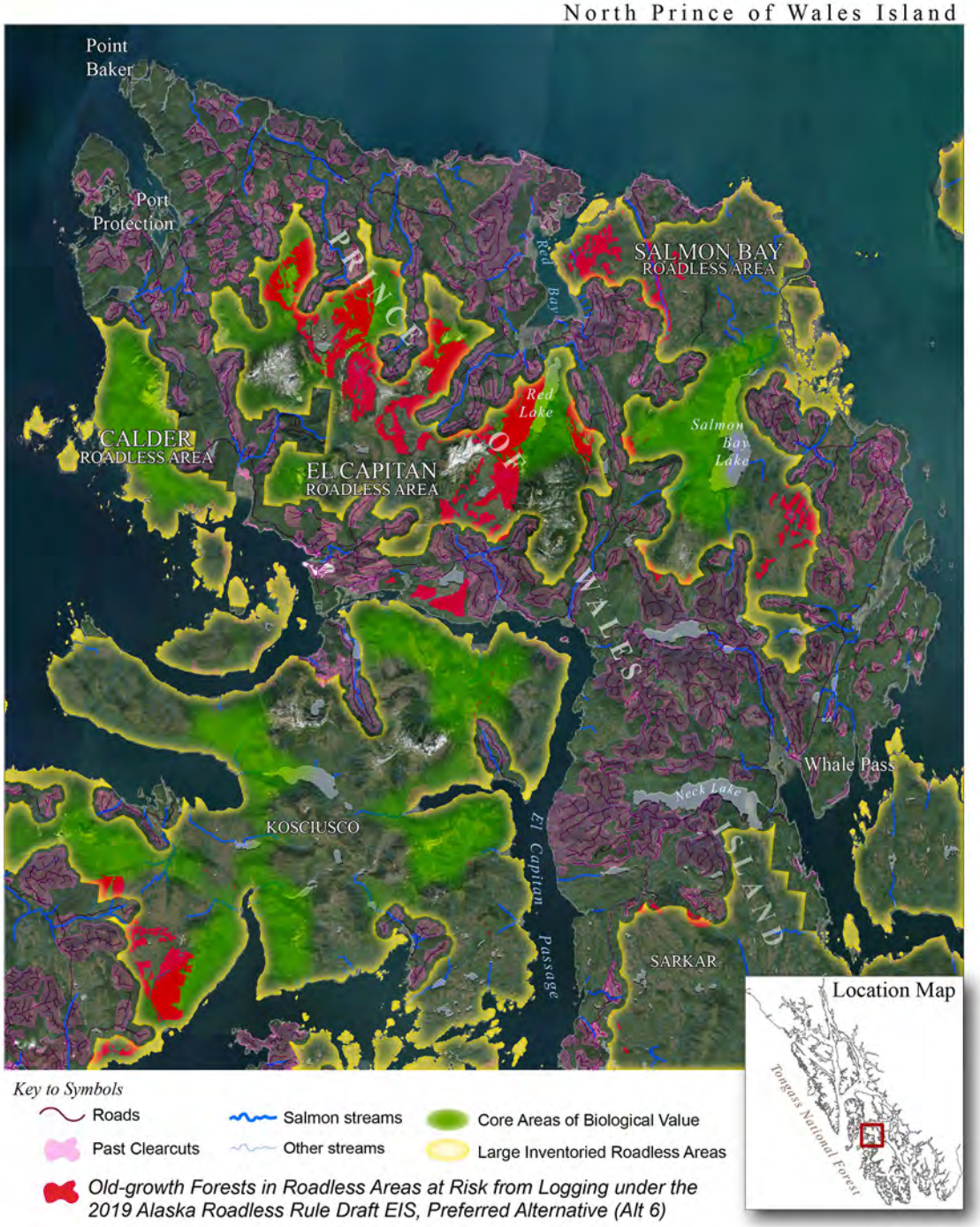
Conservation Significance of
Large Inventoried Roadless Areas
on the Tongass National Forest

North Prince of Wales Island



Key to Symbols

- Roads
- Salmon streams
- Core Areas of Biological Value
- Past Clearcuts
- Other streams
- Large Inventoried Roadless Areas
- *Old-growth Forests in Roadless Areas at Risk from Logging under the 2019 Alaska Roadless Rule Draft EIS, Preferred Alternative (Alt 6)*

Audubon ALASKA

DEFENDERS OF WILDLIFE

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David M. Albert

CONSERVATION SIGNIFICANCE OF LARGE INVENTORIED ROADLESS AREAS ON THE TONGASS NATIONAL FOREST

David M. Albert, Juneau AK

ABSTRACT

We evaluated the conservation significance of large inventoried roadless toward the goal of maintaining viable and well-distributed populations of fish and wildlife across the Tongass National Forest. We used the best available data to calculate indicators of habitat condition for 5 important species and forest systems. The significance of roadless areas was evaluated based the relative distribution of habitat values among biogeographic provinces, the degree to which habitats have been altered relative to historical conditions, the proportion of remaining values contained in large inventoried roadless areas; and the proportion of remaining values in lands potentially available for future development. No biological indicators exceeded the 40% threshold based on current alteration from original conditions region-wide, although loss of contiguous forest landscapes was approaching that value with a decline of 39.2%. However, within biogeographic provinces 25% of all indicators exceeded this threshold, with highest levels of alteration within the Prince of Wales Island group. The average decline across all indicators was 29% from historical conditions, regionwide. Consideration of lands potentially available for future development with removal of the Roadless Rule would result in a Cumulative Risk Index of 50.4% across all indicators. Large inventoried roadless areas contain approximately 48.8% of all remaining habitat values, including a high proportion of remaining contiguous old-growth forest landscapes that have been severely reduced elsewhere. Reduction of current protections for large inventoried roadless areas by the USFS would likely increase the vulnerability of remaining rare and high value habitats for fish and wildlife to future logging.

INTRODUCTION

Southeastern Alaska encompasses one of the largest remaining portion of old-growth temperate rainforest on earth (DellaSala 2011). These globally rare forests continue to support abundant populations of fish and wildlife such as brown bears (*Ursus arctos*), wolves (*Canis lupus*) and Pacific salmon (*Oncorhynchus sp*.) and other species that have declined or become threatened in southern portions of their ranges. With increasing evidence of large-scale changes in wildlife

and ecosystem function world-wide (Birdlife International 2018, Bowyer et al. 2019), and the services these systems provide to people (Millennium Ecosystem Report 2005), there is a similarly increasing need for quantitative tools to compare of management alternatives, evaluate risks and inform decision-making (Martin et al. 2009)

Industrial logging in the region increased rapidly with the Tongass Timber Act of 1947, and long-term contracts to supply pulp mills in Ketchikan and Sitka by 1954 (Beier et al. 2009). Since then, timber harvest and road construction have selectively penetrated many of the most biologically productive forest lands of region, with a disproportionate loss of the large-tree stands, low elevation valley bottom and karst forests, and landscapes of contiguous old-growth forest (Albert and Schoen 2013). This pattern of disproportionate logging also has consequences for old-growth dependent species (Shanley et al. 2013), and the ability of managers to maintain viable and well-distributed populations across this region fragmented by islands, mountains and ice fields (Cook et al. 2006, Dawson et al. 2007).

Much of the remaining high-value old-growth forests and contiguous forest landscapes only occur within roadless areas. Some portion of remaining large inventoried roadless areas were granted protection from logging under the 2001 Roadless Rule, and upheld as part of a stakeholder agreement implemented by US Forest Service in the 2016 Amendment to the Tongass National Forest Plan. However, in response to a 2018 petition by the State of Alaska, the USFS has released a Draft Environmental Impact Statement (DEIS) to consider remove these protective measures, with public review and comment available through December 2019 (USFS 2019).

In this paper we evaluate the significance of biological values associated with roadless areas on the Tongass NF. We stratified the analysis among biogeographic provinces and account for spatial isolation and biogeography effects of the Alexander Archipelago (Albert & Schoen 2007a). For each of these biogeographic provinces, we calculated indices of (1) *relative biological value* based on indicators of forest, fish and wildlife habitats, (2) *ecological condition* to estimate the proportion of habitats altered by past logging, including cumulative effects of both public and private lands, and (3) the *vulnerability* of remaining habitat within all Development Land Use Designations (LUDs) under the 2016 TLMP. Finally, we combined the indices of ecological condition and vulnerability to develop a *cumulative index of ecological risk*.

This index describes the proportion original habitats that have been altered by past logging and the proportion that may be altered under future management scenarios (Albert and Schoen 2007b). This index provides a quantitative index for stakeholders and decision-makers to weigh alternatives and design strategies to achieve desired social, ecological and economic outcomes (Martin et al. 2009).

STUDY AREA

Southeast is dominated by the Alexander Archipelago, made up of thousands of islands. This coastal ecosystem has a marine shoreline of more than 18,000 mi (30,000 km) with over 250,000 acres (101,200 ha) of intertidal habitats providing a rich environment that ranks among the most productive salmon spawning regions in the world. The climate of Southeast is maritime with cool, wet weather predominating throughout most of the year.

Although Southeast is best known for its rainforest, more than 45% of the land area of the region is unforested rock, ice, alpine, or muskeg bog, and less than one-third of the land base of Southeast is considered productive forest land. Much (~89%) of the forest land in Southeast is still old growth (>150 years old), dominated by western hemlock (*Tsuga heterophylla*)-Sitka spruce (*Picea sitchensis*) (Fig. 1). Approximately 72,000 people live in Southeast distributed throughout approximately 30 communities, of which Juneau—the state capital—is the largest. Over 500,000 acres (200,000 ha) of logging has occurred on the Tongass, and nearly 350,000 acres (141,000 ha) on state and private lands throughout Southeast, including construction of over 7,500 miles of roads.

METHODS

The study area for this project included approximately 17.6 million acres, which included the Tongass NF (~16.6 million acres) and adjacent private lands (~1.0 million acres) to account for cumulative effects of past and future logging (Fig. 2). The study further categorized the area into 20 Biogeographic Provinces representing gradients in climate, geology, vegetation and mammal diversity (McDonald & Cook 1996, Cook & McDonald 2001, USFS 1997). We used the best available data on forest conditions and habitat values using agency datasets and published models (Johnson and Blossom 2017, USFS 1998) to estimate the relative contribution of each biogeographic province to the total regional distribution (Albert & Schoen 2007b).

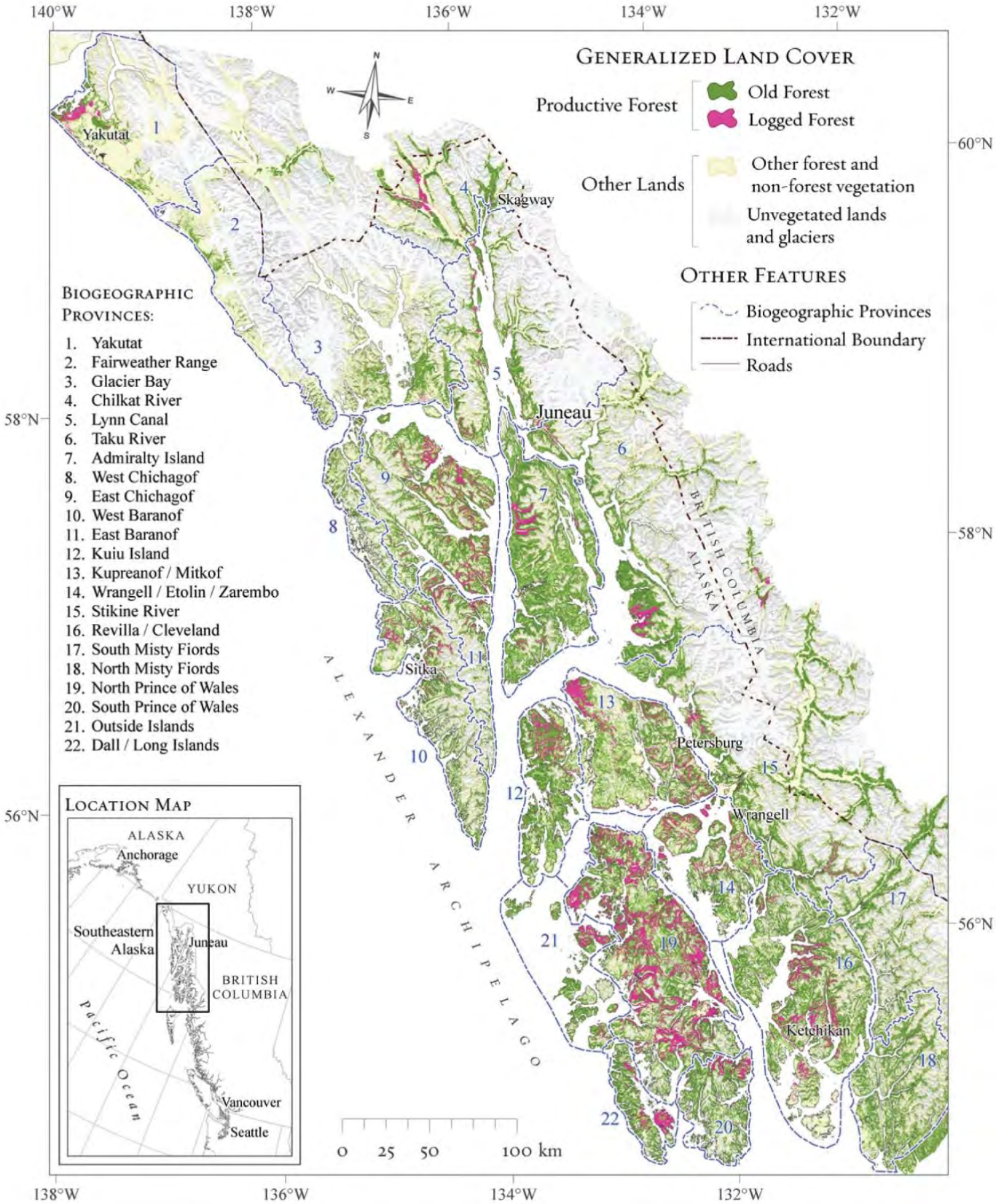

Figure 1. Forest condition and generalized landcover in Southeast Alaska (from Albert and Schoen 2013)

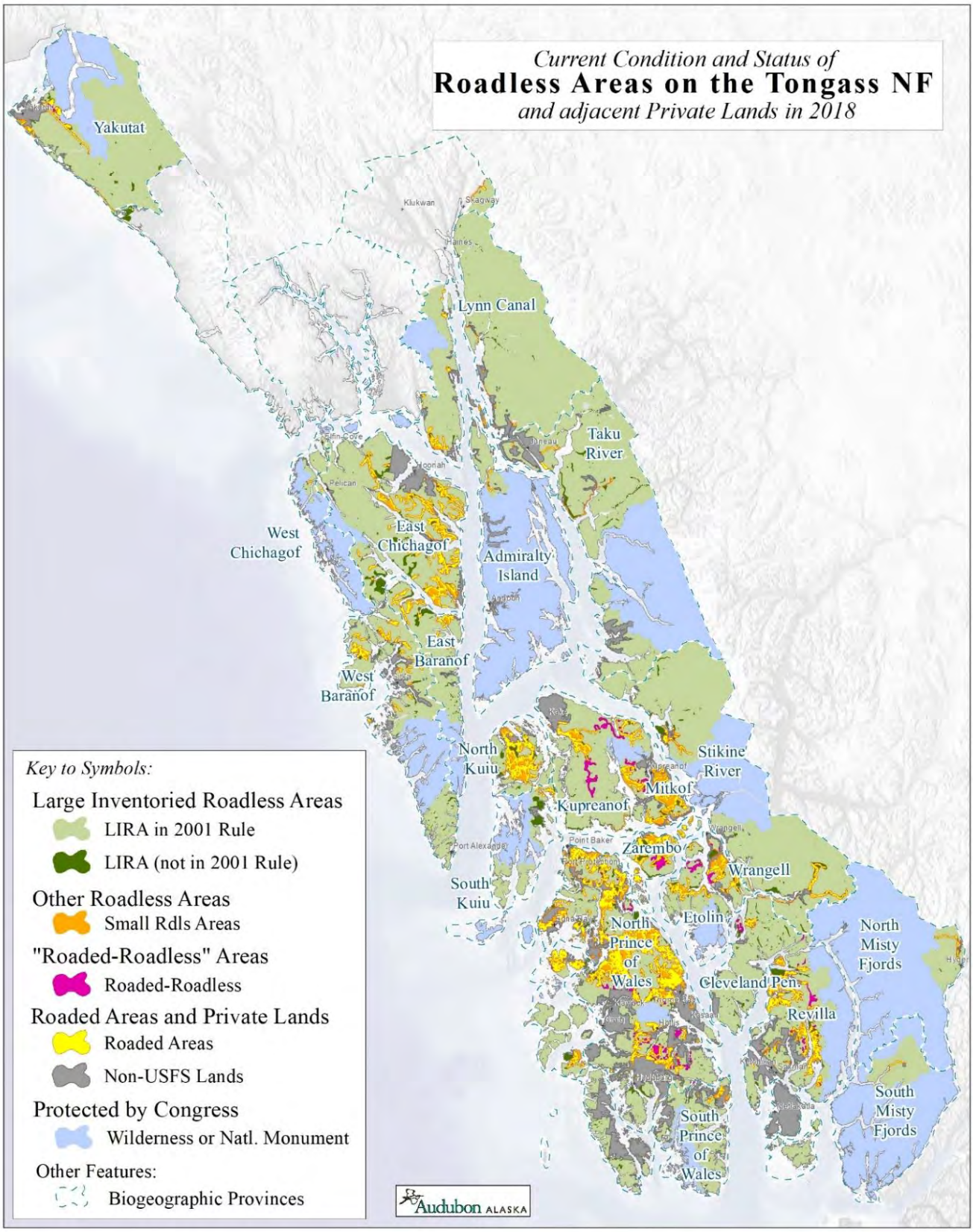

Figure 2. Study area and roadless status of the Tongass National Forest and adjacent private lands.

Indicators of Forest, Fish and Wildlife Habitat

To quantify the spatial distribution and of habitat values and evaluate change over time, we selected 5 indicators of biological value (Groves 2003), including large-tree forests (>21" quadradic mean diameter; Caouette & DeGayner 2005) and contiguous old-growth forest landscapes (Shanley et al. 2013), floodplain forest associated with 5 species of Pacific salmon (*Oncorhynchus spp*.; Paustian et al. 1992, USFS 1996, Albert & Schoen 2007a), summer habitat for brown (*Ursus arctos*) and black bear (*Ursus americanus*; Schoen et al 1994), and winter habitat for Sitka black-tailed deer (*Odocoileus hemionus sitkensis*; Schoen & Kirchhoff 1990, Suring et al. 1994). For further details on methods and model development, see Albert & Schoen (2007a). These indicators represent forest, fish and wildlife habitats with high ecological, social and economic values that are known to be sensitive to logging originally developed as part of the Audubon-TNC Conservation Assessment (Albert & Schoen 2007a). We updated the best available information on most recent forest conditions using the latest inventory of timber harvest on USFS lands, the All Lands Young-growth Inventory published as part of the 2016 Tongass Advisory Committee process and augmented with more recent harvest using Google Earth imagery. These data on forest condition were then used to update habitat models for deer, bear and floodplain forests associated with salmon streams.

We followed USFS definitions to characterize forest lands based on timber volume, tree size and stand density, as well as landscape-scale forest characteristics (Albert & Schoen 2013). Productive forests were defined by USFS as lands that contain >8 thousand board-feet (mbf) per acre, and father categorized based structural characteristics of tree size and stand density (Cauoette & DeGayner 2008). To evaluate forest composition at a landscape scale, we identified areas with >70% coverage of medium-to-high volume POG (>16 mbf / acre) within 0.39 mile^2 (1 km^2) as contiguous old-growth forest landscapes. This has been identified as a functional threshold for landscapes to support old-growth dependent species such as the Northern flying squirrel (Shanley et al. 2013). For each of these metrics, we used the best available information to estimate pre-logged forest conditions and evaluate changes over time (Albert & Schoen 2013). We used the USFS Roadless Inventory as developed in the 2003 Supplemental EIS to the Tongass Land Management Plan (TLMP; USFS 2003), the 2001 Roadless Rule, along with the current extent of roads and roadless areas to characterize the contribution of roadless areas to the remaining distribution of forests, fish and wildlife values (Fig. 3).

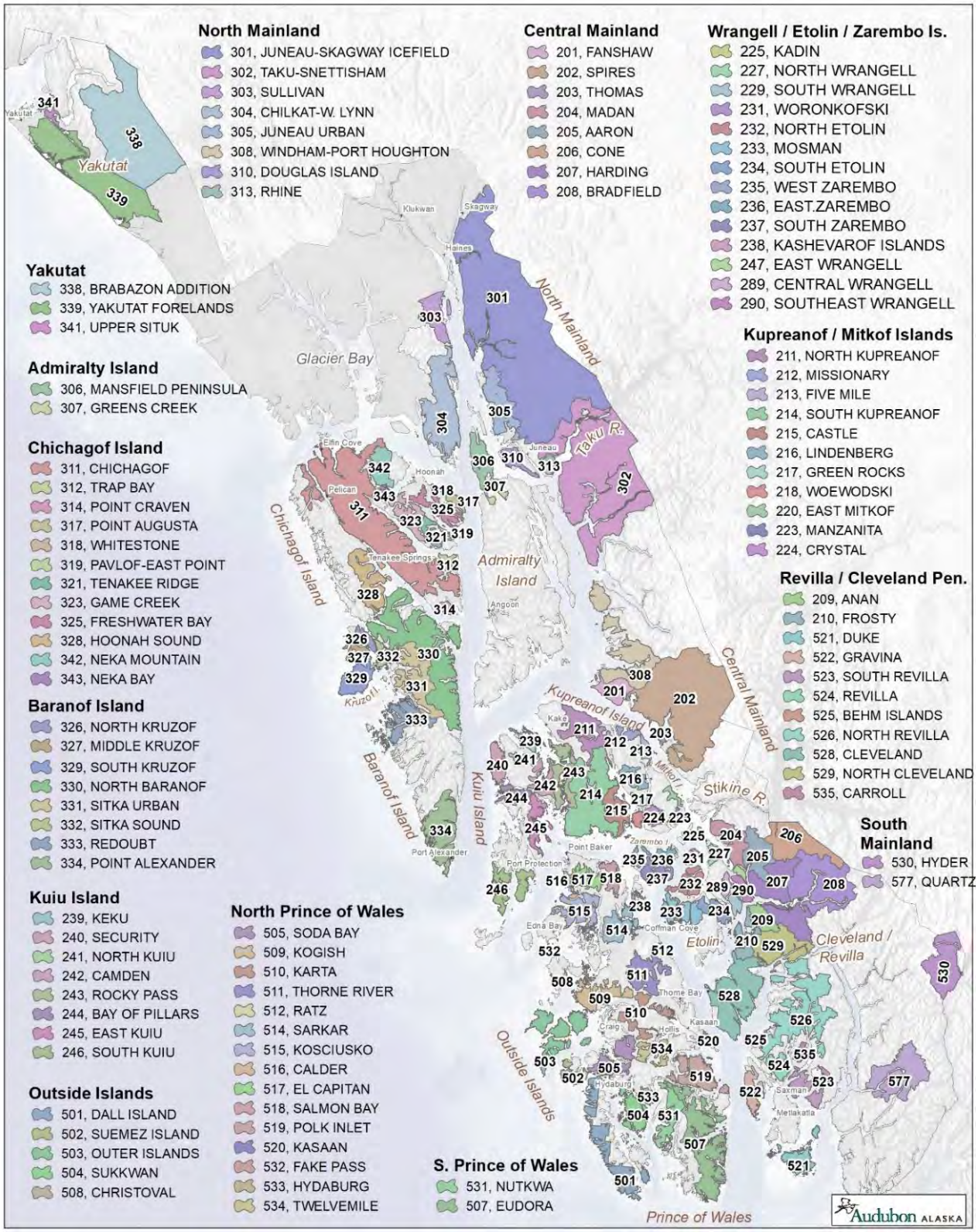

Figure 3. Named roadless areas in 2003 Tongass Roadless Inventory (USFS 2003).

Index of Relative Biological Value

As described above, we selected 5 indicators of biological value that are sensitive to changes associated with industrial logging and road construction. For this analysis, focal species included salmon, brown and black bear, Sitka black-tailed deer, large-tree forests and contiguous forest landscapes. Values in these models reflect key aspects of each species life history. Our estimate of habitat values for salmon was based on the distribution of freshwater habitat used for spawning or rearing by each of the 5 species of pacific salmon, while the distribution of forest types was based on an integrated regional database of vegetation and landcover (Albert & Schoen 2013) that was updated to reflect current conditions. These data were extensively reviewed by interagency biologists and local experts and have been judged to adequately describe the large-scale patterns of distribution and abundance of habitat values in this region. Based on these data, we were able to evaluate the current and original abundance of habitat values for each indicator, as well as their relative distribution among biogeographic provinces.

We defined an index of relative biological value (RBV) as the percent contribution of each biogeographic province to the total distribution of habitat values for each species or ecological system:

$$\mathrm{RBV}_p = \frac{\sum_{i=1}^{n}(h_p/h_{total})}{n}$$

where:

p	=	biogeographic province
n	=	number of target species or systems within province (p)
h_p	=	habitat value for species (i) contained within province (p)
h_{total}	=	total habitat for species (i) in the region

Index of Ecological Condition

The index of ecological condition is an estimate of the degree to which forests and associated habitat values have been altered as a result of past human activity. This index is presented as a percentage of the original habitat values that have been altered, rather than a strict interpretation as decline in habitat values. This reflects the complex spatial and temporal dynamics by which logging and associated activities affect habitat values for fish and wildlife, such as the time-lag in forest succession and associated habitat values (Alaback 1982, Person and Brinkman 2013).

$$\text{Index of Ecological Condition 2018}_p = \frac{\sum_{i=1}^{n}\left(1-(h_{curr}/h_{orig})\right)}{n}$$

where:

p	=	biogeographic province
n	=	number of target species or systems within province (p)
h_{curr}	=	habitat value for species (i) contained within province (p)
h_{orig}	=	original habitat for species (i) within province (p)

For each biological indicator, we estimated the original condition of the forest and wildlife values, based on the best available data. For example, we estimated that the regional distribution of large-tree forests was reduced from a total of 795,680 acres in 1954 to 542,846 acres in 2018 (Table 2). In this case, the remaining distribution is 68.2% of the 1954 total, so the Index of Current Condition is (1 – 0.682 = 0.318), reflecting a 31.8% decline in the regional distribution of large-tree forests (Table 2). The overall index is the average decline across in habitat values across all 5 indicators, including large tree forest, contiguous forest, salmon floodplain forest, bear habitat and deer habitat (Table 7).

In some cases, we needed to make informed assumptions to estimate the original condition. For example, in some cases the original forest composition was unknown, so to estimate the distribution of large-tree that had been logged, we used available data on selectivity in logging from 1986 - 2006 (Albert & Schoen 2013) as a conservative estimate of the percent change in the rare, large-tree forest types over time since 1954. We used these estimates to calculate the original distribution of large-tree forests, and to estimate the original capability of winter habitat for deer and summer habitat for brown and black bear. We estimated conditions of habitat for salmon by the percent of forests located within the floodplain of documented salmon streams that had been logged. While these estimates are not expected to directly predict trends in population size or abundance, they can be used as a conservative index to the departure from natural conditions, which in turn provides insight into the robustness of these systems for continued production of goods and services on which people rely, as well as resilience to future variability such as climate change (Orians and Schoen 2013, Person and Brinkman 2013).

Index of Relative Vulnerability

We calculated an index of relative vulnerability as a percent of remaining habitat values that occur within landscapes available for future logging or other development:

$$\text{Index of Vulnerability 2018}_p = \frac{\sum_{i=1}^{n}(h_{devp}/h_{total})}{n}$$

where:

p	=	biogeographic province
n	=	number of target species or systems within province (p)
h_{devp}	=	2018 habitat value for species (i) designated within development landscapes in province (p)
h_{total}	=	2018 total habitat for species (i) within province (p)

This index allows for comparison of relative vulnerability among scenarios as part of forest planning and public review. In this analysis, we included all TNF lands designated as Timber Production, Modified Landscape, and Scenic Viewshed under the 2016 Amendment to the Tongass Land Management Plan (USFS 2016). Lands owned by the State of Alaska, the Alaska Mental Health Land Trust, Alaska Native Corporations and other private lands were also considered to be available for development in this analysis.

This is a landscape-scale index that estimates the percentage of habitat values contained within development LUDs, and does not consider stand-scale suitability for logging (USFS 2016). Limiting such an analysis only to effects of stand-scale disturbance or suitability for logging would underestimate the cumulative vulnerability to secondary effect of logging and infrastructure on wildlife, such as habitat loss from road construction, edge effects and fragmentation, and downstream effects of altered stream hydrology and sedimentation on aquatic habitats (Lindenmeyer and Franklin, 2002).

Index of Cumulative Ecological Risk

Finally, cumulative ecological risk is an estimate of the total proportion of original habitat values that have been altered or are potentially at risk from potential future alteration under management scenarios.

This index was calculated by adding the percent of original habitat values that have been altered (Index of Current Condition) and the percent of remaining values that occur within development lands, adjusted to reflect a percent of original condition (Index of Vulnerability$_{adj}$):

$$\%\text{Cumulative Risk}_p = \%\text{Ecological Condition}_p + \%\text{Vulnerability}_{p\text{-adj}}$$

This adjustment in the Index of Vulnerability$_{adj}$ is necessary so that the Index of Cumulative Risk can be interpreted as a proportion of the total original (circa 1954) value for each individual and

combined biological indicators. For example, assume that the current distribution of a biological indicator (e.g., large-tree forest acres) is 70% of its original value. In this case, the Index of Ecological Condition is 30%. Further assume that 40% of the remaining values are designated within development landscapes or private lands, for an Index of Vulnerability of 40%. However, to calculate the cumulative risk as a measure of pre-industrial (~1954) habitat conditions, the vulnerability score needs to be adjusted as a percentage of that value. In this example, the current vulnerability (40%) is multiplied by the proportion of habitat remaining (70%) to yield an "adjusted" Index of $\text{Vulnerability}_{adj}$ of 28% (i.e., 0.4 * 0.7 = 0.28). Finally, in this example the Index of Cumulative Risk is 52% (i.e., 0.3 + 0.28 = 0.52). The interpretation is that an estimated 52% of the original distribution of a biological indicator (e.g., large-tree forest), or the average of all indicators combined, either have already been altered or are vulnerable to potential future alteration at a landscape scale under this scenario. This is simply a measure of the degree to which habitat values for this set of indicators are expected to remain intact over the current planning horizon. This does not imply that species declines will or will not occur, but simply that the risk of instability is related to the cumulative change in habitat values relative to the natural range of variability within coastal forest ecosystems (Albert & Schoen 2007b).

RESULTS

The Tongass NF and adjacent private lands cover an area of approximately 17.6 million acres. About 2/3 of this area is unvegetated, non-forest or non-commercial forest land cover types, including glaciers, alpine forests and extensive peatlands (Table 1). Productive forests cover the remaining 1/3 of the region or approximately 6.1 million acres (Table 1). Of these productive forest lands, approximately 863,000 acres (14.1 %) have been logged since 1954, including both public and private lands (Table 1). The distribution of this logging has been selective, with the highest concentrations on Prince of Wales and neighboring islands, where approximately 420,000 acres (30.4%) of productive forests have been logged. Region-wide, Prince of Wales and neighboring islands have sustained 48.6% of all logging in the region, within a group of islands that contained only 22.6% of all productive forests (Table 1).

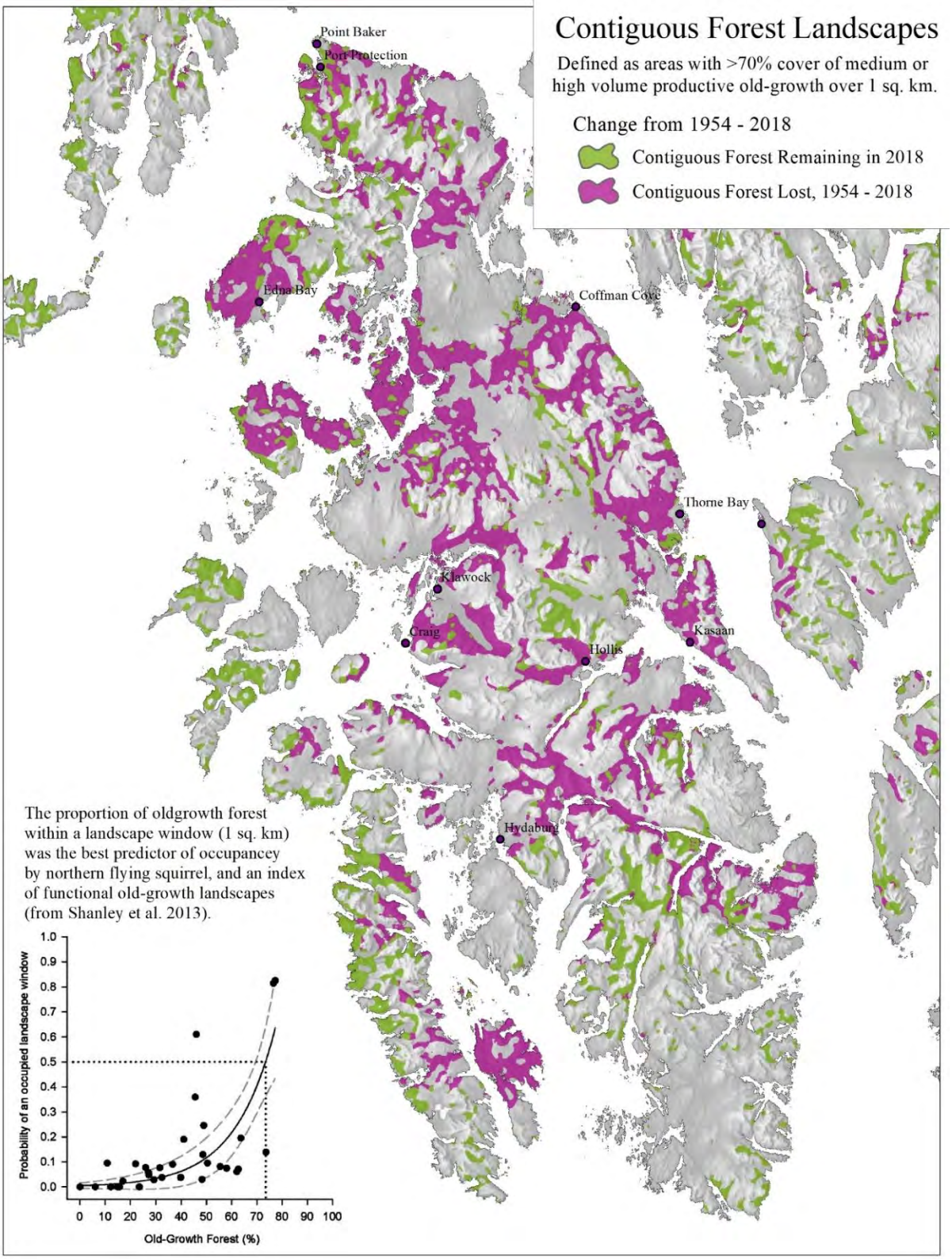

Figure 4. Change in distribution of contiguous forests at a landscape scale (1sq. km), 1954 - 2018

Large-tree POG Forests

Large-tree forests (defined as stands with tree-size >21" quadratic mean diameter) occur on approximately 542,800 acres and represent approximately 10% of all productive forest lands (Table 2). We conservatively estimate that the original distribution of large-tree old-growth forests was 795,680 acres, which represents a region-wide decline of 31.8% from pre-industrial forest conditions (Albert & Schoen 2013). In this region naturally isolated among islands and further fragmented by high elevation mountains and extensive wetlands, contiguous forest landscapes were always relatively rare. We estimate that in 1954, approximately 39.4% of all productive forests (2.4 million acres) were part of contiguous old-growth forest landscapes, and the remaining 60.6% (3.7 million acres) were in fragmented patches at a landscape scale. In 2018, only 27.6% of old-growth forests (1.5 million acres) were part of contiguous forest landscapes and the remaining 72% (3.8 million acres) were characterized by fragmented old-growth forest landscapes. Thus, contiguous forest landscapes have been reduced by 39.4% region-wide, with the highest loss evident on North Prince of Wales Island, where contiguous old-growth landscapes have been reduced by 77.5% (Table 3).

Contiguous Old-growth Forest Landscapes

Forests that are contiguous over a landscape scale (defined as >70% canopy of medium-to-high volume productive old growth forest per sq. km) originally accounted for approximately 2.5 million acres region-wide, tended to occur on the southern and central islands (Table 3). The Prince of Wales Island group originally accounted for 27.7% of the regional total, with 10.2% of that found on North Prince of Wales alone. Regionwide, these forests have been reduced by 39.2% to approximately 1.5 million acres in 2018. Likewise, the proportional loss of contiguous forest has been the most dramatic on North Prince of Wales (Fig. 4), where contiguous forests have been reduced by 77.5%, followed by Kupreanof / Mitkof (55.9% loss), East Baranof (55.5% loss) and West Baranof (50% loss). East Baranof has a very small proportion of the regional distribution (1.3%), but 93.1% of that is found in large inventoried roadless areas. Other provinces with the highest proportion of remaining contiguous forests in LRIA include East Chichagof (78.3%), West Baranof (77.4%), Dall Island Complex (76.8%), and Lynn Canal (75.9%). The province with the highest proportion of contagious forests vulnerable to future development include Kupreanof / Mitkof (48.5%), East Baranof (45.4%) and Etolin / Zarembo

(43.2%). The cumulative ecological risk region-wide, considering both past and potential for future fragmentation represents approximately 54.1% of the original distribution of these types of forests. Provinces with the highest cumulative risk include North Prince of Wales (85.2%), Kupreanof / Mitkof (77.3%), East Baranof (75.7%) and Etolin / Zarembo (70.4%) (Table 3).

Summer Habitat Capability Model for Brown and Black bear

Brown and/or black bears are present throughout the region. According to the Interagency habitat capability model, the largest contribution to the regional distribution of bear habitat was from North Prince of Wales (15.7%), East Chichagof (8.6%), Kupreanof / Mitkof Islands (8.3%) and Admiralty Islands (8.0%) (Table 4). Further, this model suggested that region wide, habitat values had declined from 1954 - 2018 by an estimated 36.9% (Table 4). Provinces with largest declines in estimated habitat capability included North Prince of Wales (62.4%), North Kuiu (56.2%), Dall & Long Islands (62.4%), followed by E. Chichagof (48.6%), Kupreanof / Mitkof (47.9%) and Etolin / Zarembo (46%). An estimated 49.6% of remaining bear habitat was found in Large Inventoried Roadless Areas, with the largest contributions found in the Outside Islands (77.6%), Yakutat Forelands (75.5%), and Taku River (74.3%). An estimated 24.8% of the remaining habitat capability for brown and black bear were located in development LUDs within the Tongass NF or on State of Alaska or private lands. The largest potential vulnerability to future development occurred on N. Kuiu Island (54.8%), Dall Island Complex (52%) and N. Prince of Wales (51.8%) (Table 4). Overall the cumulative ecological risk, including both past modification and potential future development, is estimated at 52.8% of the original habitat for brown and black bear region wide. Individual provinces with the highest cumulative risk include N. Prince of Wales (81.9%), N. Kuiu (80.2%), Dall Island Complex (78.1%), Kupreanof / Mitkof Island (75.5%), Wrangell / Etolin / Zarembo (72.5%) and E. Chichagof Island (70.3%).

Winter Habitat Capability Model for Sitka black-tailed Deer

Winter deer habitat capability as represented in the Interagency Deer Model is governed by forest type, elevation, aspect and winter snow depth. Provinces with highest contribution of winter deer habitat to the region-wide total include N. Prince of Wales (19.3%), Admiralty Island (9.3%), Revilla / Cleveland Peninsula (9.3%) and Kupreanof / Mitkof Islands (8.1%). Region-wide, we estimated that winter habitat for deer has declined by 16.7% (Table 5), with largest declines in N. Prince of Wales (35.5%), Dall Island (22.2%), N. Kuiu (21%), E. Chichagof

(20.5%) and E. Baranof (20%). Large inventoried roadless areas contain approximately 49.8% of the remaining winter deer habitat, with largest roadless contribution in E. Baranof (70.6%), Kupreanof / Mitkof (69.2%) and Stikine River (66.6%). Region-wide, an estimated 34.3% of remaining winter deer habitat is located in development lands, including both Tongass NF and adjacent state or private lands. The largest proportion of remaining habitat in development lands was found on N. Prince of Wales (55.1%), Dall Island (54.9%), Kupreanof / Mitkof (54.4%), N. Kuiu (52.7%) and Wrangell / Etolin / Zarembo (51.1%). Overall the combination of past logging and potential future development result in an Index of Cumulative Ecological Risk of 45.3% of the estimated 1954 habitat. Highest cumulative risk occurred on N. Prince of Wales (71.1%), Dall Island Complex (64.9%), N. Kuiu (62.6%) and Kupreanof / Mitkof Islands (62.6%) (Table 5).

Floodplain Forests Associated with Salmon Spawning and Rearing

Salmon streams occur on floodplains and other lands that cover approximately 1 million acres of on the Tongass NF and adjacent lands (Table 6). Of those, approximately 500,000 are characterized as productive forest, with approximately 395,484 acres (78.6%) of old-growth forest and 107,706 acres (21.4%) of post-logging young growth forest (Table 6). The largest regional contribution of anadromous floodplain old-growth forests in Southeast Alaska were found on N. Prince of Wales Island (20.6%), E. Chichagof Island (10.3%), Kupreanof / Mitkof Islands (8.0%) and the Stikine River (7.6%). Provinces with the largest percent of anadromous floodplain forests logged included E. Baranof (42%), N. Prince of Wales (36.6%), W. Baranof (34.2%) and North Kuiu (29.2%). Overall, approximately 46% of remaining old-growth anadromous floodplain forests occur within large roadless areas, and approximately 36.5% of remaining old-growth floodplain forests in Tongass Development LUDs, state or private lands Biogeographic provinces with the largest proportion of remaining old-growth salmon forests in large roadless areas include Kupreanof / Mitkof (70.6%), Yakutat Forelands (68.8%), Outside Islands (67.9%) and S. Prince of Wales Island (65.3%). Provinces with the highest proportion of remaining old-growth salmon forests located in development lands included Dall Island (59.3%), North Prince of Wales (57.5%), North Kuiu (56%) and East Chichagof (50.8%). Taken together, the Index of Cumulative Ecological Risk of past logging and potential future development represents 50.1% of all floodplain forests associated with anadromous fish in Southeast Alaska,

with highest proportions found on North Prince of Wales Island (73.1%), Dall Island (70%), North Kuiu (68.9%), East Chichagof (63.9%) and East Baranof (62%).

Combined Indicators of Forest, Fish and Wildlife Habitat

The final step in our analysis was to combine all indicators of biological value to get 'average' values for the relative distribution among biogeographic provinces (Index of Relative Biological Value), the degree to which these values have been altered by past logging and road construction (Index of Current Condition), the proportion of remaining values that are located within large roadless areas, the proportion of remaining values that are located in lands available for development, either within the Tongass NF or adjacent lands, and finally the combination of past harvest with potential future development (Index of Cumulative Ecological Risk) (Table 7). These results are similar in pattern to the previous tables: North Prince of Wales contains by far the largest proportion of biological values among any biogeographic province (20.2%), followed by Admiralty Island (9.7%), East Chichagof Island (8.4%), Revilla / Cleveland Peninsula (7.8%) and Kupreanof / Mitkof Islands (7.0%). On average, 29% of the original distribution of these value has been altered by past logging and road construction (Fig. 5a), including largest proportions altered on North Prince of Wales (51.5%), East Baranof (44.7%), Dall Island (41.4%) and Kupreanof / Mitkof (37.6%). Region-wide, an average of 48.8% of remaining values were located in large roadless areas, and an average of 29.7% of remaining values were located in lands available for development, including both Tongass NF and adjacent lands. Provinces with the highest remaining proportions in roadless areas included Lynn Canal (68.4%), Taku River (68.4%), Revilla / Cleveland (66.9%), Kupreanof / Mitkof (66.7%) and Stikine River (65.3%). Provinces with the highest proportion of remaining values located in lands available for future development include North Kuiu (51%), Kupreanof / Mitkof (50.6%), Wrangell / Etolin / Zarembo (48.7) and North Prince of Wales (48.3%). Taken together, the cumulative effect of past logging and potential for future development within lands designated on the Tongass NF or adjacent State of Alaska or private lands, represents approximately 50% of the original distribution of biological values for these indicators region-wide (Fig. 5b). Among biogeographic provinces, this cumulative ecological risk was highest on North Prince of Wales (76%), followed by Dall Island (69.1%), Kupreanof / Mitkof (68.9%), North Kuiu (68.6%), Wrangell / Etolin / Zarembo (66.9%), East Baranof (64.8%) and East Chichagof (63%) (Table 7).

Table 1. Regional distribution and current condition of productive forest lands among biogeographic provinces (all lands).

Biogeographic Provinces	Old Growth		Young Growth		All		% of POG Logged
	(acres)	(%)	(acres)	(%)	(acres)	(%)	
ABC Islands (all)	**1,412,130**	**26.9%**	**147,815**	**17.3%**	**1,559,945**	**25.5%**	**9.5%**
Admiralty Island	596,482	11.4%	32,371	3.8%	628,853	10.3%	5.1%
E. Baranof Island	88,612	1.7%	14,365	1.7%	102,977	1.7%	13.9%
E. Chichagof Island	426,305	8.1%	80,994	9.4%	507,299	8.3%	16.0%
W. Baranof Island	228,347	4.3%	20,085	2.4%	248,432	4.1%	8.1%
W. Chichagof Island	72,385	1.4%		0.0%	72,385	1.2%	0.0%
Central Islands (all)	**1,388,861**	**26.4%**	**229,482**	**26.9%**	**1,618,343**	**26.5%**	**14.2%**
Wrangell Etolin Zarembo	222,139	4.2%	46,127	5.5%	268,266	4.4%	17.2%
N. Kuiu Island	125,545	2.4%	26,822	3.2%	152,367	2.5%	17.6%
S. Kuiu Island	156,370	3.0%	4,196	0.5%	160,566	2.6%	2.6%
Kupreanof / Mitkof	343,116	6.5%	76,590	8.9%	419,706	6.9%	18.2%
Revilla Island / Cleveland	541,691	10.3%	75,748	9.0%	617,439	10.1%	12.3%
Prince of Wales Complex (all)	**959,743**	**18.3%**	**419,671**	**48.0%**	**1,379,414**	**22.6%**	**30.4%**
Dall Island Complex	97,516	1.9%	39,096	4.5%	136,612	2.2%	28.6%
North Prince of Wales	587,988	11.2%	338,944	38.6%	926,932	15.2%	36.6%
Outside Islands	112,792	2.1%	20,039	2.4%	132,831	2.2%	15.1%
South Prince of Wales	161,447	3.1%	21,591	2.5%	183,038	3.0%	11.8%
Mainland (all)	**1,409,622**	**26.8%**	**45,094**	**5.3%**	**1,454,716**	**23.8%**	**3.1%**
Lynn Canal	209,374	4.0%	6,568	0.8%	215,942	3.5%	3.0%
North Misty Fjords	215,885	4.1%	17	0.0%	215,902	3.5%	0.0%
South Misty Fjords	312,729	6.0%		0.0%	312,729	5.1%	0.0%
Stikine River	331,532	6.3%	15,679	1.9%	347,211	5.7%	4.5%
Taku River	340,101	6.5%	22,830	2.7%	362,931	5.9%	6.3%
Yakutat (all)	**81,262**	**1.5%**	**20,855**	**2.5%**	**102,117**	**1.7%**	**20.4%**
Yakutat Forelands	81,262	1.5%	20,855	2.5%	102,117	1.7%	20.4%
Grand Total	**5,251,618**	**100%**	**862,916**	**100%**	**6,114,534**	**100%**	**14.1%**

Table 2. A comparison of the regional distribution of large-tree forests, change over time, contribution of large inventoried roadless areas, and cumulative risk among biogeographic provinces (all lands).

Biogeographic Provinces	Large-tree Forest		Index of Relative Biological Value	Index of Current Condition (% altered)	% of Remaining in Large Roadless	% of Remaining in Devp. or Private Lands	Index of Cumulative Risk (% of original)
	1954	2018					
ABC Islands (all)	**186,693**	**143,383**	**23.5%**	**23.2%**	**24.0%**	**11.3%**	**31.9%**
Admiralty Island	107,848	98,364	13.6%	8.8%	7.2%	0.9%	9.6%
E. Baranof Island	6,126	1,918	0.8%	68.7%	74.4%	38.0%	80.6%
E. Chichagof Island	60,188	36,457	7.6%	39.4%	64.3%	37.2%	61.9%
W. Baranof Island	10,530	4,645	1.3%	55.9%	44.9%	23.8%	66.4%
W. Chichagof Island	2,000	2,000	0.3%	0.0%	18.2%	0.0%	0.0%
Central Islands (all)	**165,966**	**98,728**	**20.9%**	**40.5%**	**58.8%**	**44.9%**	**67.2%**
Etolin Zarembo Island Complex	25,292	11,777	3.2%	53.4%	51.5%	51.4%	77.4%
N. Kuiu Island	31,686	23,828	4.0%	24.8%	46.3%	61.6%	71.1%
S. Kuiu Island	12,424	11,194	1.6%	9.9%	49.6%	22.7%	30.3%
Kupreanof / Mitkof Islands	42,783	20,342	5.4%	52.5%	66.3%	51.9%	77.1%
Revilla Island / Cleveland Peninsula	53,780	31,586	6.8%	41.3%	69.3%	33.4%	60.9%
Prince of Wales Group (all)	**305,924**	**182,960**	**38.4%**	**40.2%**	**48.3%**	**43.6%**	**66.3%**
Dall Island Complex	20,105	8,650	2.5%	57.0%	85.1%	21.9%	66.4%
North Prince of Wales Complex	219,553	120,243	27.6%	45.2%	39.6%	47.6%	71.3%
Outside Islands	18,513	12,642	2.3%	31.7%	53.3%	33.4%	54.5%
South Prince of Wales Island	47,751	41,425	6.0%	13.2%	67.7%	37.7%	46.0%
Mainland (all)	**103,984**	**90,771**	**13.1%**	**12.7%**	**52.2%**	**27.0%**	**36.3%**
Lynn Canal / Mainland	18,186	16,261	2.3%	10.6%	71.7%	35.5%	42.3%
North Misty Fjords	16,398	16,393	2.1%	0.0%	10.0%	7.1%	7.1%
South Misty Fjords	14,105	14,105	1.8%	0.0%	32.3%	0.0%	0.0%
Stikine River / Mainland	25,301	20,707	3.2%	18.2%	55.3%	35.1%	46.9%
Taku River / Mainland	29,994	23,305	3.8%	22.3%	77.8%	44.2%	56.6%
Yakutat (all)	**33,114**	**27,003**	**4.2%**	**18.5%**	**77.4%**	**36.1%**	**47.9%**
Yakutat Forelands	33,114	27,003	4.2%	18.5%	77.4%	36.1%	47.9%
Grand Total	**795,680**	**542,846**	**100.0%**	**31.8%**	**45.9%**	**32.1%**	**53.7%**

Table 3. A comparison of contiguous landscape forests among biogeographic provinces, change over time, contribution of large inventoried roadless areas, Index of Vulnerability and Index of Cumulative Risk in Large Roadless Areas.

Biogeographic Provinces	Contiguous Landscape Forest		Index of Relative Biological Value	Index of Current Condition (% altered)	% of Remaining in Large Roadless	% of Remaining in Devp. or Private Lands	Index of Cumulative Risk (% of original)
	1954	2018					
ABC Islands (all)	**605,064**	**420,243**	**23.5%**	**30.5%**	**33.4%**	**12.0%**	**38.9%**
Admiralty Island	310,967	268,361	12.1%	13.7%	8.7%	0.9%	14.4%
E. Baranof Island	32,839	14,623	1.3%	55.5%	93.1%	45.4%	75.7%
E. Chichagof Island	203,577	101,743	7.9%	50.0%	78.3%	34.2%	67.1%
W. Baranof Island	51,281	28,815	2.0%	43.8%	77.4%	22.7%	56.5%
W. Chichagof Island	6,400	6,700	0.2%	-4.7%	21.0%	0.1%	-4.6%
Central Islands (all)	**630,766**	**358,675**	**24.5%**	**43.1%**	**62.3%**	**32.4%**	**61.6%**
Wrangell Etolin Zarembo Islands	93,938	48,972	3.6%	47.9%	66.9%	43.2%	70.4%
N. Kuiu Island	95,502	52,293	3.7%	45.2%	73.6%	39.3%	66.7%
S. Kuiu Island	67,039	61,162	2.6%	8.8%	33.3%	16.1%	23.5%
Kupreanof / Mitkof Islands	145,691	64,315	5.7%	55.9%	55.1%	48.5%	77.3%
Revilla Island / Cleveland Peninsula	228,597	131,933	8.9%	42.3%	73.0%	25.3%	56.9%
Prince of Wales Group (all)	**712,701**	**241,072**	**27.7%**	**66.2%**	**66.1%**	**29.0%**	**76.0%**
Dall Island Complex	76,128	39,636	3.0%	47.9%	76.8%	32.5%	64.9%
North Prince of Wales Complex	494,468	111,108	19.2%	77.5%	60.0%	34.1%	85.2%
Outside Islands	75,924	47,719	2.9%	37.1%	70.5%	12.1%	44.8%
South Prince of Wales Island	66,181	42,609	2.6%	35.6%	68.1%	32.0%	56.2%
Mainland (all)	579,796	517,498	**22.5%**	**10.7%**	**57.0%**	**26.3%**	**34.2%**
Lynn Canal / Mainland	90,291	80,461	3.5%	10.9%	75.9%	30.9%	38.4%
North Misty Fjords	60,747	60,824	2.4%	-0.1%	8.2%	3.3%	3.1%
South Misty Fjords	72,241	72,505	2.8%	-0.4%	30.0%	0.1%	-0.3%
Stikine River / Mainland	154,501	131,551	6.0%	14.9%	70.6%	26.2%	37.2%
Taku River / Mainland	202,016	172,157	7.8%	14.8%	66.5%	43.3%	51.7%
Yakutat (all)	**47,151**	**27,598**	**1.8%**	**41.5%**	**43.6%**	**42.3%**	**66.2%**
Yakutat Forelands	47,151	27,598	1.8%	41.5%	43.6%	42.3%	66.2%
Grand Total	**2,575,478**	**1,565,086**	**100%**	**39.2%**	**53%**	**24.5%**	**54.1%**

Table 4. A comparison of the regional distribution of brown and black bear habitat capability, change over time, contribution of large inventoried roadless areas and cumulative risk in southeastern Alaska.

Biogeographic Provinces	Habitat Capability Index		Index of Relative Biological Value	Index of Current Condition (% altered)	% of Remaining in Large Roadless	% of Remaining in Devp. or Private Lands	Index of Cumulative Risk (% of original)
	1954	2018					
ABC Islands (all)	**4,226**	**2,860**	**24.7%**	**32.3%**	**37.7%**	**16.1%**	**43.2%**
Admiralty Island	1,373	1,136	8.0%	17.2%	5.4%	1.0%	18.1%
E. Baranof Island	358	219	2.1%	38.7%	66.0%	25.6%	54.4%
E. Chichagof Island	1,463	752	8.6%	48.6%	70.3%	42.3%	70.3%
W. Baranof Island	786	525	4.6%	33.3%	58.5%	14.2%	42.7%
W. Chichagof Island	247	228	1.4%	7.6%	15.5%	0.3%	7.9%
Central Islands (all)	**4,141**	**2,305**	**24.2%**	**44.3%**	**61.1%**	**42.0%**	**67.7%**
Etolin Zarembo Island Complex	622	336	3.6%	46.0%	48.9%	49.2%	72.5%
N. Kuiu Island	380	167	2.2%	56.2%	52.0%	54.8%	80.2%
S. Kuiu Island	343	257	2.0%	25.0%	37.8%	16.4%	37.3%
Kupreanof / Mitkof Islands	1,418	739	8.3%	47.9%	72.1%	53.0%	75.5%
Revilla Island / Cleveland Peninsula	1,378	806	8.1%	41.5%	65.6%	34.6%	61.8%
Prince of Wales Group (all)	**3,624**	**1,603**	**21.2%**	**55.8%**	**56.5%**	**42.3%**	**74.5%**
Dall Island Complex	210	96	1.2%	54.4%	63.2%	52.0%	78.1%
North Prince of Wales Complex	2,683	1,009	15.7%	62.4%	50.5%	51.8%	81.9%
Outside Islands	297	212	1.7%	28.5%	77.6%	15.9%	39.8%
South Prince of Wales Island	434	286	2.5%	34.0%	60.7%	26.3%	51.4%
Mainland (all)	**4,102**	**3,376**	**24.0%**	**17.7%**	**43.4%**	**14.8%**	**29.9%**
Lynn Canal / Mainland	629	426	3.7%	32.3%	74.3%	21.7%	47.0%
North Misty Fjords	741	697	4.3%	5.9%	4.3%	0.9%	6.8%
South Misty Fjords	852	804	5.0%	5.6%	12.9%	0.0%	5.7%
Stikine River / Mainland	1,000	751	5.8%	24.9%	66.4%	26.9%	45.1%
Taku River / Mainland	879	698	5.1%	20.6%	74.3%	28.6%	43.3%
Yakutat (all)	**1,009**	**651**	**5.9%**	**35.5%**	**75.5%**	**10.6%**	**42.4%**
Yakutat Forelands	1,009	651	5.9%	35.5%	75.5%	10.6%	42.4%
Grand Total	**17,101**	**10,795**	**100.0%**	**36.9%**	**49.6%**	**24.8%**	**52.5%**

Table 5. A comparison of the regional distribution of Sitka black-tailed deer habitat capability, change over time, contribution of large inventoried roadless areas and cumulative risk in southeastern Alaska.

Biogeographic Provinces	Habitat Capability Index		Index of Relative Biological Value	Index of Current Condition (% altered)	% of Remaining in Large Roadless	% of Remaining in Devp. or Private Lands	Index of Cumulative Risk (% of original)
	1954	2018					
ABC Islands (all)	**80,272**	**70,779**	**24.4%**	**11.8%**	**37.4%**	**19.5%**	**29.0%**
Admiralty Island	30,514	28,738	9.3%	5.8%	7.0%	2.1%	7.8%
E. Baranof Island	4,573	3,657	1.4%	20.0%	70.6%	41.9%	53.5%
E. Chichagof Island	24,144	19,206	7.4%	20.5%	63.4%	44.7%	56.0%
W. Baranof Island	15,609	13,755	4.8%	11.9%	64.2%	21.9%	31.2%
W. Chichagof Island	5,432	5,423	1.7%	0.2%	15.5%	0.3%	0.5%
Central Islands (all)	**92,572**	**78,644**	**28.2%**	**15.0%**	**60.8%**	**43.3%**	**51.9%**
Etolin Zarembo Island Complex	15,229	12,407	4.6%	18.5%	53.1%	51.1%	60.2%
N. Kuiu Island	8,819	6,971	2.7%	21.0%	59.5%	52.7%	62.6%
S. Kuiu Island	11,306	10,937	3.4%	3.3%	45.5%	16.9%	19.7%
Kupreanof / Mitkof Islands	26,578	21,818	8.1%	17.9%	69.2%	54.4%	62.6%
Revilla Island / Cleveland Peninsula	30,641	26,512	9.3%	13.5%	64.1%	39.0%	47.2%
Prince of Wales Group (all)	**99,195**	**71,187**	**30.2%**	**28.2%**	**52.0%**	**47.4%**	**62.3%**
Dall Island Complex	10,553	8,211	3.2%	22.2%	55.9%	54.9%	64.9%
North Prince of Wales Complex	63,453	40,907	19.3%	35.5%	46.5%	55.1%	71.1%
Outside Islands	11,117	9,393	3.4%	15.5%	62.5%	24.0%	35.8%
South Prince of Wales Island	14,072	12,675	4.3%	9.9%	59.9%	36.1%	42.4%
Mainland (all)	**48,135**	**45,858**	**14.7%**	**4.7%**	**44.9%**	**23.7%**	**27.3%**
Lynn Canal / Mainland	7,647	7,316	2.3%	4.3%	67.7%	37.8%	40.5%
North Misty Fjords	5,666	5,671	1.7%	-0.1%	6.3%	2.5%	2.4%
South Misty Fjords	11,333	11,337	3.5%	0.0%	11.1%	0.0%	0.0%
Stikine River / Mainland	12,486	11,541	3.8%	7.6%	66.6%	34.8%	39.8%
Taku River / Mainland	11,003	9,993	3.4%	9.2%	63.4%	39.6%	45.1%
Yakutat (all)	**8,186**	**6,951**	**2.5%**	**15.1%**	**63.7%**	**23.3%**	**34.8%**
Yakutat Forelands	8,186	6,951	2.5%	15.1%	63.7%	23.3%	34.8%
Grand Total	**328,361**	**273,418**	**100.0%**	**16.7%**	**49.8%**	**34.3%**	**45.3%**

Table 6. A comparison of the regional distribution of floodplain forests associated with anadromous fish habitat, forest condition, contribution of large inventoried roadless areas and cumulative risk among biogeographic provinces in southeastern Alaska.

Biogeographic Provinces	Anadromous Flood Plain			Index of Relative Biological Value	Index of Current Condition (% altered)	% of Remaining in Large Roadless	% of Remaining in Devp. or Private Lands	Index of Cumulative Risk (% of original)
	All Lands (acres)	Old-growth forest	Young-growth forest					
ABC Islands (all)	**189,964**	**88,554**	**30,412**	**23.6%**	**25.6%**	**37.8%**	**29.2%**	**47.3%**
Admiralty Island	44,956	26,466	5,532	6.4%	17.3%	9.4%	1.5%	18.5%
E. Baranof Island	16,940	6,204	4,495	2.1%	42.0%	54.5%	34.4%	62.0%
E. Chichagof Island	76,682	38,213	13,862	10.3%	26.6%	51.8%	50.8%	63.9%
W. Baranof Island	38,419	12,544	6,523	3.8%	34.2%	55.7%	31.2%	54.8%
W. Chichagof Island	12,966	5,127	0	1.0%	0.0%	15.8%	0.1%	0.1%
Central Islands (all)	**208,840**	**94,661**	**16,863**	**22.2%**	**15.1%**	**55.8%**	**45.8%**	**54.0%**
Etolin Zarembo Complex	24,705	11,968	1,752	2.7%	12.8%	42.8%	49.6%	56.1%
Kuiu Island	17,362	10,710	676	2.3%	5.9%	36.8%	24.6%	29.0%
Kupreanof / Mitkof Islands	72,184	34,232	5,907	8.0%	14.7%	70.6%	45.7%	53.7%
North Kuiu	20,584	10,426	4,292	2.9%	29.2%	35.0%	56.0%	68.9%
Revilla Island / Cleveland Pen	74,004	27,325	4,237	6.3%	13.4%	58.2%	48.7%	55.6%
Prince of Wales (all)	**224,503**	**89,869**	**42,835**	**26.4%**	**32.3%**	**41.1%**	**53.1%**	**68.2%**
Dall Island Complex	14,816	5,657	2,024	1.5%	26.4%	56.6%	59.3%	70.0%
N. Prince of Wales	166,986	65,787	37,931	20.6%	36.6%	32.9%	57.5%	73.1%
Outside Islands	20,053	8,779	1,098	2.0%	11.1%	67.9%	36.2%	43.3%
S. Prince of Wales	22,648	9,646	1,783	2.3%	15.6%	65.3%	35.1%	45.2%
Mainland (all)	**290,805**	**105,638**	**13,379**	**23.7%**	**11.2%**	**44.6%**	**21.9%**	**30.7%**
Lynn Canal	49,240	14,612	1,996	3.3%	12.0%	57.1%	50.3%	56.3%
North Misty Fjords	60,126	16,058	0	3.2%	0.0%	4.3%	3.2%	3.2%
South Misty Fjords	40,261	23,732	0	4.7%	0.0%	32.7%	0.0%	0.0%
Stikine River	79,364	30,439	7,725	7.6%	20.2%	55.5%	18.2%	34.7%
Taku River	61,813	20,797	3,658	4.9%	15.0%	64.9%	47.1%	55.0%
Yakutat (all)	**167,569**	**16,764**	**4,217**	**4.2%**	**20.1%**	**68.8%**	**28.1%**	**42.6%**
Yakutat Forelands	167,569	16,764	4,217	4.2%	20.1%	68.8%	28.1%	42.6%
Grand Total	**1,081,681**	**395,484**	**107,706**	**100.0%**	**21.4%**	**46.0%**	**36.5%**	**50.1%**

Table 7. A comparison of combined indicators of biological value among biogeographic provinces, including current condition, % of remaining in large roadless areas, % of remaining in development lands, and cumulative ecological risk.

Biogeographic Provinces	Index of Relative Biological Value	Index of Current Condition (% altered)	% of Remaining in Large Roadless	% of Remaining in Devp. or Private Lands	Index of Cumulative Risk (% of original)
ABC Islands (all)	**24.2%**	**24.6%**	**34.4%**	**17.1%**	**37.5%**
Admiralty Island	9.7%	11.7%	7.2%	1.2%	12.6%
E. Baranof Island	1.6%	44.7%	71.5%	36.7%	64.8%
E. Chichagof Island	8.4%	36.7%	66.5%	41.0%	63.0%
W. Baranof Island	3.4%	36.4%	60.7%	22.1%	50.2%
W. Chichagof Island	1.0%	1.6%	16.4%	0.2%	0.8%
Central Islands (all)	**23.7%**	**31.6%**	**59.7%**	**41.0%**	**59.8%**
Wrangell / Etolin / Zarembo	3.5%	35.3%	51.9%	48.7%	66.9%
N. Kuiu Island	3.1%	35.4%	52.4%	51.0%	68.6%
S. Kuiu Island	2.4%	10.6%	39.8%	18.2%	26.8%
Kupreanof / Mitkof Islands	7.0%	37.6%	66.7%	50.6%	68.9%
Revilla Island / Cleveland Peninsula	7.8%	30.5%	66.9%	35.3%	55.7%
Prince of Wales Group (all)	**28.4%**	**44.5%**	**52.8%**	**42.2%**	**68.9%**
Dall Island Complex	2.3%	41.4%	67.3%	44.7%	69.1%
North Prince of Wales Complex	20.2%	51.5%	45.9%	48.3%	76.0%
Outside Islands	2.4%	24.6%	67.4%	23.2%	42.5%
South Prince of Wales Island	3.6%	21.3%	64.6%	32.3%	47.0%
Mainland (all)	**20.0%**	**11.1%**	**48.4%**	**21.9%**	**30.6%**
Lynn Canal	3.0%	14.0%	68.4%	35.8%	45.3%
North Misty Fjords	2.9%	1.2%	6.3%	3.3%	4.4%
South Misty Fjords	3.8%	1.1%	24.7%	0.0%	1.1%
Stikine River	5.2%	16.6%	65.3%	27.8%	40.1%
Taku River	5.1%	16.4%	68.4%	39.0%	49.1%
Yakutat (all)	**3.7%**	**25.6%**	**64.5%**	**29.4%**	**47.5%**
Yakutat Forelands	3.7%	25.6%	64.5%	29.4%	47.5%
Grand Total	**100.0%**	**29.0%**	**48.8%**	**29.7%**	**50.4%**

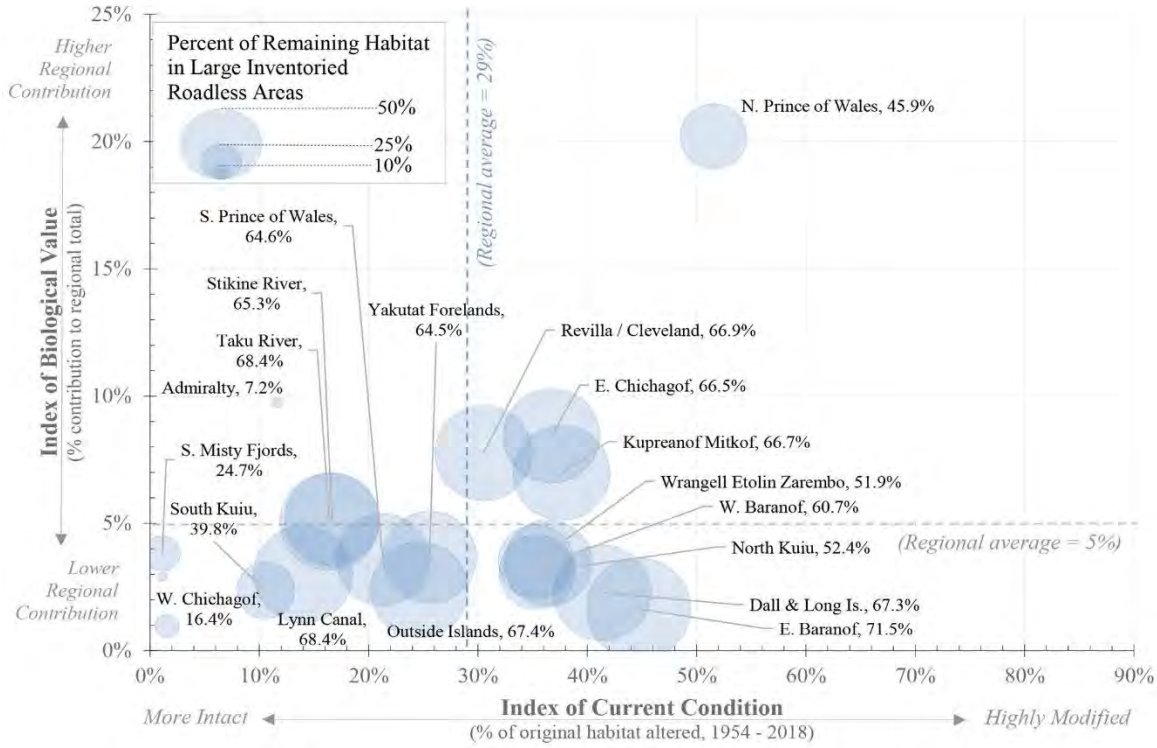

Figure 5(a). A comparison of biological value (y-axis), current condition (x-axis) and percent remaining biological values in large roadless areas (bubble size) among biogeographic provinces.

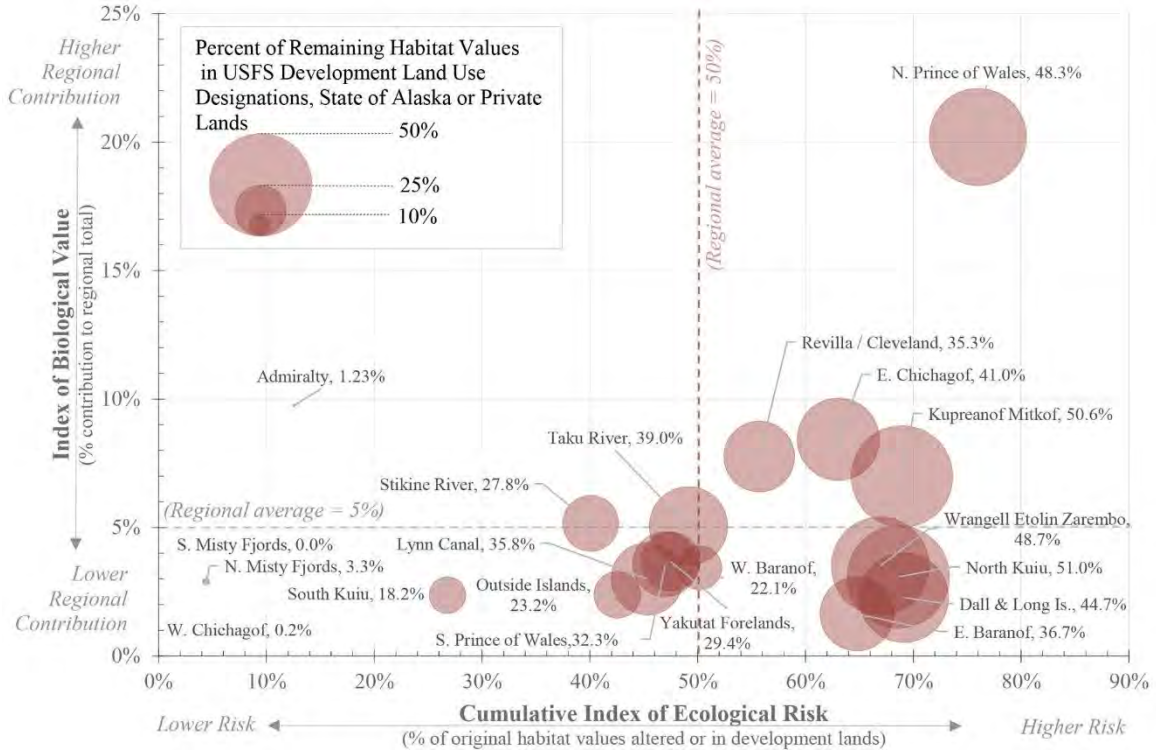

Figure 5(b). A comparison of biological value (y-axis), cumulative ecological risk (x-axis) and percent remaining biological values in development lands (bubble size) among biogeographic provinces.

DISCUSSION

In this study, we evaluated the importance of roadless areas toward the goal of maintaining viable and well distributed populations of fish and wildlife in Southeast Alaska. We based this assessment on 4 criteria: (1) the relative distribution of fish and wildlife habitat values among biogeographic provinces; (2) the current condition of habitats compared with pre-industrial conditions; (3) the proportion of remaining habitat values that are located within large roadless areas; and (4) the proportion of remaining values located in lands open to future development. Provinces that contain a disproportionate share of all habitat values within the region, those with high levels of past logging and/or vulnerable to future development, as well as those with a high proportion of remaining resources located in roadless areas are all candidates for special consideration.

Specific thresholds at which habitat alteration affects population viability is difficult to determine (Fahrig 2001). However, results of a review of habitat thresholds literature (to inform forest planning in coastal British Columbia) indicated that maintaining loss of habitat below 40% of historical abundance poses a low risk to most species, whereas declines above that level result in less confidence that risks of extirpation will remain low (Price et al. 2009). We used this general rule of thumb that modification of >40% of the original distribution for any biological indicator can no longer be considered low risk and warrants further detailed investigation.

No biological indicators exceeded the 40% threshold based on current alteration from original conditions region-wide, although loss of contiguous old-growth forest landscapes was approaching that value with a decline of 39.2% (Table 3). Within individual provinces, a total of 25 indicators exceeded the 40% threshold, including loss of contiguous forest landscapes (9 provinces; Table 3), decline in bear habitat (8 provinces; Table 4), loss of large-tree forests (7 provinces; Table 2) and logging of salmon floodplain forests (1 province; Table 6). Among provinces, 3 provinces exceeded this threshold of 40% decline for all indicators combined, including North Prince of Wales, Dall & Long Islands and East Baranof (Fig. 5a). Based on this analysis, the 2 indicators that were least sensitive to effects of past logging were total POG (Table 1) and deer habitat (Table 5).

Figure 6. North Prince of Wales Island contains the highest proportion of biological values of any biogeographic province (20.2% of regional total), but also the highest degree of alteration from historical conditions (51.5%), and the highest index of cumulative ecological risk under the 'full exemption' alternative (75%).

Potential future change in ecological conditions is particularly difficult to predict, but our Index of Cumulative Risk is a reasonable approximation to describe the proportion of fish and wildlife habitat values located within landscapes generally designated for various levels of resource development. According to this accounting, 4 of 5 indicators we evaluated surpassed the 40% threshold region-wide, with deer habitat being the only exception (Table 7). Regionwide, the average decline across all these indicators was 50.4%, and 15 of the 20 biogeographic provinces exceeded that threshold (Fig. 5b).

In this context, the remaining 48.8% ecological values contained in large inventoried roadless areas represents an important opportunity to maintain viable and well-distributed populations of fish and wildlife, and the variety of services these species provide (Table 7). Provinces with the highest proportion of remaining habitat located in large roadless areas included both highly modified provinces, such as East Baranof (71.5%), and the relatively intact provinces of the northern mainland, including Lynn Canal and Taku River (68.5%; Fig. 5a).

North Prince of Wales Island stands out as the most biologically important province in the region (Fig. 5a), with highest levels of both past logging and potential future modification based on land ownership and management (Fig 5b). Indeed, the entire Prince of Wales Island Group already exceeds the 40% threshold for habitat modification based on the average of 5 indicators we examined (Table 7). The degree of modification and fragmentation evident in the central portion of Prince of Wales Island (Fig. 6) is unique in Southeast Alaska, and more reminiscent of areas in the Pacific Northwest where populations of important fish and wildlife species have been listed under the Endangered Species Act and experience ongoing challenges (Spies et al. 2018). To avoid that fate, we recommend that remaining roadless areas on North Prince of Wales Island be given special consideration to maintain these rare forest conditions.

Of all the biological indicators that we examined, the 77.5% loss of contiguous forests on North Prince of Wales was the single greatest indicator of risk in the entire region (Fig. 4). In addition, most remaining contiguous old-growth forest landscapes only occur in large roadless areas, both on North Prince of Wales (60%) and region-wide (53%; Table 3). This was the highest association of a biological indicator to large roadless areas of any that we examined. This is of particular concern on Prince of Wales because contiguous old-growth forest landscapes there provide critical habitat for a sub-species of northern flying squirrel endemic to these islands

(Smith 2005). Moreover, recent studies have demonstrated that the composition and spacing of old-growth reserves in the Tongass Land Management Plan may not support viable populations over the long term (Pyare and Smith 2005), and habitat requirements of this species make it unlikely to persist in more fragmented landscapes (Shanley et al. 2013). The El Capitan and Salmon Bay roadless areas on the north end of Prince of Wales Island are particularly vulnerable under the Preferred Alternative to the Alaska Roadless Rule DEIS (Fig. 7).

The DEIS recognizes that proposed actions to remove protective measures for existing roadless areas would contribute to the cumulative reduction in POG and increase risks to biological diversity from fragmentation and loss of connectivity among old growth forests (USFS 2019: p. 3-68). Nonetheless, the DEIS does not calculate the degree to which human-caused fragmentation has already occurred, the relative severity of fragmentation among biogeographic provinces (Fig 5a), or adequately quantitatively evaluate the consequences of such fragmentation on biological diversity (USFS 2019, P2-28). This is a particularly serious omission given recent studies that document the degree to which past logging has disproportionately targeted contiguous, high-volume forest landscapes (Albert & Schoen 2013). Our study determined that region-wide, the loss of contiguous forest landscapes was 383% more severe (Table 3) than loss of POG in general (Table 1) and may be an important metric for evaluation in Tongass National Forest planning processes.

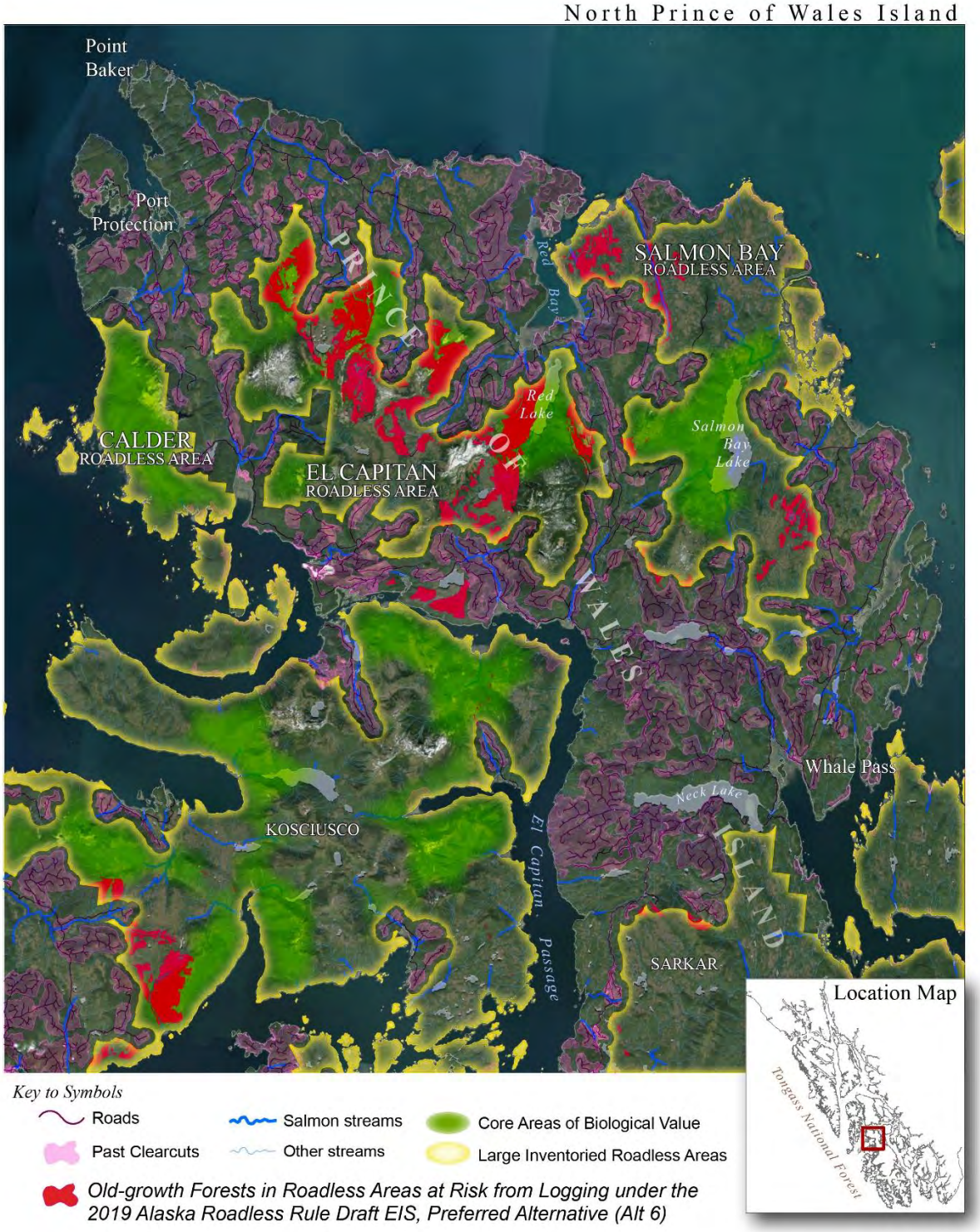

Figure 7. The 2019 Alaska Roadless Rule DEIS does not quantify landscape-scale effects to old-growth forests, yet identifies some of the last remaining contiguous old-growth landscapes on North Prince of Wales Island as "suitable" for logging under the Preferred Alternative (Alt. 6)

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Restoration Ecology
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REVIEW ARTICLE

Linkages between unpaved forest roads and streambed sediment: why context matters in directing road restoration

Robert Al-Chokhachy[1,2], Tom A. Black[3], Cameron Thomas[4], Charles H. Luce[3], Bruce Rieman[5], Richard Cissel[3], Anne Carlson[6], Shane Hendrickson[7], Eric K. Archer[8], Jeff L. Kershner[1]

Unpaved forest roads remain a pervasive disturbance on public lands and mitigating sediment from road networks remains a priority for management agencies. Restoring roaded landscapes is becoming increasingly important for many native coldwater fishes that disproportionately rely on public lands for persistence. However, effectively targeting restoration opportunities requires a comprehensive understanding of the effects of roads across different ecosystems. Here, we combine a review and a field study to evaluate the status of knowledge supporting the conceptual framework linking unpaved forest roads with streambed sediment. Through our review, we specifically focused on those studies linking measures of the density of forest roads or sediment delivery with empirical streambed sediment measures. Our field study provides an example of a targeted effort of linking spatially explicit estimates of sediment production with measures of streambed sediment. Surprisingly, our review uncovered few studies ($n = 8$) that empirically tested the conceptual framework linking unpaved forest roads and streambed sediment, and the results varied considerably. Field results generally supported the conceptual model that unpaved forest roads can control streambed sediment quality, but demonstrated high-spatial variability in the effects of forest roads on streambed sediment and the need to address hotspots of sediment sources. The importance of context in the effects of forest roads is apparent in both our review and field data, suggesting the need for in situ studies to avoid misdirected restoration actions.

Key words: restoration, road density, sediment production, streambed sediment

Implications for Practice

- Limited empirical evidence exists to specifically quantify the effects of forest roads on aquatic ecosystems and effectively target restoration.
- Sediment production from roads and delivery to streams varies substantially within and across landscape settings.
- The conceptual model linking roads to streambed sediment is supported by field data, but the contribution from different road segments can vary substantially, highlighting the importance of context in prescribing restoration actions as a means to reduce sediment contributions.

Introduction: The Need for Road Restoration

Public lands remain a critical component of large landscape conservation within western North America (Noss et al. 2002; Hauer & Muhlfeld 2010). Such public lands, however, are culturally and socioeconomically valuable to adjacent communities (Naylor et al. 2009), rendering a wide range of current and historical land use practices. Historically, unpaved roads have been constructed across the public lands of western North America to facilitate forest management, as well as associated land use practices such as timber harvest and grazing. The total extent of unpaved forest roads in Idaho, Montana, Oregon, and Washington alone is seven times the length of interstate highways across the contiguous United States. Many of these roads have been identified as causing significant impacts to streams and riparian systems through changes in hydrology, mass wasting and sediment delivery, and riparian degradation (Jones et al. 2000; Trombulak & Frissell 2000). Because there are hundreds of thousands of miles of existing unpaved roads, prioritization of restoration actions is an imperative step in effectively recovering ecosystems.

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[1]U.S. Geological Survey, Northern Rocky Mountain Science Center, 2327 University Way, Suite 2, Bozeman, MT 59715, U.S.A.
[2]Address correspondence to R. Al-Chokhachy, email ral-chokhachy@usgs.gov
[3]USDA Forest Service, 322 East Front Street, Suite 401, Boise, ID 83702, U.S.A.
[4]USDA Forest Service, 200 E. Broadway, PO Box 7669, Missoula, MT 59807-7669, U.S.A.
[5]Clearwater Resources Council, PO Box 1471, Seeley Lake, MT 59868, U.S.A.
[6]The Wilderness Society, 503 W. Mendenhall, Bozeman, MT 59715, U.S.A.
[7]USDA Forest Service, Building 24a, Fort Missoula, Missoula, MT 59804, U.S.A.
[8]USDA Forest Service, 860 N 1200 E, Logan, UT 84321, U.S.A.

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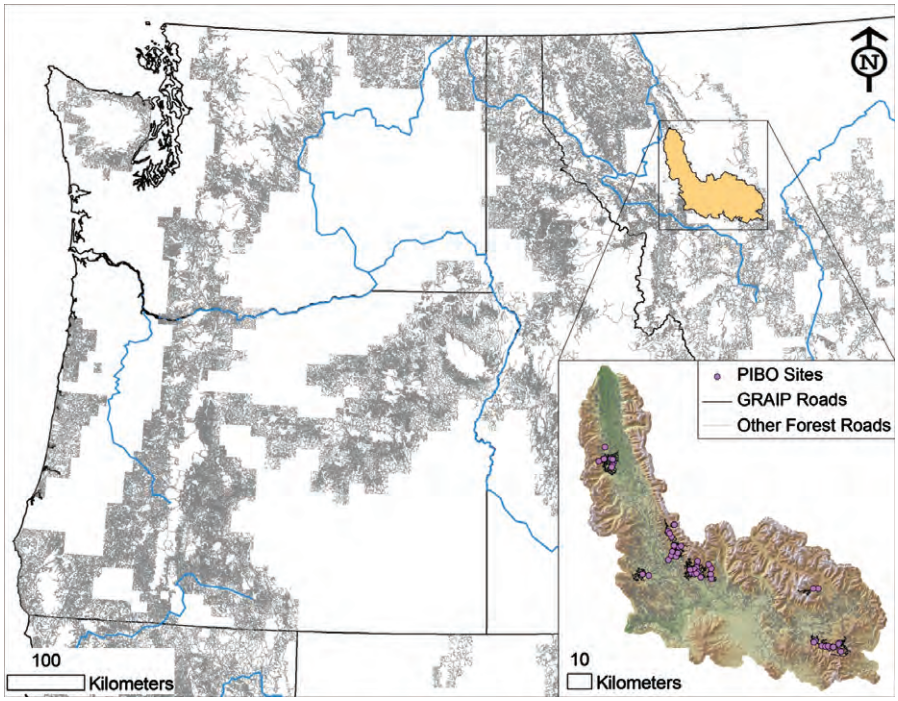

Figure 1. The majority of forest road networks (gray) and major rivers (blue) throughout the Pacific Northwest (gray), the location of the *Southwest Crown of the Continent (inset), forest roads included in the GRAIP inventory (black), and habitat sampling locations (i.e. PIBO sites).

Removing or restoring unpaved roads remains an option for mitigating ecosystem effects (Lloyd et al. 2013), but this is a daunting task given the extent of the current road network (Fig. 1), the high cost of either maintenance, relocation, or obliteration of roads, and the restrictive nature of available funding. Furthermore, there remains considerable resistance from the public to closing forest roads that have historically been open for recreation and management.

Mitigating the effects of unpaved forest roads is an important component of native salmonid (e.g. salmon and trout) conservation, a group of fishes with high ecological, socioeconomic, and culture value (Naiman et al. 2002; Stapp & Hayward 2002; Loomis 2006) that has experienced significant declines in distribution and abundance across the Rocky Mountains (Thurow et al. 1997). Headwater streams within and immediately downstream of public lands are critical for supporting multiple stages of salmonid life cycles (Northcote 1997). Anticipated changes in climate suggest that montane ecosystems are likely to be disproportionately important as areas of coldwater refugia (Wenger et al. 2011), and yet these areas contain a large portion of the existing, unpaved roads. Although roads may not represent a large portion of the total sediment budget in these areas (Goode et al. 2012), they can have a disproportionate effect on streambed sediment (Maturana et al. 2014). A key question for managers remains how and what can be done (i.e. climate adaptation) to increase species resilience to climate change through alleviating land management activities that potentially limit populations (Lawler et al. 2010).

In response to the need for forest restoration, the U.S. Congress established the Collaborative Forest Landscape Restoration Program (CFLRP) in 2009 to initiate science and community-based, large-scale restoration on public lands. The CFLRP is part of the Omnibus Public Land Management Act and encourages collaborative, science-based ecosystem restoration while specifically incorporating ecological, economic, and social sustainability components. A key component in initiating landscape restoration is science-based justification of specific locations for restoration (sensu Luce et al. 2001; Rieman et al. 2010), as well as monitoring the long-term effectiveness of such actions (e.g. ecosystem response to road restoration). Providing empirically-based rationale for the need for road restoration will likely facilitate socially and economically acceptable restoration under the CFLRP.

Effectively targeting road restoration opportunities in montane ecosystems requires spatially explicit disturbance information (i.e. road effects) and an understanding of how different disturbance processes, intensities, routing, and locations affect important attributes of aquatic ecosystems. The general understanding of road effects on aquatic ecosystems has been based largely on varied measures of road density and their associations with in-stream habitat or species/population status (e.g. Thurow et al. 1997; Hughes et al. 2004), but often lacks resolution on specific processes driving the apparent response.

Here, we refine our understanding of forest roads and their effects on streams by combining a review of existing literature linking unpaved forest roads with streambed sediment and a targeted study that integrates field data with mechanistic models of the processes thought to link roads to streams. We specifically focused on road-related sediment delivery and consequently, streambed sediment habitat attributes, as sediment

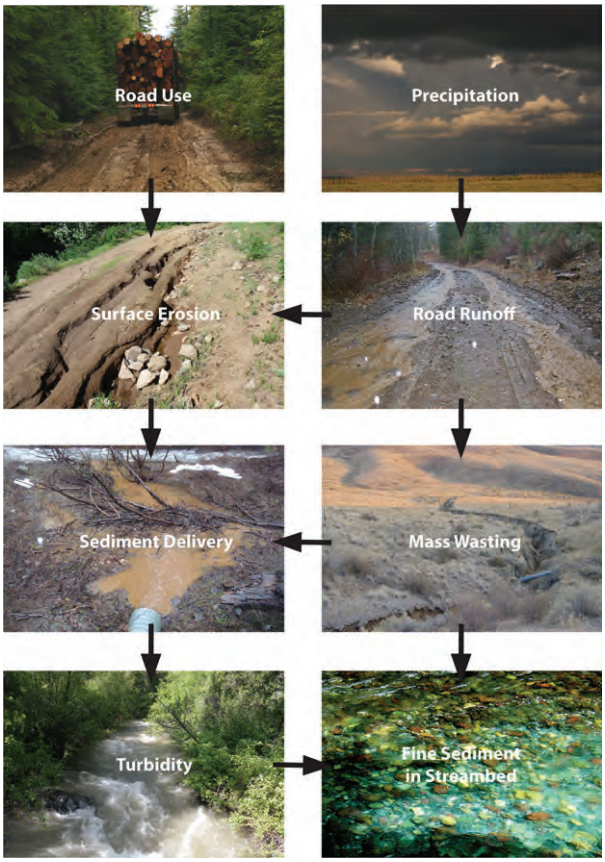

Figure 2. The conceptual model linking different sources of sediment from forest road surfaces and road networks to streambed sediment (Photos by T. Black).

strongly affects salmonid egg survival and habitat use (Tappel & Bjornn 1983; Suttle et al. 2004) and aquatic ecosystems (Wood & Armitage 1997; Allan 2004). Furthermore, the general assumption under the CFLRP is that road-related sediment is a major source of disruption of streams (Fig. 2). We merge the results from the review and the field study to address our specific objectives to evaluate: (1) the variability in sediment production across road networks; (2) the delivery of sediment from road networks to stream networks; and (3) the conceptual model (e.g. Trombulak & Frissell 2000) linking sediment production from unpaved roads to in-stream habitat characteristics (Fig. 2). Collectively, our review, which is largely driven by studies aimed at reducing the effects of roads on streams within the historical range of salmon species (*Oncorhunchus* spp.) currently listed on the U.S. Endangered Species Act list, and our case study serves to guide restoration in similar situations around the world.

Review of Current Literature

A number of reviews exist concerning the ecological effects of roads (Forman & Alexander 1998; Jones et al. 2000; Trombulak & Frissell 2000) and the delivery of sediment from mountain roads (Croke & Hairsine 2006). Here, we concentrated our efforts on peer-reviewed studies that specifically evaluated empirical relationships between sediment delivery from unpaved road surfaces and road density estimates with streambed sediment composition. We were interested in sediment from unpaved, forest roads—typically characterized as "fine" sediments (i.e. <6 mm)—as it can be mobilized by rain and melting snow, aggrade in-stream habitats and in some cases restrict the development of embryos or other life stages of fishes (Jensen et al. 2009). Our focus on these specific linkages is driven by the common and pervasive use of streambed sediment metrics to evaluate the effects of management activities on aquatic habitat (Kershner et al. 2004).

We used Web of Science to search for articles matching these criteria. We performed searches that iteratively used "forest road AND sediment or substrate" and "road density AND sediment or substrate." Subsequently, we used literature-cited references within relevant publications to identify other applicable research. In addition, we used Google Scholar to examine all publications that cited each applicable article.

What We Found; Literature Review

Focusing specifically on the effects of unpaved, forest roads on streambed sediment, our review uncovered only eight studies that specifically linked measures of sediment delivery or density of unpaved forest roads with streambed sediment metrics (Table 1). Bilby (1985) found considerably differences in turbidity levels above and below a sediment source (i.e. ditch) but no significant difference in streambed sediment characteristics. Instead of road density, Schnackenberg and Macdonald (1998) used the amount of road segments greater than 400 m within 60 m of a stream channel, and found no significant relationships between this surrogate for road density and fine sediment across catchments dominated by granitic or sedimentary geologic formations. Despite different response metrics, three of the four studies that included density of unpaved roads found significant, positive relationships between road density and measures of fine sediment. Yet, the strength of these relationships was variable and sometimes weak (e.g. $r = 0.36$; McCaffery et al. 2007). Together, these results indicate considerable variability in associations found between roads and road density and streambed sediment. At least part of this variability may be due to the anthropogenic disturbances related to roads (see below), which are not captured in road-density analyses.

Sediment delivery from locations where forest roads intersect streams (i.e. road crossings) can be particularly acute (Pechenick et al. 2014) and can comprise a large proportion of sediment production for road networks (Coe 2006). The three studies that specifically evaluated linkages between road-stream crossings and streambed sediment found significant correlations (Table 1); however, Schnackenberg and Macdonald (1998) also found the significance of the relationship varied across watershed geologies, suggesting that the effects of road crossings are context-specific. Only Cover et al. (2008) have demonstrated strong positive correlations between streambed sediments with sediment delivery from roads. Overall, given the extensive network of unpaved forest roads (Fig. 1), we consider the paucity

Table 1. Studies from a literature review that evaluated linkages between different road-related metrics and streambed sediment metrics, the location of the studies, the road metric considered, the sediment metric considered, and the correlation direction (ns, nonsignificant). [a]Relationships differed by geologic composition; nonsignificant in sedimentary-volcanic geology, significant in granitic geology. [b]Canada.

Study	*Location*	*Metric*	*Sediment Metric*	*Correlation*
Cederholm et al. (1980)	WA	Road density	Subsurface fine sediment	+
Bilby (1985)	WA	Turbidity from roads	Subsurface fine sediment	ns
Lisle and Hilton (1992)	CA	Road density	Fine sediment volume in pools	+
Eaglin and Hubert (1993)	WY	Road density, density of culverts	Surface fines and embeddedness	ns, road density; + culverts
Schnackenberg and Macdonald (1998)	CO	Density of culverts, extent of road network near channel	Surface fines	ns, +[a] culverts; ns extent of road network near channel
Kreutzweiser et al. (2005)	ON[b]	Stream crossing	Subsurface fine sediment	+
McCaffery et al. (2007)	MT	Road density, crossings	Subsurface fine sediment	+
Cover et al. (2008)	OR	Sediment supply	Surface fines, volume of fines, permeability of streambed	+, +, −

of existing literature linking observed sediment production and road density with streambed sediment surprising.

Measuring Sediment Delivery: A Case Study

We synthesized data from a field study within the 6,070 km^2 Southwestern Crown of the Continent CFLRP project in western Montana (SWCC). The SWCC is set in three adjoining National Forests (Flathead, Helena, and Lolo) and currently contains greater than 7,200 km of roads. Climate within the study area is characterized by relatively cold, wet winter and spring months and relatively warm, dry summers. Streamflows are typical of the northern Rocky Mountains with highest spring flows during May and June, which taper throughout the summer and early winter.

We selected six individual 12-digit subwatersheds in the SWCC (Hydrologic Unit Code [HUC]; http://water.usgs.gov/GIS/huc.html) to conduct field sampling based on the need and potential (i.e. logistically feasible) for road restoration. Reducing sediment contributions to aquatic ecosystems via restoration has been deemed a priority in the SWCC by State and Federal agencies and local communities. A major component of the justification for restoration is to alleviate stressors to rare and threatened species such as bull trout (*Salvelinus confluentus*), a species listed under the Endangered Species Act with specialist habitat requirements (Al-Chokhachy et al. 2010*b*).

Road Inventory and Sediment Modeling

We conducted a detailed inventory of all of the identifiable roads within the watersheds using the Geomorphic Roads Analysis and Inventory Package (GRAIP) methodology (Black et al. 2012). Through a GPS-based inventory we documented the locations of water flow along the road (i.e. ditches) and drain points where that water discharges to the hillslope or stream network (Appendix S1, Supporting Information). We recorded the volumes of gullies, landslides, and fill material that was eroded by runoff from the road and if it delivered to the stream. Next, we measured surface erosion rates for the roads in the study area at eight sediment plots between 2011 and 2014, a period ranging from below to above average precipitation (http://ncdc.noaa.gov; station ID GHCND:USC00247448). Briefly, we located 80-m plots on typical unsurfaced roads that were open ($n = 4$) and closed ($n = 4$) to public traffic. We collected runoff and sediment from the plots into settling tanks and the outflow from the tanks was connected to a filter to retain sediment (Appendix S1). Coarse sediment deposited in settling tanks was then removed and weighed wet at the site with a portable hanging scale each June and October (Black & Luce 2013). Tipping bucket rain gages were deployed near the sediment plot locations to measure liquid precipitation depth and intensity. We integrated the detailed road inventory data and erosion rates from sediment plots with 30-m elevation data (ned.usgs.gov) in the GRAIP model (Version 1.0.8, http://neng.usu.edu/cee/faculty/dtarb/graip; Cissel et al. 2012) resampled to 5 m to estimate road sediment production (E) for each road segment (Luce & Black 1999):

$$E = B \times L \times S \times V \times R$$

where B is the average erosion rate from the sediment plots (kg/m elevation), L is the road length (m), S is the slope of the road (m/m), V is the vegetation cover factor for the flow path, and R is the road surfacing factor. We employed GRAIP to route sediment from the road drain points to the downslope stream using a flow direction grid (Taudem 4, http://hydrology.usu.edu/taudem/taudem4.0/, Tarboton 1997) and the field observations of stream connection (Appendix S1). Finally, we used GRAIP to accumulate delivered road sediment in a downstream direction and normalized the total mass by watershed area to provide a specific road sediment value (mg/km^2), which allows sediment impacts to be compared between watersheds of different sizes (Cissel et al. 2012).

Streambed Sediment

Our initial sampling frame included all perennial streams in the selected SWCC subwatersheds that occurred on public lands. We used the National Elevation Dataset (30 m; http://ned.usgs.gov) to initially screen for streams with low (<6%) gradients to focus on areas most sensitive to changes in watershed conditions (e.g. Montgomery & Macdonald 2002). We then categorized the existing road densities of catchments for streams based on existing forest road information (1:24,000 scale; USDA 2014), and used a spatially balanced, stratified sampling design (generalized random tessellation stratified; Stevens & Olsen 2004) with road density categories as our strata (spsurvey package; R Development Core Team 2012) to identify specific sites for sampling (Appendix S2).

We sampled stream habitat at sites during baseflow conditions in late August and September when sediment transport is minimized (2012–2014; Kershner et al. 2004). At each sample site, we measured sediment and physical attributes that can moderate sediment characteristics (e.g. large woody debris). We sampled sediment metrics that have been demonstrated to have strong linkages with salmonid survival during the early life-history stages and are commonly used in land management monitoring programs (Al-Chokhachy et al. 2010*b*; Bryce et al. 2010). Our sediment sampling included: (1) the measures of median particle size (d_{50}) of all surface substrate particles; (2) the percent of surface fine sediment (<6 mm) in pool tails; and (3) the percentage of fine sediment (<6 mm) in subsurface sediment determined from a shovel–core sampling approach (Sutherland et al. 2010).

Quantifying Road-stream Habitat Linkages

At each of the randomly selected sites, we delineated catchments upstream of each site which encompassed all existing road information from the GRAIP surveys. We then summarized landscape-level covariates including an index of stream power (Flores et al. 2006), catchment-level estimates of drainage density, average precipitation and proportion of the catchment with less erosive Belt Series geology (i.e. sedimentary; Sugden & Woods 2007), and site-level estimates of large woody debris, and sinuosity (Appendix S3). We constrained our analyses to sample sites with field estimates of gradient (Appendix S3) less than 3.5% to align with regional monitoring programs (Kershner et al. 2004). We then developed multiple linear regression (MLR) models to evaluate relationships between road density and sediment delivery from GRAIP and streambed sediment in sites after accounting for covariates. To avoid any potential effects on model structure arising from the order in which explanatory variables entered the MLR models (i.e. spatial scales, landscape factors, and so on), we used an all-subsets modeling approach (R Development Core Team 2012).

Sediment Delivery from Roads in the SWCC

Our detailed inventory in the SWCC subwatersheds included 896 km of forest roads and 13,125 drain points. In our study, only 4–5% of the forest road network was found to be hydrologically connected to stream networks, which is dramatically different than 57% estimated in western Oregon by Wemple et al. (1996). Localized hotspots of sediment production were apparent and typically included road crossings and culverts (Fig. 3A & 3B), and only 2% of the drain points were estimated to deliver 90% of the sediment in our study watersheds. We estimated that over time road surface erosion delivered 44% of the total sediment from the road to streams, while episodic contributions from gullies, landslides, and fill erosion delivered 22%, 18% and 15%, respectively (Cissel et al. 2014).

On average roads open to travel (0.40 kg/m^2) were estimated to produce over an order of magnitude more sediment than roads closed to travel (0.03 kg/ m^2). A substantial difference with road use is consistent across studies, but the effects of use can vary considerably (e.g. Reid & Dunne 1984; Luce & Black 2001).

Road Density, Sediment Delivery, and Streambed Sediment

Estimated sediment delivered to streams varied by over an order of magnitude (Fig. 3C) and catchment-level measures of road density and sediment delivery were significantly correlated ($r = 0.43$, $p = 0.002$; Fig. 4A). Measures of streambed sediment differed across sites and we observed a range of sediment values for a given road density and sediment delivery (Fig. 4B & 4C; Fig. S2). Considering sediment criteria for bull trout, 22% of the sites exceeded what has been suggested as an optimum level for surface fines for bull trout spawning and egg incubation (Bryce et al. 2010).

After controlling for covariates, we found significant relationships between d_{50} (−), pool tail fines <6 mm (+), and percent subsurface fines (+) with road density ($p < 0.05$; Fig. 4D; Fig. S1). Although the pattern was still supportive of the conceptual model, our multimodel approach indicated less support in our data for models linking sediment production from road networks with streambed sediment than those based on simple road density (Table S1; Fig. 4C). Unfortunately, this leaves more questions than clarity, likely as sediment delivery from roads varied across a small range in the SWCC (e.g. Fig. 5 and contrast with the study of Goode et al. 2012).

Rethinking the Relationship of Roads and In-stream Sediment Effects in the SWCC

Our study clearly demonstrated a negative relationship between field-measured densities of forest roads and the quality of streambed sediment (i.e. proportion of fines, d_{50}). Data linking sediment production and streambed sediment suggest that sediment production serves as a constraint on streambed sediment (i.e. limits the possible range of conditions; Fig. 4C); indeed post hoc quantile regression analysis (0.9 quantile) supports this notion (d_{50}; $\beta = -0.09$; SE = 0.02). In other words, streams with the highest road densities or estimated sediment production and delivery rates are likely to dramatically reduce the upper range of possibilities for d_{50} (i.e. maximum possible median particle size). In concordance with this general pattern, we found an increasing range for attributes such as the percent of surface fine sediment in pool tails and percent subsurface fine sediment

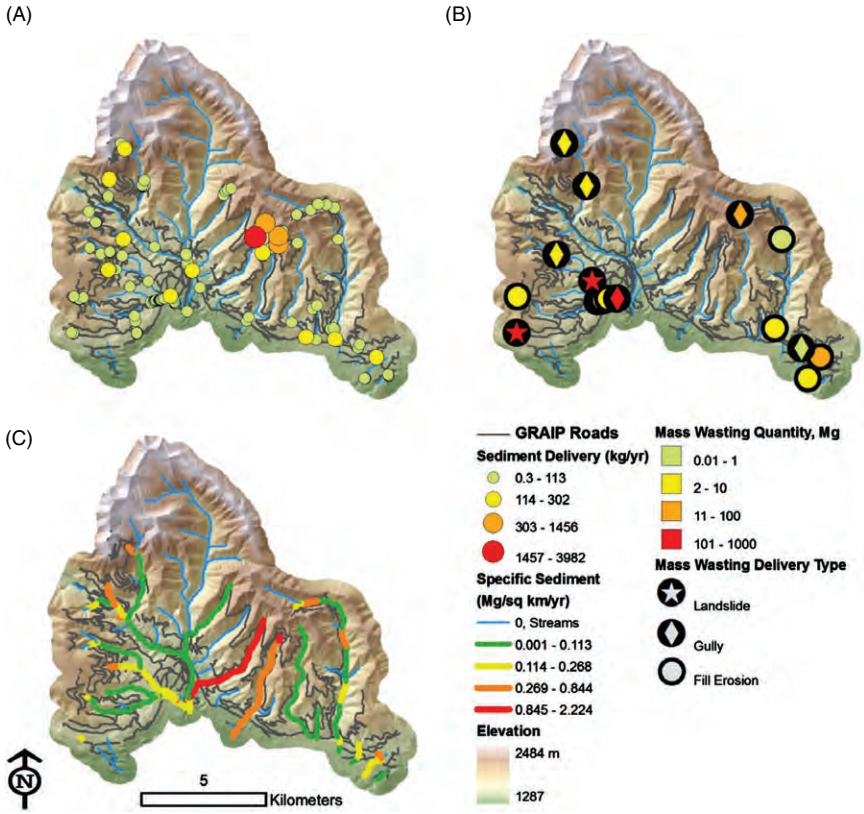

Figure 3. Spatial variability in sediment delivery from road surfaces for locations where road networks are connected to stream networks (A), locations and quantity of sediment from mass wasting (B), and the specific sediment production to stream from road networks (C) for a subwatershed in the SWCC study area.

with increasing road density but less clarity in such relationships with sediment delivery (Fig. S2). Our data support a model that natural disturbances and variability in hydrologic and geomorphic processes lead to a wide range of variability in streambed sediment that can be more and more constrained with intensive road development within a watershed.

The Importance of Context

The variability in the relationships we observed and comparisons to that of other studies (e.g. Cover et al. 2008) imply that all roads are not equal in their effects, and that topographic and geologic contexts matter. This is not surprising across inherent differences in geology, which control the characteristics of the material and the erosion rates. The SWCC is largely dominated by Belt Series metasedimentary parent materials that result in relatively low erosion rates (e.g. Sugden & Woods 2007), while the inference by Cover et al. (2008) is from a far more weathered granitic geology. But even within our study, the variability in sediment production and delivery with simple road density is substantial (Fig. 4A). Certainly, our results do not suggest that forest roads have limited effects on sediment delivery to streams everywhere in Belt Series geology but understanding the geologic as well as landscape, climatic, and so on (Pechenick et al. 2014) context matters. Borrowing inference from previous studies to justify action (i.e. road construction or restoration) or no-action may not be judicious.

Concomitantly, sediment that enters streams comes from a variety of both land management and natural sources and this complicates our ability to identify the effect on streambed sediments from just one source. For example, off road vehicle trails, cattle grazing, and forest management may also create sediment that eventually ends up in streams. Potential legacy effects (e.g. Harding et al. 1998) from past disturbances or even episodic events from road networks (e.g. mass wasting) may represent further contributions to streambed sediment that are difficult to measure with a single snapshot of existing watershed conditions. However, data identifying management intensity (e.g. livestock grazing) and legacy effects are extremely limited (Al-Chokhachy et al. 2010*a*). Isolating these sources from current road sediment contributions continues to be a major challenge but necessary to help managers prioritize where restoration of roads might be most effective.

Issues of Scale in Monitoring to Guide Road Restoration

The majority of existing stream habitat data within roaded (i.e. unpaved roads) and managed landscapes has been collected through large-scale monitoring programs by agencies

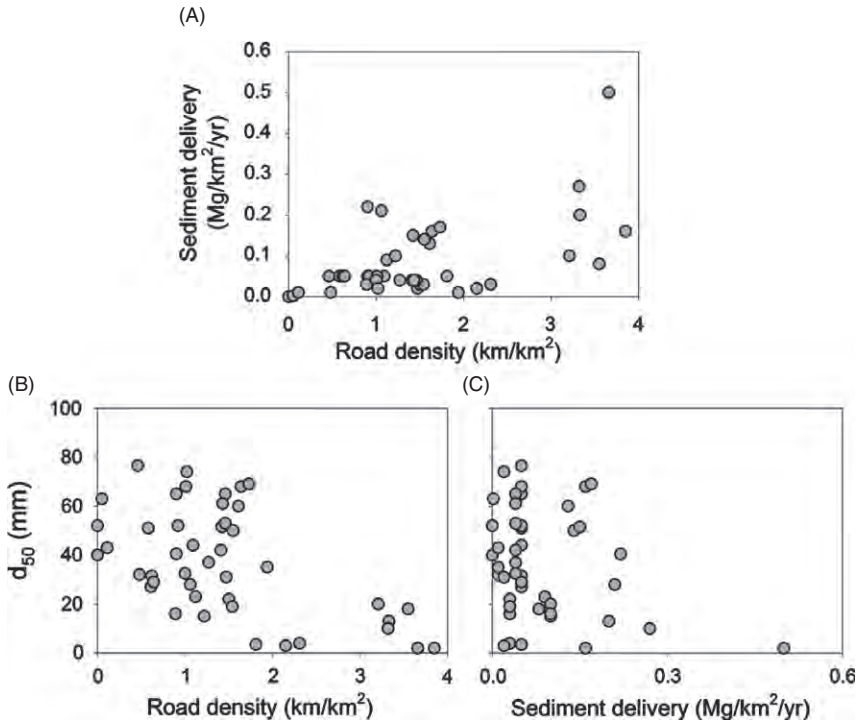

Figure 4. The relationship between estimates of sediment delivery estimated from GRAIP and road density in catchments upstream of streambed habitat sampling sites ($r = 0.43$; A) and median particle size (d_{50}) from low-gradient sites (<3.5% gradient) plotted against catchment-level estimates of road density ($r = -0.53$; B) and sediment delivery estimated from GRAIP ($r = -0.23$; C) for sampling locations in the Southwest Crown of the Continent in Montana.

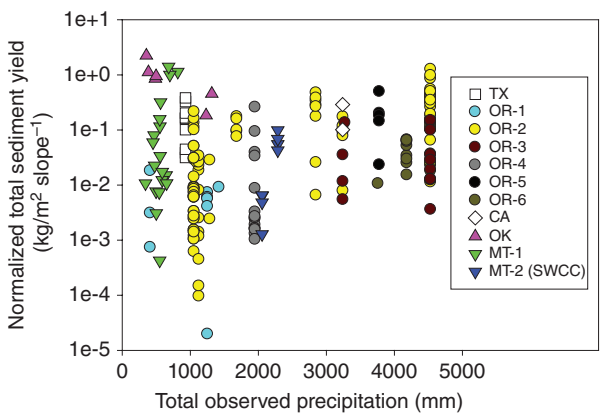

Figure 5. Estimates of sediment yield (normalized for road area and slope; log scale) across different precipitation regimes (measured in situ at individual plots) from studies in different regions of the United States. The locations of studies are indicated by U.S. state symbols; where multiple studies occurred within any given state, a number is used to denote a separate study.

including the U.S. Forest Service, U.S. Environmental Protection Agency, and so on designed specifically to monitor the effectiveness of management actions. While such programs may be essential in identifying landscape disturbance patterns across large landscapes (e.g. Kershner et al. 2004), there remains a paucity of applications where such data are aligned with detailed management disturbance information such as sediment production and delivery. This monitoring mismatch makes it challenging to accurately identify and prioritize hotspots of sediment sources, particularly through the use of office-mapped road density data alone (Fig. 1). Furthermore, in many cases road density provides an index of overall human disturbance, not the specific effects of sediment contributions from roads.

At first glance, labor intensive, in situ studies of sediment production, connectivity to stream networks, and measures of streambed sediment as demonstrated herein may seem cost prohibitive. Alternatively, the costs associated with naïve road restoration ($10,000–100,000/mile; Robinson et al. 2010) may not only have little effect on changing the conditions of streambed sediment but also detract from locations where resources could be used more effectively. For areas where streambed sediment may be limiting ecosystem processes, the costs of a detailed study that refines effective restoration choices and specifically develops defensible, sediment mitigation plans (e.g. Lisle et al. 2015) are likely to fall well below the costs of a broadly applied naïve road restoration program. Indeed the costs of a targeted study as opposed to the uncertainty of the benefits of restoration should be considered.

Understanding the Benefits of Road Restoration: Management Considerations

Part of the challenge in quantifying the benefits of road restoration to aquatic ecosystems is because of the limited number of studies that have monitored the results in a statistically rigorous

manner (Bernhardt et al. 2005). Detailed studies have examined sediment production from road segments (Reid & Dunne 1984; Sugden & Woods 2007, among others), and changes in sediment production through restoration actions (e.g. Switalski et al. 2004). However, few efforts have quantified how restoration actions (e.g. surfacing, obliteration and culvert removal) translate into changes in streambed sediment at the scale of an entire watershed.

Sediment production from road surfaces varies dramatically within and across ecological settings (Macdonald & Coe 2008; Dube et al. 2011), with over three orders of magnitude differences within similar ecoregions and precipitation regimes (Fig. 5). Differences in sediment production coupled with landscape-influenced differences in connectivity to stream networks and streambed sediment highlight the challenges in understanding the benefits of road restoration (Benda & Dunne 1997).

Although the results of our field study represent only a snapshot of time, they support management priorities that include addressing sediment sources at road crossings and closing roads to travel as options for reducing sediment risks in the SWCC. Infrastructure upgrades, best management practices, and removal/restoration provide mitigation opportunities at stream crossings (McCaffery et al. 2007). Promoting revegetation of roads through closure can dramatically reduce sediment production while still allowing some maintenance of infrastructure for future management and avoiding the high costs of complete road obliteration (Foltz et al. 2009). Ultimately, we suggest that cost-benefit analyses that explicitly consider ecological effects and financial obligations associated with different management approaches are warranted.

A Systematic Approach to Identifying Problem Roads

In some cases, road sediments that are related to culvert failures, landslides, and gullies present obvious locations for road rehabilitation or restoration. When these obvious problems are not present, a systematic approach that is more informative for identifying the need and means for remedying road-related issues may be warranted. An initial step in this approach would be to quantify if and where sediment levels (e.g. suspended sediment, streambed sediment) are exceeding regional baseline levels and/or are limiting biota (e.g. Shaw & Richardson 2001). This step is imperative, particularly where sediment effects on aquatic ecosystems are the justification for restoration. A logical next step would be to quantify road density information from readily available Geographic Information System (GIS) layers as well as potential other sources of sediment contribution (e.g. recreation) as measures of potential risk; however, the extent of forest roads may need to be verified as we documented greater than 30% more roads than reported in GIS layers. Where road networks exist upstream of sediment-impaired ecosystems, information regarding chronic and episodic source locations (e.g. mass wasting) may be necessary to identify hotspots of sediment production and effectively target restoration options (Luce et al. 2001). Region-specific measures of sediment production may be warranted given inherent differences in geology, soil types, precipitation, and the design, aggregation, and use of roads (Pechenick et al. 2014). Accumulation of studies will ultimately improve our understanding of the differences in sediment production from road networks, and concomitantly delivery to streams, across parent geology and soil types. Once refined, developing hybrid field-GIS-based approaches (e.g. Benda et al. 2007) that would allow platforms such as GRAIP to be regionally calibrated can allow managers to identify and prioritize locations for restoration activities (Fig. 3). Within this framework, we suggest that it may be imperative to incorporate monitoring approaches (Bernhardt et al. 2005) that will allow for robust assessments of the effectiveness of restoration opportunities.

Conclusions

Here, we specifically focused our review and analyses on linkages between sediment production and delivery from unpaved forest roads and streambed sediment metrics, but not changes in hydrology, fragmentation, or other alterations of terrestrial and aquatic ecosystems because of roads. The influence of forest roads on streambed sediment is often a major justification for restoration under programs such as the CFLRP.

Commonly, there is intense disagreement among biologists, hydrologists, foresters, and supervisors about the relevance of road derived sediment in streams. We find support for the conceptual model linking forest roads and streambed sediment, yet the severity of the effects of roads is highly variable—likely fostering such contention. Although we acknowledge that the inference from our review and case study mainly focused on montane areas of Northwestern United States, sediment from unpaved roads remains a global issue (e.g. Croke et al. 1999; Gruszowski et al. 2003; Sidle et al. 2006) and we consider our results germane in directing restoration efforts. Identifying hotspots of sediment delivery provides a framework for cost-effective and ecologically relevant restoration and evidence to garner public support where the closure/restoration of roads meets resistance from local residents.

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Supporting Information

The following information may be found in the online version of this article:

Appendix S1. Road inventory and sediment modeling.
Appendix S2. Streambed sediment.
Appendix S3. Quantifying road-stream habitat linkages.
Table S1. Multimodel results from multiple linear regression analyses linking road density and sediment production estimates from GRAIP with different measures of streambed sediment.
Figure S1. An example of a settling tank and outflow filter used in sediment plots for calibrating the GRAIP model.
Figure S2. The relationships between site-level estimates of percentage of fine sediment in pool tails (A, B) and percentage of subsurface fine sediment (C, D) with estimates of road density in catchments and sediment delivery estimated from GRAIP, respectively, for sampling locations in the Southwest Crown of the Continent in Montana.

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THE TONGASS NATIONAL FOREST
AMERICAN SALMON FOREST

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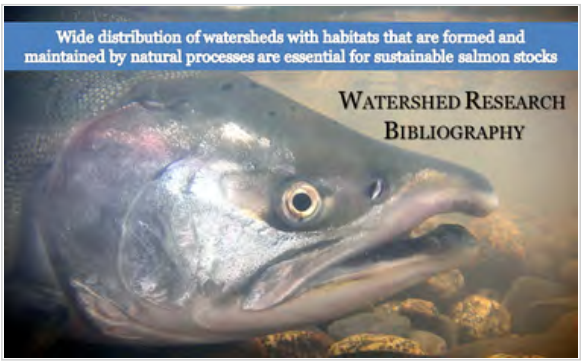

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The Tongass 77 Watersheds by Name / Acreage

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Southeast Alaska Salmon and Trout: a $1Billion Economic Engine

Tongass Economic Study – full report

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LETTER

Watershed complexity increases the capacity for salmon–wildlife interactions in coastal ecosystems

Jonathan B. Armstrong[1] | **Daniel E. Schindler**[2] | **Curry J. Cunningham**[3] | **William Deacy**[1,4] | **Patrick Walsh**[5]

[1]Department of Fisheries and Wildlife, Oregon State University, Corvallis, Oregon

[2]School of Aquatic and Fishery Sciences, University of Washington, Seattle, Washington

[3]College of Fisheries and Ocean Sciences, University of Alaska Fairbanks, Juneau, Alaska

[4]Arctic Network, U.S. National Park Service, 4175 Geist Road, Fairbanks, Alaska 99709 USA

[5]U.S. Fish and Wildlife Service, Togiak National Wildlife Refuge, Togiak, Alaska

Correspondence
Jonathan B. Armstrong, Department of Fisheries and Wildlife, Oregon State University, 104 Nash Hall, 2820 SW Campus Way, Corvallis, OR 97331.
Email: jonathan.armstrong@oregonstate.edu

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Abstract

Habitat alteration and species exploitation are fundamental issues in conservation, yet their interacting effects on food webs are rarely considered. We used a foraging model based on the Wood River basin (Alaska, USA) to explore how watershed development and commercial fisheries affect energy flow from sockeye salmon to brown bears. We found that, where salmon are abundant, fisheries can harvest large fractions of runs without substantially reducing bear consumption of salmon, but that watershed development could strongly reduce bear consumption if it shortens the duration of foraging opportunities by reducing population-level variation in salmon spawn timing. Habitats with the lowest resource abundance (small streams) were particularly profitable for bear foraging because they offer salmon at unique times of the season. This result challenges environmental impact assessments that assume ecological effects respond solely to changes in resource abundance.

KEYWORDS

ecosystem-based management, fisheries, phenology, predation, trade-off, watershed development

1 | INTRODUCTION

A primary challenge of conservation science is to quantify how the effects of human development propagate through biological communities to modify or degrade ecosystem functions and services (Hebblewhite et al., 2005; Koh et al., 2004; Sinclair & Byrom, 2006). A difficulty in anticipating such effects is that species interactions are governed not only by their abundances (Holling, 1959) but also the habitat context in which species interact (Werner, Gilliam, Hall, & Mittelbach, 1983), both of which are affected by human activities. For example, humans can reduce energy flows to predators by reducing prey abundance through harvest (Cury et al., 2011) or by homogenizing and fragmenting habitat features that mediate predator–prey interactions (Andruskiw, Fryxell, Thompson, & Baker, 2008). However, it is often not clear how

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harvest and habitat modification interact to affect ecosystem processes that support important services or species at risk. Here, we explore a case study involving marine commercial fisheries for Pacific salmon (*Oncorhynchus* spp.), flows of these marine-derived resources to brown bears (*Ursus arctos*) in coastal watersheds, and how watershed development may erode the capacities of ecosystems to support inland food webs dependent on salmon subsidies.

Anadromous Pacific salmon support a variety of ecosystem functions in coastal watersheds (Gende, Edwards, Willson, & Wipfli, 2002; Schindler et al., 2003). After accumulating most of their body mass in the ocean, salmon return to their natal rivers to spawn and then die, generating a high-quality resource pulse that supports a wide variety of aquatic and terrestrial consumers in coastal watersheds (Willson & Halupka, 1995). Consumers ranging from insects to bears scavenge or prey on salmon and their eggs. At higher latitudes where growing seasons are shorter (and winters longer), salmon provide a critical energy source for a wide variety of taxa (Armstrong & Bond, 2013; Bentley et al., 2012). Marine fisheries often harvest a large proportion of salmon returning to watersheds, leading to concerns that harvest reduces the resource base supporting terrestrial and freshwater food webs in salmon watersheds and that an ecosystem-based approach to salmon management would more explicitly consider the roles of salmon as a resource subsidy to watershed predators (Darimont et al., 2010; Levi et al., 2012).

A growing body of work demonstrates that intraspecific diversity in salmon attributes mediates energy flows to both fisheries and predators (Deacy, Leacock, Armstrong, & Stanford, 2016; Ruff et al., 2011; Schindler et al., 2013). This diversity is associated with local adaptation of salmon to specific habitat conditions such as water temperature, water depth, stream gradient, and gravel size (Fraser, Weir, Bernatchez, Hansen, & Taylor, 2011; Hilborn, Quinn, Schindler, & Rogers, 2003; Taylor, 1991). For example, geomorphic conditions affect stream hydrology and thermal regimes that cause local selection pressures on optimal spawn timing (Larson, Lisi, Seeb, Seeb, & Schindler, 2016), which can differ by weeks or even months among populations occupying a single river basin (Lisi, Schindler, Bentley, & Pess, 2013; Schindler et al., 2010). Because salmon are primarily available to terrestrial and aquatic predators only when they are actively spawning in shallow waters, seasonal variation in spawn timing among populations generates resource waves that prolong the duration of salmon foraging opportunities for inland consumers (Deacy et al., 2016; Schindler et al., 2013). Thus, consumers that can move to exploit the seasonal spatial progression of spawning salmon benefit more from salmon resources than those in systems with shorter resource waves or in ecosystems where there are impediments to movement (Armstrong, Takimoto, Schindler, Hayes, & Kauffman, 2016; Deacy et al., 2018).

Across the Northern Pacific Rim, there is increasing risk of development in watersheds that support salmon and their predators such as bears (Darimont et al., 2010). For example, the Bristol Bay region of western Alaska is famous for its prolific commercial salmon fisheries (Hilborn et al., 2003; Schindler et al., 2010), tourism focused on bear viewing (Richardson, Huber, & Loomis, 2017), and recreational fisheries for species that rely heavily on salmon resource subsidies (Bentley et al., 2012; Dye & Borden, 2018). However, the watersheds that provide spawning and nursery habitat for anadromous salmon are experiencing increasing pressure from resource extraction (Hébert & Brock, 2017), hydroelectric development (Cherry et al., 2017), and climate warming (Winfree et al., 2018). Development pressures have the potential to degrade watershed features that modify a range of evolutionary and ecological characteristics of salmon populations throughout the region.

Proponents of development projects often argue that direct footprints are relatively small and pose risks to only a small fraction of a watershed. However, empirical studies of bear–salmon interactions suggest that some small salmon populations can contribute disproportionately to the foraging opportunities of bears. Salmon populations that spawn in small tributary streams are far more vulnerable to predation than populations that spawn in rivers or on lake shores (Quinn, Wetzel, Bishop, Overberg, & Rogers, 2001). Populations returning to small streams also spawn the earliest due to colder temperatures (Lisi et al., 2013), and may be the only salmon available for a substantial window of the salmon-foraging season (Schindler et al., 2013). Thus, although not numerically abundant, salmon populations occupying small habitat features with unique local selection pressures may provide a disproportionately large contribution to flows of marine-derived resources to terrestrial food webs in coastal watersheds.

We developed an empirically parameterized simulation model to explore the interactions between watershed development (which could reduce variation in spawn timing by eliminating populations or homogenizing habitat) and commercial fishery harvest in controlling foraging opportunities for terrestrial predators. Our objectives were to (a) characterize the relationship between salmon abundance (primarily determined by fishery exploitation) and salmon population diversity (a product of watershed complexity) in regulating salmon foraging opportunities for brown bears, (b) assess the degree to which erosion of salmon population diversity affects bear foraging opportunities, and (c) quantify how alternative conservation strategies might fare in maintaining bear foraging opportunity in the face of watershed development. Our results demonstrate the need to identify and protect novel habitats that provide unique opportunities for predator–prey interactions but may not necessarily be dominant features of the landscape. We found that streams, which harbor smaller populations of salmon, contribute disproportionately to the

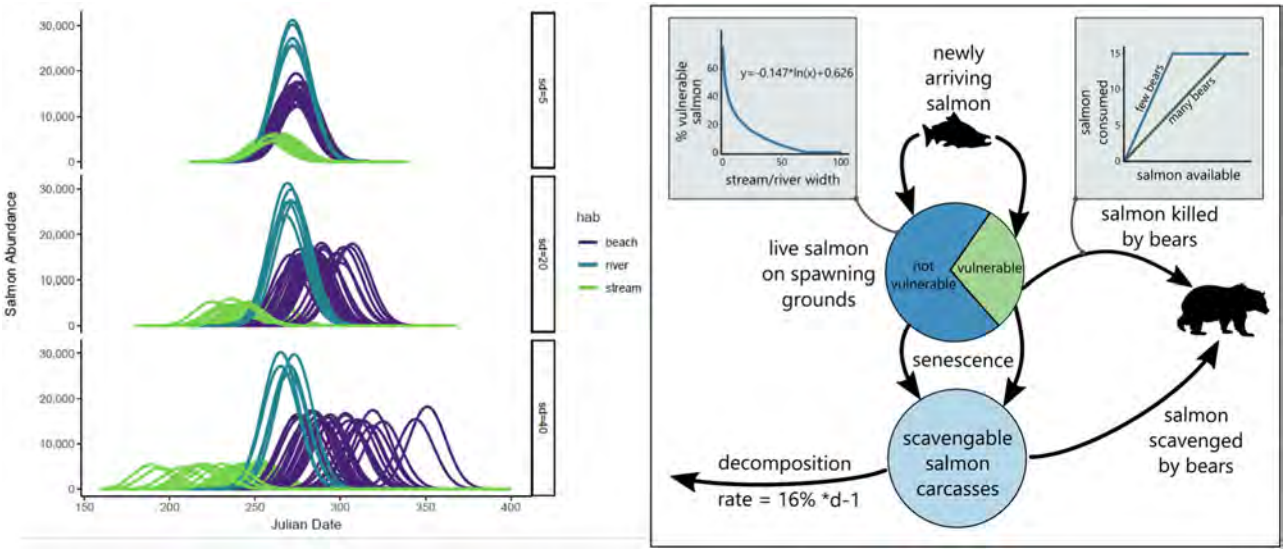

FIGURE 1 Summary of key components in the bear-salmon foraging model. (a) Plot illustrating simulated salmon portfolios in which a total of 1.5 million fish (the escapement after commercial fishing) occur in 60 discrete populations (individual lines) spawning in three habitat types (indicated by line color). Panels show different levels of phenological variation (standard deviation = 5, 20, and 40 top-to-bottom) that prolongs bear foraging opportunity. (b) Flow chart demonstrating key aspects of bear-salmon foraging represented in our simulations

flows of marine-derived resources to important species in coastal food webs, even though these habitats are commonly assumed to have little importance at the ecosystem level.

2 | METHODS

2.1 | Model summary

We simulated individual brown bears foraging across a landscape containing salmon populations that vary in abundance, spawning phenology, and vulnerability to predation (Figure 1, full details in the Supporting Information). The response variable of our simulations was the cumulative number of salmon consumed, averaged across the bear population. Salmon portfolios were based on the Wood River system of southwest Alaska, one of the only large watersheds with comprehensive data on salmon population attributes that mediate bear foraging opportunity (Schindler et al., 2013). This basin is approximately 3,200 km^2 and contains about 60 sockeye salmon populations that spawn in tributary streams, lake shores, or larger rivers. We only considered sockeye salmon (*O. nerka*) because they comprise >99% of the Pacific salmon returning to the Wood River system (Alaska Salmon Program *unpublished data*). We assume that bear foraging opportunities are regulated by five mechanisms: (a) salmon abundance (Levi et al., 2012; Quinn, Gende, Ruggerone, & Rogers, 2003; Van Daele et al., 2013), (b) habitat-mediated prey vulnerability (Quinn et al., 2001), (c) satiation (Gende, Quinn, & Willson, 2001; Hilderbrand, Jenkins, Schwartz, Hanley, & Robbins, 1999), (d) time constraints alleviated by phenological variation (Deacy et al., 2018; Schindler et al., 2013), and (e) intraspecific competition (Ben-David, Titus, & Beier, 2004; Egbert, Stokes, & Egbert, 1976; Rode, Farley, & Robbins, 2006).

2.2 | Scenarios explored

To explore the relationship between salmon fisheries and inland food webs, and how this was modified by watershed complexity, we varied escapement from 0.2 to 6 million fish to reflect observed variation around the Wood River's management target of 0.7–1.8 million (high values reflect large runs that exceed fisheries capacity; Erickson, Brazil, Chang, McKinley, & Clark, 2015). To explore how phenological variation mediated the effect of salmon abundance on bears, we simulated across three levels of variation in sockeye salmon spawn timing across the watershed ($\sigma phen$): 5, 20, and 40. The Wood River's salmon populations have a $\sigma phen$ value of approximately 22 and a total duration of approximately 3 months (Lisi et al., 2013). Areas with high geomorphic diversity and multiple salmon species, such as the Copper River Delta (Campbell, Dunham, Reeves, & Wondzell, 2018), exhibit $\sigma phen > 50$ and have spawning salmon present for >6 months (Gordon Reeves U.S. Forest Service *unpublished data*).

Next, we explored how watershed development that eliminates salmon populations (e.g., mining that degrades, dewaters, or contaminates spawning sites; U.S. EPA 2014) could affect seasonal bear consumption of salmon. We considered three alternative rules for ordering the removals: (a) "protect the most abundant salmon populations" (remove in reverse order of population size), (b) "remove populations randomly,"

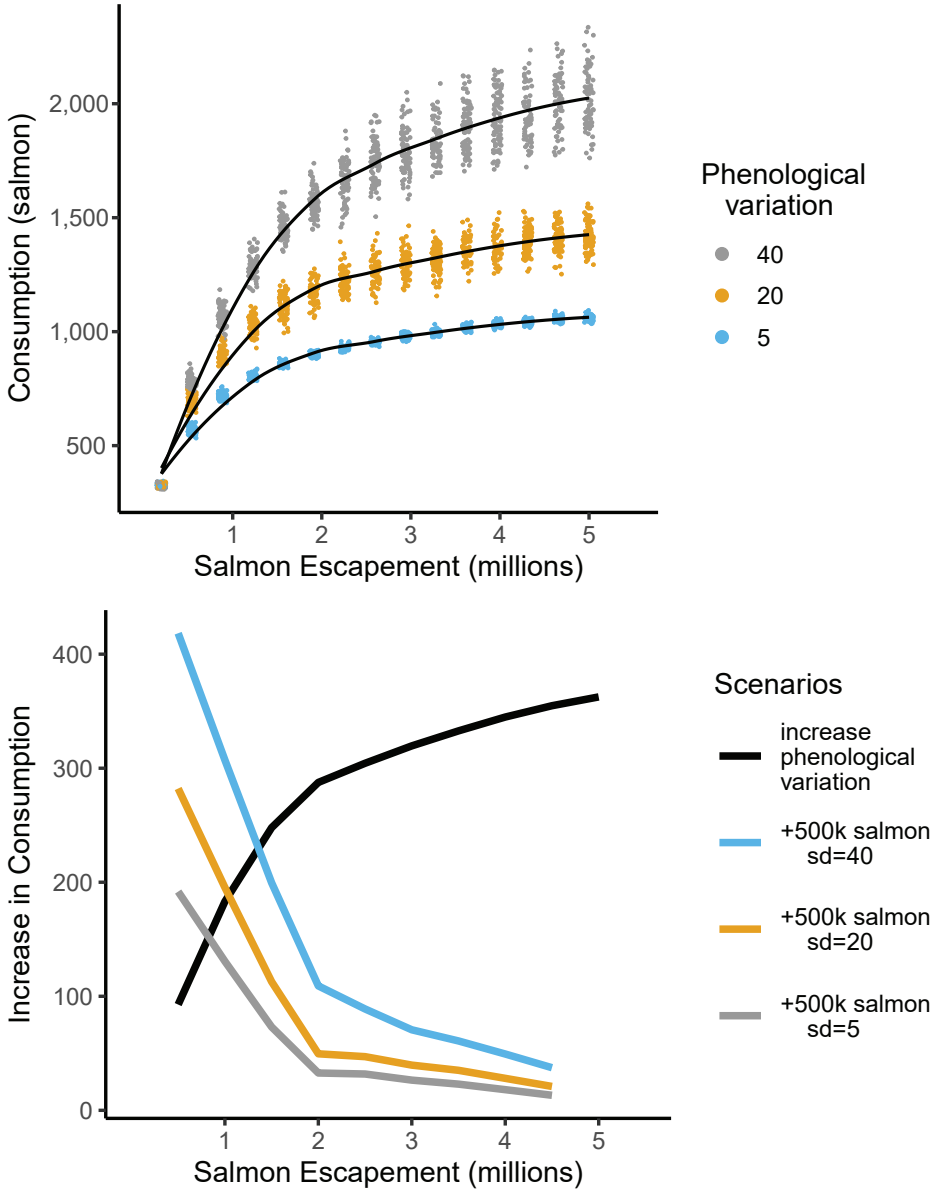

FIGURE 2 (a) Simulation results showing the relationship between salmon escapement (fish remaining after commercial fishing) and bear consumption (mean number of salmon per bear), as a function of phenological diversity among salmon populations (point colors indicate standard deviation). Lines show local polynomial regressions fit to data using weighted least squares. (b) Line plot comparing the effect sizes of salmon escapement and phenological diversity on salmon consumption by bears, across a range of salmon escapements. *X*-axis indicates initial salmon escapement level; *y*-axis shows change in salmon consumption that results from increasing the escapement by 500,000 fish (solid lines) or increasing the standard deviation of phenology from 5 to 20. For the effects of escapement, color of solid line indicates different levels of phenological variation as depicted in (a). Both plots show data for the intermediate competition scenario

and (c) "maintain phenological variation" (remove populations with the most common spawn timing).

3 | RESULTS

3.1 | Relationship between fishery exploitation and consumption of salmon by bears

Salmon escapement had a positive saturating effect on bear foraging (Figure 2). At salmon escapements less than approximately 0.5 million fish, bears consumed less than approximately 500 salmon and responded strongly to increases in salmon abundance. In contrast, at our maximum escapement levels of 6 million fish, bears consumed nearly 1,200–2,500 salmon and showed little response to further increases in salmon abundance. Phenological variation had strong positive effects on bear foraging that were enhanced by salmon escapement, but dampened by competition (Figures 2 and S1).

Phenological variation mediated trade-offs between salmon fisheries and bears by governing the rate at which

bear foraging saturated with increasing salmon abundance (Figure 2b). Specifically, variation in spawn timing increased (a) the marginal value of salmon at low escapements, (b) the escapement level at which the marginal value of salmon approached zero (the point of saturation), and (c) the foraging opportunity of bears at saturating escapements (Figure 2b). At a threshold of approximately 2 million salmon, adding additional escapement had little effect on bear consumption, whereas phenological variation could still increase bear foraging by nearly 50–100% (compared to a situation with homogeneous run timing). The effect size of phenological variation increased with increasing salmon escapement. In combination with the saturating effect of escapement, this generated two domains in the response of bears. At high salmon abundance, bear consumption was primarily sensitive to phenological variation, whereas at low salmon abundance, bear consumption was primarily sensitive to escapement levels (Figure 2b).

3.2 | Value of small populations

In our simulations, salmon populations in streams were approximately 1/3 and approximately 1/5 as abundant as those in beaches and rivers, respectively. Summed together, stream spawning salmon populations comprised only about 20% of total escapement across the watershed. However, because stream populations exhibit unique phenologies and are more vulnerable to predation, they contributed disproportionately more to bear foraging, providing 48% ± 17% (mean ± 1 standard deviation) of the total salmon consumed by bears over the course of a season. In contrast, beach spawning populations contributed 58% of escapement and provided 33% ± 9% of salmon consumption, and river populations contributed 23% of escapement and 15% ± 4% of consumption. Thus, small populations of salmon contributed roughly half of total bear consumption while representing only one-fifth of total salmon escapement (Figure 3).

3.3 | Effects of eroding salmon portfolios

Eliminating salmon populations always reduced bear foraging levels; however, the magnitude of this effect was highly sensitive to which populations were removed. In simulations that protected the most abundant salmon populations (making streams with smaller populations most vulnerable), low levels of development caused steep declines in bear foraging level (Figure 4). For example, in the low and medium competition scenarios, removing only 20% of total salmon escapement decreased bear consumption by nearly 30% (Figure 4). In contrast, the alternative strategy that maintained phenological diversity by protecting small streams showed lower declines in bear foraging (~10%, Figure 4). Higher levels of competition among bears resulted in greater sensitivity of bear foraging to development and caused the alternative prioritization strategies to become more similar in their performance (Figure 4).

4 | DISCUSSION

Intact watersheds support a wide range of life history diversity in salmon populations, some of which is expressed as variation in spawn timing that extends the duration of salmon foraging for bears and other wide-ranging consumers (Deacy et al., 2016; Schindler et al., 2013). Although there is increasing interest in managing fisheries to ensure adequate salmon abundance for inland food webs (Darimont et al., 2010; Levi et al., 2012), our results emphasize that managing landscapes and fisheries to maintain phenological diversity is probably at least as important. In our models, increasing salmon escapement within the current range of management goals had positive effects on bear consumption (Figure 2), which corroborates concerns over exploitative competition between fisheries and bears (Hilderbrand, Farley, Schwartz, & Robbins, 2004). However, increasing escapement from the average level observed (1.5 million, $n = 56$ years) to 2-times the average (3 million) only increased consumption by 15–31%. In contrast, at the average observed escapement level, increasing the phenological variation from our minimum to maximum simulated value increased bear consumption by 52–75%. This model prediction is supported by recent empirical work beyond the Wood River Basin; Kodiak brown bears that tracked phenological variation to prolong their salmon foraging exhibited nearly 40–90% higher consumption levels (Deacy et al., 2018). Similarly, salmon species diversity (a proxy for phenological variation) outperformed salmon biomass in predicting the contribution of salmon to black bear diets in coastal British Columbia (Service et al., 2019). Although it is difficult to compare abundance and diversity metrics in a standardized fashion, we believe the variation in phenology we simulated across represents realistic contrasts found in nature that are appropriate to compare to realistic contrasts in abundance.

Phenological diversity within salmon species emerges from local adaptation to different regimes of flow, temperature, and biological productivity (Fraser et al., 2011). Ongoing climate change may reduce the thermal heterogeneity currently expressed across watersheds in Bristol Bay (Armstrong & Schindler, 2013; Lisi et al., 2013) and in other parts of Alaska (Winfree et al., 2018) by reducing the effect of snowmelt on stream hydrologies (Adelfio, Wondzell, Mantua, & Reeves, 2019). Hatchery salmon production can replace diverse populations of wild fish with homogenous hatchery stocks (Brenner, Moffitt, & Grant, 2012; Hilborn & Eggers, 2000) and exacerbate the challenge of maintaining population diversity in mixed-stock fisheries (Naish et al., 2007). Salmon populations exhibit a wide range of productivities but are

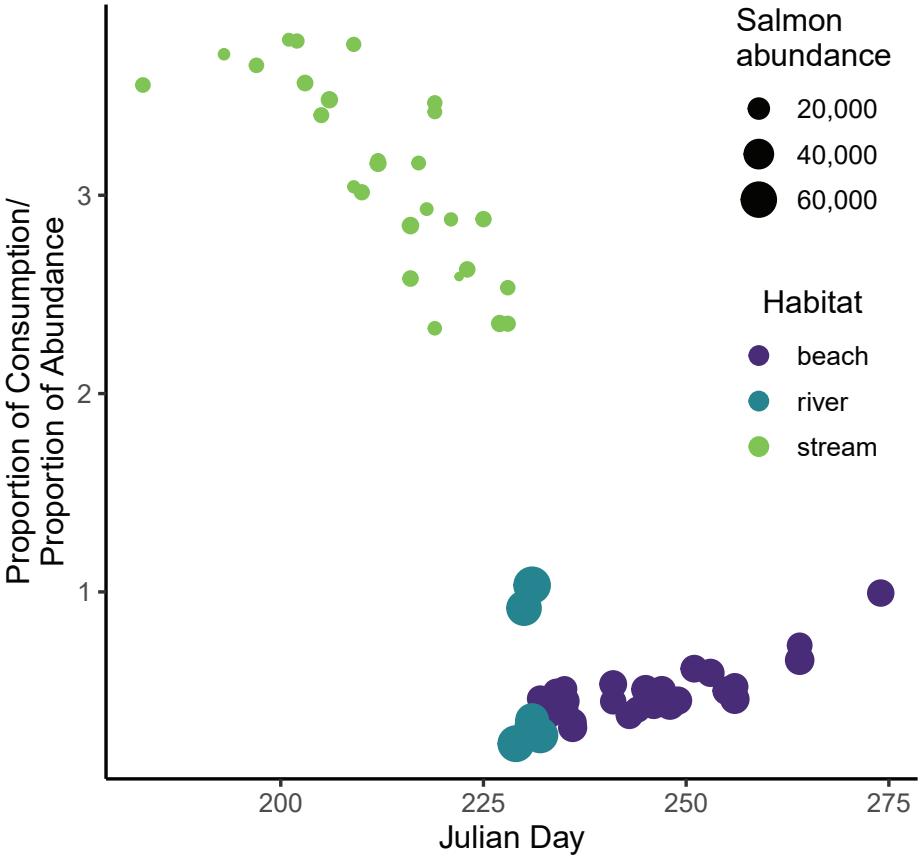

FIGURE 3 Plot depicting how salmon populations (represented as points) differentially contribute to bear consumption as a function of their habitat type and spawning phenology. *Y*-axis shows each salmon population's contribution to cumulative bear consumption divided by its contribution to total salmon abundance (i.e., watershed-level escapement). *X*-axis shows date of spawning. Color indicates habitat type and size of dot indicates population abundance

harvested in aggregate. Fishing a stock complex at maximum sustainable yield results in overfishing of less productive populations and has led to intense controversy regarding whether unproductive populations are worth protecting at the expense of fisheries yields (Price, Darimont, Temple, & MacDuffee, 2008; Walters & Martell, 2004; Wright, 1993). Our results show that some of these small populations may be particularly important for maintaining food webs at landscape scales. We found that lack of protection for small populations could severely affect bears, causing >2× more reduction in salmon consumption compared to alternative conservation strategies (Figure 4). Our results are likely conservative because we assumed habitat mediates predation but not scavenging; in reality, small streams likely offer higher efficiency for scavenging as well as predation. More empirical work is needed to understand the quantitative effect of human actions on phenological diversity and how consumers respond to uneven declines in abundance across salmon stock complexes (Price et al., 2019).

Although our model was based on the Wood River Basin, our results are likely generalizable to other basins that support major salmon fisheries. The key characteristic of the Wood River system is that its stream spawning salmon populations generate a small fraction of total salmon abundance, but provide earlier spawning salmon in habitats more conducive to bear predation. These phenomena reflect underlying mechanisms that should apply broadly. The area of suitable spawning habitat (i.e., gravel substrate) in tributary streams is typically less than that available in lake shores and large rivers (Burgner et al., 1969), limiting the average production from this habitat type. Tributary streams are generally cooler than lake shores and rivers and cooler temperatures are associated with earlier spawning phenologies in salmon. Last, tributary streams are confining habitats that render salmon more vulnerable to predators than lake shores or rivers. Although more work is needed to robustly quantify these traits across watersheds, they appear to hold across sockeye salmon stock complexes that sustain major fisheries (Deacy et al., 2016; Hilborn et al., 2003).

Our simulations demonstrate that reducing fishing pressure cannot offset losses of phenological diversity because diversity and abundance affect complementary dimensions of bear foraging. Diversity in spawn timing determines how long salmon are present on the landscape, whereas salmon

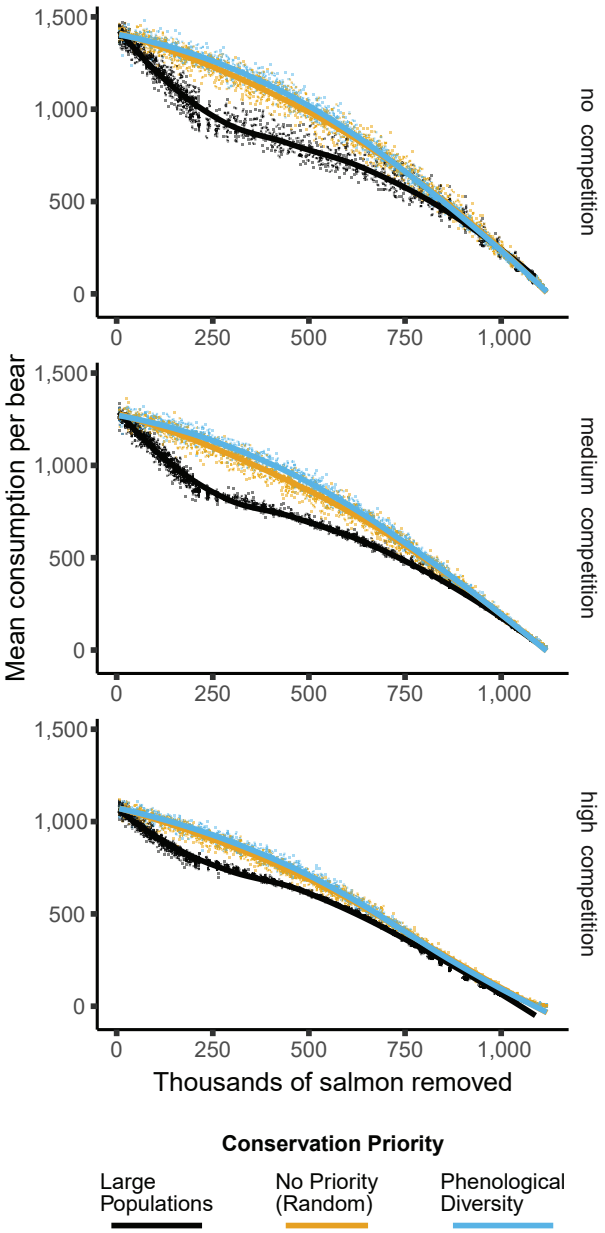

FIGURE 4 The simulated effects of alternative patterns of watershed development on the foraging opportunity of bears. In each simulation, salmon populations were removed sequentially from the watershed, following three alternative conservation prioritization strategies: (a) protect large populations (black), (b) maintain phenological diversity (blue), and (c) random (orange). Panels show results for different levels of interference competition

abundance primarily influences bear energy intake on the days when salmon are present. The positive effect of abundance is capped by physiological capacity (Hilderbrand et al., 1999), creating an asymptotic relationship between salmon density and the number of salmon that bears consume (Figure 2; Quinn et al., 2003). Watershed development that reduces phenological diversity will reduce the capacity for salmon runs to support bears and other wide-ranging salmon consumers. Our development scenarios eliminated populations entirely (or completely prevented bears from accessing them) rather than reducing their productivity or accessibility. Future work could consider how lesser impacts scale up to affect bears, or how additional ecological complexity, such as selective consumption (Gende et al., 2001) and dominance (Gende & Quinn, 2004), further mediates energy flows from salmon to bears.

A serious challenge in conservation is to understand which aspects of ecological complexity are most important for maintaining ecosystem functions. Ecosystem-based fisheries management accounts for interactions among multiple species, but typically considers simplified food webs with energy flows mediated solely by prey abundance (Christensen & Walters, 2004). We incorporated two empirically documented phenomena, intraspecific diversity in prey attributes and habitat-mediated predation risk, and found that they strongly mediate how the abundance of salmon translates into bear foraging opportunity. By considering the spatial and temporal structure of this ecosystem, we arrive at a very different conclusion about the importance of certain landscape features over others. Even habitat patches that produce small contributions to overall resource production can be disproportionately important to consumers if they provide novel opportunities in space or time. Risk assessments of watershed development activities need to account for the value of such novel features if their intent is to protect key ecosystem integrity.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

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Windthrow and recruitment of large woody debris in riparian stands

Devesh Bahuguna, Stephen J. Mitchell*, Yosune Miquelajauregui

Department of Forest Sciences, University of British Columbia, 2424-3041 Main Mall, Vancouver, BC V6T1Z4, Canada

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ABSTRACT

To document the impacts of windthrow in riparian leave strips and identify the components needed for small stream large woody debris (LWD) recruitment modeling, we monitored nine small streams at a temperate rainforest site in coastal British Columbia. This study was a component of a larger integrated study of forest management impacts on small streams. A series of small clearcuts were harvested in 1998 in a 70-year-old second growth stand that had regenerated naturally following logging and wildfire. Three cutblocks each were assigned to 10 m and 30 m buffer width treatments and three areas were assigned as unharvested controls. Seven years after the 1998 logging, all logs greater than 10 cm diameter that spanned at least part of stream channel width were measured. A total of 179 logs were recorded. Post-harvest windthrow was higher in the 10 m buffer treatment, while competition-related standing tree mortality was higher in the controls. The major windthrow events had occurred in the first and second years after logging of adjacent stands. There was no significant difference in the number of spanning and in-stream logs in the 10 m, 30 m buffer and control treatments. More than 90% of the LWD was in the 10–30 cm diameter classes. The majority of logs were oriented perpendicular to the stream channel. At the time of measurement, the majority of these trees were still suspended above the stream channel, indicating that the recruitment of logs into the stream channel is a long-term process. Time to recruitment into the channel is dependent on log and valley geometry, log size, species, and log condition prior to toppling. Log height above stream was negatively correlated with log decay class and valley width. Log length was negatively correlated with state of decay, and many windthrown logs were in an advanced state of decay before they entered the stream.

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1. Introduction

Changes in forest practice regulations in British Columbia and the northwest United States in the past two decades have lead to greater protection of small streams during timber harvesting. Forest policies require retention of treed buffer strips along relatively small stream channels if they have fish populations (Wang et al., 2002). These unharvested strips are intended to minimize impacts of forest management activities on water quality, aquatic ecosystems and riparian community diversity (BCMOF, 1995). In-stream large woody debris (LWD) helps to structure fish habitat (Bisson et al., 1987), shape channels (Swanson et al., 1976), and trap sediments (Swanson and Lienkaemper, 1978; Gurnell et al., 2005). Larger wood pieces tend to remain where they are delivered in small streams, providing long-term benefits (Gurnell et al., 2002). The transfer of LWD from streamside forests to the stream and river systems creates a strong linkage between terrestrial and aquatic ecosystems (Lienkaemper and Swanson, 1987). The process of transfer of LWD from forests to streams is complex, and occurs in phases, starting with tree fall.

In riparian buffer strips, post-harvest windthrow is a major source of tree falls. Approximately 15% of cutblock boundary segments in wind exposed areas of coastal BC are partially windthrown following harvesting (Lanquaye-Opoku and Mitchell, 2005; Scott and Mitchell, 2005), and riparian buffers are particularly susceptible (e.g. Steinblums et al., 1984; Rollerson and McGourlick, 2001). To design effective riparian prescriptions, we need to understand the ecological impacts of windthrow in both the short and longer term. Potential impacts include loss of overstory, introduction of LWD into the streams, pulse introduction of foliage and fine branch materials, loss of bank stability and exposure of sediment sources (Lewis, 1998; MacDonald et al., 2003).

There are a number of models that aim to characterize the input, storage and loss of LWD in stream systems (e.g. STREAMWOOD, Meleason, 2001; RAIS, Welty et al., 2002; AQUAWOOD, Wei, 2005b). In these models, tree fall rates are usually tied to standing tree mortality rates. Tree fall direction is either treated as random or is conditioned by the user based on expert knowledge or empirically derived data. If it is explicitly dealt with at all, windthrow in newly exposed riparian buffers is often viewed as a pulse source of LWD input (e.g. RAIS, Welty et al., 2002). Given the geometry of

* Corresponding author. Tel.: +1 604 822 4591.
E-mail address: stephen.mitchell@ubc.ca (S.J. Mitchell).

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Table 1
Description of variables.

Variable	Label	Description
Channel form	ChFrm	A category to classify the channel as narrow or broad based on valley width index
Narrow channel type	CB	Constrained by bedrock
	CH	Constrained by hill slope
	CF	Constrained by alluvial fan
Broad channel type	US	Unconstrained predominantly single channel
	UA	Unconstrained anastomosing several complex interconnecting channels
	UB	Unconstrained braided channel (numerous, small channels often flowing over alluvial deposits)
	CT	Constraining terrace
	CA	Constrained by alternating terraces
	CL	Constrained by land use
Valley form	VFrm	General description of the valley cross-section with emphasis on the configuration of valley floor and classify into narrow or broad based on valley width index
Narrow valley type	SV	Steep V-shaped valley or bed rock gorge (side slope $\geq 60^\circ$)
	MV	Moderate V-shaped valley (side slope $>30^\circ$, $<60^\circ$)
	OV	Open V-shaped valley (side slope $\leq 30^\circ$)
Broad valley type	CT	Constraining terraces. Terraces high and close to active channel
	MT	Multiple terraces. Surface with varying height and distance from the channel
	WF	Wide active flood plain. Significant portion of valley floor influenced by annual floods
Bankfull channel width	BCW	Distance across channel at bankfull flow
Valley floor width	VFW	Distance of valley across channel
Valley floor index	VFI	It is a ratio of VFW to BCW
Distance along	Dist_along	Distance of log from point of commencement
Status		SL: standing live; LL: live leaning; DB: dead broken; SD: standing dead; DL: dead leaning; UR: uprooted
Species	Spp	Hw: western hemlock; Cw: western redcedar; Fd: Douglas-fir; Ss: sitka spruce; Ep: Paper birch; Dr: red alder; Mb: Maple
Decay class	Deccls	Classification system for logs based on decay (Table 2) (Bartels et al., 1985)
Orientation	Brg	Orientation to the top of log and orientation of stream measured with compass in degrees
Diameter at breast height	DBH	Diameter at breast height measured with diameter tape
Diameter at mid-stream	DMC	Diameter of log measured at middle of stream
Bankfull channel width length	BCWL	The portion of log in active channel measured with tape
Total length	TL	The total length of log
Debris type		RN: rootwad attached to ends; N: broken ends; C: cut ends
Base diameter	Base_dia	Diameter at base of the downed logs
Length mid-stream	Len_midcreek	Distance of log from mid-stream to the base of the tree
Span length	Span_leng	Distance between two suspending points of log
Height above stream	HAS	Height of log from bankfull height of stream

small stream valleys, it is more likely that downed trees, whether from standing tree mortality or from pulses of windthrow will be suspended above small streams rather than immediately entering the channel. The process by which these spanning logs enter the channel, and log condition at the time of channel entry is not well understood or modeled. The objectives of this research project were to:

(1) Evaluate the effect of riparian buffer width on windthrow and LWD recruitment and contrast this with unharvested controls.
(2) Investigate the condition of LWD.
(3) Document the geometry of post-harvest windthrow in riparian buffers of different widths.
(4) Develop the framework for a process model that simulates the supply of LWD of windthrow origin to streams within riparian buffers.

2. Methods

This study is a component of a larger integrated study of forest management impacts on small streams in second growth forests. The riparian areas investigated in this study are in dense young-mature conifer-dominated forests that have developed following harvest and wildfire. This is a common forest type in the low elevation areas of the north west coast of North America and small, fish bearing streams are abundant in these forests. The small streams riparian buffers experiment is located in the foothills of the Coast Mountains, approximately 60 km east of Vancouver, British Columbia (Feller and Sanders, 1999; Moore and Richardson, 2003). The climate is maritime and characterized by dry, warm summers and wet, cool winters. Total precipitation ranges between 2200 mm and 3000 mm per year. Snow falls only occasionally at these low elevations (120–450 m). Soils are shallow and are derived from glacial till and glacio-marine deposits. The topography varies from flat to hilly and gently rolling, with some bedrock knolls. The underlying geology of the study site is quartz diorite, diorite and granodiorite. The stands in the study area naturally regenerated following logging and wildfire in the 1930s and are dense and uniform in structure. The tree species include western hemlock (*Tsuga heterophylla* (Raf.) Sarg.), western redcedar (*Thuja plicata* Donn ex D. Don), Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), paper birch (*Betula papyrifera* Marsh.), and red alder (*Alnus rubra* Bong.).

2.1. Experimental design

Two buffer width treatments (10 m and 30 m on each side of the stream), and unharvested controls were each replicated three times within the riparian buffers experiment. There were also three fully harvested blocks (0 m buffers), but these treatment units were not included in the LWD study since they had no post-harvest LWD recruitment. Harvesting commenced in the fall of 1998. Overstory vegetation plots were measured annually. These vegetation

Table 2
Classification of LWD decay classes (from Bartels et al., 1985; MOFR, 2007).

	Class 1	Class 2	Class 3	Class 4	Class 5
Bark	Intact	Intact	Trace	Absent	Absent
Twigs	Present	Absent	Absent	Absent	Absent
Texture	Intact	Intact to partly soft	Hard large pieces	Small, soft blocky pieces	Soft and powdery
Shape	Round	Round	Round	Round to oval	Oval
Color of wood	Original color	Original color	Original to faded	Light brown to reddish brown	Red brown to dark brown
Portion of tree on ground	Tree elevated on support points	Tree elevated on support points but sagging slightly	Tree sagging near ground	All of tree on ground	All of tree on ground
Invading roots	None	None	In sapwood	In heartwood	In heartwood

plots were 15 m long and 4 m wide, ran lengthwise parallel to the stream, and were repeated at 2 m and 15 m from the stream bank on each side of the stream. These 4 strip-plot clusters were replicated at two locations along the stream within each treatment unit. They represented vegetation dynamics for the buffers experiment (Miquelajauregui, 2008) and provided tree mortality data for this LWD study. Within each strip plot, the species, diameter at 1.3 m (DBH), height and status (standing live, standing dead, uprooted) in each year were recorded for each tree, starting the year prior to harvesting.

Transects were established up the centre of each stream within each treatment unit in the summer of 2006 using the protocol of Moore et al. (2002) as a basis for sampling and measurements. Since each treatment unit was at least 150 m long, we chose a transect length of 150 m. The streams were all under 5 m wide, and were incised, relatively straight, and constrained primarily by hill slopes and narrow fluvial terraces. To characterize stream conditions, each stream was divided into reaches whenever there was a major change in orientation of the stream, or in the channel or valley form. The bankfull channel width (BCW) and valley floor width (VFW) were measured in two locations within each reach (see Table 1 for a description of terms). The bankfull channel is the channel cross-section that would be flooded each year during peak flow. This is indicated by the absence of established perennial understory vegetation. The bearing, gradient and slope angle along the reach were recorded, as was the slope angle of the valley sides. The ratio of VFW to BCW gave valley floor index (VFI); this value indicates the degree to which the channel is constrained by fluvial terraces. The spanning logs in each transect were tagged with uniquely numbered plastic tree tags, usually near the mid-stream on the downstream side.

For downed trees to be tagged and measured they needed to be: fallen or windthrown since 1998, having at least a portion of the log length within (or above) the bankfull channel, greater than 10 cm in diameter at the mid-stream, and in decay class 1–3. We used the five class decay classification system which is extensively used in the U.S Pacific Northwest (Table 2). In this system, wood soundness or structural integrity is an explicit part of the classification and recently downed trees with intact fine branches and bark constitute the first decay class. Pre- versus post-harvest downed trees, were differentiated by the accumulation of litter, age of regeneration and state of revegetation on the exposed pit, rootwad and downed bole, using the condition of trees within the vegetation plots that were known to have windthrown immediately post-harvest as the benchmark.

Tagged, downed trees were classified according to: status (dead leaning, dead uprooted, dead broken, and live uprooted), species, and decay class. The distance of the tree from the transect point of commencement (POC) was recorded. The orientation of the tree top, DBH, length within the bankfull channel width, and total length of each downed tree were also recorded. To distinguish between the characteristics of downed material and the tree from which it was derived, we hereafter refer to downed tree boles as 'logs'. Each log retained the number of the tree from which it was derived. If the tree bole was broken into multiple pieces within the bankfull channel width, those logs received the additional notation 'a', 'b' and so on. For each log, the base diameter, diameter at mid-stream (DMC), top diameter, log length, height above stream (HAS), log angle, span length and length mid-stream were recorded. Logs were classified according to log end conditions (e.g. rootwad still attached, cut ends, broken ends). Where logs were elevated above the bankfull height of the stream channel (e.g. height above stream > 0 cm), they are referred to as 'spanning' logs.

The experimental design is a replicated design with one factor, 'treatment' (with three levels: unharvested control, 10 m buffer, and 30 m buffer). The General Linear Model procedure (GLM, SAS Institute Inc., Cary, NC) was used to determine whether the diameter at mid-stream, number of logs by decay class (1, 2, and 3), in-stream, or spanning logs varied significantly among buffer treatments. The general model with source, degrees of freedom, the mean square formula and corrected error term used for testing the significance of the particular factor is given in Table 3. Observations from all treatments were pooled for the analysis of log geometry. Pearson correlation coefficients were used for assessing the correlations between height above stream and potential predictor variables such as log diameter, valley width and decay class. Multiple linear regression was then used to assess the combined utility of these variables in predicting LWD height above the stream. Variables with non-normal distributions were transformed prior to analysis.

3. Results

At the stand level, species composition was 51% western hemlock, 38% western redcedar, 6% Douglas-fir and 5% other species, primarily birch and alder. There was very little windthrow or recently downed material in these second growth stands prior to harvesting. Windthrow in the overstory vegetation plots was greatest in the first 1–2 winters following harvest of adjacent timber, however some windthrow occurred in subsequent years. After harvest in 1998, 11% of initially standing trees were blown down in the first and second years in the 10 m buffer treatment compared to 4% in the 30 m buffer and less than 1% in unharvested controls. There was minimal new windthrow in subsequent years until 2006

Table 3
General model for ANOVA design.

Source	df formula	MS	*F*-ratio
T	$t-1$	MST	MST/MSE
$r(t)$	$t(r-1)$	MSE	
Total	$r(t)-1$		

t is treatment (10 m buffer, 30 m buffer and unharvested control); r is the number of replications.

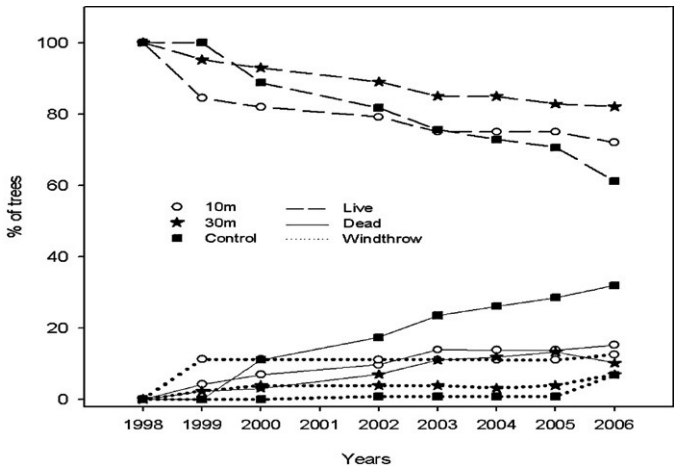

Fig. 1. The percentage of live, dead and windthrown trees in each of the three treatments for 8 years following harvest.

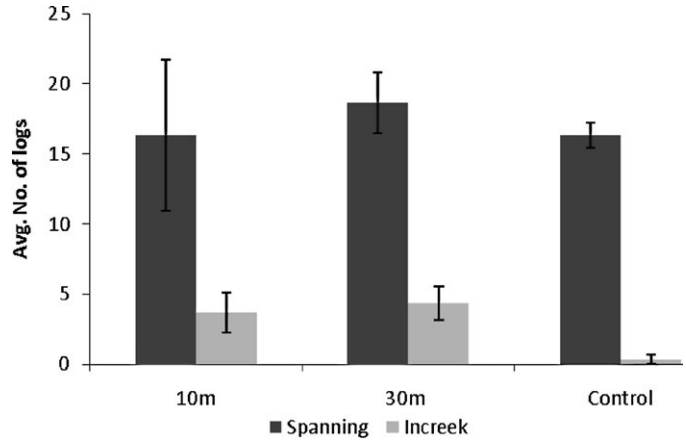

Fig. 2. Average number of spanning and in-stream logs (height above stream = 0) >10 cm diameter at mid-stream by treatment, all species, all decay class with SE bars.

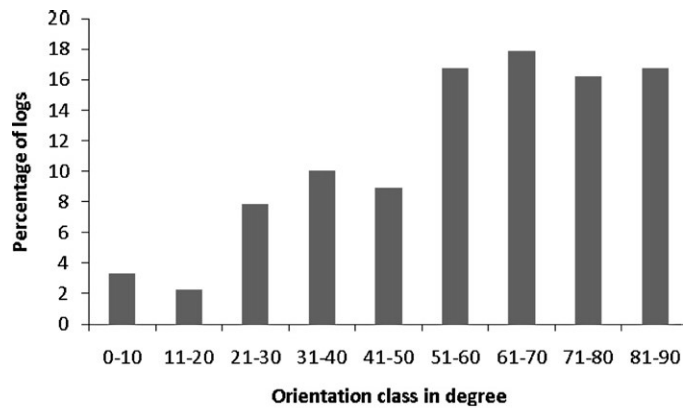

Fig. 3. Percentage of logs by orientation class in degrees where 0° is parallel to stream and 90° is perpendicular to stream, all treatments ($n = 179$).

when winter storms brought down some additional trees in both the buffer and control treatments (Fig. 1). Interestingly, in these dense second growth stands, there was still a significant amount of annual mortality in standing trees, particularly in the unharvested control, amounting to 30% of initially live trees in the control and 15% each in the 10 m and 30 m buffers after 8 years. The average diameter (DBH) of live, dead and windthrown trees were 35 cm, 13 cm, and 26 cm respectively for all treatments pooled.

A total of 179 logs (spanning and in-stream) were recorded in the stream transects (Table 4). Even after 8 years following harvesting and the post-harvest pulse of windthrow, 78% and 81% of logs in the 10 m and 30 m buffer treatments, respectively, were still elevated above the stream. In the unharvested control, 98% of downed logs were elevated above the stream. The average number of spanning logs was consistent among the treatments (16, 19 and 16 logs per 150 m of stream, for 10 m, 30 m and unharvested control respectively; Fig. 2). Few logs had dropped into the stream channel in any of the treatments (4, 4 and <1 logs per 150 m of stream for 10 m, 30 m, and unharvested control respectively). The dominant fall direction of windthrown trees was toward the west-northwest, while streams typically ran from north-south. Accordingly, most logs were oriented diagonal-perpendicular relative to the stream channel, with just 3% of logs falling parallel to the stream (Fig. 3).

Of the log variables analyzed, only the number of logs in decay class 3 was significantly different between buffer treatments ($p = 0.015$) (Table 5). There were no significant differences in the average number of decayed logs in decay classes 1 and 2 ($p = 0.30$) between buffer widths, however there were 3 times as many logs in decay class 3 in the 30 m buffer than in the 10 m and control treat-

Table 5
Analysis of variance results. Bold letters indicate significant results for $\alpha = 0.05$.

Variables	*p*-Value
Number, all logs	0.39
Number of spanning logs	0.85
decay class 1	0.40
decay class 2	0.80
decay class 3	**0.015**
Number of in-stream logs	0.51
decay class 1	–
decay class 2	0.30
decay class 3	0.87
DMC	0.94

Table 4
Stream characteristics and abundance and dimensions of post-harvest spanning and in-stream large woody debris (LWD) in the sample streams.

Stream	Watershed area (ha)	No. of reaches	BCW (m)	VFW (m)	LWD/150 m	DBH (cm)	DMC (cm)	Log length (m)	BCWL (m)	HAS (cm)
C (10 m)	89.1	11	2.94 (0.21)	11.03 (2.34)	28	24.92 (1.92)	19.48 (1.41)	24.71 (1.77)	3.51 (0.31)	141.4 (13.5)
F (10 m)	11.5	12	1.96 (0.22)	7.10 (1.21)	18	20.86 (3.18)	14.49 (1.19)	15.39 (1.86)	2.07 (0.37)	45.6 (11.9)
G (10 m)	83.5	11	3.43 (0.15)	7.57 (0.26)	14	25.09 (3.44)	17.79 (1.68)	20.26 (1.48)	2.44 (0.19)	51.6 (12.1)
D (30 m)	43.3	13	1.60 (0.20)	9.99 (1.43)	29	24.68 (2.46)	17.31 (1.35)	18.11 (2.08)	3.04 (0.25)	65.3 (13.9)
H (30 m)	55.4	9	3.79 (0.25)	23.82 (2.46)	19	28.72 (2.76)	19.23 (1.78)	16.99 (1.67)	4.40 (0.43)	61.6 (12.1)
Sk (30 m)	18.6	10	2.73 (0.24)	9.01 (1.01)	21	23.36 (2.62)	13.85 (0.76)	19.62 (2.51)	2.65 (0.24)	51.9 (11.2)
MC control	29.7	9	4.46 (0.45)	10.77 (0.74)	17	25.49 (3.39)	26.84 (3.39)	15.6 (1.76)	6.54 (0.91)	90.9 (14.4)
EC control	44.0	7	3.26 (0.33)	7.18 (0.65)	18	13.982 (0.5)	11.83 (0.31)	13.87 (0.93)	1.05 (0.17)	109.7 (17)
SC control	111.0	5	4.23 (1.12)	6.56 (1.67)	15	18.87 (1.02)	15.81 (0.94)	18.31 (1.5)	4.96 (0.39)	139.5 (15.6)
Mean (SE)	54.01 (11.3)	9.66 (0.83)	3.15 (0.32)	10.34 (1.77)	19.89 (1.7)	22.89 (2.37)	17.41 (1.42)	18.1 (1.72)	3.41 (0.36)	84.17 (13.52)

Means are followed by standard errors in parentheses.

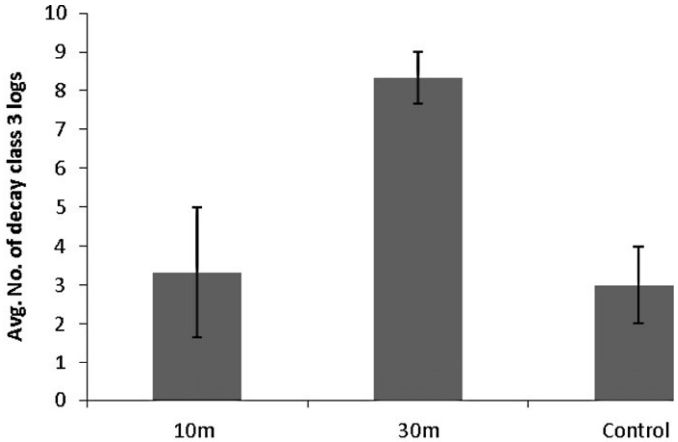

Fig. 4. Average number of logs in all size classes for decay class 3 logs, by treatment.

ments (Fig. 4). The proportion of logs in advanced stages of decay was higher for in-stream logs than for spanning logs (Fig. 5).

The mean log diameter at mid-stream (DMC) was 17 cm, with no difference between treatments ($p = 0.94$, $n = 179$). Debris with the rootwad attached to the bole was more abundant (71%) than debris with broken (28%) or cut ends (1%). As was the case for the standing timber, western hemlock was the dominant species in the stream channel, followed by western redcedar. The majority (77%) of the logs were in the 10–20 cm DMC class with only 2% larger than 40 cm DMC (Fig. 6). Since the diameter distribution of standing live trees was near normal around a mean diameter of 30 cm, the LWD

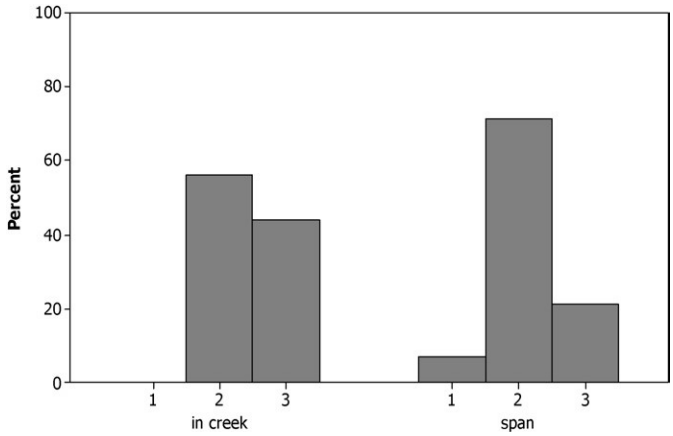

Fig. 5. Percent of spanning and in-stream logs by decay classes, all size classes and decay classes (log decay classes from Bartels et al., 1985; MOFR, 2007).

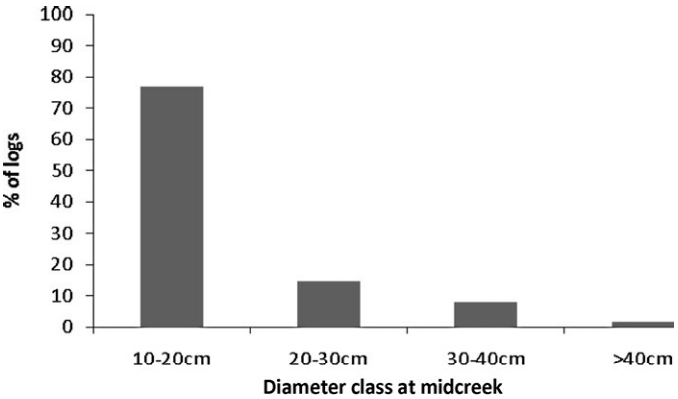

Fig. 6. Percentage of logs (in-stream and spanning logs) by diameter at mid-creek.

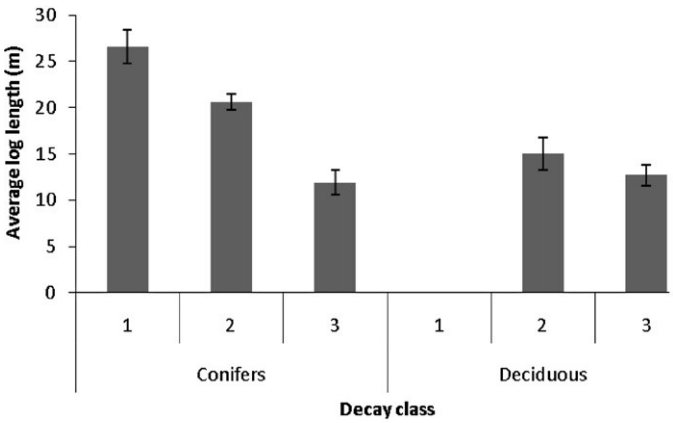

Fig. 7. Average log length (m) by decay class and species. Conifers include western hemlock, western redcedar, sitka spruce, and Douglas-fir. Deciduous include paper birch, red alder, big leaf maple, and black cottonwood. For all in-stream and spanning logs, all size classes.

was smaller on average than the standing trees. Approximately, 10% of uprooted western redcedar were still alive and spanning the stream, whereas all of the uprooted western hemlock trees were dead. A large proportion of conifer logs were in decay classes 1 and 2 whereas logs from deciduous trees were in decay classes 2 and 3 (Fig. 7).

The variables most strongly correlated with log height above stream were decay class ($r = -0.35$, $p < 0.0001$), reciprocal of valley width index ($r = 0.31$, $p < 0.0001$) and logarithm of diameter at mid-stream ($r = 0.26$, $p = 0.0005$). Accordingly, multiple linear regressions were fitted for predicting height above stream using these variables. Though the intercept and all the variables DMC, RECIVWI and Deccls were significant ($\alpha = 0.05$), the R^2 value for the best regression was low (0.21; Table 6). The average length of the logs decreased as decay class increased, for both conifer and deciduous logs. Furthermore, within decay class 2, deciduous logs were shorter than conifer logs (Fig. 7).

4. Discussion

A key reason for retaining buffers adjacent to streams is to ensure a sustained supply of LWD to the aquatic ecosystem (e.g. Van Sickle and Gregory, 1990). The numbers of trees available as source of LWD depends on stand density, width of the treed zone, valley geometry and fall directionality. The tree-fall rate will depend on the severity and timing of post-harvest windthrow events, and standing tree mortality, and how these vary with buffer condition. Grizzel and Wolff (1998) suggested that in a managed forest landscape with small, low gradient streams, windthrow in riparian buffers is likely the most significant mechanism by which LWD is recruited into stream channels. They found higher total tree fall rates in narrower buffers and a 34% increase in within-channel

Table 6
Multiple linear regression for predicting height above stream (HAS, cm) using diameter at mid-stream (DMC, cm), reciprocal of valley width index (RECIVWI) and decay class (Deccls).

Variables	Parameter estimates	Standard error	p-Value	R^2	Root MSE	n
Intercept	106.55	29.4	0.0004			
DMC	1.30	0.63	0.0407			
RECIVWI	106.80	27.13	0.0001			
Deccls	−38.69	9.90	0.0001			
MODEL				0.2142	64.72	165

HAS, height above stream; DMC, diameter at mid-stream; VWI, valley width index; Deccls, decay class of log. Level of significance, $\alpha = 0.05$.

LWD pieces in small coastal streams in Washington State. In our riparian buffers experiment, we found higher windthrow rates in the 10 m and 30 m buffers than in the control, but a higher rate of standing tree mortality due to ongoing competitive exclusion in the controls. In consequence, there was no difference in the total number of spanning or in-stream logs or the size of these logs between buffers and controls in our study. While windthrow is important, mortality due to competitive exclusion is a significant source of spanning LWD in all treatments. The number of spanning LWD pieces per linear meter of stream channel in our study is lower than reported in other studies in comparable forests, likely because we were recording only post-harvest LWD. In contrast to our value of 0.13 LWD/linear meter, Long (1987) found 0.45 LWD/linear meter of stream in an Oregon coastal stream in a 50+ year-old stand. Heimann (1988) found 0.61 LWD/linear meter of stream in an 80–140 year-old stand in coastal Oregon, and Ralph et al. (1994) found 0.25 LWD/linear meter of stream in an 80+ year-old stand in Western Washington.

When adjacent timber is harvested, the width of the unharvested buffer affects the number of stems available for recruitment as LWD. In Southeast Alaska, Martin and Grotefendt (2006) found that 95% of the LWD in streams in buffered sections was derived from within 30 m of the channel, whereas in the unlogged stand 96% of LWD recruits were derived from within 20 m. However, they found that the majority of LWD was coming from within 10 m of the channel for buffer and unlogged stands respectively. McDade et al. (1990) examined LWD sources in first to third order streams in coastal Washington and Oregon and found that 70% of LWD pieces came from trees rooted within 20 m of the stream channel. In our study at MKRF, the similarity in number of spanning and in-stream logs between the 30 m buffer and 10 m buffers, and unharvested control suggests that 10 m buffers are adequate for maintaining the supply of LWD in the short term.

We found that small diameter trees contributed disproportionately to LWD recruitment. Similar results have been reported in other LWD studies in coastal and continental forests (e.g. Hauer et al., 1999; Berg et al., 1998; Richmond and Fausch, 1995). The preferential recruitment of small LWD is in part due to the ongoing competition mortality which is a normal phenomenon in high density young-mature stands (e.g. Oliver and Larson, 1990). Windthrow has the potential to introduce larger trees (e.g. Scott and Mitchell, 2005), but in our case, the absolute difference in windthrow levels between buffers and controls was relatively small and did not result in an overall increase in LWD dimensions in the buffer treatments. The increased light levels and wind-induced thinning in the buffers may accelerate the development of larger diameter trees in these treatments. Furthermore, these buffer stands will be more exposed than controls to the effects of periodic severe storms for many years. Longer term studies will be necessary to characterize differences in wood recruitment dynamics between buffers and controls.

Windthrow in cutblock edges and buffers is typically greatest in the first 1–2 post-harvest winter seasons (Rollerson and McGourlick, 2001). We found a similar pulse of post-harvest windthrow in the MKRF buffers experiment. However, this pulse of windthrow does not immediately result in LWD within the stream channel. We found that even 8 years after harvest, 80% of windthrown logs were still spanning the stream in the 10 m and 30 m buffers. Grizzel and Wolff (1998) reported that 75% of post-harvest windthrow LWD was suspended in their study of coastal streams. Wei (2005a) and Chen et al. (2006) also found the majority of LWD was located above the bankfull height in small sized streams in the continental forests of the BC Interior. These results have important implications for debris recruitment modeling. The first is that the connection between the level of post-harvest windthrow in buffers, and the ultimate effect of this windthrow on stream channel properties is very indirect. Actual recruitment of windthrown material into the stream may take place over decades following a short-term pulse of post-harvest windthrow. Secondly, most windthrown logs enter the stream channel only after decay and subsequent breakage. This slow recruitment following a pulse of mortality appears to be similar to the process following wildfires (e.g. Powell, 2006; Jones and Daniels, 2008), however there are likely important differences in decay processes between live trees that are uprooted by wind and standing fire-killed trees.

The relationship between decay and log breakage is interesting and important for LWD recruitment modeling and understanding the subsequent role and persistence of in-stream LWD. If logs decayed uniformly from the outside-in along their entire length, one would expect that decaying logs would fail near mid suspension. However, we observed that logs decayed more rapidly from the upper broken end where they were in contact with the soil, breaking into relatively short chunks from the upper end. This manner of decay is important. If the majority of spanning material eventually enters the stream channel as short, well-decayed logs, these logs may not provide the long-term in-stream role in channel structuring suggested by Gurnell et al. (2002). Further study of the condition, residency time and role of LWD material that enters streams by this route is warranted.

The orienting effects of wind damage are also important for LWD recruitment modeling. With a sampling rule that trees must cross at least part of the stream channel in order to be tallied, it is not surprising that few downed trees that run parallel to the stream are sampled. The area for potential recruitment is much larger for trees falling perpendicular to stream direction. However, our vegetation plot results showed that windthrow was strongly oriented perpendicular to the riparian buffers. This windthrow orientation is logical given that wind exposure and wind loading is higher where wind comes from directions perpendicular to the buffer, and was also observed by Rollerson and McGourlick (2001) in their studies on Vancouver Island. Regardless of the initial orientation of material that enters streams, stream energy can re-orient it (e.g. Chen et al., 2006), but this is not typically the case in small streams (e.g. Bilby and Ward, 1989; Richmond and Fausch, 1995). Our streams were all under 5 m wide and only 3% of in-stream logs were oriented parallel to the stream.

Grizzel and Wolff (1998) stated that the time between initial tree fall and the secondary phase when logs break apart and enter the channel depends upon species, size and condition of the wood piece. Our methodology in this study did not permit time to recruitment to be measured directly, but the trends in height above stream provide some insights into the governing processes and indicate how predictive models can be developed. The initial log height above stream depends on the interaction of log orientation and stream valley width. Smaller logs decayed more rapidly than larger ones. The logs failed primarily through decay at the upper end, progressively shortening and dropping closer to the water. Accordingly, the height above stream of suspended logs provides an indication of the progression towards in-stream recruitment. To model the time to in-stream LWD deposition, it is therefore necessary to characterize initial LWD position, valley geometry, and species-specific decay rates. It is also necessary to use faster decay rates where the log contacts the ground.

Windthrow as a source of LWD recruitment is ignored in recruitment models except for CWD (Bragg et al., 2000), in which users specify rate of windthrow as a fraction of live trees per year. Neither AQUAWOOD (Wei, 2005b), RAIS (Welty et al., 2002), STREAMWOOD (Meleason, 2001) nor CWD (Bragg et al., 2000) account for the initial position of uprooted trees relative to the stream. Nor do they have submodels that specifically account for the lag between tree fall and in-stream recruitment. In order to address this conceptual gap in LWD models, it is necessary to represent the valley

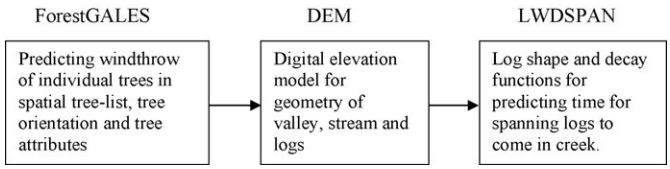

Fig. 8. Information flow between ForestGALES and LWDSPAN recruitment model for windthrow-derived large woody debris.

and stream channel geometry, the geometry and initial condition of spanning LWD, and to introduce a decay function for LWD change in condition over time. Appropriate functional forms for LWD decay functions are documented in the literature (Harmon et al., 1986), along with coefficients for common species. Decay rates vary with temperature and humidity (Mackensen et al., 2003) and riparian zones are typically more humid than upland forests, so decay rates should be locally estimated. To provide further realism in riparian buffers exposed by harvesting, the recruitment model can be driven by a windthrow prediction model such as ForestGALES (Gardiner et al., 2008). The variant of this model used in British Columbia is coupled with the stand growth and yield model TASS and produces a spatially explicit tree list. Since the tree list can be derived from plot data or growth models, it would also be possible to identify which trees will die and subsequently fail due to competition mortality. The LWDSPAN submodel determines the probable break points for a log when it hits the ground by estimating the cumulative turning moment and sectional resistive moments for any stem segments that are suspended above the ground. Resistive moments are adjusted annually based on decay and loss of effective diameter. To accurately portray log geometry, it is necessary to include a digital elevation model of terrain adjacent to the stream channel. The flow of information between ForestGALES and LWDSPAN is shown in Fig. 8.

5. Conclusion

In riparian buffers, recruitment of logs into stream channels where they can play a role in stream morphology lags behind the post-harvest pulse of windthrow by many years. This lag time depends on the size, species and initial condition of the logs, and their direction of fall relative to the geometry of the stream valley. These components can be characterized in submodels of LWD recruitment models. Logs that initially span the stream are in a state of advanced decay before they enter the stream channel. This has implications for their in-stream role and residence time and needs further investigation.

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Post-harvest windthrow and recruitment of large woody debris in riparian buffers on Vancouver Island

- Devesh Bahuguna[1],
- Stephen J. Mitchell[1] &
- Grant R. Nishio[2]

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Abstract

Large woody debris (LWD) provides structural complexity to small streams. Riparian buffers are intended to provide long-term vegetation cover and supplies of LWD, but post-harvest windthrow often occurs. To evaluate the impacts of windthrow in riparian buffers and identify the components for a small stream LWD recruitment model, we sampled 26 streams in immature and older stands in wind-exposed areas of southwestern and northern Vancouver Island. These treed buffer strips had been exposed following clearcut harvest of adja timber on both sides 1–20 years previously. For stream sections 100 m long in each buffer, all logs greate

7.5 cm diameter that spanned at least part of stream channel were measured. A total of 658 logs were recorded. Windthrown trees were comparable in characteristics to the trees that made up the buffer. The majority of logs derived from windthrown trees were oriented perpendicular to the stream channel and were suspended above the stream channel. Even 20 years after harvesting, two-thirds of the logs were still suspended above the stream. Logs in older buffers were more decayed, and the decay rate depended on tree species and initial diameter. Log height above stream was negatively correlated with log decay class and time since logging. Log length declined with time since harvest exposure and decay class. Sediment was exposed on upturned roots and within mineral soil pits. The volume of soil retained on upturned rootwads declined over time, but some soil remained even after 20 years. Very little of this exposed sediment was close enough to the creek to result in sediment delivery.

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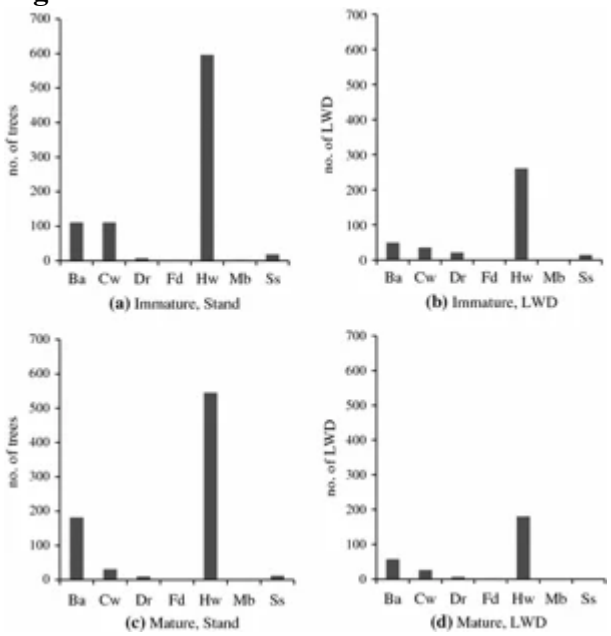

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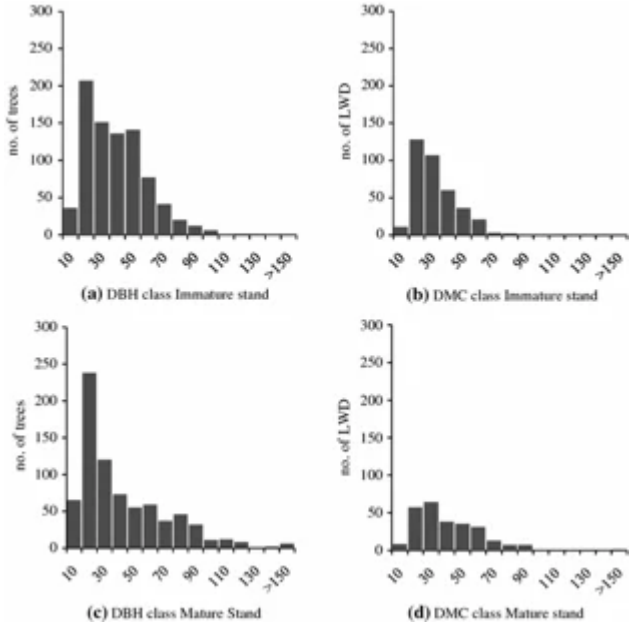

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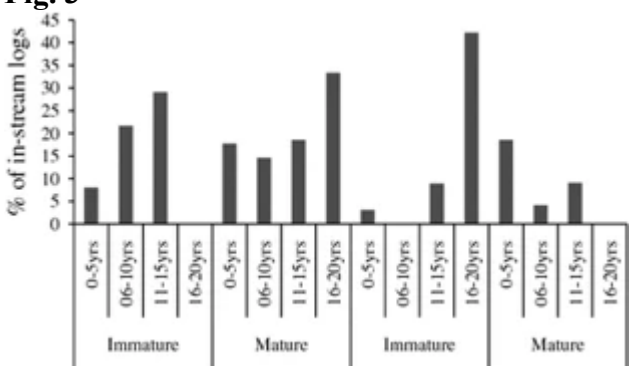

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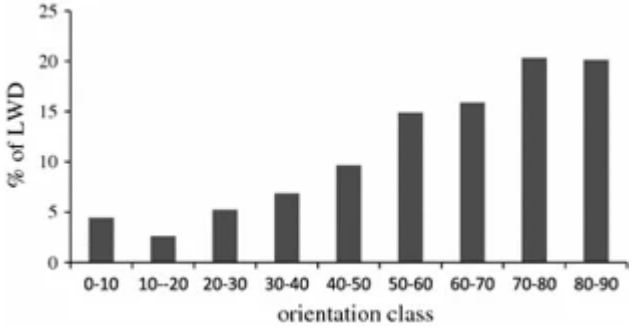

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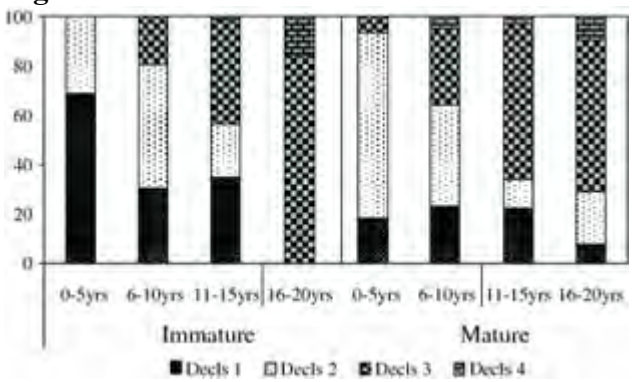

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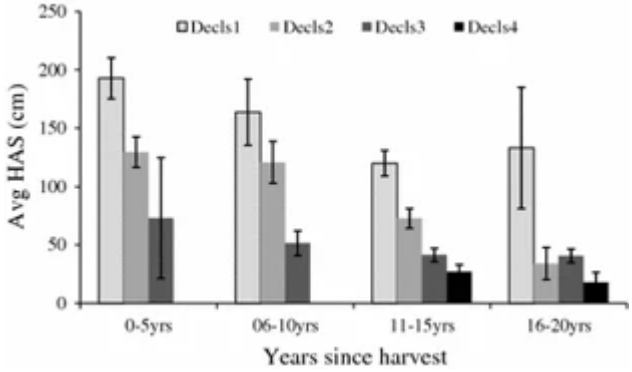

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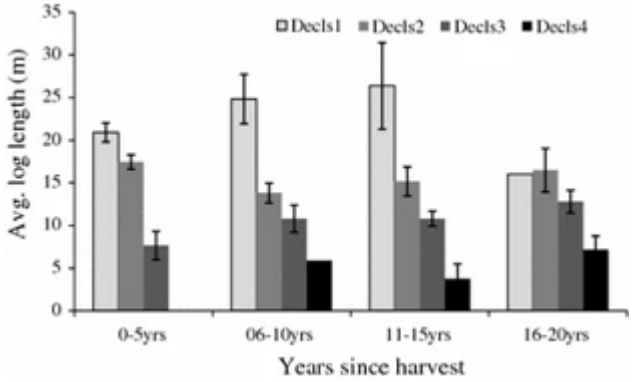

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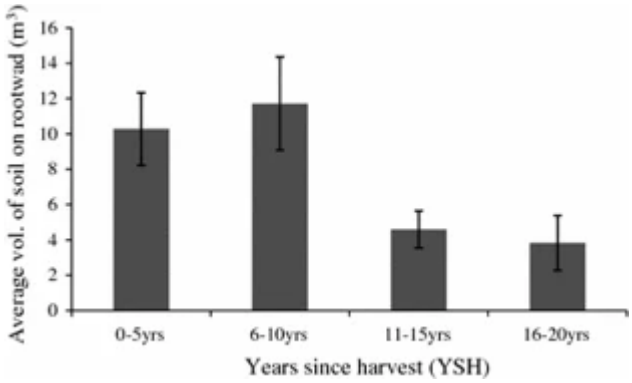

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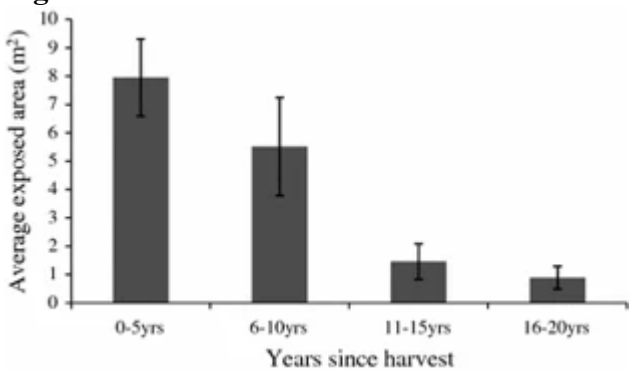

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Author information

Affiliations

1. **Department of Forest Sciences, University of British Columbia, 2424-3041 Main Mall, Vancouver, BC, V6T1Z4, Canada**

    - Devesh Bahuguna
    - & Stephen J. Mitchell

2. **FPInnovations, 2601 East Mall, Vancouver, BC, Canada**

    - Grant R. Nishio

Authors

1. **Search for Devesh Bahuguna in:**

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Corresponding author

Correspondence to Stephen J. Mitchell.

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Geomorphology, hyporheic exchange, and selection of spawning habitat by bull trout (*Salvelinus confluentus*)

Colden V. Baxter and F. Richard Hauer

Abstract: The distribution and abundance of bull trout (*Salvelinus confluentus*) spawning were affected by geomorphology and hyporheic groundwater – stream water exchange across multiple spatial scales in streams of the Swan River basin, northwestern Montana. Among spawning tributary streams, the abundance of bull trout redds increased with increased area of alluvial valley segments that were longitudinally confined by geomorphic knickpoints. Among all valley segment types, bull trout redds were primarily found in these bounded alluvial valley segments, which possessed complex patterns of hyporheic exchange and extensive upwelling zones. Bull trout used stream reaches for spawning that were strongly influenced by upwelling. However, within these selected reaches, bull trout redds were primarily located in transitional bedforms that possessed strong localized downwelling and high intragravel flow rates. The changing relationship of spawning habitat selection, in which bull trout selected upwelling zones at one spatial scale and downwelling zones at another spatial scale, emphasizes the importance of considering multiple spatial scales within a hierarchical geomorphic context when considering the ecology of this species or plans for bull trout conservation and restoration.

Résumé : La distribution et l'abondance de l'omble à tête plate (*Salvelinus confluentus*) en phase de reproduction étaient affectées par la géomorphologie et par l'échange d'eau entre la nappe sous-terraine hyporhéique et les cours d'eau à des échelles spatiales multiples dans les cours d'eau du bassin de la Swan, nord-ouest du Montana. Parmi les affluents de fraye, l'abondance des nids d'omble à tête plate augmentait avec la superficie des portions de vallées alluviales qui étaient longitudinalement confinées par des ruptures de pente. Parmi tous les types de portions de vallée, les nids d'omble à tête plate se retrouvaient principalement dans ces portions confinées, qui présentaient des régimes complexes d'échange hyporhéique et de vastes zones de remontée d'eau. L'omble à tête plate recherchait pour la fraye des tronçons fortement influencés par les remontées d'eau. Toutefois, à l'intérieur de ces tronçons sélectionnés, les nids se retrouvaient principalement dans des types de lit de transition qui présentaient une forte remontée d'eau localisée et des taux élevés d'écoulement à l'intérieur du gravier. La relation changeante régissant le mode de sélection de l'habitat de fraye – l'omble de fontaine choisissait les zones de remontée d'eau à une échelle spatiale et les zones d'enfoncement des eaux à une autre échelle spatiale – fait ressortir qu'il est important de considérer des échelles spatiales multiples à l'intérieur d'un contexte géomorphologique hiérarchique lorsqu'on étudie l'écologie de cette espèce ou qu'on fait des plans pour sa conservation et son rétablissement.

[Traduit par la Rédaction]

Introduction

Stream ecologists have long recognized the interactive relationship between a stream and its valley (Hynes 1975). Patterns and processes of stream ecosystems are closely tied to the surrounding terrestrial environment (Junk et al. 1989; Gregory et al. 1991), longitudinal gradients (Vannote et al. 1980; Minshall et al. 1985), and groundwater – stream water exchange (Grimm and Fisher 1984; Stanford and Ward 1993). Because of these interactions, it is essential that stream ecosystems be examined holistically within the context of their landscapes (Frissell et al. 1986; Swanson et al. 1988).

The flow of water over land shapes the geomorphology of stream systems. Geomorphic change, combined with the interactive response of the stream environment, results in alternating erosion and deposition. This dynamic process of cut and fill creates spatial heterogeneity in physical habitat for stream organisms. Stream habitat can be classified hierarchically into drainages, valley segments, stream reaches, and channel bedform units (pool–riffle) (Frissell et al. 1986). Each level of the stream habitat hierarchy corresponds to a distinct spatial scale, from kilometres to metres to centimetres, etc.

At the drainage spatial scale, alluvial river systems of the northern Rocky Mountains are characterized by confined and unconfined valley segments occurring in alternating series along the stream gradient. In unconfined alluvial segments, streams flow across deposits of gravel and cobble associated with alluvial floodplains (Church 1992). These reaches commonly have a vertical dimension of groundwater – surface water interaction extending tens of metres into the alluvium

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C.V. Baxter and F.R. Hauer. Flathead Lake Biological Station, University of Montana, 311 Bio Station Lane, Polson, MT 59860-9659, U.S.A.

[1]Author to whom all correspondence should be sent at the following address: Department of Fisheries and Wildlife, Oregon State University, Corvallis, OR 97331, U.S.A.
e-mail: baxterco@ucs.orst.edu

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(Stanford and Ward 1988, 1993). The size and arrangement of particles contained in vertical and lateral profiles of an alluvial segment vary with geomorphic surface. Differences in substrate porosity lead to the formation of preferential hyporheic flow pathways in longitudinal, lateral, and vertical dimensions. In contrast, confined valley segments are generally characterized by narrow valley walls, near-surface bedrock, absence of a floodplain, and relatively high stream gradient (Montgomery et al. 1996).

The response of stream organisms to habitat is scale dependent. Habitat-use patterns for any organism in an ecosystem can be related to many environmental factors. The variation of a habitat factor may be perceived by an organism as information at certain scales, while at another scale, it may be perceived as noise (Dutilleul and Legendre 1993). Consequently, understanding the distribution and abundance of species relative to habitat factors may require a multiscale approach (Wiens 1989). Relationships between fish and their habitat have been primarily examined at fine spatial scales (Bayley and Li 1992). However, there is increasing recognition of the importance of a multiscale approach (e.g., Poizat and Pont 1996; Torgersen et al. 1999) and inclusion of critical landscape features such as zones of process interaction (e.g., land–water ecotones) (Schlosser 1995).

The hyporheic zone is an ecotone that links ground and surface water processes in streams (Brunke and Gonser 1997). The boundaries of groundwater – stream water interactions are spatially and temporally dynamic because of fluctuating hydrographic regimes. Although hyporheic exchange has been hypothesized to occur at multiple spatial scales (e.g., drainage, valley segment, reach, and channel unit scales; Stanford and Ward 1993), few efforts have been made to quantify hyporheic exchange extensively within a spatial hierarchy (Boulton et al. 1998). Most investigations of groundwater – surface water interaction in streams have focused at the channel bedform scale (i.e., pool–riffle, which includes local, shallow exchange). Hyporheic exchange at this spatial scale is known to vary with discharge, gradient, bedform, sediment permeability, and water surface slope (Vaux 1968; Thibodeaux and Boyle 1987; Harvey and Bencala 1993). Few studies of hyporheic exchange have been conducted at anything larger than the reach scale; however, theoretical models predicting how the hyporheic zone changes within a drainage have recently been proposed (e.g., Stanford and Ward 1993; White 1993).

For decades, fisheries research involving groundwater – stream water exchange has primarily focused on the importance of intragravel flow to the incubation of fish embryos (e.g., Terhune 1958; Curry and Noakes 1995). Upwelling water often differs from surface water in numerous ways (e.g., dissolved oxygen, nutrients), but perhaps none more important than temperature regime. Upwelling waters in temperate climates are generally cooler in summer and warmer in winter and may provide thermal refugia for stenothermic fish species (e.g., Gibson 1966; Nielson et al. 1994).

The bull trout (*Salvelinus confluentus*) is a native char in the Pacific Northwest whose populations are fragmented and declining throughout their range due to overharvest, displacement by exotic species, and habitat degradation (Rieman and McIntyre 1993). In 1997 the U.S. Fish and Wildlife Service recommended the bull trout for listing under the U.S. Endangered Species Act. The Swan basin in northwestern Montana (Fig. 1) has received attention as a refuge where many bull trout remain (Rieman and McIntyre 1993; Baxter et al. 1999). Consequently, this basin provides an opportunity for developing a better understanding of bull trout in a system where sufficient numbers exist for effective study.

Bull trout display migratory life histories that are associated with large lake and river systems. Migratory bull trout in the Swan basin spawn in second- to fourth-order streams from August through October. Embryos and alevins overwinter in spawning gravel for more than 200 days prior to fry emergence. Juveniles remain in the natal streams for 1–3 years prior to migrating to Swan Lake where they spend 2–4 years growing before sexual maturity (Fraley and Shepard 1989). Factors such as temperature, channel stability, and cover are known to be important spawning habitat characters (for a review, see Rieman and McIntyre 1993). However, researchers have long recognized that many stream segments of ostensibly suitable habitat (i.e., substrata, gradient, and cover) were never used by spawning bull trout. For example, over 75% of the spawning in the Swan basin occurs in less than 10% of the available stream length (Leathe and Enk 1985). Although studies have noted that stream habitats selected by bull trout for spawning are often influenced by groundwater (Graham et al. 1981; Weaver and White 1985), a quantitative spatial assessment of this habitat factor and its importance to bull trout has not yet been conducted. Furthermore, no studies have focused on a multiscale spatial assessment of hyporheic exchange and spawning habitat use patterns.

The objective of this study was to examine the relationships between geomorphology, groundwater – stream water exchange patterns, and the distribution and abundance of bull trout spawning sites. General observations of geomorphic structure at the landscape scale, recent study of hyporheic exchange in large, coarse-substratum alluvial floodplains (Stanford and Ward 1993), and patterns of bull trout spawning (e.g., Weaver and White 1985) led us to hypothesize that these biophysical variables were related. Our specific research objectives were to (*i*) quantify the process of hyporheic exchange at multiple spatial scales, (*ii*) identify geomorphic features associated with variation of hyporheic exchange at each spatial scale, and (*iii*) evaluate the influence of hyporheic exchange within the context of a geomorphic hierarchy (i.e., drainage, valley segment, reach, and bedform scales) on the distribution of bull trout spawning.

Materials and methods

Study area

The Swan basin (2070 km^2) is a densely forested, north–south trending, glaciated basin between the Swan and Mission fault-block mountain ranges in northwestern Montana. From peak elevations over 2500 m that are in excess of 1500 m above the valley floor, waters drain through tributary canyons carved in Precambrian metamorphic rock. Tributary streams reach their confluence with the sinuous Swan River as it flows across morainal and alluvial deposits of the broad Swan Valley north into Swan Lake (Fig. 1). The current geomorphic character of the Swan River and its tributaries was established during two major Pleistocene glacial advances, which led to a period of fluvial transport, mass wasting, and alluvial thickening by in-filling from glacial outwash (Kleinkopf et al. 1972). Streams in the Swan basin traverse valley

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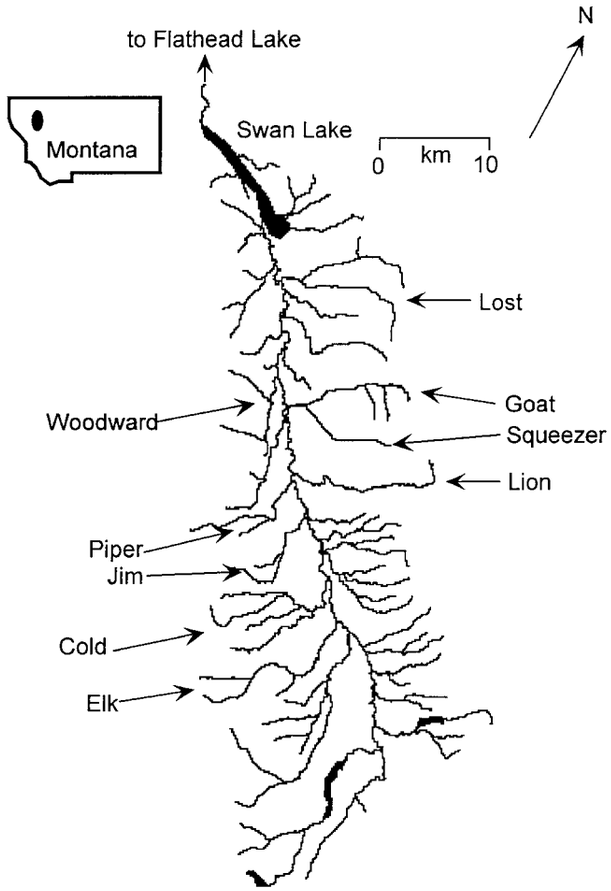
Fig. 1. Map of the Swan River basin (latitude 47°35′, longitude 113°45′). The nine primary bull trout spawning tributary streams are identified by name. The inset indicates the location of the Swan River in northwestern Montana.



floors dominated by mixed-conifer forest (e.g., *Abies*, *Pinus*, *Thuja*, *Pseudotsuga*, *Picea*, *Larix*), with deciduous trees and shrubs (e.g., *Betula*, *Populus*, *Salix*) concentrated along riparian areas and floodplains.

Our study area included nine third- and fourth-order (Strahler 1964) tributary streams, which constitute the principal bull trout spawning tributaries of the Swan River (Leathe and Enk 1985). The nine tributary drainages ranged in size from 32.9 km^2 (Piper Creek) to 81.8 km^2 (Lion Creek). These streams are typical of the northern Rocky Mountains, possessing a snowmelt-dominated hydrograph with peak flows in late May or June as much as three orders of magnitude greater than base flows. Streams are further characterized by large cobble substratum and waters that are generally very low in dissolved ions, nutrients, and suspended particulates.

Characterization of geomorphic features

At the drainage scale, we used standard U.S. Geological Survey topographic 7.5′ quadrangle maps (1 : 24 000 scale) and structural geological maps (1 : 250 000 scale) to identify and measure 28 geomorphic variables for each of the nine study drainages (Fig. 1; Table 1). We also used aerial photographs and field verification to aid in discrimination of specific features. In addition to traditional landscape-scale geomorphic metrics (Strahler 1964), we classified valley segments into three categories: (*i*) bounded alluvial valley segments, (*ii*) unbounded alluvial valley segments, and (*iii*) confined valley segments.

Bounded alluvial valley segments were characterized by unconfined stream sections at least 500 m in length and greater than 50 m in valley bottom width with a downstream geomorphic knickpoint. Generally, knickpoints were distinguished by steepening in channel slope, constriction in valley width, and a geologic formation near the surface, such as bedrock or terminal moraine deposits. By including measures of valley bottom width and the number of knickpoints, we were able to estimate the number, length, and area of bounded alluvial valley segments (Fig. 2). The area of bounded alluvial segments was examined because it helped describe the overall valley shape that we hypothesized was important to the structuring of hyporheic flow patterns (Fig. 2). Unbounded alluvial valley segments were similar to bounded alluvial valley segments but were always the furthest downstream segment of a tributary where the stream emerged onto the Swan Valley floor and flowed to the Swan River without encountering another knickpoint. We identified confined valley segments by their narrow valley bottoms (typically <50 m) and steep gradients (usually <0.05 m/m). Confined valley segments often were associated with canyons and near-surface bedrock with only a patchy, thin (<2 m) layer of cobble substratum.

Characterization of hyporheic exchange

We quantified hyporheic exchange by measuring vertical hydraulic gradient (VHG) and streambed hydraulic conductivity. We installed piezometers in the stream substratum and calculated VHG from the equation

$$\mathrm{VHG} = \mathrm{d}h/\mathrm{d}l$$

where dh is the difference in head between the water surface in the piezometer and the level of the stream surface (the hydraulic head differential, centimetres), and dl is the depth from the substratum surface to the first opening in the piezometer sidewall (the elevation head differential, centimetres). Thus, VHG is a unitless measure that is positive under upwelling conditions and negative under downwelling conditions.

Hydraulic conductivities (K) were estimated using falling head tests (Lee and Cherry 1978), which measure the rate of water-level equilibration between the piezometer and the surrounding groundwater. For some piezometers, we were able to estimate K using the Hvorslev (1951) equation:

$$K = \frac{(\pi)(D)}{(11)(T_0)}$$

where D is the inside diameter of the piezometer and T_0 is the basic time lag, a value derived from the construction of an equilibration curve (for summary of calculation, see Cedergren 1989). However, among many of the piezometers, equilibration took place so quickly that it was impossible to use the conventional falling head test to create a curve and acquire an accurate estimate of K. Thus, we derived an alternative equation:

$$K = \frac{(0.2501)(D)}{(d_t)}\left|\ln\frac{h_0}{h}\right|$$

where D is the inside diameter of the piezometer and d_t is the time it takes for the head level to drop from h_0 to h. In the situations where we were able to estimate K using the Hvorslev method as well as our modified equation, there was no appreciable difference between the two estimates. Therefore, for consistency, we used only the values obtained by the latter approach for all our analyses. Following the estimation of K, we were able to calculate the vertical component of flow using Darcy's equation for specific discharge (v, centimetres per second):

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Table 1. Drainage-scale geomorphic variables that were examined as a part of this study.

Morphometric measure	Slope class	BAVS[a] related
Drainage area	% drainage <1% slope	Maximum valley bottom width
Stream order	% drainage 1–2% slope	Mean valley bottom width
Total stream length	% drainage 2–7% slope	Variation in valley bottom width
Accessible stream length	% drainage 7–20% slope	Number of gradient steps >10%
Drainage density	% drainage 20–40% slope	Number of BAVS
Drainage length	% drainage 40–55% slope	Total area of BAVS
Average stream gradient	% drainage >55% slope	Length of accessible BAVS
Gradient of accessible stream length	% drainage >55% slope	Area of accessible BAVS
Relief ratio		
Form ratio		
Elongation ratio		
Mean bedrock dip angle		

Note: Accessibility refers to whether migratory bull trout can navigate the stream.
[a]Bounded alluvial valley segment.

$$v = -K \times \mathrm{d}h/\mathrm{d}l$$

where dh and dl are as above for the calculation of VHG (Freeze and Cherry 1979).

We quantified hyporheic exchange at the drainage scale in four of the nine principal bull trout spawning streams: Elk, Lion, Cold, and Lost creeks (see Fig. 1). We installed more than 500 piezometers in alluvial valley segments of these drainages. We did not sample confined segments for VHG because generally, it was impossible to install piezometers in the large, boulder-dominated bed material typical of the confined segments, and frequently confined segments were characterized by near-surface bedrock.

Alluvial valley segments within each of the four tributary drainages were divided into upper, middle, and lower regions, and a reach of approximately 200 m length was sampled in each. Piezometers were spaced at 10- to 15-m intervals, and a minimum of 15 piezometers were installed in each reach. At this scale of sampling, we measured only VHG for each piezometer (no estimate of K). To examine the relative potential for upwelling at each piezometer site, we transformed these VHG data by changing all negative values to zero, while all positive values remained the same. For the sake of simplicity, we refer to these values as groundwater input, although they are actually only a measure of input potential.

We examined hyporheic exchange and streambed geomorphology at the bedform (pool–riffle) spatial scale through detailed analysis of two spawning reaches in Cold and Lion creeks. Following installation of an extensive network of more than 50 piezometers in 100 m of stream length, we measured VHG and K. We then used a Lietz TC600 Total Station to survey channel bedform, topographic complexity of the surrounding floodplain, and piezometer and bull trout redd locations by acquiring 600–700 individual point readings. We then constructed detailed planimetric maps of floodplain and channel morphometry, bull trout redds, and piezometer locations using SURFER™ mapping software. We also constructed topographic contour maps and interpolated isopleths of VHG and intragravel v. Channel morphometry maps were used to separate the streambed into three categories based on longitudinal bedslope character: (*i*) concave, (*ii*) convex, and (*iii*) transitional. Concave patches typically were associated with riffles and the head of pools and convex areas with pool tailouts, and transitional patches were located at the downstream edge of pool tailouts that merge into the head of a riffle. Because we did not want to disturb bull trout eggs, piezometers were not installed directly into redds.

Bull trout redds and stream temperature

Data on the distribution and abundance of bull trout redds were collected in two ways. In all nine principal spawning streams of the Swan basin, we obtained redd data from surveys conducted by the Montana Department of Fish, Wildlife and Parks in autumn 1995. The precision of these data was sufficient for drainage and valley segment scale analyses and allowed us to position bull trout redd locations on topographic 7.5′ quadrangle maps that had geomorphic variables delineated for landscape characterization. We also conducted our own redd surveys on Elk, Lion, Cold, and Lost creeks in order to map locations of redds with the greater precision required for reach and channel unit scale analyses.

We installed Onset™ data loggers to monitor water temperature within the four primary study drainages. We placed temperature loggers in stream reaches that our VHG data indicated to be reaches of groundwater discharge and in reaches known to be either VHG neutral or VHG negative. We also conducted several winter surveys of each creek to estimate the percent ice cover and document the occurrence or absence of anchor ice in proximity to positive or negative VHG readings from our piezometers.

Data analysis

We utilized an empirical approach to assess the distribution of bull trout spawning with respect to geomorphology and hyporheic exchange. At each spatial scale (i.e., drainage, valley segment, reach, and channel unit), we expected that the distribution of bull trout redds would take one of two theoretical forms: (*i*) a uniform, even distribution or (*ii*) a discontinuous or patchy distribution. Our working hypothesis was that bull trout redds would be patchily distributed, with specific habitat preference regarding geomorphic and hyporheic exchange variables occurring at each spatial scale. The null hypothesis was that the distribution of bull trout redds is independent of geomorphic or hyporheic exchange variables.

We examined all combinations of dependent and independent variables. At the drainage scale, two correlation matrices were constructed, one that included geomorphic features and bull trout redd numbers for the nine principal bull trout spawning drainages and another that included only the four streams that were sampled for VHG. Similarly, at valley segment and reach scales, one matrix was constructed for all segments and reaches in the nine basins and another for those in which we collected VHG data. These matrices allowed us to examine individual associations between bull trout spawning and landscape-scale geomorphic variables as well as groundwater – surface water exchange variables. They also allowed us to assess covariation among variables. Where we detected significant correlations, we applied linear regression to assess the relationship between dependent and independent variables.

We used nonparametric χ^2 analysis to determine differences between expected and observed patterns of bull trout redd distribu-

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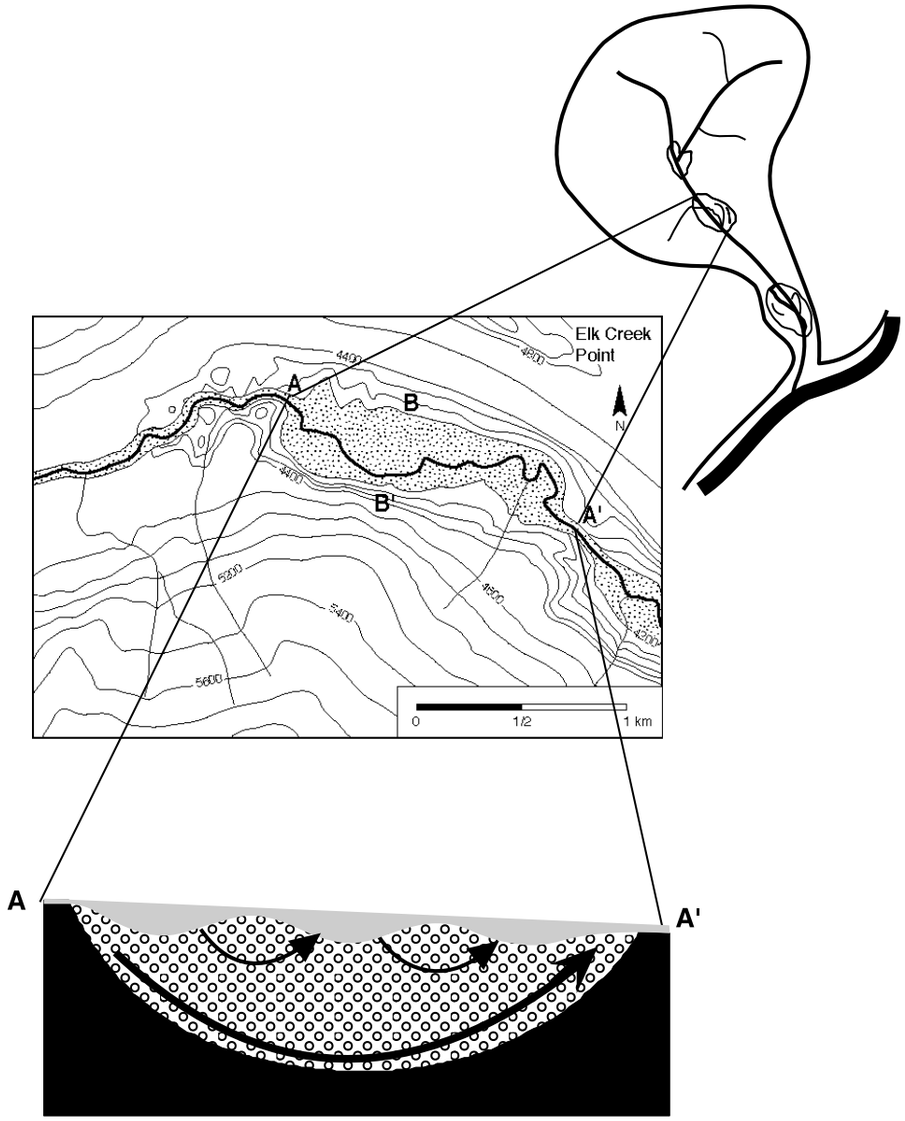
Fig. 2. One of the bounded alluvial valley segments (BAVS) in the Elk Creek drainage (latitude 47°30′, longitude 113°48′). The dimensions of the BAVS were measured directly from U.S. Geological Survey topographic 7.5′ quadrangle maps (1 : 24 000 scale) (*A*–*A*′ = length, *B*–*B*′ = maximum valley bottom width). The cross-sectional diagram (*A*–*A*′) illustrates how reach-scale (large arrow) and bedform-scale (small arrows) hyporheic exchange typically occurs within a BAVS. The stippling denotes the alluvial valley fill.



tion. We also applied nearest-neighbor analysis and Mann–Whitney *U* tests to assess spatial association between bull trout redds and VHGs. When χ^2 tests resulted in significant differences between used and available habitat, we used Jacobs' *D*, a modification of Ivlev's electivity index, as a nonstatistical indication of where differences between use and availability of habitat types occurred (Jacobs 1974). We considered statistical test results to be significant at $\alpha = 0.05$.

Results

Drainage scale

Among the four tributary streams sampled for VHG, the groundwater input, measured as the mean of positive VHG readings and as maximum positive VHG reading, was positively correlated with the area of bounded alluvial valley segments ($P < 0.05$ and $P < 0.01$, respectively). The mean of all VHG values, positive and negative, was significantly associated with only one geomorphic variable, the number of bounded alluvial valley segments ($P < 0.05$) per stream. Drainages with large and frequent bounded alluvial valley segments possessed more stream habitat that was thermally moderated by groundwater discharge than drainages dominated by confined stream segments and unbounded alluvial floodplain segments.

Among the nine principle spawning streams, the abundance of bull trout redds per tributary drainage was not associated with any of the traditional geomorphic metrics, such as drainage area or accessible stream length (Table 1). The distribution and abundance of redds were, however, posi-

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Fig. 3. Number of bull trout redds versus the proportion of the total drainage area that was bounded alluvial valley segment (BAVS) for the nine tributary spawning streams in the Swan River basin.

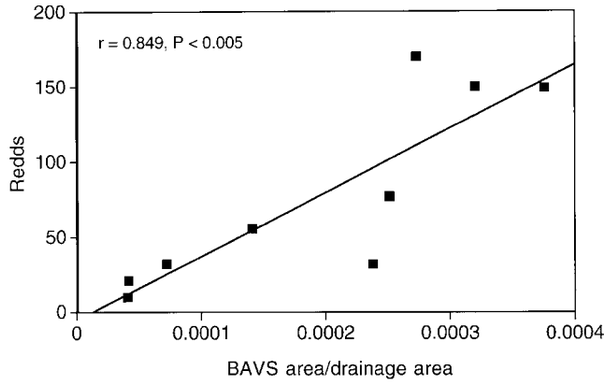
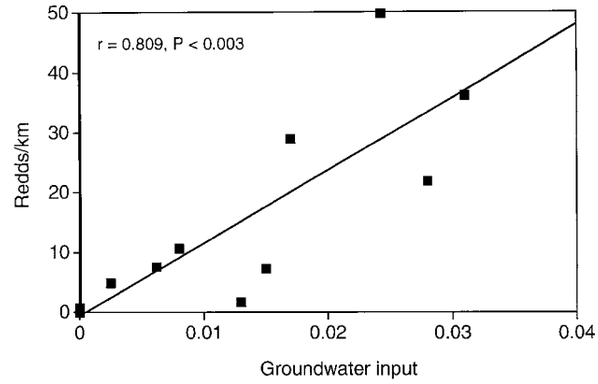


Fig. 4. Number of bull trout redds per stream kilometre in each of the alluvial valley segments sampled for VHG versus the relative strength of groundwater input (mean of transformed VHG readings, unitless) for each valley segment.



tively correlated with geomorphic measures of bounded alluvial valley segments. In particular, the number of redds was positively correlated with bounded alluvial valley area ($P < 0.005$) as well with as the proportion of the drainage area that was bounded alluvial valley ($P < 0.005$) (Fig. 3). In addition, the number of redds was associated with the total area of all bounded alluvial valley segments including those above fish barriers ($P < 0.05$) and with the proportion of the accessible stream length found within bounded alluvial valley segments ($P < 0.05$). Among the four streams sampled for VHG, we found that the relative strength of groundwater input and maximum positive VHG were positively associated with redd counts ($P < 0.05$ and $P < 0.01$, respectively).

Valley segment scale

Within the four streams sampled for VHG, alluvial valley segment types differed significantly in hyporheic exchange depending on whether they were bounded or unbounded by a downstream knickpoint (Mann–Whitney U, $P < 0.010$). All bounded alluvial valley segments possessed large and significant upwelling zones, while three of the four unbounded alluvial valley segments were exclusively losing or neutral and the fourth showed only weak zones of positive VHG in isolated sites. We also observed a close association between bounded alluvial valley segment area and the relative strength of groundwater input to a segment ($P < 0.01$). Likewise, bounded alluvial valley segments possessed more stream length that was thermally moderated by groundwater discharge than either unbounded alluvial valley segments or confined segments. They also had the greatest amount of ice-free habitat in the winter.

Among all valley segments in the nine principal spawning streams, bull trout redds were positively associated with bounded alluvial valley segments ($\chi^2 = 917$, $P < 0.0001$). Of the redds observed, 88.5% were located in bounded alluvial valley segments, whereas 8% were found in unbounded alluvial floodplain segments and only 3.5% were observed in confined valley segments. Of the available stream length, 38% was located in bounded alluvial valley segments, 56% in unbounded alluvial valley segments, and 5% in confined segments. The availability of bounded alluvial valley segments relative to their usage suggested a positive selection (Jacobs' $D = 0.450$). Additionally, among the valley segments of the four VHG-sampled tributaries, the relative strength of groundwater input in a segment was positively associated with both the number of redds ($P < 0.05$) and the redd density ($P < 0.005$) (Fig. 4). When only bounded alluvial valley segments were considered, the number of redds was weakly correlated with groundwater input ($P < 0.1$).

Reach scale

Within bounded alluvial valley segments, the geomorphic context of a reach along the longitudinal gradient was often indicative of whether it was a gaining, losing, or neutral reach. Reaches at the upstream end of a bounded alluvial valley segment were usually downwelling, those in the middle were often neutral, and those at the downstream end, just upstream of the bounding knickpoint or valley constriction, were always upwelling. The mean VHG per reach was negatively associated with the mean valley bottom width at each reach ($P < 0.005$). Reaches in the middle of bounded alluvial valley segments had very wide valley bottom widths and were typically neutral or slightly negative in VHG. In addition, several reaches located in the unbounded alluvial valley segments of the four streams had wide valley bottom widths and were strongly downwelling. The thermal regimes of gaining reaches were significantly moderated by the groundwater effect (Fig. 5). Gaining reaches possessed the least ice cover even when the reaches were exposed to extremely cold winter air temperatures. These moderating effects dissipated with increasing downstream distance from the upwelling zone.

Reaches that were gaining water (mean VHG = 0.0169) had the highest number of observed bull trout redds ($n = 94$). Among the reaches sampled, there were no significant differences in redd distribution associated with geomorphic variables (e.g., reach gradient, valley bottom width) other than the proximity to a downstream knickpoint. The number of bull trout redds per reach was positively correlated ($P < 0.001$) with the relative strength of groundwater input per reach (Fig. 6). Furthermore, since the scatter of points on this graph was somewhat clustered, we broke the reaches into categories of low (<0.03) and high (≥0.03) groundwater input and tested the observed versus random distribution of redds among reaches. We found the difference to be highly

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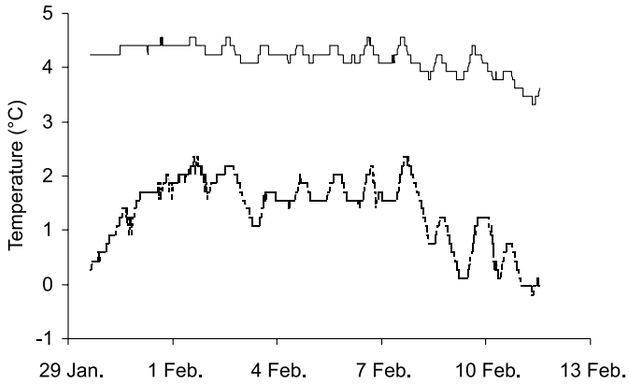
Fig. 5. Thermal profiles for two winter weeks from a gaining (solid line) and a losing (broken line) reach of Lion Creek clearly show the moderating effects (both seasonal and diel) of groundwater. The supercooled temperature of the losing reach at the end of the profile corresponded to anchor ice formation. Twenty-three bull trout redds were counted in the gaining reach, while none were found in the losing reach.



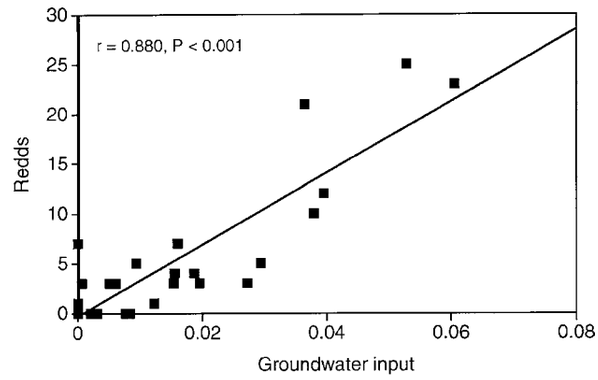
significant (χ^2 = 215, P < 0.0001), and reaches with strongly positive groundwater input were selected for spawning based on their availability (Jacobs' D = 0.795).

Pool–riffle scale

Within reaches in Lion and Cold creeks, we observed an association between bedform morphology, hyporheic exchange, and redd distribution. These reaches possessed significant sources of groundwater discharge, had moderated thermal regimes, and were free of ice throughout the winter. However, VHG varied significantly with local bedform character (Fig. 7). Convex and transitional bedforms were significantly different in VHG from concave bedforms (Mann–Whitney U = 64, P < 0.0001), in that areas of the streambed that were transitional or convex possessed negative VHG with means of –0.099 and –0.042, respectively, while concave streambed displayed upwelling VHG (mean = 0.038). In each reach, we observed several instances in which large wood contributed to local variation in bed topography and heterogeneity in the magnitude and direction of VHG. For logs that were positioned perpendicular to the current and that created a pool, shallow downwelling occurred above the logs and upwelling occurred in the pools below the logs. Estimated K ranged from 2.32×10^{-6} cm/s to a maximum of 3.37×10^{-1} cm/s, a difference of more than five orders of magnitude. Falling head test data indicated that transitional bedforms possessed significantly higher intragravel v than either convex or concave bedforms (Mann–Whitney U, P < 0.0001).

Bull trout redds were located most frequently in transitional as opposed to convex or concave bedform classes (χ^2 = 54, P < 0.0001) (Fig. 7). Of the 15 redds found in the Lion Creek and Cold Creek reaches, only one of these was located in a concave bedform unit. There were two transitional units where no redds were constructed; however, the availability of transitional areas relative to their usage was suggestive of positive selection (Jacobs' D = 0.895). Examination of individual redds revealed that spawning sites were

Fig. 6. Number of bull trout redds observed in each of the reaches sampled for VHG versus the groundwater input (mean of transformed VHG readings) for each reach.



strongly associated with piezometers having a negative VHG to the second nearest neighbor (Mann–Whitney U = 32, P < 0.001). The median distance from redds to the nearest downwelling piezometer was only 2.1 m, while the median distance to the nearest upwelling piezometer was 14.6 m. Furthermore, after accounting for the relative availability of downwelling versus upwelling habitat within these reaches, we found a significant difference between the expected and the observed distributions of redds (χ^2 = 8.99, P < 0.005), suggesting positive selection of transitional bedforms that are also downwelling areas (Jacobs' D = 0.751) (Fig. 7).

The distribution of redds was also associated with the magnitude of intragravel v. Redds were located in areas with the highest flow (indicated by piezometric falling head tests), while those with the weakest flow were never used (Fig. 7). We separated the estimates of v (i.e., v using Darcy's equation) in the two mapped reaches into three categories of low (<0.001 cm/s), medium (0.001–0.01 cm/s), and high (>0.01 cm/s). After accounting for the relative availability of habitat in each intragravel flow category, we found a significant difference between the expected and the observed distributions of redds (χ^2 = 17.98, P < 0.0001), suggesting positive selection of areas with high intragravel v (Jacobs' D = 0.727).

Discussion

The results of this study describe how groundwater – stream water exchange is structured by geomorphic features at drainage, valley segment, stream reach, and channel unit scales. Our findings also show that the selection of spawning sites by bull trout is strongly associated with hyporheic exchange at all of these scales. The mechanisms governing bull trout spawning habitat selection may vary from one scale to another within the hierarchy. Furthermore, although we are unsure of the scale to which homing occurs in this species (Rieman and McIntyre 1993), there is a transition in process that occurs between homing and redd site selection (Northcote 1978). Spawning habitat selection is a complex function of the results of differential survival by embryos and rearing juvenile fish, along with fish sensory capabilities. Although our study does not explicitly determine causal mechanisms, we illustrate how hyporheic exchange may control a variety

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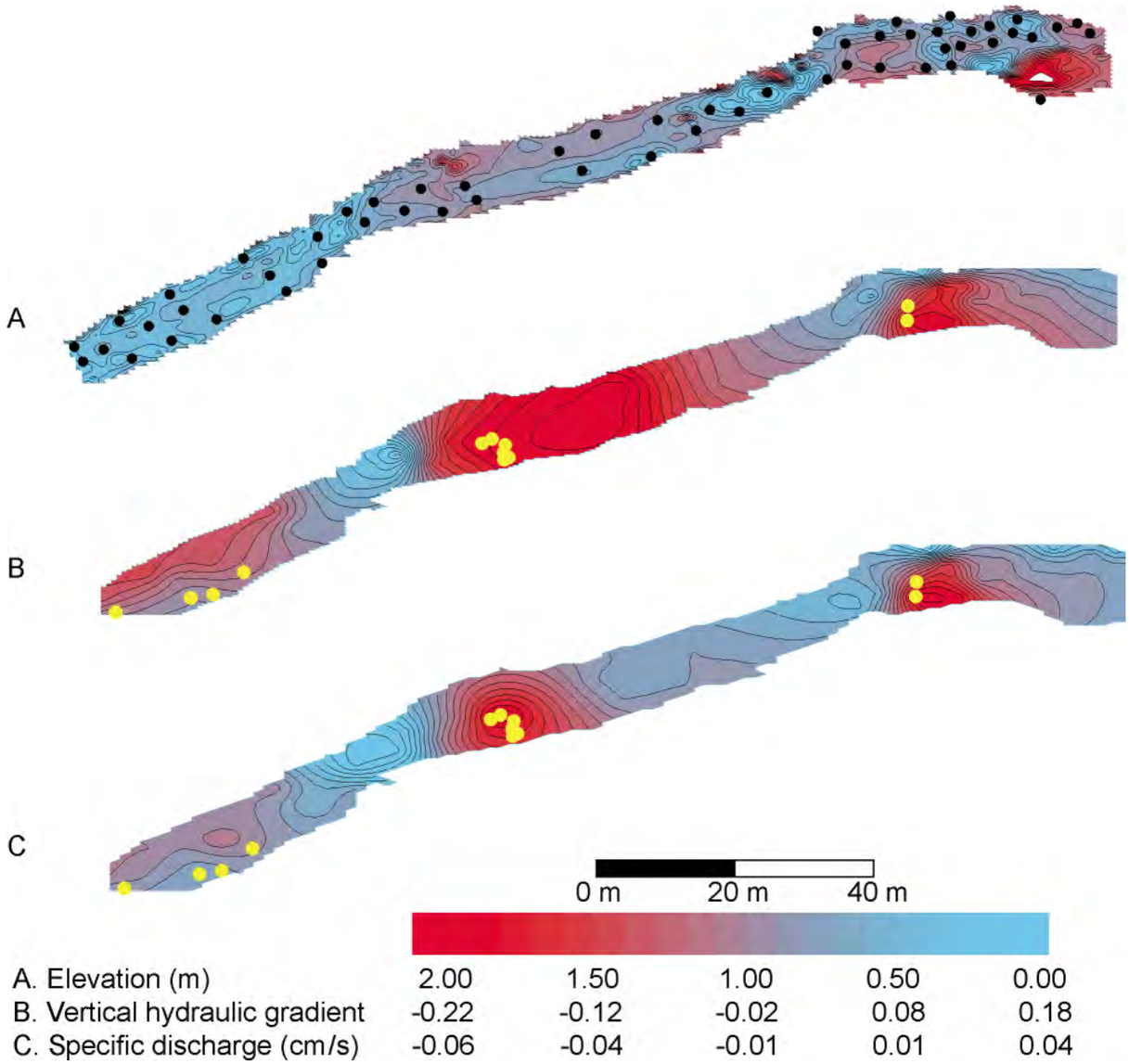
Fig. 7. Three maps of a spawning reach on Lion Creek showing (A) channel bed topography and piezometer locations, (B) isopleths of VHG grading from positive VHG (blue) to negative VHG (red), and (C) isopleths of intragravel *v* grading from high (red) to low (blue) rates. Dots indicate piezometer locations in map A and bull trout redds in maps B and C. The direction of flow is from right to left.



of habitat factors (i.e., temperature, intragravel conditions, ice conditions) that may directly affect an integrated process of spawning habitat selection.

Geomorphic controls on hyporheic exchange

Our results demonstrate that the integration of hydrology and geomorphology across spatial scales plays a critical role in structuring hyporheic exchange patterns. Within a drainage, we observed that bounded alluvial valley segments and their associated hyporheic zones were structured longitudinally like beads on a string (sensu Stanford and Ward 1993). Among the study drainages, the amount of groundwater discharge to a stream appeared to vary closely with the size and frequency of bounded alluvial valley segments. The presence of knickpoints, particularly on the downstream end of alluvial valley segments, was critical to the constraining of hyporheic flow. The importance of alluvial floodplain segments and knickpoints to ecosystem structure and function has been observed in large montane rivers (Stanford and Ward 1993). Our results are consistent with the concept that these landforms play a significant role in structuring hyporheic exchange in small tributary systems as well.

Within tributary drainages of the Swan basin, we found that alluvial valley segment types differed in their hyporheic exchange character depending on whether they were bounded or unbounded by a downstream knickpoint. Bounded alluvial valley segments had the most variation in hyporheic exchange and the strongest upwelling zones. Moreover, the greater the spatial extent of a bounded alluvial valley segment, the greater the strength of groundwater input

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was within the segment. Unbounded alluvial valley segments were always associated with a tributary's confluence with the Swan River and were characterized by reaches with strongly negative VHG. These contrasts and associations are evidence that geomorphic constraint on the length, lateral complexity, volume of alluvial fill, and character of geomorphic knickpoints in alluvial valleys affects hyporheic exchange.

Reaches with upwelling hyporheic groundwater were almost exclusively located within bounded alluvial valley segments. Within a bounded alluvial valley segment, reaches at the upstream end were usually downwelling, and reaches at the downstream end of the floodplain just upstream of the confining knickpoint were upwelling. These sites were also marked by narrowing valley widths partly because the valley walls of bounded alluvial valley segments typically begin to pinch together as the downstream knickpoint is reached. These patterns are consistent with what might be expected based on the application of basic principles of hydrogeology (Freeze and Cherry 1979) at the scale of an alluvial valley segment. Although reaches located at the downstream ends of bounded alluvial valley segments were gaining subsurface water, these reaches were not the only ones with groundwater discharge. In part, we attribute this to nonhyporheic inputs (e.g., seeps from bedrock fissures) that were independent of hyporheic flow pathways. We also observed relatively impermeable clay layers that probably contributed to localized upwelling (i.e., positive VHG) measurements in some instances. In addition, in Cold Creek, we observed negative VHG above a large beaver dam complex and positive VHG for several hundred metres downstream, a phenomenon that has been observed by others (e.g., White 1990).

Within reaches, we also observed associations between geomorphology and hyporheic exchange. We observed that VHG varied significantly with bedform character, as has been shown by others (e.g., Vaux 1968; Thibodeaux and Boyle 1987; Harvey and Bencala 1993). Convex and transitional bedforms displayed localized, shallow downwelling, while concave areas possessed upwelling. In several instances, large wood appeared to create local variation in bed topography and heterogeneity in the magnitude and direction of VHG, where shallow downwelling occurred above logs and upwelling occurred in the pools below logs.

The K (as measured using falling head tests in piezometers) and intragravel v (estimated via Darcy's equation) were highly variable within stream reaches. Transitional bedforms located at the tailout of pools (i.e., just upstream of riffles) possessed the highest values of K and v. This observation concurs with laboratory flume experiments (Cooper 1965; Thibodeaux and Boyle 1987) in which the fastest intragravel flow rates were observed at the crests of waveforms as a consequence of pressure gradients. Qualitative observation of piezometer screens indicated the patchy distribution of clay or fine sediment at varying depths. Thus, both the shape of the stream bed and the structure of the stream substratum may contribute to the spatial variability in intragravel flow. In addition, although we did not investigate conditions within redds, the construction of redds may alter very localized patterns of VHG and may positively influence the K of the disturbed site (Cooper 1965).

At the drainage scale, the arrangement of bounded alluvial valley segments may influence the cumulative thermal and flow regimes of a stream. Although our study did not explicitly address this hypothesis, we did observe that drainages with larger, more frequent bounded alluvial valley segments possessed more stream habitat that was ice free during the winter. The strength and dissipation of the groundwater plume leaving the lower end of a bounded alluvial valley segment may affect the discharge and thermal regimes of areas downstream. We observed that areas influenced by groundwater discharge possessed relatively stable thermal regimes. Furthermore, reaches influenced by upwelling groundwater had less winter ice cover and experienced no anchor ice formation. This was in contrast with frequent occurrence of anchor ice in neutral and losing reaches that were not thermally buffered by upstream groundwater discharge.

Bull trout spawning patterns

The size and distribution of bounded alluvial valley segments on the landscape influence the distribution and abundance of bull trout spawning. In the Swan basin, bull trout redds were most abundant in drainages with extensive bounded alluvial valley segments. Previous landscape-scale analyses of bull trout spawning have suggested there is a significant association between redd frequency and drainage area and (or) stream order (Graham et al. 1981). These conventional drainage metrics may be important discriminators for presence–absence of bull trout spawning at broad spatial scales, as was suggested by Rieman and McIntyre (1995). However, they do not appear to be useful predictors of between-tributary variation in distribution or abundance among the principle spawning streams of the Swan basin.

We found that bull trout selected reaches for spawning with zones of hyporheic groundwater discharge. We also observed that these reaches consistently occurred within bounded alluvial valley segments. Various reach-scale geomorphic factors, such as stream gradient, have been identified as important factors affecting bull trout spawning habitat (Graham et al. 1981; Fraley and Shepard 1989). We found that low stream gradient was a character of both bounded and unbounded alluvial valley segments. Consequently, it was not a direct indicator of spawning reach selection. Rather, bull trout used reaches with hyporheic groundwater discharge that also possessed low stream gradient as a covariate.

At drainage and valley segment scales, the importance of hyporheic exchange and its influence on bull trout spawning may be related to temperature and flow regimes. Numerous researchers have identified alluvial valley segments as critical spawning habitat for many fish species (e.g., Copp 1989; Montgomery et al. 1999). In addition to habitat influenced by groundwater discharge, these segments contain large, stable expanses of well-sorted gravel and cobble. Furthermore, the presence of floodplain surfaces in these segments allows overbank flow, which moderates sediment scour associated with flooding. In contrast, suitable spawning gravel is sparse and individual patches are often small in confined valley segments where floodwaters are not attenuated by a floodplain.

Habitat-use and spawning patterns of bull trout are influenced by the geologic legacy of a basin. Longitudinally stepped drainage forms, bowl-like floodplain shapes, deep

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alluvial fill, high gravel porosity, and position of knickpoints are features influenced by glaciation. These features create deep hyporheic flow pathways and influence bull trout spawning habitat in the Swan basin. In contrast, hyporheic exchange in similar-sized, nonglaciated stream systems of northeastern Oregon appears to occur only at shallow, small spatial scales (C. Baxter, Oregon State University, Corvallis, Oreg., unpublished data). Thus, many bull trout streams lack the deep, extensive flow pathways typical of glaciated basins and their floodplains. As a consequence of these differences, it is important to consider groundwater – surface water interactions (Creuze des Chatelliers et al. 1994) and bull trout habitat-use patterns in their biogeoclimatic context.

Reaches influenced by groundwater discharge possess relatively stable thermal and flow regimes that may be important for bull trout egg incubation, emergence success, and the survival of juvenile bull trout. Differential survival at any of these stages could influence the distribution of adult spawning. In addition, homing by bull trout may be guided by cues that are related to the unique chemical signature of groundwater. Although little is known about groundwater effects on bull trout, numerous studies have demonstrated the importance of groundwater discharge zones to a related species, the brook trout (*Salvelinus fontinalis*), and other salmonids. For example, our observations of bull trout reflect Benson's (1953) finding that groundwater-influenced reaches of the Pigeon River in Michigan had less ice cover and appeared to be selected as spawning reaches by brook trout. In general, groundwater discharge sources provide cold-water refugia for fish in summer (Gibson 1966; Nielson et al. 1994) and warmwater refugia in winter (Craig and Poulin 1975; Cunjak and Power 1986). In addition, warm groundwater temperatures in winter inhibit the formation of anchor ice, which would otherwise cause high mortality in postemergent salmonids (Benson 1955).

At the pool–riffle spatial scale, we found that bull trout redds were associated with transitional bedforms at the tailouts of pools that formed at the heads of riffles. These sites also possessed localized downwelling (i.e., negative VHG) and high intragravel flow rates. Similarly, in a spawning tributary of the North Fork of the Flathead River, Weaver and White (1985) detected greater permeability in a bull trout spawning area than in a nonspawning area. Whether it is the direction or the magnitude of hyporheic exchange, or both, that is actually involved in redd site selection, we are unsure. Intragravel flow has long been recognized by fisheries biologists as important to the growth, development, and survival of salmonid eggs and fry. Numerous salmonid species tend to spawn in the transitional zone between pools and riffles where intragravel flow rates are high, and percolation through the redd is thought to provide a constant supply of oxygen to the eggs and effectively remove metabolic waste materials (Bjornn and Reiser 1991). Finer spatial resolution in measures of VHG and intragravel flow rate, along with quantification of other factors such as substrate character and intragravel dissolved oxygen levels, would further delineate the relative importance of hyporheic exchange to spawning site selection at the microhabitat scale. Study of bull trout sensory capabilities, along with their growth and survival to emergence and the success of rearing juveniles in habitats possessing variable hyporheic exchange characters, could shed more light on the underlying biological mechanisms of these habitat-use patterns.

It appears that there is a spatially nested relationship between hyporheic exchange and bull trout spawning habitat use. Although we found at the larger spatial scales that bull trout spawning was associated with zones of positive VHG, at the pool–riffle scale, we found bull trout redd distributions associated with localized downwelling (i.e., negative VHG) and transitional bedforms between pools and riffles. Without a hierarchical approach to the interaction between hyporheic exchange and bull trout spawning, we would have obtained an erroneous picture of spawning habitat selection. For example, if we had only examined hyporheic exchange in the immediate vicinity of redds, we might have concluded that spawning habitat selection occurred within downwelling areas and had nothing to do with upwelling groundwater. Alternatively, had we ignored the smaller scale, the unique associations that we observed between VHG, transitional bedforms, intragravel flow, and specific redd placement would have been missed.

Much of the fisheries research on groundwater effects has focused on characteristics of the environment in and immediately surrounding redds (e.g., Webster and Eriksdottir 1976; Curry and Noakes 1995) and the consequences for growth and survival to emergence in redds with varying rates of subsurface flow (e.g., Sowden and Power 1985). The findings of this study suggest the drainage and valley segment scale context of a spawning site or a spawning reach are important considerations. Variation of some habitat factors may not be detectable or important to bull trout at all scales. Hyporheic exchange, however, appears to be important to spawning habitat selection processes at each of the spatial scales examined. In much of the fisheries research attempting to characterize habitat use, habitat variables have been considered as describing local conditions, regardless of their primary scales of heterogeneity. As we and other authors (e.g., Poizat and Pont 1996; Torgersen et al. 1999) have observed, this approach may obscure meaningful patterns by mixing specific organismal responses to habitat factors that vary differently at different scales.

The findings of this study emphasize the importance of considering local geomorphic context and spatial scale when describing stream processes. These factors are treated incompletely within the existing paradigm of the river continuum concept (Vannote et al. 1980; Minshall et al. 1985). Rather, our observations of the importance of local variability in hydrogeomorphic processes to habitat use by spawning bull trout may represent an example of the applicability of Montgomery's (1999) process domain concept. A landscape context is important to a better understanding of hyporheic processes, as geomorphology plays a scale-specific role in patterning variation of hyporheic exchange. Many studies of hyporheic processes have been done within a single, short stream reach (e.g., Harvey and Bencala 1993). Our observations demonstrate that the valley segment and drainage scale context of a reach could influence the findings of such a study and that quantifying hyporheic exchange in a spatially extensive manner is both critical and feasible.

Conserving the bull trout and other fish species that pos-

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sess complex migratory life histories demands a landscape perspective (Schlosser 1995). Our findings demonstrate that the conservation of bull trout and the maintenance and (or) restoration of spawning habitat will require consideration of groundwater – surface water exchange within a hierarchical, geomorphic context. For example, the distribution of bounded alluvial valley segments on the landscape may be significant to bull trout restoration plans in this region because of the importance of this distribution to understanding the spatial geometry of bull trout habitat (e.g., Rieman and McIntyre 1995; Dunham and Rieman 1998). This study has helped to identify one component of the functional combination of habitats in the landscape mosaic required by this species. Finally, these findings have contributed to the growing body of research demonstrating the importance of hyporheic processes to the structure and function of stream ecosystems.

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Reduced genetic variation in insular northern flying squirrels (*Glaucomys sabrinus*) along the North Pacific Coast

Allison L. Bidlack[1] and Joseph A. Cook[1,2]

[1]Institute of Arctic Biology and University of Alaska Museum, 907 Yukon Drive, Fairbanks, AK 99775–6960, USA
[2]Biological Sciences, Idaho State University, Pocatello, ID 83207–8007, USA

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Abstract

Nearshore oceanic archipelagos are natural laboratories that could provide valuable insight into the role of evolutionary processes such as founder effects and incipient speciation in biotic conservation. The Alexander Archipelago of Southeast Alaska is an example of such a complex, yet few biological investigations have been conducted. For the past 50 years, the region has experienced intense anthropogenic disturbances (particularly timber harvesting), causing habitat fragmentation and potential disruption of biotic communities. As part of a series of studies of mammals endemic to Southeast Alaska, we examined mitochondrial DNA sequences from 118 flying squirrels to investigate genetic diversity across Southeast Alaska. Mitochondrial sequence divergence corroborates the subspecific designation of the endemic Prince of Wales flying squirrel (*Glaucomys sabrinus griseifrons*). This island lineage exhibits severely reduced genetic variation and may be the result of an early Holocene founder event. Nearly all of the animals we examined on Prince of Wales Island and ten islands to the west had identical cytochrome *b* (52 of 53) and control region (21 of 21) sequences. In contrast, substantial polymorphism and little genetic structuring were found in comparable populations on the mainland of Southeast and Interior Alaska. Because flying squirrels in the Pacific Northwest are associated with old-growth forest, forest-use plans should aim to conserve this unique lineage of island squirrels.

INTRODUCTION

Archipelagos play a prominent role in conservation because they often harbour endemic species, which are especially susceptible to extinction (Diamond, 1989). Considerable attention has been paid to loss of diversity on remote oceanic archipelagos, like the Hawaiian and Galapagos islands, because of their unique flora and fauna. Less research has focused on nearshore oceanic archipelagos, though similar evolutionary forces shape these ecosystems. Often these islands harbour endemic taxa and unique combinations of plants and animals not found on the mainland (e.g., Cook & MacDonald, 2001); they also provide opportunities to explore metapopulation dynamics, such as colonization and extinction (e.g., Lomolino, 1994; Giles & Goudet, 1997; Conroy, Demboski & Cook, 1999). In particular, founder events can impact the genetic variation found in island populations and lead to divergent evolutionary lineages. With few exceptions (e.g., Ranta *et al.*, 1999), conservation on these nearshore archipelagos has received little attention in the literature.

High numbers of nominal endemic species and subspecies occur along the North Pacific Coast of North America. Twenty-four taxa of mammals are considered endemic to Southeast Alaska, with 12 others largely confined to the region (MacDonald & Cook, 1996). Many of these endemics are spread over the Alexander Archipelago, which consists of over 2000 named islands (Fig. 1). This complex landscape has been further fragmented by extensive timber harvests and road building during the past 50 years (Durbin, 1999). Rudimentary inventories led to uncertainty over the distribution and abundance of endemic taxa; however, recent phylogeographic work highlights the need to incorporate information on endemics and evolutionary processes on islands into management plans for the region (Cook *et al.*, 2001).

Because northern flying squirrels (*Glaucomys sabrinus*) generally are associated with old-growth forest in the Pacific Northwest (Carey, 1995, 1996; but see Rosenberg & Anthony, 1992), the Prince of Wales flying squirrel (*G. s. griseifrons*) has been the focus of conservation concern (Demboski, Cook & Kirkland, 1998*a*).

All correspondence to: Allison L. Bidlack. Current address: Environmental Science, Policy and Management, University of California, Berkeley, 201 Wellman #3112, Berkeley, CA 94720-3112; Tel: 510-643-1227; E-mail: abidlack@nature.berkeley.edu.

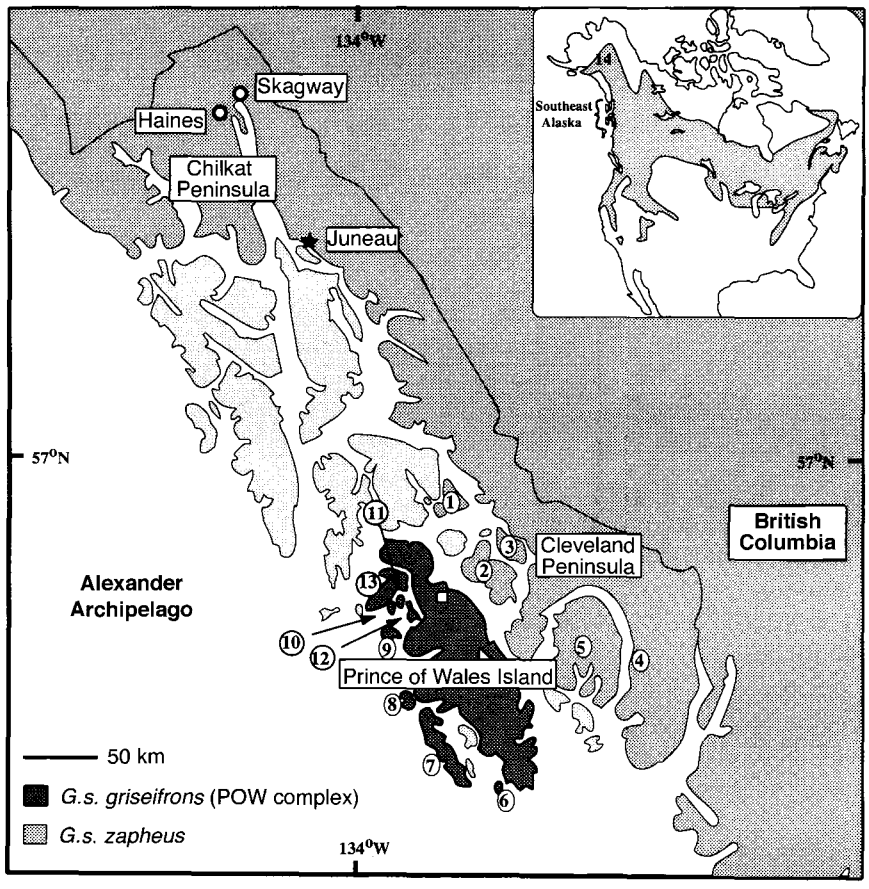

Fig. 1. Distribution of flying squirrels in Southeast Alaska, including the type locality for *Glaucomys sabrinus griseifrons* (□). Lightly shaded islands do not have flying squirrels or have not been inventoried. Inset map shows North American range of *G. sabrinus*. Numbers on map represent selected sample localities: (1) Mitkof I., (2) Etolin I., (3) Wrangell I., (4) Rudyerd Bay, (5) Revillagigedo I., (6) Barrier Island group, (7) Dall I., (8) Suemez I., (9) Heceta I., (10) Orr I., (11) El Capitan I., (12) Tuxekan I., (13) Kosciusko I., (14) Interior Alaska. Other sampling localities indicated by name.

Prince of Wales Island in the Alexander Archipelago has been heavily deforested and extensive future timber harvests are planned. Demboski, Jacobsen & Cook's (1998*b*) preliminary assessment of geographic variation in northern flying squirrels in Southeast Alaska used sequences of the mitochondrial cytochrome *b* gene. They characterized three subspecies from Alaska (*G. s. griseifrons, G. s. zapheus, G. s. yukonensis*), and found that animals from Prince of Wales, and eight nearby islands (POW complex), shared two diagnostic mutations across 1440 base pairs examined.

The mitochondrial control region evolves faster than cytochrome *b*, and usually provides greater resolution of recent evolutionary and population-level events (e.g., Thomas *et al.*, 1990; Encalada *et al.*, 1996; Pope, Sharp & Moritz, 1996; Good *et al.*, 1997). We examined control region sequence variation and expanded Demboski *et al.*'s (1998*b*) cytochrome *b* data by adding complete sequences for animals from an additional four islands and three mainland areas. The pronounced signal of a founder event on the POW complex highlights the value of documenting intraspecific diversity across nearshore archipelagos, particularly when large-scale anthropogenic disturbances may impact insular endemic populations.

MATERIALS AND METHODS

Heart tissue housed in the Alaska Frozen Tissue Collection at the University of Alaska Museum from 118 specimens (representing 15 islands and 8 mainland populations) was used for DNA extraction. Specimens, including many provided by marten trappers, were sampled from 22 locations (sample number in parentheses): Prince of Wales I. (21), Tuxekan I. (2), Suemez I. (6), Orr I. (2), Heceta I. (4), El Capitan I. (6), Barrier Islands (12), Kosciusko I. (3), Dall I. (1), Revillagigedo I. (4), Wrangell I. (1), Etolin I. (1), Mitkof I. (9), Chilkat Peninsula (2), Skagway (1), Haines (1), Juneau (3), Rudyerd Bay (1), Cleveland Peninsula (20), Yukon Territory (1), Interior Alaska (15) and Washington state (2). Extraction followed a protocol modified from Miller, Dykes & Polesky (1988). Partial (792 base pairs) and complete (1140 base pairs) cytochrome *b* gene sequences of 86 animals (n = 49 partial; n = 37 complete) were amplified using primers MVZ 04/05, 16/37 and 14/23 (Smith & Patton, 1993); 350 base pairs from the 5′ end of the mitochondrial control region were obtained from 42 of these 86 individuals using primers TDKD (5′–CCT GAA GTA GGA ACC AGA TG;

Kocher *et al.*, 1993) and CTRL-L (5′-CAC YWT YAA CWC CCA AAG CT). Control region sequences were obtained from an additional 34 animals which were not sequenced for cytochrome *b*. Sequences were amplified by polymerase chain reaction on a Perkin-Elmer 2400 thermal cycler using standard protocols (Lessa & Cook, 1998). Both forward and reverse strands were sequenced on an ABI 373 Automated Sequencer and translated and aligned with Sequence Navigator Version 1.0.1 (ABI). All amplifications included negative controls. Control region and cytochrome *b* sequences have been deposited in GenBank under Accession numbers AF011738–AF011742, AF030389–AF030394, AF271669, AF271673, AF271668, AF359135–AF359213, AF359216–AF359218, AF359220–AF359233.

The program PAUP* (Swofford, 1999) was used to calculate genetic distances (both uncorrected 'p' and GR+I+Γ) among all individuals for which complete cytochrome *b* and 350 bp of control region was sequenced (n = 32). Parameters for maximum likelihood were estimated (Sullivan, Swofford & Naylor, 1999), and likelihood ratio tests were performed among them to determine the best evolutionary model to create a maximum-likelihood tree. Sequences from 32 individuals were used to construct a maximum-likelihood tree using the general time reversible model with invariable sites plus gamma (GTR+I+Γ). Fifty bootstrap replicates were performed across the tree to determine strength of relationships.

RESULTS

Cytochrome *b* and control region sequences show a division between the POW complex and other populations, reinforcing and clarifying the preliminary conclusions of Demboski *et al.* (1998*b*). All cytochrome *b* sequences (n = 53) from the POW complex are identical, except for one from the southern end of Prince of Wales Island which has one additional base pair transition. All 53 individuals from the POW complex are characterized by two synonymous base pair changes (one transition, one transversion) compared with all other individuals examined. Cytochrome *b* sequences from animals from nearshore islands (Wrangell, Etolin, Mitkof and Revillagigedo) and mainland sites (Juneau, Skagway, Haines, Chilkat Peninsula, Cleveland Peninsula and Rudyerd Bay) differ from those of the POW complex by at least these two base pair changes (Table 1).

Excluding the divergent Washington sequences, the 24 variable sites in the control region (of 350 base pairs; Table 2(a)) comprise 22 haplotypes. All control region sequences from the POW complex (n = 21) are identical and represent a distinct haplotype. Similarly, all five individuals from Mitkof Island share a single haplotype, although this haplotype also occurs on the Cleveland Peninsula. In contrast to the monotypic POW complex samples, populations from the Cleveland Peninsula and Interior Alaska with large sample sizes (n > 14) include seven haplotypes each (Table 2(b)). Locations for which we sequenced only a few animals, such as Juneau, Revillagigedo Island and Chilkat Peninsula, were also variable. A single individual was examined for Etolin and Wrangell islands, Rudyerd Bay, Haines and Skagway.

Genetic distances (Table 3) between the POW complex and individuals from the mainland or nearshore islands (excluding Washington state) ranged from 0.33% to 0.61% (uncorrected 'p'). Differences among individuals from the mainland and nearshore islands ranged from 0.07% to 0.7%. GTR distances are similar to these uncorrected distances (Table 3). These low levels of divergence allow only two supported clades to be defined: Haines–Skagway (78%), and POW–Southern POW (80%; Fig. 2).

DISCUSSION

Mainland vs. island patterns of variation

Four minimally divergent cytochrome *b* haplotypes were found along the Southeast Alaska mainland. The level of variation among these haplotypes is consistent with cytochrome *b* diversity in other mammals sampled from the mainland (Cook *et al.*, 2001). *Glaucomys sabrinus* is unique, however, in having so little cytochrome *b* diversity across 11 islands of the Alexander Archipelago. Similarly, while only one control region haplotype was found on the POW complex, four control region haplotypes on the nearshore islands of Revillagigedo, Wrangell, Mitkof and Etolin and 12 haplotypes on the mainland were identified. Many of these haplotypes are unique to a particular area (i.e., Rudyerd Bay, Haines, Skagway; Table 2(b)); however, small sample sizes (n = 1 for several localities) and incomplete geographic sampling preclude robust analysis of population genetic

Table 1. Cytochrome *b* haplotypes and nucleotide positions of substitutions

	Cytochrome *b* position and type of base pair substitution							
Haplotype	150	318	394	468	480	574	903	1131
Mainland	T	C	G	C	T	C	T	C
POW Complex	.	.	.	.	.	T	.	A
South POW	.	.	A	.	.	T	.	A
Haines & Skagway	.	.	.	T	.	.	.	.
Juneau & Chilkat Peninsula	.	.	.	.	C	.	.	.
Wrangell Island	.	T	.	.	.	.	.	.
Rudyerd Bay	C	.	.	.	.	.	.	.
Yukon Territory	.	.	.	.	.	.	C	.

Table 2(a). Control region haplotypes and positions of substitutions from 5′ end

Haplotype	77	96	106	114	115	149	150	151	173	174	178	230	252	255	258	259	272	277	305	308	312	315	322	341
A POW Complex	C	C	G	C	G	T	C	C	C	T	C	T	A	T	C	T	T	T	A	A	A	A	T	C
B Rudyerd Bay	T	.	.	.	.	.	.	T	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
C Wrangell I. and Cleveland Pen.	.	.	.	.	A	.	.	.	.	C	.	C	.	.	.	C	.	.	.	.	G	.	.	.
D Mitkof I., Etolin I. Cleveland Pen.	T	.	.	.	A	.	.	.	.	C	.	C	.	.	.	C	.	.	.	.	G	.	.	.
E Cleveland Pen.	.	A	.	.	A	.	.	.	.	.	T	C	.	.	.	.	.	.	.	.	.	.	.	.
F Revillagigedo I.	.	.	.	.	A	.	.	.	.	T	T	C	.	.	.	C	.	.	.	.	.	.	.	
G Revillagigedo I.	.	.	.	.	A	.	.	.	T	.	T	C	G	.	.	.	C	.	.	.	.	.	.	.
H Cleveland Pen.	.	.	A	.	A	.	.	.	.	.	.	C	.	.	T	.	.	.	.	.	.	.	.	.
I Cleveland Pen.	.	.	A	.	.	.	.	.	.	.	.	C	.	.	T	.	.	.	.	.	.	.	.	.
J Cleveland Pen.	.	.	.	.	A	.	.	.	.	.	.	C	.	.	T	.	.	.	.	.	.	.	.	.
K Cleveland Pen.	T	.	.	.	A	.	.	T	.	C	.	C	.	.	.	C	.	.	.	.	G	.	.	.
L Haines	.	.	.	.	A	.	.	.	T	C	.	C	.	.	.	.	.	C	.	.	.	.	.	.
M Skagway	.	.	.	.	A	.	.	.	T	C	.	C	.	.	T	.	.	C	.	.	.	.	.	.
N Chilkat Pen.	.	.	A	.	A	.	.	.	.	.	.	C	.	.	.	.	.	.	.	.	.	G	.	.
O Juneau & Chilkat Pen.	.	.	.	.	A	.	.	.	.	.	.	C	.	.	.	.	C	C	.	.	.	G	C	.
P Interior Alaska	.	.	.	.	A	.	.	.	.	.	T	C	.	C	T	.	C	.	.	.	.	.	.	.
Q Interior Alaska	.	.	.	.	A	.	.	.	.	.	T	C	.	.	T	.	.	.	.	.	.	.	.	T
R Interior Alaska	.	.	.	.	A	.	.	.	.	.	T	C	.	.	T	.	.	.	.	.	.	.	.	.
S Interior Alaska	.	.	.	.	A	C	.	.	.	.	T	C	.	.	T	.	.	.	.	.	.	.	.	.
T Interior Alaska	.	.	A	.	A	.	.	.	.	.	.	C	.	.	T	.	.	.	.	G	.	.	.	T
U Interior Alaska	.	.	A	T	A	.	T	.	.	.	.	C	.	.	T	.	.	.	G	G	.	.	.	T
V Interior Alaska	.	.	.	T	A	.	.	.	.	.	.	C	.	.	T	.	.	.	G	.	.	.	.	T

Table 2(b). Distribution of control region haplotypes in *Glaucomys sabrinus* in Southeast and Interior Alaska

Subspecies	Sample locations	Haplotypes A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q	R	S	T	U	V	Total
G. s. griseifrons	POW I.	6	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	6
	Dall I.	1	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	1
	El Capitan I.	2	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	2
	Heceta I.	2	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	2
POW Complex	Kosiusko I.	2	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	2
	Orr I.	2	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	2
	Suemez I.	2	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	2
	Tuxekan I.	2	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	2
	Barrier Is.	2	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	2
G. s. zapheus	Mitkof I.	—	—	—	5	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	5
	Etolin I.	—	—	—	1	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	1
	Wrangell I.	—	—	1	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	1
	Revillagigedo I.	—	—	—	—	—	3	1	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	4
	Rudyerd Bay	—	1	—	—	—	—	—	—	—	—	—	—	—	—	3	—	—	—	—	—	—	—	1
	Juneau	—	—	—	—	—	—	—	—	—	—	—	—	—	1	1	—	—	—	—	—	—	—	3
	Chilkat Pen.	—	—	1	6	4	—	—	3	3	2	1	—	—	—	—	—	—	—	—	—	—	—	2
	Cleveland Pen.	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	20
	Skagway	—	—	—	—	—	—	—	—	—	—	—	—	1	—	—	—	—	—	—	—	—	—	1
	Haines	—	—	—	—	—	—	—	—	—	—	—	1	—	—	—	—	—	—	—	—	—	—	1
G. s. yukonensis	Interior AK	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	1	2	1	2	1	3	4	14
Total		21	1	2	12	4	3	1	3	2	2	1	1	1	1	4	1	2	1	2	1	3	4	74

structure. A maximum-likelihood phylogeny based on cytochrome *b* and control region sequences is poorly resolved (Fig. 2). Only two clades are well supported (> 75%), one that includes Haines and Skagway, and the other representing the two haplotypes of the POW complex. This minimal geographic structuring along the mainland possibly reflects a rapid expansion into Southeast Alaska after the last glacial maximum (22,000–18,000 BP).

Colonization of islands may result in a loss of genetic diversity, with subsequent differentiation between the original and founder populations induced by genetic drift or differential selective regimes. Many insular mammal taxa exhibit lower genetic variation than their continental counterparts (e.g., Kilpatrick, 1981; Frankham, 1997, 1998). For example, the Channel Island fox, *Urocyon littoralis*, exhibits lower allozyme and mitochondrial haplotype diversity, and lower micro- and mini-satellite allelic diversity than mainland foxes (Gilbert *et al.*, 1990; Wayne *et al.*, 1991; Goldstein *et al.*, 1999). We suspect that the distinctive POW complex haplotypes may be due to a post-Pleistocene founder event to these islands, with the extremely low level of variation on the 11 POW complex islands (a single autapomorphic

Table 3. Genetic distance between combined cytochrome *b* and control region haplotypes; uncorrected 'p' below diagonal; GTR +Γ + I distances above diagonal

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
1. POW		0.0007	0.0044	0.0044	0.0044	0.0060	0.0069	0.0045	0.0060	0.4053	0.3677	0.0060	0.0068	0.0052	0.0036	0.0060	0.0060	0.0036	0.0069	0.0052
2. Southern POW	0.0007		0.0052	0.0053	0.0053	0.0069	0.0078	0.0053	0.0069	0.4123	0.3745	0.0069	0.0077	0.0060	0.0044	0.0069	0.0070	0.0044	0.0078	0.0060
3. Revillagigedo I.	0.0040	0.0047		0.0014	0.0028	0.0014	0.0036	0.0029	0.0043	0.3795	0.3594	0.0014	0.0051	0.0021	0.0036	0.0044	0.0044	0.0051	0.0051	0.0021
4. Cleveland Pen.	0.0040	0.0047	0.0013		0.0029	0.0028	0.0052	0.0029	0.0043	0.3942	0.3652	0.0029	0.0051	0.0021	0.0037	0.0044	0.0044	0.0052	0.0052	0.0021
5. Cleveland Pen.	0.0040	0.0047	0.0027	0.0027		0.0028	0.0052	0.0014	0.0044	0.3688	0.3411	0.0044	0.0036	0.0021	0.0007	0.0044	0.0044	0.0052	0.0052	0.0021
6. Interior Alaska	0.0054	0.0060	0.0013	0.0027	0.0027		0.0051	0.0044	0.0059	0.3713	0.3514	0.0028	0.0051	0.0021	0.0036	0.0059	0.0059	0.0067	0.0067	0.0021
7. Juneau & Chilkat	0.0060	0.0067	0.0034	0.0047	0.0047	0.0047		0.0036	0.0051	0.3775	0.03651	0.0052	0.0059	0.0059	0.0061	0.0068	0.0068	0.0077	0.0077	0.0059
8. Chilkat	0.0040	0.0047	0.0027	0.0027	0.0013	0.0040	0.0034		0.0044	0.3789	0.3508	0.0045	0.0052	0.0036	0.0021	0.0045	0.0045	0.0053	0.0053	0.0036
9. Haines	0.0054	0.0060	0.0040	0.0040	0.0040	0.0054	0.0047	0.0040		0.3713	0.3514	0.0044	0.0007	0.0051	0.0052	0.0044	0.0044	0.0067	0.0051	0.0051
10. Washington	0.0848	0.0855	0.0821	0.0835	0.0808	0.0808	0.0815	0.0815	0.0808		0.0085	0.3754	0.3672	0.3754	0.3754	0.3847	0.3865	0.3942	0.3806	0.3796
11. Washington	0.0808	0.0815	0.0808	0.0808	0.0781	0.0795	0.0808	0.0788	0.0795	0.0074		0.3554	0.3474	0.3474	0.3474	0.3564	0.3580	0.3574	0.3524	0.3514
12. Revillagigedo I.	0.0054	0.0060	0.0013	0.0027	0.0040	0.0027	0.0047	0.0040	0.0040	0.0815	0.0801		0.0051	0.0036	0.0053	0.0060	0.0060	0.0068	0.0068	0.0036
13. Skagway	0.0060	0.0067	0.0047	0.0047	0.0034	0.0047	0.0054	0.0047	0.0007	0.0801	0.0788	0.0047		0.0043	0.0044	0.0051	0.0051	0.0076	0.0059	0.0043
14. Interior Alaska	0.0047	0.0054	0.0020	0.0020	0.0020	0.0020	0.0054	0.0034	0.0047	0.0815	0.0788	0.0034	0.0040		0.0029	0.0051	0.0051	0.0059	0.0059	0.0014
15. Cleveland Pen.	0.0034	0.0040	0.0034	0.0034	0.0007	0.0034	0.0054	0.0020	0.0047	0.0815	0.0788	0.0047	0.0040	0.0027		0.0053	0.0053	0.0044	0.0061	0.0029
16. Mitkof I., Etolin I. and Cleveland Pen.	0.0054	0.0060	0.0040	0.0040	0.0040	0.0054	0.0060	0.0040	0.0040	0.0828	0.0801	0.0054	0.0047	0.0047	0.0047		0.0014	0.0052	0.0007	0.0051
17. Wrangell I.	0.0054	0.0061	0.0040	0.0040	0.0040	0.0054	0.0061	0.0040	0.0040	0.0836	0.0809	0.0054	0.0047	0.0047	0.0047	0.0013		0.0068	0.0021	0.0051
18. Rudyerd Bay	0.0034	0.0040	0.0047	0.0047	0.0047	0.000	0.0067	0.0047	0.0060	0.0835	0.0795	0.0060	0.0067	0.0054	0.0040	0.0047	0.0061		0.0044	0.0059
19. Cleveland Pen.	0.0060	0.0067	0.0047	0.0047	0.0047	0.0060	0.0067	0.0047	0.0047	0.0047	0.0822	0.0795	0.0060	0.0054	0.0054	0.0054	0.0007	0.0020	0.0040	0.0059
20. Interior Alaska	0.0047	0.0054	0.0020	0.0020	0.0020	0.0020	0.0054	0.0034	0.0047	0.0821	0.0794	0.0034	0.0040	0.0013	0.0027	0.0047	0.0047	0.0054	0.0054	

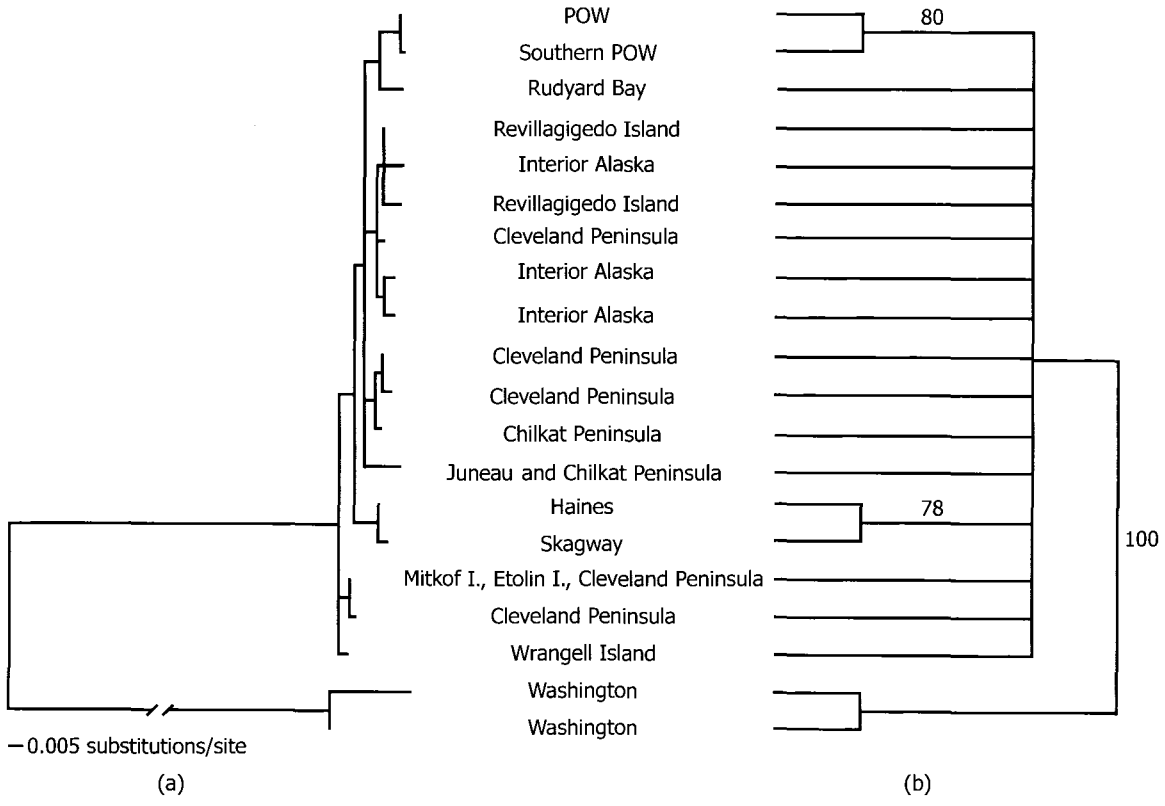

Fig. 2. Maximum-likelihood tree and bootstrapped ML tree of mitochondrial haplotypes of northern flying squirrels sampled from Southeast and Interior Alaska. Maximum-likelihood tree (a) shows minimal differentiation among haplotypes and little geographic structuring of populations. Two clades in Southeast Alaska are supported in the bootstrapped tree (b), with bootstrap percentages indicated above branches.

change in one individual) suggesting the possibility of a severe bottleneck in the ancestral population (Nei, Maruyama & Chakraborty, 1975). The apparent isolation of these islands, yet low level of genetic differentiation, indicates the Prince of Wales flying squirrel may offer an example of incipient speciation.

Phylogenetics and conservation

DNA sequencing and fingerprinting are commonly used to diagnose evolutionarily significant units (ESUs) and management units (MUs) because they provide efficient ways of gathering information about population subdivision in endangered or threatened taxa (Moritz, 1994). However, the prevailing dominance of genetic data over morphological or ecological information has raised concerns, including the possible misdiagnosis of populations (Paetkau, 1999), the potential dissociation between natural genetic markers and significant functional diversity (Pritchard, 1999; Crandall *et al.*, 2000), and the need to differentiate historical legacy from current dynamics (Crandall *et al.*, 2000). While these concerns are valid, genetic data may provide a first approximation of population subdivision, long before ecological or morphological differences can be assessed critically. Genetic data can also be used to assess diversity within a population, as limited genetic variability may be an important factor for the long-term viability of endangered taxa (Frankham, 1997; Newman & Pilson, 1997; Saccheri *et al.*, 1998). Our research is a first step towards quantifying population divergence in an insular endemic in Southeast Alaska because mitochondrial sequences can provide insight into both recent and historical genetic exchangeability (*sensu* Crandall *et al.*, 2000) among island and mainland populations.

Moritz's (1994) criteria for ESU designation were reciprocal mitochondrial monophyly and significant allele frequency divergence in nuclear alleles. MUs were recognized as populations with only significant allele frequency divergence at nuclear or mitochondrial loci. These criteria emphasized historical population structure and were based on theoretical rates of allele subdivision among populations. Moritz's criteria were not meant to be proscriptive, but rather meant to lay the groundwork for the entrance of genetic data into the ESU debate. Using Moritz's (1994) molecular criteria, mitochondrial sequences indicate that the POW flying squirrel should be minimally recognized as a MU because these populations have not experienced recent female-mediated gene flow with the mainland populations. Investigations of variability at nuclear loci may shed light on male-mediated gene flow between the POW complex and the mainland. Ecological characterization of these populations also would help evaluate functional diversity.

Because island systems tend to be simpler than continental ones, certain taxa may play different roles in these less complex systems. Both red squirrels (*Tamiasciurus*) and red-backed voles (*Clethrionomys*) are absent on the POW complex, and the effect of their absence on insular flying squirrels is unknown. All three species consume hypogeous fungi, and the two species of squirrel may compete for nesting space. In the absence of specific ecological studies, molecular research is providing a picture of historical population subdivision and limited genetic diversity in insular northern flying squirrels in the Tongass.

Conservation implications

The Tongass, covering 6.8 million coastal hectares, is the largest national forest in the United States. Unique challenges face managers across this complex landscape, and the current forest plan mandates the conservation of endemic taxa across the archipelago (United States Department of Agriculture, 1997). Mitochondrial sequences identify a unique and nearly monotypic clade of flying squirrels on the POW complex of islands, corroborating the original description of *G. s. griseifrons* (Howell, 1934) based on diagnostic morphological characters including a darker dorsal side, whiter ventral side, and greyer neck and head. To date, there have been few additional comparative morphological or ecological studies of subspecies of *G. sabrinus*.

Northern flying squirrels are associated with old-growth coniferous forests in the Pacific Northwest (Carey, 1995, 1996), where they utilize snags as nesting sites (Maser *et al.*, 1986). They play an important role in boreal forests by consuming and disseminating hypogeous fungal spores which are required by conifers for the uptake of nutrients. Flying squirrels are also a major prey item of many avian and mustelid predators (Forsman, Otto & Carey, 1991; Wilson & Carey, 1996). Habitat favoured by flying squirrels has been heavily fragmented in Southeast Alaska, with up to 46% of the old-growth of the Tongass harvested on some islands (Table 4; United States Department of Agriculture, 2000). Private land (220,000 ha) also has been heavily logged (Durbin, 1999). An additional 200 million board feet of timber is currently scheduled to be harvested from four islands within the range of *G. s. griseifrons* between the years 2000 and 2010 (Table 4). Most old-growth forest in Southeast Alaska that is protected from timber harvests (e.g., Admiralty Island) is not within the range of the POW flying squirrel.

This study is one of several concurrent studies of endemic mammals in the Alexander Archipelago. Molecular work is revealing a more dynamic picture of faunal movement and differentiation in Southeast Alaska than previously thought, with both palaeoendemics and neoendemics possibly present (Demboski, Stone & Cook, 2000; Cook *et al.*, 2001). We suggest that *G. s. griseifrons* is an example of the latter and, like several other island taxa, may be on a distinct evolutionary path. These studies emphasize the value of biogeographic studies of nearshore archipelagos; they may provide insight into incipient speciation and founder effects, and may reveal patterns relevant to conservation. Future land use should consider impacts on endemics along the North Pacific Coast. In particular, if the Prince of Wales flying squirrel is a subspecies endemic to some islands within the Tongass, as both morphologic and genetic data suggest, then the current logging pressure within its range may not be consistent with the management and conservation goals set forth in the new forest plan.

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Table 4. Timber harvests* on POW complex islands with *Glaucomys sabrinus griseifrons*; ROG, remaining old-growth habitat; H, harvested area; OGH, percentage of old-growth harvested to date; PTH, proposed timber harvest in million board feet. All areas are in hectares. Information is for USDA Forest Service land only.

	Island area	ROG	H	OGH	PTH
Barrier Islands	< 1000	498.4	0	0	0
Dall	65,820	26,046.6	1464.7	5.3	0
El Capitan	< 1000	484.2	27.3	5.3	0
Heceta	18,900	8497.0	6028.6	41.5	0
Kosciusko	48,220	20,420.3	7170.6	26.0	17
Orr	2335	1135.5	780.4	40.7	0
Prince of Wales	577,750	261,773.3	80,774.5	23.6	121–41
Suemez	15,012	8249.5	493.2	5.6	30
Tuxekan	8520	3572.1	3079.3	46.3	20

*as of April 2000

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ORIGINAL RESEARCH

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From canopy to seed: Loss of snow drives directional changes in forest composition

Sarah M. Bisbing[1] | Brian J. Buma[2] | Lauren E. Oakes[3,4] | John Krapek[5] | Allison L. Bidlack[6]

[1]Department of Natural Resources and Environmental Science, University of Nevada – Reno, Reno, Nevada, USA

[2]Department of Integrative Biology, University of Colorado, Denver, Denver, Colorado, USA

[3]Department of Earth System Science, Stanford University, Stanford, California, USA

[4]Climate Change Americas Program, Wildlife Conservation Society, Bozeman, Montana, USA

[5]Department of Natural Sciences, University of Alaska Southeast, Juneau, Alaska, USA

[6]Alaska Coastal Rainforest Center, University of Alaska Southeast, Juneau, Alaska, USA

Correspondence
Sarah M. Bisbing, Department of Natural Resources and Environmental Science, University of Nevada – Reno, 1664 N. Virginia St, Reno, NV, 89557, USA.
Email: sbisbing@unr.edu

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Abstract

Climate change is altering the conditions for tree recruitment, growth, and survival, and impacting forest community composition. Across southeast Alaska, USA, and British Columbia, Canada, *Callitropsis nootkatensis* (Alaska yellow-cedar) is experiencing extensive climate change-induced canopy mortality due to fine-root death during soil freezing events following warmer winters and the loss of insulating snowpack. Here, we examine the effects of ongoing, climate-driven canopy mortality on forest community composition and identify potential shifts in stand trajectories due to the loss of a single canopy species. We sampled canopy and regenerating forest communities across the extent of *C. nootkatensis* decline in southeast Alaska to quantify the effects of climate, community, and stand-level drivers on *C. nootkatensis* canopy mortality and regeneration as well as postdecline regenerating community composition. Across the plot network, *C. nootkatensis* exhibited significantly higher mortality than co-occurring conifers across all size classes and locations. Regenerating community composition was highly variable but closely related to the severity of *C. nootkatensis* mortality. *Callitropsis nootkatensis* canopy mortality was correlated with winter temperatures and precipitation as well as local soil drainage, with regenerating community composition and *C. nootkatensis* regeneration abundances best explained by available seed source. In areas of high *C. nootkatensis* mortality, *C. nootkatensis* regeneration was low and replaced by *Tsuga*. Our study suggests that climate-induced forest mortality is driving alternate successional pathways in forests where *C. nootkatensis* was once a major component. These pathways are likely to lead to long-term shifts in forest community composition and stand dynamics. Our analysis fills a critical knowledge gap on forest ecosystem response and rearrangement following the climate-driven decline of a single species, providing new insight into stand dynamics in a changing climate. As tree species across the globe are increasingly stressed by climate change-induced alteration of suitable habitat, identifying the autecological factors contributing to successful regeneration, or lack thereof, will provide key insight into forest resilience and persistence on the landscape.

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KEYWORDS
Callitropsis nootkatensis, climate change, community composition, diversity, forest mortality, yellow-cedar decline

1 | INTRODUCTION

Climate change is altering the conditions for tree recruitment, growth, and survival, and range shifts are a widely anticipated consequence of novel temperature and precipitation regimes. Local adaptation to historical climate is already creating mismatches between species' current distributions and suitable habitat conditions (Aitken & Bemmels, 2015; Aitken, Yeaman, Holliday, Wang, & Curtis-McLane, 2008), and these disparities are likely to be most extreme for seedlings, which have a narrower range of tolerance to climate conditions than mature individuals (Niinemets, 2010). As climate regimes shift, habitat suitable for survival of mature trees and the conditions necessary for germination and establishment may no longer correspond with each other, potentially leading to simultaneous canopy mortality and declines in regeneration, and, ultimately, shifts in species distributions (Walck, Hidayati, Dixon, Thompson, & Poschlod, 2011). The key to regeneration success and long-term survival will be continued synchronization of tree germination, establishment, and growth with local climate (Aitken & Bemmels, 2015). In contrast, asynchronization will likely lead to restructuring of forest communities through dieback (Oakes, Hennon, O'Hara, & Dirzo, 2014), regeneration failures (Holz, Wood, Veblen, & Bowman, 2015), or both (Anderegg, Kane, & Anderegg, 2013). Inhibition of conifer regeneration, for example, could lead to ecosystem type conversions (Allen & Breshears, 1998; Holz et al., 2015) and long-term changes in stand dynamics (Turner, Dale, & Everham, 1997). Thus, understanding the relative climatic tolerances of the mature and regeneration life phases of species is critical to predicting their response to climate change.

While the driving processes are difficult to disentangle, species range contractions consist of concurrent or sequential canopy mortality and regeneration failures. Climate change-induced canopy mortality is already a globally documented phenomenon (Allen, Breshears, & McDowell, 2015; Allen et al., 2010). Altered precipitation regimes (i.e., change in timing, amount, frequency, type) combined with simultaneous increases in temperature are leading to both drought mortality (Guarín & Taylor, 2005; Peng et al., 2011) and, at the other extreme, mortality attributed to earlier snowmelt and spring freezing events (Bourque, Cox, Allen, Arp, & Meng, 2005). Drought-induced canopy mortality is well-established (Allen et al., 2015; Anderegg et al., 2013), and recent studies have quantified the impacts of prolonged drought (Redmond, Weisberg, Cobb, & Clifford, 2018) and postfire drought conditions (Stevens-Rumann et al., 2017; Young et al., 2019) on regeneration. However, relatively little is known about the consequences of reduced snowpack and early season frost damage on canopy mortality and regeneration response, although these phenomena may become more common as the climate warms (Woldendorp, Hill, Doran, & Ball, 2008)—leading to root mortality and nutrient loss (Decker, Wang, Waite, & Scherbatskoy, 2003), needle and bud injury (Man, Kayahara, Dang, & Rice, 2009), canopy mortality (Buma, 2018), and seedling mortality (Camarero & Gutiérrez, 2004). Regardless of the climatic stressor, widespread canopy mortality is likely to be ongoing under the more extreme conditions predicted for the future, and under such conditions, range contraction potential will be governed by regeneration success or failure.

Not all climate-driven mortality will result in a range contraction, as regeneration after extensive mortality is dependent upon two conditions: (a) available sources of seed and/or vegetative reproduction and (b) the establishment environment (climatic, abiotic, and biotic). Seed supply generally decreases with a loss of mature trees (Tepley, Veblen, Perry, Stewart, & Naficy, 2016), and this loss can result in reductions in seedling abundances, lower recruitment into mature tree size classes, and the potential for local extirpation due to competition with nondecline-affected species (Oakes et al., 2014). Mortality of mature trees may, conversely, create more favorable establishment conditions by increasing available light and releasing advanced regeneration (Macek et al., 2017; Zeppenfeld et al., 2015) or leading to more successful germination and subsequent survival (Whitmore, 1989). Such increases in favorable microsites may partially offset canopy declines; under such conditions, regeneration of species declining in the canopy may increase despite a reduction in available seed—similar to pulses of recruitment following windthrow events (Dunn, Guntenspergen, & Dorney, 1983; Peterson & Pickett, 1995). Comparing the relative strength of seed source versus community competition and abiotic changes is therefore important in predicting the net effects of mortality and the potential for range shifts at broader scales—thus, resilience.

The effects of canopy mortality on regeneration are, however, difficult to isolate, as climate change-induced canopy mortality often leads to concurrent decline of multiple species. Concurrent declines challenge our ability to identify the factors driving reductions in suitable habitat for mature tree survival versus those leading to regeneration failures for individual species, as a variety of interspecific relationships are changing simultaneously. Simplified systems in which a single species undergoes climate-related mortality in isolation of climate effects on co-occurring species provide a means of parsing out the effects of climate change on mortality, regeneration, and the resultant ecological community.

Across the North Pacific coastal temperate rainforest (NPCTR) of southeast Alaska, USA, and British Columbia, Canada, *Callitropsis nootkatensis*, D. Don, Oesrt. Ex D.P. Little (Alaska yellow-cedar) is experiencing extensive, climate change-driven mortality over more than 400,000 ha and ten degrees of latitude (Buma et al., 2017).

Regional wintertime temperatures average 0°C; thus, a slight warming results in significant snow loss (Buma, 2018). Subfreezing weather events following snowmelt in late winter and early spring kills fine roots of mature *C. nootkatensis* (Hennon et al., 2016; Schaberg, Hennon, D'amore, & Hawley, 2008), and this phenomenon has been ongoing for several decades (Beier, Sink, Hennon, D'Amore, & Juday, 2008; Hennon, D'Amore, Schaberg, Wittwer, & Shanley, 2012). Mortality drivers in this complex pathway include the limited cold tolerance of roots, a reduction in insulating snowpack due to warmer winters, soil freezing due to lack of insulation, and a positive feedback loop in which canopy gaps in declining forests lead to further reductions in springtime snowpack (Beier et al., 2008; Schaberg et al., 2008). Given predicted future climate conditions, *C. nootkatensis* is likely to experience continued decline over most of its range (Buma, 2018) and be replaced by other regionally dominant conifers (Oakes et al., 2014). Co-occurring *Tsuga heterophylla* Raf. (Sarg) (western hemlock) and *Picea sitchensis* Bong. (Carr) (Sitka spruce) are not considered sensitive to snow loss (Buma & Barrett, 2015) and may increase in dominance when *C. nootkatensis* fails to regenerate (Oakes et al., 2014). Yet, despite extensive research on the drivers of decline (Barrett, Latta, Hennon, & Eskelson, 2012; Buma et al., 2017; Hennon et al., 2012; Hennon, Hansen, & Shaw, 1990; Hennon & Shaw, 1997; Hennon, Shaw, & Hansen, 1990; Schaberg et al., 2008), little is known about *C. nootkatensis* regeneration following canopy mortality, and the fate of *C. nootkatensis* and long-term dynamics of affected forests remain unknown.

To address these knowledge gaps and isolate the conditions differentiating habitat suitable for mature tree survival versus those key to successful regeneration in a climate mortality-affected system, we sampled declining *C. nootkatensis* forests over a five-degree latitude range in the NPCTR of southeast Alaska to ask the following: (a) "Does climate-induced mortality occur across all size classes of the affected species, and, specifically, does regeneration response correspond with the same climate conditions driving mortality?", (b) "If response is differential, what climatic and community factors drive tree mortality versus postdecline composition of the regenerating community?", and (c) "Is community composition stable or in the process of a decline-induced shift?". Our analysis provides new insight into stand dynamics in a changing climate by increasing understanding of forest ecosystem response and rearrangement following the decline of a single species. Obtaining information on life stage response to climate-induced mortality, postdecline community composition, and stand dynamics in the NPCTR and beyond will be essential to scientifically based forest management and vital to supporting conservation efforts in the face of climate change.

2 | METHODS

2.1 | Study area and species

Callitropsis nootkatensis is distributed across ~20° of latitude from northern California into Prince William Sound, Alaska (DellaSala et al., 2011). Half of the species' range occurs in the perhumid region of the NPCTR (10° of the 20° latitudinal distribution), where this study occurs, which is characterized by mild, consistently humid conditions and high annual precipitation (3,182 mm average, 621–9,332 range; extracted from ClimateWNA, Wang, Hamann, Spittlehouse, & Carroll, 2016). Across the NPCTR, *C. nootkatensis* co-occurs with *Picea sitchensis*, *Pinus contorta ssp. contorta* Douglas Ex. Louden (shore pine), *Thuja plicata* Donn ex D. Don (western redcedar), *Tsuga heterophylla*, and *Tsuga mertensiana* (Bong.) Carriere. *Sphagnum* spp. are common in areas of poor drainage and low forest productivity, decreasing in abundance with increasing slope and increasing depth to groundwater (Bisbing, Cooper, D'Amore, & Marshall, 2016; Neiland, 1971).

Callitropsis nootkatensis is locally distributed across the NPCTR's hydrologic gradient from emergent wetlands to upland forests, which corresponds to a gradient of low to high forest productivity (Hennon et al., 2012, 2016). This gradient drives the distribution, abundance, and biomass of the region's dominant tree species (Bisbing et al., 2016), with low-lying saturated peatlands limiting the success of most species but providing low-competition environments for stress-tolerant species, such as *P. contorta* (Bisbing et al., 2016) and *C. nootkatensis* (Caouette et al., 2015; Hennon, Hansen, et al., 1990). Background mortality rates in healthy *C. nootkatensis* forests average <25% (Hennon, Hansen, et al., 1990).

Callitropsis nootkatensis regeneration occurs via seed but also through vegetative reproduction. Vegetative layering is particularly common on lower-productivity peatlands with poor drainage and/or high snow cover; lower limbs will produce adventitious roots when depressed by accumulating *Sphagnum* and snow (Hennon et al., 2016). Individuals recruited through vegetative layering often persist on these lower-productivity peatlands despite mature tree mortality.

2.2 | Plot design and sampling

A total of 67 plots were compiled from published (Oakes et al., 2014) and ongoing research by the authors in the perhumid NPCTR subregion of southeast Alaska (Figure 1). This plot network represents all compatible studies in the region, with compatible defined as those including both regeneration and canopy community data, precise locations, comparable methodologies, and comparable scales. Due to aggregating across sources and studies, each with their own goals, plots were not randomly distributed across the region. However, the data products were compatible and collectively allowed for analysis of regeneration response to the canopy decline severity gradient.

Plots installed by Oakes and colleagues (2014, n = 50) were randomly selected from areas stratified by *C. nootkatensis* status—live forests (n = 20) and three time-since-mortality classes (n = 10 each in recent, mid-range, and old) on Chichagof Island and in Glacier Bay National Park (installed in 2011–2012, Figure 1). These plots represent the northernmost extent of the contiguous species distribution while also approaching the northernmost extent of documented decline. Data were collected in nested, fixed-radius plots: (a) 10.3m radius (~333 m^2) for mature trees (>25 cm diameter at breast

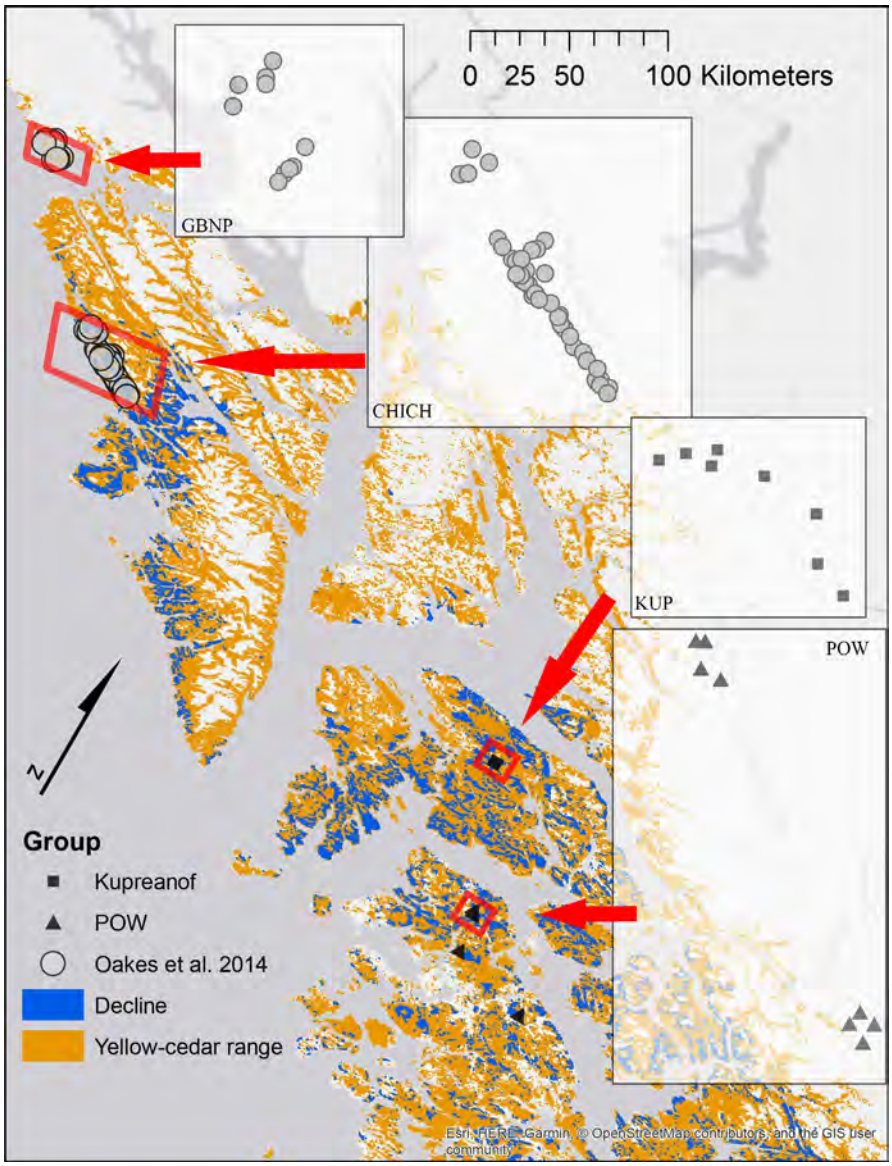

FIGURE 1 Map of *Callitropsis nootkatensis* plot network distributed across southeast Alaska. Previously established plots were installed by Oakes and colleagues in 2011 and 2012 in Glacier Bay National Park (GBNP) and Chichagof Island (CHICH), and additional plots were installed by the authors in 2015 and 2016 on Kupreanof Island (KUP) and Prince of Wales Island (POW). Inset plots for illustration of the numerical distribution of plots only (scale varies). For a complete range and decline map, see Buma et al. (2017) or Buma (2018)

height, DBH) and (b) 6m radius (~113 m^2) for saplings (<2.5 cm DBH and >1 m in height), treelets (2.5–9.9 cm DBH), and small trees (10–24.9 cm DBH, see Oakes et al., 2014 for details). In addition, conifer germinants and seedlings (<10 cm height) were sampled using eight one-m^2 quadrats (8m^2 total area sampled) installed in the cardinal directions at five and eight meters from plot center.

To extend inference across *C. nootkatensis*' distribution in southeast Alaska, we leveraged 17 additional nested, fixed-radius plots on Kupreanof and Prince of Wales Islands (installed 2015–2016, Figure 1). Mature and small trees (>10 cm DBH) were sampled over a 400-m^2 plot. Treelets, saplings, and advanced regeneration (>10 cm but <1.37m in height) were sampled in a nested 100-m^2 plot. Germinants and seedlings (<10 cm height) were tallied by species in four one-m^2 quadrats at the corners of the 400-m^2 plot (16 m^2 total area sampled). Hereafter, the term regeneration refers to the combined germinant and seedling classes, including all recently emerged germinants (i.e., cotyledons still visible) to seedlings less than 10cm in height and, presumably, <2 years of age.

Across all plots, species, DBH, tree status (live/dead), live crown condition, and snag class were sampled for all trees. Snags were identified to species, if possible, and *C. nootkatensis* snags classified into time-since-mortality classes according to Hennon, Shaw, et al. (1990). Unidentifiable snags were classified as unknown species. Additionally, regeneration was identified to the genus for *Tsuga* germinants (*heterophylla* and *mertensiana*) but to the species for all other species and size classes. Disturbed areas (e.g., windthrow, landslides) were eliminated from plot selection to avoid the confounding influence of disturbance on community composition.

2.3 | Environmental variables

Climate data were extracted from ClimateWNA v5.51 (http://www.climatewna.com, Wang et al., 2016), which provides spatially interpolated, locally downscaled climate data and derived biologically relevant climate variables. Data were obtained for three date ranges to capture drivers of ongoing decline and regeneration—the last few

decades (30-year normal, 1981–2010), the five-year period prior to and including sampling year (five-year average, range 2007–2016), and the year of sampling (annual, range: 2011–2016). Annual data were used to evaluate the influence of current year conditions on the regenerating community, while data from the five-year period were used to quantify the influence of multiyear climate on canopy mortality, potential seed availability and viability, and regenerating community composition. A five-year window was selected for two reasons: (a) *C. nootkatensis* cones take two to three years to mature and produce viable seed (Bonner & Karrfalt, 2008), and (b) we focus regeneration analysis on individuals presumed to be up to two years of age.

Prior to analysis, we ran a correlation analysis to reduce the full set of ClimateWNA variables to a minimally correlated set (<0.65) while retaining those considered biologically important in *C. nootkatensis* decline, including winter and spring temperatures and precipitation, length of the growing season, and frost period (Buma et al., 2017; Hennon et al., 2012). The following variables were selected as potential predictors: winter (December–February) and spring (March–May) minimum and maximum temperatures (°C); winter, spring, summer (June–August), and average annual precipitation (mm); number of frost-free days in winter and spring (days); average annual length of growing season (frost-free days), date of first frost (Julian date); and annual precipitation as snow (mm; Table 1).

Local topographic data were obtained from the NASA ASTER mission (30m resolution, LP DAAC 2017) and used to derive elevation, slope, and aspect. To assess the role of local soil drainage, which influences competition and rooting, we selected two metrics operating at different scales. At the local scale, we used *Sphagnum* coverage. To do so, we sampled bryophyte (n = 67) and *Sphagnum* (subset, n = 12) cover on replicate 1-m^2 quadrats (8–10 per plot). Bryophyte coverage was correlated with *Sphagnum*-specific coverage, so, on plots with bryophyte coverage only, a log-linear model was created to estimate *Sphagnum* coverage (R^2 = 0.48, F = 15.81 on 1,15 df, p < 0.001). Predicted *Sphagnum* coverages were used in overall model creation. At the landscape scale, we used contributing area derived from the ASTER elevation data, a metric of upslope area potentially contributing runoff to a location in which higher values

TABLE 1 Potential predictor variables included in regression tree modeling of *Callitropsis nootkatensis* (CANO) mortality and regeneration abundance following decline

Factors	Variables	Unit
Location	Group	Study area
Time	Time since onset of mortality	Years
Competition	CANO, PICO, PISI, THPL, TSHE, TSME	Live basal area in m^2/ha
	CANO, PICO, PISI, THPL, TSHE, TSME, Unknown, and Unknown *Tsuga*	Dead basal area in m^2/ha
	Total live basal area	m^2/ha
	Total dead basal area	m^2/ha
	Sphagnum coverage	Percent
Climate (annual, 5-year period, 30-year normal from 1981 to 2010)	Maximum and minimum temperatures: winter, spring	C
	Precipitation: winter, spring, summer, annual	mm
	Length of growing season	Days
	Frost-free days: winter, spring	Days
	Date of first frost	Julian Date
	Precipitation as snow	mm
Topographic	Elevation	Meters
	Aspect	Degrees
	Slope	Degrees
	Contributing area	Log_{10} (m^2)
	Wind exposure (Buma & Barrett, 2015)	1–8
	Landslide likelihood (Buma & Johnson, 2015)	0–1

Abbreviations: PICO, *Pinus contorta* spp. *contorta*; PISI, *Picea sitchensis*; THPL, *Thuja plicata*; TSHE, *Tsuga heterophylla*; and TSME, *Tsuga mertensiana*.

indicate wetter, lower-lying areas. All data processing and subsequent analyses were conducted in R version 3.4.1. (R Core Team, 2018)

2.4 | Data analyses

To control for the possibility of general forest declines, which would affect all species and obscure *C. nootkatensis*-specific mortality, we used simple linear regressions to assess the correlation between the proportion of dead *C. nootkatensis* (log-transformed) as compared to that of the other dominant co-occurring tree species, and Kruskal–Wallis tests for nonparametric, ranked data to assess variation in proportional mortality among species, locations, and size classes (as defined above). There were no significant concurrent declines in any co-occurring species ($p > 0.05$), so we focused subsequent mortality analyses on *C. nootkatensis* alone. We also used a Kruskal–Wallis analysis to test for significant differences in live tree abundances among species and locations. We then tested for significant differences in canopy and regenerating community composition among locations and as related to *C. nootkatensis* mortality with multivariate analysis of variance (MANOVA) tests using Bray–Curtis distances in the *vegan* package. The proportion of dead *C. nootkatensis* in the canopy (dead *C. nootkatensis* out of total *C. nootkatensis*) was also compared to regeneration density using a negative log-transformed linear regression. In Kruskal–Wallis and MANOVA tests, proportional mortality was categorized into the following mortality severity classes: low = 1%–25%, moderate = 25.1%–69.9%, high = 70%–99%, and all = 100%.

Next, we compared drivers of *C. nootkatensis* canopy mortality versus regeneration. Two random forests analyses were run—one to model the proportion of dead *C. nootkatensis* in the canopy and another the abundance of *C. nootkatensis* regeneration. Both were based on potential drivers related to climate, disturbance exposure, topography, drainage, and competition (Table 1). Random forests, an extension of regression tree analysis (Breiman, 2001), are well suited for complex, nonlinear interactions between variables and generally perform better than other methods in predictive accuracy (Prasad, Iverson, & Liaw, 2006).

We took a two-step process similar to importance-based variable selection procedures (Evans & Cushman, 2009). First, an initial forest was grown using all potential variables, and variable importance was calculated, based on the decrease in accuracy on the out-of-bag sample (independent data points not used in building the tree used for testing) when each variable is permuted compared to the original tree. This is averaged across all trees in the forest. Variables were then assessed for cross correlation with each other. The top ten important uncorrelated variables (<0.65) were retained. This was necessary as, while random forests are not generally subject to overfitting due to correlated variables, correlation between variables often means several highly correlated variables may all be simultaneously considered of high importance. While this may not be an impediment to modeling accuracy (Fox et al., 2017), it interferes with our ability to interpret the random forest outputs versus our hypotheses. Instead, the top uncorrelated variables were used to create a second, final model. Our model was then investigated for the marginal influence of the most important variables on proportion dead and regeneration density by running the final model while varying the single variable of interest and plotting projected values (sometimes called a partial plot). We used the *randomForest* package for analyses.

To determine regenerating community types and identify potential shifts in community composition, we analyzed conifer regeneration abundances across all sampling locations using a cluster analysis with the Bray–Curtis distance measure and Ward's hierarchical agglomerative method in the *vegan* package. Community types were determined with an indicator species analysis within the *indicspecies* package; the appropriate number of community types was classified by maximizing the number of statistically significant indicator species in each group (Dufrêne & Legendre, 1997). We assessed differences in community composition and cluster types with Kruskal–Wallis tests for nonparametric, ranked data, including location, time since mortality, and proportional mortality severity class as potential predictors.

We then used a suite of nonparametric, multivariate analyses to compare patterns in and identify drivers of regenerating community composition. First, we performed nonmetric multidimensional scaling (NMS) ordinations on regeneration abundances based on Bray–Curtis dissimilarity with the *vegan* package (Oksanen et al., 2007). Nonmetric multidimensional scaling avoids the assumption of linear or unimodal responses so is well suited to non-normal plant community data (McCune, Grace, & Urban, 2002). We employed permutational vector fitting (999 permutations) on biologically significant yet minimally correlated variables (<0.65, detailed above, Table 1) using a multiple linear regression technique with the *envfit* function to assess relationships between NMS ordinations of community structure and this reduced set of climate, community, and stand-level variables. Variables identified as significant using *envfit* were then evaluated with a generalized additive model to test for linear fit, and variables representing nonlinear relationships were removed from the final model.

3 | RESULTS

3.1 | Community composition, canopy mortality, and regeneration

Tree diversity is generally low in the region (Caouette et al., 2015; Neiland, 1971), and despite the latitudinal range, all plots were similar in species composition. The canopy was a mixed-conifer forest of *Tsuga (heterophylla* and/or *mertensiana*), *P. sitchensis*, and *C. nootkatensis*. *Picea sitchensis*, although common in the region, was rare on the plot network, with only one tree documented on many of the plots. *Pinus contorta* occurred in peatlands on Chichagof Island, while *T. plicata* was found only at higher elevations on Prince of Wales Island (other plots fell outside *T. plicata*'s range). Total basal area ranged from 9 to 87 m^2/ha, and higher *C. nootkatensis* basal areas were generally found on higher productivity, upland forests or in areas north of the decline (Table A1). *Callitropsis nootkatensis* regeneration abundance ranged from 0 to 9.6 per m^2 (mean 1.9/m^2, median = 1/m^2; Table A1).

Mortality was documented in each of the most common conifers on the plot network (*C. nootkatensis*, *P. sitchensis*, *T. heterophylla*, *T. mertensiana;* Figure 2) and across all size classes, but, with the exception of *C. nootkatensis*, proportions were in line with or lower than expectations of snag abundances for the region (Deal, Oliver, & Bormann, 1991; Hennon, Hansen, et al., 1990; Hennon & McClellan, 2003). Proportional mortality in mature *P. sitchensis* was high on Kupreanof Island (Figure 2), but this was driven by a lack of trees on the plot network (two dead of three total trees on eight plots). Mortality of less common *P. contorta* occurred only on Chichagof Island (mean = 30% ± 3% for mature and small trees), and no mortality was documented for *T. plicata*.

Callitropsis nootkatensis mortality did occur across all tree size classes (mature, small, and treelet; Figure 2), and estimated time since *C. nootkatensis* mortality ranged from 0 (in healthy stands) to 75 years (Table A1). Mortality was not documented in sapling and seedling size classes, likely due to the short-lived nature of this fine material. The proportion of dead *C. nootkatensis* averaged 74% (median = 75%, range = 4%–100%) across the extent of decline, while background mortality in healthy stands north of the decline averaged 16% (median = 12%, range = 0.01%–43%). Mortality of common co-occurring conifers was significantly lower than that of *C. nootkatensis* (Kruskal–Wallis chi-squared = 160.45, $df = 3$, $p < 0.001$). Canopy mortality was also significantly different among co-occurring species (Kruskal–Wallis chi-squared = 33.79, $df = 3$, $p < 0.001$); however, within-species mortality was not significantly different between size classes (NS within species, Kruskal–Wallis chi-squared (species pooled) = 2.77, $df = 2$, $p > 0.05$) or locations (NS within species, Kruskal–Wallis chi-squared (species pooled) = 6.81, $df = 3$, $p > 0.05$).

Species abundances were significantly different within both the canopy and regenerating communities. Live tree abundances varied by species (Kruskal–Wallis chi-squared = 43.17, $df = 5$, $p < 0.001$), but significance was driven by variation in live *C. nootkatensis* among sampling locations (Kruskal–Wallis chi-squared = 9.83, $df = 4$, $p < 0.05$). Live, mature *C. nootkatensis* continued to dominate the canopy in Glacier Bay National Park (mean dead = 8%), an area currently north of the region of decline, but was a minor component (>50% dead across all plots, mean = 75%) at the southern end of decline on Prince of Wales Island (Figure 3a). Canopy community composition (Figure 3a) varied significantly by location (MANOVA $R^2 = 0.13$, $df = 3$, $p < 0.001$) and as a function of the severity of *C. nootkatensis* mortality (MANOVA $R^2 = 0.29$, $df = 3$, $p < 0.001$). Regenerating community composition (Figure 3b) was highly variable among locations (MANOVA $R^2 = 0.29$, $df = 3$, $p < 0.001$) but closely related to canopy condition (Figure 3a) and the severity of *C. nootkatensis* mortality (MANOVA $R^2 = 0.09$, $df = 3$, $p < 0.05$). Areas of high severity *C. nootkatensis* mortality had the lowest abundances of *C. nootkatensis* regeneration (Kruskal–Wallis chi-squared = 23.02, $df = 3$, $p < 0.001$, Figure 3a, b). *Tsuga* species dominated the regenerating community across all locations and were the principal regenerating species in areas of high decline (Figure 3b).

3.2 | Climate and community drivers of mortality and regeneration

In the random forests model, higher *C. nootkatensis* mortality was correlated with cool winter temperatures and lower winter precipitation as well as two metrics of soil drainage—moderate *Sphagnum* percentage and higher slopes (mean squared residuals = 0.06, variance explained = 0.40; Figure 4). As a result, the highest proportional mortality on the plot network was found in cooler, relatively drier regions and on higher productivity upland forests.

Callitropsis nootkatensis regeneration abundances were strongly correlated with stand condition and canopy composition (variance

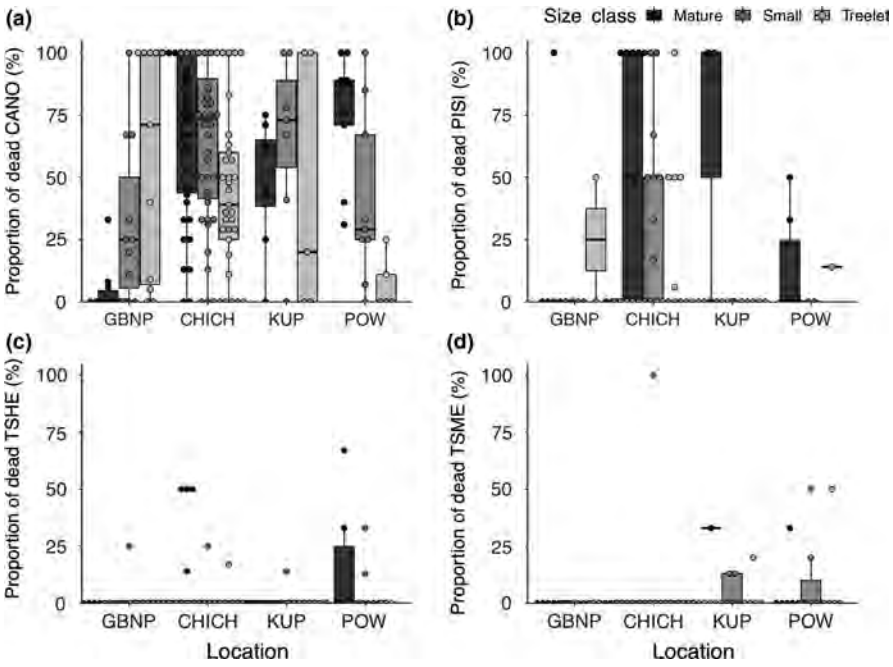

FIGURE 2 Proportional mortality (dots = plot-level data, boxplot = median and range) of common canopy species of southeast Alaska: (a) *Callitropsis nootkatensis* (CANO), (b) *Picea sitchensis* (PISI), (c) *Tsuga heterophylla* (TSHE), and (d) *Tsuga mertensiana* (TSME). Proportional mortality was significantly different among species ($p < 0.001$) but not among size classes within a species ($p > 0.05$). CANO mortality was not significantly different among locations ($p > 0.05$). CHICH, Chichagof Island; GBNP, Glacier Bay National Park; KUP, Kupreanof Island; and POW, Prince of Wales Island. Note that for KUP PISI, proportional mortality is pulled from only three trees across all sampled plots, so it is not representative of regional PISI mortality

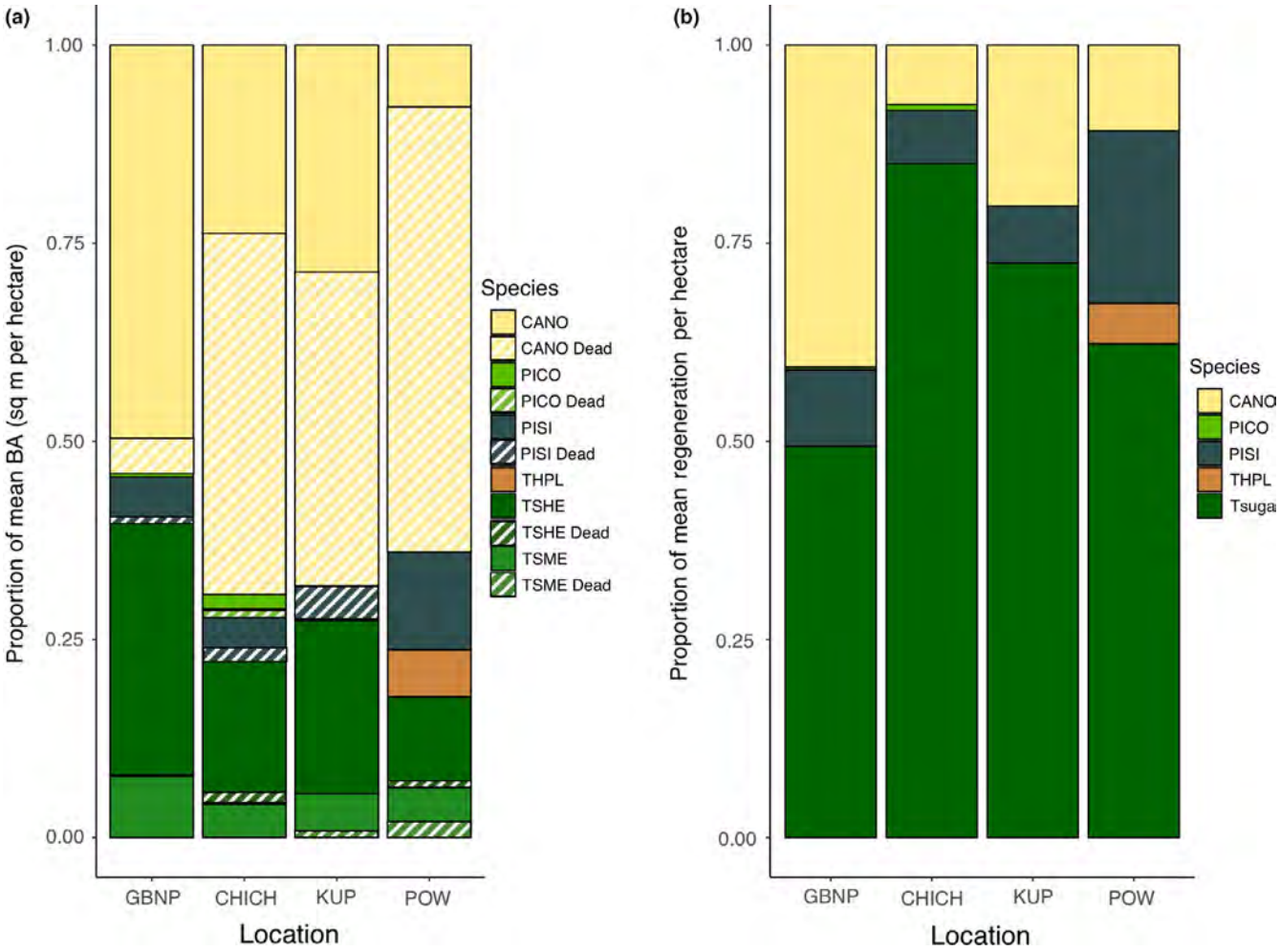

FIGURE 3 (a) Proportion of average canopy (mature trees, small trees, treelets, and saplings) live and dead basal area per hectare by species and location across southeast *Callitropsis nootkatensis* plot network. (b) Proportion of average regeneration densities (germinants and seedlings per hectare) by species and location across southeast *C. nootkatensis* plot network. CANO, *Callitropsis nootkatensis*; CHICH, Chichagof Island; GBNP, Glacier Bay National Park; KUP, Kupreanof Island; PICO, *Pinus contorta*; PISI, *Picea sitchensis*; POW, Prince of Wales Island; THPL, *Thuja plicata*; *Tsuga*, hemlock species. Community composition was significantly different among locations ($p < 0.001$) for the canopy and regenerating communities and across the severity of *C. nootkatensis* mortality ($p < 0.05$) for the regenerating community

explained = 0.45; mean squared residuals = 2.02). The top four variables in the model were basal area of dead and live *C. nootkatensis*, total live stand basal area (all species), and contributing area (Figure 5). *Callitropsis nootkatensis* basal area was the most important variable in explaining regeneration abundance, with higher regeneration in areas with a greater proportion of live *C. nootkatensis* in the canopy and a lower proportion of dead *C. nootkatensis* (Figure 5). This was corroborated by the direct comparison between canopy mortality and regeneration; regeneration (log-transformed) was negatively correlated with increasing mortality ($p < 0.001$, $r^2 = 0.38$, $F = 40.64$ on 1,65 *df*; Figure 6).

3.3 | Response and stability in regenerating community composition

Four regenerating community types were identified in the cluster and indicator species analysis. Community types were largely determined by extent of *C. nootkatensis* decline, with communities assigned to mixed-conifer (Mixed), *C. nootkatensis* (CANO), *Tsuga-P. sitchensis* (Tsuga-PISI), or *P.contorta* (PICO) clusters (Figure 7). The Mixed type was highly variable in location, climate, and drainage condition (no significant indicator species scores (ISS), $p > 0.05$; Figure 7). The CANO type occurred in low slope, lower-productivity plots in areas of low *C. nootkatensis* mortality (ISS = 70, $p < 0.01$). The Tsuga-PISI type was found on steeper slope, higher productivity plots and consisted of *Tsuga* (ISS = 81, $p < 0.01$) and *P. sitchensis* (ISS = 72, $p < 0.05$) in areas of moderate to complete *C. nootkatensis* mortality. The PICO type was assigned to low slope, lower-productivity peatlands with high bryophyte coverage (ISS = 60, $p < 0.05$). Plots of similar condition generally clustered into the same community types; plots with high to complete *C. nootkatensis* mortality were regenerating to Tsuga-PISI or PICO, with *C. nootkatensis* regeneration dominating in low mortality, lower-productivity peatlands. Location (Kruskal–Wallis chi-squared = 19.07, *df* = 3,

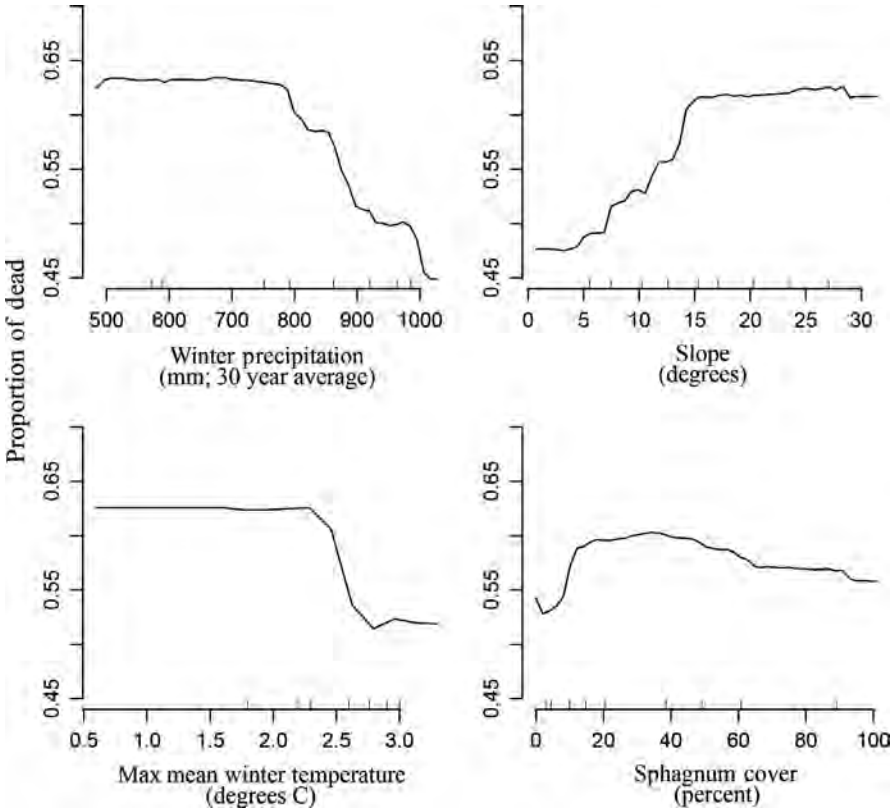

FIGURE 4 Change in predicted percent of mortality (modeled as proportion of dead *Callitropsis nootkatensis*) on the plot network as driven by the top four most significant uncorrelated variables in the final model. Y-axis is the modeled mortality percentage as a function of the overall random forest model while varying the top for variables, respectively. For example, mortality generally declines in areas of higher winter precipitation and lower slopes

$p < 0.001$), severity of *C. nootkatensis* mortality (Kruskal–Wallis chi-squared = 19.94, $df = 3$, $p < 0.001$), and time since mortality (Kruskal–Wallis chi-squared = 15.78, $df = 3$, $p < 0.01$) were significant predictors of regenerating community types, indicating that climate and both the timing and the extent of decline are primary determinants of community composition stability.

In areas of high *C. nootkatensis* decline, *C. nootkatensis* regeneration was low and replaced by *Tsuga* spp. (Figure 7), and community

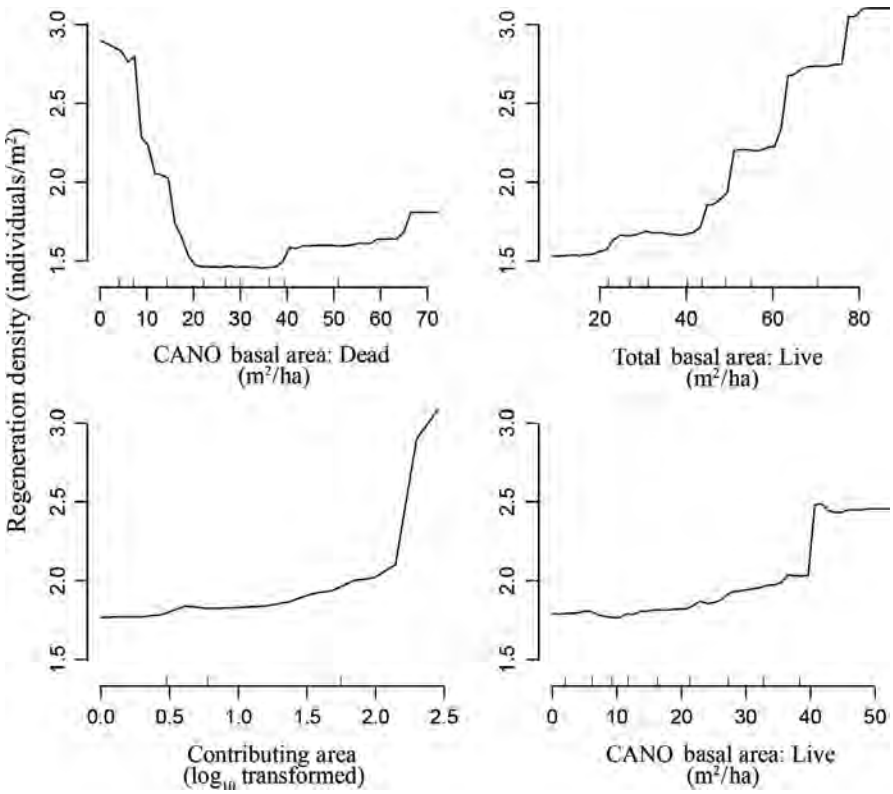

FIGURE 5 Change in predicted *Callitropsis nootkatensis* regeneration densities on the plot network as driven by the top four most significant uncorrelated variables in the final model. There is a clear relationship between forest health, seed source, and regeneration, with higher densities seen in areas of lower *C. nootkatensis* mortality, higher live *C. nootkatensis* (presumably seed source), and higher overall basal area. Higher densities are also found in wetter landscapes, with a higher contributing area

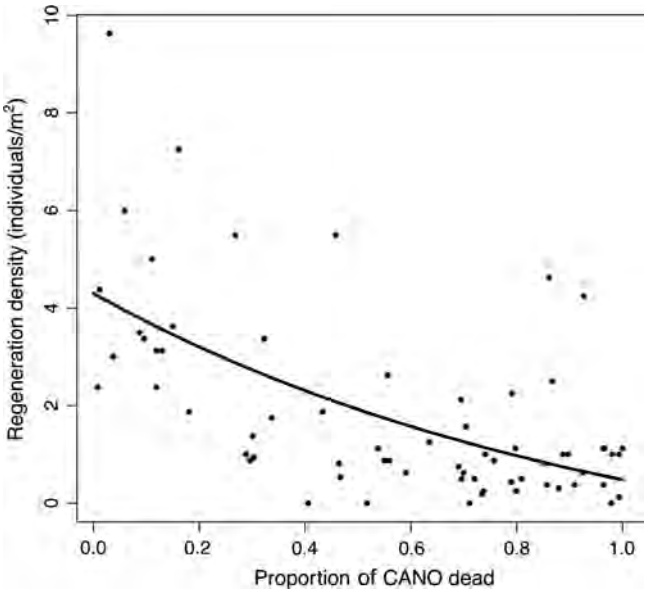

FIGURE 6 *Callitropsis nootkatensis* (CANO) regeneration densities (individuals per square meter) per plot (black, filled circles) as a function of the proportion of dead *C.nootkatensis* in the canopy. Regeneration densities decline with increasing severity of canopy mortality ($p < 0.001$)

composition was best explained by the following climatic variables in NMS analysis and vector fitting: maximum winter temperature ($R^2 = 0.30$, $p < 0.001$) and five-year average precipitation as snow ($R^2 = 0.22$, $p < 0.001$; Figure 7). Canopy community composition and stand characteristics additionally explained regenerating community composition; *P. contorta* ($R^2 = 0.27$, $p < 0.001$) and *P. sitchensis* ($R^2 = 0.12$, $p < 0.05$) live basal area plus slope ($R^2 = 0.10$, $p < 0.05$) drove community clustering (Figure 7). Variation in community composition was best explained by two axes (final stress = 0.10, nonmetric $R^2 = 0.99$, linear $R^2 = 0.96$). Axis 1 relates to a gradient of canopy mortality and local hydrologic condition (and associated ecosystem productivity), with *C. nootkatensis* successfully regenerating on low slope peatlands and *Tsuga* species dominating well-drained upland forests (Figure 7). Axis 2 divides forest ecosystem types by productivity (*P. contorta*-dominated peatlands from *Tsuga*–*P. sitchensis* upland forests).

4 | DISCUSSION

As the climate changes, successful tree regeneration is threatened by novel climate and disturbance conditions and associated shifts in canopy community composition. Our study highlights how species-specific sensitivity to climate change can lead to restructuring of the forest community following canopy mortality. We demonstrate that reductions in both mature tree and regeneration abundances after single-species mortality events occur via the same mechanism—death of mature, seed-producing trees. In the case of *C. nootkatensis*, our work suggests that climate-induced forest mortality is driving alternate successional pathways in forests where *C. nootkatensis* was once a major component, which is likely to lead to long-term shifts in community composition and stand dynamics. As suitable habitat conditions shift with ongoing global climate change, both mature trees and regeneration will be increasingly exposed to novel conditions, and identifying the autecological factors contributing to successful regeneration, or lack thereof, is an essential first step toward predicting forest response and resilience to climate change.

4.1 | Climate impacts on mortality and regeneration in Callitropsis nootkatensis

In the NPCTR, climate change-induced canopy mortality is restricted to a single species, *C. nootkatensis*, allowing for a focused examination of the effects of climate, seed source, and local conditions on regenerating community response and subsequent stand dynamics. Mature tree mortality across the study area corresponded with a reduction in *C. nootkatensis* regeneration (Figure 6) and a shift to surviving canopy species, predominantly *Tsuga* (Figure 3), a widespread species known to dominate the regenerating community following disturbance (Alaback and Tappeiner II (1991); Deal & Farr, 1994). *Callitropsis nootkatensis* regeneration constituted less than 20% of the regenerating community in decline areas (Figure 3) and appears to be limited by a lack of available seed and vegetative source, although drainage conditions, as determined by slope, and maximum winter temperatures also contribute to structuring regenerating community composition (Figure 7).

Local hydrologic regime and associated ecosystem productivity are key factors in both species' distributions (Bisbing et al., 2016) and in the extent of *C. nootkatensis* decline (D'Amore & Hennon, 2006). Mortality across this plot network was most extensive on wetter, lower-productivity peatlands; however, proportional mortality was greatest in productive upland forests. The apparent lack of congruity between these results and previous research, which reports higher mortality on lower slopes (D'Amore & Hennon, 2006), is explained by our use of proportional mortality rather than total. Although mortality is most extreme on saturated peatlands, it is likely that surviving individuals will allow for ongoing perpetuation of the species on lower-productivity peatlands.

Nonetheless, the loss of *C. nootkatensis* from both the canopy and the regenerating community across the extent of decline in southeast Alaska indicates that canopy trees and regeneration are in sync in their responses to climate change. Cold, low-snow winters have led to widespread mortality over our study area (Figure 4) and across the species' range (Buma et al., 2017), and local drainage conditions (i.e., slope, hydrologic regime) amplify or mitigate response of the canopy (Figure 4) and regenerating community (Figures 5 and 7). While this study cannot conclusively determine the mechanism for the lack of regeneration, the close correspondence between the canopy and regenerating communities is striking. These plot-level findings are consistent

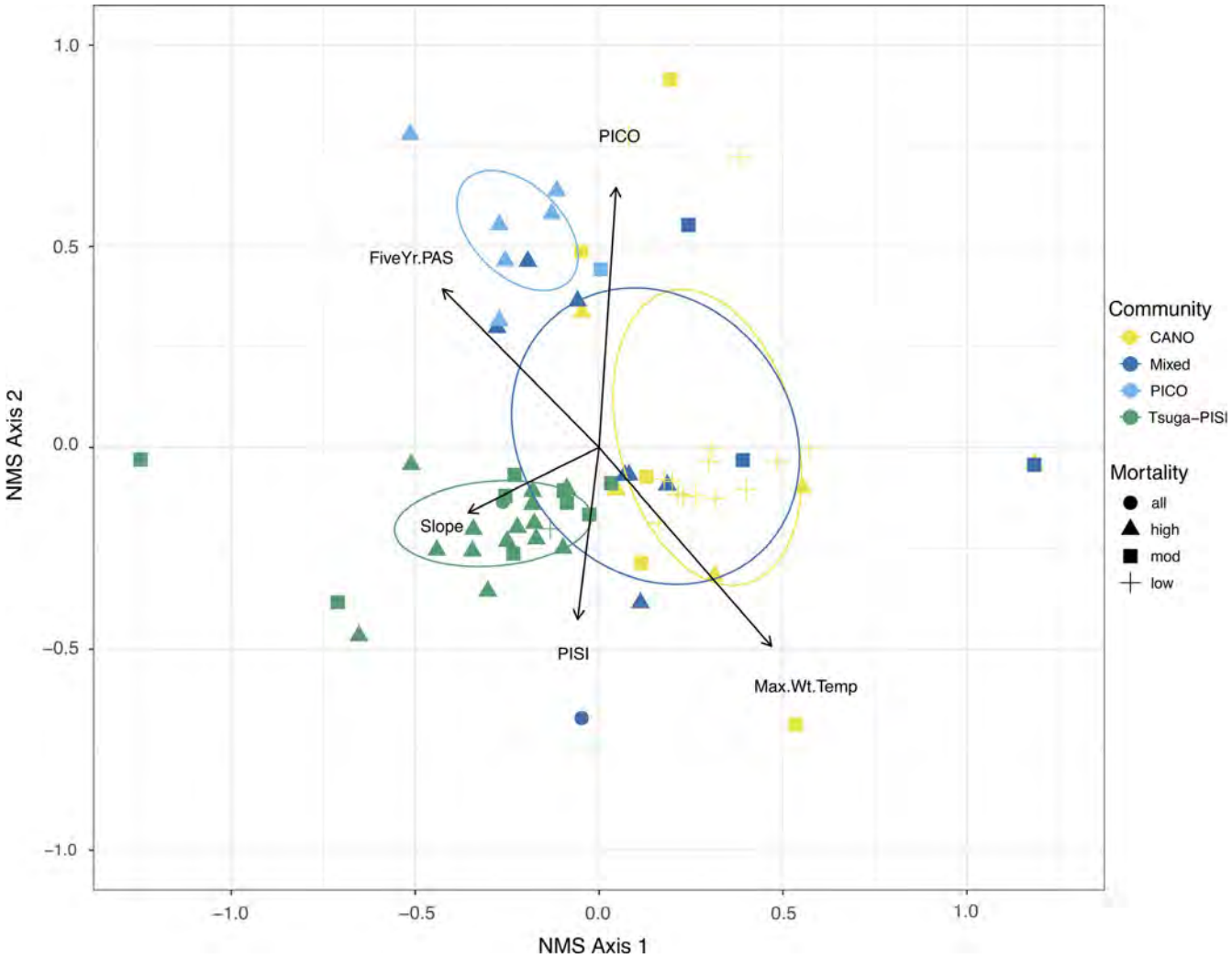

FIGURE 7 Nonmetric multidimensional scaling (NMS) of the regenerating community by classification of the severity of *Callitropsis nootkatensis* canopy mortality (shapes) and community type (colors), with each point representing individual sample plots. Mortality severity classes: low = 1%–25%, moderate = 25.1%–69.9%, high = 70%–99%, and all = 100%. Ellipses represent the mean scores of each NMS community cluster. Species labels represent the most abundant species in each NMS community cluster. CANO, *Callitropsis nootkatensis*; PICO, *Pinus contorta* spp. *contorta*; PISI, *Picea sitchensis*; THPL, *Thuja plicata*; and Tsuga, *Tsuga heterophylla* and *mertensiana*. Variables and associated vector arrows indicate the direction and magnitude of effects of the canopy community, local stand conditions, and local annual and five-year climate on regenerating community composition. Arrow length is proportional to the magnitude of correlation. Variables identified as both significant ($p < 0.05$) and minimally correlated include Max.Wt.Temp = maximum winter temperature (°C), Five.Yr.PAS, five-year average precipitation as snow (mm), average plot slope (°), PICO, *Pinus contorta* spp. *contorta* basal area; and PISI, *Picea sitchensis* basal area

with recent modeling work on drivers of canopy mortality that identified winter temperatures and slope as conditions leading to decline at broad scales (Buma et al., 2017). Mortality is predicted to be ongoing in areas above the winter snow threshold, and up to 50% of current suitable habitat is expected to experience climate conditions favorable to decline (Buma et al., 2017). Continued mortality of this conifer is likely to lead to more widespread shifts in community composition (Oakes et al., 2014; data presented here). The close relationship between seed source and composition of the regenerating community suggests that no compensation in the form of increased recruitment will make up for the loss of the *C. nootkatensis* canopy.

4.2 | *Single* versus *multiple species decline*

Mortality events specifically attributed to climate change (Breshears et al., 2005; Van Mantgem & Stephenson, 2007; Williams et al., 2010) do not typically discriminate but, instead, lead to decline or mortality of multiple species in the affected ecosystem. Sierra Nevada mixed-conifer forests have, for example, seen concurrent declines in shade tolerant *Abies* and shade-intolerant *Pinus* following years of temperature-driven drought stress (Van Mantgem & Stephenson, 2007), while extreme droughts in the arid southwest led to mortality in *Pinus edulis* and *Juniperus monosperma* (Mueller et al., 2005). In both cases, no immediate impacts on regenerating community

composition were evident, although higher *P. edulis* mortality indicated that long-term shifts are likely (Mueller et al., 2005). The climate change-induced *C. nootkatensis* mortality in the NPCTR is unique in its isolated effect on a single species but also in its observable, and now documented, postdecline shift in species dominance.

Regenerating community response in other single-species mortality events has been highly variable and largely driven by the severity of a biotic disturbance, such as insect attack (Burr & McCullough, 2014), on the canopy community and variability in local establishment conditions (Kayes & Tinker, 2012). In some cases, regeneration proceeds successfully following the loss of canopy species (Diskin, Rocca, Nelson, Aoki, & Romme, 2011; Macek et al., 2017), while, in others, species experience loss of dominance (Pelz & Smith, 2012), increasing the probability of ecosystem type conversions (Burr & McCullough, 2014; Klooster et al., 2014). Self-replacement of a single species following canopy morality has been documented following extreme drought (Suarez & Lloret, 2018) and bark beetle attack (Diskin et al., 2011), where co-occurring species established but failed to dominate. These instances of ongoing success, however, occurred in forest types where the impacted species dominated the canopy prior to the disturbance and either did not experience wholesale mortality, as is the case in *Nothofagus* drought mortality (Suarez & Lloret, 2018), or possessed a canopy seed bank, as occurs with *Pinus contorta* ssp. *latifolia* forests (Diskin et al., 2011).

Regeneration declines or failures have also been documented in cases of single-species mortality (Burr & McCullough, 2014; DeRose & Long, 2007; Klooster et al., 2014; Pelz & Smith, 2012). Laminated root rot in the Northwest has led to similar compositional changes as seen in *C. nootkatensis* forests, where *T. heterophylla* assumes dominance as canopy mortality of *Pseudotsuga menziesii* proceeds (Hansen & Goheen, 2000). Additionally, the 1990s *Dendroctonus rufipennis* attack on *Picea engelmannii* led to *Abies lasiocarpa* and *Populus tremuloides* dominance in the regenerating community, precluding *P. engelmannii* from returning to dominance (DeRose & Long, 2007). As with *C. nootkatensis* response to canopy mortality, the primary limiting factor in all cases is an available seed source.

Regenerating community response to canopy mortality is clearly influenced by a number of factors, such as establishment environment conditions, postdisturbance climate, seed availability, and herbivory—but available seed source is regularly identified as a primary limiting factor, regardless of disturbance type (e.g., fire vs. beetle attack) and magnitude (e.g., single vs. multiple species). Consistent with studies in other forest types and under different climate-induced disturbances, our findings suggest that seed source (DeRose & Long, 2007; Redmond et al., 2018; Urza & Sibold, 2017) and microsite/establishment environments (Harvey, Donato, & Turner, 2016; Redmond et al., 2018; Urza & Sibold, 2017) are the most important factors in regenerating community response. Species tolerant of postdecline conditions will have a higher likelihood of future dominance, further reducing available seed and vegetative source of the declining species.

Documenting changes in communities following canopy mortality will be crucial to identifying the ecological, and potentially economic, consequences of these losses. Identifying successional trajectories, in particular, will reduce the large uncertainty around the long-term impacts of climate change-driven forest mortality events. Forest response to novel disturbance conditions will be hard to predict, and the lessons learned from these cases can help shift our expectations of postdisturbance stand dynamics, particularly in climate-impacted systems.

4.3 | Ecosystem resilience and transition to alternate stable states

The consequences of rapid climate change on forest resilience remain uncertain and are likely to be highly variable, based on a particular forest's ecological memory (i.e., information legacies of ecological adaptations to disturbance; Johnstone et al., 2016). For forests to be resilient and resist transitions to alternate states, there must be synchrony between both information legacies (i.e., genetic adaptations to disturbance, like sprouting in avalanche-prone ecosystems) and material legacies (i.e., the physical legacies, such as seedbanks, that are present after a disturbance event; Johnstone et al., 2016). Historically, *C. nootkatensis'* shallow root system and early response to spring warming were beneficial, allowing for early, rapid growth each growing season (Hennon et al., 2016); however, this once advantageous legacy is now a deleterious adaptation. Mortality in mature trees limits the potential for biological inertia (Young et al., 2019) due to a lack of available seed. Instead, advanced regeneration via vegetative reproduction remains a successful strategy even in areas of severe decline (unpublished data/personal observation). This may help the species maintain a presence on the landscape during periods of mature tree mortality until periods more favorable for sexual reproduction, similar to the "orphaned cohort" example of *Fraxinus* in response to *Agrilus planipennis* invasion (Klooster et al., 2014). The immediate loss of seed-producing trees and seedlings (Figures 2 and 3), as well as competition from faster growing species, means these forests are lacking the biological inertia needed for resilience and are likely to transition to *T. heterophylla*-dominated forests even if *C. nootkatensis* maintains some temporary or disjunct presence on the landscape. For species or forest types undergoing similar climate-induced mortality events, ecologists will need to determine which legacies contribute to ecosystem resilience, or if novel climate or disturbance will remove these legacies that were historically critical.

Ecosystem resilience may also vary as a function of regional climate fluctuations over space and time, known as transitional climate mortality (Buma, 2018), where mortality is highest within a particular range of climatic conditions but decreases above or below this range. Currently, the mid-range of the *C. nootkatensis'* distribution is experiencing mortality, but, if emissions scenarios continue toward worst-case trajectories, it is possible that the mortality "donut-hole" will be relatively short-lived (Buma, 2018). Short, intense climate fluctuations may therefore be less deleterious to ecosystem resilience if there are deep enough ecological legacies to sustain species over time.

5 | CONCLUSIONS AND NEXT STEPS

Widespread mortality of tree species due to changing climate is a major concern in forests worldwide, but the potential for resilience is rarely assessed during an ongoing mortality event, a gap we have attempted to fill here. Species' responses to climate change-induced mortality will vary widely based on species-specific traits, sensitivity to climatic extremes, biotic stressors, and abiotic conditions of the establishment environment, thus requiring autecological studies on factors limiting versus promoting success. In *C. nootkatensis* forests, there is no increase in *C. nootkatensis* regeneration abundances to offset canopy mortality. As a result, this forest type is not resilient to mortality associated with ongoing snow loss, and a type change appears to be underway. This example of climate change-driven mortality in a single species highlights how species-specific sensitivity can lead to shifts in community composition and stand dynamics following canopy mortality via the same mechanism—death of mature seed trees.

Few strategies or solutions exist for forests vulnerable to ecosystem type conversion due to mature tree mortality and associated loss of seed source, and numerous knowledge gaps remain. In the case of *C. nootkatensis*, common garden studies using seed sources from across the species' range could allow for identification of genotypes with fine-root frost tolerance. Planting gardens across decline severity gradients would also allow for targeted research on regeneration response to concurrent canopy mortality and associated establishment conditions. Given the potential for a wave of transitional mortality across the *C. nootkatensis* range (Buma, 2018), however, survival of local versus foreign seed source is hard to predict. Long-term monitoring in our plot network will allow us to track forest response to the predicted transitional mortality phenomenon and provide demographic information on the impacts of climate on different size classes as well as on the growth and survival rates of regeneration across the decline severity gradient. Demographic studies will also resolve the confounding effects of seed availability, canopy mortality, and establishment conditions on regeneration. Simultaneously investigating climate-induced tree mortality and subsequent postmortality resilience gives a clearer view of the long-term consequences of climate change on forest health, and this observational study has just scratched the surface of filling the critical knowledge gaps essential to understanding and predicting long-term forest resilience to climate change.

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CONFLICT OF INTEREST

None declared.

AUTHORS CONTRIBUTION

A.B., B.B, and S.B. conceived of the study. A.B, B.B., S.B., J.K, and L.E.O. collected the data. B.B. and S.B analyzed the data. S.B. led the writing with contributions from all authors.

DATA ACCESSIBILITY

All plot-level and summary data generated for this study are available through the Dryad Digital Repository (https://doi.org/10.5061/dryad.23420f6).

ORCID

Sarah M. Bisbing https://orcid.org/0000-0002-5534-9352

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APPENDIX

TABLE A1 Plot-level locational and compositional information from across network in southeast Alaska, USA

Plot ID	Location	Year	Latitude	Longitude	Dominant tree species - live BA	Time since CANO death (yrs)	CANO live BA per ha	CANO dead BA per ha	CANO regeneration per m^2	CANO regeneration per ha
1.1	KUP	2015	56.85047°N	133.53033°W	*C. nootkatensis*	35	38.0	26.0	0.0	0
1.2	KUP	2015	56.85076°N	133.53135°W	*T. heterophylla*	35	8.0	30.0	0.4	4,375
1.3	KUP	2015	56.85082°N	133.53009°W	*C. nootkatensis*	35	39.0	37.0	0.9	9,375
1.4	KUP	2015	56.85062°N	133.53244°W	*C. nootkatensis*	35	24.0	1.0	0.5	5,333
2.1	KUP	2015	56.85022°N	133.52821°W	*C. nootkatensis*	35	18.0	26.0	0.6	6,250
2.2	KUP	2015	56.84827°N	133.52614°W	*C. nootkatensis* – *T. heterophylla*	35	10.0	58.0	0.8	8,125
2.3	KUP	2015	56.84755°N	133.52513°W	*T. heterophylla*	35	3.0	22.0	0.3	3,125
2.4	KUP	2015	56.84936°N	133.52615°W	*C. nootkatensis*	35	15.0	13.0	0.8	8,125
3.1	POW	2016	56.21524°N	133.26588°W	*C. nootkatensis*	10	12.2	33.8	0.2	1875
3.2	POW	2016	56.21558°N	133.26656°W	*P. sitchensis*	10	0.3	47.0	1.0	10,000
3.3	POW	2017	56.21528°N	133.26707°W	*P. sitchensis*	10	17.4	41.4	1.6	15,625
3.4	POW	2017	56.21476°N	133.26645°W	*T. mertensiana*	10	17.3	14.6	5.5	55,000
4.1	POW	2016	56.22578°N	133.27412°W	*T. heterophylla*	10	2.6	31.9	0.6	6,250
4.2	POW	2016	56.22504°N	133.27389°W	*P. sitchensis*	10	6.1	39.7	2.5	25,000
4.3	POW	2017	56.22577°N	133.27365°W	*T. heterophylla*	10	15.2	37.4	<0.01	1
4.4	POW	2017	56.22473°N	133.27292°W	*C. nootkatensis*	10	25.1	26.9	<0.01	1
5.3	POW	2017	55.91775°N	132.85709°W	*T. heterophylla*	35	5.5	70.0	4.3	42,500
6.3	POW	2017	56.03667°N	133.23947°W	*T. heterophylla*	35	9.2	22.2	NA	NA
GB_010_066	GBNP	2011	58.22538°N	136.63204°W	*C. nootkatensis*	0	31.7	4.7	3.2	31,250
GB_011_071	GBNP	2011	58.24562°N	136.61262°W	*C. nootkatensis*	0	24.5	0.2	2.4	23,750
GB_012_064	GBNP	2011	58.21917°N	136.63780°W	*T. heterophylla*	0	29.5	5.2	3.6	36,250
GB_014_068	GBNP	2011	58.23110°N	136.62305°W	*C. nootkatensis*	0	45.4	5.7	5.0	50,000
GB_02_024	GBNP	2011	58.29816°N	136.67510°W	*C. nootkatensis*	0	52.2	0.6	4.4	43,750
GB_03_019	GBNP	2011	58.28641°N	136.68011°W	*C. nootkatensis*	0	41.7	1.3	9.6	96,250
GB_04_042	GBNP	2011	58.27781°N	136.71230°W	*C. nootkatensis*	0	41.8	2.6	6.0	60,000
GB_07_016	GBNP	2011	58.28207°N	136.67820°W	*C. nootkatensis*	0	28.1	3.8	3.1	31,250
GB_08_048	GBNP	2011	58.26283°N	136.71370°W	*C. nootkatensis*	0	36.0	3.8	3.4	33,750
GB_09_067	GBNP	2011	58.22911°N	136.62730°W	*C. nootkatensis*	0	44.6	4.3	3.5	35,000
H_02_062	CHICH	2011	57.55681°N	136.01001°W	*C. nootkatensis*	0	35.9	27.5	1.9	18,750

(Continues)

TABLE A1 (Continued)

Plot ID	Location	Year	Latitude	Longitude	Dominant tree species - live BA	Time since CANO death (yrs)	CANO live BA per ha	CANO dead BA per ha	CANO regeneration per m^2	CANO regeneration per ha
H_03_071	CHICH	2011	57.57203°N	136.04004°W	*C. nootkatensis*	0	35.9	6.9	7.3	72,500
H_04_091	CHICH	2011	57.62267°N	136.12257°W	*C. nootkatensis*	0	35.4	18.1	1.8	17,500
H_06_068	CHICH	2011	57.56611°N	136.02991°W	*C. nootkatensis*	0	37.8	1.5	3.0	30,000
H_08_124	CHICH	2011	57.57882°N	135.96251°W	*C. nootkatensis*	0	33.0	14.2	1.4	13,750
H_09_082	CHICH	2011	57.63800°N	136.07839°W	*T. heterophylla*	0	22.5	8.3	5.5	55,000
H_10_126	CHICH	2011	57.56995°N	135.97246°W	*C. nootkatensis*	0	38.8	5.2	2.4	23,750
H_13_088	CHICH	2011	57.62485°N	136.10913°W	*C. nootkatensis*	0	22.7	9.2	1.0	10,000
H_15_084	CHICH	2011	57.64685°N	136.11008°W	*C. nootkatensis*	0	30.3	12.7	0.9	8,750
H_16_129	CHICH	2011	57.56802°N	135.98041°W	*C. nootkatensis*	0	27.2	6.0	1.9	18,750
M_01_029	CHICH	2011	57.50626°N	135.90751°W	*T. heterophylla*	35	0.6	31.4	1.0	10,000
M_02_057	CHICH	2011	57.54192°N	135.98833°W	*T. heterophylla*	35	3.9	39.2	0.4	3,750
M_03_044	CHICH	2011	57.52819°N	135.96468°W	*T. heterophylla*	35	0.8	35.9	0.0	0
M_04_035	CHICH	2011	57.5157°N	135.91531°W	*T. heterophylla*	35	7.1	22.3	0.9	8,750
M_05_029	CHICH	2011	57.5045°N	135.90228°W	*C. nootkatensis*	35	9.4	26.9	1.0	10,000
M_06_029	CHICH	2011	57.50793°N	135.90987°W	*C. nootkatensis*	35	7.9	22.3	0.3	2,500
M_07_026	CHICH	2011	57.48985°N	135.88183°W	*C. nootkatensis*	35	11.1	28.5	0.5	5,000
M_08_031	CHICH	2011	57.51351°N	135.91613°W	*P. contorta* ssp. *contorta*	35	3.2	28.1	1.0	10,000
M_09_057	CHICH	2011	57.53963°N	135.98675°W	*C. nootkatensis*	35	9.6	22.0	0.5	5,000
M_10_051	CHICH	2011	57.53694°N	135.97525°W	*C. nootkatensis*	35	29.1	37.0	0.9	8,750
O_01_006	CHICH	2011	57.46517°N	135.82757°W	*T. mertensiana*	75	0.2	37.2	0.1	1,250
O_02_010	CHICH	2011	57.46275°N	135.83273°W	*C. nootkatensis*	75	16.3	36.3	0.8	7,500
O_03_022	CHICH	2011	57.4867°N	135.86483°W	*T. heterophylla*	75	6.5	51.4	1.0	10,000
O_04_005	CHICH	2011	57.45955°N	135.81151°W	*T. heterophylla*	75	6.9	43.0	4.6	46,250
O_05_028	CHICH	2011	57.49871°N	135.89103°W	*T. heterophylla*	75	1.3	32.8	1.1	11,250
O_06_019	CHICH	2011	57.48018°N	135.85594°W	*T. heterophylla*	75	11.4	14.3	2.6	26,250
O_08_011	CHICH	2011	57.46494°N	135.83600°W	*T. mertensiana*	75	1.1	29.9	0.4	3,750
O_09_010	CHICH	2011	57.46065°N	135.82668°W	*T. heterophylla*	75	21.2	25.9	0.9	8,750
O_10_015	CHICH	2011	57.47464°N	135.83629°W	*T. heterophylla*	75	2.5	72.2	1.1	11,250
O_14_002	CHICH	2011	57.45436°N	135.81272°W	*T. mertensiana*	75	8.5	36.1	0.5	5,000
R_01_133	CHICH	2011	57.55577°N	136.00360°W	*T. heterophylla*	10	13.2	30.1	2.1	21,250

(Continues)

TABLE A1 (Continued)

Plot ID	Location	Year	Latitude	Longitude	Dominant tree species - live BA	Time since CANO death (yrs)	CANO live BA per ha	CANO dead BA per ha	CANO regeneration per m^2	CANO regeneration per ha
R_02_055	CHICH	2011	57.53862°N	135.98150°W	*C. nootkatensis*	10	21.4	37.3	1.3	12,500
R_03_037	CHICH	2011	57.52466°N	135.93290°W	*C. nootkatensis*	10	15.3	58.2	2.3	22,500
R_04_059	CHICH	2011	57.54938°N	135.98393°W	*C. nootkatensis*	10	27.6	13.2	3.4	33,750
R_06_047	CHICH	2011	57.53040°N	135.96152°W	*C. nootkatensis*	10	9.6	58.0	0.4	3,750
R_07_058	CHICH	2011	57.54386°N	135.99047°W	*C. nootkatensis*	10	10.8	42.9	0.3	2,500
R_08_101	CHICH	2011	57.54978°N	135.95184°W	*C. nootkatensis*	10	20.5	23.9	1.1	11,250
R_09_133	CHICH	2011	57.55844°N	135.99534°W	*T. heterophylla*	10	0.0	62.0	1.1	11,250
R_10_041	CHICH	2011	57.52584°N	135.95378°W	*T. heterophylla*	10	12.9	50.7	1.1	11,250
R_11_059	CHICH	2011	57.54442°N	135.99834°W	*C. nootkatensis*	10	16.9	39.4	0.6	6,250

Abbreviations: BA, basal area (m^2/ha); CANO, *Callitropsis nootkatensis*; CHICH, Chichagof Island; GBNP, Glacier Bay National Park; KUP, Kupreanof Island; and POW, Prince of Wales Island.

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SPECIAL ISSUE

On humans and wildlife in Mediterranean islands

Jacques Blondel*

CEFE-CNRS, 1919 Route de Mende, 34293 Montpellier Cedex 5, France

ABSTRACT

Aim To investigate the effects of human-induced landscape changes in Mediterranean islands on the ecological and evolutionary responses of bird communities and populations. The combination of mass extinction of large mammals and massive deforestation by humans was hypothesized to produce new selection regimes to which organisms were likely to respond. Habitat selection and niche breadth have been investigated at the scale of species, and phenotypic variation at the scale of local populations.

Location The study was carried out along habitat gradients and in habitat mosaics at different spatial scales on the island of Corsica and in areas of similar size and structure in continental France.

Methods Two sets of gradients have been used for investigating habitat selection and niche breadth: gradients of altitude, and gradients of vegetation structure. Population studies focused on the blue tit, *Cyanistes caeruleus*. Large samples of breeding attempts by this species in 10 habitats provided detailed data on phenotypic variation of fitness-related traits both on Corsica and on the mainland.

Results The extent of niche space used by birds differed substantially depending on which habitat gradient was considered. Many species have been found to contract their habitat niche along the elevation gradient on Corsica compared with the mainland, whereas all species in the vegetation gradient broadened their niche on the island. Breeding patterns of the blue tit differed considerably depending on whether they settle in deciduous oaks (*Quercus humilis*) or in evergreen sclerophyllous oaks (*Quercus ilex*). Phenotypic variation of breeding traits was much higher on the island, where more populations were correctly timed for the best breeding period than on the mainland, a pattern that is likely to result from lower dispersal of organisms on the island.

Main conclusions The differences in observed niche breadth between the two series of habitat gradients is explained both by the species-specific ecology of the species and the human-induced environmental history of Corsica. Large-scale landscape changes provided new opportunities for island colonization by non-forest species, which are isolated as small, 'fugitive' local populations. In both gradients, forest species that are typical components of the Corsican bird fauna definitely expanded their niche and occupied a wider range of habitats on Corsica than on the mainland. At the population scale, landscapes included habitat patches with contrasted selection regimes, which resulted in high phenotypic variation for many fitness-related traits. Reduced dispersal of birds on the island resulted in a much higher degree of local differentiation on Corsica than on the mainland.

Keywords

Blue tit, gene flow, habitat gradient, human disturbance, insular syndrome, landscape changes, local differentiation, mass extinction, niche breadth, phenotypic variation.

*Correspondence: Jacques Blondel, CEFE-CNRS, 1919 Route de Mende, 34293 Montpellier Cedex 5, France.
E-mail: jacques.blondel@cefe.cnrs.fr

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INTRODUCTION

Mediterranean islands provide a fascinating framework for studying the impact of past human activities on the distribution, dynamics and evolution of vertebrate faunas. Large series of comprehensive palaeobotanical, palaeontological and archaeological studies provide a wealth of data for the past 10 millennia, the Holocene period, especially in Corsica and Sardinia (Reille, 1992; Vigne, 1992; Reille *et al.*, 1996; Vigne *et al.*, 1997). For many issues concerning environmental changes, islands provide more insight than mainland areas because drastic transformations in the composition of animal communities and the structure of landscapes have been more important and better studied on islands (Vigne & Valladas, 1996). In addition, islands are more vulnerable and less resilient to environmental changes than are mainland areas, which makes these changes easier to analyse (Walter, 2004). As pointed out by Butzer (2005), understanding cause-and-effect relationships in ecological change is difficult without an integrative and interdisciplinary methodology combining natural sciences and human sciences. Corsica is one of the few examples of a successful integration of the two approaches.

Colonization of most of the larger Mediterranean islands by modern humans dates back to the 11th–10th millennia BP (Cherry, 1990). During this long period, the islands have been deeply modified by human activity through two types of change: an almost complete turnover of mammal faunas, and active deforestation by fast-growing human societies that needed open, grassy spaces for developing agriculture and animal husbandry. These changes deeply modified the structure, composition and design of landscapes. For example, thousands of teeth and bones from Corsican archaeological sites comprise an almost complete succession of mammal species from the end of the Pleistocene (*c.* 10,000 years BP) to the present (Vigne & Valladas, 1996). Human activities brought about the extinction of the entire autochthonous mammalian fauna and the gradual introduction of more than 25 taxa of mammals, which constitute the present wild and domestic fauna of these islands. Of the endemic mammals, only four small-sized species survived until the Iron Age: *Prolagus sardus* (Lagomorpha), *Rhagamys orthodon* (Muridae), *Tyrrhenicola henseli* (Microtidae) and *Episoriculus corsicanus* (Soricidae). Human influence on vegetation, mostly through deforestation, was apparent as early as the end of the Neolithic (Reille, 1992) and has not really stopped since then, despite many ups and downs resulting from political, demographic and economic pulses (Thirgood, 1981; Vigne & Valladas, 1996; Blondel & Aronson, 1999; Butzer, 2005). Pollen analyses indicated that the most severe deforestation period occurred after the conquest of the island by the Romans, and did not stop through classical antiquity up to the 13th century. The mechanisms of landscape change are fairly well known through various techniques, including palynological, pedoanthracological, palaeontological and archaeological analyses (Thinon, 1978; Reille, 1984, 1992; Butzer, 2005). Fossil assemblages of small mammals determined from the contents of fossil pellets of the barn owl *Tyto alba* proved good indicators of changes in vegetation cover and landscape structure over several millennia (Libois, 1984; Vigne & Valladas, 1996). For example, a global decrease of the small mammals *Rhagamys* and *Apodemus*, associated with a sudden increase of *Mus* some 2500 years ago, indicates a strong increase of human impact with large-scale opening up of vegetation. Signatures of vegetation change from various cues have been used by Vigne & Valladas (1996) as indicators of agricultural cycles affecting the landscape structure at different periods of the Middle Ages. The most dramatic vegetation changes were the partial replacement of deciduous tree species (e.g. downy oak, *Quercus humilis*) by sclerophyllous tree species (e.g. holm oak, *Quercus ilex*). When humans first colonized Corsica, most of the lowlands of the island were covered by downy oaks, and human action progressively resulted in the replacement of this species by the evergreen sclerophyllous holm oak (Reille, 1992). This replacement has been associated, as almost everywhere in the Mediterranean Basin, with a general desiccation of the region (Blondel & Aronson, 1999; Butzer, 2005).

This paper aims first to set the scene of environmental transformation by documenting the role of humans in biodiversity decline, changes and faunal turnover, then to show that the human-induced transformation of landscapes has resulted in two different wildlife responses (only birds are studied here): (1) changes in immigration processes and habitat selection by the new colonizers (niche breadth), and (2) ecological and evolutionary responses of populations to habitat heterogeneity and fragmentation (phenotypic variation). A nested scaling from landscapes to communities and populations was used.

CHANGES IN BIODIVERSITY

Landscapes have been designed and redesigned by humans for almost 10,000 years in the eastern part of the Basin, and for 8000 years in the western part (Le Houérou, 1981; Braudel, 1985; Pons & Quézel, 1985; Butzer, 2005). Discussing whether human impact has been beneficial or detrimental to biodiversity in the Mediterranean would be a challenging issue, and is not the aim of this paper, but two contrasting theories are worth mentioning briefly. The first is the 'ruined landscape' or 'lost Eden' theory advocated by Attenborough (1987), which argues that human action (deforestation and overgrazing) resulted in a progressive and cumulative degradation and desertification of Mediterranean landscapes. Challenging this pessimistic view, the second approach dismisses the supposedly detrimental effects of humans, arguing that the imaginary past idealized by artists and scientists does not reflect reality. According to this school of thought, humans actually contributed to maintaining Mediterranean landscapes as they progressively established since the last glacial episode, and favoured biodiversity by shaping a variety of cultural landscapes (Grove & Rackham, 2001; see Butzer, 2005; Blondel, 2006). Blondel & Aronson (1995, 1999) argued that many traditional land-use practices act as surrogates of natural

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disturbance regimes, with the consequence that, according to the intermediate disturbance hypothesis (Huston, 1994), several components of biodiversity have actually been higher in landscapes shaped by humans than in primitive plant communities such as oak woodlands.

The ups and downs in biodiversity on Mediterranean islands are not discussed in detail here; only events with the most relevant consequences for the ecology and evolution of ecological systems are considered. If the current biodiversity crisis is indisputable, with many plant and animal species being at risk in the Mediterranean Basin (Blondel & Médail, in press), especially on islands, the most important events occurred in the past, especially at the end of glacial times and the first millennia of the Holocene, so the current level of extinction in the Mediterranean is perhaps not significantly higher than during geological times. The fact that drastic human-induced changes occurred long ago is interesting in that it resulted in significant ecological and evolutionary responses that would not have happened if these changes had been more recent.

Perhaps the most dramatic, and still controversial, event of the Late Pleistocene, with many consequences for habitats and landscapes, is the tempo and mode of the mass extinction of large mammals. The question of whether this mass extinction was caused mainly by humans (as advocated by the overkill hypothesis; Martin, 1984), or resulted from natural environmental changes, or a combination of both, is still open and hotly debated, but recent studies emphasize the importance of human impact (e.g. Brook & Bowman, 2004; Burney & Flannery, 2005). The decimation of large endemic mammals continued well into the Holocene, as sadly illustrated by the human-induced extinction of all the large 'mega-nano-mammals' of Mediterranean islands, for example the dwarf hippos, elephants and deer of Cyprus, Malta, Sicily and other islands following the colonization of these islands by humans (Simmons, 1988, 1991). No fewer than 12 species of dwarf descendants of the ancestor elephant *Palaeoloxodon antiquus* inhabited Mediterranean islands (Lister, 1996). These species were very variable in size, and are an interesting example of evolutionary convergence because the different species evolved independently from *P. antiquus*. Even the small island of Tilos (64 km^2) in the Aegean Sea had its own species of dwarf elephant. The smallest species, *Palaeoloxodon falconeri*, less than 1 m high, from Sicily, probably gave rise to the myth of the Cyclops Polyphem in the *Odyssey* of Homer because of the large frontal hollow of the nostrils, which looks like an enormous single eye. In addition to these mammals, tortoises, giant rodents and flightless owls, which populated most Mediterranean islands, were also decimated by humans as soon as they invaded them (Blondel & Vigne, 1993; Vigne *et al.*, 1997). Recent studies showed that human population pressures, which increased abruptly during the late Palaeolithic and again during the Upper and Epi-Palaeolithic periods, prompted people to expand into new territories and colonize islands, as exemplified by the colonization of Cyprus (Davis, 2001). Hunting pressures became so high that they resulted in a gradual shift in culinary remains from very large animals, which became less and less common, to small animals, especially fish, birds and shellfish (Stiner *et al.*, 1999). These demographic pulses are evidenced by increasing reliance on agile, fast-reproducing, small animals at the expense of slowly reproducing but easily caught tortoises and marine shellfish, which became less common as a result of overharvesting. Demographic pressure, with more and more people to feed, ultimately stimulated both animal husbandry for food, and the search for new territory overseas, which is clearly the case for Cyprus (Morales *et al.*, 1998; Stiner *et al.*, 1999; Davis, 2001). Such high human pressure must have had a great impact on habitats and landscapes. Although we still lack clear evidence for the co-existence of man and the endemic Cypriot hippos and elephants, Simmons (1988) suggested that their bones, found with cultural remains at Akrotiri, indicate the possibility of a pre-Neolithic 'kill-site' of these animals. It is reasonable to assume that soon after the arrival of man, these pygmy animals became extinct through overharvesting or through competition with the feral pig, which had been imported from the nearby mainland (Simmons, 1991).

All the endemic mammals of Mediterranean islands were doomed to extinction, and a complete turnover of the mammal fauna took place over the course of the past 10 millennia or so. More than 20 mammal species, both domestic and wild, immigrated from the nearby mainland, voluntarily or involuntarily introduced by humans (Vigne, 1992). Few species of birds have been introduced by humans on Mediterranean islands, but one interesting example is the partridges of the genus *Alectoris*. A striking characteristic of the fossil avifauna of the Palaeolithic and Holocene on Mediterranean islands is the absence of galliform species, except the quail (*Coturnix coturnix*), perhaps because galliforms are mostly sedentary and do not cross the sea channels that separate the islands from the continent (Alcover *et al.*, 1992; Blondel & Vigne, 1993). The four partridge species that occur today around the Basin (*Alectoris rufa*, *A. graeca*, *A. chukar*, *A. barbara*) have been introduced as game species several times on each of the larger islands. However, only one species of partridge occurs today on each island, which is that currently occurring on the nearest mainland. This is an interesting example of competitive exclusion of allospecies – species that are differentiated enough to be considered as full species, but are ecologically too similar to live in sympatry.

LANDSCAPE CHANGES AND HABITAT SELECTION: NICHE BREADTH OF BIRDS IN CORSICA

A classical component of the insular syndrome is niche enlargement, which has repeatedly been shown to occur on islands (MacArthur *et al.*, 1972; Williamson, 1981; Blondel, 2000; Whittaker & Fernández-Palacios, 2007). Contradictory results have recently been reported on Corsica, depending on which components of the niche are considered. This discrepancy is probably due to secondary human-induced changes in habitats and landscapes, which have modified patterns of

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habitat use by birds and provided new opportunities for several species to colonize the island. In a recent study, Prodon *et al.* (2002) found that most species contract their habitat niche on Corsica, a result that runs counter to the classical tenet of island biology, as well as with previous results that also analysed the habitat niche of land birds on this island (Blondel *et al.*, 1988; Pavoine *et al.*, 2007). Prodon and colleagues compared the altitudinal extension of bird species along the slopes of mountain ranges from sea level to 2784 and 2710 m a.s.l. in the Pyrenees (southern France) and in Corsica, respectively. The study of Blondel and colleagues focused on the distribution of birds along seven stages of two forested habitat gradients, starting from very low matorral and ending with an old mature forest of the Mediterranean holm oak. In both studies, the mainland and island systems matched each other reasonably well, with a similar zonation of vegetation belts (for details and methods see Blondel *et al.*, 1988; Prodon *et al.*, 2002). Prodon and colleagues found that a majority of bird species contracted their habitat niche on the elevation gradient, with only a small proportion of them enlarging it, whereas Blondel and colleagues found the opposite on the vegetation structure gradient. The reasons why the two studies differ are interesting to analyse.

Following the taxon-cycle rationale (Ricklefs & Cox, 1978), species have been assigned by Prodon and colleagues to one of six categories according to their degree of morphological differentiation compared with their mainland counterparts. This scale, referred to as the 'immigration-differentiation' scale, was assumed to correspond to the time since the species colonized the island. Categories I and II included irregular breeders or species recorded on Corsica only within the previous 20 years; category III included undifferentiated regular breeders recorded on Corsica over more than a century; and categories IV and V encompassed species that are well differentiated at the subspecific level. Category VI corresponds to the two insular endemic species that have no mainland counterparts (the Corsican nuthatch *Sitta whiteheadi* and the Marmora's warbler *Sylvia sarda*). Fossil remains have also been considered by Prodon *et al.* (2002) for assigning species to these categories, but this does not make sense in an island that is visited twice a year and in winter by tens of millions of migrants and wintering birds in the course of their migration between Africa and Eurasia.

The number of species in common in Corsica and on the mainland was 85 in the elevation gradient and 26 in the vegetation structure gradient. This large difference is due to two factors in the elevation gradient: (1) a larger diversity of species as a result of a larger variety of habitats; and (2) a large number of species with small groups of individuals that are highly localized and do not constitute established permanent self-sustainable populations. In contrast, all species in the vegetation structure gradient are members of well established communities that have evolved traits of the insular syndrome and are widespread everywhere in forested habitats of Corsica. The number of habitats per species, which is a proxy for measuring niche breadth (Cox & Ricklefs, 1977), is reported in Table 1. In Table 2 the niche space occupied by the species is expressed as the ratio of the size of this space on the island to that on the mainland (island/mainland amplitude ratio, IMAR; Prodon *et al.*, 2002). Species enlarge their niche for a ratio > 1 and reduce it for a ratio < 1. Mean niche breadth for these two estimates of habitat selection differed considerably between the two studies, and was clearly much narrower along the altitudinal gradient than along the vegetation gradient. On average, species extended over 6.33 intervals of 200 m altitude in Corsica, compared with 8.43 on the mainland (Prodon *et al.*, 2002), whereas species occupied 4.07 habitats on Corsica compared with 2.22 on the mainland (Blondel *et al.*, 1988). With 66 species (78%) exhibiting an IMAR < 1, Prodon *et al.* (2002, p. 1294) concluded that 'there was an overall trend in favour of range compression on Corsica', which runs counter to the classical tenet of niche enlarging on islands. It could be argued that species respond differently to these two components of niche space (altitude and habitat structure), but the diversity of habitats along the elevation gradient is greater than that in the vegetation structure gradient, so species should be expected to exhibit an even larger niche breadth along this gradient.

Table 1 Average number of habitats occupied by bird species along the two gradients ± 1 SD.

	Elevation gradient (number of 200-m elevation stripes per species)	Habitat gradient (number of habitats per species)
Mainland	8.43 ± 2.71	2.88 ± 1.58
Corsica	6.33 ± 3.24	4.54 ± 1.79

For the elevation gradient, the number of elevation stripes and bird species were 12 and 85, respectively (Prodon *et al.*, 2002; original data in Ecological Archives EO83-021-A1). For the habitat gradient, the number of habitats and bird species were seven and 26, respectively (original data in Blondel *et al.*, 1988).

Table 2 Island/mainland ratios of niche breadth for species that occur in both Corsican and mainland habitat gradients.

Categories	Altitude gradient*	Vegetation gradient†
I	0.29 ± 0.26 (11)	1.60 ± 1.12 (15)
II	0.67 ± 0.32 (7)	
III	0.82 ± 0.40 (49)	
IV	0.84 ± 0.13 (8)	2.79 ± 1.68 (10)
V	1.18 ± 0.25 (10)	

Results ± 1 SD; numbers of species in parentheses. Categories refer to the degree of morphological differentiation of Corsican populations, which is assumed to be proportional to the time since the species colonized the island (see text; Prodon *et al.*, 2002, p. 1297).
*Data from Prodon *et al.* (2002).
†Data from Blondel *et al.* (1988). Categories of the 'immigration-differentiation' scale are: I and II, irregular and/or recent breeding birds; III, undifferentiated regular breeders; IV and V, species with well differentiated subspecies; see text.

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Niche changes on islands have traditionally been considered as a proximate plastic response to the number of potential competitors: a population contracts its niche when meeting more competitors, and enlarges it when meeting fewer competitors. The competition paradigm has been hotly debated, but the basic rationale of niche changes on islands has not been fundamentally questioned. Whether the causal mechanisms that lead to niche shifts on islands result from changes in interspecific interactions such as predation, competition or parasite loads (e.g. density compensation; MacArthur *et al.*, 1972), or another causal mechanism related to population-specific responses to new environments (e.g. population spillover resulting from density inflation; Crowell, 1983; Blondel *et al.*, 1988), these shifts are evolutionary responses that require time to become established as population-specific life-history traits. In the framework of the taxon-cycle hypothesis, it has been argued that niche expansion is common in new colonizers, and that the new selection regimes prevailing on islands lead to local differentiation and ultimately to niche contraction as a result of local specialization. This suggests that niche breadth should be inversely proportional to the time since a population colonized an island (Ricklefs & Bermingham, 1999). Niche shifts have been discussed in the framework of a taxon cycle by Prodon *et al.* (2002), as components of evolutionary changes that characterize populations after they colonized an island, but their interpretation differed from the taxon cycle rationale. Steps of a taxon cycle do not involve niche enlarging, but instead a progressive reduction of range, specialization and eventually extinction (Ricklefs & Cox, 1978; Ricklefs & Bermingham, 1999). Following the rationale of insular evolution, one may predict that species that colonized an island a long time ago have a larger niche than recent colonizers. In both studies (habitat gradients and elevation gradients), species of categories IV and V (which have been pooled together in the vegetation structure gradient because of low sample size) exhibit a larger habitat niche than those of lower categories on the immigration–differentiation scale (categories I–III). Thus the studies of Prodon *et al.* (2002) and Blondel *et al.* (1988) came to the same conclusion for those (mostly forest) species that colonized the island a long time ago. The increase in mean IMAR along the five categories of increasing morphological differentiation suggests that species tend to enlarge their habitat niche with increasing time since colonizing Corsica (Prodon *et al.*, 2002, fig. 4), which is the expected trend on an island.

Species that have the narrowest altitudinal niche in Prodon *et al.*'s (2002) study, and which occur as small localized populations, with many not breeding regularly each year, raise two crucial questions in island biogeography: What is an island population? and To what extent does immigration differ from colonization? These points have been addressed by Diamond & May (1977) in analyses of species turnover on the islands of Farnes and Skokholm off the coast of the British Isles. These authors showed that most 'insular populations' on these islands were actually fugitive members of mainland populations that failed to establish themselves on the island. As pointed out by Haila & Hanski (1993), isolated breeding pairs or small groups of birds that breed sporadically in small habitat patches cannot be considered as demographically functional populations belonging to an insular fauna *sensu* Ebenhard (1991). Local groups of birds that breed on islands not far from the mainland are often parts of larger populations that occur on much larger areas. These erratic and ephemeral individuals do not contribute to insular faunal turnover, and cannot be considered as colonizers. Williamson (1981) and Haila *et al.* (1993) gave several similar examples of pseudo-extinction in continental habitat islands in the British Isles (Eastern wood, Surrey, England) and in the Finnish archipelago, respectively. In Corsica, no fewer than 15 species have been reported to immigrate to the island and to breed in one or two breeding attempts, but to have failed to become true colonizers (Thibault & Bonaccorsi, 1999; personal observation). These species, included in the list of Prodon *et al.* (2002), strongly contribute to 'contracting' the average niche of the species. Examples include *Asio otus*, *Phoenicurus ochruros*, *Monticola saxatilis*, *Hippolais polyglotta*, *Sylvia hortensis*, *Phylloscopus bonelli*, *Phylloscopus collybita*, *Pica pica* and *Sturnus vulgaris*, to cite just a few. All are recent ephemeral colonizers or vagrants (the so-called 'tramp' strategists of Diamond, 1974). They belong to categories I and II, and are localized or patchily distributed in the lower parts of the island. The presence on Corsica of these 'tramp' species is presumably a secondary feature mostly due to human-induced habitat changes that have made many new habitats suitable for them.

On the other hand, forest species have occurred on the island for a long time and many have differentiated as subspecies, involving a significant enlargement of their habitat niche, as noted in the two studies. They repeatedly followed the vegetation belts as the alternation of glacial and interglacial cycles moved up and down the limits of forests during the Pleistocene. Prodon *et al.* (2002) rightly pointed out that they evolved traits that allowed them to survive these climatic vicissitudes, as shown previously by Blondel & Farré (1988); Blondel & Vigne (1993), and confirmed by Pavoine *et al.* (2007). They extended over the whole range of forest belts, which were much narrower during full-glacial than during inter-glacial periods such as the present one (Beug, 1967; Reille *et al.*, 1997, 1999) and enlarged their altitudinal range accordingly (see fig. 12 in Blondel, 1988). Most Corsican endemic subspecies or well differentiated forms are forest birds that experienced these shifts in altitudinal range in response to Pleistocene climatic changes.

HABITAT HETEROGENEITY AND POPULATION CHANGES: A REACTION NORM APPROACH

Scaling down the effects of human action on living systems, this section examines the consequences of habitat heterogeneity and patchiness on life-history evolution at the population level. Landscape fragmentation is a consequence of ancient practices throughout the Mediterranean Basin, adding to the

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natural diversity of landscapes and habitats, and potentially producing new selection regimes to which populations may show evolutionary responses (Blondel, 2006; Blondel *et al.*, 2006). If habitat fragmentation often threatens biodiversity by increasing inbreeding and extinction risks of small, isolated populations, the process is not necessarily detrimental to all aspects and metrics of biodiversity, as shown by the consequences of traditional land-use practices such as the ancient agro–sylvo–pastoral systems on alpha, beta and gamma diversity of plant and animal communities in man-made habitat mosaics (Blondel & Aronson, 1999).

Human impact on Mediterranean ecosystems has two consequences of great importance for wildlife. The first is the fragmentation of forests into patches of various sizes and degrees of isolation. The second is that most broad-leaved forests are dominated either by deciduous trees (e.g. downy oak, *Q. humilis*) or by evergreen sclerophyllous trees (e.g. holm oak, *Q. ilex*). This makes forest landscapes into mosaics of habitat patches dominated by one or the other of these two oak species, with more than 95% of the trees of each forest patch usually being of the same species. A long-term programme designed for investigating the response of blue tit (*Cyanistes caeruleus*) populations to these two features gave an insight into the huge phenotypic variation of this small passerine (Blondel *et al.*, 2006). This man-made landscape design is a unique natural experiment, because the different physiology of the two oak species has many consequences for several components of food chains that depend on the oaks. A crucial element of the story is that the spring flush of leaves, measured as leaves developing from dormant buds to fully developed leaves, starts 1 month earlier in deciduous oaks than in evergreen oaks (Figs 1 & 2). Bud burst is a key event because it triggers the hatching of leaf-eating caterpillars, the preferred prey of blue tits, which will start to feed on young leaves. This event activates food chains, which cascade from oak leaves through caterpillars to insectivorous blue tits, and follow different temporal trajectories in the two types of oak forest. In addition to this large qualitative difference in timing, there is also a large quantitative difference. Caterpillars are much more abundant in the deciduous downy oaks because the whole foliage is renewed each year, as compared to only 30% renewal in evergreen oaks (Blondel *et al.*, 1993). Only young leaves are edible by caterpillars, so the window of food availability is short, 2 weeks at most, which makes the best time for tit breeding rather short (Blondel *et al.*, 1999). The 1-month difference in bud burst and leaf production between the two oak morphotypes results in an early- and high-amplitude peak in caterpillar production in downy oak forest, which contrasts with a late- and low-amplitude peak in holm oak forest (Fig. 2). This combination of large differences in phenology and abundance of caterpillars is crucial because food supply has repeatedly been shown proximately and ultimately to

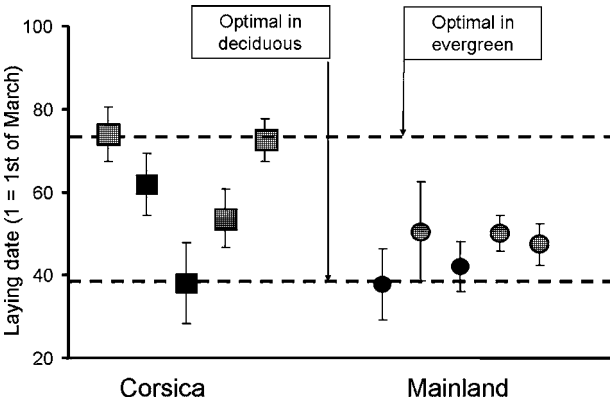

Figure 1 Variation of laying date (± 1 SD) for 10 blue tit (*Cyanistes caeruleus*) populations in landscapes including habitat patches dominated either by deciduous downy oak (*Quercus humilis*) or by evergreen holm oak (*Quercus ilex*). Horizontal dotted lines denote the best date for starting to breed relative to food availability in deciduous downy oaks and in evergreen holm oaks, respectively. Laying dates, 1 = 1 March, 32 = 1 April, etc.). Black symbols, deciduous habitats; hatched symbols, evergreen habitats. Note the much higher variation of laying dates in Corsica than on the mainland (modified from Blondel *et al.*, 2006).

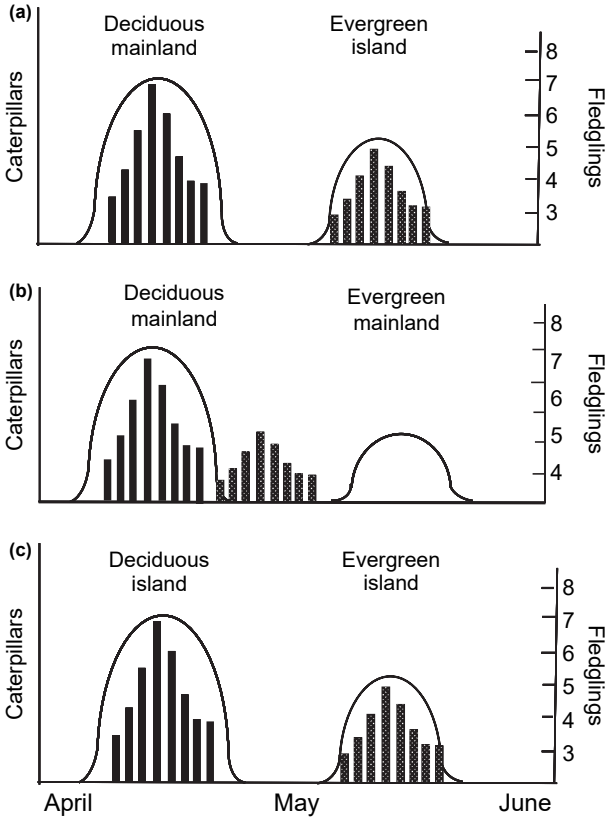

Figure 2 Caterpillar abundance (curves) estimated by caterpillar droppings falling from oak leaves and number of blue tit (*Cyanistes caeruleus*) fledglings (histograms) in deciduous downy oaks (*Quercus humilis*) and evergreen holm oaks (*Quercus ilex*) on the mainland and on Corsica. Scale not given on left vertical axis because of the huge between-habitat variation of caterpillar abundance. (a) Between-landscape scale; (b) within-landscape scale on the mainland; (c) within-landscape scale on Corsica. See text (simplified from Blondel *et al.*, 2006).

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determine the values of breeding traits such as laying date, clutch size and breeding success in income breeders such as small passerines (Lack, 1968; Drent & Daan, 1980). The question arises: how do blue tits respond to these large differences in timing and abundance of food resources? To answer this question, we examined the phenotypic variation and reaction norms of a series of populations from an experimental design that included two landscapes with two deciduous habitat patches and three evergreen habitat patches each. One landscape was near Montpellier on the mainland, southern France, and the other on the island of Corsica. Each habitat patch was equipped with nest boxes for monitoring the breeding process of tits, and trays under the canopy of trees were used to collect the droppings of caterpillars and thus to quantify the amount of food available for tits (for details see Blondel *et al.*, 1993, 2006). An important point that must be kept in mind is that deciduous oakwoods are more common than evergreen oakwoods on the mainland, whereas the reverse is true on the island.

The following briefly summarizes some results of this study that are relevant to discussion of the evolutionary consequences for wildlife of human impact on landscapes. The dotted lines in Fig. 1 (laying dates, 1 = 1 March; 32 = 1 April, etc.) represent the best date for starting to breed in each type of oakwood. These lines represent the time when the leaf buds start to burst, triggering the hatching of caterpillars that become available as food for tits. On the mainland, only the two populations breeding in deciduous oaks can be considered as more-or-less 'correctly' timed on the caterpillar peak, but the three populations breeding in evergreen oaks obviously miss the best time to breed because they started too early in relation to caterpillars. On Corsica, three populations were correctly timed, two in evergreen and one in deciduous oaks, which made the phenotypic variation much higher on the island than on the mainland despite the fact that the two landscapes match each other fairly well (Fig. 1).

In a first step, we examined breeding patterns of tits in a deciduous oakwood on the mainland, where deciduous oaks dominate landscapes, and in an evergreen oakwood on the island, where evergreen oaks dominate landscapes. In both habitats, blue tits nicely synchronized their breeding time in such a way that the supply/demand ratio of food was optimal in the two habitats (Fig. 2a). On average, tits started to lay 3–4 weeks earlier in deciduous mainland oaks than in evergreen island oaks (6 April vs. 12 May); produced larger clutches (9.8 vs. 6.4 eggs); and raised on average seven fledglings in the food-rich deciduous oakwood compared with only four to five fledglings in food-poor evergreen oakwood. Common garden experiments have shown that the > 3-week difference in breeding time was genetically determined (Lambrechts *et al.*, 1997).

In a second step, we examined the breeding patterns of blue tits in the two types of oakwood, but at the within-landscape scale on the mainland. We found that tits started to breed at least 2 weeks too early in the evergreen oakwood, so they missed the best period for raising their young (Fig. 2b). As a consequence, they produced offspring of poor quality, which had low prospects of being recruited into the breeding population. Several lines of evidence from molecular genetics (Dias *et al.*, 1996) and quantitative genetics (Charmantier *et al.*, 2004) have shown that this mistiming is a consequence of gene flow across habitat patches. Density-dependent mechanisms have resulted in birds that are specialized in deciduous oaks, where they developed habitat-specific adaptations, migrating into evergreen oaks. Such patterns resulted in a source–sink population structure that involved asymmetrical dispersal from the former (source) to the latter (sink). In the mainland landscape, where birds were assumed to disperse freely across habitat patches, little local differentiation of both breeding and morphometric traits has been found within a range of *c.* 40 km, supporting the hypothesis of gene swamping between populations (Blondel *et al.*, 2001).

In a third step, a null hypothesis predicted that a similar maladaptation would occur in Corsica, but the other way round, with blue tits being nicely synchronised to the peak of caterpillars in the evergreen oakwoods (which are more abundant on the island) and maladapted to this peak in deciduous oakwoods. Interestingly, blue tits proved to be equally well synchronized to the caterpillar peak in the two habitat types, which were no more than 25 km apart (Fig. 2c). In addition, the two populations differed in a number of traits, including demographic and morphometric traits, suggesting a genetically based local adaptation (Blondel *et al.*, 1999). This case study shows that local specialization may occur between populations at a scale that is much smaller than the scale of dispersal of the organism and potential gene flow. This difference between the mainland and Corsica has been interpreted as resulting from reduced dispersal ranges of birds on islands, and supports the divergence-with-gene-flow model of speciation (Blondel *et al.*, 1999) that had been already proposed long ago by Maynard Smith (1966) and Felsenstein (1976). What explains the large difference between the mainland and the island is a greatly different gene flow/selection tension, with the effects of gene flow being counteracted by local selection on the island, but not on the mainland. Lower dispersal on the island, which is a component of the insular syndrome, explains that populations may respond to strong directional selection even if there is some gene flow. Reduced dispersal combined with assortative mating and oak-specific habitat selection makes birds more tightly linked to their local habitat on the island than on the mainland.

To summarize, from a reaction-norm approach (Kawecki & Stearns, 1993), mainland and Corsican populations of blue tits belong, so to speak, to two genotypes, a 'deciduous genotype' on the mainland and an 'evergreen genotype' on the island of Corsica. The reaction norms of these genotypes do not cross, and correspond to a genetically determined local specialization to local selection regimes. In addition to this local specialization, there is some phenotypic plasticity both in time and in space. Plasticity in time is expressed by the proximate year-to-year variation of breeding time depending on yearly variation of temperatures and food (standard deviation around the

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mean laying date). Plasticity in space results from the fact that tits always breed earlier in deciduous than in evergreen oakwoods, whatever the region, and always earlier in the wild than in aviaries, whatever the dominant tree species.

CONCLUSION

Eight millennia of human impact on ecosystems have resulted in dramatic changes in the composition and structure of landscapes in Mediterranean islands. These changes were probably more important (and easier to demonstrate) on islands than on the mainland because islands are less resistant and resilient to disturbance than mainland areas. They deeply modified the distribution and structure of habitats and opened the route to invasions by new colonizers, which may lead to misleading conclusions about habitat selection and niche shifts on islands.

After so many centuries of co-habitation between humans and nature, most Mediterranean ecosystems are so inextricably linked to human interventions that evolutionary responses to human-induced changes are difficult to decipher. There could have been a myriad of case studies exemplifying evolutionary processes such as those described in this paper with the blue tit model. Such studies are badly needed because a detailed analysis of potential evolutionary responses of organisms to human-induced changes could help evaluate the tempo and mode of the forthcoming response of organisms to the many changes that will occur in the Mediterranean in the future. According to scenarios of the IPCC (2001), the Mediterranean region will be particularly affected by global warming, with a strong decrease in rainfall.

Studies on habitat niche breadth in Corsica showed that species do not all respond in the same way to environmental changes. This means that sorting processes may shape new species assemblages as a result of changes in probabilities of extinction and colonization as well as changes in the migratory behaviour of many species. On the other hand, the blue tit studies provide good evidence that micro-evolutionary changes can occur rapidly in fitness-related traits with high heritability values, such as those related to breeding processes in birds (Visser *et al.*, 2003; Charmantier *et al.*, 2004) or flowering time in plants (Peñuelas *et al.*, 2002). Palynological studies have shown that most lowland forested areas in Corsica were covered by deciduous trees before humans destroyed them. Blue tits responded within a few generations to the substitution of tree species, and succeeded in adapting to the new human-made design of landscapes. Such rapid evolutionary responses suggest that birds will be able to cope with the new environmental conditions associated with global warming.

Once again, islands shed an interesting light on several aspects of evolutionary responses to environmental heterogeneity. In particular, the much higher phenotypic plasticity of blue tits observed on Corsica than on the mainland demonstrates that species impoverishment in islands may, to some extent, be compensated by an increase in intraspecific diversity, as shown by the high differentiation of populations at small spatial scales. Comparing the responses of less-dispersive island birds and highly dispersive mainland birds to the same spatial diversity of habitat gives us a practical demonstration of the relationships between dispersal, spatially variable selection and local adaptation.

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BIOSKETCH

Jacques Blondel is Director of Research emeritus at the Centre d'Ecologie Fonctionnelle et Evolutive, CNRS, Montpellier, France. His main interest is the biogeography, evolutionary ecology and population biology of birds at different spatial scales, from the whole Mediterranean Basin to local habitats, focusing on island–mainland comparisons.

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The following recent literature summary was prepared by Benjamin Bograd (Princeton University) of road impacts and roadless area importance using 3 search engines (ScienceDirect, Scopus, Google Scholar) to gather the road impact pdfs in our comments. The Forest Service must assess the information described in the following studies in the EIS for the proposed rule.

Link/Title/ Author(s)	**Year**	**Region**	**Abstract/Synopsis**
Roadless in the Pacific Northwest: Ecology and History - James Furnish	2019	PNW	A notable but unexpected outcome of this dramatic policy shift is a significant increase in stored forest carbon following logging reductions. Such carbon gains team with other "free" environmental services like clean water and air, and high quality wildlife and fish habitat to illustrate important non-commercial forest values.
Roadless Areas as Key Approach to Conservation of Functional Forest Ecosystems - Monika Hoffman, Stefan Kreft, Vassiliki Kati, Pierre Ibisch	2019	General	Among all terrestrial ecosystems, roadless forests are the single most important strongholds of regulating ecosystem services: among others, soil protection, water retention, buffering of the local and regional climate and mitigation of global climate change via capturing of atmospheric carbon. Large roadless areas can serve as a measurable surrogate for the most pristine and functional ecosystems. Roadlessness is a property of areas, which are not impacted by roads; it can be used as a proxy for assessing ecosystem integrity and the absence of many anthropogenic disturbances (note – ecosystem or ecological integrity is covered under the 2012 forest planning rule). We recommend that policy-makers give roadless areas conservation priority over areas that have already been fragmented. It is essential to establish roadlessness as a criterion for the planning of ecosystem-based, cost-effective sustainable development.. Even if "climate-friendly" renewable energy was available for road transport on a large scale, the construction, existence and operation of roads would continue to severely impair ecosystem functionality.
On the Variable Effects of Climate Change	2019	Columbia River Basin	Water temperature has manifold effects on the biology of Pacific salmon. Thermal optima enable Pacific salmon to maximize growth while temperatures above thermal

on Pacific Salmon - Xiao Zhang, Hong-Yi Li, Zhiqun Deng, L. Ruby Leung, John Skalski, Steven Cooke			optima can induce stress and lead to mortality. This study investigated the impacts of climatic changes and water management practices on Chinook and Steelhead smolts in the Columbia River Basin using an integrated earth system model and a multiple regression model that incorporated nonlinear survival responses to water temperature. Results revealed that the effects would vary significantly with the species, location, and climate change scenario. Mean survival rates may increase by more than 10% in Upper Columbia River, while reduce by 1~13% and 2~35% for Chinook and Steelhead smolts respectively, in the Lower Columbia River by 2080s. This study highlights the importance of integrating the nonlinear response of survival rate to river temperature and water management effects in climate change vulnerability analysis for salmonid stocks.
Assessing the risk to the conservation status of temperate rainforest from exposure to mining, commercial logging, and climate change: A Tasmanian case study - Brendan Mackey, Sean Cadman, Nicole Rogers, Sonia Hugh	2017	Tasmania, Australia	Allowing structural degradation and fragmentation to intact rainforest blocks will reduce their capacity to buffer meso-climatic variability and resist fire events thereby undermining their ecosystem integrity.
Assessing the value of roadless areas in a conservation reserve strategy: biodiversity and landscape connectivity in	2005	Northern Rockies	• Roadless areas on Forest Service lands hold significant potential for the conservation of native biodiversity and ecosystem processes, primarily because of their size and location. • Roadless areas, when added to existing federal-protected areas in the northern Rockies, increase representation of virtually all land-cover types, some by more than 100%, and increase the protection of relatively undisturbed lower elevation lands, which

the northern Rockies - Michele Crist, Bo Wilmer, Gregory Aplet			are exceedingly rare in the northern Rockies. In fact, roadless areas protect more rare and declining land-cover types, such as aspen, whitebark pine, sagebrush and grassland communities, than existing protected areas. • Roadless areas adjacent to protected areas increase connectivity by creating larger and more cohesive protected area 'patches.' Roadless areas enhance overall landscape connectivity by reducing isolation among protected areas and creating a more dispersed conservation reserve network, important for maintaining wide-ranging species movements. We advocate that the USDA Forest Service should retain the Roadless Area Conservation Rule and manage roadless areas as an integral part of the conservation reserve network for the northern Rockies.
A global map of roadless areas and their conservation status - Pierre L. Ibisch, Monika T. Hoffmann, Stefan Kreft, Guy Pe'er, Vassiliki Kati, Lisa Biber-Freudenberger, Dominick A. DellaSala, Mariana M. Vale, Peter R. Hobson,Nuria Selva	2017	Comprehen-sive and global in scope	Roads fragment landscapes and trigger human colonization and degradation of ecosystems, to the detriment of biodiversity and ecosystem functions. About 80% of Earth's terrestrial surface remains roadless, but this area is fragmented into ~600,000 patches, more than half of which are <1 square kilometer and only 7% of which are larger than 100 square kilometers. Global protection of ecologically valuable roadless areas is inadequate. International recognition and protection of roadless areas is urgently needed to halt their continued loss.
USDA Forest Service Roadless Areas: Potential Biodiversity Conservation Reserves - Colby Loucks, Nicholas Brown, Andrea	2003	United States	We found that more than 25% of IRAs are located in globally or regionally outstanding ecoregions and that 77% of inventoried roadless areas have the potential to conserve threatened, endangered, or imperiled species. IRAs would increase the conservation reserve network containing these species by 156%. We further illustrate the conservation potential of IRAs by highlighting their contribution to the conservation of the grizzly bear (Ursos

Loucks, and Kerry Cesareo			arctos), a wide-ranging carnivore. The area created by the addition of IRAs to the existing system of conservation reserves shows a strong concordance with grizzly bear recovery zones and habitat range. Based on these findings, we conclude that IRAs belonging to the U.S. Forest Service are one of the most important biotic areas in the nation, and that their status as roadless areas could have lasting and far-reaching effects for biodiversity conservation.
Roadless and Low-Traffic Areas as Conservation Targets in Europe – Nuria Selva • Stefan Kreft • Vassiliki Kati • Martin Schluck • Bengt-Gunnar Jonsson • Barbara Mihok • Henryk Okarma • Pierre L. Ibisch	2011	Europe	With increasing road encroachment, habitat fragmentation by transport infrastructures has been a serious threat for European biodiversity. Areas with no roads or little traffic (‘‘roadless and low-traffic areas’’) represent relatively undisturbed natural habitats and functioning ecosystems. They provide many benefits for biodiversity and human societies (e.g., landscape connectivity, barrier against pests and invasions, ecosystem services). Roadless and low-traffic areas, with a lower level of anthropogenic disturbances, are of special relevance in Europe because of their rarity and, in the context of climate change, because of their contribution to higher resilience and buffering capacity within landscape ecosystems. We propose that the few remaining roadless and low-traffic areas in Europe should be an important focus of conservation efforts; they should be urgently inventoried, included more explicitly in the law and accounted for in transport and urban planning. Considering them as complementary conservation targets would represent a concrete step towards the strengthening and adaptation of the Natura 2000 network to climate change.
A review of environmental impacts of winter road maintenance - Hrefna Run Vignisdottir, Babak Ebrahimi, Gaylord Kabongo Booto, Reyn O'Born, Helge Brattebø, Holger	2018	Cold Climates	The need for winter road maintenance (WRM) is changing in cold regions due to climate change. How the different modes of WRM will contribute to future overall emissions from infrastructure is therefore of great interest to road owners with a view to a more sustainable, low-carbon future. In the quest for near-zero-emissions transport, all aspects of the transport sector need to be accounted for in the search for possible mitigation of emissions. This study used 35 peer-reviewed articles published between 2000 and 2018 to map available information on the environmental impacts and effect of WRM and reveal any research gaps. The articles were categorized according to their research theme and focus. They were

Wallbaum, Rolf André Bohne			found to focus mainly on the local effects of WRM with emphasis on effects on water. Of the reviewed works, 27 contain information related to the environmental effects on a local level while five focused on global impact, which was mainly caused by fuel consumption. Only two articles took a holistic look at the system to identify emission sources and the effectiveness of possible changes in operations methods or material selection. Furthermore, a life-cycle approach could reveal ways to mitigate emissions through effectively comparing possible changes in the system without shifting the problem to other aspects of road transport.
Land use and climate change impacts on lake sedimentation rates in western Canada - Erik Schiefer, Ellen L. Petticrew, Richard Immell, Marwan A. Hassan, Derek L. Sonderegger	2014	Western Canada	Although sedimentation was highly variable, increasing trends in accumulation corresponded with cumulative land use and, to a lesser degree, with climate change. Road density was the most important variable, but the inclusion of timber harvesting density further improved model fits significantly.
Biodiversity, roads, & landscape fragmentation: Two Mediterranean cases - Matteo Marcantonio, Duccio Rocchini, Francesco Geri, Giovanni Bacaro, Valerio Amici	2013	Mediterranean	The most pervasive threats to biological diversity are directly or indirectly linked to the road networks. For this reason, over the last few decades, interest in the study of the ecological characteristics of the edges associated with roads has increased. Our findings indicated a clear relationship between road distance and different plant biodiversity facets, which showed its maximum effect in the first 0–20 m forest-to-road segment and a mitigation after the 200 m threshold. The few remnants of core forest habitats in the Mediterranean basin highlight the need to recognize that road construction and maintenance have several ecological implications and accordingly require long-term monitoring programs.
Does the effect of forest roads extend a few	2010	France	We studied the effect of forest road distance on plant understory diversity at 20 sites in young and adult oak stands in a French lowland forest with a long history of

meters or more into the adjacent forest? A study on understory plant diversity in managed oak stands - Catherine Avon, Laurent Bergès Yann Dumas Jean-Luc Dupouey			management and road construction. Even if the depth of forest road effect measured in lowland managed stands was narrow, building of a new forest road has non-negligible effects on plant population dynamics. Forest managers should take into account the impacts of roads on biodiversity, since the expected intensification of silviculture in response to global changes is set to accentuate the effect of forest roads.
A Global Strategy for Road Building - William F. Laurance, Gopalasamy Reuben Clements, Sean Sloan, Christine S. O'Connell, Nathan D. Mueller, Miriam Goosem, Oscar Venter, David P. Edwards, Ben Phalan, Andrew Balmford, Rodney Van Der Ree & Irene Burgues Arrea	2014	Global	The number and extent of roads will expand dramatically this century. Globally, at least 25 million kilometres of new roads are anticipated by 2050; a 60% increase in the total length of roads over that in 2010. Nine-tenths of all road construction is expected to occur in developing nations, including many regions that sustain exceptional biodiversity and vital ecosystem services. Roads penetrating into wilderness or frontier areas are a major proximate driver of habitat loss and fragmentation, wildfires, overhunting and other environmental degradation, often with irreversible impacts on ecosystems.
Estimating Diesel Fuel Consumption and Carbon Dioxide Emissions from Forest Road Construction - Dan Loeffler, Greg Jones,	2008	General	Forest access road construction is a necessary component of many on-the-ground forest vegetation treatment projects. However, the fuel energy requirements and associated carbon dioxide emissions from forest road construction are unknown. We present a method for estimating diesel fuel consumed and related carbon dioxide emissions from constructing forest roads using published results from a study designed to measure road construction costs together with machine productivity and fuel consumption rates. Our resulting estimate of diesel

Nikolaus Vonessen, Sean Healey, Woodam Chung			fuel required per mile of road constructed on slopes up to 50% using a cut-fill construction method is 590 gallons, with 13,400 pounds of carbon dioxide emitted per mile of road built. Using a full bench road construction method on slopes greater than 50% where volume of material handled and moved is very sensitive to hill slope and soil type, we estimated between 3,265 and 8,000 gallons of diesel fuel are required per mile of road emitting between 74,400 to 182,700 pounds of carbon dioxide.
Biodiversity Impact Assessment of roads: an approach based on ecosystem rarity - David Geneletti	2003	General	A sound Biodiversity Impact Assessment (BIA) in road planning and development needs to be coupled to other commonly considered aspects. This paper presents an approach to contribute to BIA of road projects that focuses on one type of impact: the direct loss of ecosystems. The first step consists in mapping the different ecosystem types, and in evaluating their relevance for biodiversity conservation. This is based on the assessment of ecosystem's rarity. Rarity is a measure of how frequently an ecosystem type is found within a given area. Its relevance is confirmed by the fact that the protection of rare ecosystems is often considered as the single most important function of biodiversity conservation. Subsequently, the impact of a road project can be quantified by spatially computing the expected losses of each ecosystem type. To illustrate the applicability of the methodology, a case study is presented dealing with the assessment of alternative routes for a highway development in northern Italy.

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Multifunctionality and biodiversity: Ecosystem services in temperate rainforests of the Pacific Northwest, USA

Patric Brandt [a,b,*], David J. Abson [c], Dominick A. DellaSala [d], Robert Feller [b], Henrik von Wehrden [b,c]

[a] Karlsruhe Institute of Technology, Institute of Meteorology and Climate - Atmospheric Environmental Research, Kreuzeckbahnstraße 19, 82467 Garmisch-Partenkirchen, Germany
[b] Leuphana University Lüneburg, Centre for Methods & Institute of Ecology, Faculty of Sustainability, Scharnhorststr. 1, 21335 Lüneburg, Germany
[c] Leuphana University Lüneburg, FuturES Research Center, Scharnhorststr. 1, 21335 Lüneburg, Germany
[d] Geos Institute, 84-4th St., Ashland, OR 97520, USA

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ABSTRACT

Forests produce a myriad of ecosystem related benefits known as ecosystem services. Maximizing the provision of single goods may lead to the overexploitation of ecosystems that negatively affects biodiversity and causes ecosystem degradation. We analyzed the temperate rainforest region of the Pacific Northwest, which offers a multitude of ecosystem services and harbors unique biodiversity, to investigate linkages and trade-offs between ecosystem services and biodiversity. We mapped nine actual and potential ecosystem services, grouped into provision, supporting, regulating and cultural ecosystem service categories, as well as species richness of four taxonomic groups (mammals, birds, trees, and amphibians). We analyzed linkages and tradeoffs between ecosystem services, their overall diversity, and species richness as well as different levels of taxon diversity. We also tested if ecosystem service categories, in addition to climate and land cover parameters, could indicate species richness. We found significant positive linkages between ecosystem service diversity and species richness of all considered taxa. The provision of the majority of ecosystem services was higher in areas of high taxon diversity, indicating both positive relationships and slight trade-offs in maximizing single ecosystem services. In general, ecosystem service categories were a comparable indicator of species richness as climate. Our findings show that multifunctionality largely coincides with high levels of biodiversity within the study region. Hence, an integrative ecosystem management approach that incorporates ecosystem services and biodiversity concerns is needed to both provide diverse ecosystem benefits and conserve biological diversity.

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1. Introduction

Ecosystem services and biodiversity conservation have become the two dominant, and potentially conflicting (Bullock et al., 2011; Marrs et al., 2007; McShane et al., 2011) management aims in conservation science during the last decades. Ecosystem services are the numerous benefits people directly or indirectly appropriate from the functioning of ecological systems and provide the foundations for human well-being (Daily, 1997; Nelson et al., 2009). The ecosystem services concept combines resource use, ecosystem management – including adaptation to impacts of driving forces such as land use and climate change – and the valuation of nature (Maskell et al., 2013), making it a key concept that bridges social and ecological systems (Carpenter et al., 2009). Biodiversity is vital for maintaining ecosystem processes and functioning (Duffy, 2009; Hector and Bagchi, 2007). Its loss has been shown to cause ecosystem degradation (Hooper et al., 2012). Hence, biodiversity is seen as essential requirement for the provisioning of ecosystem services (Diaz et al., 2006). Here it should be noted that as well as an instrumental value related to the provision of ecosystem services, the conservation of biodiversity is also a normative goal in its own right (Mace et al., 2012). Biodiversity conservation is therefore not solely contingent on the instrumental contribution to human well-being it may provide.

The increasing number of studies on the functional relationships between biodiversity and ecosystem services reveal mostly positive patterns (Gamfeldt et al., 2013; Hector and Bagchi, 2007; Maskell et al., 2013). However, many of these diversity-ecosystem services studies focus on a single facet of diversity such as one species group and a single ecosystem service, such as primary productivity (Costanza et al., 2007), pest control (Simon et al., 2010) or agricultural yields (Di Falco and Chavas, 2006). Managing an ecosystem for a single ecosystem service is potentially problematic as it may result in trade-offs in terms of associated biodiversity (Ingram et al., 2012; Ridder, 2008; Rodriguez et al., 2006) and thereby compromises conservation efforts. The interplay between the provision of multiple ecosystem services and biodiversity

* Corresponding author. Tel.: +49 4131 677 1571.
E-mail address: patricbrandt@gmx.de (P. Brandt).

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represents an important knowledge gap (Geijzendorffer and Roche, 2013; Sircely and Naeem, 2012), potentially limiting our ability to effectively manage multifunctional landscapes for both ecosystem services provision and biodiversity conservation.

In this regard, it becomes vital to analyze ecosystems that are managed for diverse societal needs. Multifunctional ecosystems offer several services simultaneously to satisfy social, cultural, economic and environmental demands (O'Farrell et al., 2010). Hence, a diverse set of ecosystem services needs to be considered when assessing the relations between biodiversity conservation and ecosystem service provision in multifunctional ecosystems (Chan et al., 2006; Tallis and Polasky, 2009). This includes services that cannot be straightforwardly linked to specific ecosystem functions such as cultural services (Hernandez-Morcillo et al., 2013). Potential synergies between ecosystem services and biodiversity are expected, though they might vary across ecosystems and depend on the specific ecosystem services and aspects of biodiversity taken into consideration (Mace et al., 2012).

Forests are of immense global importance in delivering a myriad of benefits to humanity (Bonan, 2008; FAO, 2010; Schwenk et al., 2012). In particular, temperate rainforests represent an ecologically complex, unique ecosystem with high biodiversity importance, subjected to multiple human demands. We analyzed a region along the Pacific coastline of North America harboring the world's largest remaining extents of temperate rainforests (DellaSala, 2011). While currently offering a broad range of goods and services such as salmon (*Oncorhynchus* spp.), timber, water regulation and recreation, these rainforests are threatened by climate and land use changes (DellaSala, 2011; Fitzgerald et al., 2011).

We addressed three key questions to investigate linkages and trade-offs between ecosystem services and biodiversity across the temperate rainforest region of the Pacific Northwest: (1) How is ecosystem service diversity related to species richness across different taxonomic groups? (2) How are ecosystem services and their diversity linked to different diversity levels of the considered taxa? (3) In order to untangle the interrelations among the environment, ecosystem services and species richness we tested if the provision of ecosystem services, grouped by the millennium ecosystem service assessment (MA) categories, alongside environmental variables such as climate and land cover, indicate species richness. Here, we did not seek to explain the functional relations between biodiversity and ecosystem services categories. Rather, we described the patterns (Shmueli, 2010) between the types of ecosystem services provided, their diversity and biodiversity across the temperate rainforest region of the Pacific Northwest and discussed the implications of these patterns for multifunctional landscape management and conservation at a regional scale.

Recent studies focusing on the relationship between ecosystem services and biodiversity have taken a functional perspective and mostly considered limited ecosystem service categories such as provisioning or regulating services (e.g. Balvanera et al., 2006; Costanza et al., 2007; Schwenk et al., 2012) and single species groups such as plant species (e.g. Gamfeldt et al., 2013; Maskell et al., 2013; Tilman et al., 2012). By involving multiple taxa and multiple ecosystem services including supporting and cultural services, we aim to identify more comprehensive patterns relating ecosystem services to biodiversity.

Here we note that the direct quantification of ecosystem services is often problematic and that there might be considerable differences between the ecosystem services that potentially flow from a given ecosystem and the actual services that are appropriated at a given point in time. For example, timber harvest is an indicator for the appropriation of timber but provides limited information regarding the capacity of a given system to sustainably provide timber. Similarly it can be argued that benefits received (i.e. the direct quantification of services) from physically appropriated goods such as timber must be related to how those physical goods contribute to human well-being (Fischer et al., 2009). Given the importance of both the actual appropriation and the potential capacity to supply ecosystem services and the difficulty in directly and accurately quantifying multiple ecosystem services across large spatial and temporal extents, we focus on the mapping of proxy datasets that indicate nine important potential and actual ecosystem services within the temperate rainforest region of the Pacific Northwest. The following proxy data for ecosystem services were modeled: timber harvest, salmon abundance, deer hunting, net primary productivity, carbon storage in vegetation, organic matter in soil, forest importance for drinking water supply, landscape aesthetics, and park visitation. These proxies for ecosystem services were grouped into the MA categories of provision, supporting, regulating, and cultural ecosystem services (MA, 2005). Biodiversity was quantified in terms of spatially explicit species richness data for higher taxa, including mammals, birds, trees and amphibians. Diversity metrics were derived for ecosystem services and the higher taxa. Subsequently, we computed univariate models to reveal the patterns between ecosystem service diversity and species richness. Potential and actual ecosystem services and their diversity were linked to the higher taxon diversity. Multivariate direct gradient analyses were performed to assess if the MA ecosystem service categories are able to indicate species richness in interaction with and untangled from environmental variables such as climate and land cover.

2. Materials and methods

2.1. Study area

The study area was based on the original coastal temperate rainforest extent of the Pacific Northwest region (DellaSala, 2011) that shows an overall high proportion of forest coverage. All US counties that intersect the original coastal rainforest extent, including a buffer of 15 km, were incorporated into the study area of 325,614 km^2. This broad extent was chosen to ensure that climate and land cover gradients are well represented. Due to limited data availability, coastal rainforest regions located in British Columbia and Alaska were excluded from our analyses. All metrics related to species richness, ecosystem services and environmental data were mapped at a resolution of approximately 8×8 km – 3997 grid cells in total. It is important to note that the study extent, while dominated by forests, encompasses a spatially heterogeneous matrix of different land uses that in turn create spatially heterogeneous patterns of ecosystem service provision and biodiversity. The study extent comprised 55% forest, 33% scrub- and grassland, 7% cultivated areas and 5% developed/urban regions. Public lands in this region are managed under the Northwest Forest Plan that governs ecosystem management and biodiversity conservation (DellaSala and Williams, 2006). However, non-federal landowners frequently focus on timber management as the primary ecosystem service. ARCGIS 10.1 was used for all geo-processing work.

2.2. Ecosystem service data

The proxy data used refer either to the actual goods or services people appropriate from nature, known as 'ecosystem services', or to the capacity of the ecosystem to deliver those goods and services to society, conceptualized here as 'potential ecosystem services' (Vira and Adams, 2009). The data were based on physical occurrence of actual and potential ecosystem services, rather than the monetary or non-monetary values associated with those services.

All data were gathered from publically available datasets, further processed and linked to spatial data or were readily available in a spatially explicit format. We compiled GIS-layers indicating the nine potential and actual ecosystem services, which were also grouped into MA categories – i.e. provisioning, regulating, supporting, and cultural ecosystem services (MA, 2005). Detailed descriptions of the datasets and data sources can be found in the online appendix (Online appendix, Table A1).

2.2.1. Provisioning services

2.2.1.1. Timber harvest. Timber is one of the most prominent resources derived from forest ecosystems and has been intensively harvested from temperate rainforests in this region. This layer depicts the total volume of timber harvested in 2010 measured in thousands of board feet. Derived tabular data are based on the county level.

2.2.1.2. Salmon abundance. Salmon are an important economic and food resource for the entire coastal rainforest region in North America and they are the key for trophic dynamics and energy transfer (DellaSala, 2011). The salmon abundance data are based on observed (1998–2005) and modeled data at the watershed level (Pinsky et al., 2009). Watershed based data were normalized and then converted into gridded data.

2.2.1.3. Deer hunting. Hunting has been taking place for millennia across the coastal temperate rainforest region of North America (Schoonmaker et al., 1997). Hence, hunting can be considered as a traditional source of local food resources. This layer indicates overall deer hunting successes for 2010 measured in counted deer kills. Census data are obtained and mapped based on hunting management units defined by State Departments of Fish & Wildlife.

2.2.2. Supporting services

2.2.2.1. Net primary productivity. The Pacific coastal rainforests belong to the most productive ecosystems worldwide (DellaSala, 2011), and primary productivity is a key ecological function from which many other, directly used, ecosystem services flow. Gridded information on NPP is derived from NASA's MODIS satellite data in a 10 km^2 grid cell resolution based on monthly values averaged for 2010 in gC m^{-2} day^{-1}.

2.2.3. Regulating services

2.2.3.1. Carbon storage in vegetation. The storage of atmospheric carbon in biomass is essential to climate regulation and climate change mitigation. The coastal temperate rainforests in North America show high carbon densities compared to other forest ecosystems (DellaSala, 2011; Woodbury et al., 2007). Gridded data show total mean carbon content in vegetation for 1961–1990 and originate from outputs of the MC1 dynamic vegetation model (Bachelet et al., 2001a,b) in an 8.8 km^2 grid cell resolution.

2.2.3.2. Organic matter in soil. Organic matter strongly influences soil properties such as water retention, erodibility and fertility (Ontl and Schulte, 2012). Furthermore, soil represents a large carbon pool. The sequestration of atmospheric carbon in soil organic matter contributes to climate change mitigation (Lal, 2004). The data used indicates the total content of organic matter in soil expressed as percent by weight of the 2 mm soil fraction at the watershed level.

2.2.3.3. Forest importance for drinking water supply. Forests are known to serve as important regulators of drinking water, particularly in this region (DellaSala et al., 2011). This layer combines precipitation intensity, proportion of forests and population density per watershed and was derived from the USDA 'forests to faucets' dataset (Barnes et al., 2009). We used these data as proxy for water regulation (Todd and Weidner, 2010).

2.2.4. Cultural services

2.2.4.1. Landscape aesthetics. The possibility to experience landscapes that are largely undisturbed by human pressure is usually accepted as a great benefit that ecological systems may offer in terms of recreation (Gobster et al., 2007). The compiled dataset consists of several spatial layers related to infrastructure such as roads, railroads and settlements, and natural elements such as lakes, rivers and forests that are undisturbed by human influences. All layers were weighted according to their naturalness. Terrain roughness was incorporated as proxy for physical landscape heterogeneity. Each layer was weighted either positively or negatively except for terrain roughness that was weighted based on three states, low roughness as negative, medium roughness as neutral and high roughness as positive. The resulted 'landscape aesthetics' layer is considered as a potential ecosystem service since the quantification of the actual cultural values associated with the landscapes of the study region was beyond the scope of our analysis.

2.2.4.2. Park visitation. State and national parks represent essential recreation areas in the US (Daniel et al., 2012), facilitating environmental education and sustainable tourism. We mapped the tabular park visitation data for 2010 on state and national parks, derived from the PAD-US protected area database (v. 1.2). Subsequently spatial data were aggregated on county level since most of the state parks do not match the working resolution and hence would not have been visible for the analyses. We used this dataset as proxy for the provision of space for recreation and cultural experiences.

All data based on unequally sized areas were normalized based on area. Thus, every layer refers to equal area units. For further analyses all potential and actual ecosystem service layers were transformed to a standardized scale based on their maximum values (Raudsepp-Hearne et al., 2010). Hence, all ecosystem service values range between 0 and 1.

2.3. Species data

Spatially explicit species richness data for higher taxa, including mammals (between 1 and 85 species recorded), birds (88–223 species), trees (1–50 species) and amphibians (2–38 species) were obtained as gridded layer from several resources (Online appendix, Table A2). For tree species, we compiled a richness layer through aggregating range polygon data (Little, 1978). Selected species groups represent major parts of the overall species diversity that exists across the Pacific coastal temperate rainforests and contain numerous species of economic, cultural and conservation importance. Reptiles, as a further terrestrial vertebrate group, were not included into the analyses since they are not well represented, nor particularly abundant, across the Pacific coastal temperate rainforests compared to other regions of their occurrence (Böhm et al., 2013).

2.4. Applied statistical approaches

All statistical analyses were undertaken using R 2.15, including the packages 'raster' (v. 2.1.12) for handling spatial data, 'car' (v. 2.0.16) for building generalized linear models (GLMs), 'spdep' (v. 0.5.56) for correcting autocorrelation patterns, 'vegan' (v. 2.0.6) to obtain diversity indices and to perform principal component analyses (PCAs) as well as redundancy analyses (RDAs).

2.4.1. Simpson diversity metrics

Diversity metrics were derived by using the Simpson diversity index for potential and actual ecosystem services (Raudsepp-Hearne et al., 2010) and higher taxa ranging between 0 (low diversity) and 1 (high diversity). The Simpson index is illustrated by the following formula:

$$D = 1 - \sum_{i=1}^{R} p^2 i$$

R is the richness of taxa/ecosystem services and p_i is the proportion of abundances for the ith taxon/ecosystem service.

The Simpson diversity measure takes abundances into account and equals the probability that two entities taken at random from the dataset represent the same type (Simpson, 1949). The Simpson diversity of higher taxonomic groups was used as biodiversity metric that is comparable to the Simpson diversity of ecosystem services. A color map was compiled illustrating the degree of spatial correspondence between the diversity metrics across the study area.

2.4.2. Univariate linkage modeling: ecosystem service diversity – species richness

In order to model the relationship between ecosystem service diversity and species richness, we chose a univariate model approach using GLMs (Crawley, 2007). Due to the non-normalized distribution of model residuals, we opted for GLMs with Poisson error structure. Species richness data were selected as dependent variables and ecosystem service diversity as independent variable since Poisson-GLMs require real count data. Hence, we follow a descriptive approach rather than explaining the causal relationship between biodiversity and ecosystem services. GLMs also included quadratic terms and were reduced based on the Akaike information criterion (AIC), to avoid overfitting (Sakamoto et al., 1986). Since model residuals revealed patterns of spatial autocorrelation, we applied spatial eigenvector filtering to incorporate spatial autocorrelation structures (Dray et al., 2006; Griffith and Peres-Neto, 2006). Spatial eigenvectors are derived from a neighborhood matrix spanning a distance of 100 km, which was chosen due to highest spatial autocorrelation values within that distance. The number of incorporated spatial eigenvectors was based on Moran's I significance values for each GLM. Eigenvectors were included until they exceeded a significant Moran's I value ($p < 0.05$).

2.4.3. Ordination techniques: ecosystem service categories, land cover and climate – species richness

A multivariate direct gradient analysis was applied to investigate the proportion of species richness variances captured by potential as well as actual ecosystem services grouped into the MA categories and environmental variables. Initial analyses of data distribution and gradient lengths showed that linear models are a correct general assumption for our data. Hence we used a PCA to reduce multicollinearity inherent to the climatic parameters applied and a partially constrained RDA as overall multivariate model (Legendre and Legendre, 2012) to partition the explained variance of four different variable groups, such as potential and actual ecosystem services for each MA category, 'climate', 'land cover' and 'geography'. Species richness data from the considered taxonomic groups served as response variables and were subjected to Hellinger transformations as proposed for analyzing heterogeneous community datasets (Legendre and Gallagher, 2001). Climatic data were derived by performing a PCA, including 19 BIOCLIM variables that were obtained as downscaled spatial grids in a 2.5 arc-min resolution (Hijmans et al., 2005). The PCA scores from the first two principal components were extracted and subsequently used as 'climate' variable group for the RDAs (Hanspach et al., 2011). Land cover data were derived from the USGS land use survey 2006 comprising 16 land cover classes, including developed, forested, cultivated, wetland, herbaceous, scrubland and barren land cover types at a grid cell resolution of 30 meters. The original dataset was spatially downscaled to match the working resolution. To account for spatial autocorrelation effects, we defined latitude and longitude as a further variable group named 'geography'.

3. Results

3.1. Ecosystem service diversity and species richness

The compiled spatial layers of potential as well as actual ecosystem services, the derived Simpson diversity metrics of the considered taxa and ecosystem services varied across the study area (Fig. 1). The diversity of taxa and the diversity of ecosystem services were highly correlated, indicated by Spearman's rho = 0.719 ($p < 0.001$). Species richness maps for mammals, birds, trees and amphibians are shown in online appendix (Fig. A1).

Ecosystem service diversity showed significant positive interactions with the richness of mammal, bird, tree, and amphibian

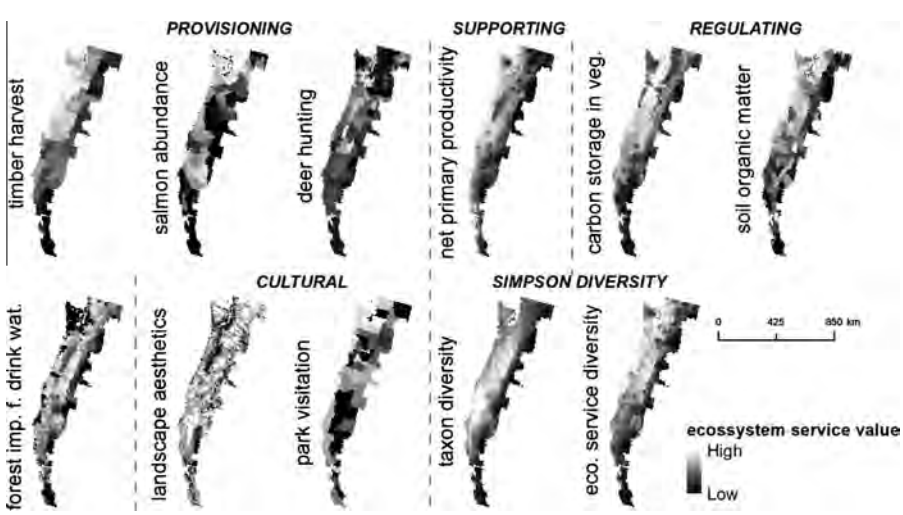

Fig. 1. Spatial distribution of nine potential and actual ecosystem services as well as the Simpson diversity of considered taxa and ecosystem services across the coastal temperate rainforest region of the Pacific Northwest, USA.

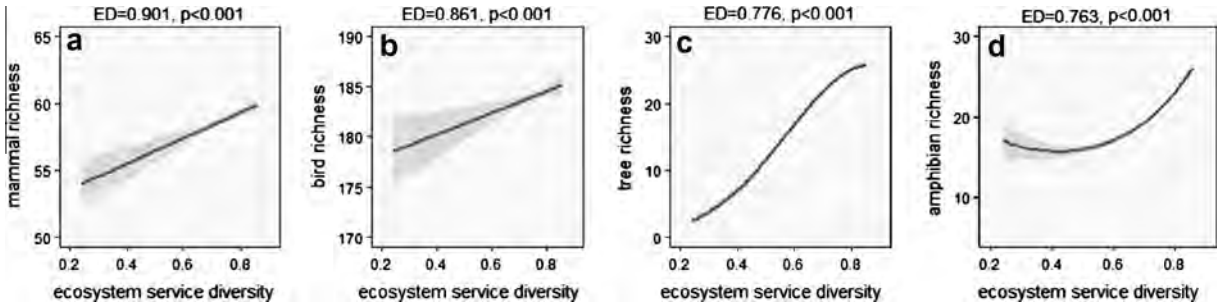

Fig. 2. GLM results for linkages between ecosystem service diversity and mammal, bird, tree, and amphibian richness. Incorporated spatial eigenvectors were kept at mean level. Light gray areas indicate the 95% confidence intervals of prediction errors (ED = explained deviance).

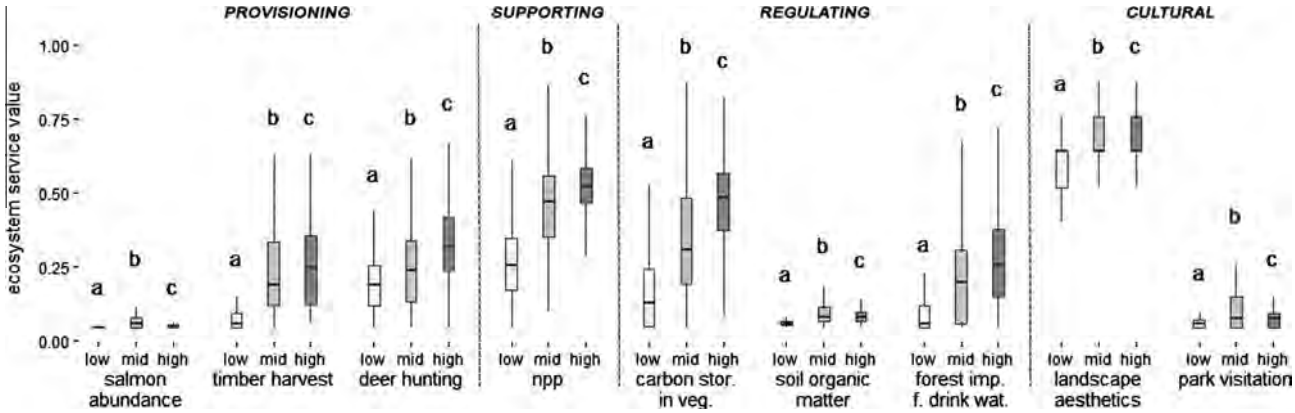

Fig. 3. Potential and actual ecosystem services for three levels of overall Simpson diversity of considered taxa (low = 0.335–0.514, mid = 0.514–0.573, high = 0.573–0.634). Grouping maintained equal sample sizes within each level. A Wilcoxon rank sum test was performed to assess the mean differences between Simpson diversity levels ($p < 0.001$). *P*-values were Bonferroni corrected to account for multiple testing.

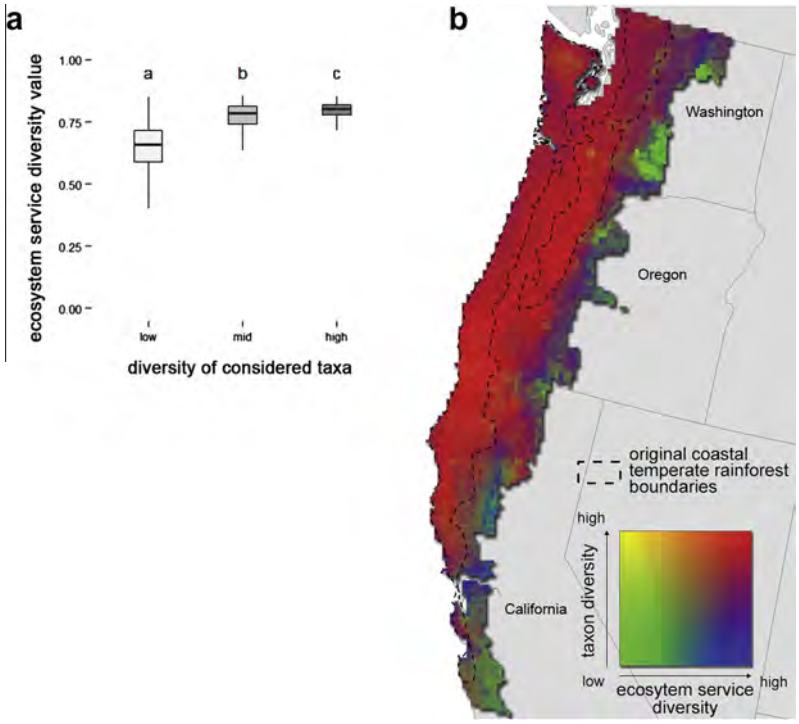

Fig. 4. (a) Ecosystem service diversity for three levels of overall Simpson diversity of the four considered taxonomic groups (low = 0.335–0.514, mid = 0.514–0.573, high = 0.573–0.634). Grouping maintained equal sample sizes within each level. A Wilcoxon rank sum test was performed to assess the mean differences between Simpson diversity levels ($p < 0.001$). *P*-values were Bonferroni corrected to account for multiple testing. (b) Spatial correspondence between ecosystem service diversity and diversity of considered taxa. Mapped pixel colors were assigned based on a RGB color space defined by ecosystem service diversity on the *x*-axis and diversity of four considered taxa on the *y*-axis.

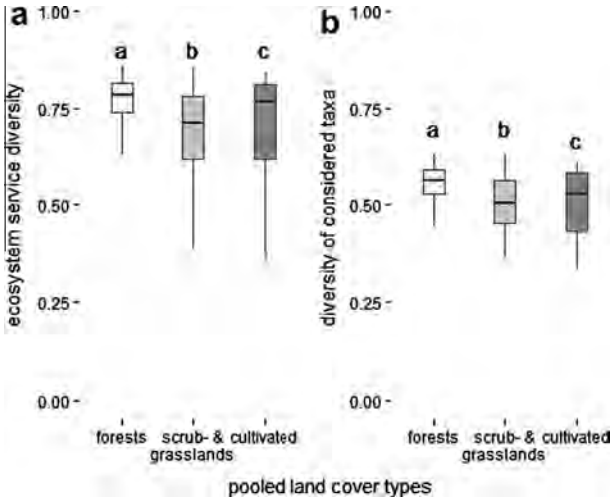

Fig. 5. Ecosystem service diversity (a) and diversity of considered taxa (b) for three pooled land cover types: forests, scrub- and grasslands, and cultivated areas. Developed land cover types were excluded due to minor relevance for the provisioning of analyzed ecosystem services.

species (Fig. 2, $p < 0.001$). Mammal and bird species richness increased linearly with higher ecosystem service diversity (Fig. 2a and b), tree richness showed a sigmoidal relationship indicating a saturation effect of tree species richness at the highest levels of ecosystem service diversity (Fig. 2c). Amphibian richness increased steeply with elevated ecosystem service diversity (Fig. 2d). The GLM on mammal richness had the highest model fit expressed as explained deviance (ED = 0.901), the model that considered amphibian richness the lowest (ED = 0.763).

3.2. Ecosystem services and taxon diversity

Higher values of ecosystem services were related to higher levels of the overall Simpson taxon diversity for most of the applied potential and actual ecosystem services across all MA categories (Fig. 3). However, salmon abundance, soil organic matter and park visitation differed from that pattern, indicating trade-offs between maxima of single ecosystem services and diversity of involved taxa. No pronounced ecosystem service gradient could be detected based on a PCA, including all modeled ecosystem services (not shown). The first two PCA axes together explained 46% of the overall variance.

Higher ecosystem service diversity was significantly linked to elevated taxon diversity (Fig. 4a, $p < 0.001$). However, less pronounced differences between medium and high levels of taxon diversity suggested a nonlinear relationship resulting in a saturation effect for ecosystem service diversity in areas of high taxon diversity. High spatial correspondence between ecosystem service diversity and the diversity of included taxa was shown within coastal temperate rainforest regions throughout most of the Pacific Northwest (Fig. 4b).

3.3. Ecosystem service categories as indicators for species richness

To assess both the distribution of ecosystem service diversity and the diversity of higher taxa for major land cover types, we pooled the detailed land cover types into three groups, namely, 'forests', 'scrub- and grasslands', and 'cultivated areas'. Highest diversity values for ecosystem services as well as considered taxa were significantly higher for forests (Fig. 5a and b, $p < 0.05$). Groups differed significantly as assessed through a one-way analysis of

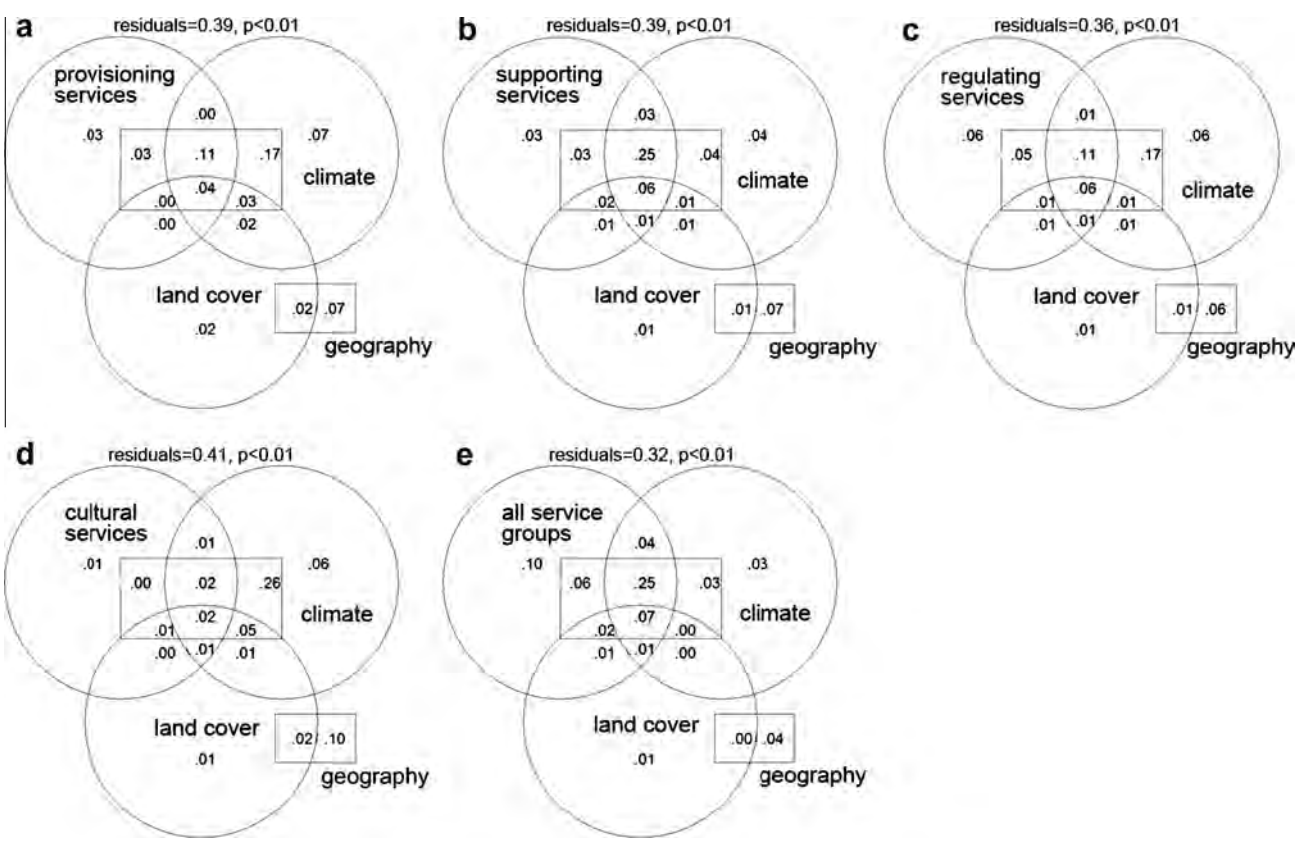

Fig. 6. RDA variance partitioning results for species richness including mammals, birds, trees and amphibians separately indicated by different ecosystem service categories: (a) provisioning, (b) supporting, (c) regulating, (d) cultural and (e) all potential and actual ecosystem services, climate, land cover (circles) and geography (rectangle). Displayed values show captured variances as adjusted R^2 for all single (non-overlapping parts of circles and the rectangle) and combined effects (overlapping parts of circles and the rectangle).

variance and a subsequent paired *t*-test ($p < 0.05$). P-values were Bonferroni corrected to account for multiple testing.

Constrained RDAs revealed that ecosystem service categories significantly indicated species richness in a comparable magnitude of land cover and climate (Fig. 6, $p < 0.001$). Among the RDAs that were fitted with single ecosystem service categories, the RDA with regulating services showed the lowest model error (Fig. 6c, residuals = 0.36) and the one that included cultural services the highest error (Fig. 6d, residuals = 0.41). The RDA incorporating the entire set of potential and actual ecosystem services as variable group showed the lowest model error among all RDAs (Fig. 6e, residuals = 0.32). However, for the majority of RDAs the climatic space was, after geography, the variable group that captured most of the species richness variances. This reflected both the prevailing climatic gradient that shapes diversity patterns across the temperate rainforests of the Pacific Northwest and an inherent autocorrelation pattern. Focusing on interactions between variable groups the climate-geography interactions showed the strongest effects followed by the ecosystem service category–climate interactions.

4. Discussion

4.1. Ecosystem service diversity and species richness

We found broad, positive relationships between ecosystem service diversity and species richness. Such a pattern confirms the findings of Egoh et al. (2009) who illustrated a spatial congruency between ecosystem services and biological diversity in South Africa. No trade-offs were observed in our study between ecosystem service diversity and species richness. Other studies reported both trade-offs and concordances between ecosystem service hotspots and biodiversity or its conservation (Chan et al., 2006; Turner et al., 2007). Our results clearly show that high levels of biodiversity are found in areas that provide diverse actual and potential ecosystem services across the coastal temperate rainforest region of the Pacific Northwest. This pattern was also apparent when all considered taxa were combined to one diversity index, particularly within the original coastal temperate rainforest boundaries. Though, some minor areas showed a contrasting pattern of low biodiversity but high ecosystem service diversity. These scattered areas were mostly distributed at the inland edges of our study region indicating transition zones to other ecosystems that might start to harbor different species inventories not included in our study.

Saturation effects were revealed for tree species and overall taxon diversity suggesting that further ecosystem service increases in regions of highly diverse ecosystem service provision coincide with marginally higher biodiversity levels. This might relate to redundancies of present species in terms of the necessary ecosystem functions that are required to maintain considered ecosystem services (Duffy, 2009; Hector and Bagchi, 2007). Notwithstanding, including more services and thus more ecosystem functions would probably incorporate more biodiversity needed to sustain these functions (Gamfeldt et al., 2008). Moreover, biodiversity reduces the vulnerability of ecosystems to disturbances, serving both as a backup for functional degradation and to ensure diverse and fast responses to perturbations hence improving overall ecosystem resilience (Mori et al., 2013).

4.2. Ecosystem services and their diversity for different levels of taxon diversity

The majority of our results indicate positive relationships between single ecosystem services included and the overall diversity of the considered taxa. Similar patterns are found in recent studies (Balvanera et al., 2006; Schneiders et al., 2012), in particular for productivity and biodiversity (Gamfeldt et al., 2013; Tilman et al., 2012) – though Costanza et al. (2007) found a temperature dependent relationship. The relation between timber harvest and taxon diversity was most surprising and probably, in part, resulted from a scale artifact inherent to the data used. It is important to note here that the established relationships do not imply causality. Yet, intense forest management is usually considered to have negative impacts on biodiversity (Bengtsson et al., 2000). The data used in our study did not include any information on how the forests are managed for timber harvest on a local scale. Hence, it is beyond the scope of our analyses to assess the effects of forest practices on biodiversity patterns.

Despite the largely positive patterns found, a few trade-offs were noticeable in our results. Salmon abundance, soil organic matter and park visitation were highest in areas with moderate levels of taxon diversity. Non-supporting patterns or trade-offs among ecosystem services are postulated (Bennett et al., 2009) and reported on a regional (Raudsepp-Hearne et al., 2010), continental (Haines-Young et al., 2012) and global scale (Naidoo et al., 2008). Though, in our case, no clear trade-offs among the studied ecosystem services could be detected.

4.3. Ecosystem service categories as indicators for species richness

Both, ecosystem service diversity and taxon diversity were highest in forested extents within the study region. Although the differences among land cover types were only marginal, it suggests that forests provide conditions most suited for supplying ecosystem services and biodiversity.

Using a multivariate approach, including the nine actual and potential ecosystem services grouped into MA categories as well as environmental variables showed that the different MA ecosystem service categories indicated species richness of the four different taxa in a comparable magnitude to climate and land cover. The capability of all ecosystem service categories, in interaction with climate, to indicate species richness illustrates that a management focusing on multiple ecosystem-based benefits and the current climatic conditions are synergistic for both ecosystem services and biodiversity. These findings support the idea that the ecosystem service approach could be used to monitor and manage biodiversity (Egoh et al., 2009). However, cultural services showed an overall weak link, probably due to the most indirect relationship to richness for instance compared to the considered regulating ecosystem services. Nevertheless, the management of ecosystems based on providing a diversity of ecosystem services might have co-benefits in terms of biodiversity conservation.

4.4. Ecosystem service approach and multifunctional ecosystems

Temperate rainforests of the Pacific Northwest simultaneously offer a multitude of ecosystem-based benefits. We were able to show that such a multifunctional ecosystem might serve as indicator of biodiversity and its conservation while delivering important goods and services to society. Our results are restricted to one region and spatial scale as well as one point in time. Thus, extrapolating these results to other regions featuring different ecosystem properties and species should be done with considerable caution. However, high biodiversity levels in multifunctional landscapes also have been shown before for areas with heterogeneous land use or agricultural regions (O'Farrell et al., 2010; Schneiders et al., 2012; Sircely and Naeem, 2012). Managing for multiple ecosystem services may also create conditions for higher levels of biodiversity. Given the co-occurrence of biodiversity and diverse ecosystem service provision, we suggest that biodiversity conservation should be integrated into the management of

multifunctional ecosystems and not only take place in areas explicitly designated for conservation.

4.5. Threats to ecosystem services and biodiversity: land use and climate change

North America's temperate rainforests are fragmented by logging, road building, and other human disturbances (DellaSala, 2011). Coinciding biodiversity loss and the degradation of ecosystem functions are expected due to habitat fragmentation and increasing land use intensity (Foley et al., 2005). However, sustainable trajectories of land use changes and restoration efforts have been positively linked to ecosystem service provision and biodiversity conservation (Nelson et al., 2009; Rey Benayas et al., 2009).

Ecosystem services in this region are threatened by a changing climate regime and projected vegetation shifts in Western North America (Wang et al., 2012). Dominant tree species and vegetation types in our study area are predicted to shift substantially until the end of the 21st century (Coops and Waring, 2011; Gonzalez et al., 2010; McKenney et al., 2007), probably detrimentally affecting both current ecosystem service and biodiversity patterns. Hence, an adaptive ecosystem management approach is needed to mitigate estimated impacts.

5. Conclusions

Our results confirm that multifunctional landscapes, here largely covered by temperate rainforests, co-occur with high levels of biodiversity. Thus, the management of ecosystem services should not substitute, but rather incorporate, biodiversity conservation since the two concepts are interdependently related through maintaining the functioning of ecosystems on the one hand and the management for goods and services on the other hand (Ingram et al., 2012; Mace et al., 2012). Based on our results, we derive the following management and research recommendations for the coastal temperate rainforest region across the Pacific Northwest.

1. The concepts of ecosystem services and biodiversity are not only linked, they act in concert. Based on our analysis, an integrative approach of ecosystem management that incorporates both ecosystem services and biodiversity is indeed beneficial in providing goods and services to society while maintaining biodiversity. We therefore support the perspective that multifunctional ecosystems should become a key for sustainable ecosystem management in this region, particularly in a way that optimizes land-use and strives for compatibility in management among different ecosystem services.
2. Our findings generally show that land managers who are interested in the provisioning of diverse ecosystem services are also able to maintain biodiversity. For instance, large landscape level management efforts inherent to the Northwest Forest Plan (DellaSala and Williams, 2006) represent approaches in which ecosystem management and biodiversity conservation on public lands are capable to produce multiple ecosystem benefits and, hence, help to maintain multifunctionality.
3. Research at finer spatial scales, incorporating time series data and information on how local forest management practices determine possible relationships between timber harvest, other ecosystem services and biodiversity would be useful for our study region. Standardized surveys and sampling protocols are required and data on socio-economical dynamics and ecosystem service valuations should be linked to assess the compatibility of (potentially competing) provisioning ecosystem services at the local and regional scale. Scenario driven analyses (Carpenter et al., 2006; Nelson et al., 2009) that consider climate and land-use changes are necessary since they may offer valuable insights about possible future trajectories of biodiversity and ecosystem service patterns in this region.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.biocon.2013.12.003.

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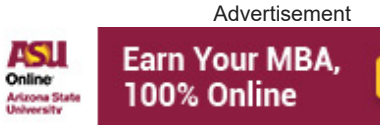

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Shifting habitat mosaics and fish production across river basins

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Abstract

Watersheds are complex mosaics of habitats whose conditions vary across space and time as landscape features filter overriding climate forcing, yet the extent to which the reliability of ecosystem services depends on these dynamics remains unknown. We quantified how shifting habitat mosaics are expressed across a

range of spatial scales within a large, free-flowing river, and how they stabilize the production of Pacific salmon that support valuable fisheries. The strontium isotope records of ear stones (otoliths) show that the relative productivity of locations across the river network, as both natal- and juvenile-rearing habitat, varies widely among years and that this variability is expressed across a broad range of spatial scales, ultimately stabilizing the interannual production of fish at the scale of the entire basin.

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... The opposite was true in the Little Colorado River, characterized by low mainstem hydrologic connectivity, where perennial streams had more unique species composition. Despite the fact that the strength of our inferences are limited by including just two basins for comparison, these findings support the long recog- nized value of complementarity in conservation planning (Brennan et al., 2019; Vane-Wright, Humphries, & Williams, 1991).Intermittent sites provide periodic increases in network con- nectivity, expanding habitat availability and facilitating dispersal within riverine metacommunities when flowing(Jaeger et al., 2014).Intermittent site contributions to beta diversity remained fairly sta- ble through time and were invariant to high or low departures from historical annual flow conditions. Because high flow years showed little association with the relative contributions of intermittent and perennial sites to beta diversity, species' dispersal and coloni- zation events may have played similar roles in perennial and inter- mittent sites, or else environmental conditions other than flow may have affected local species composition(Cadotte & Tucker, 2017).Alternatively, the annual temporal resolution of community surveys may have been insufficient to capture intra-annual metacommunity and beta diversity dynamics in

response to seasonal drying and re- wetting periodsRuhí et al., 2017). ...

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Effective planning of a large-scale restoration project is challenging, because of the range of factors that need to be considered (e.g. restoration of multiple habitats with varying degradation levels, multiple restoration goals and limited conservation resources). Ecological restoration planning studies typically focus on biodiversity and ecosystem services, rather than employment and other ... [Show full abstract]

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Global Change Biology

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Spatial and topographic trends in forest expansion and biomass change, from regional to local scales

BRIAN BUMA[1] and TARA M. BARRETT[2]

[1]*University of Alaska Southeast, 11120 Glacier Highway, Juneau, AK 99801, USA,* [2]*Pacific Northwest Research Station, USDA Forest Service, 1133 N. Western Ave., Wenatchee, WA 98801, USA*

Abstract

Natural forest growth and expansion are important carbon sequestration processes globally. Climate change is likely to increase forest growth in some regions via CO_2 fertilization, increased temperatures, and altered precipitation; however, altered disturbance regimes and climate stress (e.g. drought) will act to reduce carbon stocks in forests as well. Observations of asynchrony in forest change is useful in determining current trends in forest carbon stocks, both in terms of forest density (e.g. Mg ha^{-1}) and spatially (extent and location). Monitoring change in natural (unmanaged) areas is particularly useful, as while afforestation and recovery from historic land use are currently large carbon sinks, the long-term viability of those sinks depends on climate change and disturbance dynamics at their particular location. We utilize a large, unmanaged biome (>135 000 km^2) which spans a broad latitudinal gradient to explore how variation in location affects forest density and spatial patterning: the forests of the North American temperate rainforests in Alaska, which store >2.8 Pg C in biomass and soil, equivalent to >8% of the C in contiguous US forests. We demonstrate that the regional biome is shifting; gains exceed losses and are located in different spatio-topographic contexts. Forest gains are concentrated on northerly aspects, lower elevations, and higher latitudes, especially in sheltered areas, whereas loss is skewed toward southerly aspects and lower latitudes. Repeat plot-scale biomass data (n = 759) indicate that within-forest biomass gains outpace losses (live trees >12.7 cm diameter, 986 Gg yr^{-1}) on gentler slopes and in higher latitudes. This work demonstrates that while temperate rainforest dynamics occur at fine spatial scales (<1000 m^2), the net result of thousands of individual events is regionally patterned change. Correlations between the disturbance/establishment imbalance and biomass accumulation suggest the potential for relatively rapid biome shifts and biomass changes.

Keywords: climate change, disturbance, forest change, North America, temperate rainforest biome

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Introduction

Forests comprise a significant amount of the global carbon pool (861.1 ± 66.4 Pg) and are currently a significant carbon sink (1.11 ± 0.8 Pg yr^{-1}; Pan *et al.*, 2011a). This sequestered carbon is a critical part of the carbon cycle and potentially an important offset to anthropogenic emissions. Much of this can be attributed to afforestation (e.g. afforestation in China; Piao *et al.*, 2009) and shifting land use practices (e.g. recovery from historic land clearing; Birdsey *et al.*, 2006; Zhang *et al.*, 2012). Global climate change and increasing N deposition are also expected to alter forest biomass substantially (Pan *et al.*, 2011a), although the direction and magnitude of change are difficult to predict due to competing factors. While N deposition and carbon fertilization may increase growth rates in many locations (Hember *et al.*, 2012; Zhang *et al.*, 2012), increased evaporative demand, altered precipitation patterns, and altered disturbance regimes may decrease forest biomass in others (Van Mantgem *et al.*, 2009). The development of an improved understanding of forest dynamics and forest extent change is vital for supporting accurate assessments of changes in forest biomass and improving future monitoring efforts.

While the overall carbon balance in a forest is a function of afforestation, historic and current management, and natural dynamics, understanding patterns and trends in unmanaged (i.e. 'natural') forests are particularly important. Afforestation may serve to compliment natural forest biomass accumulation, but in some cases may mask a decline: For example, China has seen a net gain in forested carbon stocks via afforestation which mask a slight decline in natural forests (Fang *et al.*, 2001). Similarly, regrowth from historic logging in the United States (Pan *et al.*, 2011b) is currently offsetting a decline due to increasing disturbances in many western states (Zhang *et al.*, 2012). Predicting global and regional carbon balance in the future, understanding how afforestation efforts may fare in the future, and anticipating the sustainability of current efforts requires an improved understanding of how natural forests (e.g.

Correspondence: Brian Buma, tel. 1 907 796 6410, fax 1 907 796 6447, e-mail: brian.buma@uas.alaska.edu

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forests not managed historically or presently) respond to climate change.

Differences in the relative magnitude of forest disturbance, establishment, growth, and mortality is the mechanism by which unmanaged forest biomass and distributions will shift in response to a warming climate (Dale *et al*., 2001; Johnson & Miyanishi, 2010; Vanderwel & Purves, 2014). Identifying where ecosystem dynamics are in disequilibrium is a means to determining where change is being manifested on the landscape, its magnitude, and relative rate, and requires understanding the spatial distribution of disturbance and establishment events and field plots which can record the magnitude of change. In a hypothetical stable system, the spatio-topographic distribution and magnitude of those fine-scale disturbance/establishment events would be similar at appropriately broad spatial and temporal scales (large enough to encompass many individual events while still comparable in terms of disturbance drivers, e.g. climate, species composition, Johnson & Miyanishi, 2010), although this requires the assumption that enough observations of change can be made to adequately describe ongoing spatiotemporal dynamics for the system under study (see Materials and methods). Derivations from this hypothetical stable system should be observable as differences in the spatial and topographic distribution of disturbance and recovery events.

While the importance of forests as a carbon sink is well recognized, understanding how that carbon is changing in response to climate change at multiple scales (both in terms of forest extent and biomass density) is difficult. Here, we explore the relationships between latitudinal and topographic patterns of forest change (disturbance and recovery) and forest biomass change in a significant carbon storehouse, the North Pacific temperate rainforests. If similar drivers are controlling each process, then natural systems may respond rapidly to climate change (e.g. an increase in biomass density and an increase in extent in areas where the warming climate favors forests). If not, then forest carbon stock response to climate change may be attenuated (e.g. a warming climate favors denser forests but also more disturbance/less recovery). As the strength and direction of this relationship may vary by location, such an analysis must be carried out in a spatially explicit fashion. Here, we address the following questions:

1 Are spatial patterns of forest change similar to spatial patterns of biomass change?
2 Are the topographic distributions of forest change similar to the topographic distributions of biomass change?
3 How are the temperate rainforests of the North Pacific changing in terms of aboveground, live biomass, and where are those changes most significant? Do these changes correspond with previous hypotheses about the effects of climate change on this substantial carbon pool?

Materials and methods

Study region

Southeast Alaskan temperate rainforests provide the unique opportunity to look at a consistent system, in terms of species, climate, and disturbance regimes across a broad latitudinal gradient, which allows us to explore the relationship between disequilibrium in forest change and forest biomass changes. In addition, they are ecologically significant. Temperate rainforests contain the world's densest stocks of carbon (Leighty *et al*., 2006; Keith *et al*., 2009), a variety of important ecosystem services (Brandt *et al*., 2014), and a multitude of endemic species (Cook *et al*., 2001). They are also experiencing (and will experience) profound changes due to climatic warming (Meehl *et al*., 2004; Wolken *et al*., 2011; Orians & Schoen, 2013). Alaskan temperate rainforests contain an estimated >2.8 ± 0.5 Pg C in biomass and soil, equivalent to >8% of the C in all the forests of the contiguous United States (Leighty *et al*., 2006). Despite the broad latitudinal gradient, the climate and species composition are similar across the region, facilitating a regional analysis (DellaSala, 2011). Temperatures are mild and precipitation uniformly high (cool maritime, Köppen climate classification Cfb) due to the moderating oceanic influence. Species diversity is relatively low and composition is similar across the region as well, primarily *Picea sitchensis – Tsuga heterophylla* forests. *Pinus contorta* var. *contorta*, *Thuja plicata,* and *Cupressus nootkatensis* are found in boggy locations throughout the region, and *Alnus viridis* is often found in recently disturbed locations. Critically, disturbance drivers are consistent across the region: predominately fine-scale, non-episodic mortality events, primarily a function of small gaps forming through stem decay, wind, or snow damage (Ott & Juday, 2002; Hennon & McClellan, 2003). There are no indications of historic fire or large-scale insect outbreaks (including in the paleo record), and major windstorms (Harris, 1999; Kramer *et al*., 2001; Orians & Schoen, 2013) and landslides (Swanston, 1974; Buma & Johnson, 2015), also rare, affect limited spatial regions (exposed southerly slopes and steep terrain) consistently across the region. Because the small-scale gap dynamics are relatively frequent when considering the entire region, using a space-for-time conceptual framework a good description of the spatial-topographic distribution of the regional disturbance regime can be constructed. This assumption and generalization allows us to draw empirically based conclusions regarding the biomass trajectory of this large region while also incorporating disturbances, a task which would be difficult in other systems where disturbance patterns, species, and change are more variable (e.g. infrequent, severe fires).

Current expectations for the region (excluding historic or currently harvested areas) are expansion and accretion of biomass (Wolken *et al*., 2011), with the region operating as a potentially significant carbon sink. This is expected because of

observed increases in temperature driving increased growth, declining snowpacks allowing for expansion of forests at treeline, a lack of drought stress due to continuous, high precipitation, and the absence of fire. Testing these expectations, however, is difficult, because of the inaccessible nature of the terrain and the lack of infrastructure (the majority of the region is not on a road system). Here, we use a combination of remotely sensed resources and repeat plot measurements to determine where forest change is occurring and the magnitude of that change in terms of biomass.

Forest change map

Using 30 m resolution forest loss and gain maps (Hansen et al., 2013), the distribution of forest loss events was contrasted against establishment events (where 'events' are contiguous patches) over the entire temperate rainforest biome of southeast Alaska (135 000 km^2; regional scale) and by 1° latitudinal bands (subregional scale). To isolate natural dynamics, areas of historic and current logging were removed prior to analysis. Maps were built from available satellite imagery (2000–2012) and represent either the loss or gain of forest cover over that time period. This layer was then limited to locations within the defined perhumid temperate rainforest biome (Ecotrust, 1995) between latitude 54.7° (Alaska–Canada border) and 59° where harvest maps were available; harvested areas (either active loss or regrowth from historic harvesting) were not considered in this analysis (USFS, 2013). Private lands (a minor component of the landscape) do not consistently track forest harvest spatially and were excluded. Spatially contiguous gain or loss pixels were combined into single polygonal 'events;' note that a single patch expanding over multiple years is considered a single event. Accuracy of the forest change map was assessed manually via high resolution remote sensing (overall accuracy 91%), see Data S1.

Spatio-topographic data

Topographic variables were calculated from the ASTER Global DEM V2 dataset. The ASTER dataset has reasonable accuracy (NASA, 2011), with a mean error of 0.94 m for evergreen forest. Aspect was transformed to 0 N, 180 S (E/W equivalent) to avoid circularity. Exposure was calculated for the entire study area using methodology calibrated in southeast Alaska (Kramer *et al.*, 2001) and represents topographic sheltering to regional prevailing winds [southeast, south, southeast, (Harris, 1999)]. 2001 landcover maps (Homer *et al.*, 2007) were used to correspond to the beginning of the observation period. A resolution of 30 m was used throughout, which while useful for disturbances at meso- and broad scales, limits our ability to identify small, subpixel single-tree mortality; this is addressed using FIA data.

FIA Data

To investigate change at finer scales (e.g. single-tree mortality) and tie observed spatial patterns to biomass change, US Forest Service Forest Inventory and Analysis (FIA) data within the region were investigated (n = 1465 plots, with 759 forested plots; ~500 m^2). FIA plots are established on a regular grid across all locations (regardless of ownership), provide repeated measurements of biomass accumulation and mortality at regular intervals, and are designed to provide an unbiased assessment of forest biomass change at regional scales, along with appropriate measures of uncertainty. FIA plots were partitioned according to their spatio-topographic position to assess changes in biomass in the same fashion as changes in forest extent and disturbance patterns from the remote sensing analysis.

Remeasured inventory data include all trees above 12.7 cm diameter; biomass change is calculated based on increases in size on trees within each plot, observed mortality, and recruitment of smaller trees into the >12.7-cm diameter pool. Soil and dead material was not remeasured and thus not available for this spatially explicit design. Plots were revisited between 2004 and 2010; only plots with no history of management were included here. Wilderness areas were only visited once and thus not used for biomass difference calculations; rather observed change was scaled by relative area to include wilderness areas for the final net totals. Estimates were produced using the standard national method (Bechtold & Patterson, 2005).

The use of remeasured FIA plots allows us to: (1) characterize the implications of observed forest disturbance by directly tying change to biomass totals, (2) infer causal agents by noting species differences, and (3) observe single-tree mortality events, which would likely be missed by the 30-m resolution satellite record. Thus, we can monitor the entire spatial spectrum of forest change, from single tree to large events, and compare spatio-topographic patterns across multiple scales of investigation.

All statistical and geospatial analyses were conducted in R. Significance of any skew in the distribution, indicating topographic bias in disturbance or gain, was calculated using D'Agostino K^2 tests (D'Agostino *et al.*, 1990). It should be noted that complex relationships (e.g. bimodal distributions) are not well suited to statistical and significance summaries and distribution results should be interpreted primarily via the histograms/density plots.

Results

Results indicate that the temperate rainforests of southeast Alaska are growing, shifting their distribution and density at large spatial scales through thousands of small events (loss: >17 400 patches; gain: >27 107 patches; Fig. 1). This change can be observed in three ways: (1) more spatially extensive establishment events than loss events, (2) differing spatial and topographic patterns in loss vs. gains, and (3) biomass accumulation within forested areas outpacing mortality.

Total area

At the regional scale, ~25% more area was gained than lost, although that difference was not distributed

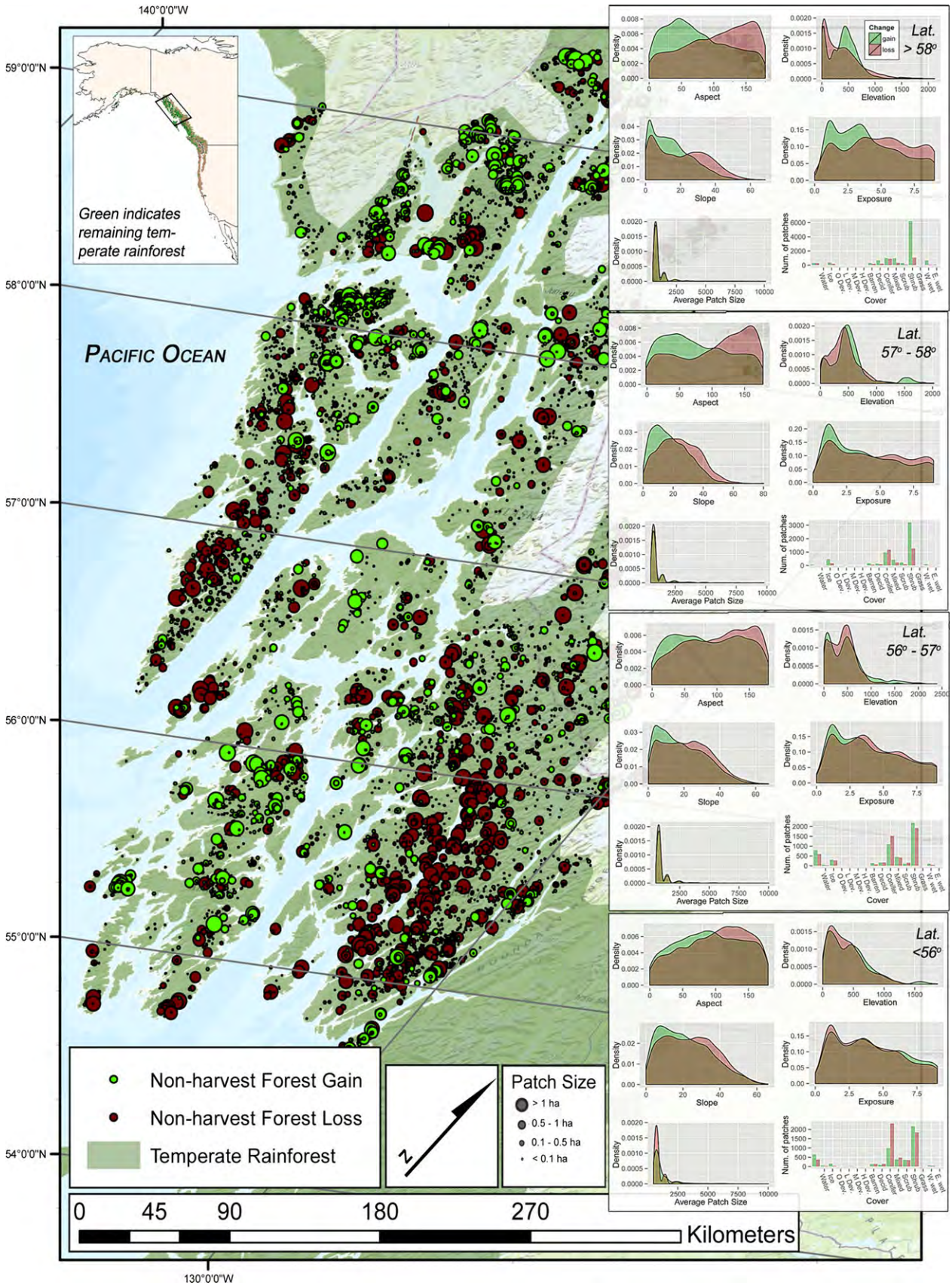
140°0'0"W
59°0'0"N
58°0'0"N
57°0'0"N
56°0'0"N
55°0'0"N
54°0'0"N
130°0'0"W
Green indicates remaining temperate rainforest
PACIFIC OCEAN
Non-harvest Forest Gain
Non-harvest Forest Loss
Temperate Rainforest
Patch Size
> 1 ha
0.5 - 1 ha
0.1 - 0.5 ha
< 0.1 ha
0 45 90 180 270
Kilometers
Lat. > 58°
Lat. 57° - 58°
Lat. 56° - 57°
Lat. <56°
Change
gain
loss
Aspect
Elevation
Slope
Exposure
Average Patch Size
Num. of patches
Cover
Density

Fig. 1 Distribution of forest loss/gain events and topographical distributions of gain and loss. The North Pacific perhumid rainforest zone of SE Alaska, with gain and loss events. Events located in harvested area, or private land, are not shown. Density plots: Subregional density plots (corresponding to latitudinal bands in main map) are shown for each topographic variable. Topographic disequilibrium can be seen by non-overlapping distributions, such as in the north, where forest gain is skewed toward northerly facing slopes (note aspect transformation, see Materials and methods); loss toward southerly. Cover abbreviations: O/L/M/H Dev: Open, low, medium, high developed; Decid: Deciduous; Mixed: Mixed forest, W. Wet: Woody wetlands; E. Wet: Emergent herbaceous wetlands. Unlabeled column is undefined cover types, primarily located along the US–Canadian border.

evenly, with more gains in the north. Losses are, in general, evenly distributed, although a slight skew toward the south is apparent, where they slightly outweigh gained forest (Fig. 2). The relative difference peaks sharply in the north, where gained areas greatly outpace losses and are concentrated primarily in shrubland (such as above treeline, areas exposed by receding glaciers, and floodplains; Fig. 3). In all cases, the average patch size (for both losses and gains) is small, on the order of 1000 m^2 (Table 1).

Topographic differences

Forest losses and gains are distributed differently for the topographic variables considered ($P < 0.05$), with the strongest differences seen in aspect and slope (subregionally: Fig. 1; regionally: Fig. 3). Forest gains are essentially uniformly distributed with respect to aspect in the south, but take on a skew toward northerly aspects in the northerly portions of the rainforest. In contrast, forest loss events are skewed toward southerly aspects across the region, although the effect is strongest in the north. Gains and losses are distributed similarly across elevation; regional medians: 391 m (loss), 441 m (gain) except in the far north, where gained area is distributed biomodally, with most occurring either at sea level or around 500-m elevation (Fig. 1). Gains were typically seen on lower slopes throughout the region (median = 15.7 deg), and losses more uniformly distributed across slopes (median 22.1 deg). Establishment and losses relative to wind exposure were broadly similar except in the north, where gains were skewed toward sheltered locales and loss toward exposed areas.

Biomass change

Within the previously forested areas on the landscape, there was an estimated net increase in biomass of 0.24 Mg ha^{-1} yr^{-1} (SE 0.19 Mg ha^{-1} yr^{-1}), or approximately 985.5 Gg per year across the preexisting forest (4 403 000 ha; Table 2). Fine-scale, single-tree mortality occurred (~1.371 Mg ha^{-1} yr^{-1}, SE: 0.156 Mg ha^{-1} yr^{-1}), but was more than compensated for by growth of remaining individuals (1.523 Mg ha^{-1} yr^{-1}, SE: 0.12 Mg ha^{-1} yr^{-1}) and recruitment of smaller individuals into the measured pool (0.095 Mg ha^{-1} yr^{-1}, SE: 0.006 Mg ha^{-1} yr^{-1}). This estimate does not include forest expansion in previously non-forested areas (which was not measured in the FIA system) and thus is a conservative estimate of biomass accumulation.

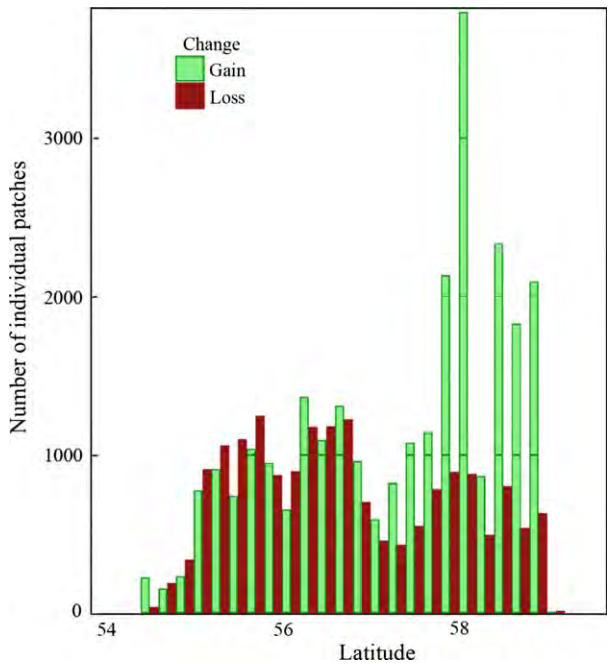

Fig. 2 Summary of latitudinal distributions of losses and gains in the North Pacific perhumid rainforest. In sum, there are 17 400 individual patches of forest loss and 27 107 patches of forest gain (excluding harvest and private landholdings). While forest loss is spread across the latitudinal range, a sharp skew in forest gain to the north indicates forest expansion in that region. Patch areal and topographic statistics given in Table 1; topographic distributions in Fig. 3.

Patterns in biomass accumulation were roughly similar to broad scale forest disturbance and establishment spatio-topographic patterning, although more variable and less significant, as heterogeneity increased at finer scales. Net accumulation was most significant in the north (0.79 Mg ha^{-1} yr^{-1}, SE 0.36 Mg ha^{-1} yr^{-1}), similar to the observed expansion patterns. Lower latitudes showed mortality roughly balanced with biomass

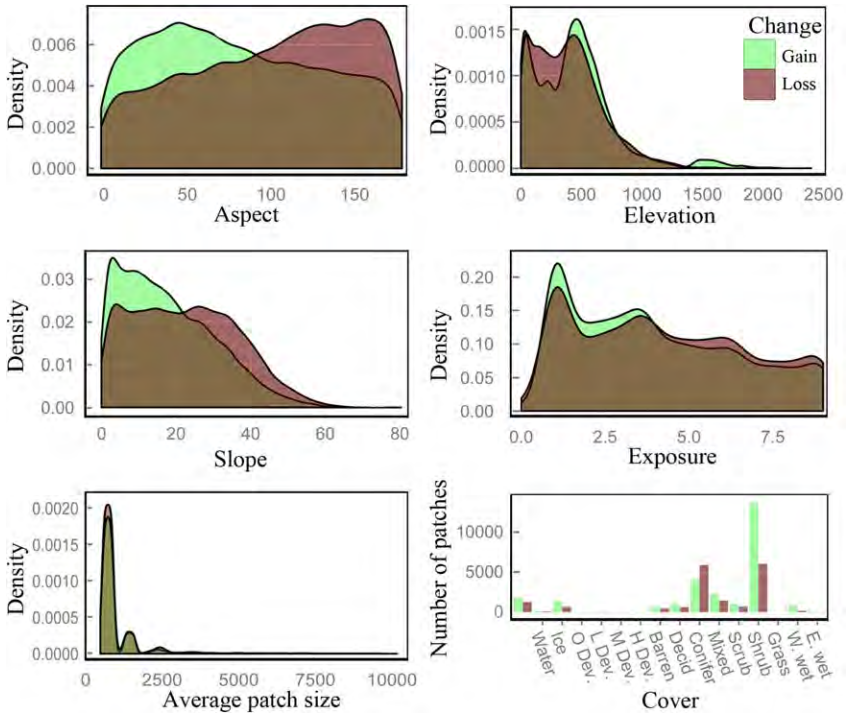

Fig. 3 Regional-scale relationship between topographic variables and forest change. Data include all points in Fig. 1. Forest loss is strongly skewed to steeper, southerly slopes, whereas gains are skewed toward shallow, northerly slopes. There is little difference in the distributions in relation to exposure or elevation at the regional scale, although a slight skew toward higher elevations is apparent in forest gain locations. A majority of the gains occur in shrublands, which includes lowlands, postglacial landscapes, and above tree-line.

Table 1 Change in forest extent and patch sizes by region and subregion

	Latitude range				
	<56°	56°–57°	57°–58°	>58°	Regional
Loss area (ha)	728	576	339	362	2006
Gain area (ha)	534	503	525	1088	2650
Patches lost	5754	5182	3116	3348	17 400
Mean patch size (ha)	0.13	0.11	0.11	0.07	0.12
Median patch size (ha)	0.07	0.07	0.07	0.11	0.07
Patches gained	5027	5386	5773	10 921	27 107
Mean patch size (ha)	0.11	0.09	0.11	0.07	0.1
Median patch size (ha)	0.07	0.07	0.07	0.1	0.07

accumulation, although variance was high. At the regional scale, biomass accumulation was biased toward low elevations (<350 m) with an estimated net change of +0.37 Mg ha^{-1} yr^{-1} (SE 0.25 Mg ha^{-1} yr^{-1}) and low slopes (<20 deg.) where biomass is increasing at a rate of 0.64 Mg ha^{-1} yr^{-1} (SE 0.19 Mg ha^{-1} yr^{-1}). In the northerly portions of the region, accumulation was highest on north slopes (Table S1), although that pattern was not significant at the regional scale (Table 2).

Discussion

Forest change is driven by disturbance, establishment, and growth. Disturbances happen at multiple scales, from single-tree mortality events at the plot scale which open up resources to the understory and reduce standing live biomass, to larger, patch-forming events which are easily discernable from the satellite record, often driven by winter storms or landslides. Establishment of forests in non-forested locations

Table 2 Change in regional biomass within existing forest across topographic context. Totals estimated from remeasured FIA plots within each group

	Observed change								
Group	Growth total	SE	Ingrowth total	SE	Mortality total Gg	SE	Net total*	SE	Forest area total 1000 ha
Elev < 350 m	2440	290	140	12	2002	276	1086	386	3170
Elev 350–700 m	1212	120	69	9	1286	275	−5	273	1001
Elev > 700 m	35	11	5	2	32	16	20	18	232
Slope < 20 deg	2200	211	121	12	1391	197	1427	272	2421
Slope > 20 deg	1487	231	93	10	1929	329	−675	375	1982
Flat – no aspect	613	101	22	5	248	67	741	102	552
South aspect	1582	230	114	12	1670	302	42	360	1922
North aspect	1492	194	78	8	1402	240	282	282	1928
All forest	3687	301	214	14	3320	380	985	469	4403

*Final net change (total growth–mortality) scaled from observed plot data to total unmanaged forested area including wilderness areas (see Materials and methods).

allows for the accumulation of new biomass in woody material, and growth of existing forests, if not balanced by ongoing mortality, increases biomass density within the biome.

Differences in spatial distributions suggest ongoing change in the total area and spatial location of the forest, occurring via small, isolated patches. Forest extent appears to be exhibiting a slight decline in the south, and expanding in the north on northerly facing, low angle, sheltered slopes. This is potentially a signal, in part, of lower snowpacks and earlier melt dates resulting in longer growing seasons in previously short-season areas (Barrett & Christensen, 2011; Wolken *et al.*, 2011). The similar skew toward areas of lower exposure would correspond with this hypothesis, as higher success in sheltered areas would be expected if amelioration of environmental limitations is the driving factor in the observed forest gains, as opposed to declines in the intensity of disturbance drivers (higher exposure) or changes in solar energy inputs potentially driven by changes in cloud patterns (southerly slopes). Observed gains are also consistent with the hypothesis that climate change is allowing for colonization of previously non-forested locations, including above treeline and low elevation, recently deglaciated land (Fastie, 1995; Wolken *et al.*, 2011). Wetland encroachment, found in all subregions (Fig. 1), could also explain much of the biomass and spatial gains at low slopes. Because some of these shifts are reinforcing (establishment skewed to northerly aspects, mortality skewed to southerly), relatively rapid changes in forest distributions are possible.

These broad spatial patterns were somewhat reflected in the finer-scale plot-level changes, which include subpixel forest mortality, although variance was high due to the relatively small sample size (759 forested plots). Broadly, plots are accumulating biomass in the north (similar to expansion) while remaining neutral in the south. At present, this increase in biomass is relatively modest at fine scales (0.24 Mg ha^{-1} per year), but because of the substantial area, total biomass accretion is large, on the order of 986 Gg per year; assuming 50% C, approximately 0.5 Tg C yr^{-1} in natural, unmanaged forest. The observed increases in biomass are an underestimation of carbon/biomass accumulation, as a change in inventory methods prevented the inclusion of biomass accretion in newly forested areas (e.g. treeline or periglacial areas) or soil/detrital pools. Increases in forest extent are apparent in the comparison of change by landcover types (subregional: Fig. 1) and on the change map, such as in the Glacier Bay region (59°N, 136°W), which has seen extensive glacial retreat and forest establishment (Fastie, 1995). We refrain from extrapolating observed biomass accumulation within forested areas to the observed and potential future expansion outside of forested areas as it is outside the scope of our questions and would be highly speculative; however, given the large amount of coastal, periglacial, and treeline in the region, the amount could be significant depending on the rate of change. Forest expansion has implications for albedo and absorbed solar radiation in formerly treeless areas (~1.7x increase across these latitudes), a substantial potential feedback mechanism which appears to enhance subsequent growth (Sturm *et al.*, 2005). However, increases in precipitation in the future may decrease growth if soils become waterlogged, although slower

decomposition may balance that decrease in terms of overall ecosystem carbon balance (Orians & Schoen, 2013). Further research on biomass trends during a shrub-to-forest transition in the region (for example) as well as the potential for future precipitation increases and its effect on growth is needed to resolve these potential feedbacks and magnitudes.

The pattern of both increased extent and plot-level biomass was primarily a northern phenomenon, less apparent in the south. An emerging disturbance, yellow cedar mortality, may be driving much of this spatial pattern. Currently affecting >250 000 ha and resulting in >70% mortality in mainly southerly locations, this mortality is a result of early snowpack melting and shallow rooting in wet soils, predisposing fine roots to freezing (Hennon *et al.*, 2012). FIA data do not show an overall decline in yellow cedar biomass, although biomass increases for yellow cedar are highest on steeper, north facing slopes (which are likely to be better drained and hold snow longer) and nonexistent on south facing, shallow slopes, contrary to the other major tree species and the overall trends of higher mortality at higher slopes (Table S2). Successional trends following this large-scale morality are relatively unknown; one study indicated a shift to western hemlock dominance (Oakes *et al.*, 2014), although that study was in an area with a relatively small tree species pool. This represents a climatic shift that explicitly disfavors one species while favoring other tree species, and thus expansion and increased biomass accumulation in the forest may proceed while certain species are lost.

Because the FIA plots used here were only remeasured once, it is impossible to estimate changes in rates, for example, if the observed increase in plot-scale biomass is due to increased growth rates, decreased mortality rates, or both. Regional disturbance patterns suggest that decreased mortality is at least partially responsible, as growth is outpacing mortality at both the plot scale and in terms of disturbance events. An alternate explanation would be a decrease in disturbance events in the recent past; however, there has been no apparent change in high-speed wind events (driving blowdown) or extreme cold events (driving yellow cedar mortality; Fig. S1) or in precipitation totals, an important driver of landslide likelihood (Buma & Johnson, 2015). Thus, it appears the signal is likely more from changes in growth and endogenous mortality, rather than external disturbances. Because of the need for spatially explicit, repeat measurements, results are limited to a subset of the ecosystem (live trees >12.7 cm DBH) for which those data were available. Total ecosystem accumulation in the region is likely significantly higher, as mature forests typically sequester more carbon in slowly decomposing dead material and the soil, especially in the relatively cool and wet temperate rainforest region. Unforested areas, including muskeg bogs, also sequester immense amounts of carbon belowground, and harvested areas are a strong and potentially accelerating carbon sink in these forests (Hember *et al.*, 2012), although limited temporally. Overall all cover types and forest age classes, estimated carbon sequestration in the study region is >2.8 Pg C in total biomass and soil (Leighty *et al.*, 2006).

The data used here are remotely sensed and empirical, intended to test expectations related to forest growth and expansion and explore the spatio-topographic distribution of those changes. This work also represents the first spatially explicit estimate of change in this large carbon sink. The results presented in this study indicate that regional carbon sink is not only driven by afforestation and historic harvest, but also expansion and densification of mature forest, along spatio-topographic patterns expected from a warming climate. Certain aspects, elevations, slopes, and contexts are favored over others, indicating the ongoing shifts are not following historic drivers. Ongoing monitoring, attributing observations to location-specific mechanisms, and following the implications for other cover types will be important over subsequent decades. True equilibrium (e.g. complete balance between growth and loss) can be a useful theoretic concept, although unlikely over short-time periods and small spatial scales due to stochastic variation in growth/loss processes. However, the nature of the perhumid temperate rainforest (similar climate and species, gap-dominated disturbance regime) makes them particularly amenable to this conceptual framework, although rare large events in the future could disrupt the observed patterns. Ongoing work is needed to determine whether the observed imbalance is maintained, increases, or decreases, especially if novel (e.g. fire) or large-scale disturbances affect the region as the climate shifts. Nonetheless, the data presented here represent an imbalance over a decadal time period and an important first data point on regional (spatial) response to climate change.

Generally, spatio-topographic trends in forest extent change are similar (though not identical) to spatio-topographic trends in forest biomass change, representing two independent datasets which support the hypothesis that climate change is causing directional change at the regional scale, although spatial and topographic variation is significant. The temperate rainforests of North America represent one of the globe's great storehouses of ecological diversity (Brandt *et al.*, 2014), endemic flora and fauna (Cook

et al., 2001; Orians & Schoen, 2013), and carbon stocks (Leighty *et al*., 2006; Luyssaert *et al*., 2008; Keith *et al*., 2009). Any change or directionality to their dynamics is important to consider. Overall, the significant differences in total forested area and the spatial/topographic distributions of growth and loss, consistent with expectations of altered forest dynamics due to a warming climate, suggest a directional shift due to climate influences. The process is playing out via small, subhectare scale patches non-randomly distributed across the landscape; unremarkable in isolation but significant in aggregate, these events represent a large, interesting shift in forest dynamics.

Acknowledgements

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Table S1. Changes in live biomass within existing forest by latitude and topographic context.
Table S2. Regional changes in live biomass by species across all topographic contexts.
Figure S1. Reconstructed extreme wind events for study region from 1958 to 2010. Drivers of disturbance – high winds and low temperatures – are shown for the northern and southern portions of the region, as reconstructed using CMIP5 GCM daily data for the time period. Data indicates no general trend in disturbance drivers which may influence results. Data and visualizations provided by Matthew Leonawicz and SNAP (Scenarios Network for Arctic Planning).
Data S1. Accuracy Assessment. Accuracy assessment of remotely sensed dataset for the region using high resolution imagery.

Mason D. Bryant and **Fred H. Everest,** Aquatic/Land Interactions Research Program, Pacific Northwest Research Station, U.S.D.A. Forest Service, 2770 Sherwood Ln., 2A, Juneau, Alaska 99801

Management and Condition of Watersheds in Southeast Alaska: The Persistence of Anadromous Salmon

Abstract

In contrast to most of North America and Europe, numerous intact or lightly disturbed watersheds are present throughout southeast Alaska. These watersheds support abundant and diverse populations of anadromous salmonids. While the watersheds throughout the northern hemisphere have been exposed to human disturbance from millennia to centuries, significant human disturbance to the watersheds of southeast Alaska did not begin until the 1950's with the start of industrial logging. Although management of watersheds has evolved to reduce risks to aquatic habitat, the most intensive logging occurred during the first 20 years of timber harvest when few restraints were placed on timber harvest in watersheds. As a result, a legacy of streams with deteriorating habitat remains. While few salmon stocks in southeast Alaska appear to be in decline, escapement records on specific watersheds, particularly those most severely affected by management are non-existent or qualitative. The present status of salmon stocks may be attributed to abundant intact watersheds, high marine survival, and escapement levels that fully seed most watersheds. The numerous intact watersheds throughout southeast Alaska are a critical factor in maintaining sustainable salmon stocks iu southeast Alaska.

Introduction

Hundreds of large and small, and mostly pristine watersheds are dispersed through the islands of the Alexander Archipelago of southeast Alaska. These watersheds are located in a small percent of the remaining temperate rain forests in the world that have not been moderately or severely altered by human activity. Furthermore, these watersheds support a disproportionately high proportion of the wild anadromous salmonid stocks remaining in the Pacific Northwest. Throughout the western United States and British Columbia, Canada, numerous stocks of anadromous salmonids are at risk of extinction, declining, or of serious concern to resource managers and users (Nehlsen et al. 1991; Slaney et al. 1996). Southeast Alaska is an exception with few stocks that have been identified in decline (Baker et al. 1996; Halupka et al. in press). This can be attributed largely to the absence of dams, agricultural and urban development, and intact watersheds in southeast Alaska. Intensive human exploitation of watersheds in southeast Alaska began only a few decades ago and is likely to continue as demand for resources grows with increasing human populations. The most significant potential risks to salmonid stocks in southeast Alaska include large scale habitat degradation and fishing pressure (Baker et al. 1996; Halupka et al. in press).

The purpose of this paper is to examine historical timber harvest patterns, management history, distribution of managed and intact watersheds, and their relationship to anadromous salmonid populations in southeast Alaska. The focus is on land managed by the U. S. Forest Service, the Tongass National Forest (TNF). Although the area of the Tongass National Forest is large, a relatively small portion is suitable for timber production and this portion often coincides with highly productive fish habitat. We briefly describe the development of "Best Management Practices" implemented by the U.S. Forest Service from the 1950s through 1996 and their application in watersheds in southeast Alaska. We examine timber harvest patterns to show the extent of disturbance in managed watersheds and how these patterns have changed from the start of industrial timber harvest to the present. We hypothesize that improvements in forest management practices within the past 5 years and retention of intact watersheds will be an important factor in maintaining healthy anadromous salmon populations in southeast Alaska.

The TNF is part of the temperate rain forest biome (Alaback 1991). Globally, the temperate rain forest consists of about 34.4 million ha (90 million acres) which includes parts of North America, Chile, New Zealand, Tasmania, and the former Soviet Union (Hagenstein 1993; TLMP

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Revision 1997). About 50% of this biome occurs in North America with most located along the coasts of British Columbia and southeast Alaska (Ecotrust et al. 1995). About 14% (5.1 million ha) occurs in the TNF. About 2.0 million ha of the TNF is classified as old-growth forests and is commercial forest[1]. The remaining area consists of alpine, ice fields, muskegs, and areas with less than 10% tree cover. More than 70% of the commercial forest is below 492 m (1500 ft) elevation and of that more than 40% is below 262.5 m (800 ft) elevation (Figure 1). The area less than 262.5 m (800 ft) includes 1,450,000 ha of commercial and non-commercial forests and comprises the most productive valley bottoms. Of this, about 263,000 ha are riparian corridors (TLMP Revision 1997).

[1]commercial forest defined as having >8,000 board feet (b.f.) per acre.

Sitka spruce (*Picea sitchensis*) and western hemlock (*Tsuga heterophylla*) are the dominant trees at the lower elevations and in the stream valley bottoms (Viereck and Little 1972). These were the only economically valuable tree species until five years ago when yellow cedar (*Chamaecyparis nootkatensis*) also became economically important. Alaska yellow cedar extends throughout coastal south central Alaska, southeast Alaska, and southward through British Columbia (Viereck and Little 1972). Western redcedar (*Thuja plicata*) occurs in the southern portion of southeast Alaska. Mountain hemlock (*Tsuga mertensiana*) is found at higher elevations. Red Alder (*Alnus rubra*) is common along disturbed sites, either natural or anthropogenic, such as landslides, abandoned roads, and gravel bars along riparian areas. Understory of old-growth forests is generally comprised of younger trees and various shrubs and

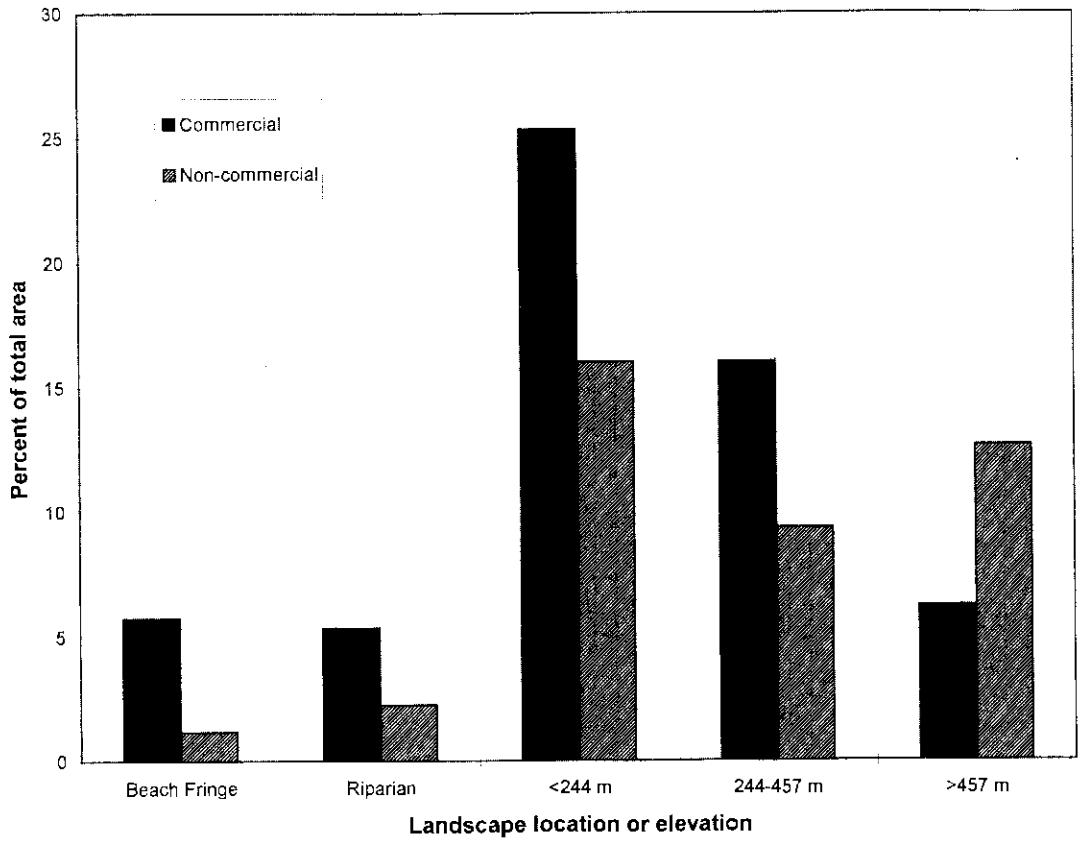

Figure 1. Distribution (% of area) of old-growth forest in southeast Alaska classified into commercial and non-commercial forest by landscape and elevation (TLMP 1997).

herbs including blueberry (*Vaccinium* spp), fool's huckleberry (*Menziesia ferruginea*), stink current (*Ribes bracteosum*), and devils club (*Oplopanax horridus*). A complete description of the vegetation is provided by Alaback (1980).

The landscape of southeast Alaska is geologically young. Most areas were glaciated during the late-Wisconsin glaciation which ended about 10,000 yr BP (Heusser 1960; Pewe 1975; Pielou 1992). The terrain is mountainous and characterized by steep-sided mountains and U-shaped valleys sculpted by glaciers (Harris et al. 1974). High rainfall and heavy snowpack at higher elevations (>300 m) combined with the steep topography drive a complex hydrological cycle that creates fall floods and high water in the spring as the snow melts (Schmiege et al. 1974). The steep, irregular topography and high precipitation contribute to a pattern of frequent natural disturbance events that include landslides and blowdown of large old-growth trees. The combination of varying flows, natural disturbance, and large old-growth trees creates complex stream ecosystems that support large and diverse communities of anadromous salmonids. These, in turn, support valuable commercial, sport, and subsistence fisheries for pink (*Oncorhynchus gorbuscha*), coho, (*O. kisutch*), sockeye, (*O. nerka*), chum (*O. keta*), and chinook salmon (*O. tshawytscha*) (Meehan 1974). The large salmon runs are a significant resource to terrestrial mammals and birds (Shochat 1993; Willson and Halupka 1995) as well as an important nutrient source for aquatic organisms (Cederholm et al.1989; Wipfli in press).

Historical Perspective of Watershed Disturbance

Rivers and their watersheds have been focal points for human occupation and large scale anthropogenic modifications and disturbances have occurred for millennia. Zuaderer (1985) documents the development of canals and irrigation systems of the Tigres and Euphrates river systems to support agriculture for the ancient city states of Ninevah and the Assyrian empire about 3000 yr BP. Watershed and riparian development throughout Europe began centuries ago, beginning with deforestation, agriculture, and more recently, industrial development (Petts 1989; Petts et al. 1989). As a result, few unmodified reaches remain. Atlantic salmon (*Salmo salar*) have been extirpated from many of the rivers. Remaining populations are maintained by hatcheries in numerous rivers throughout western Europe (Mills 1991).

Widespread exploitation of watersheds in North America has spanned three centuries beginning on the Atlantic coast with permanent European settlements in the early 1700's. River valleys were cleared for agricultural development, and streams and rivers were dammed for water power and small mills. Along the east coast of the United States, New England, in particular, loss of Atlantic salmon populations occurred more than 100 years ago and was largely unrecorded. Dams throughout New England extirpated natural runs of Atlantic Salmon throughout the region (Stolte 1981). Wissmar et al. (1994) reviewed chronology of disturbance on the watersheds of the Pacific Northwest which began with the fur trade in 1811. The following decades brought the development of mining, early settlement for agriculture, and livestock grazing throughout the Pacific Northwest. During the course of three centuries of watershed development in North America, no major watershed remains completely free of extensive human disturbance in the contiguous United States. By 1990, less than 20% of the old-growth forests remained in the Pacific Northwest, virtually all on public land (Lehmkuhl and Ruggiero 1991).

The literature reviewed by Wissmar et al. (1994) and McIntosh et al. (1994) suggested that most of the adverse effects of the development and exploitation of the watersheds in Pacific Northwest were largely unreported until the 1970's. They cited only three pre-1940 reports on habitat condition of these watersheds. During the 1980's, reports on the condition of habitat and salmon stocks appeared in greater numbers and by the 1990's several comprehensive reports on habitat condition revealed that large scale deterioration of habitat in the Columbia River basin occurred during the past century (cited in Wissmar et al.1994). The cumulative effects of habitat degradation, combined with over-exploitation, and declining ocean productivity for these stocks has placed 214 native, naturally spawning stocks of salmonids in Washington, Oregon, and California at varying degrees of risk of extinction (Nehlsen et al. 1991; FEMAT 1993). The list includes chinook, coho, sockeye, chum, and pink salmon, as well as steelhead (*O. mykiss*) and sea-run cutthroat trout (*O. clarki*). The combined effects of these factors has created the "salmon crisis" in the last decade. The degree of effect seems to be

related to the duration and density of human occupation.

Large scale anthropogenic influence on southeast Alaska watersheds is a relatively recent phenomenon. Even though European exploration of southeast Alaska began in the 1700's, it was primarily related to the development of the fur trade—sea otter and seals— and whaling and effects on the landscape were localized and relatively minor (McDougall 1993). Even after the purchase of Alaska in 1867 by the United States, timber harvest was restricted to easily accessible areas near the shoreline. By 1900, 14 sawmills were operating and harvested 8.45 million board feet. From 1910 to 1920, another 410 million board feet were harvested (Harris and Farr 1974). Sporadic timber harvest continued through 1950. Large scale harvest did not begin until after a 50-year timber contract for 8.25 billion board feet of timber was awarded to the Ketchikan Pulp Company in 1951 by the U.S.D.A. Forest Service (Harris and Farr 1974). In 1956, a second 50-year timber contract for 5.31 billion board feet of timber was awarded to the Alaska Lumber and Pulp Company. Timber harvest for the two 50-year contracts marks the beginning of industrial forestry and major watershed disturbance in southeast Alaska.

From 1950 through 1992, more than 14 billion board feet of timber were removed from the Tongass National Forest (Murray 1970; Ruderman 1985; Warren 1996). From 1980 through 1994, about 5.9 billion board feet were harvested from private land in Alaska, most of which occurred in southeast Alaska (Ruderman 1985; Warren 1996). The peak harvest on the TNF occurred during the late 1960's through the early 1970's (Figure 2). From 1982 through 1986 harvest was about 1.4 billion board feet. As National Forest land was transferred to private ownership, timber harvest on private lands exceeded that on the TNF. The combined harvest from the TNF and from private

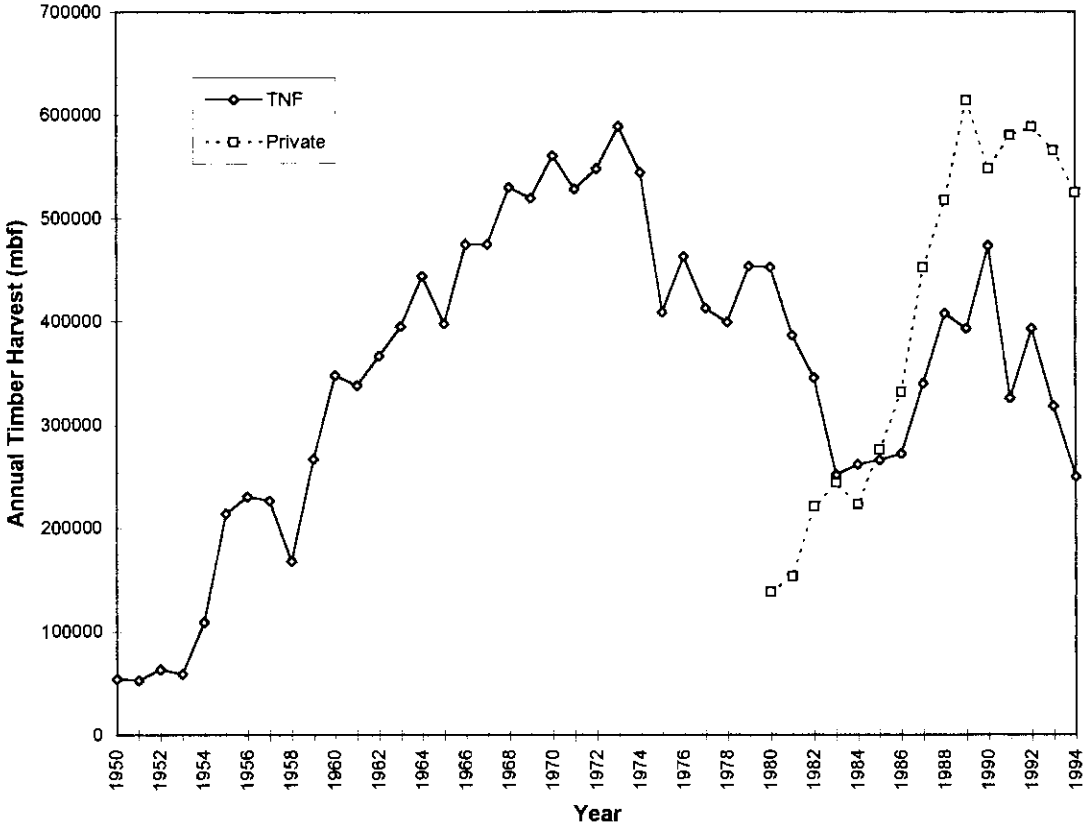

Figure 2. Annual timber harvest in southeast Alaska from the Tongass National Forest (TNF), and state and private lands from 1950 through 1994.

land from 1988 to 1994 was greater than the highest harvest on TNF lands at any point. The mill in Sitka closed in 1994 and a decline in timber harvest occurred on the TNF. The pulp mill in Ketchikan closed in 1997 further decreasing the demand for timber from the TNF.

History of Management Practices in Southeast Alaska

We separate management of riparian areas and stream habitat in the TNF into five intervals: 1950 to 1970; 1971 to 1982; 1983 to 1990; 1991 to 1996; and from 1997. The last period began with the adoption of the Tongass Land Management Plan (TLMP) by the U.S. Forest Service in May 1997. The intervals do not represent precise changes in management policies with the exception of 1991 when buffer strips 32.5m (100 ft) along fish streams were mandated (TTRA 1990). Changes throughout other periods were less apparent in administrative documentation, but the degree of protection to aquatic resources during timber harvest increased during each time interval.

1950-1970

From 1950 to 1970, logging generally proceeded up an entire stream basin with clear-cut logging from the streambank up to an elevation of about 324 m (800 ft). Although felling timber into streams and yarding across streams was prohibited, this was not strictly enforced. Trees were felled into creeks and removed during logging.[2] Logs were often dragged across streams or in some cases logs were dragged along streams. Tractor logging was often used in low gradient riparian areas and road crossings were made through streams. Heavy equipment was operated in streams and along stream banks. Log jams (large woody debris), both natural or resulting from logging operations, were systematically removed from streams either because they were perceived as blocks to migration or created pools that removed spawning areas. Evaluation of the effects of logging was generally accomplished by site inspections which frequently reported equivocal effects.[2] For example, "The tractor road channel may or may not be detrimental." Few guidelines for road layout or design that addressed the protection of aquatic habitat were available (U.S.D.A. Forest Service 1960).

[2]unpublished report. The effect of logging on twelve salmon streams in southeast Alaska. On file with the primary author.

1970-1982

The Logging and Fish Habitat Pamphlet of 1976 outlined improved forest practices for fish streams (U.S.D.A. Forest Service 1976). It identified the significance of small tributaries to juvenile salmonids, steelhead, and non-anadromous species such as Dolly Varden (*Salvelinus malma*) and cutthroat trout. Although the pamphlet recognized the importance of woody debris as cover for fish and cautioned against over-zealous debris removal, it did not recognize the significance of large wood on stream morphology and suggested that large accumulations were blocks to migration and affected spawning habitat. Extensive stream cleaning continued unabated from 1971 through 1982 (Bryant 1983). Examples of road construction, bridge and culvert crossing design, and construction timing to minimize the effect of sediment in streams were given in the Logging and Fish Habitat Pamphlet (U.S.D.A. Forest Service 1976). Significant improvements in aquatic habitat protection were implemented during this period in comparison to the previous 20 years, but streams in most watersheds that were logged during this period had no buffer strips and removal of large wood from most main stem and tributary streams was a common practice. Culverts were often installed that were too small for frequent floods in autumn and washed out. Others were improperly installed and were blocks to adult and juvenile salmonid movement (Bryant personal observation).

1983-1990

The 1979 Tongass Land Management Plan (TLMP 1979), the Alaska Regional Guide (U.S.D.A. Forest Service 1983), and a 1984 review of the Plan (U.S.D.A. Forest Service 1984) incorporated extensive sets of recommendations for stream habitat protection. The Plan in 1979 classified streams with anadromous fish as class I; streams with non-anadromous fish as class II; and tributaries or upstream sections of streams without fish that flow into fish bearing streams as class III. Each stream class was given a different level of protection with the highest afforded to class I streams. Riparian vegetation was retained to provide shade for temperature sensitive streams. Small tributary streams were recognized as habitat for resident trout and Dolly Varden. Riparian habitat was managed to provide large wood to streams that were identified as rearing habitat for identified

species. The recommendation for these streams was to retain all trees that were along the streambank and less than 30.5 cm in diameter at about 1.5 m above ground. Directional felling away from the stream channel was required as was full suspension of timber over streams (i.e. logs were completely elevated above the ground) during yarding operations. Logging in the riparian zone continued as a common practice. Where buffer strips were left, they were generally no wider than one tree height and usually were smaller, non-merchantable trees. In some cases, old-growth buffer strips were left along low gradient floodplain areas such as sections in the upper section of the South Fork of Staney Creek on Prince of Wales Island.

Retention of large wood in streams was included as a management practice in the 1986 Aquatic Habitat Management Handbook as a result of literature demonstrating the importance of large wood in the streams of the Pacific Northwest and Alaska (Swanson et al. 1976; Keller and Talley 1979; Bryant 1983; and Dolloff 1983 among others). Removal of woody debris was restricted to logging slash less than 4 cm diameter, although logging slash was to be prevented from entering the stream in the first place. The handbook specifically recommended against "removal of small debris jams in rearing streams". The large scale removal of woody debris prevalent before 1983 was largely discontinued throughout southeast Alaska (Bryant personal observation).

Significant improvements and specific recommendations for road layout, construction, and stream crossings were included in the Aquatic Habitat Management Handbook (U.S. Department of Agriculture 1986). Practices were incorporated to provide fish passage for both adult and juvenile salmonids and to minimize sediment production. These recommendations included information from an increasing number of reports on methods for forest road construction and stream crossing (Dane 1978; Anderson and Bryant 1980). Although the improved practices were applied and served to reduce effects of roads and stream crossing structures on aquatic habitat, implementation or effectiveness was not systematically monitored and numerous instances where procedures were either not followed or improperly applied occurred throughout the region. More significantly as roads were closed, many stream crossing structures were either not removed or improperly removed. As a result long term, chronic erosion and sediment transport sources were created.

1991-1996

The most obvious change in management in 1991 was the requirement for buffer strips that were at least 32.8 m (100 ft) wide where no trees would be cut along class I, and II streams that flowed into anadromous fish streams (Tongass Timber Reform Act 1990). It included provisions of the Aquatic Management Handbook (U.S. Department of Agriculture 1986). The "Best Management Practices" in the Soil and Water Conservation handbook (U.S. Department of the Agriculture 1993) were applied to protect riparian habitat and stream habitat in managed watersheds. Some significant additions in the Soil and Conservation Handbook were the recognition of riparian areas and floodplains, and specification of management practices to protect habitat beyond the mandated 32.8 m buffer zone. It states that "preferential consideration to riparian dependent resources will be given when conflicts among land use activities occur". The handbook also presents monitoring as an "essential" part of "Best Management Practices" and describes procedures and three stages of monitoring to evaluate management practices. Specific prescriptions are given to maintain channel integrity of all streams including non-fish streams. Guidelines are provided to reduce the potential for mass failures from road construction and location, road crossings, and log decks.

The set of management practices used from 1990 through 1996 were a significant improvement from those in place at the onset of industrial timber harvest and represented an accumulation of management and research information during that period. Throughout the evolution of "Best Management Practices" implementation was unevenly applied. In the absence of effective monitoring, detrimental streamside harvest practices such as debris removal, improper road and culvert installation and maintenance continued even after they were identified in various administrative directives. To the benefit of some watersheds, management practices exceeded those required by various directives. In many watersheds, buffer strips were extended beyond 32.8 m, particularly in floodplains. While the practices

used from 1991 through 1996 generally provided good protection to streams used by anadromous and non-anadromous salmonids, they did not address many of the important intermittent and higher gradient streams that are an integral part of the drainage systems in watersheds. Most of these systems are at elevations above 262.5 m (800 ft) and may not support perennial fish populations. However, they can have a significant influence on downstream habitat and fish populations (Heede 1972; Benda 1990; and Lamberti et al. 1991).

In 1994, the U.S. Congress directed the USDA Forest Service to review existing habitat protection for salmonid streams on the Tongass National Forest to determine their effectiveness (U.S. Department of Agriculture 1995). The report included an extensive literature review on Pacific salmon and their habitat in southeast Alaska and the Pacific Northwest, examined existing data and information on salmonid populations and their habitat in the Tongass National Forest, and reported results from a field evaluation of seven watersheds by a staff of Federal fisheries and watershed professionals (U.S. Department of Agriculture 1995). The assessment also included a pilot watershed analysis of three watersheds in southern, central, and northern southeast Alaska. Some of the key recommendations in the report were for increased protection of headwater streams, ephemeral streams, and non-fish bearing streams that are tributaries to resident and anadromous salmonid streams, site specific buffers on flood plains and alluvial streams, management of fish streams "to preserve biological integrity", and implementation of watershed analysis. The results and recommendations from the Anadromous Fish Habitat Assessment Report formed the cornerstone for the provisions for fish stream and watershed management in the Revision of the Tongass Land Management Plan (TLMP 1997).

1997

In May 1997, the revision of the Tongass Land Management Plan (TLMP) was completed and included increased protection for watersheds and associated riparian areas (TLMP 1997). A significant element of the TLMP is to consider fish habitat in the context of the whole watershed from the headwaters to the ocean during management planning. The plan directly addresses riparian areas and assigns management objectives that include maintaining natural conditions for fish and aquatic life, managing for biodiversity, and to maintain streambank and stream channel processes (TLMP 1997). In most floodplain stream reaches, buffer areas are required over the active floodplain. Changes to prescribed riparian buffers require a watershed analysis to assess "important riparian and aquatic habitat values and geomorphic processes within a watershed". While maintaining and reinforcing protection afforded to anadromous salmonid — class I streams, resident salmonid —class II streams, and non-fish —class III streams, TLMP identifies an additional group of streams—Class IV streams. These include intermittent, ephemeral, and small perennial channels with low flows or low sediment transport capabilities. Although these are not specifically addressed for riparian protection, buffer strips in V-notches of higher gradient class III streams, which may be ephemeral, are included. The management strategy of the standards and guides is to retain natural stream function and processes in watersheds that fall into management categories that will be used for timber harvest or other development activities.

History of Timber Harvest

Timber harvest in southeast Alaska was not located randomly throughout time or location (TLMP Revision 1997). The sites with easy access and high volumes of wood were harvested first. High volume timber sites were frequently located in the valley bottoms less than 244 m (800 ft) in elevation and along riparian corridors. More than 70% of the total timber volume on the TNF was harvested from 1950 through 1982, the period when minimum riparian protection was given to streams (Table 1). Of the 156,600 ha (387,000 acres) logged from 1950 through 1994, 38% (59,896 ha) were harvested from 1950 through 1970 (Table 1). Furthermore, timber harvest was concentrated on selected islands such as Chichagof, Kuiu, Prince of Wales, and Revillagigado islands (Figure 3). Other locations such as Admiralty Island, most of which has been designated as a National Monument, and Baranof Island which contains large wilderness areas, are less developed. They also contain extensive areas of non-commercial forests.

During the first 20 years of large scale timber harvest, most harvest occurred in the valley bottoms where low gradient streams are located. These

TABLE 1. Timber harvest of the Tongass National Forest by time period and by elevation.

Time period	Area in hectares and (percent harvested) during each time period			
		Elevation (m)		
	Total	<244	244-457	>457
1950-70	59.896	54027.6 (90.2)	5665.8 (9.5)	202.35 (0.34)
1971-1982	53.420	45164.5 (84.5)	8094.0 (15.2)	161.88 (0.30)
1983-1990	28.329	20639.7 (73.6)	6879.9 (24.5)	526.11 (1.88)
1991-1994	14.974	9308.1 (62.2)	5261.1 (35.1)	404.7 (2.70)

streams and their associated riparian habitats (i.e. beaver ponds) also are the most productive locations for anadromous salmonids. From 1950 to 1970, 90% of the total area harvested was less than 244 m in elevation (Table 1). As timber harvest progressed, proportionally more area above 244 m was harvested and by 1991 more than 37% of the timber harvest occurred above 244 m in elevation (Table 1). Less than 30% of the area was harvested while measures were in place that recognized small rearing streams and afforded increased protection to non-anadromous species. Less than 10% of the total area was harvested when buffer strips were an integral and required part of watershed management.

Mayheso Creek, located on Prince of Wales Island, was harvested during the 1950's. Harvest began at the bottom of the valley and continued up to 366 m (1,200 ft) in elevation during one entry (Figure 4). All merchantable timber along the riparian zone was harvested which was the general pattern of harvest for the first 20 years of timber harvest. In subsequent years, harvest patterns changed and by 1970 smaller patches (usually < 40 ha) were cut. The earlier harvest units tended to be in the valley bottoms. Most of the harvest units during 1974 through 1976 in the Corner Creek watershed were located along the stream, whereas, those cut in 1986 or later were located at higher elevations away from the main channel (Figure 5). A similar pattern occurred in the Hamilton Creek watershed with later units placed further upstream and on higher elevations (Figure 6). The concentration of intense timber harvest in the riparian zone before 1980 left a legacy of watersheds with little or no riparian protection.

The effect of time and elevation of timber harvest is reflected in the number of kilometers of stream that were within 100 m of a timber harvest unit. More than 75% of the length of anadromous streams were near or adjacent to a timber harvest unit that was cut from 1950 to 1982 (Table 2). As changes in forest management that moved harvest units away from the riparian zone were implemented, the amount of harvest in these areas decreased. The largest part of timber harvest occurred in the valley bottoms adjacent to streams before adequate management practices were in place to reduce even the most obvious detrimental effects of timber harvest on stream habitats.

The Persistence of Salmon

Despite the legacy of watersheds that were logged with poor riparian management practices throughout southeast Alaska, such as the Maybeso Creek watershed, few documented declines of salmon stocks are recorded (Baker et al. 1996). However, few watersheds in southeast Alaska have escapement records that are dependable before 1960 and no systematic or quantitative studies document juvenile or smolt numbers from watersheds before the onset of large scale logging in southeast Alaska (Halupka et al. in press).

Natural fluctuations at a region-wide scale caused by changes in marine survival, weather cycles, or other environmental factors as well as production of salmon from intact watersheds will obscure declines in a single watershed. The limitations of catch statistics as indicators of stock abundance are generally acknowledged (see Bisson et al. 1992), but the cycles of the commercial catch of coho salmon from 1893 through 1998 demonstrate the magnitude of region-wide variation that obscure trends

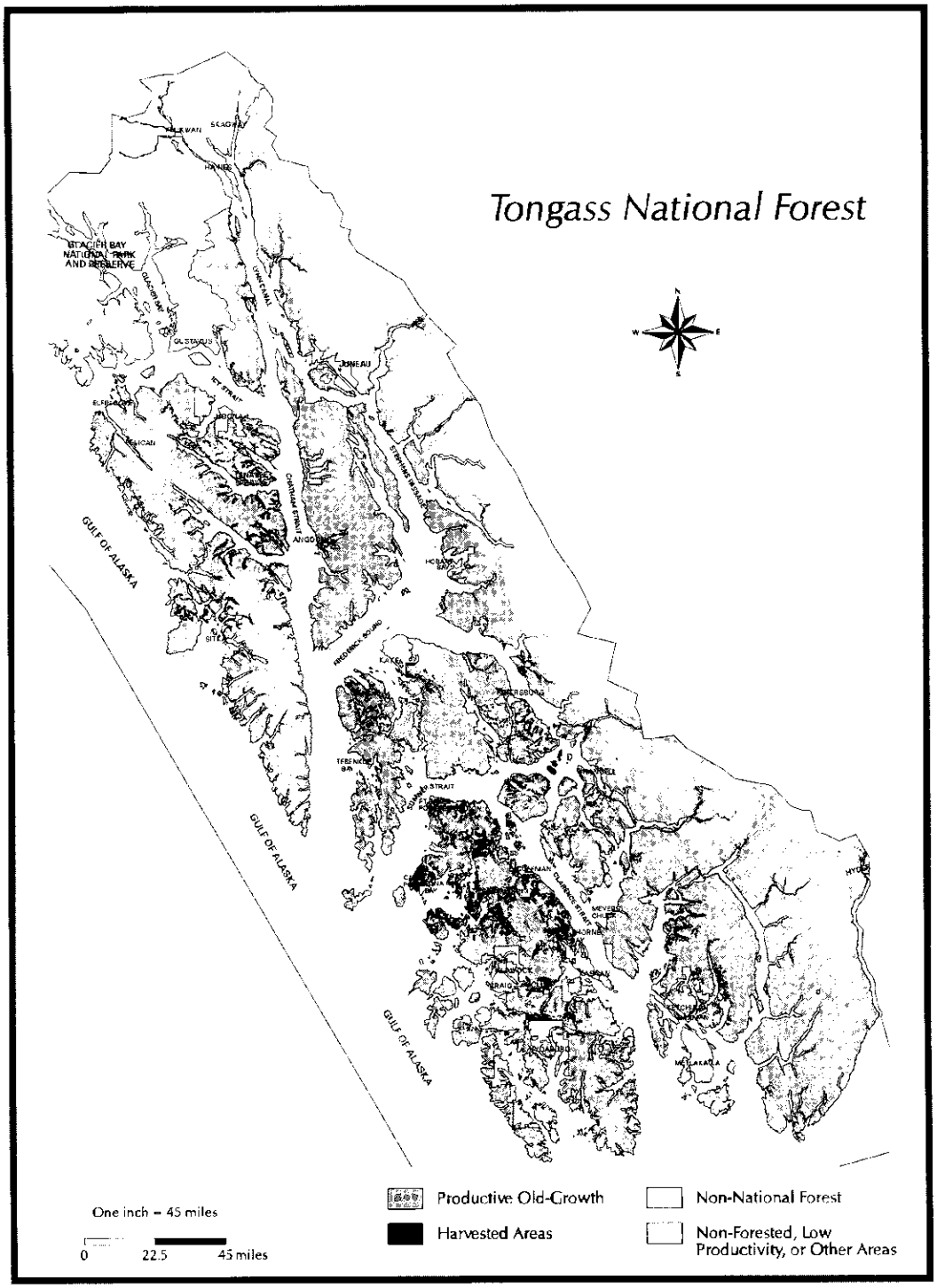

Figure 3. Distribution of commercial old-growth forest and timber harvest throughout southeast Alaska. "Productive Old-growth" was defined as forest cover of old-growth Sitka Spruce and Hemlock ≥ 22.8 cm (9 in) diameter breast height (dbh) and >150 years old at a volume of > 8 million board feet/acre. Timber harvest on non-National Forest land is not identified.

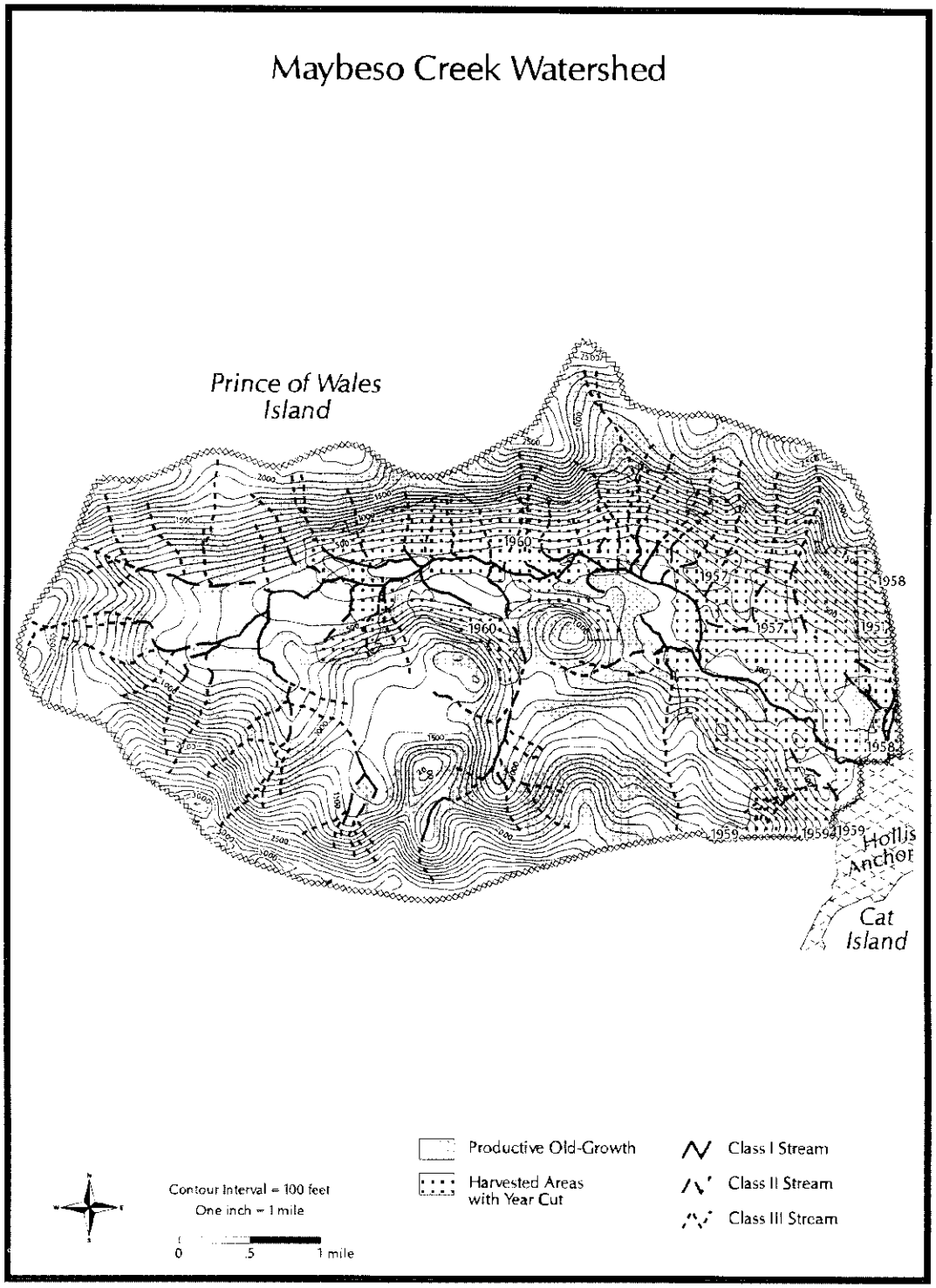

Figure 4. Timber harvest, dates of harvest, and remaining old-growth forest on the Maybeso Creek watershed, Prince of Wales Island.

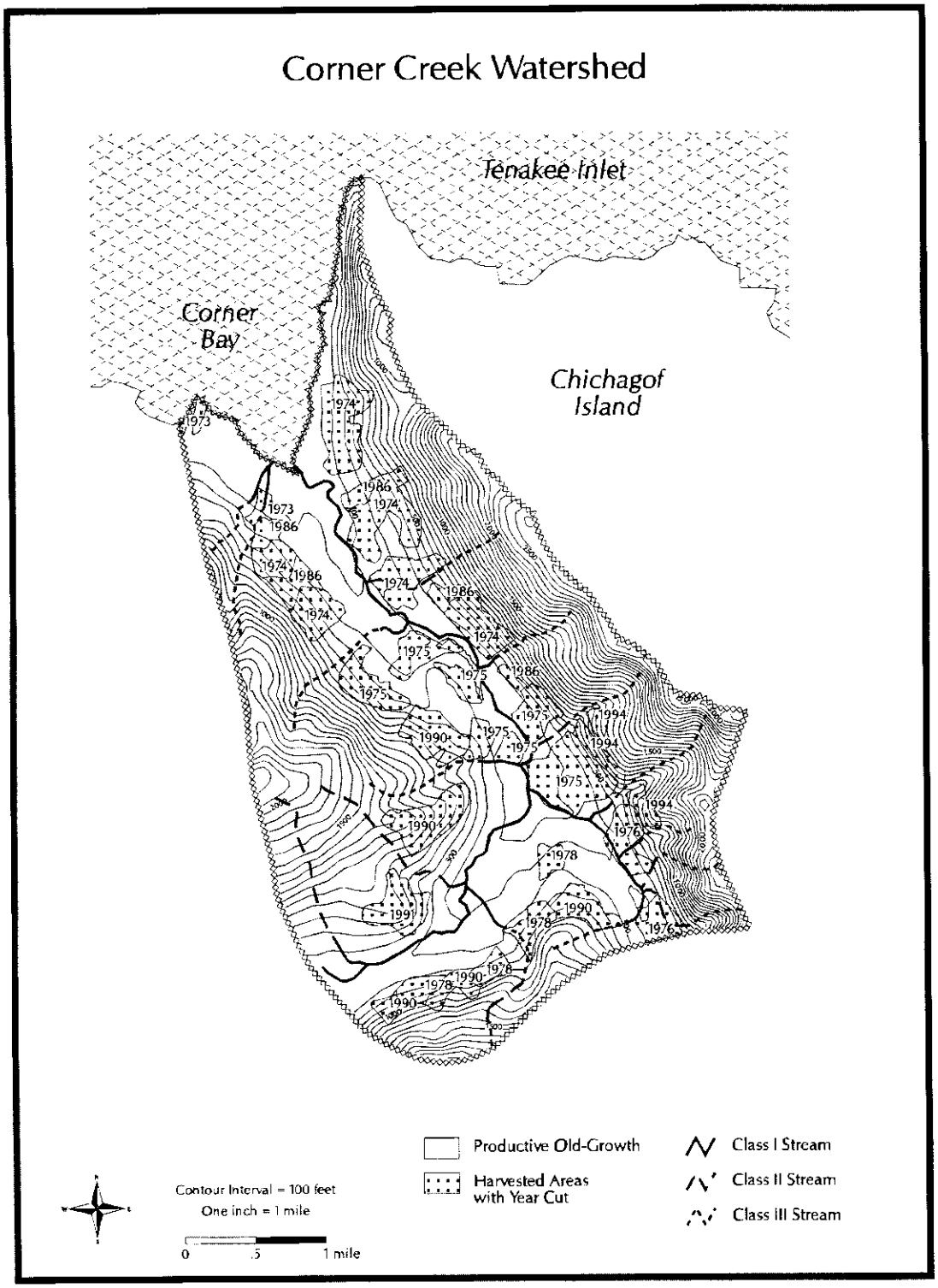

Figure 5. Timber harvest, dates of harvest, and remaining old-growth forest on the Corner Creek watershed, Chichagof Island.

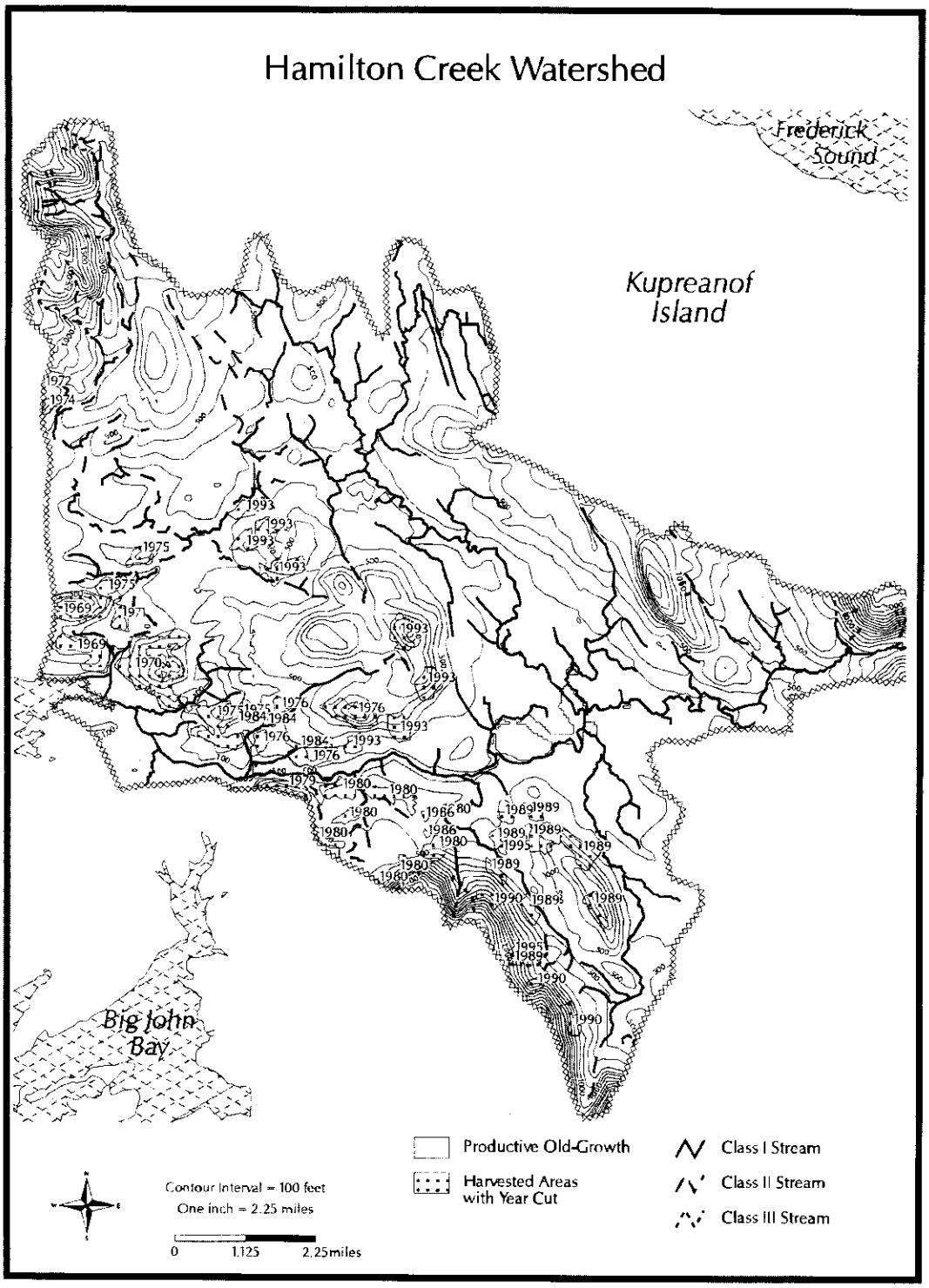

Figure 6. Timber harvest, dates of harvest, and remaining old-growth forest on the Hamilton River watershed, Kupreanof Island.

TABLE 2. Kilometers and (percent) of anadromous and non-anadromous streams within 100 m of a timber harvest unit.

Year Harvested	Anadromous (Class I) Streams	Resident Class II Streams	Total
1950-1970	880.3 (44)	326.8 (32)	1207.9 (40)
1971-1982	678.0 (34)	372.6 (38)	1051.2 (35)
1983-1990	114.4 (6)	127.0 (13)	241.5 (8)
1991-1995	319.8 (16)	189.1 (19)	509.2 (17)

for individual watersheds (Figure 7). As a result, any declines in salmon production in individual watersheds have gone undetected.

High marine survival rates during the past decade have been implicated in increased catch and escapement to all watersheds (Hofmeister et al. 1988; Shaul 1994; Beamish et al. 1997). Changes in marine survival will have a large effect on adult returns and increased marine survival will offset decreased freshwater survival as a result of habitat loss in individual watersheds. For example, estimates from Hugh Smith Lake in southeast Alaska were 51,789 smolt for 1983 and 23,499 smolt for 1985; the estimated marine survival rates for the two years were 7.4% and 19.1%, respectively (Shaul 1994). Adult returns, using these two survival estimates, were 3,832 fish in 1983 and 4,488 fish in 1985. The higher marine survival in 1985 produced a higher return from 50% fewer smolts. An increase in marine survival can easily mask a 50% loss of juvenile salmonid production in freshwater.

The effects of logging on salmon populations under various management prescriptions and particularly in watersheds with poor protection of riparian and stream habitat have been documented and reviewed in several sources (Salo and Cundy 1987; Meehan 1991; Hicks et al. 1991; Bisson et al.1992; Woodsmith and Buffington 1996). Habitat loss and reduced survival of anadromous salmonids have been documented in several studies of

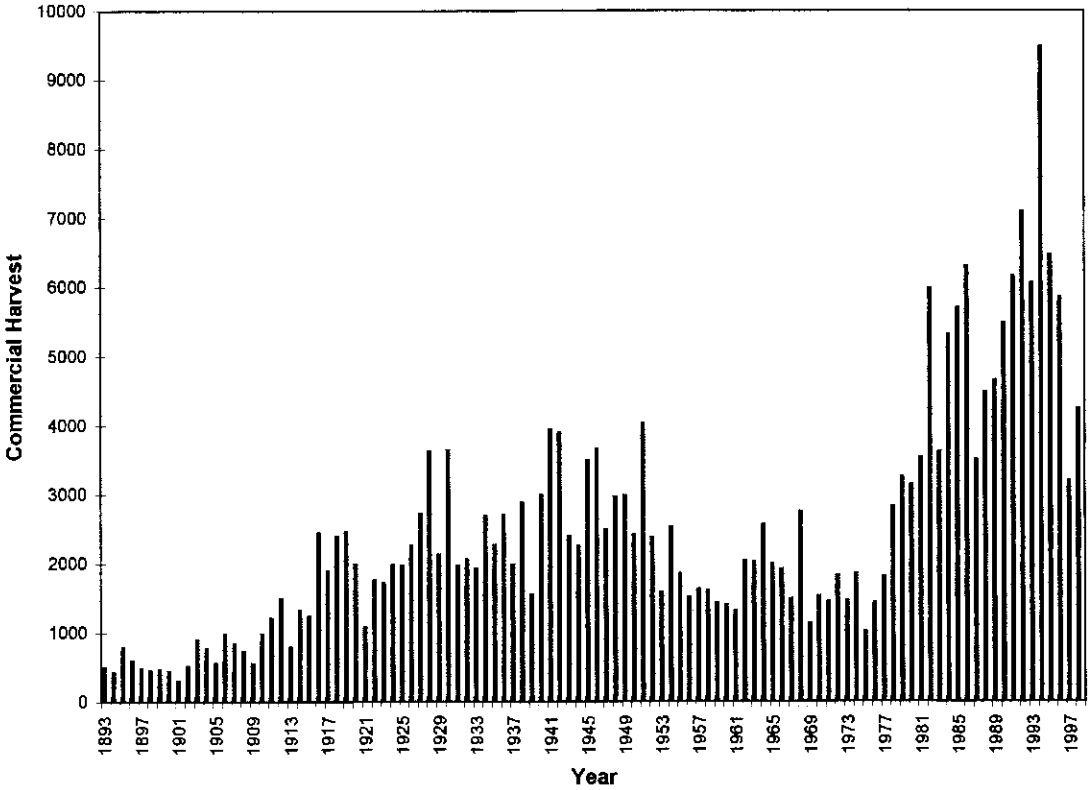

Figure 7. Annual commercial harvest of coho salmon in southeast Alaska from 1900 through 1992 (Rigby et al. 1991 and unpublished data Alaska Department of Fish and Game, Division of Commercial Fisheries, Juneau, AK).

watersheds in southeast Alaska that were logged before 1980 (Bryant 1985a; Thedinga et al. 1989; Montgomery et al.1995). The common theme among these studies is loss of stream channel complexity and simplification of habitat. However, even in intensively logged watersheds, populations of anadromous salmonids remain, but these populations are closely linked with complex habitats maintained by large wood (Dolloff 1983; Bryant 1985a). These complex habitats have been maintained by large wood that was present in streams before timber harvest. For example, in severely impacted watersheds, such as Maybeso Creek and the Harris River, large wood from old-growth trees that fell into the stream before logging provide the only anchors for large wood accumulations. The residence time of large trees in stream channels of southeast Alaska spans decades (Swanson et al. 1984; Murphy and Koski 1989) and the effects of their loss will be gradual and appear decades after the cessation of timber harvest in the watershed (Bryant 1980). A commonly accepted misconception is that stream recovery begins shortly after the cessation of activity in the watershed. In reality, the loss of habitat quality is likely to continue for more than 100 years after logging until riparian trees become large enough to maintain stream channel complexity (Bryant 1980; 1985a; Beechie et al. 1994).

Persistence of fish in some of the more severely affected watersheds can be attributed to escapements which fully seed available habitats, and the ability of some species of salmonids, coho salmon in particular, to exploit marginal habitats. The presence and expansion of beaver pond habitats in intensively logged watersheds may be an important factor maintaining coho salmon population in some of these watersheds (Bryant 1985b; Sampson 1994). Small, low gradient 2nd to 3rd order tributaries, often less than 2 meters in width, are common rearing habitat for juvenile coho salmon in southeast Alaska (Elliott and Hubbartt 1978). The size of the tributaries may allow retention of smaller wood debris (Bilby 1985). Small streams with greater amounts of large wood in intensively managed watersheds maintain higher densities of juvenile coho salmon than those with smaller amounts of wood and few pools (Dolloff 1983). The persistence of salmonids in larger watersheds is maintained by complexity that includes both instream structure, most of which is related to residual large wood, and heterogeneity of habitats within watersheds which includes a mixture of off-channel habitats, beaver ponds, and small tributaries.

On a region-wide scale, the existence and retention of intact watersheds may be a significant factor in maintaining the current status of salmon stocks in southeast Alaska. More than 75% of the Tongass National Forest is classified into Land Use Designations (LUD) I or II which include wilderness areas, National Monuments, and roadless areas (Tongass Land Management Revision 1997). This includes a substantial part of Admiralty Island and the Misty Fjord which are designated as National Monuments. Although large portions of the area in LUD I and LUD II categories are alpine, glacial, or areas of marginal economic value with respect to timber harvest, numerous highly productive watersheds in old-growth forests are also included in these designations. The watersheds in these protected landscapes represent the largest group of intact watersheds in the northern hemisphere. Other designations are LUD III which is primarily recreation use and LUD IV which allows intensive resource use such as timber harvest and mining.

The existence of these low impact land use designations (LUD I, III, and III) has resulted in a relatively high percent of anadromous streams (56%) —measured by length— that are located in watersheds that have not been affected by timber harvest (Figure 8). A total of 9,911 km (6,160 miles) (or 44%) of anadromous stream flow through watersheds that have been exposed to varying degrees of timber harvest. Of this group of watersheds, 6% of the stream length is located in watersheds that have had more than 20% of the area logged. These are likely to be the most severely affected stream systems. The largest percent of anadromous stream length is in watersheds with less than 20% of the area logged (Figure 8). While it is likely that habitat has been lost in these systems, especially in the most intensively harvested watersheds such as Maybeso Creek (Bryant 1980; 1985a), a decrease in the number of fish may be obscured by natural fluctuations and by production of fish from intact streams on a region-wide basis.

We propose the following hypothesis: 1) anadromous salmon production—measured by smolt output—will be more variable in intensively logged (>20% of area) watersheds than in intact

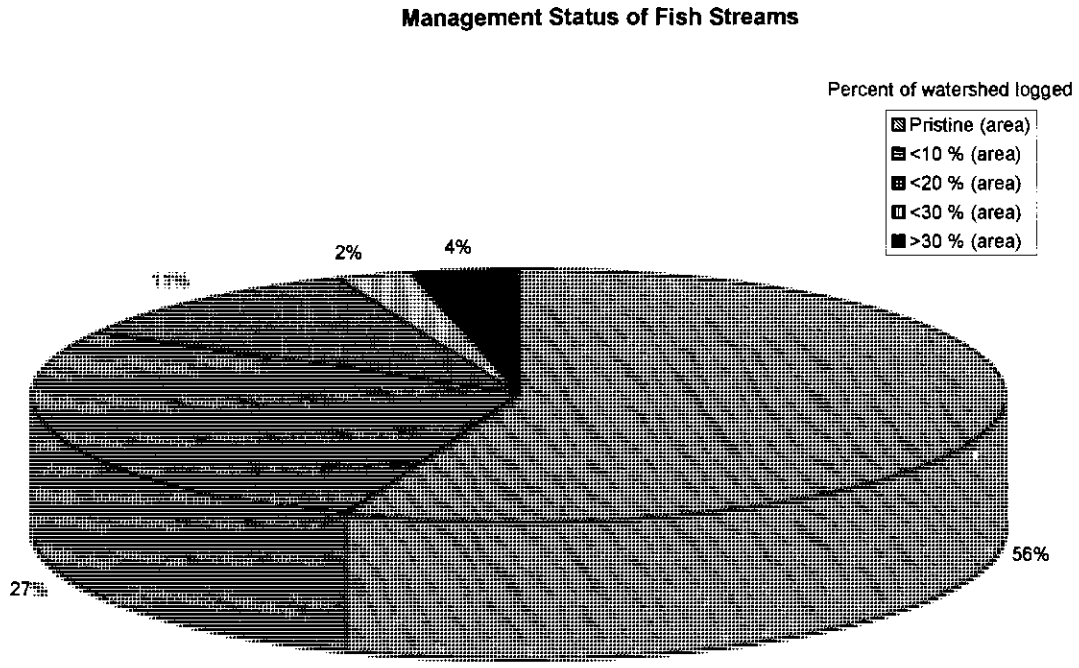

Figure 8. Relative length (%) of anadromous streams flowing through watersheds with varying percent of area logged in the Tongass National Forest.

watersheds; 2) decreases in marine survival, including natural and fishing mortality, will contribute to greater variability of smolt production in intensively managed watershed than in intact watersheds; and 3) environmental disturbances—droughts, severe floods, landslides—will have a proportionally greater effect on variability in intensively managed watersheds than in intact watersheds. The unifying theme in these hypothesis is the test of resiliency of watersheds and the prediction that intensively managed watershed will be less resilient than intact watersheds. The outcome is that salmon production in watersheds that have been intensively logged such as those during the first 30 years of industrial timber harvest in southeast Alaska may not be measurably different than many intact watersheds during benign environmental conditions and high ocean productivity, but are likely to be less resistant to adverse effects during periods of environmental stress. The streams in the unmanaged (i.e. pristine) watersheds are likely to be the most effective buffer for freshwater production of anadromous salmon region-wide.

Summary

Although "Best Management Practices" have substantially improved during the past 40 years of commercial timber harvest in southeast Alaska, most timber harvest on the Tongass National Forest occurred during the first 20 years of industrial logging and was disproportionately distributed in the valley bottoms before adequate riparian protection was in place. As a result, a legacy of watersheds with second-growth vegetation with limited potential for recruitment of large wood, as well as other adverse effects rising from poor timber harvest practices remains. While loss of salmonid production from these watersheds may be significant, it has been undocumented. However, more than 50% of the length of anadromous stream habitat remains in pristine watersheds of the Tongass National Forest. The number and distribution of intact watersheds in the TNF are critical elements for sustainable salmon populations in the face of habitat loss elsewhere in southeast Alaska and the Pacific Northwest.

The catastrophic declines in salmon stocks that have occurred in other areas of the Pacific Northwest are the result of multiple effects that include urban and agricultural development, timber harvest practices, and large dams such as those on the Columbia River which may have been exacerbated by cyclical decreases in marine productivity (Bisson et al. 1992). Furthermore, the patchwork of landowners along nearly all the watersheds of the Pacific Northwest complicates watershed management and restoration (Kohonen 1996). The multiple effects present in the Pacific Northwest are generally absent in southeast Alaska; therefore, the prospect of sustainable salmon stocks in southeast Alaska is good, but depends on a combinations of factors. The retention of existing intact watersheds is a key element. Improving riparian management in managed watersheds is important to prevent further loss of aquatic productivity. A systematic and well-planned and monitored watershed restoration effort is possible and could accelerate the recovery of damaged watersheds. A continuing conservative harvest strategy on salmon that accommodates changes in marine and freshwater conditions to allow spawning escapements that maintain freshwater productivity is essential.

Globally, few intact watersheds remain in the northern hemisphere and nearly all are in the northern regions of the hemisphere (Dynesius and Nilsson 1994; Arthington and Welcome 1995). Most of the remaining intact watersheds of the temperate rain forest occur in southeast Alaska. As the land base within southeast Alaska is further separated into old-growth forests and nonforest land, the relative amount of unmanaged old-growth forest becomes smaller. The relatively small proportion of old-growth forests supports some of the most productive salmonid habitats in North America, and a disproportionately large segment of the salmonid fishery value for the Pacific Northwest. Furthermore, these watersheds represent calibration sites for management of disturbed watersheds within the northern temperate zone and are of global significance.

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Global climate change and potential effects on Pacific salmonids in freshwater ecosystems of southeast Alaska

M. D. Bryant

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Abstract General circulation models predict increases in air temperatures from 1°C to 5°C as atmospheric CO_2 continues to rise during the next 100 years. Thermal regimes in freshwater ecosystems will change as air temperatures increase regionally. As air temperatures increase, the distribution and intensity of precipitation will change which will in turn alter freshwater hydrology. Low elevation floodplains and wetlands will flood as continental ice sheets melt, increasing sea-levels. Although anadromous salmonids exist over a wide range of climatic conditions along the Pacific coast, individual stocks have adapted life history strategies—time of emergence, run timing, and residence time in freshwater—that are often unique to regions and watersheds. The response of anadromous salmonids will differ among species depending on their life cycle in freshwater. For pink and chum salmon that migrate to the ocean shortly after they emerge from the gravel, higher temperatures during spawning and incubation may result in earlier entry into the ocean when food resources are low. Shifts in thermal regimes in lakes will change trophic conditions that will affect juvenile sockeye salmon growth and survival. Decreased summer stream flows and higher water temperatures will affect growth and survival of juvenile coho salmon. Rising sea-levels will inundate low elevation spawning areas for pink salmon and floodplain rearing habitats for juvenile coho salmon. Rapid changes in climatic conditions may not extirpate anadromous salmonids in the region, but they will impose greater stress on many stocks that are adapted to present climatic conditions. Survival of sustainable populations will depend on the existing genetic diversity within and among stocks, conservative harvest management, and habitat conservation.

M. D. Bryant (✉)
Southern Research Station, P.O. Box 227, Stoneville, MS 38776, USA
e-mail: mdbryant@fs.fed.us

1 Introduction

Global temperatures are increasingly driven by greenhouse gas accumulation over the last 50 years (IPCC 2007). The response of Pacific salmon in Alaska to climate change is of ecological and social consequence. Alaska supports the greatest proportion of salmon harvested along the west coast of North America (Clark et al. 2006; Meacham and Clark 1994; Howe et al. 2001; Shaul et al. 2003). In contrast to most salmon stocks along the west coast of the US, most stocks in southeast Alaska maintain sustainable harvests and escapements (Baker et al. 1996; Halupka et al. 2000). The condition of salmon stocks in Alaska can be attributed largely to the absence dams, agriculture, and urbanization that have contributed to the decline of salmon stocks elsewhere (Nehlson 1997). Over-harvest has been an issue in the past in southeast Alaska (Meacham and Clark 1994) and in some watersheds and timber harvest has been and is still a concern (Koski et al. 1966; Murphy et al. 1986; Thedinga et al. 1989). However, the large land base, low human population, and numerous intact watersheds throughout southeast Alaska mitigate these effects on a region-wide scale (Bryant and Everest 1998). Furthermore, the large number of stocks in southeast Alaska represents an immense genetic reservoir that provides resiliency across the region for anadromous salmonid stocks (Halupka et al. 2003). The genetic diversity is manifested in a wide range of phenotypic adaptations to both small and large scale features. This biocomplexity is an important factor in the resilience of salmon stocks to a range of natural and anthropogenic disturbances (Hilborn et al. 2003).

The genetic and phenotypic diversity represents stocks that have adapted over centuries to environmental conditions particular to geographic location—north to south, mainland to island, east to west—and watershed conditions (Halupka et al. 2000). The physical geometry of watersheds and landscapes provides a template for some of these differences as does stock separation and isolation (Waples 1991). In addition to these, climatic factors play an important role in the development of specific stock characteristics. Temperature and hydrology control several critical stages in the life cycle of salmonids. During periods of rapid climate change, these can have significant effects on anadromous salmonid populations.

Southeast Alaska supports a diverse set of salmonids that depend on freshwater ecosystems. The five Pacific salmon include pink (*Oncorhynchus gorbuscha*), chum (*O. keta*), coho (*O. kisutch*), sockeye (*O. nerka*), and chinook (*O. tshawytscha*) salmon. These form the basis of a significant commercial, sport, and subsistence fishery. Steelhead and rainbow trout (*O. mykiss*), cutthroat trout (*O. clarki*), Dolly Varden (*Salvelinus malma*) are important in varying degrees to both sport and subsistence fisheries. These fish employ a wide range of life histories and their response to climate change will be complex and differ among species.

Life history strategies of anadromous salmonids in Alaska separate into two groups on the basis of how they use the freshwater environment (Groot and Margolis 1991). Pink and chum salmon spawn in freshwater and the fry emerge from the gravel from April through May and migrate to the ocean in a few weeks (Fig. 1). Pink and chum salmon often spawn in the lower reaches of rivers; however, some chum salmon may migrate considerable distances upstream in rivers such as the Yukon and Frazier (Salo 1991). Sockeye, Chinook, and coho salmon spawn in freshwater, but the fry typically spend one or more years in freshwater after they emerge from the gravel.

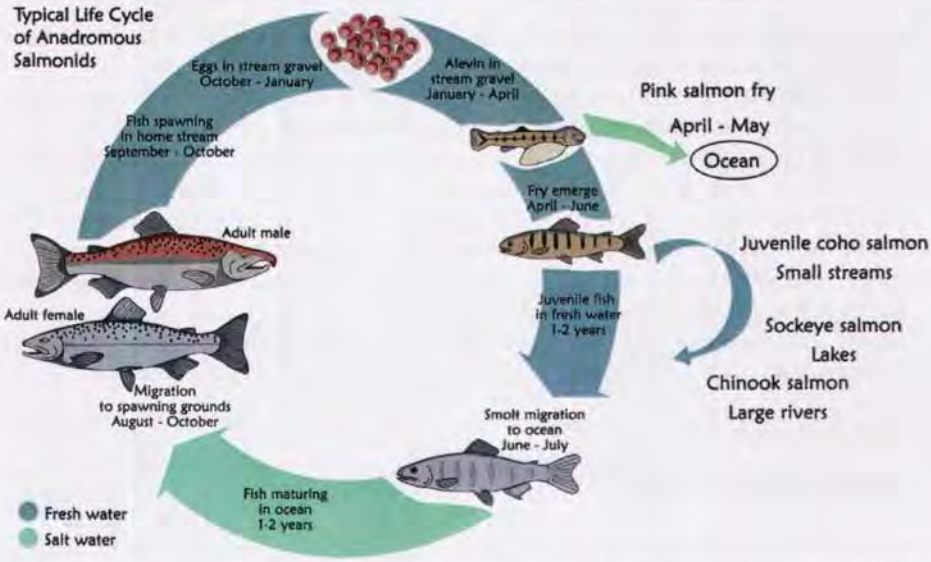

Fig. 1 Life cycle strategies of the five species of salmon (*Oncorhynchus*) found in southeast Alaska with those that rear in freshwater and those that migrate directly to the ocean

As a general pattern, Chinook salmon spawn in large rivers, coho salmon use small streams and sockeye salmon spawn in streams associated with lakes and to a lesser degree along lake margins. Juvenile sockeye salmon usually rear in lakes from 1 to 2 years (Burgner 1991; Quinn 2005).

Extreme variation occurs within the group, for example, some coho salmon juveniles may use lakes (Bryant et al. 1996). Some Chinook and sockeye stocks have subgroups that may migrate to the ocean shortly after emergence (Groot and Margolis 1991). Small populations of Chinook salmon are found in some small streams on Admiralty Island (Halupka et al. 2000). Scott and Crossman (1973) and Mecklenburg et al. (2002) describe the various life histories for each group. These variations are significant in as much as they illustrate the plasticity of salmonids to adapt to a range of environmental conditions and may be important in an environment of rapidly changing climatic conditions.

Salmon populations respond to variation in climate patterns in the marine environment (Beamish et al. 1999). Ocean temperatures driven by the Pacific Decadal Oscillation (PDO) have been shown to affect their survival (Hare and Mantua 2000; Mueter et al. 2002). Among other things, their results indicated that northern stocks, including Alaska, had higher survival rates during warmer periods, whereas, the southern stocks had lower early marine survival during warm periods. Similar results are reported by Hare et al. (1999). Sockeye salmon average length appeared to decrease with increasing temperatures (Pyper and Peterman 1999). However, ocean surveys of salmon populations suggest that thermal boundaries limit southern feeding areas in the Pacific Ocean. As the warm thermal boundary moves northward with global warming, ocean foraging area will decrease (Welch et al. 1998a, b).

Similar studies of the effects of changes in temperature regimes on Pacific salmon populations in the freshwater environment are limited.

This assessment examines potential responses of the five species of Pacific salmon found in southeast Alaska to climate change. The focus is on the freshwater phase of the life cycle of each species and potential effects of climatic changes. The three major climate change variables are temperature, precipitation (hydrology), and sea level. The purpose of this assessment is to identify potential effects and responses by each species, and to propose questions that can be used to quantitatively evaluate some of the effects and responses. Although the emphasis of the assessment and examples of potential effects are drawn from Pacific salmon stocks in southeast Alaska, the responses and effects may apply to stocks throughout their range. In regions where stocks are stressed potential responses may be more severe.

2 Climate change scenarios

Most assessments of the effects of climate change on both marine and freshwater ecosystems are based on general circulation models (GCM) that are large scale and complex (Smith and Tirpak 1989; Kalkstein 1991; IPCC 2007). Hauer et al. (1997) discuss the limitations of these models in their examination of climate change and freshwater ecosystems of the Rocky Mountain region of the USA. They use a methodology that nests GCMs into regional climate models (RCM) to simulate climate changes for their assessment of freshwater ecosystems of the Rocky Mountain region. In all scenarios commonly used in GCMs, increasing concentrations of CO_2 in the atmosphere result in increased atmospheric temperatures (Magnuson et al. 1990; Hauer et al. 1997; Schindler 1997). A RCM specific for Southeast Alaska is needed, but GCMs generally predict an increase from 1°C to 4.5°C during the next 100 years and that the effect will be greater near polar regions (Hengeveld 1990).

Three major effects are likely from increasing atmospheric temperatures. Temperatures in ocean and freshwater habitats will increase, precipitation intensity and distribution will change, and sea level will rise. The response of precipitation depends on location and season. Inland regions may experience increased drought, whereas, coastal regions may receive more rain during the fall and winter. One scenario suggests that intensity of rainfall is likely to increase (Trenberth 1999). Changes in winter snow pack and the time of spring melt will also occur (Stewart et al. 2004). As ocean temperatures increase, sea level is projected to increase (Wigley and Raper 1993). However, the increase of sea level relative to land is complicated by isostatic rebound (Peltier 2001).

Changes in temperature and precipitation are likely to have important consequences in freshwater ecosystems. In their systematic assessment of the effects of climate change in freshwater ecosystems of the Rocky Mountains, USA, Hauer et al. (1997) identify changes in community structure as temperatures increase in both streams and lakes with elimination or isolation of cold water fish species. They also point to the intensification of anthropogenic effects that include habitat degradation and introduction of non-native species. Eaton and Scheller (1996) estimate that habitat for cold and cool water fish could be reduced by about 50% under most scenarios in US streams. Using a bioenergetics model, McDonald et al. (1996) predicted a substantial decrease in lake trout in a lake in northern Alaska resulting

from increased metabolic demands from higher temperatures accompanied by either reduction or no increase in food supply available to young of the year trout. In streams, increases in thermal regimes may reduce or eliminate cold water pool refuges important during the summer for cold water fish (Bagiun et al. 2000).

Various GCMs predict that mean temperatures will increase and most studies that examine fish response to an increase in temperatures within the predicted ranges show a trend of decreasing thermal habitat for cold water fish in both lakes and streams (Hauer et al. 1997; McDonald et al. 1996; Meisner 1990; Schindler 1997). In southeast Alaska, stream water temperatures range from freezing to about 20°C and generally are less than 10°C in most streams during most of the year (Fig. 2). Anadromous salmon in southeast Alaska are cold water species and have spawning, incubation, and growth characteristics that are tuned to these temperature ranges (e.g. Hodgson and Quinn 2002).

Precipitation directly influences hydrology, and stream and river discharges respond to precipitation. Salmonid life histories are intimately connected to discharge timing and stage in streams and rivers. Spawning generally occurs during periods of high flows for most species and changes in discharge patterns can adversely affect spawning success (Moscrip and Montgomery 1997). Coho salmon production has been shown to be related to discharge (Smoker 1955). Timing of high flows is critical both to spawning migration and to emigration of smolt in the spring and spring discharge regimes are often highly dependent on snow accumulations during the winter (Mote et al. 2003; Lawson et al. 2004; Stewart et al. 2004). Summer habitats are maintained by suitable flow regimes for stream rearing species and fall freshets provide access to off-channel refuges for over-winter habitats (Peterson 1982). Poff and Allan (1995) found ecological relationships between fish assemblages and stream flow variability, which would be modified with changes in hydrological regimes induced by climate change. Alterations in stream flows may also affect lake catchments. Schindler (1997) suggests that acidification may occur in streams accompanied by alkalization of lakes with decreasing discharge regimes, depending upon the surrounding geochemistry of the land base.

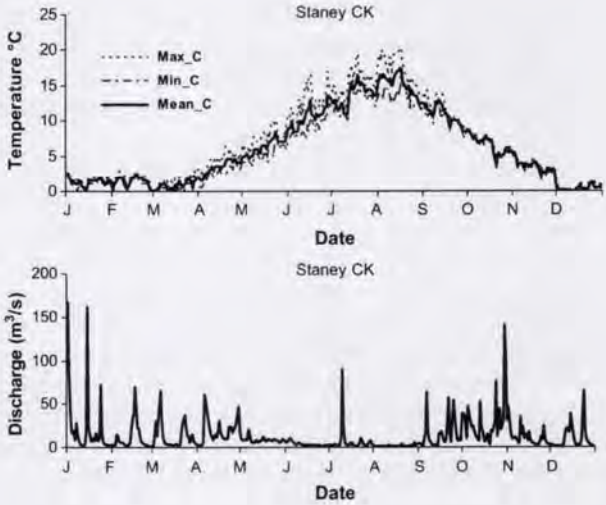

Fig. 2 The average daily mean, maximum, and minimum temperature and discharge for Staney Creek, Prince of Wales Island for 2007 (USGS)

Shifts in the amount, intensity, and form (snow vs. rain) of precipitation will alter the hydrological regimes of streams in southeast Alaska. Warming trends will also affect streams and rivers fed by glaciers. Most aspects of the life history of salmon are linked to hydrological patterns and seasonal freshets. Pacific salmon spawn during periods of high flows during the fall and fry migrate to the ocean as spring flows increase with snowmelt. Species rearing in fresh water depend on maintenance flows during the summer that may be sustained by snow melt and ground water sources that depend on snow melt at higher elevations (Hauer et al. 1997; Harr and McCorison 1979). The terrain in southeast Alaska is mountainous and weather patterns can be highly localized; however, stream and river discharge patterns are linked to precipitation and climatic events (Neal et al. 2002). Neal et al. (2002) observed lower summer flows and higher winter flows in an old-growth river in southeast Alaska for warmer periods than colder periods during Pacific Decadal Oscillations (PDO). A similar response could be expected with warmer atmospheric temperatures as a result of global climate change.

Sea level is linked to climate change and is projected to increase with global warming (Warrick et al. 1993; Douglas et al. 2001). A wide range of factors can influence sea level and historical evidence reviewed by Gornitz (1993) shows substantial variation in mean sea level during the past 200 years. Predictions of net increase in sea level vary widely and are likely to vary locally with glacio-isostatic and tectonic activity (Gornitz 1995). Most of the increase in sea level is expected to occur from melting of the large land-based ice sheets in Greenland and Antarctica (Alley et al. 2005). They predict an increase of about 70 m if these completely melted; however, more common ranges are from no change to slightly more than 1 m by 2100 (Neumann et al. 2000). More recent models indicate that sea-level may increase between 0.5 and 1.4 m by 2100 (Rahmstorf 2007). The response of sea-level to atmospheric temperature increase is complex and may exceed current projections.

Neumann et al. (2000) cite assessments that estimate flooding of 18,000 km^2 of coast land in the US with an increase in sea level of 0.5 and 35,000 km^2 with an increase of 1.0 m of sea level. The social and economic consequences are huge and the effects on wetlands and associated ecosystems are substantial (Titus 1990). Galbraith et al. (2002), using a series of rates of increase in sea level over 100 years, predicted a loss of 20% to 70% of habitat for shore birds at four sites along the coast of the US. In coastal freshwater ecosystems, changes in mean sea level can affect salmonids in freshwater habitats. A large proportion of both spawning and rearing habitat for anadromous salmonids occurs in low elevation flood plains that could be flooded by rapid increases in mean sea level.

Rising sea levels will flood low elevation habitats converting freshwater habitats into brackish or saline environments (Titus 1990). Habitats above the immediate effects of flooding will become inter-tidal and subject to periodic tidal flooding and pulses of saline water. Habitats not directly affected by incursion of salt water and tides are likely to be affected by channel readjustment resulting from changes in sediment deposition or scour (Blum and Tornqvist 2000). In other geographic locations, loss of wetlands through rising sea level may be compensated by conversion of low gradient uplands (Titus 1990). Throughout most of southeast Alaska, the terrain is precipitous and rises rapidly to elevations above 1,000 m from coastal margins (Nowacki et al. 2001) and replacement of low gradient habitats along coastal margins is not likely to compensate for habitats lost to marine incursions (Fig. 3).

Fig. 3 Southeast Alaska topography from shuttle radar topography with elevations 20 m (best DEM resolution for all of southeast Alaska is 20 m) *shaded dark gray* and intertidal area *shaded light gray*

In Alaska, the amount of increase relative to the present land mass is complicated by glacio-isostatic rebound and tectonic activity such as the 1964 earthquake. Glacio-isostatic rebound also varies from location to location throughout southeast Alaska (Larsen et al. 2005). In recently de-glaciated areas such as Glacier Bay, land emergence with respect to sea-level has been reported at 3.96 cm/year (Hicks

and Shofnos 1965). Recent observations show uplift of 10 mm/year near Glacier Bay decreasing southward to 2 mm/year, varying from location to location (Larsen et al. 2005). Therefore, prediction of the actual increase (or decrease) of sea level in relation to land is difficult.

Each salmon species in southeast Alaska has a unique life history that can be affected in different ways and intensity by rapid changes in environmental conditions. Some effects are likely to be common across two or more species and some effects may be more severe on one species than another.

3 Potential consequences of climate change on salmonids

3.1 Pink and chum salmon

Pink and chum salmon enter freshwater to spawn during the late summer when stream temperatures are close to the maximum for the season. Mean run times (approximately the peak time of entry into freshwater) range from July 25 through September 10 (Halupka et al. 2000). Usually the highest temperatures occur during August and begin to decrease in September. In more southern streams such as Staney Creek temperature can exceed 20°C (Fig. 2). They enter streams in large numbers over several weeks where they often hold in deep pools and move upstream following periods of rain and pulses in stream flow. The consequences of increasing temperatures are inter-related with decreases in summer precipitation and low flow. Low flows will delay migration and fish will hold in pools at various locations in freshwater and inter-tidal areas. Low flows and high temperatures usually coincide in August (Fig. 2). During periods of low flows extending for several weeks, the combination of high temperatures, low flows, and high fish density rapidly depletes dissolved oxygen content of stream water leading to high pre-spawning mortality (Murphy 1985; Pentec Environmental 1991). Under present temperature regimes, large pre-spawner mortality events are not common occurrences and generally occur in the southern region of southeast Alaska. As temperatures increase and summer flows decrease, these events are likely to become more common and occur in more northern watersheds of southeast Alaska.

Pink and chum salmon fry migrate to the ocean shortly after they emerge from the gravel (Heard 1991; Salo 1991). Time of emergence is related to temperature during incubation (see references in Heard 1991 and Salo 1991) and changes during the early part of incubation can affect time of emergence. Fry of both species migrate to the ocean in large schools, particularly pink salmon, at small size (<50 mm). Timing of their migration is generally thought to be related to food resources in inter-tidal areas (Mortensen et al. 1999 and references cited therein). As temperatures increase, egg incubation rates will increase and time to emergence and migration will decrease for both species and entry into the ocean will occur earlier in the season. Presently, both species enter the near shore marine environment beginning in late March and continuing through May when productivity is higher than in January or February. Early entry into the marine environment when food resources are low or absent will decrease growth and survival. Spawning later in the season may compensate for shorter incubation times.

Landslides, debris avalanches, and mass soil movement can have severe consequences on salmon eggs and embryo during their residence in streambeds (Everest

and Meehan 1981; Tripp and Poulin 1986a; Swanson et al. 1987). In southeast Alaska, these events are closely associated with intense rainfall on saturated soils (Swanston 1970; Johnson et al. 2000). A frequent outcome of several global climate change models is a shift to more frequent high intensity precipitation events where precipitation increases (Hauer et al. 1997; Melack et al. 1997; Trenberth 1999). In erosion prone landscapes, landslides and mass failures are likely to increase, which will increase egg and embryo mortality as landslides scour redds where they run through spawning locations and deposit fine sediment on downstream locations (Lisle 1989; Swanson et al. 1987; Tripp and Poulin 1986a, b).

Pink and chum salmon that spawn in low salinity inter-tidal reaches of rivers or in the lower reaches of rivers and streams will be susceptible to changes in sea level and tidal influence even with relatively small changes. As sea level rises the deposition zones for sediment at the river outfall will change (Blum and Tornqvist 2000). The response is complicated by basin morphology, upslope sediment supply, and local rates of isostatic rebound or subsidence (Blum and Tornqvist 2000). These processes will affect spawning locations used by pink and chum salmon that spawn in the lower reaches of rivers.

3.2 Sockeye salmon

Small populations, relative to those in western Alaska, of sockeye salmon occur throughout southeast Alaska in watersheds with lakes of varying sizes. They usually enter freshwater to spawn from late summer through early fall. However, they usually enter and hold in lakes before they spawn in streams. Some may spawn along lake margins. Mean summer temperatures in lakes throughout Alaska seldom rise above 14°C and are below 10°C most of the time (Cartwright et al. 1998; Rogers and Rogers 1998; Westley and Hilborn 2006). Changes in temperature and discharge regimes may shift spawning timing and time of emergence of sockeye fry with unknown effects on growth and survival. Low flows may impede access to lakes and spawning locations. Higher water temperatures in lakes can also increase stress on adult sockeye holding in lakes. More high intensity rainfall events in the fall that increase the number of landslides in upstream high gradients streams will increase sediment deposition in both spawning streams and in lakes where sockeye rear.

Temperature increases are likely to affect growth and survival of sockeye fry that rear in lakes for one to two summers. Increasing temperature in the epilimnion zone may increase growth rates for juvenile salmonids. McDonald et al. (1996) found that food requirements for young-of-the-year lake trout increased as metabolic requirements increased with temperature in a lake in interior Alaska. However, food resources (zooplankton) did not increase and their model predicted a decline in lake trout populations with increasing temperature. Similar studies have not been done for lake rearing sockeye salmon. Climate changes can also affect the trophic structure of lakes. Winder and Schindler (2004) observed shifts in the time of vernal and autumnal mixing and the length of summer stratification associated with PDOs. The time of stratification increased and vernal mixing occurred earlier with increasing temperature in PDO cycles. Spring peak in phytoplankton and zooplankton shifted with changes in the thermal dynamics of Lake Washington.

The effects of changes in precipitation and hydrology on lake rearing sockeye are likely to be less direct, but may affect nutrient dynamics, turbidity, and temperature

regimes which in turn will affect productivity. Carpenter et al. (1992) discuss the effects of water renewal on lake ecosystems. Increased precipitation can increase nutrient inputs and amplify anthropogenic disturbances such as urban and agricultural runoff. Decreased precipitation and increased evaporation may increase concentrations of chemicals and nutrients resulting in increased primary production. Coupled with decreased water turn-over, this could lead to severe reductions in hypolimnetic oxygen. Although most of the severe anthropogenic effects are absent in southeast Alaska, greater evaporation and lower water replacement can magnify acidification in lakes. This could have significant consequences for sockeye salmon rearing in lakes located in watersheds of southeast Alaska with large areas of acidic wetlands.

Predation can have a substantial effect on juvenile sockeye salmon in lakes (Cartwright et al. 1998). Alterations in the trophic status of lakes may also change dynamics of other species. The effect of higher temperatures in Alaskan lakes on cutthroat trout is not known; however, higher temperatures may increase metabolic requirements for cutthroat trout and increase predation rates on juvenile sockeye salmon.

Small lakes throughout southeast Alaska that support populations of sockeye salmon are located near salt water and are often connected to salt water by short streams. As the sea level rises, lakes at lower elevations may be subject to salt water intrusion as higher tidal cycles push salt water into the lake. As heavier salt water enters these lakes, it will sink to the bottom creating a layer of salt water. This will create a stable chemocline in the bottom layer of water that will effectively isolate the bottom surface of the lake from mixing during vernal and autumnal isothermal periods (Hutchinson 1957). The result will decrease productivity of the lake as nutrients are isolated in bottom sediments and are not released when vertical mixing occurs in the spring and fall (see Winder and Schindler 2004).

3.3 Chinook salmon

Most populations of Chinook salmon that spawn in southeast Alaska are less than 1,000 fish (Halupka et al. 2000). Larger numbers migrate through the large transboundary rivers, such as the Taku, Stikine, and Alsek and spawn in the interior in Canadian territory (Halupka et al. 2000). Climatic conditions in the interior are different from those in coastal Alaska and the manifestations of climate change will be different as well (Rouse et al. 1997). For example, it is possible that warmer drier conditions may occur in interior habitats than those in coastal habitats. Juvenile Chinook salmon from upstream spawning locations migrate into coastal river habitats where they will be exposed to changes imposed on the coastal climate (Murphy et al. 1989; Pahlke 1995). However, in both locations the general trend is expected to be warmer. Chinook salmon travel long distances to spawning grounds and increased temperatures will increase metabolic costs during migration (Berman and Quinn 1991; Torgersen et al. 1995).

As with most salmonids, Chinook salmon egg development responds to temperature changes and an increase in incubation temperature will decrease the amount of time to hatch and subsequently fry emerge from the gravel earlier (Alderdice and Velsen 1978; Beer and Anderson 2001). Higher temperatures and early emergence may affect feeding and growth (Heming et al. 1982) with subsequent effects on

survival and time of smoltification (Berggren and Filardo 1993; Taylor 1990). Early emergence may alter life history patterns.

Life history strategies of Chinook salmon appear to be related to geographic location, distance from saltwater, and growth. Chinook salmon display two life history strategies in freshwater that occur in southeast Alaska (Healey 1991; Halupka et al. 2000). Ocean-type fish migrate to sea after 2–3 months in freshwater; stream-type fish remain in fresh water for one or more years. The fish with the stream-type life cycle tend to rear further upstream, experience slower growth and are located in more northern locations (Taylor 1990). In southeast Alaska, both strategies often occur in the same river system from apparently similar stocks (Murphy et al. 1989). Increasing temperatures may increase the occurrence of ocean-type fish in these rivers. Beechie et al. (2006) observed that life history strategies of Chinook salmon in Puget Sound responded to hydrologic regimes. They reported that stream-type fish were more common in watersheds dominated by snowmelt and suggest that their numbers would decline with decreasing snowmelt with global warming. It is unclear if this would be the case for populations in southeast Alaska, but it does indicate that temperature changes may cause a shift in life history strategies for Chinook salmon. In most cases, higher marine survival may be expected from stream-type smolt that migrate to the ocean at a larger size. However, there is a trade-off with survival probabilities in fresh water.

The effects on fish rearing in the large trans-boundary rivers will be complex as the effects from weather patterns in the interior interact with changes in coastal environments. Warmer drier weather in the interior may increase temperatures in rearing habitats for Chinook salmon in large rivers and their tributaries. Increases in temperature have been shown to affect juvenile salmon growth. In a watershed in California, Sommer et al. (2001) observed higher growth rates for Chinook salmon juveniles that reared in floodplain habitats that were 5°C higher than those in an adjacent river reach. They attribute the higher growth rates in the floodplain to higher food consumption. Beckman et al. (1998) found that Chinook grew faster at higher temperatures, but Bisson and Davis (1976) found that Chinook in artificial channels with elevated temperatures did not grow as fast as those in control channels. Many of the rivers used by Chinook salmon are fed by glaciers which would moderate potential increases in temperatures.

Dry conditions during the summer in the interior will reduce flows. Lower flows during the summer may reduce rearing habitat; however, in larger glacial rivers increased temperature during the summer may increase flows as glacial melting increases. Migration of both adult and fry may be affected by changes in flows. Healey (1991) proposed that downstream migration was initiated by spring freshets; however, the relationship is not clear in other studies. Low flows may also affect upstream migration timing (Berggren and Filardo 1993). Juvenile Chinook salmon respond to alterations in discharge regimes; however, it is unclear if it will be positive or negative with respect to survival in Southeast Alaska.

3.4 Coho salmon

Coho salmon spawn over an extended period of time in southeast Alaska. Summer run stocks may enter freshwater in late July (Bryant et al. 1999; Halupka et al. 2000). Most coho salmon migrate into freshwater spawning areas from September through

November, although some stocks may spawn as late as December (Halupka et al. 2000). During the fall, freshwater temperatures rapidly decline and flows tend to be high. Increasing stream temperatures during the fall and winter are not likely to substantially increase mortality of spawning fish; however, as with other species alterations in the temperature regime during incubation may affect development rates and emergence timing (Tang et al. 1987). In an analysis of temperature effects of clear cut logging, Holtby et al. (1989) observed that coho salmon fry emigrated from spawning locations earlier as temperatures increased; however, they did not observe a significant relationship between smolt emigration and temperature.

In southeast Alaska, most juvenile coho salmon spend two summers in freshwater (Halupka et al. 2000). Although not well documented, the 2 year freshwater residence may be attributable to the cool temperatures of most rearing habitats used by juvenile coho salmon. In most streams in old-growth watersheds, mean daily temperatures usually do not exceed 5°C until late May and by mid September are below 6°C (Figs. 2 and 4). Throughout southeast Alaska, most fry are less than 60 mm fork length by fall of their first year of freshwater residence (Thedinga et al. 1989; Bramblett et al. 2002; Bryant et al. 2004). A combination of increased temperatures and a longer period where temperatures are within physiological growth boundaries may lead to increased growth and potentially a shift from a two-summer life cycle to a one-summer freshwater life cycle. The ultimate effect on survival may depend in part upon the size of the smolt as they migrate to the ocean (Holtby et al. 1990). If one-summer smolt are smaller than two-summer smolt, marine survival may be less for one summer smolt than for two summer smolt.

Although small streams are important rearing areas for juvenile coho salmon, a substantial but largely undocumented number use off-channel habitats such as beaver ponds and sloughs (Bryant 1984b; Murphy et al. 1989; Swales and Levings 1989; Bryant 1991; Pollock et al. 2004). These are often used as over-winter habitat, but substantial numbers remain in the ponds during the summer (Bryant 1984b; Sampson 1994). Fish in the ponds tend to be larger than those in adjacent stream reaches and most are parr (age 1+) fish. Temperatures in these ponds may be 2° to 4°C warmer than the adjacent free flowing stream during the summer (Fig. 4). The response to warmer temperatures in these habitats may become less favorable as summer temperatures approach or exceed physiological limits for growth. With increasing temperatures, water quality may decrease as biological oxygen demand increases with temperature.

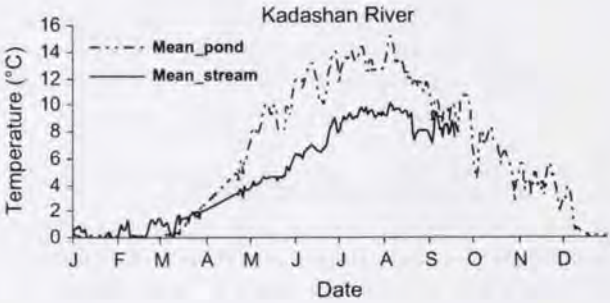

Fig. 4 Mean daily temperatures for the Kadashan River and an adjacent beaver pond for 2000

Alterations to seasonal flow regimes caused by shifts in precipitation patterns are most likely to affect the freshwater residence of juvenile coho salmon. In southeast Alaska, the summer season is short and coho salmon fry grow relatively slowly (Wright and Bryant unpublished manuscript[1]). Reduced flows during the summer restrict pool habitats and may increase density and competition among coho fry (Nickelson et al. 1992; Rosenfeld et al. 2000). Low flows that reduce the size and depth of pools accompanied by increases in temperature will impose higher mortality and reduced growth rates among coho fry. Lower flows will also decrease the amount of invertebrate drift further reducing growth (Wilzbach and Hall 1985; Hetrick et al. 1998; Hughes 1998).

Most coho salmon spawn during the fall and are not likely to be affected by reduced summer flows. However, a few stocks move into fresh water during the summer and may encounter barriers to upstream migration at low flows (Halupka et al. 2000). Spawning success may be adversely affected by intensification of precipitation events that trigger landslides, which will scour spawning areas in low gradient headwater streams often used by spawning coho salmon. Downstream locations would be subject to increased sedimentation resulting from landslides (Tripp and Poulin 1986a).

Landslides induced by high intensity rainfall events will scour streambeds and remove habitat in small streams commonly used for rearing by juvenile coho salmon (Tripp and Poulin 1986b; Cederholm and Reid 1987; Hogan 1987; Benda 1990; Tripp 1998) and remove alcoves and dammed pools may also limit over-winter survival (Nickelson et al. 1992). However, in some Oregon watersheds, landslides can be significant sources of large wood to streams (Reeves et al. 2003). In the U-shaped valleys of southeast Alaska, most landslides and debris flows deposit large wood in the moderate (4–7%) gradient zones of small tributaries and do not enter larger third and fourth order streams (Johnson et al. 2000). In most cases landslides and mass wasting events will have a deleterious effect on winter survival of juvenile coho and if they become a chronic occurrence, they will have long term effects on populations.

Juvenile coho salmon use a wide range of habitats that are common in small low elevation streams that may be affected by rising sea levels (Elliott and Reed 1974; Bryant 1984a). Habitats within 1 to 5 m of existing sea level include small tributaries in the lower reaches of watersheds (Bryant 1984a), off-channel and main stream habitats near the inter-zone (Murphy et al. 1989), and flood plain habitats in large river deltas (Bryant 1991; Schaberg 2006). Some of these areas are extensive such as the Copper River Delta and the Yakutat Forelands located a few hundred miles north of southeast Alaska (Bryant 1991; Schaberg 2006). Numerous other low elevation river delta and intertidal areas that support large numbers of several species of anadromous salmon are scattered throughout southeast Alaska (Fig. 3). The Kadashan River watershed (see inset in Fig. 3) is one example of a low level alluvial floodplain associated with an extensive intertidal area that would be susceptible to a relatively small (<1 m) increase in sea level (Fig. 5). Rising sea level will increase salinity in the lower reaches and increase sediment deposition as flows are altered by tidal effects (Blum and Tornqvist 2000). Rising sea level as a result of global warming

[1] Wright, B. E., and Bryant, M. D. Unpublished. How Wild Steelhead Win the Summer Growth Race in Southeast Alaska. Unpublished manuscript, Juneau Forestry Sciences Laboratory, USDA Forest Service, 2770 Sherwood Ln 2A., Juneau, AK 99801, USA.

Fig. 5 The lower watershed and intertidal area of the Kadashan River showing area that would be affected by an increase in sea level of 1 m

may be compensated by isostatic rebound as result of recent de-glaciation in many areas of Southeast Alaska.

4 Discussion

Many of the predicted outcomes from scenarios for climate change are not favorable for anadromous salmonids (Table 1). However, some may be positive. In several instances the outcome is not known and may depend on interacting events. In all cases, the magnitude is speculative. It is highly unlikely that there will be a wholesale extirpation of salmon stocks in southeast Alaska. All anadromous species found in southeast Alaska are also found in more southern locations with thermal regimes that might be expected in southeast Alaska under most predictions of climate change. Nonetheless, stocks with small numbers or unique habitat requirements may be lost. The small run of Chinook salmon in two streams on Admiralty Island are examples of "at risk" stocks (Halupka et al. 2000). Other examples are stocks of summer run coho salmon and small populations of sockeye salmon in small lakes.

The most pervasive anthropogenic effect on the landscape throughout southeast Alaska is timber harvest and the effects of poor management practices in the past (Heifetz et al. 1986; Murphy et al. 1986; Thedinga et al. 1989); however, a large number of intact watersheds with little or no anthropogenic disturbance on the habitat remain throughout southeast Alaska (Bryant and Everest 1998). These are important buffers to many of the effects that may be imposed by climate change. Although the harvest of salmon by commercial, sport, and subsistence fisheries appears to be sustainable under current management practices, a substantial portion

Table 1 Summary of potential effects of climate change on anadromous salmonids in freshwater habitats of southeast Alaska

Pink salmon and chum salmon
Increased frequency and extent of pre-spawner mortality resulting from increasing temperatures and decreasing summer flows
Earlier emergence time and entry into the marine environment with less favorable conditions for early feeding and growth
Deterioration of spawning habitats
Greater upslope landslide activity increasing scour and sediment infiltration
Incursion of saltwater from rising sea levels into spawning areas
Alterations in sediment dynamics with changes in sea level
Alterations in run timing as a result of shifts in temperature and discharge
Sockeye salmon
Shifts in spawning time with subsequent changes in time of emergence of fry
Spawning habitat deterioration from upslope landslides induced by increased rainfall intensity
Changes in growth and survival resulting from alteration of trophic status of lakes
Shifts in zooplankton availability
Changes in lake physical and chemical dynamics resulting from either increases or decreases in water recharge
Decreasing rearing capacity and secondary production from saltwater intrusion
Increased predation as thermal characteristics become more favorable for natural or introduced predators
Chinook salmon
Changes in run timing forced by temperature and/or discharge regimes
Increased stress and mortality during spawning migration resulting from loss of thermal refuges in large pools
Deterioration of spawning habitat caused by increased frequency of upslope landslides
Loss of rearing habitat as thermal refuges are lost
Coho salmon
Deterioration of spawning habitat from landslides that scour spawning beds and deposit sediment on downstream spawning areas
Changes in fry emergence timing and emigration
Effects of climate change induced temperatures on growth and survival of juvenile coho salmon
Increased growth as temperatures in streams increase above 10°C but remain below 18°C
Decreased survival as metabolic demands increase but food supplies become limited
Loss of rearing habitats
Decrease in summer rearing habitats as flow decreases and pool abundance and quality decrease
Deterioration of off-channel habitats as temperatures exceed optimum ranges
Loss of off-channel habitats through more frequent high intensity rainfall events that remove instream structure and beaver dams during fall and winter
Intrusion of salt water into low elevation rearing areas

of a run for a given stock may be harvested depending on the run size and the escapement goals (Clark et al. 2006; Eggers 2007). High harvest rates are a significant anthropogenic disturbance with unknown results in an environment of rapid climate change. Rapid and sustained changes in climate regimes occurring over a period of decades are likely to impose increased stress on stocks that have adapted over centuries to specific environments (Halupka et al. 2003).

A favorable outcome for Alaskan salmon depends upon a few assumptions. The first is that they can acclimate to the rapid (i.e. decades rather than centuries) changes in temperature regimes that are likely to occur with climate change. The ability of

individual stocks to adapt to and withstand the effects of precipitation and sea-level will also influence the outcome. The issue of climate change and ocean conditions and its effects on salmon populations is not directly addressed in this analysis. Recruitment to freshwater ecosystems depends on oceans conditions which may also influence the response in freshwater.

Ocean conditions impose another element of complexity to potential effects of climate change on salmon in freshwater (Beamish 1995). Beamish and Bouillon (1993) identified trends in catch associated with weather patterns for pink and chum salmon in the northern Pacific Ocean using historical catch and weather data. Recent studies have shown that the abundance of salmon production responds to ocean temperature cycles associated with PDO and El Niño (ENSO) events (Beamish et al. 1999; Hare et al. 1999; Hollowed et al. 2001). Temperature increases appear to have contradictory effects on salmon production with increasing survival rates for northern stocks and decreasing rates for southern stocks (Hare et al. 1999; Mueter et al. 2002). Ocean surveys by Welch et al. (1998a) suggest that as thermal boundaries move northward, the ocean rearing area available for sockeye will decrease with increasing ocean temperatures. The response of salmon to climate change in the ocean is complex and is likely to differ by geographic region and is likely to have a bearing on freshwater populations.

The combination of the three outcomes—positive, negative, or neutral—for the freshwater and ocean environments results in nine combinations of outcomes when the responses in each environment are considered. Three may be considered as positive outcomes, a positive outcome in both environments and a positive in one and a neutral in the other. Neutral outcomes in each are possible. It is uncertain if a positive outcome in one environment would "cancel out" a negative outcome in the other environment. The remaining outcomes are negative. All of this assumes equal weighting for each outcome. A qualitative evaluation of the evidence suggests that the probability for negative outcomes is greater than that for positive outcomes with respect to salmon response to global climate change. Furthermore, negative outcomes are more likely where salmon stocks are already stressed or are at the more southern range of their distribution than those in regions where they tend to have robust populations.

Pacific salmon as a group occupy habitats that range from the Beaufort Sea where they occur in relatively small numbers to San Francisco Bay where they are nearly extirpated (Craig and Haldorson 1986; Nehlson 1997; Groot and Margolis 1991). Furthermore, they have been introduced to diverse geographic locations in both the northern and southern hemispheres (Quinn et al. 2001; Hansen and Holey 2002). They readily exploit new habitats opened with fish ladders and major changes in watersheds (Hendry et al. 1998, 2000; Bryant et al. 1999). These characteristics suggest that Pacific salmon in southeast Alaska may be fairly resilient in the face of global temperature increases. If the temperatures in southeast Alaska increase by 2–4°C, then the climate might be similar to that in southern British Columbia or Washington State. Both locations have climates that are generally favorable to Pacific salmon and have or had robust salmon stocks.

A common concern in several assessments of the effect of climate change on aquatic ecosystems is the interaction with existing effects of anthropogenic disturbance (Magnuson et al. 1990; Hauer et al. 1997; Schindler 1997; Mote et al. 2003). Among these effects are chemical and thermal pollution, dams, habitat deterioration,

and introduced species. These effects are not widespread in southeast Alaska and where they do occur they tend to be localized near urban areas that are considerably smaller than the large urban areas of the west coast. Nearly all of the landscapes where Pacific salmon are found along the west coast of the US are exposed to one or more of these stressors. Most of the effects on the salmon stocks of southeast Alaska are likely to be similar for stocks in more southern parts of North America; however, they are likely to be exacerbated by existing stressors described by Hauer et al. (1997). Furthermore, depleted and extirpated stocks combined with introgression of hatchery stocks are likely to have altered the genetic structure of existing stocks. The potential effects described for salmon in southeast Alaska may be more acute for stressed stocks elsewhere; therefore, their ability to respond to rapidly changing environmental conditions imposed by climate change bears considerable attention.

The preponderance of evidence supports a scenario of increasing temperatures globally (Hauer et al. 1997; Mote et al. 2003; Schindler 1997; Smith and Tirpak 1989). The potential effects of climate change and management responses for salmonid populations in Southeast Alaska (as well as elsewhere along the Pacific coast) are to a large degree speculative and based on studies that address many of the issues indirectly (e.g. Levy 1992; Mote et al. 2003; Winder and Schindler 2004). Most management practices for salmonids in southeast Alaska are designed to sustain salmon populations under present climatic conditions. Given that rapidly changing climatic conditions will impose additional stress on populations, a more conservative management strategy may be appropriate (Halupka et al. 2003). In general, these include land management practices that preserve thermal refugia, critical habitats, and intact watersheds. More intense monitoring of harvest rates that focus on specific stocks will allow managers to respond rapidly to changes in productive capacity of watersheds that may be most susceptible to potential changes in climate conditions. Development of effective management guidelines is greatly restricted by a lack of quantitative information with respect to the potential changes and the response salmonid populations in southeast Alaska.

5 A set of questions

The set of questions proposed in Table 2 provide a starting point for research to predict and anticipate how anadromous salmonids may respond to climatic changes. Most require the development of predictive models; however, models require reliable quantitative information. Some is available. For example, the relationship between rates of development, rate of embryos and temperature (Alderdice and Velsen 1978; Tang et al. 1987). Use of GIS databases can provide estimates of the amount of area affected by changing climatic conditions. Bioenergetics models such as the one applied by McDonald et al. (1996) may provide useful estimates of growth rates for specific species, such as juvenile coho salmon.

The outline in Table 2 presents a set of topics that can provide useful information that will contribute to the understanding of how salmon species may respond to changing climatic conditions in southeast Alaska. The format begins with a major question to be addressed (1) followed by a brief statement of a general hypothesis (a) under the question. The last element in the outline (i) is a one sentence description of a proposed line of research. Each in turn may be developed into a detailed study

Table 2 Questions, hypotheses, and study topics

1. How will temperature and flow interact to affect spawning conditions for pink salmon?
   (a) Predicted changes in the global climate will increase frequency and extent of pre-spawner mortality in southeast Alaska streams
      (i) Develop a model based on current and predicted ranges of temperature and discharge to predict potential mortality of pre-spawner pink salmon
   (b) Pink salmon fry will emerge earlier as stream temperatures increase and enter the marine environment during periods of low food availability
      (i) Determine the effects of increasing temperatures on development and emergence of pink salmon in Southeast Alaska streams and relationships to near-shore marine productivity on early life history survival
2. How will increasing stream temperature affect growth of juvenile coho salmon?
   (a) Atmospheric warming will lengthen growing season for juvenile coho salmon and increase temperatures that can increase growth rates
      (i) Construct a bioenergetic model to examine growth rates in southeast Alaska streams and associated habitats to determine growth under varying temperatures and food conditions
3. How will changes in lake trophic conditions affect sockeye salmon rearing in southeast Alaska lakes?
   (a) Increasing temperatures will shift thermodynamics of lakes with earlier vernal mixing and later autumnal mixing with subsequent shifts in food availability (quantity and quality) to sockeye salmon and will alter growth of juvenile sockeye salmon
      (i) Establish relationships between lake thermodynamics, lake trophic conditions, and time of entry of sockeye fry into lakes of southeast Alaska
      (ii) Determine response of juvenile sockeye salmon growth to changes in temperature and food abundance and quality in southeast Alaska lakes using a bioenergetic model
4. How much area will be lost to salmonid production through elevation of sea level throughout southeast Alaska?
   (a) Global climate change will increase temperature that will result in rising sea levels that will shift freshwater spawning and rearing into marine or intertidal habitats
      (i) Determine net loss of freshwater wetlands and habitat under a series of scenarios of sea level rise using GIS models
      (ii) Estimate the potential effects on rearing habitat for juvenile salmonids with loss of freshwater wetlands and habitats
      (iii) Determine shifts in spawning habitats of pink salmon as lower reaches of spawning locations are affected by sea water

proposal. Most statements are directed at the level of a post-doctoral or Ph.D. research assignment. Although it is not listed in the table, there is a need for a Regional Climate Model for southeast Alaska and the north Pacific coast. This is not an easy task given the complex topography, severe weather patterns, and intense interaction with the marine environment.

Most of the existing studies of the response of fish to climate change tend to remain within the realm of biological response; the multidisciplinary paper by Hauer et al. (1997) is an exception. Most of the studies suggested in Table 2 tend to be narrowly focused on specific response by species or group of species. However, understanding the potential responses and underlying causes requires an interdisciplinary approach. The questions address relatively specific aspects of the life history of each species and the response to changes in its physical environment, specifically temperature and precipitation, and in some cases changes in habitat imposed by higher sea level. These interactions require a strong interdisciplinary component as a basis for all of

the suggested studies. Development of a RCM for southeast Alaska and the northern Pacific Northwest coast region would be a prerequisite for realistic modeling of potential responses of salmon and other biota to climate change in the region.

As an example of the interdisciplinary nature of the studies, determination of the effects of rising sea level on salmon populations will require a complex analysis of physical, geographic, and biological responses. As a starting point it will require a GIS assessment of area that may be affected including the development of a fine scale digital elevation model (DEM). Current DEMs that cover most of southeast Alaska are 20 m. Isostatic rebound is an important process in most of coastal Alaska and varies from location to location and must be included in the assessment. Once these processes have been quantified, effects on habitat used by salmon and biological responses can be modeled. Taken as a whole, it poses as difficult question, but with an interdisciplinary approach each element has a solution.

Even under the most draconian preventative measures climate change will occur and global atmospheric temperatures will increase. Many of the corresponding effects cannot be avoided; however, with increased information on the potential effects and responses of economically and socially important resources such a salmon, management strategies may be adopted that will mitigate or reduce adverse effects on them. The studies suggested in this paper are a starting point for understanding the consequences of climate change on anadromous salmon. They are oriented toward salmon populations in southeast Alaska; however, they may be applied to salmon populations in more southern regions of the Pacific coast where current condition and stressors may be far more critical than those in southeast Alaska.

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The Case for Salmon Conservation at the Watershed Scale in Southeast Alaska: An annotated bibliography

Trout Unlimited Alaska

Mason D. Bryant, Ph.D.
Douglas Island Aquatic Ecology
P.O. Box 240152
Douglas, AK 99824
mdbiii45@gmail.com

The numerous and diverse watersheds of southeast Alaska, in contrast to those elsewhere in the Pacific Northwest, support abundant and sustainable runs of five species of Pacific salmon (Baker et al. 1996; Halupka et al. 2000; Lohr and Bryant 1999). This can be attributed to the large number of intact watersheds and relatively low intensity of human impacts in many watersheds throughout the region (Bryant and Everest 1998). Wide distribution of watersheds with habitats that are formed and maintained by natural processes and conservative harvest practices are essential for sustainable salmon stocks (Baker et al. 1996; Meacham and Clark 1994; Schindler et al. 2008). The biological and physical processes that support these stocks are reasonably well documented in many cases throughout the Pacific Northwest and Alaska.

The purpose of this bibliography is to compile this information into a single, searchable source. The scope of the bibliography includes relevant studies, primarily from peer-reviewed literature, throughout the Pacific Northwest, including British Columbia, Washington, Oregon, and Idaho that would support the case for protecting whole watersheds as an integral tool in salmon conservation. Studies from other geographic locations are included if they are relevant at the watershed scale. The objective of the bibliography is to assemble a set of published studies that describe the structure and function of watersheds that are important to spawning and rearing habitats

of salmon in southeast Alaska. These include the five Pacific salmon; Chinook (*Oncorhynchus tshawytscha*), coho (*O. kitsutch*), sockeye (*O. nerka*), chum (*O. keta*), and pink salmon (*O. gorbuscha*); and steelhead (*O. mykiss*). These species also share habitats with cutthroat trout (*O. clarkii*) and Dolly Varden (*Salvelinus malma*). Citations include those that describe and discuss watershed structure and function as well as specific studies of various parts of the watershed, such as headwaters, tributaries, and main stream. The studies include those that typically describe the relationships between habitats and salmon populations and connectivity of elements within the watershed.

The bibliography consists of the complete citation listed by author and date, the publication, volume and pages, a set of keywords, and an abstract or brief summary of the article. Citations listed by author and date are organized into groups by keywords that describe their location within the watershed beginning at the watershed scale. The groups include studies that focus on the watershed scale, main stream, tributaries, headwaters, and estuaries. A group of studies that discuss effects or potential effects of climate change on salmon and their habitat is also included. Many studies may cover multiple parts of the watershed and the groups will include duplicate entries. The complete citation with the abstract and keywords is listed once alphabetically by author.

All of the listed citations are available in a searchable database using Endnote. The Endnote database can be searched using a robust combination of keywords, author, date, and by words or phrases found in the title or abstract. All citations in the database can be edited to include additional keywords or additional information in the abstract or summary. It can be updated to include addition references as they are found or become available.

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Murphy et al. 1986; Murphy et al. 1989; Nass et al. 1996; Newbold et al. 1981; Nickelson et al. 1992; Rice et al. 2008; Rosenfeld et al. 2000; Rosenfeld and Boss 2001; Rosenfeld and Huato 2003; Solazzi et al. 2000; Stednick 2008; Swain and Holtby 1989; Swales et al. 1988; Thedinga et al. 1989; Tripp and Poulin 1986; Triska et al. 1989; Wissmar et al. 1997)

Off Channel

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Climate Change

(Beacham and Murray 1990; Beamish et al. 1997; Beamish et al. 1999; Berggren and Filardo 1993; Bryant 2009; Carpenter et al. 1992; Connor et al. 2003; Decamps 1993; Eaton and Adams 1995; Eaton and Scheller 1996; Finney et al. 2000; Hilborn et al. 2003; Lawson et al. 2004; Mantua 2009 ; McFarlane et al. 2000; Mote et al. 2003; Schindler et al. 2008; Shirvell 1994; Taylor 2008; Williams et al. 2009)

Literature Summaries

Abrahams, A. D. 1984. Channel networks: a geomorphological perspective. Water Resources Research 20(2):161-168.

'KEY WORDS'
Watershed Scale
Channel morhology
Geomorpholgy
'ABSTRACT'

The study of channel networks has been dominated since 1966 by the random model. However, recent work has shown (1) that although the topological properties of small networks conform to the random model more closely than those of large ones, even small networks exhibit systematic deviations from topological randomness and (2) that the topological and length properties of channel networks are controlled to a large degree by the spatial requirements of subbasins and the need for these subbasins to fit together in space, by the size, sinuosity, and migration rate of valley bends, and by the length and steepness of valley sides. The factors that control the density properties of channel networks vary with the scale of the investigation and the geomorphic processes governing channel initiation. Although progress has been made toward a satisfactory stream junction angle model, further work is needed. The evolution of channel networks has been investigated by a variety of methods, including the development of conceptual and simulation models, the monitoring of small-scale badland and experimental drainage basins, and the substitution of space for time. The morphology of most channel networks is largely inherited from the past or strongly influenced by inherited forms. Inasmuch as there is no way of ever knowing the origin or complex history of such networks, the use of stochastic models in their study seems unavoidable.

Allan, J. D., M. S. Wipfli, J. P. Caouette, A. Prussian, and J. Rodgers. 2003. Influence of sreamside vegetation on inputs of terrestrial invertebrates to salmonid food webs. Canadian Journal of Fisheries and Aquatic Sciences 60:309-320.

'KEY WORDS'
Tributaries
Southeast Alaska
Coho salmon
Feeding ecology
Riparian vegetation
'ABSTRACT'

Salmonid food webs receive important energy subsidies via terrestrial in-fall, downstream transport, and spawning migrations. We examined the contribution of terrestrially derived invertebrates (TI) to juvenile coho (Oncorhynchus kisutch) in streams of southeastern Alaska by diet analysis and sampling of TI inputs in 12 streams of contrasting riparian vegetation. Juvenile coho ingested 12.1 mg·fish−1 of invertebrate mass averaged across all sites; no significant differences associated with location (plant or forest type) were detected, possibly because prey are well mixed by wind and water dispersal. Terrestrial and aquatic prey composed approximately equal fractions of prey ingested. Surface inputs were estimated at ~80 mg·m−2·day−1, primarily TI. Direct sampling of invertebrates from the stems of six plant species demonstrated differences in invertebrate taxa occupying different plant species and much lower TI biomass per stem for conifers compared with overstory and understory deciduous plants. Traps placed under red alder (Alnus rubra) and conifer (mix of western hemlock (Tsuga heterophylla) and Sitka spruce (Picea sitchensis)) canopies consistently

captured higher biomass of TI under the former. Management of riparian vegetation is likely to influence the food supply of juvenile coho and the productivity of stream food webs.

Allred, M. 1980. A re-emphasis on the value of the beaver in natural resource conservation. Journal of the Idaho Academy of Science 16(1):3-10.
'KEY WORDS'
Off Channel
Beaver
Conservation
'ABSTRACT'
Beaver ponds provide a range of habitats for fish,wildlife, & vegetation succession.

Amoros, C., and G. Bornette. 2002. Connectivity and biocomplexity in waterbodies of riverine floodplains. Freshwater Biology 47:761-776.
'KEY WORDS'
Watershed scale
Theory
floods
biodiversity
river dynamics
succession
'ABSTRACT'
In river corridors, water plays a key role in connecting various landscape patches. This `hydrological connectivity' operates on the four dimensions of fluvial hydrosystems: longitudinal, lateral, vertical, and temporal. The present review focuses on: (1) lateral connectivity that links the main course of a river with floodplain waterbodies; and (2) vertical connectivity, the exchanges between the surface and groundwater via infiltration into the alluvial aquifer and exfiltration of phreatic water from the hillslope aquifer.The biocomplexity of fluvial hydrosystems results from interactions between processes operating at various spatial and temporal scales. Differences in the nature and intensity of hydrological connectivity contribute to the spatial heterogeneity of riverine floodplains, which results in high alpha, beta and gamma diversity. Differences in connectivity also provide complementary habitats that are required for the parts of life cycles and life-cycles of some species. Hydrological connectivity also produces antagonistic effects, even within the same waterbody. Two temporal scales are distinguished in connectivity dynamics. River level fluctuations within years lead to a pulsing connectivity that drives the functioning of floodplain ecosystems, namely the exchange of organic matter and inorganic nutrients and the shift between production and transport phases. On the scale of decades to centuries, the interactions between various processes increase the biocomplexity of floodplains; for example, river dynamics, which create highly connected waterbodies, compensate for succession that tends towards disconnection. Alternatively, river-bed incision leads to the reduction of fluvial dynamics and to the disconnection of waterbodies, although river incision may increase vertical connectivity where waterbodies are supplied by the hillslope aquifer.

Anderson, N. H., and D. M. Lehmkuhl. 1968. Catastrophic drift of insects in a woodland stream. Ecology 49(2):198-206.
'KEY WORDS'
Tributaries
Headwater streams
Oregon
Aquatic insects
Drift
'ABSTRACT'
The effect of early fall rains on the downstream drift or displacement of insects was studied for two seasons by collecting the entire streamflow at one point through a drift net. Drift rate

increased within 24 hr after the start of each rainy period, with the increase approximately proportional to the increase in stream flow. Freshets due to less than 1 in. (2.5 cm) of rain caused a fourfold increase in numbers and fivefold to eightfold increase in biomass. Major components of the drift were Ephemeroptera, Plecoptera, Diptera and terrestrial insects. Plecoptera and Ephemeroptera retained the day—night periodicity of behavioral drift during freshets, but drift of Chironomidae (Diptera) was attributed to catastrophic and constant drift. Mean weight per individual of several taxa was greater at night than day, in freshet than nonfreshet periods, and in drift compared with benthos samples. Though catastrophic drift due to fall freshets displaced large numbers of individuals, the standing crop of the benthos increased during the fall because of hatching. The drift may be beneficial in dispersing aggregations of young larvae. Removal of allochthonous food by increased water flow could be more detrimental to benthos populations than the direct mortality caused by catastrophic drift.

Andrus, C. W., B. A. Long, and H. A. Froehlich. 1988. Woody debris and its contribution to pool formation in a coastal stream 50 years after logging. Canadian Journal of Fisheries and Aquatic Sciences 45:2080--2086.

'KEY WORDS'
Tributaries
Main Stream
Oregon
Large wood
Stream habitat
Effects of logging
Stream succession
'ABSTRACT'

Large quantities of woody debris persisted 50 yr after logging and fire in stream channels of a small coastal Oregon watershed. Debris from the current stand represented only 14% of total debris volume and 8% of debris volume responsible for creating pools. The greatest number of pools were located in downstream sections of the watershed where gradient was reduced, discharge was increased, and streambed material was finer. Seventy percent of pools with a volume greater than 1.0 m3 were associated with woody debris in the channel. Composition of the current riparian forest varied with topography. Alder stands dominated moist terrace sites adjacent to channels, whereas slopes contained a mixture of alder and conifer. Study results indicate that riparian trees must be left to grow longer than 50 yr to ensure that an adequate, long-term supply of woody debris is available to stream channels. Debris from previous stands plays a crucial role in the interim and should not be removed from stream channels.

Baker, T. T., and coauthors. 1996. Status of Pacific salmon and steelhead escapements in Southeast Alaska. Fisheries 21(10):6-18.

'KEY WORDS'
Anadromous salmonids
steelhead
Escapement
Stock status
'ABSTRACT'

We evaluated the status of Pacific salmon and steelhead (*Oncorhynchus mykiss*) in southeastern Alaska. Of 9,296 spawning aggregates identified in this region, some data were available for 4,009 (43%), and 928 (10%) had sufficient information to analyze for escapement trend. Of those analyzed, 333 (36%) were increasing, 556 (60%) were stable, 37 (4%) were declining, and 2 (< 1%) showed precipitous declines. We evaluated risk of extinction of spawning aggregates using criteria similar to surveys outside Alaska. We rated 918 (99%) at no or low risk, 8 (≈ 1%) at moderate risk, and 2 (< 1%) at high risk. No spawning aggregates were identified as extinct based on our evaluation of escapement data dating back to 1960. Prior to

1960, two spawning aggregates, one sockeye salmon (*0. nerka*) and one chum salmon (*0. keta*), were identified as extinct based on responses to a postal questionnaire. The Alaska Department of Fish and Game grouped spawning aggregates into management units for each species. Management units vary in number and size for each species. Of 141 management units defined, 129 (92%) had enough information to evaluate; all had stable or increasing escapement trends. However, escapement data for this study had limitations for two reasons. First, monitoring spawning aggregates for escapement in all spawning locations in southeastern Alaska is impractical; interpretations are based instead on estimates of key indicator streams (usually commercially important runs) in the region. Second, most escapement estimates are based on aerial surveys rather than more accurate methods such as weirs or sonar estimates. Our results, based on the limited data, indicated that Pacific salmon in the region were generally in good health at two levels of salmon population structure: spawning aggregates and management units. The small amount of escapement data for steelhead precludes a generalization about their status and overall health.

Baron, J. S., and coauthors. 2002. Meeting ecological and societal needs for freshwater. Ecological Applications 12(5):1247-1260.

'KEY WORDS'
Watershed scale
Ecological integrity
Ecosystem protection
Ecosystem services
Freshwater ecosystems
Restoration, rivers
Waterflow
Water management policy

'ABSTRACT'

Human society has used freshwater from rivers, lakes, groundwater, and wetlands for many different urban, agricultural, and industrial activities, but in doing so has overlooked its value in supporting ecosystems. Freshwater is vital to human life and societal well-being, and thus its utilization for consumption, irrigation, and transport has long taken precedence over other commodities and services provided by freshwater ecosystems. However, there is growing recognition that functionally intact and biologically complex aquatic ecosystems provide many economically valuable services and long-term benefits to society. The short-term benefits include ecosystem goods and services, such as food supply, flood control, purification of human and industrial wastes, and habitat for plant and animal life—and these are costly, if not impossible, to replace. Long-term benefits include the sustained provision of those goods and services, as well as the adaptive capacity of aquatic ecosystems to respond to future environmental alterations, such as climate change. Thus, maintenance of the processes and properties that support freshwater ecosystem integrity should be included in debates over sustainable water resource allocation.

The purpose of this report is to explain how the integrity of freshwater ecosystems depends upon adequate quantity, quality, timing, and temporal variability of water flow. Defining these requirements in a comprehensive but general manner provides a better foundation for their inclusion in current and future debates about allocation of water resources. In this way the needs of freshwater ecosystems can be legitimately recognized and addressed. We also recommend ways in which freshwater ecosystems can be protected, maintained, and restored.

Freshwater ecosystem structure and function are tightly linked to the watershed or catchment of which they are a part. Because riverine networks, lakes, wetlands, and their connecting groundwaters, are literally the "sinks" into which landscapes drain, they are greatly influenced by terrestrial processes, including many human uses or modifications of land and water. Freshwater ecosystems, whether lakes, wetlands, or rivers, have specific requirements in terms of quantity, quality, and seasonality of their water supplies. Sustainability normally requires these systems to fluctuate within a natural range of

variation. Flow regime, sediment and organic matter inputs, thermal and light characteristics, chemical and nutrient characteristics, and biotic assemblages are fundamental defining attributes of freshwater ecosystems. These attributes impart relatively unique characteristics of productivity and biodiversity to each ecosystem. The natural range of variation in each of these attributes is critical to maintaining the integrity and dynamic potential of aquatic ecosystems; therefore, management should allow for dynamic change. Piecemeal approaches cannot solve the problems confronting freshwater ecosystems.

Scientific definitions of the requirements to protect and maintain aquatic ecosystems are necessary but insufficient for establishing the appropriate distribution between societal and ecosystem water needs. For scientific knowledge to be implemented science must be connected to a political agenda for sustainable development. We offer these recommendations as a beginning to redress how water is viewed and managed in the United States: (1) Frame national and regional water management policies to explicitly incorporate freshwater ecosystem needs, particularly those related to naturally variable flow regimes and to the linking of water quality with water quantity; (2) Define water resources to include watersheds, so that freshwaters are viewed within a landscape, or systems context; (3) Increase communication and education across disciplines, especially among engineers, hydrologists, economists, and ecologists to facilitate an integrated view of freshwater resources; (4) Increase restoration efforts, using well-grounded ecological principles as guidelines; (5) Maintain and protect the remaining freshwater ecosystems that have high integrity; and (6) Recognize the dependence of human society on naturally functioning ecosystems.

Beacham, T. D., and C. B. Murray. 1990. Temperature, egg size, and development of embryos and alevins of five species of Pacific salmon: a compararive analysis. Transactions of the American Fisheries Society 119:927-945.

'KEY WORDS'
Climate change
Temperture effects
Salmon
Climate change
Embryo development

'ABSTRACT'

We examined rate of development to alevin hatching and fry emergence, embryo and alevin survival, and alevin and fry size for five Pacific salmon species. There was little difference among values for hatching and emergence time predicted by a modified thermal sums model, power law model (log-inverse Belehrádek), or quadratic model. Coho salmon Oncorhynchus kisutch had the fastest rates of development to hatching and emergence of the five species investigated; rankings for the other species depended upon temperature range. Coho salmon embryos had the highest survival rates at low incubation (1.5°C) temperatures. Embryos of pink salmon O. gorbuscha had the lowest survival at temperatures less than 4°C. For all five species, incubation temperature was the more important factor in determining alevin length, and egg size was the more important factor in determining alevin weight. Egg weight was a major determinant of fry weight at emergence. Rates of development to hatching and emergence, and alevin and fry size, differed by species in response to changes in temperature, Coho salmon alevins and fry were proportionately larger at 4°C than at 8°C or 12°C, but alevins and fry of pink salmon and chum salmon O. keta were largest at 8°C. Variation in development characters of Pacific salmon reflected adaptations to each species' life history pattern.

Beamish, R. J., C. M. Neville, and A. J. Cass. 1997. Production of Fraser River sockeye salmon (Oncorhynchus nerka) in relation to decadal-scale changes in the climate and the ocean. Canadian Journal of Fisheries and Aquatic Sciences 54:543-554.

'KEY WORDS'

Climate change
Watershed scale
Decadal climate shift
Marine survival
Sockeye salmon
'ABSTRACT'

The abundance of Fraser River sockeye salmon (Oncorhynchus nerka) stocks was low in the 1960s, increased to high levels in the 1980s, and possibly entered a period of low abundance in recent years. The abundance changes of the combined stocks can be separated into productivity regimes that correspond to changes in climate trends. The most distinct change occurred when there was a major change in the climate over the Pacific Ocean in the winter of 1976-1977. The existence of natural shifts in abundance trends means that the high returns that occur during periods of high productivity would not be expected to occur during the low-productivity periods. The response of Fraser River sockeye to climate changes may be a specific example of a more general response by a number of species of fishes in the Pacific and perhaps in other oceans. Because the shift from one regime to the other occurred quickly in the 1970s, future shifts could also occur quickly. It is necessary to detect natural shifts in productivity when attempting to manage fishing impacts to ensure that economic expectations are sound and that overfishing does not occur.

Beamish, R. J., and coauthors. 1999. The regime concept and natural trends in the production of Pacific salmon. Canadian Journal of Fisheries and Aquatic Sciences 56:516-526.
'KEY WORDS'
Climate change
Temperature regimes
Production
'ABSTRACT'

Large fluctuations in the trends of Pacific salmon production in this century have been linked to trends in climate in the Pacific that are in turn associated with climate trends throughout the Northern Hemisphere. The close correspondence in the persistence of climate trends and the synchrony of the changes is evidence that a common eventmay cause the regime shifts. The trends or regimes can be characterized by stable means in physical data series or multiyear periods of linked recruitment patterns in fish populations. The regime concept is important in fisheries management because the natural shifts in abundance may be large and sudden, requiring that these natural impacts be distinguished from fishing effects. An equally important consideration is that biological and physical mechanisms may change when regimes shift, resulting in conditions that may not be characterized in the earlier part of the data series. Fluctuations in Pacific salmon abundance in this century were synchronous with large fluctuations in Japanese sardine abundance, which can be traced back to the early 1600's. The synchrony in the fluctuations suggests that Pacific salmon abundance may have fluctuated for centuries in response to trends in climate. The concept of regimes and regime shifts stresses the need to improve our understanding of the mechanisms that regulate the dynamics of fish and their ecosystems

Beechie, T., E. Beamer, and L. Wasserman. 1994. Estimating coho salmon rearing habitat and smolt production losses in a large river basin, and implications for habitat restoration. North American Journal of Fisheries management 14:797-811.
'KEY WORDS'
Main Stream
Restoration
Coho salmon
Large rivers
'ABSTRACT'

To develop a habitat restoration strategy for the 8,270-km2 Skagit River basin, we estimated changes in smolt production of coho salmon Oncorhynchus kisutch since European settlement began

in the basin, based on changes in summer and winter rearing habitat areas. We assessed changes in coho salmon smolt production by habitat type and by cause of habitat alteration. We estimated that the coho salmon smolt production capacity of summer habitats in the Skagit River basin has been reduced from 1.28 million smolts to 0.98 million smolts (–24%) and that the production capacity of winter habitats has been reduced from 1.77 million to 1.17 million smolts (–34%). The largest proportion of summer non-main-stem habitat losses has occurred in side-channel sloughs (41%), followed by losses in small tributaries (31%) and distributary sloughs (29%). The largest loss of winter habitats has occurred in side-channel sloughs (52%), followed by losses in distributary sloughs (37%) and small tributaries (11%). By type of impact, hydromodification (diking, ditching, dredging) associated with agricultural and urban lands accounts for 73% of summer habitat losses and 91% of winter habitat losses. Blocking culverts on small tributaries account for 13% of the decrease in summer habitat and 6% of the decrease in winter habitat. Forestry activities account for 9% of summer habitat losses and 3% of winter habitat losses. Limitations of the analysis and implications for developing a habitat restoration strategy are discussed.

Beechie, T. J., and T. H. Sibley. 1997. Relationships between channel characteristics, woody debris, and fish habitat in Northwestern Washington streams. Transactions of the American Fisheries Society 126:217-229.

'KEY WORDS'
Tributaries
Main stream
Washington
Forest management
Large woody debris
Salmon habitat
Channel morphology

'ABSTRACT'

Relationships between large woody debris (LWD) and pool area or pool spacing varied with channel slope and channel width for streams in second-growth forests in northwest Washington. Pool spacing (expressed as the number of channel widths between pools) decreased as number of woody debris increased in both moderate-slope (0.02 < slope < 0.05) and low-slope (0.001 < slope ≤ 0.02) channels, but the relationship was stronger in moderate-slope channels. Percent pool was also more strongly correlated with woody debris volume in moderate-slope channels than in low-slope channels. Multiple-regression analyses showed that pool spacing and percent pool were correlated with an interaction term between LWD abundance and channel slope, suggesting that the influence of LWD on pool formation changes with channel slope. Analysis of pool-forming mechanisms indicated that low-slope channels are less sensitive to LWD abundance because pools are formed by mechanisms other than LWD when LWD abundance is low. Size of LWD that formed pools increased with increasing channel width, but was not related to channel slope. Percent gravel (proportion of the bed in patches of gravel 16–64 mm in diameter) was best explained by channel slope and channel width, and there was no significant relationship between woody debris and percent gravel. A regression between median particle size of sediment on the stream bed and basal shear stress showed that the relationships among percent gravel, channel width, and channel slope are adequately explained by the channel's capacity to transport particles of various sizes.

Beedle, D. L. 1991. Physical dimensions and hydrological effects of beaver ponds on Kuiu Island in southeast Alaska. MS Thesis. Oregon State University.

'KEY WORDS'
Off Channel
Southeast Alaska
Beaver ponds
Hydrology
Stream morphology

'ABSTRACT'
Describes characteristics of beaver dams and their reponse to season flows.

Benda, L., K. Andras, D. Miller, and P. Bigelow. 2004. Confluence effects in rivers: Interactions of basin scale, network geometry, and disturbance regimes. Water Resources Research 40:1-15.
'KEY WORDS'
Watershed scale
River ecology
Landscape ecology
Tributaries
Channel morphology
Disturbance
'ABSTRACT'
We reviewed 14 studies documenting the effects of tributaries on river morphology at 167 confluences along 730 km of river spanning seven orders of magnitude in drainage area in western United States and Canada. In both humid and semiarid environments the probability of observing significant confluence-related changes in channel and valley morphology due to tributary influxes of sediment (e.g., changes in gradient, particle size, and terraces, etc.) increased with the size of the tributary relative to the main stem. Effects of confluences on river morphology are conditioned by basin shape and channel network patterns, and they include the nonlinear separation of geomorphically significant confluences in river networks. Other modifying factors include local network geometry and drainage density. Confluence-related landforms (i.e., fans, bars, terraces, etc.) are predicted to be dominated by older features in headwaters and younger features downstream, a pattern driven by the frequency and magnitude of floods and punctuated sediment supply that scale with watershed size.

Benda, L., T. J. Beechie, A. Johnson, and R. C. Wissmar. 1992. The geomorphic structure of salmonid habitats in a recently deglaciated river basin, Washington state. Canadian Journal of Fisheries and Aquatic Sciences 49(6):1246-1256.
'KEY WORDS'
Watershed scale
Main Stream
Tributaries
Washington
Salmon habitat
Geomorphology
Landslides
Headwater streams
'ABSTRACT'
Morphology and distribution of salmonid habitats were related to the geomorphology of a river basin at three spatial scales including reach (102–103 m2), subbasin (2–26 km2), and the watershed (240 km2). Stream reaches on a young fluvial terrace (1700 yr old) adjacent to the main river contain the most extensive areas of rearing and spawning habitats. In tributary subbasins, the area of spawning habitat varies according to discharge rates and channel gradients. The most extensive salmonid habitats are located along wide glacial deposits in geologically unconstrained areas of the main valley floor. During the early Holocene (~10 000 – 12 000 years before present (B.P.)), the recently deglaciated watershed of the South Fork Stillaguamish River was extremely erosive and vegetated by alpine forest. Fish habitats then were less suitable for salmonid rearing and spawning. A much lower erosion rate after 8000 yr B.P., and the advent of old growth conifer forests after 6000 yr B.P., indicates that stream habitats attained their present-day morphology between 8000 and 6000 yr ago. Although habitats increased in quality with increasing watershed stability and evolution of forests, they decreased in quantity after 7000 yr B.P as landforms changed because of continuous river incision into glacial deposits.

Berggren, T., and M. Filardo. 1993. An analysis of variables influencing the migration of juvenile salmonids in the Columbia River basin. North American Journal of Fisheries Management 13:48-63.

'KEY WORDS'
Climate change
Anadromous migrations
Flow measurement
Modelling
Smolts
Regression analysis
Migration
Oncorhynchus tshawytscha
Oncorhynchus mykiss
Usa, columbia r.
Usa, columbia r. Basin

'ABSTRACT'

The amount of time that it takes juvenile chinook salmon Oncorhynchus tshawytscha and steelhead O. mykiss to migrate (travel time) at different river flows through index reaches in the Snake and Columbia rivers was analyzed with bivariate- and multiple-regression models. Smolt travel time estimates for yearling chinook salmon and steelhead in the Snake River, steelhead in the middle Columbia River, and subyearling chinook salmon in the lower Columbia River were inversely related to average river flows. In the multiple-regression analyses, additional predictor variables that were related either to flow or to smoltification were used. These predictor variables were calculated over the same time period as the travel time estimates. Flow-related variables were referenced at a key hydroelectric site within each index reach, and included average river flow, minimum river flow, and absolute change in river flow. The smoltification-related variables provided indirect indices of smoltification. They included water temperature, date of entry into an index reach, chinook salmon race, and travel time prior to entry into an index reach. The final models included those predictor variables explaining significant variation in smolt travel time. The variables in the final multiple-regression models explained 74% and 39% of the variation in the travel time for yearling chinook salmon within the Snake and middle Columbia river index reaches, respectively; 90% and 62% for steelhead within the Snake and middle Columbia reaches; and 65% for subyearling chinook salmon in the lower Columbia reach. Average river flow made the largest contribution to explaining variation in smolt travel time in the majority of the multiple-regression models. Additional variation in smolt travel time could be explained by including other flow- and smoltification-related variables in the models.

Bilby, R. E., and L. A. Mollot. 2008. Effect of changing land use patterns on the distribution of coho salmon (Oncorhynchus kisutch) in the Puget Sound region. Canadian Journal of Fisheries and Aquatic Sciences 65:2138-2148.

'KEY WORDS'
Watershed scale
Pacific Northwest
Puget Sound
Salmon
Land use

'ABSTRACT'

Population increase in the Pacific Northwest of North America over the last century has led to the removal of forests for various purposes. Evidence of salmon response to these alterations in land use is rare owing to a scarcity of fish population data and a high degree of interannual variation in abundance. We examined the relationship between the spatial distribution of spawning coho salmon (Oncorhynchus kisutch) and changes in land use from 1984 through 2001 at 84 sites in four rivers draining into northern Puget Sound. Changes in land use over this period were determined from LandSat imagery, county zoning designations, and aerial

photographs. Substantial reduction in forest cover occurred in many of the index watersheds during this time. The proportion of salmon using sites subjected to increased urban land use over the study period declined about 75%. Increases were observed at forested sites and those with increased rural residential use. Maintaining salmon populations in rapidly developing areas may require the identification and protection of sites that support large salmon populations and steering development to areas supporting few fish.

Bilby, R. E., and J. W. Ward. 1989. Changes in characteristics and function of woody debris with increasing size of streams in western Washington. Transactions of the American Fisheries Society 118:368-378.

'KEY WORDS'
Main stream
Tributaries
Watershed scale
Washington
Large wood
Channel morpholgy

'ABSTRACT'

In second- to fifth-order streams that drain old-growth timber in western Washington, characteristics and function of woody debris changed in relation to stream size. Average diameter, length, and volume of pieces of wood increased as stream size increased, whereas the frequency of occurrence of woody debris decreased. In streams with channel widths less than 7 m, 40% of the pieces of debris were oriented perpendicularly to the axis of flow; in streams with channel widths over 7 m, more than 40% of the pieces were oriented downstream. The types of pools most commonly associated with pieces of wood changed from plunge pools in small streams (42%) to debris scour pools in larger systems (62%). Pool area was correlated with the volume of the piece of wood forming the pool in streams of all sizes. However, this relationship was most evident in larger channels. Nearly 40% of the pieces of wood in channels less than 7 m wide were associated with sediment accumulations. Less than 30% of the pieces retained sediment in channels from 7 to 10 m wide, and less than 20% retained sediment in channels greater than 10 m wide. Surface area of sediment accumulations and the volume of the piece of wood forming the accumulation were related in all streams, but the relationship was clearest in the larger channels. Accumulations of particulate organic matter associated with woody debris were more frequent in small streams but were larger in large streams. No relationship was observed between the volume of fine particulate organic matter accumulated by a piece of wood and the piece of wood's volume.

Bilby, R. E., and J. W. Ward. 1991. Characteristics and function of large woody debris in streams draining old-growth, clear-cut, and second-growth forests in southwestern Washington. Canadian Journal of Fisheries and Aquatic Sciences 48:2499-2508.

'KEY WORDS'
Watershed scale
Tributaries
Main Stream
Effects of logging
Woody debris
Washington

'ABSTRACT'

Amount of large woody debris (LWD) surveyed in 70 stream reaches flowing through old-growth, clear-cut, and second-growth forests decreased with increasing stream size for all stand types but was greatest at old-growth sites. Average piece volume was larger at old-growth sites than at other stand types in streams >10 m wide, but no differences were seen in smaller streams. Scour pools accounted for 90% of the wood-associated pools at second-growth and clear-cut sites but only 50% at old-growth sites, which contained more pools than other stand types, particularly for larger streams. Pool size was similar for all stand types in smaller streams,

but averaged 10 m2 in streams >10 m wide at old-growth sites and 4 m2 for other stand types. Pool size was similar for all stand types in smaller streams. Sediment and fine organic matter retained by woody debris decreased with increasing stream size for ail stand types, but old-growth sites contained greater amounts of both materials than other stand types. The frequency of pool formation, the type of pool formed, and sediment accumulation were influenced by the amount of fine debris associated with LWD. Changes in LWD amount, characteristics, and function occurred very rapidly following removal of streamside vegetation.

Bisson, P. A., J. B. Dunham, and G. H. Reeves. 2009. Freshwater Ecosystems and Resilience of Pacific Salmon: Habitat Management Based on Natural Variability. Ecology and Society 14(1):np.

'KEY WORDS'
Watershed Scale
Oregon
restoration
salmon
'ABSTRACT'

In spite of numerous habitat restoration programs in fresh waters with an aggregate annual funding of millions of dollars, many populations of Pacific salmon remain significantly imperiled. Habitat restoration strategies that address limited environmental attributes and partial salmon life-history requirements or approaches that attempt to force aquatic habitat to conform to idealized but ecologically unsustainable conditions may partly explain this lack of response. Natural watershed processes generate highly variable environmental conditions and population responses, i.e., multiple life histories, that are often not considered in restoration. Examples from several locations underscore the importance of natural variability to the resilience of Pacific salmon. The implication is that habitat restoration efforts will be more likely to foster salmon resilience if they consider processes that generate and maintain natural variability in fresh water. We identify three specific criteria for management based on natural variability: the capacity of aquatic habitat to recover from disturbance, a range of habitats distributed across stream networks through time sufficient to fulfill the requirements of diverse salmon life histories, and ecological connectivity. In light of these considerations, we discuss current threats to habitat resilience and describe how regulatory and restoration approaches can be modified to better incorporate natural variability.

Bjornn, T. C., S. C. Kirking, and W. R. Meehan. 1991. Relation of cover alterations to the summer standing crop of young salmonids in small Southeast Alaska Streams. Transactions of the American Fisheries Society 120:562-570.

'KEY WORDS'
Tributaries
Southeast Alaska
Coho Salmon
Steelhead
'ABSTRACT'

Summer abundance of young coho salmon Oncorhynchus kisutch, steelhead O. mykiss, and Dolly Varden Salvelinus malma was assessed in small streams on Prince of Wales Island, Alaska, in an attempt to measure the response of these fish to various types of cover alterations. The standing crop of subyearlings decreased during summer, but none of the decrease could be attributed to the changes in cover we made. Subyearling coho salmon (about 75% of the fish present) did not respond either to the removal of natural riparian vegetation or to the addition of simulated riparian canopy, large boulders, woody debris, or simulated undercut banks. Localized movements within the streams were sufficient to provide relatively rapid recolonization of the experimental habitat units. The forms of cover we evaluated were relatively unimportant in regulating abundance of young coho salmon in small streams.

Bonneau, J., and D. Scarnecchia. 1998. Seasonal and diel changes in habitat use by juvenile bull trout (Salvelinus confluentus) and cutthroat trout (Oncorhynchus clarki) in a mountain stream. Canadian Journal of Zoology 76:783-790.

'KEY WORDS'
Main stream
Idaho
Bull Trout
Cutthroat trout
Habitat selection
Movement
Water depth
Ecological distribution
Environmental factors
Stream flow
Substrate preferences
Diurnal variation
Salvelinus confluentus
Oncorhynchus clarki

'ABSTRACT'
Habitat use by juvenile bull trout (Salvelinus confluentus) and cutthroat trout (Oncorhynchus clarki) in Trestle Creek, Idaho, changed seasonally and dielly. Both cutthroat and bull trout selected pools over riffles in both summer and winter. Both species used a wide range of depths at night but were absent from shallow water (<15 cm) during the day in summer and winter. During summer, juveniles of both species occupied areas of lower velocity water at night than during the day. Both species also occupied lower velocity water during winter days than summer days. During winter days, juvenile bull trout were located below or directly on cobble substrate, whereas cutthroat trout often formed aggregations suspended in the water column of large pools. Both species were more closely associated with cover during the day, and made the greatest use of cover during winter days. Land management activities resulting in decreased pool habitat, instream cover, and stream-bed stability may be especially detrimental to bull trout and cutthroat trout in winter.

Bramblett, R. G., M. D. Bryant, B. E. Wright, and R. G. White. 2002. Seasonal use of small tributary and main-stem habitats by juvenile steelhead, coho salmon, and Dolly Varden in a Southeastern Alaska drainage basin. Transactions of the American Fisheries Society 131:498-506.

'KEY WORDS'
Tributaries
Main Stream
Movement
Steelhead
Coho salmon
Dolly Varden
Southeast Alaska

'ABSTRACT'
The movement of juvenile salmonids between small tributaries and main-stem habitats in southeast Alaska watersheds is poorly understood. We observed movements of steelhead Oncorhynchus mykiss, coho salmon O. kisutch, and Dolly Varden Salvelinus malma between main-stem and tributary habitats at weirs located on tributaries in the Staney Creek watershed in southeast Alaska. We used seasonal relative abundance (catch per unit effort) in eight main-stem reaches and eight tributaries to corroborate observed movement in the two streams with weirs. We observed juvenile steelhead and coho salmon moving through the weirs into tributaries during the fall as flows increased and temperatures decreased. The relative abundance of steelhead was greater in main-stem sites than in tributaries during the summer, whereas during spring and fall relative abundance in the tributaries was similar to that in the main stem. Juvenile coho salmon were abundant in tributaries during all seasons. The relative

abundance of Dolly Varden was greater in the tributaries than in the main-stem during all seasons. These results underscore the significance of links between main-stem habitats and small tributaries for production of juvenile salmonids.

Bryant, M. D. 1980. Evolution of large, organic debris after timber harvest: Maybeso Creek, 1949 to 1978. U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, General Technical Report PNW-101, Portland, OR.
'KEY WORDS'
Main Stream
Southeast Alaska
Large wood
'ABSTRACT'
Hand drawn scale maps of Maybeso Creek were used to describe the dynamics of large wood shortly before, during and after logging in Maybeso Creek from 1952 to 1960. Large accumulations were present during logging, declined in the few years after logging but many large accumulations remained.

Bryant, M. D. 1983. The role and management of woody debris in west coast salmonid nursery streams. North American Journal of Fisheries Management 3:322-330.
'KEY WORDS'
Watershed scale
southeast Alaska
large wood
'ABSTRACT'
Debris removal is a frequently used management technique for small streams in logged watersheds, but many stream-cleaning techniques overlook important habitat requirements of juvenile salmonids. Reviews of some past management practices show little systematic evaluation or monitoring of physical or biological effects. A review of several studies (most of them not associated with debris removal) shows the importance of woody debris as salmonid habitat. The role of organic debris in small stream systems is discussed and a set of criteria for debris removal is proposed.

Bryant, M. D. 1984a. Distribution of salmonids in the Trap Bay Basin, Tenakee Inlet. Pages 17-31 *in* W. R. Meehan, T. R. Merrell, and T. A. Hanley, editors. Fish and Wildlife Relationships in Old-Growth Forests: Proceedings of a symposium. American Institute of Fishery Research Biologists, Juneau, AK.
'KEY WORDS'
Watershed scale
Main Stream
Tributaries
Southeast Alaska
Coho Salmon
Dolly Varden
Cuthroat trout
'ABSTRACT'
Describes distribution of anadromous salmonids in the Trap Bay watershed. Coho salmon were found throughout the watershed. Density declined as gradient increased. Dolly Varden and cutthrout trout were common in the headwaters and tributaries.

Bryant, M. D. 1984b. The role of beaver ponds as coho salmon habitat in southeast Alaska streams. Pages 183-192 *in* J. M. Walton, and D. B. Houston, editors. Proceedings of the Olympic Wild Fish Conference, Port Angles, WA.
'KEY WORDS'
Off channel Habitat
Southeast Alaska

Beaver ponds
'ABSTRACT'
Beaver ponds are distributed througout southeast Alaska. Previously dams were thought to be barriers to juvenile coho salmon. Results from this study demonstrate that beaver ponds are widely used by coho salmon in southeast Alaska.

Bryant, M. D. 2009. Global climate change and potential effects on salmonids in freshwater ecosystems of southeast Alaska. Climatic Change 95:169-193.
'KEY WORDS'
Climate change
Southeast Alaska
Beaver ponds
Stream habitat
Life history
Salmonids
'ABSTRACT'
General circulation models predict increases in air temperatures from 1°C to 5°C as atmospheric CO2 continues to rise during the next 100 years. Thermal regimes in freshwater ecosystems will change as air temperatures increase regionally. As air temperatures increase, the distribution and intensity of precipitation will change which will in turn alter freshwater hydrology. Low elevation floodplains and wetlands will flood as continental ice sheets melt, increasing sea-levels. Although anadromous salmonids exist over a wide range of climatic conditions along the Pacific coast, individual stocks have adapted life history strategies—time of emergence, run timing, and residence time in freshwater—that are often unique to regions and watersheds. The response of anadromous salmonids will differ among species depending on their life cycle in freshwater. For pink and chum salmon that migrate to the ocean shortly after they emerge from the gravel, higher temperatures during spawning and incubation may result in earlier entry into the ocean when food resources are low. Shifts in thermal regimes in lakes will change trophic conditions that will affect juvenile sockeye salmon growth and survival. Decreased summer stream flows and higher water temperatures will affect growth and survival of juvenile coho salmon. Rising sea-levels will inundate low elevation spawning areas for pink salmon and floodplain rearing habitats for juvenile coho salmon. Rapid changes in climatic conditions may not extirpate anadromous salmonids in the region, but they will impose greater stress on many stocks that are adapted to present climatic conditions. Survival of sustainable populations will depend on the existing genetic diversity within and among stocks, conservative harvest management, and habitat conservation.

Bryant, M. D., and F. H. Everest. 1998. Management and condition of watersheds in southeast Alaska: The persistence of anadromous salmonids. Northwest Science 72:249-267.
'KEY WORDS'
Watershed Scale
Southeast Alaska
Effects of logging
Forest management
Riparian management
'ABSTRACT'
In contrast to most of North America and Europe, numerous intact or highly disturbed watersheds are present throughout southeast Alaska. These watersheds support abundant and diverse populations of anadromous salmonids. While the watersheds throughout the northern hemisphere have been exposed to human disturbance from millennia to centuries, significant human disturbance to the watersheds of southeast Alaska did not begin until the 1950's with the start of industrial logging. Although management of watersheds has evolved to reduce risks to aquatic habitat, the most intensive logging occurred during the first 20 years of timber harvest when few restraints were placed on timber harvest in watersheds. As a result,

a legacy of streams with deteriorating habitat remains. While few salmon stocks in southeast Alaska appear to be in decline, escapement records on specific watersheds, particularly those most severely affected by management are non-existent or qualitative. The present status of salmon stocks may be attributed to abundant intact watersheds, high marine survival, and escapement levels that fully seed most watersheds. The numerous intact watersheds throughout southeast Alaska are a critical factor in maintaining sustainable salmon stocks in southeast Alaska

Bryant, M. D., B. J. Frenette, and S. J. McCurdy. 1999. Colonization of a watershed by anadromous salmonids following the installation of a fish ladder in Margaret Creek, southeast Alaska. North American Journal of Fisheries Management 19:1129-1136.

'KEY WORDS'
Watershed scale
Southeast Alaska
Coho salmon
Migration

'ABSTRACT'

We evaluated the colonization of a watershed blocked by a 7-m falls following the installation of an Alaska steeppass fish ladder to provide access for anadromous salmonids. Coho salmon Oncorhynchus kisutch, pink salmon O. gorbuscha, and chum salmon O. keta were present below the falls. Fry of sockeye salmon O. nerka were stocked into Margaret Lake once in 1988 and annually from 1990 through 1994. Pink salmon were the most numerous species to colonize habitat above the falls. Coho salmon moved up the ladder during all years. However, progeny from 25,000 coho salmon presmolts that were stocked in 1991 were the greatest proportion of the returns in 1992, 1995, and 1996. During the study, only 1,595 sockeye salmon returned from more than 1.4 million that were stocked. The rapidity of colonization by naturally occurring anadromous salmonids, including cutthroat trout O. clarki, Dolly Varden Salvelinus malma, and steelhead O. mykiss, underscores a life history strategy of exploiting newly accessible habitat as it becomes available. Although anadromous salmonids successfully colonized the watershed, effects of interbreeding among stocked and natural runs of coho salmon may not be observed for several generations.

Bryant, M. D., D. N. Swanston, R. C. Wissmar, and B. E. Wright. 1998. Coho salmon populations in the karst landscape of north Prince of Wales Island, southeast Alaska. Transactions of the American Fisheries Society 127:425-433.

'KEY WORDS'
Tributaries
Karst
Southeast Alaska
Coho Salmon

'ABSTRACT'

Karst topography is a unique and distinct landscape and its geology may have important implications for salmon productivity in streams. The relationship between salmonid communities and water chemistry and the influence of habitat was examined in a set of streams on north Prince of Wales Island, southeast Alaska. Streams in karst landscapes showed higher alkalinities (1,500–2,300 μeq/L) than streams not influenced by karst landscapes (750–770 μeq/L). A significant, positive relationship was observed between alkalinity and density of coho salmon parr Oncorhynchus kitsutch. Backwater pools supported higher densities of coho salmon than did other habitat units. Both coho salmon fry and parr tended to be larger in most karst-influenced streams than in nonkarst streams. Although past timber harvest practices in the riparian areas of several of the streams appeared to influence stream habitat and water temperature, streams flowing through karst landscapes had a distinct water chemistry. Furthermore, these streams appeared to support more fish than nonkarst streams.

Bryant, M. D., and R. D. Woodsmith. 2009. The Response of salmon populations to geomorphic measurements at three scales. North American Journal of Fisheries Management 29(3):549-559.

'KEY WORDS'
Watershed scale
Main stream
Southeast Alaska
Habitat
Coho salmon
Cutthroat trout

'ABSTRACT'

Protocols to assess stream channel response to disturbances often focus on physical aspects of the stream at the reach scale without measurements of fish populations. In this study, estimates of juvenile salmon abundance in 511 habitat units within 25 reaches of 12 streams were made over 4 years and juxtaposed with measurements of physical habitat at the habitat unit, reach, and watershed scales. Fish ranged in size from about 50 to 160 mm fork length. The amount of variation among densities differed by species and geographic scale. For most species, the habitat unit scale accounted for the most variation. Relationships between salmon density and measurements at the habitat unit scale varied. At the reach scale, we observed a negative relationship between abundance of coho salmon Oncorhynchus kisutch parr and number of pools. A positive relationship appeared between coho salmon parr and large wood. At the watershed scale, a positive relationship was observed between coho salmon parr and valley morphology. Valley morphology also entered the model for cutthroat trout O. clarkii. Differences in salmonid densities observed between northern and southern watersheds were attributed to differences in landforms, geology, and soils among islands in southeast Alaska. Simple habitat measures, such as pool counts, were not good predictors of fish abundance. However, geomorphic measures from multiple scales that are accompanied by estimates of fish abundance can provide managers with an integrated picture of watershed productivity and a better means to evaluate features that influence productivity.

Bugert, R. M., T. C. Bjornn, and W. R. Meehan. 1991. Summer habitat use by young salmonids and their responses to cover and predators in a small Southeast Alaska stream. Transactions of the American Fisheries Society 120:474-485.

'KEY WORDS'
Tributaries
Southeast Alaska
Coho Salmon
Dolly Varden
Steelhead
Habitat use

'ABSTRACT'

We observed young coho salmon Oncorhynchus kisutch, steelhead O. mykiss, and Dolly Varden Salvelinus malma in a second-order stream on Prince of Wales Island, Alaska, to assess differences between species in habitat use and response to cover and predators. Habitat use by subyearlings of the three species differed primarily in depth of water and position in the water column. Coho salmon selected the relatively deep areas of the small stream; steelhead were more evenly spread across the bottom, regardless of depth; and Dolly Varden were close to the bottom in water less than 20 cm deep. All three species selected lower positions in the water column in pools without cover than in pools with riparian or instream cover. We detected no shift in habitat use in response to fish predators.

Burchsted, D., M. Daniels, R. Thorson, and J. Vokoun. 2010. The River Discontinuum: Applying Beaver Modifications to Baseline Conditions for Restoration of Forested Headwaters. BioScience 60(11):908-922.

'KEY WORDS'

Off channel
Beaver
Fluvial geomorphology
Hierarchical patch dynamics
Sream ecology
Rriver continuum concept
River restoration
'ABSTRACT'
Billions of dollars are being spent in the United States to restore rivers to a desired, yet often unknown, reference condition. In lieu of a known reference, practitioners typically assume the paradigm of a connected watercourse. Geological and ecological processes, however, create patchy and discontinuous fluvial systems. One of these processes, dam building by North American beavers (Castor canadensis), generated discontinuities throughout precolonial river systems of northern North America. Under modern conditions, beaver dams create dynamic sequences of ponds and wet meadows among free-flowing segments. One beaver impoundment alone can exceed 1000 meters along the river, flood the valley laterally, and fundamentally alter biogeochemical cycles and ecological structures. In this article, we use hierarchical patch dynamics to investigate beaver-mediated discontinuity across spatial and temporal scales. We then use this conceptual model to generate testable hypotheses addressing channel geomorphology, natural flow regime, water quality, and biota, given the importance of these factors in river restoration.

Burnett, K. M., and coauthors. 2007. Distribution of salmon-habitat potential relative to landscape characteristics and implications for conservation. Ecological Applications 17:66-80.
'KEY WORDS'
Watershed scale
Oregon
Coho salmon
Landscape models
Habitat
salmon
'ABSTRACT'
The geographic distribution of stream reaches with potential to support high-quality habitat for salmonids has bearing on the actual status of habitats and populations over broad spatial extents. As part of the Coastal Landscape Analysis and Modeling Study (CLAMS), we examined how salmon-habitat potential was distributed relative to current and future (+100 years) landscape characteristics in the Coastal Province of Oregon, USA. The intrinsic potential to provide high-quality rearing habitat was modeled for juvenile coho salmon (Oncorhynchus kisutch) and juvenile steelhead (O. mykiss) based on stream flow, valley constraint, and stream gradient. Land ownership, use, and cover were summarized for 100-m analysis buffers on either side of stream reaches with high intrinsic potential and in the overall area encompassing the buffers. Past management seems to have concentrated nonindustrial private ownership, agriculture, and developed uses adjacent to reaches with high intrinsic potential for coho salmon. Thus, of the area in coho salmon buffers, 45% is either non-forested or recently logged, but only 10% is in larger-diameter forests. For the area in steelhead buffers, 21% is either non-forested or recently logged while 20% is in larger-diameter forests. Older forests are most extensive on federal lands but are rare on private lands, highlighting the critical role for public lands in near-term salmon conservation. Agriculture and development are projected to remain focused near high-intrinsic-potential reaches for coho salmon, increasing the importance of effectively addressing nonpoint source pollution from these uses. Percentages of larger-diameter forests are expected to increase throughout the province, but the increase will be only half as much in coho salmon buffers as in steelhead buffers. Most of the increase is projected for public lands, where policies emphasize biodiversity protection. Results suggest that widespread recovery of coho salmon is unlikely unless habitat can be improved in high-intrinsic-potential reaches on private

lands. Knowing where high-intrinsic-potential stream reaches occur relative to landscape characteristics can help in evaluating the current and future condition of freshwater habitat, explaining differences between species in population status and risk, and assessing the need for and feasibility of restoration.

Bustard, D. R., and D. W. Narver. 1975. Aspects of the winter ecology of juvenile coho salmon (Oncorhynchus kisutch) and steelhead trout (Salmo gairdneri). Journal of the Fisheries Research Board of Canada 32(5):667-680.

'KEY WORDS'
Tributaries
British Columbia
Winter habitat
Coho Salmon

'ABSTRACT'

The major physical characteristics of overwintering areas for juvenile coho salmon (oncorhynchus kisutch) and steelhead trout (salmo gairdneri) were described for a small, unlogged stream. Mean depths of water, maximum water velocities, water temperatures, winter cover selection, feeding and hiding behavior, tributary movements and environmental change effects were discussed for the coho salmon and steelhead trout.

Campbell, E. Y., and coauthors. 2011. Timber harvest intensifies spawning-salmon disturbance of macroinvertebrates in southeastern Alaskan streams. Journal of the North American Benthological Society 30(1):49-59.

'KEY WORDS'
Main Stream
Southeast Alaska
Effects of logging
Oncorhynchus spp
Benthic invertebrates
Ecosystem engineering
Bioturbation
Watershed management

'ABSTRACT'

We addressed how timber harvest can interact with salmon (Oncorhynchus spp.) spawning activities to influence benthic macroinvertebrate communities in streams on Prince of Wales Island, Alaska. We predicted that spawning salmon would cause greater disturbance to macroinvertebrates in streams in watersheds with high than with low timber-harvest intensity because finer sediments that accumulate in streams after timber harvesting would be readily dislodged by spawning salmon and lower volumes of large wood in the streams would decrease availability of macroinvertebrate refugia from salmon activity. We used a modified Hess sampler to collect benthic macroinvertebrates from 6 riffles in each of 7 streams before and during the annual salmon run. Diptera biomass was lower and Plecoptera biomass was higher during the salmon run than before it. During the run, insect biomass, scraper biomass, and Ephemeroptera biomass were higher in streams with low than with high timber-harvest intensity, possibly because the finer sediments in the high timber-harvest intensity streams were more readily dislodged by salmon. Macroinvertebrate community structure (density and biomass) differed between before and during the run. Epeorus longimanus, Baetis, Seratella tibialis, Suwallia, Chironomidae, and Simuliidae were significant indicators of before-salmon benthic communities, whereas Sweltsa and Zapada cinctipes typified during-salmon communities. Some taxa probably are more tolerant of salmon disturbance than others, and their life histories might be adapted to the autumn salmon run. Overall, our results indicate that strong interactive effects can occur between anthropogenic activities and natural disturbance and that timber-harvest activity can intensify the effects of spawning-salmon disturbance on macroinvertebrates.

Carlson, J. Y., C. W. Andrus, and H. A. Froehlich. 1990. Woody debris, channel features, and macroinvertebrates of streams with logged and undisturbed riparian timber in Northeastern Oregon, U.S.A. Canadian Journal of Fisheries and Aquatic Sciences 47:1103-1111.

'KEY WORDS'
Tributaries
Main stream
Effects of logging
Invertebrates
Oregon
Riparian habitat
Woody debris

'ABSTRACT'

Macroinvertebrate communities and several aspects of fish habitat were examined for 16 northeastern Oregon stream segments, 11 with undisturbed riparian forests and five where 26–54% of the riparian forest had been harvested 6 to 17 yr previously. Amounts of woody debris in streams and pools formed by the debris were similar between undisturbed and logged sites. Pool volume was inversely related to stream gradient and directly related to the amount of woody debris in the stream. Stream surface substrate composition was not significantly different between streams in logged and undisturbed areas. Macroinvertebrate density was 20 to 113 percent greater at the logged sites and diversity was similar at logged and undisturbed sites. Macroinvertebrates were most abundant at lower elevation streams and at streams that were shaded less by the surrounding vegetation. Timber harvesting activities do not appear to have damaged aquatic insect habitat and pool abundance was not altered, suggesting the habitat's carrying capacity for fish was not affected.

Carpenter, S. R., S. G. Fisher, N. B. Grimm, and J. F. Kitchell. 1992. Global change and freshwater ecosysytems. Annual Review of Ecology and Systematics 23:119-139.

'KEY WORDS'
Climate change
Aquatic ecosystems
Lakes
Streams
Wetlands

'ABSTRACT'

Reviews potential effects of climate change on lake and stream ecosystems. Considers effects on distribution and interactions of species. Consequences depend on spacial and temporal scales. Some generalizations are that precipitation and evapotranspiration will increase, precipitation and runoff are non-linear, some regions will become wetter, other drier. Increasing temperature regimes will place additional pressure on cold water species.

Cederholm, C. J., and L. M. Reid. 1987. Impact of Forest Management on coho salmon (*Oncorhynchus kitsutch*) popualtions of the Clearwater River, Washington: Project Summary. Pages 373-397 *in* E. O. Salo, and T. W. Cundy, editors. Streamside Management: Forestry and Fishery Interactions, Seattle, WA.

'KEY WORDS'
Main Stream
Washington
Landslides
Forest Mangement
Effects of logging

'ABSTRACT'

Reviews major effects of logging in the clearwater drainage. Landslides and roads increase sediment inputs.

Clark, J. H., A. McGregor, R. D. Mecum, P. Krasnowski, and A. M. Carrol. 2006. The commercial fishery in Alaska. Alaska Fishery Research Journal 12(1):1-146.

'KEY WORDS'
Watershed Scale
Fishery management
Econonic value
salmon harvest
History

'ABSTRACT'

Alaska's commercial fsalmon fisheries harvested and average of 172 million salmon annually since 1990 with a range of 123-221 million fish. In 1950's annual harvest was 41 million fish. Policies and harvest regulation were insituted to rebuild the run over the ensuing years to it's present sustainable levels. Primary causes for depressed stocks was over-fishing.

Coates, R., L. Collins, J. Florsheim, and D. Kaufman. 1985. Channel change, sediment transport, and fish habitat in a coastal stream: effects of an extreme event. Environmental Management 9(1):35-48.

'KEY WORDS'
Main stream
California
Channel change
Sediment transport
Fish habitat
Floods

'ABSTRACT'

A study on sediment transport and channel change was conducted on Zayante Creek and the lower San Lorenzo River in Santa Cruz County, California. A rainstorm with a recurrence interval locally in excess of 150 years occurred during the study year, 1982 WY. Stream surveys indicated that significant aggradation occurred during and after the peak flood. Upper study reaches were substantially recovered after high flows of early April, but the lower study reaches still had significant filling of pools and burial of riffles by sand. Increases in width-depth ratio were minor and localized in upper reaches, but were significant in lower reaches. Large inputs of sand, primarily from landsliding, altered the sediment transport regime. A higher proportion of the bedload is now transported by lower flows than before the January event. Roads and sand quarries contributed significantly to sediment input to the stream. A proposed dam may alter the sediment transport regime of Zayante Creek. Mitigating the effects of this dam on downstream fish habitat may require occasional bankfull discharges.

Collen, P., and R. J. Gibson. 2001. The general ecology of beavers (Castor spp.) as related to their influence on stream ecosystems and riparian habitats, and subsequent effects on fish-- a review. Reviews in Fish Biology 10:439-461.

'KEY WORDS'
Off channel
North America
Beaver
Beaver ponds
Stream habitat
Life history
Salmonids

'ABSTRACT'

The Eurasian and North American beavers are similar in their ecological requirements, and require water deep enough to cover the entrance to their lodge or burrow. A food cache is often built next to the lodge or burrow, exceptin some southern areas. On small streams (upto fourth order) dams are frequently built tocreate an impoundment, generally on low gradient streams, although at high populationdensities dams may be built on steeper gradientstreams.

On large rivers or in lakes, simply alodge with its food cache may be built. The beaver is a keystone riparian species in thatthe landscape can be considerably altered byits activities and a new ecosystem created. The stream above a dam changes from lotic tolentic conditions. There are hydrological,temperature and chemical changes, depending ontypes of dams and locations. Although theinvertebrates may be fewer per unit area, totalnumber of organisms increases, and diversityincreases as the pond ages. In cool, smallorder streams, the impoundments provide betterhabitat for large trout, possibly creatingangling opportunities. However, at sites wherewater temperatures rise above their optimum preferenda, salmonids may be replaced by other species, such as cyprinids, catostomids,percids or centrarchids. As the habitat isaltered, interactions amongst co-habiting species may change. For example, brown troutor brook trout (charr) may become dominant overAtlantic salmon. In warm water streams theremay be a shift from faster water dwellers topond dwellers. Larger bodied fish, such ascentrarchids and esocids may displace smallerbodied fish such as cyprinids, providing betterangling. Refugia from high or low water flows,low oxygen or high temperatures, may beprovided in adverse conditions in winter orsummer. However, in some cases dams areobstructions to upstream migration, andsediment may be deposited in former spawningareas. The practicality and benefits ofintroducing or restoring beaver populationswill vary according to location, and should beconsidered in conjunction with a managementplan to control their densities.

Collins, B. D., D. R. Montgomery, and A. D. Haas. 2002. Historical changes in the distribution and functions of large wood in Puget Sound lowland rivers. Canadian Journal of Fisheries and Aquatic Sciences 59:66-76.

'KEY WORDS'
Main stream
Washington
Large wood
Stream morphology
Historical
Habitat

'ABSTRACT'

We examined changes in wood abundance and functions in Puget Lowland rivers from the last ~150 years of land use by comparing field data from an 11-km-long protected reach of the Nisqually River with field data from the Snohomish and Stillaguamish rivers and with archival data from several Puget Lowland rivers. Current wood abundance is one to two orders of magnitude less than before European settlement in the Snohomish and Stillaguamish basins. Most importantly, wood jams are now rare because of a lack of very large wood that can function as key pieces and low rates of wood recruitment. These changes in wood abundance and size appear to have fundamentally changed the morphology, dynamics, and habitat abundance and characteristics of lowland rivers across scales from channel unit to valley bottom. Based on our field studies, rivers had substantially more and deeper pools historically. Archival data and field studies indicate that wood jams were integral to creating and maintaining a dynamic, anastomosing river pattern with numerous floodplain channels and abundant edge habitat and routed floodwaters and sediment onto floodplains. Establishing the condition of the riverine landscape before European settlement sets a reference against which to evaluate contemporary conditions and develop restoration objectives.

Colyer, W. T., J. L. Kershner, and R. H. Hilderbrand. 2005. Movements of fluvial bonneville cutthroat trout in the Thomas Fork of the Bear River, Idaho. North American Journal of Fisheries Management 25:954-963.

'KEY WORDS'
Main stream
Cutthroat trout
Idaho

Wyoming
Movement
Radio tracking
'ABSTRACT'

The majority of interior subspecies of cutthroat trout Oncorhynchus clarkii have been extirpated from large rivers by anthropogenic activities that have fragmented habitats and introduced nonnative competitors. Selective pressures against migratory behaviors and main-stem river occupation, coupled with conservation strategies that isolate genetically pure populations above barriers, have restricted gene flow and prevented expression of the fluvial life history in many populations. Existing knowledge about the movements and home range requirements of fluvial cutthroat trout is, therefore, limited. Our objectives in this study were to (1) determine the extent of seasonal home ranges and mobility of Bonneville cutthroat trout O. c. utah (BCT) in the Thomas Fork and main-stem Bear River and (2) evaluate the role of a water diversion structure functioning as a seasonal migration barrier to fish movement. We implanted 55 BCT in the Thomas Fork of the Bear River, Idaho, with radio transmitters and located them bimonthly in 1999–2000 and weekly in 2000–2001. We found fish to be more mobile than previously reported. Individuals above the diversion barrier occupied substantially larger home ranges than those below the barrier (analysis of variance: P = 0.0003; median = 2,225 m above barrier; median = 500 m below barrier) throughout our study, and they moved more frequently (mean, 0.89 movements/contact; range, 0.57–1.00) from October 2000 through March 2001 than fish below the barrier (mean, 0.45 movements/contact; range, 0.00–1.00). During the spring of both years, we located radio-tagged fish in both upstream and neighboring tributaries as far as 86 km away from our study site. Our results document the existence of a fluvial component of BCT in the Bear River and its tributaries and suggest that successful efforts at conservation of these fish must focus on main-stem habitats and the maintenance of seasonal migration corridors.

Connor, W. P., H. L. Burge, J. R. Yearsley, and T. C. Bjornn. 2003. Influence of Flow and Temperature on Survival of Wild Subyearling Fall Chinook Salmon in the Snake River. North American Journal of Fisheries Management 23(2):362-375.

'KEY WORDS'
Main Stream
Climate change
Idaho
Chinook
Fishery management
Smolts
Dams
Water temperature
Tagging
Temperature effects
Hydroelectric power plants
'ABSTRACT'

Summer flow augmentation to increase the survival of wild subyearling fall chinook salmon Oncorhynchus tshawytscha is implemented annually to mitigate for the development of the hydropower system in the Snake River basin, but the efficacy of this practice has been disputed. We studied some of the factors affecting survival of wild subyearling fall chinook salmon from capture, tagging, and release in the free-flowing Snake River to the tailrace of the first dam encountered by smolts en route to the sea. We then assessed the effects of summer flow augmentation on survival to the tailrace of this dam. We tagged and released 5,030 wild juvenile fall chinook salmon in the free-flowing Snake River from 1998 to 2000. We separated these tagged fish into four sequential within-year release groups termed cohorts (N = 12). Survival probability estimates (mean +/- SE) to the tailrace of the dam for the 12 cohorts when summer flow augmentation was implemented ranged from 36% +/- 4% to 88% +/- 5%. We fit an ordinary least-squares multiple regression model from indices of flow and

temperature that explained 92% (N = 12; P < 0.0001) of the observed variability in cohort survival. Survival generally increased with increasing flow and decreased with increasing temperature. We used the regression model to predict cohort survival for flow and temperature conditions observed when summer flow augmentation was implemented and for approximated flow and temperature conditions had the summer flow augmentation not been implemented. Survival of all cohorts was predicted to be higher when flow was augmented than when flow was not augmented because summer flow augmentation increased the flow levels and decreased the temperatures fish were exposed to as they moved seaward. We conclude that summer flow augmentation increases the survival of young fall chinook salmon.

Crone, R. A., and C. E. Bond. 1976. Life history of coho salmon (*Oncorhynchus kisutch*), in Sashin Creek, southeastern Alaska. Fishery Bulletin 74(4):897-923.
'KEY WORDS'
Main Stream
Southeast Alaska
Coho salmon
Life history
'ABSTRACT'
Extensive review of life history of coho salmon in Sashin Creek from 1963-1968. Reports age composition and fecundity of returning adults, escapement numbers, and mean redd life. Reports age composition of smolt, 37% 1-y 59% 2-yr , 4% 3-yr. Average fork length 83 mm, 105, and 104 respectively.

Cummins, K. W. 1974. Structure and function of stream ecosystems. BioScience 24(11):631.
'KEY WORDS'
Watershed scale
Aquatic insects
Ecology
Foraging strategies
Functional groups
Invertebrates
'ABSTRACT'
General functional, interrelated components have been defined and initially dimensioned for representative stream ecosystems. Additional or alternative functional criteria are sought to replace classical taxonomic units. Attention is now focused on the efficient conversion of organic matter to CO2 and the maintenance of a minor role for in-stream plant growth.

Davies, B. R., and K. F. Walker. 1986. The ecology of river systems, volume 60. DR W. Junk, Dordrecht.
'KEY WORDS'
Watershed scale
Mian Stream
Large rivers
fish
invertebrate
'ABSTRACT'
Reviews chemical, physical, and biological aspects of large river systems in Africa, North America, Asia, & Europe, includes McKenzie River system. Papeers include dscriptions of fish and invertebrate populations with an emphasis on rivers as eclogicla units.

Decamps, H. 1993. River margins and environmental change. Ecological Applications 3(3):441-445.
'KEY WORDS'
Riparian vegetation
Ecosystem mangement
Climate change

'ABSTRACT'
The paper discusses how variability of river margins interacts with riparian function at the landscape level, in order to develop inferences about the future of this interaction with respect to potential effects of a global climatic change. A riparian approach to the ecology of river landscapes should be useful in that it offers an opportunity to improve our understanding and management of the effects of environmental change at the ecosystem level.

Dolloff, C. A. 1987. Seasonal population characteristics and habitat use by juvenile coho salmon in a small southeast Alaska stream. Transactions of the American Fisheries Society 116(6):829-838.

'KEY WORDS'
Tributaries
Southeast Alaska
Coho Salmon
Effects of logging
Habitat use

'ABSTRACT'
The density, growth, production, and movements of juvenile coho salmon Oncorhynchus kisutch from a wild population were evaluated after the fish were transplanted into five types of habitat (clear-cut, forest, meadow, slough tributary, forest tributary) in a small southeastern Alaska stream. Instantaneous growth ranged from 0.0066 in the clear-cut habitat to 0.0055 in the slough tributary. Daily increase in fork length was about 0.10 mm/d system-wide. Annual production of coho salmon in each habitat type was: meadow, 3.32 g/m2; slough tributary, 2.47 g/m2; clear-cut, 1.75 g/m2; forest, 1.59 g/m2; and forest tributary, 1.34 g/m2. During all sampling periods, most fish were recaptured at the site where they were released; those fish that moved neither selected nor avoided specific habitat types. These findings suggest that all habitats should be managed to meet both the summer and winter needs of juvenile coho salmon because most fish do not move among habitats after the initial population adjustment in the spring. The ability of a stream to produce fish depends not only on the amount and accessibility of habitat, but also on the distribution of habitat types.

Dolloff, C. A., and G. H. Reeves. 1990. Microhabitat partitioning among stream-dwelling juvenile coho salmon,(*Oncorhynchus kisutch*), and Dolly Varden,(*Salvelinus malma*). Canadian Journal of Fisheries and Aquatic Sciences 47(12):2297-2306.

'KEY WORDS'
Tributaries
Southeast Alaska
Large wood
Coho salmon
Dolly Varden

'ABSTRACT'
Microhabitat use and partitioning among age 0+ and 1+ coho salmon, Oncorhynchus kisutch, and age 0+, 1+ and 2+ Dolly Varden, Salvelinus malma, was studied in small (<2 m wide), natural streams on Prince of Wales Island, Alaska, and in laboratory stream channels. Coho salmon occupied midwater positions that they defended from other fish. Dolly Varden were more closely associated with the stream bottom and were seldom territorial. For each species, the depth of water, depth of focal point, and distance to nearest fish increased with fish size, whereas the distance to nearest cover decreased as fish size increased. Most fish selected focal point velocities between 0.0–9.0 cm·s−1.Woody debris was the most frequently used cover type and most fish occurred over gravel substrates ranging from 2–100 mm particle diameter. Habitat use by each species in the laboratory was similar to the pattern observed in the field. Each species occupied similar habitats both when alone and when the other species was present. Although habitat use by juveniles of coho salmon and Dolly Varden overlapped among several key parameters, each species primarily exploited resources not readily available to or selected by the other in the natural streams we studied.

Duncan, W. F. A., and M. A. Brusven. 1985. Energy dynamics of three low-order southeast Alaska streams; Allochthonous Processes. Journal of Freshwater Ecology 3(2):233-248.

'KEY WORDS'
Tributaries
Southeast Alaska
Allochthonous debris
Organic debris
Riparian vegetation
Woody debris

'ABSTRACT'

Physical and biotic processes of three low-order southeast Alaska streams located on Prince of Wales Island were studied. These streams drained an undisturbed watershed representing a coniferous climax forest, a recently logged watershed with little riparian regeneration, and a logged watershed with heavy riparian growth. Community respiration and production were measured in closed, recirculating 12-L Plexiglas metabolism chambers using the dissolved oxygen method. Gross production among the streams varied from 0.1-2.7 g O sub(2)/m super(2)/d; respiration varied from 0.1-1.0 g O sub(2)/m super(2)/d. Highest rates of production and respiration occurred in the recently logged stream; lowest rates were measured in the mature, climax forest stream. Seasonal differences in production and respiration were apparently influenced by logging.

Dunning, J. B., B. J. Danielson, and H. R. Pulliam. 1992. Ecological processes that affect populations in complex landscapes. Oikos 65(1):169-175.

'KEY WORDS'
Watershed scale
Ecological theory
Landscape ecology
Population dynamics

'ABSTRACT'

We describe a general framework for understanding the ecological processes that operate at landscape scales. The composition of habitat types in a landscape and the physiognomic or spatial arrangement of those habitats are the two essential features that are required to describe any landscape. As such, these two features affect four basic ecological processes that can influence population dynamics or community structure. The first two of these processes, landscape complementation and landscape supplementation, occur when individuals move between patches in the landscape to make use of non-substitutable and substitutable resources, respectively. The third process, source-sink dynamics, describes the consequences of having different individuals in the same population occupy habitat patches of different qualities. The fourth process, the neighborhood effect, describes how landscape effects can be amplified when the critical resources are in the landscape immediately surrounding a given patch. Definition of these landscape features and general processes will allow a better synthesis of how landscape variation affects populations and communities.

Eaton, D. W., and F. J. Adams. 1995. The effects of global warming on the distribution of steelhead trout populations on the Alaska Peninsula, Alaska. King Salmon Fishery Resource Office, U.S. Fish and Wildlife Service, King Salmon, Alaska.

'KEY WORDS'
Steelhaed ttrout
Climate change
Temperature
Alaska

'ABSTRACT'

Eaton, J. G., and R. M. Scheller. 1996. Effects of climate warming in fish thermal habitat in streams of the United States. Limnology and Ocenaography 41(4):1109-1115.

'KEY WORDS'
Climate change
Thermal tolerance
General circulation model
Fish habitat

'ABSTRACT'

The effects of climate warming on the thermal habitat of 57 species of fish of the U.S. were estimated using results for a doubling of atmospheric carbon dioxide that were predicted by the Canadian Climate Center general circulation model. Baseline water temperature conditions were calculated from data collected at 1,700 U.S. Geological Surveys stream monitoring stations across the U.S. Water temperatures after predicted climate change were obtained by multiplying air temperature changes by 0.9, a factor based on several field studies, and adding them to baseline water temperatures at stations in corresponding grid cells. Results indicated that habitat for cold and cool water fish would be reduced by ~ 50%, and that this effect would be distributed throughout the existing range of these species. Habitat losses were greater among species with smaller initial distributions and in geographic regions with the greatest warming (e.g. the central Midwest). Results for warm water fish habitat were less certain because of the poor state of knowledge regarding their high and low temperature tolerance; however, the habitat of many species of this thermal guild likely will also be substantially reduced by climate warming, whereas the habitat of other species will be increased.

Ebersole, J. L., and coauthors. 2006. Juvenile coho salmon growth and survival across stream network seasonal habitats. Transactions of the American Fisheries Society 135:1681-1697.

'KEY WORDS'
Watershed scale
Oregon
Columbia River
Willamette River
Movement

'ABSTRACT'

Understanding watershed-scale variation in juvenile salmonid survival and growth can provide insights into factors influencing demographics and can help target restoration and mitigation efforts for imperiled fish populations. We assessed growth, movement, and apparent overwinter survival of individually tagged juvenile coho salmon Oncorhynchus kisutch in a coastal Oregon watershed from June 2002 to June 2003 and related growth and survival parameters to stream characteristics. Fall body size of juvenile coho salmon was a good predictor of smolt size and survival, but smolt size was also influenced by overwintering location. This was due to strong spatial patterns in winter growth rates associated with residency and movement into a small intermittent tributary. Though nearly dry in midsummer, this stream supported high densities of spawning coho salmon in the fall, and juveniles rearing there exhibited relatively high growth rates and emigrated as larger smolts. Improved winter growth and survival of juvenile coho salmon utilizing tributary habitats underscore the importance of maintaining connectivity between seasonal habitats and providing a diversity of sheltering and foraging opportunities, particularly where main-stem habitats have been simplified by human land uses.

Fausch, K. D. 1993. Experimental analysis of microhabitat selection by juvenile steelhead (Oncorhynchus mykiss) and coho salmon (O. kisutch) in a British Columbia stream. Canadian Journal of Fisheries and Aquatic Sciences 50:1198-1207.

'KEY WORDS'
Main Stream
British Columbia

Water velocity
Coho salmon
Steelhead
Habitat selection
'ABSTRACT'

Replicate experiments were conducted in the Salmon River, British Columbia, during early summer 1990 to test the relative importance of velocity refuge, visual isolation, and overhead cover to microhabitat selection by steelhead (Oncorhynchus mykiss) parr and age-0 coho salmon (O. kisutch). Four types of artificial Plexiglas structures, the first three of identical construction, had different portions painted to provide increasing habitat complexity: velocity refuge alone, velocity refuge with visual isolation, all three features combined, and overhead cover alone. Steelhead parr selected structures with overhead cover alone or all three features significantly more often than those without overhead cover. Steelhead also selected structures adjacent to the swiftest velocities available and closest to other natural overhead cover, which accounted for most differences in use of the same structure in different locations. In contrast, few age-0 coho salmon used any structures. Those that did selected the three types of structures with velocity refuge about equally, but significantly more often than those with overhead cover alone, regardless of their location. Field experiments such as this hold promise for elucidating mechanisms of habitat selection by stream salmonids.

Fausch, K. D., and T. G. Northcote. 1992. Large woody debris and salmonid habitat in a small coastal. Canadian Journal of Fisheries and Aquatic Sciences 49:682-693.
'KEY WORDS'
Tributaries
British Columbia
Woody debris
Pools
Coho salmon
Cutthroat trout
'ABSTRACT'

Sections of a small coastal British Columbia stream that had previously been cleaned of large woody debris (LWD) were compared with sections where most debris was left and with others where debris had been relatively undisturbed for at least 40 yr. Three sections where debris had been removed had simple habitat that was less sinuous, wider, and shallower and had less pool volume and overhead cover than four sections with more complex habitat where debris was retained. Habitat in four relatively undisturbed sections was generally similar to complex sections. Most pools in all sections were scour or plunge pools formed by LWD or large roots oriented perpendicular to the flow or angled downstream. Standing crop (kolograms per hectare) and individual weights of age 1 + and older coho salmon (Oncorhynchus kisutch) and cutthroat trout (O. clarki) were significantly greater ($P < 0.02$) in complex than in simple sections. Biomass of age 1 + and older salmonids was closely related to section pool volume ($r2 = 0.92$, $P = 0.0006$). Projections based on this model and average habitat conditions suggest that during 1990 a total of 8.0 kg of salmonid biomass, 5 times the current standing crop, was forgone in the 332-m simple reach due to prior debris removal.

Feller, M. C., and J. P. Kimmins. 1984. Effects of clearcutting and slash burning on streamwater chemistry and watershed nutrient budgets in southwestern British Columbia. Water Resources Research 20(1):29-40.
'KEY WORDS'
Tributaries
Watershed Scale
British Columbia
Chemistry-stream
Effects of logging
Nutrient budget

Slash burning
Stream management
Watershed study
'ABSTRACT'
Two small forested watersheds near Haney in southwestern British Columbia were partially clearcut, and the slash on one of them was subsequently burned. Streamwater chemistry was monitored in these treated watersheds and an undisturbed control watershed for 2 years prior to treatment and up to 9 years after treatment. The chemical parameters that were studied responded differently to the treatments, but there was a general pattern of increased concentrations and fluxes in Streamwater for the first 2–3 years following treatment followed by a decline to, and sometimes below, pretreatment values. The most pronounced increases were observed for K and NO3. It was not possible to determine the exact causes of these changes in concentrations and fluxes, due to the great variability in the ecosystems present. This variability precluded determination of statistically significant changes in annual terrestrial nutrient fluxes and pools. Stream nutrient exports usually were <10 kg/ha/yr for each of N, P, K, and Mg, <20 kg/ha/yr for Na and Cl, and <30 kg/ha/yr for Ca. These values were considerably less than nutrient exports in harvested logs and in losses to the atmosphere during the slashburn. Clearcutting and burning caused greater nutrient losses than Clearcutting alone, particularly in the case of N, where the Clearcutting and Clearcutting and burning treatments resulted in total losses of 245 kg/ha and 1293 kg/ha, respectively, for the first two years after treatment.

Finney, B. P., I. Gregory-Eaves, J. Sweetman, M. S. V. Douglas, and J. P. Smol. 2000. Impacts of Climatic Change and Fishing on Pacific Salmon Abundance Over the Past 300 Years. Science 290(5492):795-799.
'KEY WORDS'
Climate change
Alaska
Sockeye salmon
'ABSTRACT'
The effects of climate variability on Pacific salmon abundance are uncertain because historical records are short and are complicated by commercial harvesting and habitat alteration. We use lake sediment records of δ15N and biological indicators to reconstruct sockeye salmon abundance in the Bristol Bay and Kodiak Island regions of Alaska over the past 300 years. Marked shifts in populations occurred over decades during this period, and some pronounced changes appear to be related to climatic change. Variations in salmon returns due to climate or harvesting can have strong impacts on sockeye nursery lake productivity in systems where adult salmon carcasses are important nutrient sources.

Frissell, C. A., W. J. Liss, C. E. Warren, and M. D. Hurley. 1986. A Hierarchical Framework for Stream Habitat Classification: Viewing Streams in a Watershed Context. Environmental Management 10(2):199-214.
'KEY WORDS'
Watershed scale
Habitat classification
Stream habitat
Variation-spacial
Variation-temporal
'ABSTRACT'
Classification of streams and stream habitats is useful for research involving establishment of monitoring stations, determination of local impacts of land-use practices, generalization from site-specific data, and assessment of basin-wide, cumulative impacts of human activities on streams and their biota. This article presents a frame-work for a hierarchical classification system, entailing an organized view of spatial and temporal variation among and within stream systems. Stream habitat systems, defined and classified on several spatiotemporal

scales, are associated with watershed geomorphic features and events. Variables selected for classification define relative long-term capacities of systems, not simply short-term states. Streams and their watershed environments are classified within the context of a regional biogeoclimatic landscape classification. The framework is a perspective that should allow more systematic interpretation and description of watershed-stream relationships.

Gard, R. 1961. Effects of beaver on trout in Sagehen Creek California. Journal of Wildlife Management 25:221-242.
'KEY WORDS'
Off channel
Beaver
Habitat use
Invertebrates
Spawning
Trout
'ABSTRACT'
Beaver ponds are productive habitats for trout

Good, T. P., J. Davies, B. J. Burke, and M. H. Ruckelshaus. 2008. Timber harvest transforms ecological roles of salmon in southeast alaska rain forest streams. Ecological Applications 18(1):246-257.
'KEY WORDS'
Watershed scale
Pacific Northwest
Puget Sound
Catastrophic loss
Chinook salmon
Evolutionarily significant unit (ESU)
Onchorhynchus tshawytscha
Population viability
Risk analysis
'ABSTRACT'
Catastrophic die-offs can have important consequences for vertebrate population growth and biodiversity, but catastrophic risks are not commonly incorporated into endangered-species recovery planning. Natural (e.g., landslides, floods) and anthropogenic (e.g., toxic leaks and spills) catastrophes pose a challenge for evolutionarily significant units (ESUs) of Pacific salmon listed under the Endangered Species Act and teetering at precariously low population levels. To spread risks among Puget Sound chinook salmon populations, recovery strategies for ESU-wide viability recommend at least two viable populations of historical life-history types in each of five geographic regions. We explored the likelihood of Puget Sound chinook salmon ESU persistence by examining spatial patterns of catastrophic risk and testing ESU viability recommendations for 22 populations of the threatened Puget Sound chinook salmon ESU. We combined geospatial information about catastrophic risks and chinook salmon distribution in Puget Sound watersheds to categorize relative catastrophic risks for each population. We then analyzed similarities in risk scores among regions and compared risk distributions among strategies: (1) population groups selected using the ESU viability recommendations of having populations spread out geographically and including historical life-history diversity, and (2) population groups selected at random. Risks from individual catastrophes varied among populations, but overall risk from catastrophes was similar within geographic regions. Recovery strategies that called for two viable populations in each of five geographic regions had lower risk than random strategies; strategies that included life-history diversity had even lower risks. Geographically distributed populations have varying catastrophic-risks profiles, thus identifying and reinforcing the spatial and life-history diversity critical for populations to respond to environmental change or needed to rescue

severely depleted or extirpated populations. Recovery planning can promote viability of Pacific salmon ESUs across the landscape by incorporating catastrophic risk assessments.

Gowan, C., M. K. Young, K. D. Fausch, and S. C. Riley. 1994. Restricted movement in resident stream salmonids: A paradigm lost? Canadian Journal of Fisheries and Aquatic Sciences 51:2626-2637.
'KEY WORDS'
Watershed scale
Movement
Trout
North America
'ABSTRACT'
Gerking (1959. Biol. Rev. 34: 221–242) proposed a theory about the restricted movement of stream fishes that may be considered a paradigm in salmonid biology. The restricted movement paradigm (our term) hold that resident stream salmonids are sedentary. Numerous studies have supported the restricted movement paradigm, but nearly all have relied on the recapture of marked fish from the same areas in which they were released, an approach we believe is biased against detecting movement. We found substantial movement of trout in streams in Colorado and Wyoming using two-way weirs and radio telemetry. A review of the research on Lawrence Creek, Wisconsin, also showed that movement was important in the response of the trout population to habitat enhancement. Movement of resident stream fish has profound implications for research (e.g., measuring production and habitat models) and management (e.g., habitat enhancement, special regulations, and stocking hatchery fish). Methods capable of detecting fish movement could be incorporated into many studies to assess its importance in systems of interest. New theories and experiments are needed to understand the mechanisms that cause stream salmonids to move.

Greene, C. M., D. W. Jensen, G. R. Pess, E. A. Steel, and E. Beamer. 2005. Effects of Environmental Conditions during Stream, Estuary, and Ocean Residency on Chinook Salmon Return Rates in the Skagit River, Washington. Transactions of the American Fisheries Society 134(6):1562-1581.
'KEY WORDS'
Watershed scale
Estuarine
Washington
Chinnook salmon
Models
'ABSTRACT'
We predicted 22 years of return rates for wild Chinook salmon Oncorhynchus tshawytscha as a function of environmental conditions experienced during residency in freshwater, tidal delta, bay, and ocean habitats as well as as an indicator of density dependence (based on egg production) across life stages. The best predictors of return rate included the magnitude of floods experienced during incubation, a principal components factor describing environmental conditions during bay residency, a similar factor describing conditions experienced during the third ocean year, and an estimate of egg production. Our models explained up to 90% of the variation in return rate and had a very high forecasting precision, yet environmental conditions experienced during ocean residency explained only 5% of the variation. Our results suggest that returns of wild Chinook salmon can be predicted with high precision by incorporating habitat residency and that freshwater and nearshore environmental conditions strongly influence the survival of Skagit River Chinook salmon.

Gregory, S. V., F. J. Swanson, W. A. McKee, and K. W. Cummins. 1991. An ecosystem perspective of riparian zones. BioScience 41(8):540-551.
'KEY WORDS'
Watershed scale

Ecology
Stream ecology
'ABSTRACT'
A conceptual model of riparian zones integrates the physical processes that shape the landscapes, the succession of terrestrial plant communities on the geomorphic surfaces, the formation of habitat, and the production of nutritional resources for aquatic ecosystems. This ecosystem perspective of riparian zones emphasizes lotic ecosystems and the geomorphic organization of fluvial landforms. The linear nature of lotic ecosystems enhances the importance of riparian zones in landscape ecology. River valleys connect montane headwaters with lowland terrains, providing avenues for the transfer of water, nutrients, sediment, particulate organic matter, and organisms. Frequent disturbance events in riparian zones create complex mosaics of landforms and associated biological communities that often are more heterogeneous and diverse than those associated with upslope landscapes. Flooding is frequent, often annual or even more frequent, and a single flood may modify hundreds of square kilometers of river valley. Interactions between terrestrial and aquatic ecosystems include modification of microclimate, alteration of nutrient inputs from hillslopes, contribution of organic matter to streams and floodplains, and retention of inputs. It is concluded that an ecosystem perspective of riparian zones provides a rigorous ecological basis for identifying riparian zone management objectives, evaluating current land-use practices, and developing future resource alternatives.

Groot, C., and L. Margolis, editors. 1991. Pacifc Salmon Life Histories. UBC Press, Vancouver, BC.
'KEY WORDS'
Watershed scale
Life history
Pacific salmon
coho
sockeye
pink
chum
chinook
'ABSTRACT'
Provides separate life history for the five species of Pacific salmon by separate authors

Halupka, K. C., M. D. Bryant, M. F. Willson, and F. H. Everest. 2000. Biological characteristics and population status of anadromous salmon in southeast Alaska. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. General Technical Report GTR-PNW-468, Portland, OR.
'KEY WORDS'
Watershed scale
Southeast Alaska
Assessments
Population biology
Salmonid biology
Stock dynamics
'ABSTRACT'
Reviews various life history and morphometric feature to identify unique characteristics of salmon in southeast Alaska. Provides a review of a range of life history features of stock, including run size and timing. Identifies some stocks with unique feeatures and run timing.

Halupka, K. C., M. F. Willson, M. D. Bryant, F. H. Everest, and A. J. Gharret. 2003. Conservation of population diversity of Pacific Salmon in southeast Alaska. North American Journal of Fisheries Management 23:1057-1086.
'KEY WORDS'
Watershed scale

Southeast Alaska
Conservation biology
Anadromous salmonids
Population diversity
'ABSTRACT'

We analyzed intraspecific variation in selected biological characteristics of five species of Pacific salmon Oncorhynchus spp. in southeast Alaska and adjacent areas of Canada with a particular interest in describing the variation among populations and suggesting conservation priorities to preserve existing variation. We identified traits that showed high levels of among-population variation, evaluated the interspecific consistency of variation patterns, and noted the relationship of these traits to potential adaptive variation. In addition, we graphically identified populations with distinctive phenotypic and demographic characteristics as outliers from the distribution of mean values of traits taken from populations throughout the region. We also reviewed allozyme surveys to identify populations that differed in terms of the geographic clustering patterns of allele frequencies. Approximately 9,000 salmon populations occur in the study area, and sufficient data were available from 2,062 (23%) of them to analyze at least one characteristic. We identified 47 populations represented by adequate data sets that have distinctive characteristics. An additional 35 populations, represented by limited samples or unusual nominal traits, may be regionally distinctive. Of the 47 adequately sampled, distinctive populations, 22 met our criteria for conservation consideration: (1) high potential for adaptive variation (including distinctive run timing), (2) a distinctive trait combined with high spawner abundance or allozyme frequencies that diverge from geographic clustering patterns, and (3) more than one distinctive characteristic or freshwater habitat shared with other distinctive populations. Freshwater habitats for 6 of those 22 populations are located in watersheds that do not have restrictive land use designations and warrant the highest conservation priority.

Hartman, G. F., and J. C. Scrivener. 1990. Impacts of forestry practices on a coastal stream ecosystem, Carnation Creek, British Columbia. Canadian Bulletin Fisheries and Aquatic Sciences 223.
'KEY WORDS'
Main Stream
British Columbia
Carnation Creek
Habitat
Effects of logging
Forest management
Stream ecology
'ABSTRACT'

Results from the first 17 yr of a multi-disciplinary study about the effects of logging activities on a small stream ecosystem in the coastal rainforest of British Columbia have been reviewed. The main hydrological, fluvial-geomorphological, thermal, and production relationships are integrated in four schematic illustrations. The study has revealed that each activity conducted within an overall forest management plan may affect the physical components of an ecosystem differently.

Hartman, G. F., J. C. Scrivener, and M. J. Miles. 1996. Impacts of logging in Carnation Creek, a high-energy coastal stream in British Columbia, and their implications for restoring fish habitat. Canadian Journal of Fisheries and Aquatic Sciences 53(Suppl. 1):237-251.
'KEY WORDS'
Watershed scale
Main channel
British Columbia
Salmon habitat
Carnation Creek
Movement

Sediment
Effects of logging
Habitat restoration
'ABSTRACT'
Review of studies and result on the Carnation Creek watershed and effects of logging. Reviews effects on sediment., temperature, and habitat. Speceis include chum, coho, steelhead, and trout. Applies result to problems of restroration.Describes logging effects over > 20 year period that include reduced rearing and spawning habitat as a result of landslids, bank erosion, and loss of large wood. Presents a case for habitat improvement projects that include the life history of all species in the system and processes that maintain production and those that limit production.

Harvey, B. C. 1998. Influence of large woody debris on retention, immigration, and growth of coastal cutthroat trout (Oncorhynchus clarki clarki) in stream pools. Canadian Journal of Fisheries and Aquatic Sciences 55:1902-1908.
'KEY WORDS'
Tributaries
Headwater streams
California
Cutthroat trout
Large wood
Pools
'ABSTRACT'
Over 4 months and about 1 year, coastal cutthroat trout (Oncorhynchus clarki clarki) age-1 in Little Jones Creek, California, remained at similar rates in pools with and without large woody debris. This result was based on attempts in July and November 1995 to collect and tag all fish in 22 pools and three collections of fish from the same pools in November 1995, May 1996, and August 1996. Retention of fish appeared to be greater in pools with large woody debris in May 1996. The presence of large woody debris in pools did not influence immigration or growth of cutthroat trout. However, both immigration and growth increased downstream over the 3850-m study reach. Low retention and substantial immigration of cutthroat trout into experimental pools indicate that movement is important in the dynamics of this population. First- and second-order channels appear to be important sources of fish for the third-order study reach, while the study reach may export significant numbers of fish to downstream reaches accessible to anadromous fish.

Harvey, B. C., R. J. Nakamoto, and J. L. White. 1999. Influence of large woody debris and a bankfull flood on movement of adult resident coastal cutthroat trout (Oncorhynchus clarki) during fall and winter. Canadian Journal of Fisheries and Aquatic Sciences 56(1):2161-2166.
'KEY WORDS'
Tributaries
Cutthroat trout
California
Large wood
Pools
Habitat
Movement
Radio telemetry
'ABSTRACT'
To improve understanding of the significance of large woody debris to stream fishes, we examined the influence of woody debris on fall and winter movement by adult coastal cutthroa*t trout (Oncorhynch*us clarki) using radiotelemetry. Fish captured in stream pools containing large woody debris moved less than fish captured in pools lacking large woody debris or other cover. Fish from pools lacking cover commonly moved to habitats with large boulders or brush, particularly during the day. Movements by fish over 1-day periods were strongly

influenced by large woody debris or other elements providing cover. Fish initially found in habitats lacking large woody debris, large boulders, or brush cover moved the most extensively, while fish initially found in pools with large woody debris moved the least. Fish did not move extensively in response to a bankfull flood, although some moved to habitat downstream of large woody debris in tributaries or secondary channels. Habitat downstream of woody debris in the main channel was not used during the flood, apparently because of extreme turbulence. Overall, these observations provide additional evidence for the value of habitat complexity to some stream fishes and support previous observations of minimal effects of flooding on adult fish.

Hauer, F., G. Poole, J. Gangemi, and C. Baxter. 1999. Large woody debris in bull trout (Salvelinus confluentus) spawning streams of logged and wilderness watersheds in northwest Montana. Canadian Journal of Fisheries and Aquatic Sciences 56:915-924.

'KEY WORDS'
Main Stream
Montana
Large Wood
Habitat
Debris flow
Spawning
Forest mamagement
Transport processes
Salvelinus confluentus
bull trout

'ABSTRACT'

Large woody debris (LWD) was measured in 20 known bull trout (Salvelinus confluentus) spawning stream reaches from logged and wilderness watersheds in northwestern Montana. Mean bankfill width of stream reaches was 14.1 m ranging from 3.9 to 36.7 m. Streams were large enough to move LWD and form aggregates. The characteristics were determined of individual pieces of LWD that were interactive with the stream channel. Large, short pieces of LWD attached to the stream bank were the most likely to be positioned perpendicular to stream flow, while large, long pieces either tended to be parallel to the flow or, when attached, were most apt to extend across the channel. Observations indicated that the majority of pools were formed as scour pools by either very large LWD pieces that were perpendicular to the stream or multipiece LWD aggregates. Among reaches in wilderness watersheds, ratios of large to small LWD, attached to unattached LWD, and with and without rootwads were relatively consistent. Among reaches with logging in the watershed, these ratios varied substantially. Results suggest that logging can alter the balance of delivery, storage, and transport of LWD in northern Rocky Mountain streams, and therefore, the likely substantive change in stream habitats.

Hayes, S. A., and coauthors. 2008. Steelhead Growth in a Small Central California Watershed: Upstream and Estuarine Rearing Patterns. Transactions of the American Fisheries Society 137(1):114-128.

'KEY WORDS'
Esturaries
Main Stream
California
Steelhead
Growth

'ABSTRACT'

We monitored growth and life history pathways of juvenile steelhead Oncorhynchus mykiss and compared growth rates between the upper watershed and estuary in Scott Creek, a typical California coastal stream. Growth in the upper watershed was approximately linear from May to December for age-0 fish. For passive integrated transponder (PIT) tagged, age-1+ fish,

growth transitioned to a cyclic pattern, peaking at 0.2% per day during February–April, when maximum flows and temperatures of 7–12°C occurred. Growth of PIT-tagged fish then slowed during August–September (0.01% per day), when temperatures were 14–18°C and flows were low. During each spring, smolts (mean fork length [FL] ± SE = 98.0 ± 1.2 mm) and fry migrated to the estuary; some fish remained there during summer–fall as low flows and waves resulted in seasonal sandbar formation, which created a warm lagoon and restricted access to the ocean. Growth in the estuary–lagoon was much higher (0.2–0.8% per day at 15–24°C). Our data suggest the existence of three juvenile life history pathways: upper-watershed rearing, estuary–lagoon rearing, and combined upper-watershed and estuary–lagoon rearing. We present a model based upon the above data that reports size at age for each juvenile life history type. The majority of fish reaching typical steelhead ocean entry sizes (150–250 mm FL; age 0.8–3.0) were estuary–lagoon reared, which indicates a disproportionate contribution of this habitat type to survival of Scott Creek steelhead. In contrast, steelhead from higher latitudes rear in tributaries during summer, taking several years to attain ocean entry size.

Heggenes, J., T. G. Northcote, and A. Peter. 1991a. Seasonal habitat selection and preferences by cutthroat trout ({IOncorhynchus clarki}) in a small, coastal stream. Canadian Journal of Fisheries and Aquatic Sciences 48:1364-1370.

'KEY WORDS'
Tributaries
British Columbia
Cutthroat trout
Water velocity
Habitat use
'ABSTRACT'

Habitat selection by cutthroat trout (Oncorhynchus clarki) larger than 9 cm total length was monitored during winter and summer. The trout had strong preferences for depths >25 cm and areas where instream and overhead cover exceeded 40% of the local surface area. The fish selected a variety of substrate sizes. Stream areas with mean water velocities <20 cm/s were preferred. Compared with previous studies, the trout used low-velocity areas more, and we suggest that this is due to less competitive interaction from other young salmonids. The trout used the larger pools (>20 m2) considerably less during winter than during summer. Otherwise, little seasonal variation in habitat use was found. A composite measure of water depth and cover appeared to be the most important of the measured environmental factors influencing habitat selection in the stream. The larger trout, which were presumably dominant, occupied the deepest pool areas. The trout selected spatial habitats in proportions significantly different from the available habitat, demonstrating strong habitat preferences. It is concluded that observations of habitat occupancy without considering habitat availability may give biased results.

Heggenes, J., T. G. Northcote, and A. Peter. 1991b. Seasonal habitat selection and preferences by cutthroat trout (Oncorhynchus clarki) in a small, coastal stream. Canadian Journal of Fisheries and Aquatic Sciences 48:1364-1370.

'KEY WORDS'
Tributaries
British Columbia
Cutthroat trout
Habitat selection
'ABSTRACT'

Habitat selection by cutthroat trout (Oncorhynchus clarki) larger than 9 cm total length was monitored during winter and summer. The trout had strong preferences for depths >25 cm and areas where instream and overhead cover exceeded 40% of the local surface area. The fish selected a variety of substrate sizes. Stream areas with mean water velocities <20 cm/s were preferred. Compared with previous studies, the trout used low-velocity areas more, and we suggest that this is due to less competitive interaction from other young salmonids. The

trout used the larger pools (>20 m2) considerably less during winter than during summer. Otherwise, little seasonal variation in habitat use was found. A composite measure of water depth and cover appeared to be the most important of the measured environmental factors influencing habitat selection in the stream. The larger trout, which were presumably dominant, occupied the deepest pool areas. The trout selected spatial habitats in proportions significantly different from the available habitat, demonstrating strong habitat preferences. It is concluded that observations of habitat occupancy without considering habitat availability may give biased results.

Heggenes, J., T. G. Northcote, and A. Peter. 1991c. Spatial stability of cutthroat trout (*Oncorhynchus clarki*) in a small, coastal stream. Canadian Journal of Fisheries and Aquatic Sciences 48:757-762.

'KEY WORDS'
Tributaries
British Columbia
Cutthroat trout
Movement
Pools

'ABSTRACT'

Spatial stability and local movement of a cutthroat trout (Oncorhynchus clarki) population were studied from winter to late summer in a small, coastal stream. The majority of the population was static and resided within a home range < 22 m2, while a small fraction of the fish apparently was more mobile. Local movement was very restricted; 32.4% of the individually marked cutthroat were recaptured within 1 m of their original capture and marking site, and 48% remained within 3 m of that site. Only 17.9% of the fish moved more than 50 m. This behaviour was stable during winter, spring, and summer and may be of adaptive significance. Fish occupying pool areas moved considerably less than fish occupying shallow habitats, indicating that pool dwellers were dominant fish.

Heifetz, J., M. L. Murphy, and K. V. Koski. 1986. Effects of logging on winter habitat of juvenile salmonids in Alaskan streams. North American Journal of Fisheries Management 6:52-58.

'KEY WORDS'
Tributaries
Southeast Alaska
Coho salmon
Effects of logging
Overwinter survival

'ABSTRACT'

Effects of logging on preferred winter habitats of juvenile salmonids in southeastern Alaskan streams were assessed by comparing the area of preferred winter habitat in 54 reaches of 18 streams. Three types of streams were sampled at each of six locations: a stream in a mature, undisturbed forest; a stream in a clear-cut area but logged on at least one bank; and a stream in a clear-cut area with strips of forest (buffer strips) along the stream bank. To identify preferred winter habitats, we classified stream areas in 12 of 18 streams into discrete habitat types and compared the density of salmonids within these habitat types with average density of the entire reach. Most wintering coho salmon (Oncorhynchus kisutch), Dolly Varden (Salvelinus malma), and steelhead (Salmo gairdneri) occupied deep pools with cover (i.e., upturned tree roots, accumulations of logs, and cobble substrate). Riffles, glides, and pools without cover were not used. Seventy-three percent of all pools were formed by large organic debris. Reaches in clear-cut areas without buffer strips had significantly less area of pool habitat than old-growth reaches. Buffer strips protected winter habitat of juvenile salmonids by maintaining pool area and cover within pools. In some cases, blowdown from buffer strips added large organic debris to the stream and increased the cover within pools.

Hetrick, N. J., M. A. Brusven, T. C. Bjornn, R. M. Keith, and W. R. Meehan. 1998a. Effects of canopy removal on invertebrates and diet of juvenile coho salmon in a small stream in southeast Alaska. Transactions of the American Fisheries Society 127:876-888.

'KEY WORDS'
Tributaries
Southeast Alaska
Coho salmon
Invertebrates

'ABSTRACT'
We assessed changes in availability and consumption of invertebrates by juvenile coho salmon Oncorhynchus kisutch in a small stream in southeast Alaska where patches of dense second-growth riparian vegetation bordering the stream had been removed. Benthic invertebrate populations were assessed during summer 1988 and 1989 with a Hess sampler. Aerial invertebrates were sampled during summer 1989 with wire-mesh sticky traps hung just above the water surface and with floating clear-plastic pan traps. Invertebrate drift was assessed during summer 1989 with nets placed at the downstream end of closed- and open-canopy stream sections. Diets of age-0 and age-1 coho salmon were sampled by flushing stomach contents of fish collected from closed- and open-canopy stream sections. Abundance and biomass of benthic invertebrates were larger in open- than in closed-canopy stream sections and were primarily dipterans, ephemeropterans, and plecopterans. More insects were caught on sticky traps in open than in closed sections on two of four dates sampled, and composition of the catch was primarily dipterans (74% in both closed- and open-canopy sections). Catch rates of invertebrates in the pan traps were significantly higher in closed than in open sections on 12 July and were greater in open than in closed sections on 11 August. No significant canopy effect was detected with regard to dry weight of insects captured in pan traps. Composition of the pan-trap catches was primarily dipterans in both closed and open sections (65% and 72%). Abundance of invertebrates in the drift was significantly higher in closed sections than in open sections on two of four dates sampled; dry weight of invertebrate drift did not differ significantly between canopy types. Dry weight of stomach contents of age-0 and age-1 coho salmon was greater for fish sampled in closed- than open-canopy sections on one of four dates sampled; no significant canopy effect was detected for the other three dates sampled. Aerial insects were more abundant in drift and in diets of age-0 and age-1 coho salmon in closed- than in open-canopy sections. Diet of age-0 coho salmon in both closed and open sections and diet of age-1 fish in closed sections overlapped significantly with composition of the drift. Diet of age-1 coho salmon in open sections, however, did not overlap significantly with drift, an indication of selectivity in feeding behavior. Based on higher abundance of aerial invertebrates above the water surface and increased standing crop of benthic invertebrates that we observed in open- versus closed-canopy sections of Eleven Creek, it appears that canopy removal has the potential to increase the carrying capacity of juvenile coho salmon in streams where populations are food limited.

Hetrick, N. J., M. A. Brusven, W. R. Meehan, and T. C. Bjornn. 1998b. Changes in solar input,water temperture, periphyton accumulation, and allochthonous input and storage after canopy removal along two small salmon streams in southeast Alaska. Transactions of the American Fisheries Society 127:859-875.

'KEY WORDS'
Tributaries
Southeast Alaska
Coho Salmon
Effects of logging

'ABSTRACT'
Changes in solar radiation, water temperature, periphyton accumulation, and allochthonous inputs and storage were measured after we removed patches of deciduous, second-growth riparian vegetation bordering two small streams in southeast Alaska that produce coho salmon Oncorhynchus kisutch. Solar radiation and leaf litter input were measured at the water

surface at random locations dispersed through six alternating closed- and open-canopy stream sections. Water temperature, periphyton, and stored organic samples were collected near the downstream end of each section. Solar radiation intensity was measured with digital daylight integrators and pyronometers, periphyton biomass and chlorophyll a were measured on red clay tile substrates, allochthonous input was measured with leaf litter baskets, and benthic organic matter was measured with a Hess sampler. Average intensity of solar radiation that reached the water surface of open-canopy sections was significantly higher than in closed-canopy sections of two streams measured during daylight hours in summer 1988 and of one stream measured day and night in summer 1989. Average daily water temperature was similar in the two canopy types in summer 1988, but was higher in open- than in closed-canopy sections in 1989. Accumulation of periphyton biomass was significantly higher in open- than in closed-canopy sections of the two streams studied in the summer of 1988 and of the one stream sampled in 1989. Accrual of periphyton biomass on tiles placed in the stream for 30-d colonization periods during the summer months of 1989 was also significantly higher in the open than closed sections. Accumulation of chlorophyll a was significantly higher in the open- than in closed-canopy sections of the two streams in 1988 but did not differ significantly between canopy types in 1989. Thirty-day accrual of chlorophyll a was greater in open- than in closed-canopy sections of the one stream studied in 1989. Allochthonous input to the streams decreased after canopy removal, but the amount of organic material stored in the substrate did not differ significantly between open- and closed-canopy sections. Weather was predominantly overcast and rainy in summer 1988 and mostly sunny with infrequent rain in 1989. We speculate that advective heat transfer and high stream discharge from frequent rains moderated the effect of canopy removal and increased solar radiation on water temperatures in open-canopy stream sections in 1988. In 1989, solar radiation was a significant factor in regulating water temperature, especially when streamflows were low. Using a model, we predicted that water temperatures would change little in a 160-m open-canopy reach of Eleven Creek during any weather condition when flows were high. With low flows, however, stream temperatures in open sections of Eleven Creek were predicted to exceed the optimum for growth of juvenile coho salmon in about 20 m during clear sunny weather and in about 50 m when cloudy and overcast.

Hilborn, R., T. P. Quinn, D. E. Schindler, and D. E. Rogers. 2003. Biocomplexity and fisheries sustainability. Proceedings of the National Academy of Sciences of the United States of America 100(11):6564-6568.

'KEY WORDS'
Watershed Scale
Biological diversity
Climate change

'ABSTRACT'

A classic example of a sustainable fishery is that targeting sockeye salmon in Bristol Bay, Alaska, where record catches have occurred during the last 20 years. The is an amalgamation of several hundred discrete spawning populations. Structured within lake systems, individual populations display diverse life history characteristics and local adaptations to the variation in spawning and rearing habitats. This biocomplexity has enabled the aggregate of populations to sustain its productivity despite major changes in climatic conditions affecting the freshwater and marine environments during the last century. Different geographic and life history components that were minor producers during one climatic regime have dominated during others, emphasizing that the biocomplexity of fish stocks is critical for maintaining their resilience to environmental change.

Hilderbrand, R. H., and J. L. Kershner. 2000. Movement patterns of stream-resident cuthroat trout in Beaver Creek, Idaho-Utah. Transactions of the American Fisheries Society 129:1160-1170.

'KEY WORDS'
Main Stream
Idaho

Utah
Movement
'ABSTRACT'
We used mark–recapture, radiotelemetry, and two-way traps to determine daily, seasonal, and annual movements of cutthroat trout Oncorhynchus clarki in Beaver Creek, Idaho–Utah. We recaptured 26 of 167 (16%) passive integrated transponder (PIT)-tagged cutthroat trout; 16 of the fish were recaptured less than 300 m from the point of capture 1 year earlier, whereas 10 fish were recaptured a median of 1,407 m (range 331–3,292 m) from their captured point of the previous year. Radio-tagged individuals moved less frequently and shorter distances (median = 0 m) during autumn and winter, more frequently and farther during spring in association with spawning (median = 576 m), and variably and sporadically during summer (median = 55 m). We found substantial local movements during a diel period that would not have been found using a once weekly observation period. Frequency of cutthroat trout movement through two-way traps was greatest in July and early August and had stopped almost entirely by early September. Movement timing and frequency were similar between the traps and the radio-tagged fish. Our results demonstrate the mobility potential of cutthroat trout and the importance of selecting appropriate spatial and temporal scales of observation when studying their ecology.

Hill, A. C., T. S. Bansak, B. K. Ellis, and J. A. Stanford. 2010. Merits and Limits of Ecosystem Protection for Conserving Wild Salmon in a Northern Coastal British Columbia River. Pages np *in* Ecology and Society.
'KEY WORDS'
Watershed scale
Pacific Northwest
Conservation
Ecology
Resilience
Fisheries
Resource management
Salmon
Habitats
'ABSTRACT'
Loss and degradation of freshwater habitat reduces the ability of wild salmon populations to endure other anthropogenic stressors such as climate change, harvest, and interactions with artificially propagated fishes. Preservation of pristine salmon rivers has thus been advocated as a cost-effective way of sustaining wild Pacific salmon populations. We examine the value of freshwater habitat protection in conserving salmon and fostering resilience in the Kitlope watershed in northern coastal British Columbiaâ€"a large (3186 km2) and undeveloped temperate rainforest ecosystem with legislated protected status. In comparison with other pristine Pacific Rim salmon rivers we studied, the Kitlope is characterized by abundant and complex habitats for salmon that should contribute to high resilience. However, biological productivity in this system is constrained by naturally cold, light limited, ultra-oligotrophic growing conditions; and the mean (Â± SD) density of river-rearing salmonids is currently low (0.32 Â± 0.27 fish per square meter; n = 36) compared to our other four study rivers (grand mean = 2.55 Â± 2.98 fish per square meter; n = 224). Existing data and traditional ecological knowledge suggest that current returns of adult salmon to the Kitlope, particularly sockeye, are declining or depressed relative to historic levels. This poor stock statusâ€"presumably owing to unfavorable conditions in the marine environment and ongoing harvest in coastal mixed-stock fisheriesâ€"reduces the salmon-mediated transfer of marine-derived nutrients and energy to the systemâ€™s nutrient-poor aquatic and terrestrial food webs. In fact, Kitlope Lake sediments and riparian tree leaves had marine nitrogen signatures (Î´15N) among the lowest recorded in a salmon ecosystem. The protection of the Kitlope watershed is undoubtedly a conservation success story. However, 'salmon strongholds' of pristine watersheds may not adequately sustain salmon populations and foster social and ecological

resilience without more holistic and risk-averse management that accounts for uncertainty and interactions between ecosystem fertility, harvest, climate dynamics, and food web dynamics in the marine and freshwater environments encompassed by the life cycle of the fish.

Hocking, M. D., and T. E. Reimchen. 2009. Salmon species, density and watershed size predict magnitude of marine enrichment in riparian food webs. Oikos 118(9):1307-1318.

'KEY WORDS'
Watershed scale
British Columbia
Nutrients
Marine derived nutrients
'ABSTRACT'

Resource subsidies across habitat boundaries can structure recipient communities and food webs. In the northern Pacific region, bears Ursus spp. foraging on anadromous salmon Oncorhynchus spp. provide a key link between marine and terrestrial ecosystems, with salmon density, fish size and watershed size as potential predictors of the magnitude of marine subsidy to terrestrial habitats. We use nitrogen and carbon stable isotopes to provide an assessment of the patterns of marine-enrichment in riparian plants (11 species, 4 guilds) and litter invertebrates (4 guilds) sampled from 27 watersheds in coastal British Columbia, Canada. Watersheds occurred in three geographical regions (Vancouver Island, mainland midcoast and Haida Gwaii) and varied in size, and in biomass (kg m−1 of spawning length) and species of salmon (chum O. keta, pink O. gorbuscha and coho O. kisutch). δ15N values in all plant species and invertebrate guilds were positively predicted by total salmon biomass (kg m−1) and negatively predicted by watershed size. We observed replicated parallel slopes among plant species and invertebrate guilds across the gradient in salmon biomass, with differences in means hypothesized to be due to plant fractionation and animal trophic position. As such, we derived a watershed δ15N-index averaged across guilds, and using an information theoretic approach we find that the biomass of chum salmon is a much stronger predictor of the δ15N-index than either pink or coho salmon, or the sum biomass of all species. The top linear model contained chum biomass and watershed size. Chum salmon biomass independently predicted δ15N-index variation in all three regions of British Columbia. Chum salmon are larger than pink or coho and provide an energetic reward for bears that facilitates carcass transfer, tissue selective foraging, and nutrient distribution by insect scavengers. Analyses of biodiversity and habitat data across many watersheds moves towards a long-term goal in fisheries ecology to better integrate ecosystem values in salmon conservation.

Hogan, D. L., and M. Church. 1989. Hydraulic Geometry in Small, Coastal Streams: Progress Toward Quantification of Salmonid Habitat. Canadian Journal of Fisheries and Aquatic Sciences 46:844-852.

'KEY WORDS'
Tributaries
British Columbia
Stream morphology
Salmon habitat
'ABSTRACT'

It is difficult to quantify in-stream physical attributes of salmonid habitats, yet quantification is necessary if conditions are to be compared within or between streams over time or space. This paper presents an objective method based on hydraulic geometry to quantify hydraulic characteristics of fish habitat. Two small streams in coastal British Columbia provide examples. Morphological mapping and streamflow measurements were used to generate the bivariate distributions of mean depth and mean velocity at cross sections within the study reaches at various discharges. The distributions are used to generate measures of potentially useable area within the streams. Survey criteria and numerical adjustments are presented to improve comparability between channels. The streams respond similarly to a change in

discharge. The main hydraulic difference is a decrease in area useful to coho salmon (Oncorhynchus kisutch) for rearing at higher discharges in Bonanza Creek (logged) compared with Hangover Creek (unlogged). The example indicates that quantitative comparisons can be made, at comparable flows, between reaches and between streams, or over time.

Holtby, L. B., T. E. McMahon, and J. C. Scrivener. 1989. Stream temperatures and inter-annual variability in the emigration timing of coho salmon (Oncorhynchus kisutch) smolts and fry and chum salmon(O. keta) fry from Carnation Creek, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 46:1396-1405.

'KEY WORDS'
Main Stream
British Columbia
Carnation Creek
Temperature
Emigration
Coho salmon
Chum salmon
'ABSTRACT'

Variability in average stream temperatures between peak spawning and fry emergence accounted for 82 and 77% of the variance in the median emigration date of fry of chum (Oncorhynchus keta) and coho salmon (O. kisutch) respectively over a 9 to 10-yr period. The modeled relationships were indistinguishable from laboratory models that predicted time to maximum alevin wet weight. Variability in stream temperatures during the spring accounted for 60% of the variability in the median date of coho smolt emigration. As stream temperatures increased, the predicted thermal summations required for emigration were nearly constant for coho salmon fry, increased moderately for chum salmon fry and increased strongly for coho salmon smolts The duration of the emigration period also differed between the groups: 50% of the chum salmon fry emigrated over a 1-wk period compared with a 2- to 3-wk period for coho salmon fry and smolts. We speculate that the emigration timing —temperature relationships and timing of adult spawning represent adaptations for synchronizing emigration with "windows of opportunity" in the ocean or stream. The windows are of different widths and levels of predictability for coho and chum salmon fry and coho salmon smolts.

Holtby, L. B., and J. C. Scrivener. 1989. Observed and simulated effects of climatic variability, clear-cut logging and fishing on the numbers of chum salmon (Oncorhynchus keta) and coho salmon (O. kisutch) returning to Carnation Creek, British Columbia. Pages 62-81. *in* C. D. Levings, L. B. Holtby, and M. A. Henderson, editors. Proceedings of the National Workdhop on effects of habitat alteration on salmon stocks. Canadian Special Publication of Fisheries and Aquatic Sciences 105.

'KEY WORDS'
Main Stream
British Columbia
Effects of Logging
Salmonid habitat
Temperature models
'ABSTRACT'

The population dynamics of coho and chum salmon have been studied at Carnation Creek since 1970 as part of a multi-disciplinary study of the effects of logging on a small salmon stream in a coastal rainforest. The authors have developed models that predict the numbers of chum and coho salmon from correlative relationships between survival and growth at various life stages and (1) climatic, hydrologic and physical variables, (2) indices of those features of the stream habitat that were affected by logging and, (3) exploitation rates in the fishery. The authors suggest that overall variability in the salmon abundance will tend to increase in the wake of land-use activities, particularly when accompanied by high levels of exploitation and adverse environmental conditions.

Hood, W. 2007. Landscape allometry and prediction in estuarine ecology: Linking landform scaling to ecological patterns and processes. Estuaries and Coasts 30(5):895-900.

'KEY WORDS'
Watershed scale
Esturaries
Land forms
landscape
'ABSTRACT'

Spatial variation in landforms and associated physical processes can often be described by allometric scaling relationships, similar to those describing organismal allometry. Because plant and animal distribution, abundance, and behavior are generally affected, if not sometimes controlled, by the physical environment, landform scaling likely causes parallel scaling of ecological patterns and process across the landscape, i.e., landscape allometry. Organismal allometry has a long history, well-established tradition, and well-developed body of theory. Landscape allometry is a newly emerging conceptual framework that offers explanation for ecological patterns and utility for practical issues such as allowing landscape-scale replication of experimental and control treatments, providing landscape-scale predictions of ecological pattern and process, providing design guidelines for landscape management, and providing diagnostic methods for assessing historical anthropogenic effects to landscapes. Organismal and landscape allometry could be used in complementary fashion, and perhaps ultimately integrated, to form a useful theoretical framework for ecology.

Imhof, J. G., J. Fitzgibbon, and W. K. Annable. 1996. A hierarchical evaluation system for characterizing watershed ecosystems for fish habitat. Canadian Journal of Fisheries and Aquatic Sciences 53(Suppl. 1):312-326.

'KEY WORDS'
Watershed scale
Habitat
Classifciation
Watershed management
'ABSTRACT'

Presents a hierarachical classification based on scale habitat unit to landscape.Evaluatesion of watershed system to determine relationships between processes such as watershed geometry and physical features and there use by fish. Employs a framework that encompasses three hierarchies, watershed, reach, and site to model and analyze processes that affect life histories of fish.

Johnson, C. A., and R. J. Naiman. 1987. Boundary dynamics at the aquatic-terrestrial interface: The influence of beaver and geomorphology. Landscape Ecology 1:47-57.

'KEY WORDS'
Off channel
Watershed scale
Beaver
Geomorphology
Landscape ecology
Riparian habitat
Trophic ecology
'ABSTRACT'

Beaver (Castor canadensis) impoundments are used to illustrate the effect of large animals on the boundary dynamics of lsquopatch bodiesrsquo, volumetric landscape units which have surficial boundaries with upper and lower strata, and lateral boundaries with adjacent patches within the same stratum. Patch bodies created by beaver impoundments include the beaver pond, the aerobic soil beneath the pond, and the underlying anaerobic soil. Beaver herbivory in the riparian zone creates an additional patch body concentric to the pond.

Beaver and water are the primary biotic and abiotic vectors mediating fluxes across lateral patch body boundaries; vegetation and microbes are the primary biotic vectors mediating fluxes across surficial patch body boundaries. Basin geomorphology affects the permeability of pond boundaries (i.e., their ability to transmit, energy and materials) by affecting the kinetic energy of water, the surface-to-volume ratio of the impoundment, and the movement of beaver between the pond and the riparian foraging zone. We suggest that: (1) permeability of lateral boundaries to abiotic vectors is a function of kinetic energy; (2) within-patch retention of particulate matter transferred by abiotic vectors across lateral boundaries is maximized by a decrease in kinetic energy; (3) lateral patch boundaries between safe refuge and a resource used by an animal vector are most permeable when they are narrow; and (4) total amount of energy and materials transferred across surficial boundaries is maximized by increasing surface area.

Johnson, S. L., J. D. Rodgers, M. F. Solazzi, and T. E. Nickelson. 2005. Effects of an increase in large wood on abundance and survival of juvenile salmonids (Oncorhynchus spp.) in an Oregon coastal stream Canadian Journal of Fisheries and Aquatic Sciences 62:412–424 'KEY WORDS'
Tributaries
Main Stream
Oregon
Large wood
Population size
Steelhead
Coho
'ABSTRACT'
We examined the effect of an increase in large wood on the summer population size, smolt abundance, and freshwater survival of steelhead (Oncorhynchus mykiss), coastal cutthroat trout (Oncorhynchus clarki clarki), and coho salmon (Oncorhynchus kisutch). We examined these parameters for five brood years prior to the addition of wood and five brood years after in Tenmile Creek, a direct ocean tributary on the Oregon coast. Over the same time frame, a nearby reference stream, Cummins Creek, was also sampled for the same parameters. The input of large wood into Tenmile Creek resulted from a planned habitat restoration project in 1996 and an unplanned addition of wood from a winter storm the same year. Steelhead smolt abundance, steelhead freshwater survival, and coho salmon freshwater survival increased in Tenmile Creek after the input of large wood. Steelhead age-0+ summer populations and steelhead smolt populations increased in the reference stream, although steelhead freshwater survival did not. Coho salmon populations remained nchanged in the reference stream. Our results illustrate the potential shortcomings of the before-aftercontrol-impact study design under field conditions and the potential for is interpreting results had we employed amore modest sampling plan.

Johnston, C. A., and R. J. Naiman. 1990. Aquatic patch creation in relation to beaver population trends. Ecology 71:1617-1621.
'KEY WORDS'
Off channel
Watershed scale
Minnesota
Beaver
Disturbance
Landscape ecology
'ABSTRACT'
The creation of aquatic patches by beaver (Castor canadensis) in the boreal forest of northern Minnesota, USA, was studied to determine how the population dynamics of a disturbance—causing animal are linked to rates of patch formation and growth over a period of population expansion and stabilization. Using six series of aerial photographs taken between 1940 and 1986, we determined the size and growth rates of individual patches, and the numbers, area,

density, and establishment rate of the patch population. The rate of patch formation was much higher during the first two decades of colonization than during the subsequent two decades. The average area of all ponds sites, which included both filled and drained ponds, remained at °ha throughout the time period, but the average area of new ponds decreased significantly over time. Ponds established by 1961 constituted 75% of the total number and 90% of the total pond—site area as of 1986. When pond sites of similar age but different pond cohort (i.e, decade of establishment) were compared, the average area per pond site was always significantly larger for the earlier cohort. Although the rate of pond creation paralleled the increase in number of beaver colonies between 1961 and 1986, the rate of new pond creation prior to 1961 greatly exceeded the increase in number of beaver colonies. We conclude that the rate of patch formation after the first two decades of beaver colonization was constrained by geomorphology, which limited the availability of sites at which a beaver dam could impound a large area of water.

Jones, N. E. 2010. Incorporating lakes within the river discontinuum: longitudinal changes in ecological characteristics in streamlake networks. Canadian Journal of Fisheries and Aquatic Sciences 67:1350-1362.

'KEY WORDS'
Watershed scale
Lakes
Aquatic networks
Landscape ecology
'ABSTRACT'

Lakes and rivers are intimately connected in an alternating series of lentic and lotic reaches in many regions. The study of lakes and their outlets in hierarchical and branching river networks has not gained the attention of stream ecologists, and little effort has been focused on synthesizing the ecology of lake-stream interactions within a drainage network. Rapid and predictable changes in the ecological characteristics of streams occur at the interface with lakes. The influence that a lake might have on a stream is dependent on its position within the stream, stream type and size, lake size and shape, and the inlet and outlet positions. Little is known about the influences of multiple lakes within stream-lake networks and how these influences are determined by network shape and pattern. Fruitful collaborations and novel insights will come from the combined efforts of limnologists, stream ecologists, and landscape ecologists. Geographic information systems and network analyses will play an important role in summarizing aquatic landscape characteristics and creating a predictive science of aquatic networks. Lakes need to be more explicitly incorporated into ecological concepts in stream ecology, and reciprocally, streams need to be incorporated into ecological concepts involving lakes for the successful management and conservation of our aquatic resources

Junk, W. G., P. B. Bayley, and R. E. Sparks. 1989. The flood pulse concept in river-flooding systems. Pages 110-127 *in* D. P. Dodge, editor Proceedings of the International Large River Symposium. Canadian Special Publications in Fisheries and AquaticSciences.

'KEY WORDS'
Watershed scale
Floodplain
Floods
Stream ecology
'ABSTRACT'

Time and duration of floods are an important effect on river and flood plain productivity. Low order stream pylses tend to be brief and unpredictable, but may be important to transport of nutrients and organisn across the flood plain. A predictble, long duration pulse allows organism to adapt and use the flood pulse for activities such as feeding and reproduction.

Kahler, T. H., P. Roni, and T. P. Quinn. 2001. Summer movement and growth of juvenile anadromous salmonids in small western Washington streams. Canadian Journal of Fisheries and Aquatic Sciences 58:1947-1956.

'KEY WORDS'
Main Stream
Tributaries
Washington
Movement
Juvenile salmonids
Coho salmon
Cutthroat trout
Steelhead

'ABSTRACT'

Movements of juvenile coho salmon (*Oncorhynchus kisutch*), cutthroat trout (*Oncorhynchus clarki clarki*), and steelhead trout (*Oncorhynchus mykiss*) were studied by observations and recapture of marked individuals in three western Washington streams to test the hypotheses that few fish would move, downstream movement would predominate, movers would be initially smaller and grow slower after movement than residents, and habitat quality would influence movement. Contrary to predictions, from 28 to 60% of marked fish moved at least one habitat unit, and immigration of unmarked fish also indicated considerable movement. Upstream movement predominated but the stream with the step-pool/cascade channel type had fewer upstream movers and greater distances moved downstream. Coho movers were not smaller than nonmovers, as predicted based on assumptions that movement results from competitive exclusion. Habitat units that coho left were smaller and shallower but lower in density than units where coho remained. Thus, movement is a common phenomenon rather than an aberration, and may reflect habitat choice rather than territorial eviction. Moreover, movers grew faster than nonmovers, so the "mobile fraction" of the population was not composed of competitively inferior fish but rather individuals that thrived. The phenomenon of small-scale habitat- and growth-related movements should be considered when planning and interpreting studies of juvenile salmonid ecology in streams.

Keith, R. M., T. C. Bjornn, W. R. Meehan, N. J. Hetrick, and M. A. Brusven. 1998. Response of juvenile salmonids to riparian and instream cover modifcations in small streams flowing through second-growth forests of southeast Alaska. Transactions of the American Fisheries Society 127:889-907.

'KEY WORDS'
Tributaries
Southeast Alaska
Coho Salmon
Dolly Varden
Riparian Habitat

'ABSTRACT'

We manipulated the canopy of second-growth red alder Alnus rubra and instream cover to assess the effects on abundance of juvenile salmonids in small streams of Prince of Wales Island, southeast Alaska, in 1988 and 1989. Sections of red alder canopy were removed to compare responses of salmonids to open- and closed-canopy sections. At the start of the study, all potential instream cover was removed from the study pools. Alder brush bundles were then placed in half the pools to test the response of juvenile salmonids to the addition of instream cover. Abundance of age-0 coho salmon Oncorhynchus kisutch decreased in both open- and closed-canopy sections during both summers, but abundance decreased at a higher rate in closed-canopy sections. More age-0 Dolly Varden Salvelinus malma were found in open-canopy sections than in closed-canopy during both summers. Numbers of age-1 and older coho salmon and Dolly Varden were relatively constant during both summers, and there was no significant difference in abundance detected between open- and closed-canopy sections. Abundance of age-0 coho salmon decreased in pools with and without additional instream

cover during both summers. Abundance of age-1 and older coho salmon and age-0 Dolly Varden did not differ significantly in pools with or without added cover during either summer. Abundance of age-1 and older Dolly Varden was higher in pools with added instream cover than in pools without cover during both summers. Age-0 coho salmon decreased in abundance throughout the summer in both years. Emigration was measured in 1989 and accounted for most of the decrease in abundance. Age-0 coho salmon emigrants were significantly smaller than age-0 coho salmon that remained in the stream.

Kiffney, P. M., and coauthors. 2009. Changes in fish communities following recolonization of the Cedar River, WA, USA by Pacific salmon after 103 years of local extirpation. River Research and Applications 25(4):438-452.

'KEY WORDS'
Main Stream
Washington
Barriers
Habitat fragmentation
Migration corridors
Salmon

'ABSTRACT'

Migration barriers are a major reason for species loss and population decline of freshwater organisms. Significant efforts have been made to remove or provide passage around these barriers; however, our understanding of the ecological effects of these efforts is minimal. Installation of a fish passage facility at the Landsburg Dam, WA, USA provided migratory fish access to habitat from which they had been excluded for over 100 years. Relying on voluntary recruitment, we examined the effectiveness of this facility in restoring coho (Oncorhynchus kisutch) salmon populations above the diversion, and whether reintroduction of native anadromous species affected the distribution and abundance of resident trout (O. mykiss and O. clarki). Before the ladder, late summer total salmonid (trout only) density increased with distance from the dam. This pattern was reversed after the ladder was opened, as total salmonid density (salmon + trout) approximately doubled in the three reaches closest to the dam. These changes were primarily due to the addition of coho, but small trout density also increased in lower reaches and decreased in upper reaches. A nearby source population, dispersal by adults and juveniles, low density of resident trout and high quality habitat above the barrier likely promoted rapid colonization of targeted species. Our results suggest that barrier removal creates an opportunity for migratory species to re-establish populations leading to range expansion and potentially to increased population size.

Lawson, P. W., E. A. Logerwell, N. J. Mantua, R. C. Francis, and V. N. Agostini. 2004. Environmental factors influencing freshwater survival and smolt production in Pacific Northwest coho salmon (Oncorhynchus kisutch). Canadian Journal of Fisheries and Aquatic Sciences 61:360-373.

'KEY WORDS'
Main Stream
Washington
Coho salmon
Production
Survival
Climate change
Stream flow
Temperature

'ABSTRACT'

Climate variability is well known to affect the marine survival of coho salmon (Oncorhynchus kisutch) in Oregon and Washington. Marine factors have been used to explain up to 83% of the variability in Oregon coastal natural coho salmon recruitment, yet about half the variability in coho salmon recruitment comes from the freshwater life phase of the life cycle. This seeming

paradox could be resolved if freshwater variability were linked to climate and climate factors influencing marine survival were correlated with those affecting freshwater survival. Effects of climate on broad-scale fluctuations in freshwater survival or production are not well known. We examined the influence of seasonal stream flows and air temperature on freshwater survival and production of two stock units: Oregon coastal natural coho salmon and Queets River coho salmon from the Washington Coast. Annual air temperatures and second winter flows correlated strongly with smolt production from both stock units. Additional correlates for the Oregon Coast stocks were the date of first fall freshets and flow during smolt outmigration. Air temperature is correlated with sea surface temperature and timing of the spring transition so that good freshwater conditions are typically associated with good marine conditions.

Leidbold-brunner. 1992. Beaver dam location and effects on the distribution and abundance of coho salmon fry in an Oregon stream. Northwest Science 66:218-223.
'KEY WORDS'
Off channel
Oregon
Coho Salmon
'ABSTRACT'
Beaver (Castor canadensis) dams and coho salmon (Oncorhynchus kisutch) fry were examined for their relationships in two coastal Oregon streams in 1987. Our initial spring survey of 19 km of stream found only one dam still complete after winter. By autumn, the number of dams had increased to 1.1 and 1.2 per km on the two streams. Beaver dams increased summer pool habitat 7 to 14% over unmodified conditions. Although density of coho (per m super(2) and m super(3)) was similar among pool types, beaver ponds were larger and contained more coho fry than non-beaver pools; thus, beaver increased rearing habitat for coho during the late summer low flow. Beaver represent a low-cost tool deserving more consideration for stream rehabilitation projects.

Leisca, P., and F. W. Allendorf. 1995. When are peripheral populations valuable for conservation. Conservation Biology 9(4):753-760.
'KEY WORDS'
Watershed scale
Conservation
Genetic diversity
Natural selection
Population biology
'ABSTRACT'
A great deal of effort is spent protecting geographically peripheral populations of widespread species. We consider under what conditions it is appropriate to expend resources to protect these populations. The conservation value of peripheral populations depends upon their genetic divergence from other conspecific populations. Peripheral populations are expected to diverge from central populations as a result of the interwoven effects of isolation, genetic drift, and natural selection. Available empirical evidence suggests that peripheral populations are often genetically and morphologically divergent from central populations. The long-term conservation of species is likely to depend upon the protection of genetically distinct populations. In addition, peripheral populations are potentially important sites of future speciation events. Under some circumstances, conservation of peripheral populations may be beneficial to the protection of the evolutionary process and the environmental systems that are likely to generate future evolutionary diversity.

Lisle, T. E. 1986. Effects of woody debris on anadromous salmonid habitat, Prince of Wales Island, Southeast Alaska. North American Journal of Fisheries Management 6:538-550.
'KEY WORDS'
Tributaries

Southeast Alaska
Riparian habitat
Large wood
Stream morphology
'ABSTRACT'

The effects of woody debris on anadromous salmonid habitat in eight streams on Prince of Wales Island, southeast Alaska, were investigated by comparing low-gradient (1-9%) first- or second-order streams flowing through either spruce-hemlock forests or 6-10-year-old clear-cuts, and by observing changes after debris was selectively removed from clear-cut reaches. Woody debris decreased the rate of shallowing as discharge decreased, thus helping to preserve living space for fish during critical low-flow periods. Debris dams were more frequent in clear-cut streams (14.9/100 m), which contained more debris, than in forested streams (4.2/100 m). As a result, total residual pool length (length when pools are filled with water but there is no flow) and length of channel with residual depth greater than 14 cm--the depth range occupied by 84% of coho salmon (Oncorhynchus kisutch)--were greater in clear-cut streams than in forested streams. Greater volumes of woody debris in clear-cut streams produced greater storage of fine sediment (<4-mm diameter) unless the stream gradient was sufficiently high to flush sediment from storage. One-half of the debris dams broke up or were newly formed over a 3-year period, which suggests that they usually released sediment and woody debris before the pools they formed were filled with sediment. Woody debris removal decreased debris-covered area, debris dam frequency, and hydraulic friction in some cases but, in others, these variables were unaffected or recovered within 2 years after erosion and adjustment of the streambed. No consistent differences in pool dimensions were found between treated and untreated clear-cut reaches. Comparisons of habitat in forested and clear-cut streams suggested that removing debris from clear-cut streams reduced salmonid carrying capacity. Retention and natural reformation of debris dams in cleared reaches prevented the expected deterioration of habitat. However, the removal and destabilization of existing woody debris may cause depletion of debris before riparian trees can regrow and furnish new material to the clear-cut streams.

Lohr, S. C., and M. D. Bryant. 1999. Biological Characteristics and Population status of steelhead (Oncorhyncus mykiss) in southeast Alaska. General Technical Report PNW-GTR-407. U.S.D.A. Forest Service, Pacific Northwest Research Station, Portland, OR.
'KEY WORDS'
Watershed scale
Southeast Alaska
Anadromous salmonids
Life history
Steelhead trout
Stock dynamics
'ABSTRACT'

Reveiws life history and population characteristics of steelhead to identify unique or endangered stocks. Provides run timing and escapement where known.

Lotspeich, F. B. 1980. Watersheds as the basic ecosystem: this conceptual framework provides a basis for a natural classification system. Journal of the American Water Resources Association 16(4):581-586.
'KEY WORDS'
Watershed Scale
Climate
'ABSTRACT'

A scheme is outlined to classify watersheds as ecosystems, based on their natural attributes. Two physical factors of the environment, climate and geology, are selected as state factors. Climate is the master factor that supplies energy and water to all ecosystems; geologic structure supplies the materal from which the forces of climate carve landforms to establish

ecosystems. At the next lower level, soil and vegetation interact in a succession of transactions to produce a mosaic of tesseras within each watershed. It is these interacting tesseras that moderate climate and store energy within the ecosystem that influences the embedded stream. At the bottom of the scale is the stream with its passive role and inability to interact with the higher factors of the ecosystem. Thus, we have a controlling force consisting of two elements (climate and geology), a reacting force (soil and vegetation) that responds by circular conditioning to controlling forces, and at the lowest level, the stream which responds to all factors of the living system within its watershed.

Mantua, N. J. 2009 Patterns of change in climate and Pacific salmon production. American Fisheries Society Symposium 70.

'KEY WORDS'
Climate change
Pacific salmon
Pacific Decadal Ossillation
'ABSTRACT'

For much of the 20th century a clear north-south inverse production pattern for Pacific salmon had a time dynamic that closely followed that of the Pacific Decadal Oscillation (PDO), which is the dominant pattern of North Pacific sea surface temperature variability. Total Alaska salmon production was high during warm regimes of the PDO, and total Alaska salmon production was relatively low during cool regimes of the PDO. Leading hypotheses for the link between climate and Pacific salmon production have focused on changes in early ocean survival for juvenile salmon, but it is clear that climate also affects freshwater life stages and influences productivity. Over broad spatial scales, the PDO-related patterns in climate and Pacific salmon production were less prominent in the period 1990-2004 than in earlier decades of the 20th century, yet the regional associations between salmon production and temperatures were generally the same: warm periods coincided with high salmon production in Alaska, and cool periods off the west coast of the continental U.S. and British Columbia coincided with high salmon production in those regions. A case study of Norton Sound pink salmon provides one regional perspective on the links between changes in climate and salmon production. In the period 1962-1995, anomalously warm winter and spring climate in western Alaska and warm spring/summer temperatures in the eastern Bering Sea generally coincided with high pink salmon production. However, especially warm conditions for freshwater and early marine life stages for Norton Sound pink salmon coincided with both high and low levels of recruits per spawner for brood years 1996-2003, a period that experienced large inter-annual variations in Bering Sea, North Pacific, and tropical Pacific Ocean conditions.

Martin, D. J., and L. E. Benda. 2001. Patterns of instream wood recruitment and transport at the watershed scale. Transactions of the American Fisheries Society 130:940-958.

'KEY WORDS'
Main Stream
Southeast Alaska
Large woody debris
Channel morphology
'ABSTRACT'

A wood budget was constructed for the Game Creek basin (132 km2) in southeast Alaska to identify spatial and temporal controls on the abundance and distribution of large woody debris (LWD). Field measurements of wood storage, size, and age were used to estimate volumetric rates of LWD recruitment and transport. Mortality recruitment did not follow a spatial pattern and ranged from 0.1 to 8.1 m3·km−1·year−1 (recruitment corresponded to forest mortality rates of 0.1–2.6% per year). Wood recruitment by bank erosion increased with increasing drainage area and ranged from 1 m3·km−1·year−1 at the smallest drainage areas to about 16 m3·km−1·year−1 at 60 km2. Bank erosion recruitment exceeded the maximum mortality recruitment at a drainage area of approximately 20 km2 (about 10-m-wide channel). Recruitment from land-sliding was only locally significant. The contribution of

fluvial transport (flux) to total LWD storage increased with drainage area to an asymptotic maximum of 50% at about 50 km2 (about 20-m-wide channel). Mean predicted transport distances for mobile LWD over the lifetime of individual pieces ranged from about 200 m in small, jam-rich streams to about 2,500 m in larger channels with fewer jams. Fluvial transport of LWD increased interjam spacing and jam size and decreased jam age with increasing distance downstream. Constructing LWD budgets at the watershed scale has numerous geomorphic and ecological implications, including identifying spatial controls on the abundance and diversity of aquatic habitats. In addition, information on LWD budgets may be useful for determining how and where to protect LWD sources to streams.

McFarlane, G. A., J. R. King, and R. J. Beamish. 2000. Have there been recent changes in climate? Ask the fish. Progress In Oceanography 47(2-4):147-169.

'KEY WORDS'
Climate change
Climate models
Decadal climate shift

'ABSTRACT'

It is generally accepted that a climate shift occurred about 1977 that affected the dynamics of North Pacific marine ecosystems. Agreement on the possibility of further climate shifts in 1989 and the late 1990s is yet to be achieved. However, there have been changes in the dynamics of key commercial fishes that indicate changes in their environment occurred in the early 1990s, and possibly around 1998. One method of measuring climate change is to observe the dynamics of species that could be affected. Several studies have described decadal-scale changes in North Pacific climate-ocean conditions. Generally, these studies focus on a single index. Using principal components analysis, we use a composite index based on three aspects of climate ocean conditions: the Aleutian Low Pressure Index, the Pacific Atmospheric Circulation Index and the Pacific Interdecadal Oscillation Index. We link this composite index (Atmospheric Forcing Index) to decadal-scale changes in British Columbia salmon and other fish populations. Around 1989 there was a change from intense Aleutian Lows (above average south-westerly and westerly circulation patterns and warming of coastal sea surface temperatures) to average Aleutian Lows (less frequent south-westerly and westerly circulation and slightly cooler coastal sea surface temperatures in winter). These climate-ocean changes were associated with changes in the abundance and ocean survival of salmon (Oncorhynchus spp.), distribution and spawning behaviour of hake (Merluccius productus) and sardines (Sardinops sagax) and in recruitment patterns of several groundfish species.

McHenry, M. L., E. Shott, R. H. Conrad, and G. B. Glenn B. Grette. 1998. Changes in the quantity and characteristics of large woody debris in streams of the Olympic Peninsula, Washington, U.S.A. (1982.1993). Canadian Journal of Fisheries and Aquatic Sciences 55:1395-1407.

'KEY WORDS'
Main stream
Washington
Large wood
Stream habitat

'ABSTRACT'

We assessed the changes in large woody debris (LWD) abundance and composition at 28 sites in 27 low-gradient Olympic Peninsula streams between 1982 and 1993. The average number of pieces of debris was virtually identical (P = 0.98) in both years (50.7 versus 50.6). However, we found a significant (P< 0.01) reduction in the total volume of LWD material in the stream sites surveyed (51.7 m3×100 m 1 in 1982 to 38.2 m3×100 m 1 in 1993). While the mean volume of second-growth derived LWD increased from 3.6 to 10.9 m3×100 m 1 (P < 0.01), the increase was insufficient to offset the loss of old-growth derived LWD. The mean volume of old-growth derived LWD for all sites decreased from 48.1 to 27.4 m3×100 m 1 between sample years (P < 0.01). The mean diameter of second-growth derived LWD was significantly larger in 1993 than in 1982, although still smaller than old-growth derived pieces. We

measured a significant increase in the percentage of LWD pieces rated as highly decayed from 1982 to 1993. The results indicate that the loss of old-growth derived LWD following the removal of old-growth riparian forests is initially very rapid, followed by a slower rate of depletion associated with watershed destabilization. Inputs of LWD from second-growth riparian forests up to 73 years old were characterized by small diameter, high mobility, and high decay rates.

McMahon, T. E., and L. B. Blair Holtby. 1992. Behaviour, Habitat Use, and Movements of Coho Salmon (Oncorhynchus kisutch) smolts during Seaward Migration. Canadian Journal of Fisheries and Aquatic Sciences 49:1478-1 485.
'KEY WORDS'
Main Stream
Esturaries
British Columbia
Movement
Large wood
'ABSTRACT'
Coho salmon (Oweorhynchus kisutch) smolts formed aggregations in pools with large woody debris during their migration downstream and into the Carnation Creek estuary, British Columbia. Smolts utilized the estuary throughout the smolt run, with periods of high outmigration coinciding with spring tides which brought warmer, more saline water into the estuary. Smolt abundance in the stream and estuary was positively related to debris volume, and 82% of the 1260 srnolts observed during underwater counts occurred within 1 m of debris. Debris volume and srnolt density were significantly lower in clearcut than in buffered stream sections. Our observations support the need to retain and manage large woody debris for smolt habitat in streams and estuaries.

Meacham, C. P., and J. H. Clark. 1994. Pacific Salmon Management--The view from Alaska. Alaska Fishery Bulletin 1:76-80.
'KEY WORDS'
Watershed Scale
Southeast Alaska
Catch
salmon harvest
sustainability
'ABSTRACT'
Presents case for ADF&G management policies and rebuilding and ustaining salmon stocks

Meehan, W. R. 1991. Influences of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Society, Bethesda.
'KEY WORDS'
Watershed Scale
Ecosystem management
Forest management
Habitat management
Salmonid habitat
'ABSTRACT'
A compendeum of papers on land management and salmon throughout the Pacific Northwest, including Alaska and Idaho.

Miller, B. A., and S. Sadro. 2003. Residence Time and Seasonal Movements of Juvenile Coho Salmon in the Ecotone and Lower Estuary of Winchester Creek, South Slough, Oregon. Transactions of the American Fisheries Society 132(3):546-559.
'KEY WORDS'
Estuaries

Off channel
Oregon
Coho salmon
movement
'ABSTRACT'
The juvenile life history of coho salmon Oncorhynchus kisutch in the stream–estuary ecotone of Winchester Creek, South Slough, Oregon, was investigated in 1999–2001. Seines and a rotary screw trap were used to capture fish for dye-marking, and residence time within the ecotone was determined for recaptured marked fish. In the lower estuary, ultrasonic transmitters were used to document residence time and patterns of movement for smolts migrating to the ocean. Nearly half of each brood year moved to the estuary as subyearlings. A portion of age-0 juveniles that moved downstream during spring lived in the ecotone through summer for up to 8 months, then most moved back upstream to overwinter. Fish that moved to the ecotone during fall and winter had mean minimum residence times of 48 d in 1999 and 64 d in 2000. Some of the fish that moved to the ecotone during fall and winter moved into an off-channel beaver pond and resided there for a mean of 49 d. Spring age-1 smolts had a mean minimum residence time in the ecotone of 18 d for both years and used recently restored salt marshes and other off-channel habitats. Smolts implanted with ultrasonic transmitters lived in the lower estuary for an average of 5.8 d, during which their direction of movement corresponded to the direction of tidal flow.

Milner, A. M., and R. G. Bailey. 1989. Salmonid colonization of new streams in Glacier Bay National Park, Alaska. Aquaculture Research 20:179-192.
'KEY WORDS'
Watershed Scale
Southeast Alaska
Coho salmon
Dolly varden
Glacial rivers
Riparian vegetation
Sockeye salmon
Species diversity
Woody debris
'ABSTRACT'
Following the rapid recession of a neoglacial ice sheet within the last 250 years, colonization of recently deglaciated streams by salmonid fishes was investigated in Glacier Bay National Park, south-eastern Alaska. The primary factors governing the establishment, species diversity composition and abundance of salmonids in Glacier Bay streams were water temperature, sediment loading and stream discharge. No salmonids were found in the turbid meltwater streams emerging from retreating ice. Coho, Oncorhynchus kisutch (Walbaum), and sockeye, Oncorhynchus nerka (Walbaum). salmon and Dolly Vardcn, Salvelinus malma (Walbaum), charr were the first salmonids to colonize the youngest clearwater stream. Juvenile Dolly Varden were more abundant than juvenile coho salmon in the most recently formed clearwater stream because of the characteristic absence of pool habital. Densities of juvenile coho salmon were six times greater in a stream with a series of lakes compared with a stream of similar age without lakes. Future advancement of salmonid stocks will probably depend upon the rate and extent of the development of riparian vegetation and inputs of large woody debris from the developing forest to provide further instream cover, habitat variation and channel stabilization.

Minshall, G. W., and coauthors. 1985. Developments in stream ecosystem theory. Canadian Journal of Fisheries and Aquatic Sciences 42:1045-1055.
'KEY WORDS'
Watershed scale
Watershed theory

Aquatic Ecology
'ABSTRACT'
Four significant areas of thought, (1) the holistic approach, (2) the linkage between streams and their terrestrial setting, (3) material cycling in open systems, and (4) biotic interactions and integration of community ecology principles, have provided a basis for the further development of stream ecosystem theory. The River Continuum Concept (RCC) represents a synthesis of these ideas. Suggestions are made for clarifying, expanding, and refining the RCC to encompass broader spatial and temporal scales. Factors important in this regard include climate and geology, tributaries, location-specific lithology and geomorphology, and long-term changes imposed by man. It appears that most riverine ecosystems can be accommodated within this expanded conceptual framework and that the RCC continues to represent a useful paradigm for understanding and comparing the ecology of streams and rivers.

Moir, H. J., and G. B. Pasternack. 2008. Relationships between mesoscale morphological units, stream hydraulics and Chinook salmon (Oncorhynchus tshawytscha) spawning habitat on the Lower Yuba River, California. Geomorphology 100(3-4):527-548.
'KEY WORDS'
Main Stream
Chinook salmon
California
Spawning
Morphological units
Hydraulics
Two-dimensional modeling
Fluvial geomorphology
'ABSTRACT'
An expert-based approach was used to identify 10 morphological unit types within a reach of the gravel bed, regulated Yuba River, California, that is heavily utilized by spawning Chinook salmon (Oncorhynchus tshawytscha). Analysis of these units was carried out using two-dimensional hydrodynamic modeling, field-based geomorphic assessment, and detailed spawning surveying. Differently classified morphological units tended to exhibit discrete hydraulic signatures. In most cases, the Froude number adequately differentiated morphological units, but joint depth-velocity distributions proved the most effective hydraulic classification approach. Spawning activity was statistically differentiated at the mesoscale of the morphological unit. Salmon preferred lateral bar, riffle, and riffle entrance units. These units had moderately high velocity (unit median > 0.45 m s- 1) and low depth (unit median < 0.6 m), but each exhibited a unique joint depth-velocity distribution. A large proportion of redds (79%) were associated with conditions of convective flow acceleration at riffle and riffle entrance locations. In addition to reflecting microhabitat requirements of fish, it was proposed that the hydraulic segregation of preferred from avoided or tolerated morphological units was linked to the mutual association of specific hydraulic conditions with suitable caliber sediment that promotes the provision and maintenance of spawning habitat.

Montgomery, D. 1999. Process domains and the River Continuum. Journal of the American Water Resources Association 35:397-410.
'KEY WORDS'
watershed scale
Topography
Geology
Geomorphology
Catchment Areas
Stream classification
TectonicsWater resources
Watersheds

Mountain streams
'ABSTRACT'
The concept of process domains is proposed as an alternative to the River Continuum Concept for the influence of geomorphic processes on aquatic ecosystems. Broadly defined, the Process Domain Concept is a multi-scale hypothesis that spatial variability in geomorphic processes governs temporal patterns of disturbances that influence ecosystem structure and dynamics. At a coarse scale, regional climate, geology, vegetation, and topography control the suite of geomorphic processes that are distributed over a landscape. Within the broad context so defined, stream channel classification can guide identification of functionally similar portions of a channel network, but the response of otherwise similar reaches can depend upon their geologic and geomorphic context. Within geomorphic province defined by differences in topography, climate history, and tectonic setting, areas with generally similar geology and topography define lithotopo units, which are useful for stratifying different suites of dominant geomorphic processes. Process domains are spatially identifiable areas characterized by distinct suites of geomorphic processes, and the Process Domain Concept implies that channel networks can be divided into discrete regions in which community structure and dynamics respond to distinctly different disturbance regimes. The concepts of process domains and lithotopo units provide both a framework for the application of patch dynamics concepts to complex landscapes and context for addressing the effects of watershed processes on the ecology of mountain drainage basins.

Montgomery, D., G. Grant, and K. Sullivan. 1995a. Watershed analysis as a framework for implementing ecosystem management. Water Resources Bulletin 31:369-386.
'KEY WORDS'
Ecosystem management
Watershed Scale
Environmental protection
Land management
Planning
Watershed management
'ABSTRACT'
Implementing ecosystem approaches to land use decision making and land management requires new methods for linking science and planning. Greater integration is crucial because under ecosystem management sustainable levels of resource use are determined by coupling management objectives to landscape capabilities and capacities. Recent proposals for implementing ecosystem management employ analyses organized at a hierarchy of scales for analysis and planning. Within this hierarchy, watershed analysis provides a framework for delineating the spatial distribution and linkages between physical processes and biological communities in and appropriate physical context: the watershed. Several such methods are currently in use in the western United States, and although there is no universal procedure for either implementing watershed analysis or linking the results to planning, there are a number of essential elements. A series of questions on landscape-level ecological processes, history, condition, and response potential guide watershed analysis. Individual analysis modules are structured around answering these questions through a spatially-distributed, process-based approach. The planning framework linked to watershed analysis uses this information to either manage environmental impacts or to identify desired conditions and develop land management prescriptions to achieve these conditions. Watershed analysis offers a number of distinct advantages over contemporary environmental analyses for designing land management scenarios compatible with balancing environmental and economic objectives.

Montgomery, D. R., J. M. Buffington, R. D. Smith, K. M. Schmidt, and G. Pess. 1995b. Pool spacing in forest channels. Water Resources Research 31:1097-1105.
'KEY WORDS'
Watershed scale

Southeast Alaska
Geomorphology
Large wood
Stream habitat
Woody debris
'ABSTRACT'

Field surveys of stream channels in forested mountain drainage basins in southeast Alaska and Washington reveal that pool spacing depends on large woody debris (LWD) loading and channel type, slope, and width. Mean pool spacing in pool-riffle, plane-bed, and forced pool-riffle channels systematically decreases from greater than 13 channel widths per pool to less than 1 channel width with increasing LWD loading, whereas pool spacing in generally steeper, step-pool channels is independent of LWD loading. Although plane-bed and pool-riffle channels occur at similar low LWD loading, they exhibit typical pool spacings of greater than 9 and 2-4 channels widths, respectively. Forced pool-riffle channels have high LWD loading, typical pool spacing of <2 channel widths, and slopes that overlap the ranges of free-formed pool-riffle and plane-bed channel types. While a forced pool-riffle morphology may mask either of these low-LWD-loading morphologies, channel slope provides an indicator of probable morphologic response to wood loss in forced pool-riffle reaches. At all study sites, less than 40% of the LWD pieces force the formation of a pool. We also find that channel width strongly influences pool spacing in forest streams with similar debris loading and that reaches flowing through previously clear-cut forests have lower LWD loading and hence fewer pools than reaches in pristine forests.

Montgomery , D. R., T. M. Massong , and S. C. S. Hawley 2003. Influence of debris flows and log jams on the location of pools and alluvial channel reaches, Oregon Coast Range. GSA Bulletin (Geological Society of America) 115:78-88.
'KEY WORDS'
Watershed scale
Oregon
Alluvial channels
Debris flows
Log jams
Oregon coast range
Pools
'ABSTRACT'

We investigated the influence of debris-flow deposits and log jams on the location of pools and alluvial channel reaches in three Oregon Coast Range watersheds. Our surveys reveal differences in the type and location of log jams and the associated influences on pool formation and the extent of alluvial channel beds between channels flowing through old-growth and industrial forests. In channels we surveyed, debris-flow deposits formed 3% of log jams in reaches flowing through old-growth forest and 12% and 25%, respectively, in the two industrial forest channels. Pools formed by the direct effects of debris flows accounted for 4%-7% of all pools in reaches surveyed in both old-growth and industrial forest channels. Logs and log jams accounted for about half of the pools formed in old-growth reaches, but just 12%-13% of pools in reaches flowing through industrial forest. The distribution of bedrock and alluvial reaches was influenced by drainage area, channel-reach slope, sediment trapping by log jams, and boulders deposited by debris flows. Although debris-flow deposits can locally create or influence aquatic habitat, our field observations suggest general contrasts between old-growth and industrial forest in both log jam locations and the relative importance of debris-flow processes in the formation of pools and alluvial reaches.

Moore, J. N., S. N. Luoma, and D. Peters. 1991. Downstream effects of mine effluent on an intermontaine Riparian system. Canadian Journal of Fisheries and Aquatic Sciences 48:222-232.
'KEY WORDS'

Watershed scale
Minning effects
Contamination
Metals
'ABSTRACT'
Metal concentrations were determined in benthic biota, fish livers, water, and fine-grained sediment through 215 km of an intermontane river system (Blackfoot River, Montana, USA) affected by headwater inputs of acid-mine effluent. Solute and particulate contaminants decreased rapidly downstream from headwater sources, but some extended through an extensive marsh system. Particulate contaminants penetrated through the marsh system, effectively resulting in food web contamination downstream of the marshes. Metals differed in their bioavailability within and below the marsh system. Cadmium was most consistently accumulated in the food web, and the general order of downstream mobilization of bioavailable metals appears to be Cd, Zn > Cu > As, Ni. Depauperate benthic communities and reduced fish populations occurred coincident with the sediment contamination.

Morley, S. A., P. S. Garcia, T. R. Bennett, and P. Roni. 2005. Juvenile salmonid(Oncorhynchus spp.) use of constructed and natural side channels in Pacific Northwest rivers. Canadian Journal of Fisheries and Aquatic Sciences 62:2811-2821.
'KEY WORDS'
Off channel
Coho salmon
Pacific Northwest
Sloughs
Enhancement
Artificial channels
'ABSTRACT'
Off-channel habitats, critical components in the life histories of Pacific salmonids (Oncorhynchus spp.), have become increasingly rare in human-modified floodplains. The construction of groundwater-fed side channels is one approach that has been used in the Pacific Northwest to recreate off-channel habitats. We evaluated the effectiveness of this technique by comparing 11 constructed side channels with paired reference sites (naturally occurring channels fed by mixed groundwater and surface water) in western Washington. While total salmonid densities were not significantly different between channel types, coho salmon (Oncorhynchus kisutch) densities were higher in constructed channels and trout densities were higher in reference channels during the winter. Constructed channels were deeper than reference channels and warmer in the winter and cooler in the summer but had lower physical habitat diversity, wood density, and canopy coverage. We did not detect significant differences in water chemistry or invertebrate parameters between channel types. Summer coho density was inversely correlated with minimum daily temperature and with total nitrogen and total phosphorous concentrations. Relative to other stream habitats, both constructed and reference channels supported high densities of juvenile coho salmon during the summer and winter.

Mote, P. W., and coauthors. 2003. Preparing for Climatic Change: The Water, Salmon, and Forests of the Pacific Northwest. Climatic Change 61(1):45-88.
'KEY WORDS'
Climate change
climate models
Snowpack
Pacific Decadel Oscillation
PDO
Management
'ABSTRACT'

The impacts of year-to-year and decade-to-decade climatic variations on some of the Pacific Northwest's key natural resources can be quantified to estimate sensitivity to regional climatic changes expected as part of anthropogenic global climatic change. Warmer, drier years, often associated with El Niño events and/or the warm phase of the Pacific Decadal Oscillation, tend to be associated with below-average snowpack, streamflow, and flood risk, below-average salmon survival, below-average forest growth, and above-average risk of forest fire. During the 20th century, the region experienced a warming of 0.8 °C. Using output from eight climate models, we project a further warming of 0.5–2.5 °C (central estimate 1.5 °C) by the 2020s, 1.5–3.2°C (2.3 °C) by the 2040s, and an increase in precipitation except in summer. The foremost impact of a warming climate will be the reduction of regional snowpack, which presently supplies water for ecosystems and human uses during the dry summers. Our understanding of past climate also illustrates the responses of human management systems to climatic stresses, and suggests that a warming of the rate projected would pose significant challenges to the management of natural resources. Resource managers and planners currently have few plans for adapting to or mitigating the ecological and economic effects of climatic change.

Muhar, S., and M. Jungwirth. 1998. Habitat integrity of running waters — assessment criteria and their biological relevance. Hydrobiologia 386(1):195-202.

'KEY WORDS'
Watershed scale
Habitat assessment
Ecological integrity
Habitat integrity
Habitat quality
'ABSTRACT'

Based upon well-known relations between abiotic and biotic components of river systems, habitat assessment can serve as an indirect procedure to evaluate the ecological integrity by demonstrating the degree of man-induced alterations of the physical environment. Detailed habitat analysis has been an integral part of aut- or synecological investigations in aquatic environments. In many countries, however, the assessment of the physical environment on a larger spatial scale has become a regular part of watershed management programmes. A primary methodological aspect of large-scale habitat assessment is the selection and definition of evaluation criteria specifically designed to identify key functions and processes of intact river systems. Therefore, special emphasis has to be given to integrative parameters, that reflect the spatial and temporal dynamics of running waters as well as the interactions between rivers and their wetlands. Four groups of evaluation criteria are discussed within the framework of their relevance for aquatic biocoenoses and their validity to identify severe human impacts on running water ecosystems: (1) discharge regime; (2) morphological character; (3) lateral connectivity and (4) longitudinal corridor.

Murphy, M., J. Heifetz, S. Johnson, K. Koski, and J. Thedinga. 1986. Effects of clear-cut logging with and without buffer strips on juvenile salmonids in Alaskan streams. Canadian Journal of Fisheries and Aquatic Sciences 43:1521-1533.

'KEY WORDS'
Tributaries
Southeast Alaska
Effects of logging
Coho salmon
Karst
Habitat
'ABSTRACT'

To assess short-term effects of logging on juvenile Oncorhynchus kisutch, Salvelinus malma, Salmo gairdneri , and Salmo clarki in southeastern Alaska, the authors compared fish density and habitat in summer and winter in 18 streams in old-growth forest and in clearcuts with and

without buffer strips. Buffered reaches did not consistently differ from old-growth reaches; clear-cut reaches had more periphyton, lower channel stability, and less canopy, pool volume, large woody debris, and undercut banks than old-growth reaches. In summer, if areas had underlying limestone, clear-cut reaches and buffered reaches with open canopy had more periphyton, benthos, and coho salmon fry (age 0) than old-growth reaches. In winter, abundance of parr (age > 0) depended on amount of debris. If debris was left in clear-cut reaches, or added in buffered reaches, coho salmon parr were abundant (10-22/100 m super(2)). If debris had been removed from clear-cut reaches, parr were scarce

Murphy, M. L., J. Heifetz, J. F. Thedinga, S. W. Johnson, and K. V. Koski. 1989. Habitat utilization by juvenile Pacific salmon (Oncorhynchus) in the glacial Taku River, southeast Alaska. Canadian Journal of Fisheries and Aquatic Sciences 46:1677-1685.

'KEY WORDS'
Main Stream
Tributaries
Off channel
Southeast Alaska
Beaver ponds
Chinook
Coho salmon
Sockeye salmon
Taku river

'ABSTRACT'
Habitat utilization by juvenile Pacific salmon (Oncorhynchus) was determined in summer 1986 by sampling 54 sites of nine habitat types: main channels, backwaters, braids, channel edges, and sloughs in the river; and beaver ponds, terrace tributaries, tributary mouths, and upland sloughs on the valley floor. Physical characteristics were measured at all sites, and all habitats except main channels (current too swift for rearing salmon) were seined to determine fish density. Each species of Oncorhynchus was absent from about one-quarter of the seining sites of each habitat type. The lower Taku River provides important summer habitat for juvenile salmon, but many suitable areas were unoccupied possibly because of their distance from spawning areas and poor access for colonizing fish.

Naiman, R. J., R. E. Bilby, and P. A. Bisson. 2000. Riparian Ecology and Management in the Pacific Coastal Rain Forest. BioScience 50(11):996-1011.

'KEY WORDS'
watershed scale
Riparian habitat
Pacific northwest

'ABSTRACT'
Summarize the recent advances of the last decade on the ecology Pacific coastal rainforests and application to management of streams anad watrersheds

Naiman, R. J., P. A. Bisson, R. C. Lee, and M. C. Turner. 1997. Approaches to Management at the Watershed Scale K. Kohn, and J. F. Franklin, editors. Creating a forestry for the 21st century: The science of ecosystem management. Island Press, Washington,DC.

'KEY WORDS'
Watershed scale
Management

'ABSTRACT'
Integrates several diverse issues that affect management decisions, social needs, management policies, and natural sciences. Lists four watershed scale issues affecting management, variability in time and space, invasive species, connectivity, and human influence.

Naiman, R. J., H. DeCamps, J. Pastor, and C. A. Johnston. 1988a. The potential importance of boundaries to fluvial ecosystems. Journal of the North American Benthological Society 7(4):289-396.
'KEY WORDS'
Watershed Scale
Ecotone
Riparian forests
Riparian habitat
Rivers
Stream ecology
'ABSTRACT'
Boundaries separating adjacent resource patches are dynamic components of the aquatic landscape. This article addresses some fundamental questions about boundary structure and function in lotic ecosystems. We give examples of longitudinal and lateral boundaries associated with stream systems, demonstrate the application of chaos theory to understanding the inherent variability of boundary properties, and compare characteristics of boundaries in an arctic-tropical transect. We conclude that studies of resource patches, their boundaries, and the nature of exchange with adjacent patches will improve our perspective of drainage basin dynamics over a range of temporal and spatial scales.

Naiman, R. J., C. A. Johnston, and J. C. Kelley. 1988b. Alteration of North American streams by beaver. BioScience 38:753-762.
'KEY WORDS'
Off channel
Beaver Ponds
Alaska
North America
'ABSTRACT'
The objectives of this article are to briefly summarize the history of beaver in North America, then describe some of the ecosystem-level responses of streams to beaver-induced alterations and, finally, to describe beaver-induced changes in the landscape that take place over broad spatial and temporal scales. This research has been conducted in Quebec, Minnesota, Montana, and Alaska, and it should be representative of northern regions.

Nass, B., K. English, and H. Frith. 1996. Assessment of summer rearing habitat and juvenile coho abundance in the Kwinageese River, B.C., 1992. Canadian Manuscript Report of Fisheries and Aquatic Sciences, Department of Fisheries and Oceans, 2375, Prince Rupert, BC Canada.
'KEY WORDS'
Main Stream
British Columbia
Limiting factors
Rearing
Ponds
Pools
Tributaries
Coho Salmon
Oncorhynchus kisutch
'ABSTRACT'
Habitat use by juvenile coho salmon (Oncorhynchus kisutch) was examined in the Kwinageese River, British Columbia, as part of the 1992-1993 Nisga'a Interim Measures Program (IMP). Foot and snorkel surveys were carried out during Aug and Sep to quantify wetted area and juvenile coho abundance by habitat and cover type. To determine if coho production was limited by available habitats, comparisons of observed total coho abundance and densities (by habitat type) were made with those presented in the literature. Linear densities of coho fry were the highest in small tributaries and pools with cover, followed by runs with cover. Runs and riffles with no cover had the lowest densities. Densities between habitats with cover

and habitats without cover were significantly different in some comparisons. Side channels accounted for the greatest total linear habitat and the highest total abundance of juvenile coho. Pools contributed only 11.7% to total linear habitat but accounted for 16.5% of total abundance. Total estimated coho fry abundance was only 27% of the potential abundance estimated using a coho production model. Comparison of maximum density and biomass estimates for 29 B.C. streams revealed that the Kwinageese maximum for age 2 coho maximum was substantially higher than the maximum for all other streams surveyed. The average value for the 5 sites surveyed on the Kwinangeese River was only 20% of these maximum levels. Factors such as escapement or winter rearing habitat are more likely to be limiting coho production in the Kwinageese River than summer rearing habitat.

Newbold, J. D., J. W. Elwood, R. V. O'Neill, and W. Van Winkle. 1981. Measuring nutrient spiraling in streams. Canadian Journal of Fisheries and Aquatic Sciences 38:860-863.

'KEY WORDS'
Tributaries
Watershed scale
Nutrient cycling
Phosphorus
Nutrient spiralling
Stream ecology

'ABSTRACT'
Nutrient cycling in streams involves some downstream transport before the cycle is completed. Thus, the path traveled by a nutrient atom in passing through the cycle can be visualized as a spiral. As an index of the spiralling process, we introduce spiralling length, defined as the average distance associated with one complete cycle of a nutrient atom. This index provides a measure of the utilization of nutrients relative to the available supply from upstream. Using 32P as a tracer, we estimated a spiralling length of 193 m for phosphorus in a small woodland stream.

Nickelson, T., and P. Lawson. 1998. Population viability of coho salmon, Oncorhynchus kisutch, in Oregon coastal basins: application of a habitat-based life cycle model. Canadian Journal of Fisheries and Aquatic Sciences 55:2383-2392.

'KEY WORDS'
Watershed scale
Oregon
Coho salmon
Habitat
Model

'ABSTRACT'
To assess extinction risk for Oregon coastal coho salmon, Oncorhynchus kisutch, we developed a life cycle model based on habitat quality of individual stream reaches estimated from survey data. Reach-specific smolt output was a function of spawner abundance, demographic stochasticity, genetic effects, and density- and habitat-driven survival rates. After natural mortality and ocean harvest, spawners returned to their natal reaches. Populations in reaches with poor habitat became extinct during periods of low marine survival. With favorable marine survival, high productivity reaches served as sources for recolonization of lower quality reaches through straying of spawners. Consequently, both population size and distribution expanded and contracted through time. Within a reach, populations lost resilience at low numbers when demographic risk factors became more important than density-dependent compensation. Population viability was modeled for three coastal basins having good, moderate, and poor habitat. With constant habitat conditions, extinction risk in 99 years was negligible in basins with good and moderate habitat and 5-10% in the basin with poor habitat. Reductions in habitat quality up to 60% in 99 years resulted in reduced coho salmon populations in all basins and significantly increased extinction risk in the basin with poor habitat.

Nickelson, T. E., J. D. Rodgers, S. L. Johnson, and M. F. Solazzi. 1992. Seasonal changes in habitat use by juvenile coho salmon (*Oncorhynchus kisutch*) in Oregon coastal streams. Canadian Journal of Fisheries and Aquatic Sciences 49:783-789.

'KEY WORDS'
Main channel
Tributaries
Off channel
Seasonal use
Coho salmon
Oregon
Beaver pond

'ABSTRACT'

Habitat use by juvenile coho salmon (Oncorhynchus kisutch) during spring, summer, and winter was examined in Oregon coastal streams. Coho salmon fry were most abundant in backwater pools during spring. During summer, juvenile coho salmon were more abundant in pools of all types than they were in glides or riffles. During winter, juvenile coho salmon were most abundant in alcoves and beaver ponds. Because of the apparent strong preference for alcove and beaver pond habitat during winter and the rarity of that habitat in coastal streams, we concluded that if spawning escapement is adequate, the production of wild coho salmon smolts in most coho salmon spawning streams on the Oregon Coast is probably limited by the availability of adequate winter habitat.

Northcote, T. G., and G. F. Hartman. 2004. Fishes and Forestry: Worldwide watershed interaction. Blackwell Publishing Company, Oxford, UK.

'KEY WORDS'
Watershed scale
Effects of logging
Forest conservation
Watershed management
Riparian management
Rivers
Lakes

'ABSTRACT'

The books presents a series of papers as chapters that cover worldwide issues on logging, fores practices and interaction with fish. Topics include forest ecology stream ecology, estuarine ecology as it relates to fish and the forest. Aspects of fish biology and life history, migration, reproduction, and feeding that are related to forest practices are also covered in separate papers. A review of various forest harvest activities are discussed. Effects of forest practices on rivers and stream, lakes, and estuarine systems are presented. Papers on fish-forestry interactions from a range of major geographic areas from North America, South American, Europe, and Asia are included in several chapters.

Northcote, T. G., and M. C. Healey. 2007. Fundamental Aspects of Estuarine Ecology Relevant to Fish-Forestry Interactions. Blackwell Science Ltd.

'KEY WORDS'
Estuaries
fish-forestry interactions
physical and chemical characteristics
biological characteristics

'ABSTRACT'

Summary 10.1002/9780470995242.ch5.abs This chapter contains sections titled: * Introduction * Physical and chemical characteristics * Biological characteristics * Concluding remarks

Paustian, S. J. 1992. A channel type users guide for the Tongass National Forest, southeast Alaska. The heirarchial system is based on stream channel gradient, incision, size, and other geomorphic

features within stream reaches. . Technical Paper 26. U.S.D.A. Forest Service, Region 10 (Alaska Region), Juneau, AK.
'KEY WORDS'
Watershed Scale
Stream classification
Watershed management
Habitat management
'ABSTRACT'
Presents a process driven stream classification system used throughout southeast Alaska.

Pess, G. R., and coauthors. 2002. Landscape characteristics, land use, and coho salmon (*Oncorhynchus kisutch*) abundance, Snohomish River, Wash., U.S. A. Canadian Journal of Fisheries and Aquatic Sciences 59:613-623.
'KEY WORDS'
Watershed scale
Land management
Coho salmon
Stream gradient
Habitat
Habitat models
Geology
Urbanization
Forest management
Agricultural effects
'ABSTRACT'
We used temporally consistent patterns in the spatial distribution of returning adult coho salmon (Oncorhynchus kisutch) to explore relationships between salmon abundance, landscape characteristics, and land use patterns in the Snohomish River watershed, Wash. The proportion of total adult coho salmon abundance supported by a specific stream reach was consistent among years, even though interannual adult coho salmon abundance varied substantially. Wetland occurrence, local geology, stream gradient, and land use were significantly correlated with adult coho salmon abundance. Median adult coho salmon densities in forest-dominated areas were 1.5–3.5 times the densities in rural, urban, and agricultural areas. Relationships between these habitat characteristics and adult coho salmon abundance were consistent over time. Spatially explicit statistical models that included these habitat variables explained almost half of the variation in the annual distribution of adult coho salmon. Our analysis indicates that such models can be used to identify and prioritize freshwater areas for protection and restoration.

Peterson, N. P. 1982a. Immigration of juvenile coho salmon (Oncorhynchus kisutch) into riverine ponds. Canadian Journal of Fisheries and Aquatic Sciences 39:1308-1310.
'KEY WORDS'
Off channel
Coho Salmon
Oregon
Migration
Ponds
Winter habitat
'ABSTRACT'
Riverine ponds on the Olympic Peninsula provide important winter refuge for juvenile coho salmon (Oncorhynchus kisutch). A total of 9530 juvenile coho migrated into two riverine ponds of the Clearwater River in 1977, principally during fall freshets. Extensive movement by marked fish (as much as 32.6 km downstream) before entering the ponds suggests that a system-wide approach to habitat management is important in maintaining freshwater production potential of large river systems.

Peterson, N. P. 1982b. Population characteristics of juvenile coho salmon (*Oncorhynchus kisutch*) overwintering in riverine ponds. Canadian Journal of Fisheries and Aquatic Sciences 39:1303-1307.

'KEY WORDS'
Off channel
Oregon
Coho salmon
Ponds
Overwinter survival
'ABSTRACT'

Survival and growth from immigration to smolt outmigration differed substantially between pond populations of juvenile coho salmon (Oncorhynchus kisutch). In Pond 1 (the deeper, less-productive pond) overall survival was 78% but average fish weight increased only 49%, whereas in Pond 2 (the shallow, more-productive pond) survival was only 28% but average fish weight increased 94%. Diet of resident coho in the early spring was characterized by chironomid larvae and newly emerged adults in Ponds 1 and 2, respectively. Manipulation of pond morphometry may have potential for enhancing coho stocks.

Poff, N. L., and coauthors. 1997. The Natural Flow Regime. BioScience 47(11):769-784.

'KEY WORDS'
Watershed Scale
River ecology
Floods
Hydrology
Discharge
Biodiversity
'ABSTRACT'

Present a case for preserving natural flow regimes in conservation.

Pringle, C. M. 2001. Hydrologic connectivity and the management of biological reserves: a global perspective. Ecological Applications 11(4):981-998.

'KEY WORDS'
Watershed scale
biological reserves
connectivity
'ABSTRACT'

Increasingly, biological reserves throughout the world are threatened by cumulative alterations in hydrologic connectivity within the greater landscape. Hydrologic connectivity is used here in an ecological sense to refer to water-mediated transfer of matter, energy, and/or organisms within or between elements of the hydrologic cycle. Obvious human influences that alter this property include dams, associated flow regulation, groundwater extraction, and water diversion, all of which can result in a cascade of events in both aquatic and terrestrial ecosystems. Even disturbances well outside the boundaries of reserves can have profound effects on the biological integrity of these "protected" areas. Factors such as nutrient and toxic pollution and the spread of nonnative species are perpetuated by hydrologic connectivity, and their effects can be exacerbated by changes in this property. Hydrological alterations are now affecting reserves through increasingly broad feedback loops, ranging from overdrawn aquifers to atmospheric deposition and global climate change. Such alterations are often beyond the direct control of managers because they lie outside reserve boundaries, and data on hydrologic connection between reserves and surrounding landscapes are scant. The subject of water has also been typically excluded from the literature pertaining to both theoretical and practical aspects of reserve size, isolation, and design. This results, in part, from early management strategies developed when the landscape matrix outside of reserves

was not excessively fragmented, and when awareness of hydrologic connectivity was in its infancy.

The location of a given reserve within a watershed, relative to regional aquifers and wind and precipitation patterns, can play a key role in its response to human disturbance transmitted through the hydrologic cycle. To illustrate this point, I discuss reserves of varying sizes from diverse regions throughout the world. Reserves located in middle and lower watersheds often suffer direct hydrologic alterations that cause severe habitat modification and exacerbate the effects of pollution. In contrast, reserves in upper watersheds may have intact physical habitat and contain important source populations of some native biota, yet hydrologic disturbances in lower watersheds may cause extirpation of migratory species, cascading trophic effects, and genetic isolation. Worldwide, <7% of land area is either strictly or partially protected, and many reserves are in danger of becoming population "sinks" for wildlife if we do not develop a more predictive understanding of how they are affected by hydrologic alterations that originate outside of their boundaries.

Reeves, G. H., K. M. Burnett, and E. V. McGarry. 2003. Sources of large wood in the main stem of a fourth-order watershed in coastal Oregon. Canadian Journal of Forest Research 33:1363-1370.

'KEY WORDS'
Main Stream
Oregon
Large wood
Landslides
Debris flows
Salmon habitat
'ABSTRACT'

We compared the contribution of large wood from different sources and wood distributions among channel zones of influence in a relatively pristine fourth- order watershed in the central Coast Range of Oregon. Wood in the main stem of Cummins Creek was identified as coming from either (i) streamside sources immediately adjacent to the channel or (ii) upslope sources delivered by landslides or debris flows more than 90 m from the channel. About 65% of the number of pieces and 46% of the estimated volume of wood were from upslope sources. Streamside sources contributed about 35% of the number of pieces and 54% of the estimated volume of wood. The estimated mean volume of upslope- derived pieces was about one-third that of streamside-derived pieces. Upslope- derived pieces were located primarily in the middle stream reaches and in the zones of influence that had the most contact with the low-flow channel. Streamside-derived pieces were more evenly distributed among the examined reaches and were predominately in the influence zones that had the least contact with the low-flow channel. Our findings suggest that previous studies that examined only streamside sources of wood have limited applications when designing and evaluating riparian management approaches in landslide-prone areas. The failure to recognize the potential sources of wood from upslope areas is a possible reason for the decline of large wood in streams in the Pacific Northwest.

Reeves, G. H., F. H. Everest, and J. R. Sedell. 1993. Diversity of juvenile anadromous salmonid communities in basins in coastal Oregon, U.S.A. with different levels of timber harvest activity. Transactions of the American Fisheries Society 122(3):309-317.

'KEY WORDS'
Main Stream
Oregon
Juvenile salmonids
Effects of logging
'ABSTRACT'

We examined the relationships of timber harvest, stream habitat complexity, and diversity of juvenile anadromous salmonid assemblages in 14 small- to intermediate-sized basins in coastal Oregon between 1985 and 1989. Diversity (the inverse of a species dominance index) of assemblages in streams in basins with low harvest levels (≤25% of the basin area harvested) was greater than in streams in basins with high harvest levels (>25% of the basin area harvested) (P = 0.02). Assemblages in basins with high levels of harvest were more dominated by a single species than were assemblages in basins with low harvest, Percent of basin harvested was more strongly associated with assemblage diversity (P = 0.07) than were basin area (P = 0.90) or gradient (P = 0.22) when the influence of the other two factors was controlled. Habitat features were compared between three pairs of streams. Streams in basins with low timber harvest had more complex habitat, as manifested by more large pieces of wood per 100 m (P < 0.01). We conclude that a community and basin-level perspective is necessary to fully assess the effects of timber harvest and other human activities on stream fish.

Rice, S. P., P. R. Kiffney, C. Greene, and G. R. Pess. 2008. The Ecological Importance of Tributaries and Confluences. John Wiley & Sons, Ltd.

'KEY WORDS'
Tributaries
Pacific Northwest
Confluences
River ecology
Spawning patterns
Beaver pond utrient levels enhancement

'ABSTRACT'

This chapter contains sections titled: * Introduction * Tributaries, confluences and river ecology * Tributaries, ecosystem functions and river management * Constraints on understanding and progress * A case study * Conclusion * Acknowledgments * References

Robison, E. G., and R. L. Beschta. 1990. Characteristics of coarse woody debris for several coastal streams of southeast Alaska, USA. Canadian Journal of Fisheries and Aquatic Sciences 47:1684-1693.

'KEY WORDS'
Watershed scale
Southeast Alaska
Woody debris

'ABSTRACT'

Coarse woody debris (> 0.2 m in diameter and 1.5 m long) was measured along five undisturbed low-gradient stream reaches; volume, decay class, and horizontal orientation in relation to channel flow of first-, second-, third-, and fourth-order coastal streams were determined. Debris was also classified into four influence zones based on stream hydraulics and fish habitat. Average debris length, diameter, and volume per piece increased with stream size. Eighty percent of debris volume of the first-order and the smaller second-order streams was suspended above or lying outside the bankfull channel, while less than 40% was similarly positioned in the fourth-order stream. Approximately one-third of all debris was oriented perpendicular to stream flow, regardless of stream size. First-, second-, and third-order streams had a higher proportion of recent debris in the channel than the fourth-order stream (19 vs. 8%), most new debris being attributable to a major 1984 windstorm. Tree blowdown had a major influence on debris distribution along the smaller stream reaches. Debris jams and accumulations in the largest stream were formed from floated debris. These characterizations are useful for evaluating the distribution and amount of woody debris associated with land-management activities.

Rosenfeld, J., M. Porter, and E. Parkinson. 2000. Habitat factors affecting the abundance and distribution of juvenile cutthroat trout (Oncorhynchus clarki) and coho salmon (Oncorhynchus kisutch). Canadian Journal of Fisheries and Aquatic Sciences 57:766-774.

'KEY WORDS'
Tributaries
British Columbia
Habitat
Pools
Abundance
Cutthroat trout
Oncorhynchus clarki
Coho salmon

'ABSTRACT'
The distribution, abundance, and habitat associations of juvenile anadromous coastal cutthroat trout (Oncorhynchus clarki) and coho salmon (Oncorhynchus kisutch) were evaluated using survey data from 119 sites in coastal British Columbia. Both cutthroat and coho occurred at their highest densities in very small streams (<5 m channel width), and bankfull channel width was the single best predictor of cutthroat presence (p = 0.0001) and density (R super(2) = 0.55). Within a channel, densities of coho and larger (yearling and older) cutthroat parr were highest in pools, while densities of young-of-the-year cutthroat were significantly lower in pools and highest in shallower habitats. Abundance of larger cutthroat parr and pool habitat were positively correlated with large woody debris (LWD) within a subset of intermediate-gradient gravel-cobble streams, where pools appear to be limiting to larger cutthroat parr abundance. More than 50% of pools were formed by scour associated with LWD in streams ranging from 1.2 to 11 m channel width, and pools formed by LWD scour were on average 10% deeper than pools formed by other mechanisms. Disproportionate use of small streams by cutthroat indicates that protection of small stream habitat is important for long-term conservation of sea-run populations.

Rosenfeld, J. S., and S. Boss. 2001. Fitness consequences of habitat use for juvenile cutthroat trout: energetic costs and benefits in pools and riffles. Canadian Journal of Fisheries and Aquatic Sciences 58:585-593.

'KEY WORDS'
Tributaries
British Columbia
Cutthroat trout
Bioenergetics
Habitat use
growth

'ABSTRACT'
To assess freshwater habitat requirements of juvenile anadromous cutthroat trout, Oncorhynchus clarki, we measured habitat preference and growth rates of young-of-the-year (YOY) and 1- to 2-year-old fish confined to either pools or riffles in Husdon Creek, British Columbia, during 1999. YOY preferred pools to riffles in habitat-preference experiments, despite normally occurring at lower densities in pools. YOY grew in both pools and riffles when experimentally confined to either habitat, but growth rates were higher in pools. Larger juvenile cutthroat trout, on average, grew in pools, but consistently lost weight in riffles, indicating that pools are a habitat preference for YOY but a requirement for larger fish. A bioenergetic cost–benefit analysis (based on swimming costs and energy intake from invertebrate drift) indicates that energetics alone are sufficient to account for avoidance of riffles by larger cutthroat trout, without having to invoke greater predation risk in shallow habitats. Energetics modeling demonstrates that the smaller size and energetic needs of YOY allow exploitation of habitats (e.g., pocket pools in riffles) that are unavailable to larger fish.

Rosenfeld, J. S., and L. Huato. 2003. Relationship between large woody debris characteristics and pool formation in small coastal British Columbia streams. North American Journal of Fisheries Management 23:928-938.

'KEY WORDS'
Tributaries
British Columbia
Large wood
Stream ecology
Pools
Salmonid habitat

'ABSTRACT'
The characteristics and function of large woody debris (LWD) were measured in 41 small (1.2–11.2-m bank-full channel width), fish-bearing streams in coastal British Columbia to determine how total LWD abundance and the features of individual LWD pieces (diameter, length, orientation, and presence of a rootwad) influenced the effectiveness of pool formation. Pool spacing (the number of channel widths between channel-spanning pools) was a decreasing power function of total LWD abundance, but the relationship was relatively weak. Stratification of sites by channel gradient improved the model fit, steeper streams (≥2% gradient) having a significantly lower pool spacing than lower-gradient streams (<2%). The proportion of LWD that formed pools increased from 6% for pieces with a diameter of 15–30 cm to 43% for pieces with a diameter of more than 60 cm. Large woody debris more than 60 cm in diameter formed a higher proportion of pools across all channel widths. A simple, size-structured model of LWD abundance in small streams suggests that loss of LWD larger than 60 cm in diameter will greatly decrease pool frequency across all channel widths but have the greatest impact on large streams. Models that estimate pool frequency based on total LWD abundance irrespective of size distribution may underestimate the impact of riparian management that reduces the number of larger-diameter trees recruiting to the stream channel.

Rosenfeld, J. S., E. Raeburn, P. C. Carrier, and R. Johnson. 2008. Effects of Side Channel Structure on Productivity of Floodplain Habitats for Juvenile Coho Salmon. North American Journal of Fisheries Management 28(4):1108-1119.

'KEY WORDS'
Off channel
Artificial channels
Coho salmon

'ABSTRACT'
Numerous artificial side channels have been constructed in British Columbia and the Pacific Northwest to compensate for habitat loss from floodplain development. We reviewed data from published studies on natural and restored side channel habitats to determine how design features influence productive capacity for juvenile coho salmon Oncorhynchus kisutch. Average density and biomass of coho salmon parr were significantly higher in stream-type side channels (3.4 parr/m2 and 8.01 g/m2, respectively) than in pond-type side channels (0.8 parr/m2 and 2.37 g/m2). Although total parr biomass was three times higher in stream-type side channels, average parr weight was 47% lower, suggesting greater density-dependent limitation of growth from higher recruitment of juveniles to stream-type habitats. Parr abundance declined from late summer to early spring in both side channel types but appeared to decrease more quickly in stream-type side channels, suggesting greater self-thinning in stream-type habitat from mortality or immigration to slower pond or main-stem habitat as fish sought lower velocities for overwintering. Fish density in a single off-channel complex that contained both stream and pond habitats (fish were able to move between habitats) was also higher in stream habitats, although fish were significantly larger in pond habitats than in stream habitats. Parr density in stream-type side channels was constant with increasing channel size, whereas density in pond-type side channels was a decreasing function of side channel area. Smolt production data were more limited and variable, and production was not

significantly different between stream- and pond-type side channels. Smolt density (smolts produced/m2 of channel habitat) was also a decreasing function of total side channel area, indicating that the optimal side channel habitat size (or pond size within a side channel complex) was below 5,000–10,000 m2. Side channels that incorporate a diversity of flowing- and standing-water areas are most likely to provide the variety of habitats (i.e., spawning, summer rearing, and overwintering) required by salmonids to complete their life cycle.

Rot, B. W., R. J. Naiman, and R. E. Bilby. 2000. Stream channel configuration, landform, and riparian forest structure in the Cascade Mountains, Washington. Canadian Journal of Fisheries and Aquatic Sciences 57:699-707.

'KEY WORDS'
Watershed scale
Large wood
Stream ecology
Channel morphology
'ABSTRACT'

The hierarchical relationship of five key elements, valley constraint, riparian landform, riparian plant community, channel type, and channel configuration, are described for 21 sites in mature to old-growth riparian forests of the western Cascades Mountains, Washington, U.S.A. Channel type (bedrock, plane-bed, and forced pool-riffle) was closely related to channel configuration (especially large woody debris (LWD) volume, density, and LWD-formed pools) at the smallest spatial scale and valley constraint at the largest. Valley constraint significantly influenced off-channel habitat (r2 = 0.71) and LWD volume within forced pool-riffle channels (r2 = 0.58). Riparian plant community composition was differentiated by four landform classes: three alluvial landforms based on height above the channel and one based on hillslope. Just above the active channel, floodplain landforms contained more deciduous stems than conifer and greater conifer basal area than deciduous. Conifers dominated other landforms. The diameter of in-channel LWD increased with the age of the riparian forest (r2 = 0.34). In old-growth forests, LWD diameter was equivalent to or greater than the average riparian tree diameter for all sites. In younger forests, the mixed relationship between LWD and riparian tree diameter may reflect a combination of LWD input from the previous old-growth stand and LWD input from the existing stand.

Schindler, D. E., and coauthors. 2008. Climate Change, Ecosystem Impacts, and Management for Pacific Salmon. Fisheries 33(10):502-506.

'KEY WORDS'
Climate change
Management
'ABSTRACT'

As climate change intensifies, there is increasing interest in developing models that reduce uncertainties in projections of global climate and refine these projections to finer spatial scales. Forecasts of climate impacts on ecosystems are far more challenging and their uncertainties even larger because of a limited understanding of physical controls on biological systems. Management and conservation plans that explicitly account for changing climate are rare and even those generally rely on retrospective analyses rather than future scenarios of climatic conditions and associated responses of specific ecosystems. Using past biophysical relationships as a guide to predicting the impacts of future climate change assumes that the observed relationships will remain constant. However, this assumption involves a long chain of uncertainty about future greenhouse gas emissions, climate sensitivity to changes in greenhouse gases, and the ecological consequences of climate change. These uncertainties in forecasting biological responses to changing climate highlight the need for resource management and conservation policies that are robust to unknowns and responsive to change. We suggest how policy might develop despite substantial uncertainties about the future state of salmon ecosystems.

Schmetterling, D. A. 2001. Seasonal Movements of Fluvial Westslope Cutthroat Trout in the Blackfoot River Drainage, Montana. North American Journal of Fisheries Management 21(3):507-520.

'KEY WORDS'

Main stream
Montana
Cuthroat trout
Movement
Radio telemetry

'ABSTRACT'

I studied the seasonal movements and habitat use of fluvial westslope cutthroat trout Oncorhynchus clarki lewisi from 1997 to 1999 in the Blackfoot River drainage in western Montana to help guide restoration efforts and lead to a better understanding of this subspecies. Of 22 radio-tagged fish, 16 migrated during the spawning period (mean length, 371 mm). Ten of the 22 fish were tracked over a 2-year period, and 2 of these fish migrated in both 1997 and 1998. Migrations to tributaries occurred during the rising limb of the hydrograph in both years and lasted for an average of 10 d (range, 1–14 d) in 1998. Migratory fish moved both upriver and downriver to reach spawning tributaries during both years. In 1998 the mean distance traveled to access tributaries was 31 km (range, 3–72 km). Fish staged at the mouths of tributaries for up to 14 d before entering near the peak in the hydrograph. They remained in tributaries for an average of 27 d (range, 4–63 d), the duration varying with size of tributary and flow year. Once in tributaries, fish generally remained within a 200-m reach, but frequently moved within the area. In four tributaries to the Blackfoot River, actively spawning fish were observed in May 1998 as flows subsided after the peak discharge. Neither of the two repeat migrants spawned within 3 km of their previous year's spawning location, though both spawned in the same tributaries. After leaving tributaries, fish moved both up- and downriver to overwintering areas and did not move more than 100 m thereafter. Fish did not exhibit fidelity to their prespawning main-stem locations. At least six fish died after spawning (38%). Westslope cutthroat trout movements, prespawning and postspawning, exhibited a plasticity not previously reported in Montana and demonstrate the large spatial extent to which fluvial westslope cutthroat trout utilize aquatic resources. To enable continued improvement of the westslope cutthroat trout population in the Blackfoot River drainage, I recommend riparian timber management that continues long-term input of large woody debris to tributaries, continued closure of the Blackfoot River watershed to angling harvest, and the use of culvert designs that will pass spawning fish under most flow conditions.

Sedell, J. R., and J. L. Froggatt. 1984. Importance of streamside forests to large river: The isolation of the Willamette River, Oregon, U.S.A., from its floodplain by snagging and streamside forest removal. Verhandlungendern Internationalen Vereinigung Limnologie 22(1828-1834).

'KEY WORDS'

Watershed Scale
Debris removal
Riparian habitat
River continuum
Woody debris

'ABSTRACT'

Historical perspective of the develpment of riparian habitat following debris removal from snagging etc following European settlement

Sharma, R., and R. Hilborn. 2001. Empirical relationships between watershed characteristics and coho salmon (Oncorhynchus kisutch) smolt abundance in 14 western Washington streams. Canadian Journal of Fisheries and Aquatic Sciences 58:1453-1463.

'KEY WORDS'

Watershed Scale
Washington
Coho Salmon

Large wood
Pools
Population number
Smolts
Escapement
'ABSTRACT'

Data were assembled on coho salmon (Oncorhynchus kisutch) from 14 streams in western Washington, including annual smolt counts and annual escapement, either as absolute counts or as an index. Data was also compiled on large woody debris, road densities in the watersheds, gradient of the streams, valley slope adjacent to the stream, drainage area in the watershed, and pool, pond, and lake areas. The relationships between habitat variables and two measures of coho production, the maximum production of smolts in the stream (capacity) and the maximum smolts/spawner (productivity) were explored. Using the 11 streams with pool and pond counts, it was found that pool and pond densities served as good predictors of smolts density (r super(2) = 0.85 for pools and 0.68 for ponds, independently). Pools produced 0.39 smolts times m super(-2) and ponds produced 0.07 smolts times m super(-2) in the multiple regression fit, accounting for 92% of the residual error. It was also found that lower valley slopes, lower road densities, and lower stream gradients were correlated with higher smolt density.

Sheer, M. B., and E. A. Steele. 2006. Lost watersheds: Barriers, aquatic habitat connectivity, and salmon persistence in the Willamette and lower Columbia Basins. Transactions of the American Fisheries Society 135:1654-1669.

'KEY WORDS'
Watershed scale
Oregon
Columbia River
Connectivity
Migration
Barriers
'ABSTRACT'

Large portions of watersheds and streams are lost to anadromous fishes because of anthropogenic barriers to migration. The loss of these streams and rivers has shifted the distribution of accessible habitat, often reducing the diversity of accessible habitat and the quantity of high-quality habitat. We combined existing inventories of barriers to adult fish passage in the Willamette and Lower Columbia River basins and identified 1,491 anthropogenic barriers to fish passage blocking 14,931 km of streams. We quantified and compared the stream quality, land cover, and physical characteristics of lost versus currently accessible habitat by watershed, assessed the effect of barriers on the variability of accessible habitats, and investigated potential impacts of habitat reduction on endangered or threatened salmonid populations. The majority of the study watersheds have lost more than 40% of total fish stream habitat. Overall, 40% of the streams with spawning gradients suitable for steelhead (anadromous rainbow trout Oncorhynchus mykiss), 60% of streams with riparian habitat in good condition, and 30% of streams draining watersheds with all coniferous land cover are no longer accessible to anadromous fish. Across watersheds, hydrologic and topographic watershed characteristics were correlated with barrier location, barrier density, and the impacts of barriers on habitat. Population-based abundance scores for spring Chinook salmon O. tshawytscha were strongly correlated with the magnitude of habitat lost and the number of lowland fish passage barriers. The characteristics of barrier and habitat distribution presented in this paper indicate that barrier removal projects and mitigation for instream barriers should consider both the magnitude and quality of the lost habitat.

Shirvell, C. S. 1990. Role of instream rootwads as juvenile coho salmon (Oncorhynchus kisutch) and steelhead trout (O. mykiss) cover habitat under varying streamflows. Canadian Journal of Fisheries and Aquatic Sciences 47:852-860.

'KEY WORDS'
Main Stream
British Columbia
Coho salmon
Steelhead
Rootwads
'ABSTRACT'

Coho salmon fry (Oncorhynchus kisutch) and steelhead parr (O. mykiss) occupied previously infrequently-used mid-channel areas of Kloiya Creek, British Columbia, Canada, once artificial rootwads were placed there. Ninetynine percent of all coho salmon fry and 83% of steelhead parr occupied positions downstream of natural or artificial rootwads during artificially created drought, normal, and flood streamflows. Fish associated with rootwads regardless of distance from shore, but steelhead parr preferred rootwads away from shore while coho salmon fry preferred rootwads next to shore. Coho salmon fry increased their use of natural rootwads where currents were slow during floods, while steelhead parr increased their use of artificial and natural rootwads where light remained low during droughts. Young fish apparently selected areas having slower water 80% of the time because they provided shelter from adverse current, and areas having reduced light intensities 20% of the time because they provided protection from predators, juvenile coho salmon and steelhead in Kloiya Creek selected locations with slower water velocities and reduced light intensities irrespective of the physical structure that caused them.

Shirvell, C. S. 1994. Effect of changes in streamflow on microhabitat use and movements of sympatric juvenile coho salmon (Oncorhynchus kisutch) and chinook salmon (O. tshawytscha) in a natural stream. Canadian Journal of Fisheries and Aquatic Sciences 51:1644-1652.
'KEY WORDS'
Climate change
Steamflow
Stream habitat
Behavior
Salmonids
Coho salmon
Chinook salmon
'ABSTRACT'

The microhabitats at positions selected by juvenile coho (Oncorhynchus kisutch) and chinook salmon (O. tshawytscha) following a change in streamflow differed from microhabitats occupied at normal streamflows. At drought streamflow (37% mean seasonal streamflow (MSF)), juvenile coho salmon selected slower, darker, and higher sites above the streambed ($P < 0.05$) than sites selected at normal (75% MSF) or flood (159% MSF) flows. Juvenile chinook salmon microhabitat use changed similarly with changes in streamflow, but the differences were not significant. Up to one fifth of the fish chose positions with faster water velocities than those available either 30 cm above or 30 cm lateral to them. These fish chose positions inconsistent with the hypothesis of optimal position selection based on maximizing net energy gain. On average, fish moved 6.8 m following a change in streamflow. Juvenile coho salmon generally moved upstream in response to decreasing streamflows and downstream in response to increasing streamflows. Juvenile chinook salmon tended to move offshore and downstream in response to all streamflow changes. These results show that juvenile coho and chinook salmon will move to find suitable microhabitat following a change in streamflow and that the microhabitats are not the same at all streamflows.

Solazzi, M. F., T. Nickelson, S. Johnson, and J. Rodgers. 2000. Effects of increasing winter rearing habitat on abundance of salmonids in two coastal Oregon streams. Canadian Journal of Fisheries and Aquatic Sciences 57:906-914.
'KEY WORDS'
Tributaries

Oregon
Abundance
Habitat improvement (physical)
Overwinter habitat
Steelhead
Oncorhynchus mykiss
Coho salmon
Oncorhynchus kisutch
Cutthroat trout
Oncorhynchus clarki
'ABSTRACT'
A BACI (before-after-control-impact) experimental design was used to examine the effects of increasing winter habitat on the abundance of downstream migrant salmonids. Two reference streams and two treatment streams were selected in the Alsea and Nestucca basins of Oregon. Population parameters for juvenile coho salmon (Oncorhynchus kisutch), age-0 trout (Oncorhynchus spp.), steelhead (Oncorhynchus mykiss), and coastal cutthroat trout (Oncorhynchus clarki) were estimated each year for 8 years in each stream. Stream habitat was modified to increase the quality and quantity of winter habitat during the summers of 1990 (Nestucca Basin) and1991 (Alsea Basin). Complex habitat was constructed by adding large woody debris to newly created alcoves and dammed pools. Numbers of coho salmon summer juveniles and smolts increased in the treatment streams relative to the control streams during the posttreatment period. Overwinter survival of juvenile coho salmon also increased significantly in both treatment streams posttreatment. Summer trout populations in the treatment streams did not change, but downstream migrant numbers the following spring did increase. These increases suggest that winter habitat was limiting abundance of all three species.

Stanford, J. A., R. F. Hauer, and J. V. Ward. 1988. Serial discontinuity in a large river system. Verhandlungendern Internationalen Vereinigung Limnologie 23:1114-1118.
'KEY WORDS'
Watershed scale
Stream Ecology
'ABSTRACT'
The paper reviews the River Continuum Concept and suggests modifications on the basis of dams in the watershed. They discuss the ability of dam to retain sediment and organic matter which will alter downstream tranport and processing of organic matter.

Stanford, J. A., and J. V. Ward. 1993. An Ecosystem Perspective of Alluvial Rivers: Connectivity and the Hyporheic Corridor Journal of the North American Benthological Society 12:48-60.
'KEY WORDS'
Watershed scale
hyporheic zone
connectivity
'ABSTRACT'
Floodplains of large alluvial rivers are often expansive and characterized by high volume hyporheic flow through lattice-like substrata, probably formed by glacial outwash or lateral migration of the river channel over long time periods. River water downwells into the floodplain at the upstream end; and, depending on bedrock geomorphology and other factors, groundwater from the unconfined aquifer upwells directly into the channel or into floodplain springbrooks at rates determined by head pressure of the water mass moving through the floodplain hydrologic system. These large scale (km³) hyporheic zones contain speciose food webs, including specialized insects with hypogean and epigean life history stages (amphibionts) and obligate groundwater species (stygobionts). Biogeochemical processes in the hyporheic zone may naturally load groundwaters with bioavailable solutes that appear to exert proximal controls on production and biodiversity of surface benthos and riparian

vegetation. The effect is especially evident in floodplain springbrooks. Dynamic convergence of aquifer-riverine components adds physical heterogeneity and functional complexity to floodplain landscapes. Because reaches of aggraded alluvium and attendant ecotonal processes occur serially, like beads on a string, along the river continuum, we propose the concept of a hyporheic corridor in alluvial rivers. We expect predictable zonation of groundwater communities and other aquifer-riverine convergence properties within the corridor from headwaters to river mouth. The landscape-level significance and connectivity of processes along the hyporheic corridor must be better understood if river ecosystems, especially those involving large floodplain components, are to be protected and/or rehabilitated.

Statzner, B., and B. Higler. 1985. Questions and comments on the river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 42:1038-1044.
'KEY WORDS'
Watershed scale
River continum concpet
Aquatic ecology
'ABSTRACT'
The River Continuum Concept (RCC) is a generalized conceptual framework for characterization of pristine running water ecosystems. Of the numerous tenets of the concept we particularly reevaluated the following: biological analogues of energy equilibrium and entropy in the physical system; maximization of energy consumption through continuous species replacement over a year; absence of succession in stream ecosystems, which can thus be viewed in a time-independent fashion; and maximization of biotic diversity in midreaches of streams as a result of the occurrence of highest environmental variability there together with spatial abundance shifts of insects, molluscs, and crustaceans. When emphasis is placed on rapid changes in the downstream hydraulics dependent on discharge and slope (both of which are expressed by stream order in the RCC and are key factors of the concept) and on results from tropical studies, some of these tenets are partly refuted or need extension. Some of them are in conflict with the current state of knowledge in other domains of stream ecology or are at least open to various interpretations. Therefore, we advocate modifications of the theoretical background of the RCC.

Stednick, J. D. 2008. Long-term Streamflow Changes Following Timber Harvesting. Pages 139-155 *in* Hydrological and Biological Responses to Forest Practices, volume 199. Springer New York.
'KEY WORDS'
Tributaries
Effects of logging
Hydological regime
Riparian management
'ABSTRACT'
Studies under virtually every environmental condition indicate that vegetation removal results in increased annual water yield (Rothacher 1970; Harr 1976, 1979, 1983; Bosch and Hewlett 1982; Stednick 1996). However, treatment responses are variable.

Steel, E. A., and coauthors. 2008. A Spatially Explicit Decision Support System for Watershed-Scale Management of Salmon. Ecology and Society 13(2):np.
'KEY WORDS'
Watershed scale
Oregon
Salmon
Watershed management
'ABSTRACT'
Effective management for wide-ranging species must be conducted over vast spatial extents, such as whole watersheds and regions. Managers and decision makers must often consider results of

multiple quantitative and qualitative models in developing these large-scale multispecies management strategies. We present a scenario-based decision support system to evaluate watershed-scale management plans for multiple species of Pacific salmon in the Lewis River watershed in southwestern Washington, USA. We identified six aquatic restoration management strategies either described in the literature or in common use for watershed recovery planning. For each of the six strategies, actions were identified and their effect on the landscape was estimated. In this way, we created six potential future landscapes, each estimating how the watershed might look under one of the management strategies. We controlled for cost across the six modeled strategies by creating simple economic estimates of the cost of each restoration or protection action and fixing the total allowable cost under each strategy. We then applied a suite of evaluation models to estimate watershed function and habitat condition and to predict biological response to those habitat conditions. The concurrent use of many types of models and our spatially explicit approach enables analysis of the trade-offs among various types of habitat improvements and also among improvements in different areas within the watershed. We report predictions of the quantity, quality, and distribution of aquatic habitat as well as predictions for multiple species of species-specific habitat capacity and survival rates that might result from each of the six management strategies. We use our results to develop four on-the-ground watershed management strategies given alternative social constraints and manager profiles. Our approach provides technical guidance in the study watershed by predicting future impacts of potential strategies, guidance on strategy selection in other watersheds where such detailed analyses have not been completed, and a framework for organizing information and modeled predictions to best manage wide-ranging species.

Swain, D., and B. Holtby. 1989. Differences in morphology and behavior between juvenile coho salmon (*Oncorhynchus kisutch*) rearing in a lake and in its tributary stream. Canadian Journal of Fisheries and Aquatic Sciences 46:1406-1414.

'KEY WORDS'
Tributaries
British Columbia
Coho salmon
Lakes
Stream
Competition
'ABSTRACT'

Juvenile coho salmon (Oncorhynchus kisutch) rear both in Mesachie Lake, B.C., and in its inlet stream. The duration and frequency of aggressive behavior were greater among stream-rearing than lake-rearing juveniles in mirror image stimulation and stream tank tests of agonistic behavior. Lateral displays made up a higher proportion of total behavior among the stream-rearing fish compared with the lake-rearing fish. Lake-type fish had more posteriorly placed pectoral fins, shallower bodies and smaller, less brightly colored dorsal and anal fins than did stream-type fish, even after 2 mo of laboratory rearing in a common environment. Diminished aggression, a shift in aggressive behavior away from lateral displays, a more streamlined shape, and reduced coloration, all appear to be adaptations to a schooling lifestyle in the open waters of the lake.

Swales, S., F. Caron, J. R. Irvine, and C. D. Levings. 1988. Overwintering habitats of coho salmon (Oncorhynchus kisutch) and other juvenile salmonids in the Keogh River system, British Columbia. Canadian Journal of Zoology 66:254=261.

'KEY WORDS'
Tributaries
British Columbia
Overwinter habitat
Coho salmon
'ABSTRACT'

Catches of overwintering juvenile coho salmon (Oncorhynchus kisutch) in the Keogh River system, Vancouver Island, were higher in two small (8 and 25 ha), shallow (mean depth 2 - 3 m) lakes and their outlet and inlet streams than in the main river, where steelhead trout (Salmo gairdneri) were predominant. Dolly Varden char (Salvelinus mulmu), cutthroat trout (Salmo clarki), and threespine stickleback (Gasterosteus aculeatus) were also present in the lakes. The distribution of coho salmon in the lakes was restricted largely to areas close to the bank, with few fish being captured in offshore areas or in mid-water. Aapparent differences in the abundance of coho salmon between the two lakes may have been related to differences in fish community composition, with sticklebacks being particularly numerous in Misty Lake, where catches of coho salmon were lower than in Long Lake. The population density and biomass of coho salmon overwintering in Long Lake were estimated to be 176 fishlha and 1.14 kg ha-', respectively. The mean length of coho salmon in the lakes was greater than that of coho salmon in the tributary streams and main river, and the mean length of the salmon in the lakes generally increased with distance away from shore.

Swales, S., R. B. Lauzier, and C. D. Levings. 1986. Winter habitat preferences of juvenile salmonids in two interior rivers in British Columbia. Canadian Journal of Zoology 64:1506-1514.

'KEY WORDS'
Off Channel
Main channel
British columbia
Coho salmon
Off-channel habitat
Salmonid ecology
Salmonid habitat
Steelhead trout
Winter habitat

'ABSTRACT'

The winter distribution and abundance of juvenile salmonids was investigated in various main channel and off-channel habitats in the Coldwater and Nicola rivers in the southern interior of British Columbia. Catches were generally low in all main channel habitats, with coho salmon (Oncorhynchus kisutch) and steelhead trout (Salmo gairdneri) being most abundant and chinook salmon and Dolly Varden char being present in smaller numbers. Coho salmon and steelhead trout catches were generally highest in pools with abundant instream and riparian cover. Steelhead trout was the main species in riprap bank protected areas, although catches were generally low. Highest overall catches were recorded in side channels and off-channel ponds, where water temperatures were usually several degrees higher than in the main river. The authors conclude that juvenile salmonids in the rivers investigated showed considerable habitat segregation during the winter. As in coastal rivers, juvenile coho salmon made extensive use of off-channel ponds, while rainbow trout and chinook salmon were generally most abundant in riprap and deep pools containing log debris, respectively.

Swales, S., and C. D. Levings. 1989. Role of off-channel ponds in the life cycle of coho salmon (Oncorhynchus kisutch) and other juvenile salmonids in the Coldwater River, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 46:232-242.

'KEY WORDS'
Off channel
British Columbia
Coho Salmon
Steelhead

'ABSTRACT'

Off-channel ponds in the upper reaches of the Coldwater River, British Columbia, were major rearing areas for juvenile coho salmon (Oncorhynchus kisutch). Chinook salmon (Oncorhynchus tshawytscha), steelhead trout (Salmo gairdneri), and Dolly Varden char (Salvelinus malma) were generally scarce in the ponds, although they were numerous in the main river. Coho

salmon were predominant at "natural" river sites while steelhead trout was the main species at sites with "rip-rap" bank stabilization. Catches of juvenile coho were much lower in the main river than in the ponds where they were the main species, and were more variable in the river. Population density and biomass estimates of juvenile coho in the ponds ranged from 0.100 fish·m−2 and 1.00 g·m−2 to 1.00 fish·m−2 and 5.15 g·m−2, compared with density estimates of 0.08–0.23 fish·m−2 in the river. The coho population in the ponds consisted of 0+ and 1+ age-groups in similar proportions, while in the main river the 0+ age-group was much more abundant. The growth rate of coho in the ponds was faster than in the main river, with pond fish reaching mean lengths of 62–79 mm at the end of the first growing season, compared with 53 mm in the main river. Smolt outmigration from the main study pond occurred in late spring with peak outmigration in May and June coinciding with peak river discharge and increasing water temperatures in the main river and pond.

Swanston, D. N. 1969. Mass Wasting in coastal Alaska. Institute of Northern Forestry, Pacific Northwest Forest and Range Experiment Station, Portland, OR.
'KEY WORDS'
Watershed scale
Southeast Alaska
Effects of logging
Erosion control
Landslides
Sediment
Water quality
'ABSTRACT'
Present various aspects of landslides in southeast Alaska. Idendifies landslide prone landscapes.

Taylor, S. G. 2008. Climate warming causes phenological shift in Pink Salmon, Oncorhynchus gorbuscha, behavior at Auke Creek, Alaska. Global Change Biology 14(2):229-235.
'KEY WORDS'
Climate change
Southeast Alaska
Migration timing
Oncorhynchus gorbuscha
Pink salmon
Water temperature
'ABSTRACT'
Abstract Thirty-four years (1972–2005) of water temperature data and extensive biological observations at Auke Creek, Alaska indicate a general warming trend that affected the native pink salmon (Oncorhynchus gorbuscha) population. Serial environmental records at nearby Auke Bay, Alaska over 46 years show trends of increasing air and sea surface temperatures. Trends of increased total precipitation and earlier date of ice out on nearby Auke Lake also occurred, but not at significant rates. Average water temperatures during the incubation of pink salmon in Auke Creek increased at a rate of 0.03 °C yr−1 over the 34-year period. For the 1972–2005 broods, midpoints of fry migrations from Auke Creek ranged between April 2 and May 7, and there was a trend of earlier migration of pink salmon fry at a rate of −0.5 days yr−1. The migration timing of adult salmon into Auke Creek also showed a trend toward earlier timing. The earlier adult migration combined with warmer incubation temperatures are related to earlier migration of pink salmon fry. If the observed warming trend continues, Auke Creek may become unsuitable habitat for pink salmon. Given the trend for salmon fry to migrate earlier, a larger portion of the population may become mismatched with optimum environmental conditions during their early marine life history. If salmon adults continue to migrate into the creek earlier when water temperatures are commonly high, it will result in increased prespawning mortality.

Thedinga, J. F., M. L. Murphy, J. Heifetz, K. V. Koski, and S. W. Johnson. 1989. Effects of Logging on Size and Age Composition of Juvenile Coho Salmon (*Oncorhynchyus kisutch*) and Density of Presmolts in Southeast Alaska Streams. Canadian Journal of Fisheries and Aquatic Sciences 46:1383--1391.

'KEY WORDS'
Tributaries
Southeast Alaska
Effects of Logging
Coho salmon
'ABSTRACT'

Short-term effects of logging on age composition and size of juvenile coho salmon (Oncorhynchus kisutch) were studied in 18 streams in Southeast Alaska in 1982 and 1983; studies were in old-growth and clear-cut reaches with or without buffer strips. The number of fry (age 0) in summer and winter was proportionately higher in buffered and clear-cut reaches than in old-growth reaches, and all treatments averaged a 20% decrease in fry from summer to winter. Fry length and condition factor were greater for buffered and clear-cut reaches than for old-growth reaches, whereas parr (age 1 and older) size did not differ among treatments. Fry and parr were larger in the southern than in the northern regions and their length and weight were directly related to periphyton biomass and benthos density.

Tiegs, S. D., and coauthors. 2008. Timber Harvest Transforms Ecological Roles Of Salmon In Southeast Alaska Rain Forest Streams
Ecological Applications 18(1):4-11.

'KEY WORDS'
Watershed scale
Southeast Alaska
salmon
Nutrients,
Stream ecosystem
Tongass National Forest
'ABSTRACT'

Although species commonly modify habitats and thereby influence ecosystem structure and function, the factors governing the ecological importance of these modifications are not well understood. Pacific salmon have repeatedly been shown to positively influence the abundance of benthic biota by annually transferring large quantities of nutrients from marine systems to the nutrient-poor freshwaters in which they spawn. Conversely, other studies have demonstrated that salmon can negatively influence the abundance of freshwater biota, an effect attributed to bioturbation during upstream migration and nest construction. The factors determining which of these contrasting ecological effects predominates are unknown, including how human activities, such as land use, influence ecological responses to salmon. We sampled a key basal food resource, sediment biofilm, in seven southeast Alaskan streams impacted to varying degrees by timber harvest. Biofilm abundance (measured as chlorophyll a and ash-free dry mass) was positively related to timber-harvest intensity prior to salmon arrival. However, during the salmon run, an inverse relationship emerged of more abundant biofilm in less-harvested watersheds. Among-stream variability in biofilm response to salmon was largely explained by sediment particle size, which was larger in less-harvested watersheds. Collectively, these results suggest that, by altering stream sediment size, timber harvest transformed the dominant effect of salmon from nutrient enrichment to physical disturbance, thus modifying nutrient linkages between marine and freshwater ecosystems.

Tripp, D. B., and V. A. Poulin. 1986. The effects of logging and mass wasting on salmonid spawning habitat in streams on the Queeen Charlotte Islands. Research Branch, Ministry of Forests and Lands, Victoria, Briitish Columbia.

'KEY WORDS'
Main Stream

Tributaries
British Columbia
Effects of logging
Landslides
Salmonid habitat
Spawning habitat
'ABSTRACT'
Describes the efects of landslides on fish habitat in streams including increased sedimentation, washout of LWD, and erosion of streambanks.

Triska, F. 1984. Role of Wood Debris in Modifying Channel Geomorphology and Riparian Areas of a Large Lowland River under Pristine Conditions: A Historical Case Study. Verhandlung Internationale Vereinigung Limnologie 22(3):1876-1892.
'KEY WORDS'
Main Stream
Channel morphology
Geomorphology
Riparian areas
History
Debris
Red River
large Wood
Alluvial deposits
Debris dams
Floods
Snags
'ABSTRACT'
Historical records were reviewed to estimate the impact of wood debris on a 400-500 km reach of the Red River, Louisiana. In the primal condition the banks consisted of unstable alluvial deposits easily eroded during floods. Sediment and riparian trees from eroded banks formed organic debris dams which blocked the channel and promoted channel aggradation. Over a period > 375 years wood debris dams formed in series. The longest instantaneous length of impacted channel was 225 km. The average rate of channel blockage was 1.3 to 1.6 km/year between 1793-1876. Maximum debris accumulation recorded in a single flood was 8.1 km. Exposed wood covered 80-120 km of channel. Debris dams remained in place 80-150 years. Wood debris impacted adjacent riparian areas by flooding forests and forming a series of large lakes. Under natural conditions lakes could become nearly permanent features of the riparian landscape if tributary channels filled with alluvium and organic debris. After removal of organic debris the most recently formed lakes drained over a period of approximately 30 years. Flow reversal in tributaries resulted in channel enlargement and diversion of one half to three quarters of the river 's discharge adjacent to riparian lowlands. Wood debris reduced the river 's width from about 185 m to approximately 40 m, and aggraded the river bed a maximum of 7 m. Tributary channels were dammed or filled with organic debris, and the river was permanently opened to navigation in 1873. Restoration of full channel flow exposed previously buried logs and eroded forested banks. By 1904, seventy years of debris dams and snag removal, levee projects, dredging and cutting bankside trees resulted in a cleared, wide, meandering channel, which might today be mistaken as typical of a pristine lowland river.

Triska, F. J., V. C. Kennedy, R. J. Avanzino, G. W. Zellweger, and K. E. Bencala. 1989. Retention and Transport of Nutrients in a Third-Order Stream in Northwestern California: Hyporheic Processes Ecology 70:1993-1905.
'KEY WORDS'
Tributaries
California
Nutrients

Hyporhreic zone
Nutrient cycles
'ABSTRACT'

Chloride and nitrate were coinjected into the surface waters of a third-order stream for 20 d to examine solute retention, and the fate of nitrate during subsurface transport. A series of wells (shallow pits) 0.5-10 m from the adjacent channel were sampled to estimate the lateral interflow of water. Two subsurface return flows beneath the wetted channel were also examined. The conservative tracer (chloride) was hydrologically transported to all wells. Stream water was >88% of flow in wells <4 m from the wetted channel. The lowest percentage of stream water was 47% at a well 10 m perpendicular to the stream. Retention of solutes was greater in the hyporheic zone than in the channel under summer low-flow conditions. Nominal travel time (the interval required for chloride concentration to reach 50% of the plateau concentration) was variable by well location, indicating different flow paths and presumably permeability differences in subsurface gravels. Nominal travel time was M 24 h for wells <5 m from the wetted channel. Coinjected nitrate was not conservative. Two wells were significantly (P < .05) higher in nitrate-N than would be predicted from chloride, while four were significantly lower. Wells 2.0-4.0 m from the wetted channel tended to have higher nitrate concentration than predicted, whereas nitrate sink locations tended to have transport distances >4.3 m. The capacity of the hyporheic zone for transient solute storage and as potential biological habitat varies with channel morphology, bed roughness, and permeability. A conceptual model that considers the groundwater-stream water interface as the fluvial boundary is proposed. Emerging paradigms of the riverine network should consider the hyporheic zone and associated nutrient cycling as an integral component of fluvial structure and function.

Tschaplinski, P. J., and G. F. Hartman. 1983. Winter distribution of juvenile coho salmon (Onchorhynchus kisutch) before and after logging in Carnation Creek, British Columbia, and some implications for overwinter survival. Canadian Journal of Fisheries and Aquatic Sciences 40:452-461.

'KEY WORDS'
Main Stream
British Columbia
Carnation creek
Coho salmon
Effects of logging
Winter habitat

'ABSTRACT'

Numbers of juvenile coho salmon (Oncorhynchus kisutch) in streams are reduced substantially in winter compared to those that occur in summer. Most of this reduction occurs early in autumn with the onset of the first seasonal freshets. Stream sections containing adequate winter habitat in the form of deep pools, log jams, and undercut banks with tree roots and debris lost fewer fish during freshets and maintained higher numbers of coho in winter than sections without these habitat characteristics. These features provide shelter and reduce stream velocities. Microhabitats occupied by coho juveniles in winter after logging were unchanged from those described before logging — all microhabitats were characterized by low water velocities (≤ 0.3 m/s). Up to 48% of the coho population inhabiting stream sections with adequate shelter remained there by midwinter (Jan. 3). This percentage was typical of stream sections where at least some trees remained after logging. Streamside trees stabilized the banks and prevented their collapse. In contrast, two of three study sections that had been clear-cut logged had unstable banks which collapsed during winter freshets. Almost no coho remained in these sections in winter. Many coho emigrate from the main stream to seek the shelter of low-velocity tributaries and valley sloughs concurrent with the decline of coho populations in Carnation Creek during autumn and early winter. This seasonal shift in distribution reverses in the spring when large numbers of coho reenter the main stream. Fish overwintering in these sites have a high apparent survival rate. Before logging a 4-yr mean of

169 ± 44 coho entered one tributary (a slough called 750-m site) in autumn. Of these numbers entering, 72.2% came out in spring. During and after logging, an annual mean of 288 coho entered the same site. The apparent survival rate during and after logging was 67.4%, essentially unchanged from the prelogging value. Logging has neither reduced the numbers of coho juveniles that enter such sites in autumn to overwinter, nor reduced the numbers leaving these sites to reenter Carnation Creek in spring.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37:130-137.

'KEY WORDS'
Watershed Scale
River continuum
Stream ecosystems
Ecosystem structure
Ecosystem stability
Community succession
Stream geomorphology

'ABSTRACT'

From headwaters to mouth, the physical variables within a river system present a continuous gradient of physical conditions. This gradient should elicit a series of responses within the constituent populations resulting in a continuum of biotic adjustments and consistent patterns of loading, transport, utilization, and storage of organic matter along the length of a river. Based on the energy equilibrium theory of fluvial geomorphologists, we hypothesize that the structural and functional characteristics of stream communities are adapted to conform to the most probable position or mean state of the physical system. We reason that producer and consumer communities characteristic of a given river reach become established in harmony with the dynamic physical conditions of the channel. In natural stream systems, biological communities can be characterized as forming a temporal continuum of synchronized species replacements. This continuous replacement functions to distribute the utilization of energy inputs over time. Thus, the biological system moves towards a balance between a tendency for efficient use of energy inputs through resource partitioning (food, substrate, etc.) and an opposing tendency for a uniform rate of energy processing throughout the year. We theorize that biological communities developed in natural streams assume processing strategies involving minimum energy loss. Downstream communities are fashioned to capitalize on upstream processing inefficiencies. Both the upstream inefficiency (leakage) and the downstream adjustments seem predictable. We propose that this River Continuum Concept provides a framework for integrating predictable and observable biological features of lotic systems. Implications of the concept in the areas of structure, function, and stability of riverine ecosystems are discussed.

Waples, R. S., T. Beechie, and G. R. Pess. 2009. Evolutionary History, Habitat Disturbance Regimes, and Anthropogenic Changes: What Do These Mean for Resilience of Pacific Salmon Populations?

'KEY WORDS'
Watershed scale
Estuaries
Pacific Northwest
North America
Water Resources

'ABSTRACT'

Because resilience of a biological system is a product of its evolutionary history, the historical template that describes the relationships between species and their dynamic habitats is an important point of reference. Habitats used by Pacific salmon have been quite variable throughout their evolutionary history, and these habitats can be characterized by four key attributes of disturbance regimes: frequency, magnitude, duration, and predictability. Over the past two centuries, major anthropogenic changes to salmon ecosystems have dramatically

altered disturbance regimes that the species experience. To the extent that these disturbance regimes assume characteristics outside the range of the historical template, resilience of salmon populations might be compromised. We discuss anthropogenic changes that are particularly likely to compromise resilience of Pacific salmon and management actions that could help bring the current patterns of disturbance regimes more in line with the historical template.

Ward, J. V., and J. A. Stanford. 1983. The serial discontinuity concept of lotic ecosystems. Pages 29-42 *in* T. D. Fontaine, and S. M. Bartell, editors. Dynamics of Lotic Ecosystems. Ann Arbor Scientific Publishers, Ann Arbor, MI.

'KEY WORDS'
Watershed scale
Ecological theory
Ecology
Rivers
Serial dicontinuity
Streams
'ABSTRACT'
The papee expands on river continuum paradigm with the role of dams on flow and connectivity of stream systems. They suggest that dams alter the function of river systems.

Wigington, P. J., and coauthors. 2006. Coho salmon dependence on intermittent streams. Frontiers in Ecology and the Environment 4(10):513-518.

'KEY WORDS'
Headwater streams
Off channel
Oregon
Coho salmon
Movement
'ABSTRACT'
In February 2006, the US Supreme Court heard cases that may affect whether intermittent streams are jurisdictional waters under the Clean Water Act. In June 2006, however, the cases were remanded to the circuit court, leaving the status of intermittent streams uncertain once again. The presence of commercial species, such as coho salmon (Oncorhynchus kisutch), can be an important consideration when determining jurisdiction. These salmon spawn in the upper portions of Oregon coastal stream networks, where intermittent streams are common. In our study of a coastal Oregon watershed, we found that intermittent streams were an important source of coho salmon smolts. Residual pools in intermittent streams provided a means by which juvenile coho could survive during dry periods; smolts that overwintered in intermittent streams were larger than those from perennial streams. Movement of juvenile coho into intermittent tributaries from the mainstem was another way in which the fish exploited the habitat and illustrates the importance of maintaining accessibility for entire stream networks. Loss of intermittent stream habitat would have a negative effect on coho salmon populations in coastal drainages, including downstream navigable waters

Williams, J. E., A. L. Haak, H. M. Neville, and W. T. Colyer. 2009. Potential Consequences of Climate Change to Persistence of Cutthroat Trout Populations. North American Journal of Fisheries Management 29(3):533-548.

'KEY WORDS'
Climate change
cutthroat trout
Pacific Norhtwest
'ABSTRACT'
Warmer water, changes in stream flow, and the increasing frequency and intensity of other disturbances are among the factors associated with climate change that are likely to impact

native trout populations in the western USA. We examined how three of these factors—increased summer temperatures, uncharacteristic winter flooding, and increased wildfires—are likely to affect broad-scale population persistence among three subspecies of cutthroat trout Oncorhynchus clarkii. Our results suggest that as much as 73% of the habitat currently occupied by Bonneville cutthroat trout O. c. utah, 65% of that occupied by westslope cutthroat trout O. c. lewisi, and 29% of that occupied by Colorado River cutthroat trout O. c. pleuriticus will be at high risk from one or more of the these three factors. Within the next 50 years, wildfire, floods, and other disturbances may have a greater impact on population persistence than increasing water temperature alone. Our results also suggest that the risk will vary substantially within subspecies. For each subspecies, our analyses identified large portions of their ranges where all populations either currently fail to meet basic persistence criteria, are at high risk from climate change, or both, indicating a high likelihood of losing the genetic and life history diversity in those areas. Stress from climate change is likely to compound existing problems associated with habitat degradation and introgression from introduced salmonids. Recognition of the increased risk from climate change may warrant altering the management paradigm of isolation and require increased control efforts for invasive nonnative species. Regardless of the management avenue chosen, more populations are likely to become isolated and vulnerable in the near future. Our results argue for immediate restoration actions within certain subbasins to increase the resistance and resilience of at-risk populations and habitats to additional disturbances caused by rapid climate change.

Wissmar, R. C., D. N. Swanston, M. D. Bryant, and K. McGee. 1997. Factors influencing stream chemistry in catchments on Prince of Wales Island, Alaska. Freshwater Biology 38:301-314.

'KEY WORDS'
Tributaries
Southeast Alaska
Geochemistry
Karst

'ABSTRACT'

Factors influencing the water chemistry of streams were evaluated for remote catchments on Prince of Wales Island, Alaska, a high-latitude geologically diverse landscape. We evaluated the hypothesis that weathering rates of dominant geological formations of catchments would be the major factor influencing the water chemistry of streams. Catchments were compared by synoptic sampling of stream and cave waters, mapping to define the distribution of geological formations, and laboratory studies of rock weathering. Carbonation was identified as the major mechanism influencing the weathering of rocks. High Pco2 levels of cave and upwelling waters in streams suggested CO2, supplied by soil respiratory processes, was the major factor controlling the concentration of dissolved CO2, carbonic acid dissociation, and H+ replacement of cations on rock surfaces. Additional evidence of carbonation included the relationship between HCO31− and Ca2+ + Mg2+ (r2 = 0.95) for low- and high-alkalinity waters. The relationship suggested that the high-alkalinity waters were associated with the weathering of calcareous rocks. Waters with high alkalinities (> 1254 μeq l−1), pH (> 7.0), Ca : Mg ratios (> 6.0.) and saturation values for Ca2+ (SIc = −0.59−0.06) indicated that karst (limestone) formations with calcite minerals were the major sources of calcium (> 1266 μeq l−1). Waters with the lowest alkalinity, pH, HCO31− and cation concentrations were associated with granodiorite (igneous) rocks. Laboratory studies substantiated these findings, with weathering being highest when waters contacted Heceta and Bay of Pillars limestone formations and lowest with igneous rocks. Weathering of rocks, and possibly soils, appeared to be facilitated by surface and subsurface movements of CO2 and water through fractured karst formations, and water availability in rain-dominated forests. The proximity of the sea and sulphur in marine aerosols and rainfall, and sulphate reacting with water, may also supply hydrogen ions to weathering reactions. This study provides a basis for developing a better understanding of the influences of surface−subsurface geological and hydrological factors, and climatic conditions, on stream chemistry and biota in high-latitude ecosystems.

Wissmar, R. C., R. K. Timm, and M. D. Bryant. 2010. Radar-derived digital elevation models and field-surveyed variable to predict distribution of juvenile coho salmon and Dolly Varden in remote streams of Alaska. Transactions of the American Fisheries Society 139:288-302.

'KEY WORDS'
Watershed scale
Southeast Alaska
Landscape models
Coho salmon
Distribution
Remote sensing
Digital elevation models

'ABSTRACT'

We used field-surveyed and digital elevation model (DEM) parameters to predict density distributions of juvenile coho salmon Oncorhynchus kisutch and Dolly Varden Salvelinus malma in remote streams on Prince of Wales Island, southeast Alaska, and to identify influences of associated reach-level habitat characteristics. Fish densities were highest in two reach types (low-gradient floodplain reaches and moderately contained reaches) characterized by numerous pool habitats, low pool spacing distances, and abundant large woody debris. Regression analyses indicated that channel gradients measured during field surveys (M-gradients) accounted for significant amounts of the variance in coho salmon and Dolly Varden juvenile densities. The DEMs derived from a 2000 space-shuttle radar mapping mission were used to estimate channel gradients (DEM-gradients) and gradient zones: (0–3, >3–6, >6–15, and >15%) in stream networks. A high correlation between DEM-gradients and M-gradients (r = 0.92), along with the regressions of M-gradient and fish density, provided a template for comparing DEM-gradient and M-gradient predictions of fish densities within gradient zones of the watersheds. Juvenile coho salmon and Dolly Varden densities (number/100 m2) predicted from the DEM-gradients and M-gradients were consistent for a majority of the gradient zones. Our results and expanded analyses should facilitate development of paradigms for assessing fish distributions in remote landscapes.

Wright, K. K., and J. L. Li. 2002. From continua to patches: examining stream community structure over large environmental gradients. Canadian Journal of Fisheries and Aquatic Sciences 59:1404-1417.

'KEY WORDS'
Watershed scale
Oregon
Community structure
Landscape ecology
Stream ecology
Stream habitat
Distribution
Fish
Invertebrates
Algae
Primary production

'ABSTRACT'

We present an approach that integrates a conceptual framework with multivariate ordination techniques and traditional parametric analyses to examine biotic and abiotic gradients in stream ecosystems. Ordinations were used to examine multivariate patterns along an environmental gradient, with individual variables used to interpret those patterns across spatial scales. The conceptual framework provides a consistent context to compare community distributions and consequently allows for hypothesis testing using ordinations. To illustrate the approach, we examined the physical template, fish and benthic macroinvertebrate communities, and algal biomass and production along a 1st- through 5th-order stream gradient in eastern Oregon. We hypothesized that longitudinal distributions of

physical habitat characteristics, fishes, macroinvertebrates, and periphyton would reflect highly variable, discontinuous gradients. Multivariate patterns were determined by rotating nonparametric ordinations to a common set of variables and comparing them to conceptual models of (i) an ideal continuum, (ii) a random distribution, and (iii) discrete patches. Physical habitat and fishes reflected strong longitudinal gradients, macroinvertebrates were the most patchy, and algal biomass and production were highly variable. Distributions of individual variables from site and stream-order perspectives revealed how different factors, potentially influencing stream communities, may be continuous or patchy depending on spatial scale.

Mason D. Bryant and **Fred H. Everest,** Aquatic/Land Interactions Research Program, Pacific Northwest Research Station, U.S.D.A. Forest Service, 2770 Sherwood Ln., 2A, Juneau, Alaska 99801

Management and Condition of Watersheds in Southeast Alaska: The Persistence of Anadromous Salmon

Abstract

In contrast to most of North America and Europe, numerous intact or lightly disturbed watersheds are present throughout southeast Alaska. These watersheds support abundant and diverse populations of anadromous salmonids. While the watersheds throughout the northern hemisphere have been exposed to human disturbance from millennia to centuries, significant human disturbance to the watersheds of southeast Alaska did not begin until the 1950's with the start of industrial logging. Although management of watersheds has evolved to reduce risks to aquatic habitat, the most intensive logging occurred during the first 20 years of timber harvest when few restraints were placed on timber harvest in watersheds. As a result, a legacy of streams with deteriorating habitat remains. While few salmon stocks in southeast Alaska appear to be in decline, escapement records on specific watersheds, particularly those most severely affected by management are non-existent or qualitative. The present status of salmon stocks may be attributed to abundant intact watersheds, high marine survival, and escapement levels that fully seed most watersheds. The numerous intact watersheds throughout southeast Alaska are a critical factor in maintaining sustainable salmon stocks iu southeast Alaska.

Introduction

Hundreds of large and small, and mostly pristine watersheds are dispersed through the islands of the Alexander Archipelago of southeast Alaska. These watersheds are located in a small percent of the remaining temperate rain forests in the world that have not been moderately or severely altered by human activity. Furthermore, these watersheds support a disproportionately high proportion of the wild anadromous salmonid stocks remaining in the Pacific Northwest. Throughout the western United States and British Columbia, Canada, numerous stocks of anadromous salmonids are at risk of extinction, declining, or of serious concern to resource managers and users (Nehlsen et al. 1991; Slaney et al. 1996). Southeast Alaska is an exception with few stocks that have been identified in decline (Baker et al. 1996; Halupka et al. in press). This can be attributed largely to the absence of dams, agricultural and urban development, and intact watersheds in southeast Alaska. Intensive human exploitation of watersheds in southeast Alaska began only a few decades ago and is likely to continue as demand for resources grows with increasing human populations. The most significant potential risks to salmonid stocks in southeast Alaska include large scale habitat degradation and fishing pressure (Baker et al. 1996; Halupka et al. in press).

The purpose of this paper is to examine historical timber harvest patterns, management history, distribution of managed and intact watersheds, and their relationship to anadromous salmonid populations in southeast Alaska. The focus is on land managed by the U. S. Forest Service, the Tongass National Forest (TNF). Although the area of the Tongass National Forest is large, a relatively small portion is suitable for timber production and this portion often coincides with highly productive fish habitat. We briefly describe the development of "Best Management Practices" implemented by the U.S. Forest Service from the 1950s through 1996 and their application in watersheds in southeast Alaska. We examine timber harvest patterns to show the extent of disturbance in managed watersheds and how these patterns have changed from the start of industrial timber harvest to the present. We hypothesize that improvements in forest management practices within the past 5 years and retention of intact watersheds will be an important factor in maintaining healthy anadromous salmon populations in southeast Alaska.

The TNF is part of the temperate rain forest biome (Alaback 1991). Globally, the temperate rain forest consists of about 34.4 million ha (90 million acres) which includes parts of North America, Chile, New Zealand, Tasmania, and the former Soviet Union (Hagenstein 1993; TLMP

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Revision 1997). About 50% of this biome occurs in North America with most located along the coasts of British Columbia and southeast Alaska (Ecotrust et al. 1995). About 14% (5.1 million ha) occurs in the TNF. About 2.0 million ha of the TNF is classified as old-growth forests and is commercial forest[1]. The remaining area consists of alpine, ice fields, muskegs, and areas with less than 10% tree cover. More than 70% of the commercial forest is below 492 m (1500 ft) elevation and of that more than 40% is below 262.5 m (800 ft) elevation (Figure 1). The area less than 262.5 m (800 ft) includes 1,450,000 ha of commercial and non-commercial forests and comprises the most productive valley bottoms. Of this, about 263,000 ha are riparian corridors (TLMP Revision 1997).

[1]commercial forest defined as having >8,000 board feet (b.f.) per acre.

Sitka spruce (*Picea sitchensis*) and western hemlock (*Tsuga heterophylla*) are the dominant trees at the lower elevations and in the stream valley bottoms (Viereck and Little 1972). These were the only economically valuable tree species until five years ago when yellow cedar (*Chamaecyparis nootkatensis*) also became economically important. Alaska yellow cedar extends throughout coastal south central Alaska, southeast Alaska, and southward through British Columbia (Viereck and Little 1972). Western redcedar (*Thuja plicata*) occurs in the southern portion of southeast Alaska. Mountain hemlock (*Tsuga mertensiana*) is found at higher elevations. Red Alder (*Alnus rubra*) is common along disturbed sites, either natural or anthropogenic, such as landslides, abandoned roads, and gravel bars along riparian areas. Understory of old-growth forests is generally comprised of younger trees and various shrubs and

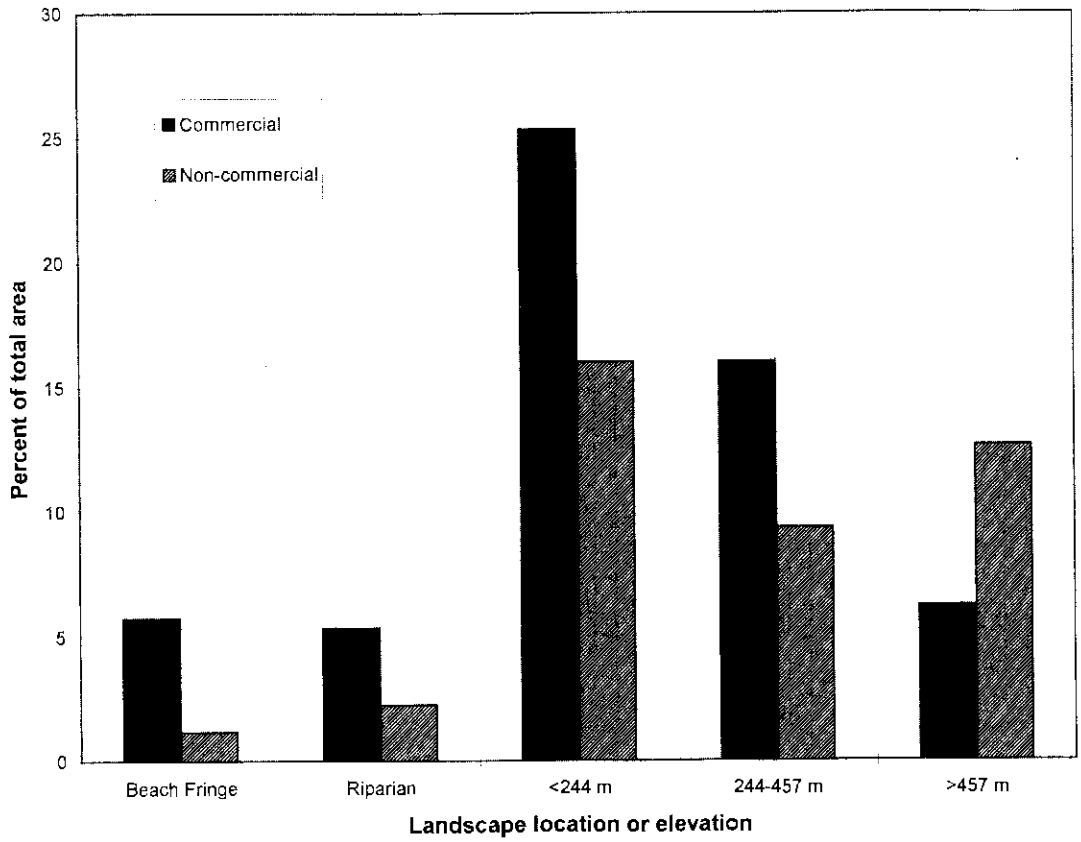

Figure 1. Distribution (% of area) of old-growth forest in southeast Alaska classified into commercial and non-commercial forest by landscape and elevation (TLMP 1997).

herbs including blueberry (*Vaccinium* spp), fool's huckleberry (*Menziesia ferruginea*), stink current (*Ribes bracteosum*), and devils club (*Oplopanax horridus*). A complete description of the vegetation is provided by Alaback (1980).

The landscape of southeast Alaska is geologically young. Most areas were glaciated during the late-Wisconsin glaciation which ended about 10,000 yr BP (Heusser 1960; Pewe 1975; Pielou 1992). The terrain is mountainous and characterized by steep-sided mountains and U-shaped valleys sculpted by glaciers (Harris et al. 1974). High rainfall and heavy snowpack at higher elevations (>300 m) combined with the steep topography drive a complex hydrological cycle that creates fall floods and high water in the spring as the snow melts (Schmiege et al. 1974). The steep, irregular topography and high precipitation contribute to a pattern of frequent natural disturbance events that include landslides and blowdown of large old-growth trees. The combination of varying flows, natural disturbance, and large old-growth trees creates complex stream ecosystems that support large and diverse communities of anadromous salmonids. These, in turn, support valuable commercial, sport, and subsistence fisheries for pink (*Oncorhynchus gorbuscha*), coho, (*O. kisutch*), sockeye, (*O. nerka*), chum (*O. keta*), and chinook salmon (*O. tshawytscha*) (Meehan 1974). The large salmon runs are a significant resource to terrestrial mammals and birds (Shochat 1993; Willson and Halupka 1995) as well as an important nutrient source for aquatic organisms (Cederholm et al.1989; Wipfli in press).

Historical Perspective of Watershed Disturbance

Rivers and their watersheds have been focal points for human occupation and large scale anthropogenic modifications and disturbances have occurred for millennia. Zuaderer (1985) documents the development of canals and irrigation systems of the Tigres and Euphrates river systems to support agriculture for the ancient city states of Ninevah and the Assyrian empire about 3000 yr BP. Watershed and riparian development throughout Europe began centuries ago, beginning with deforestation, agriculture, and more recently, industrial development (Petts 1989; Petts et al. 1989). As a result, few unmodified reaches remain. Atlantic salmon (*Salmo salar*) have been extirpated from many of the rivers. Remaining populations are maintained by hatcheries in numerous rivers throughout western Europe (Mills 1991).

Widespread exploitation of watersheds in North America has spanned three centuries beginning on the Atlantic coast with permanent European settlements in the early 1700's. River valleys were cleared for agricultural development, and streams and rivers were dammed for water power and small mills. Along the east coast of the United States, New England, in particular, loss of Atlantic salmon populations occurred more than 100 years ago and was largely unrecorded. Dams throughout New England extirpated natural runs of Atlantic Salmon throughout the region (Stolte 1981). Wissmar et al. (1994) reviewed chronology of disturbance on the watersheds of the Pacific Northwest which began with the fur trade in 1811. The following decades brought the development of mining, early settlement for agriculture, and livestock grazing throughout the Pacific Northwest. During the course of three centuries of watershed development in North America, no major watershed remains completely free of extensive human disturbance in the contiguous United States. By 1990, less than 20% of the old-growth forests remained in the Pacific Northwest, virtually all on public land (Lehmkuhl and Ruggiero 1991).

The literature reviewed by Wissmar et al. (1994) and McIntosh et al. (1994) suggested that most of the adverse effects of the development and exploitation of the watersheds in Pacific Northwest were largely unreported until the 1970's. They cited only three pre-1940 reports on habitat condition of these watersheds. During the 1980's, reports on the condition of habitat and salmon stocks appeared in greater numbers and by the 1990's several comprehensive reports on habitat condition revealed that large scale deterioration of habitat in the Columbia River basin occurred during the past century (cited in Wissmar et al.1994). The cumulative effects of habitat degradation, combined with over-exploitation, and declining ocean productivity for these stocks has placed 214 native, naturally spawning stocks of salmonids in Washington, Oregon, and California at varying degrees of risk of extinction (Nehlsen et al. 1991; FEMAT 1993). The list includes chinook, coho, sockeye, chum, and pink salmon, as well as steelhead (*O. mykiss*) and sea-run cutthroat trout (*O. clarki*). The combined effects of these factors has created the "salmon crisis" in the last decade. The degree of effect seems to be

related to the duration and density of human occupation.

Large scale anthropogenic influence on southeast Alaska watersheds is a relatively recent phenomenon. Even though European exploration of southeast Alaska began in the 1700's, it was primarily related to the development of the fur trade—sea otter and seals— and whaling and effects on the landscape were localized and relatively minor (McDougall 1993). Even after the purchase of Alaska in 1867 by the United States, timber harvest was restricted to easily accessible areas near the shoreline. By 1900, 14 sawmills were operating and harvested 8.45 million board feet. From 1910 to 1920, another 410 million board feet were harvested (Harris and Farr 1974). Sporadic timber harvest continued through 1950. Large scale harvest did not begin until after a 50-year timber contract for 8.25 billion board feet of timber was awarded to the Ketchikan Pulp Company in 1951 by the U.S.D.A. Forest Service (Harris and Farr 1974). In 1956, a second 50-year timber contract for 5.31 billion board feet of timber was awarded to the Alaska Lumber and Pulp Company. Timber harvest for the two 50-year contracts marks the beginning of industrial forestry and major watershed disturbance in southeast Alaska.

From 1950 through 1992, more than 14 billion board feet of timber were removed from the Tongass National Forest (Murray 1970; Ruderman 1985; Warren 1996). From 1980 through 1994, about 5.9 billion board feet were harvested from private land in Alaska, most of which occurred in southeast Alaska (Ruderman 1985; Warren 1996). The peak harvest on the TNF occurred during the late 1960's through the early 1970's (Figure 2). From 1982 through 1986 harvest was about 1.4 billion board feet. As National Forest land was transferred to private ownership, timber harvest on private lands exceeded that on the TNF. The combined harvest from the TNF and from private

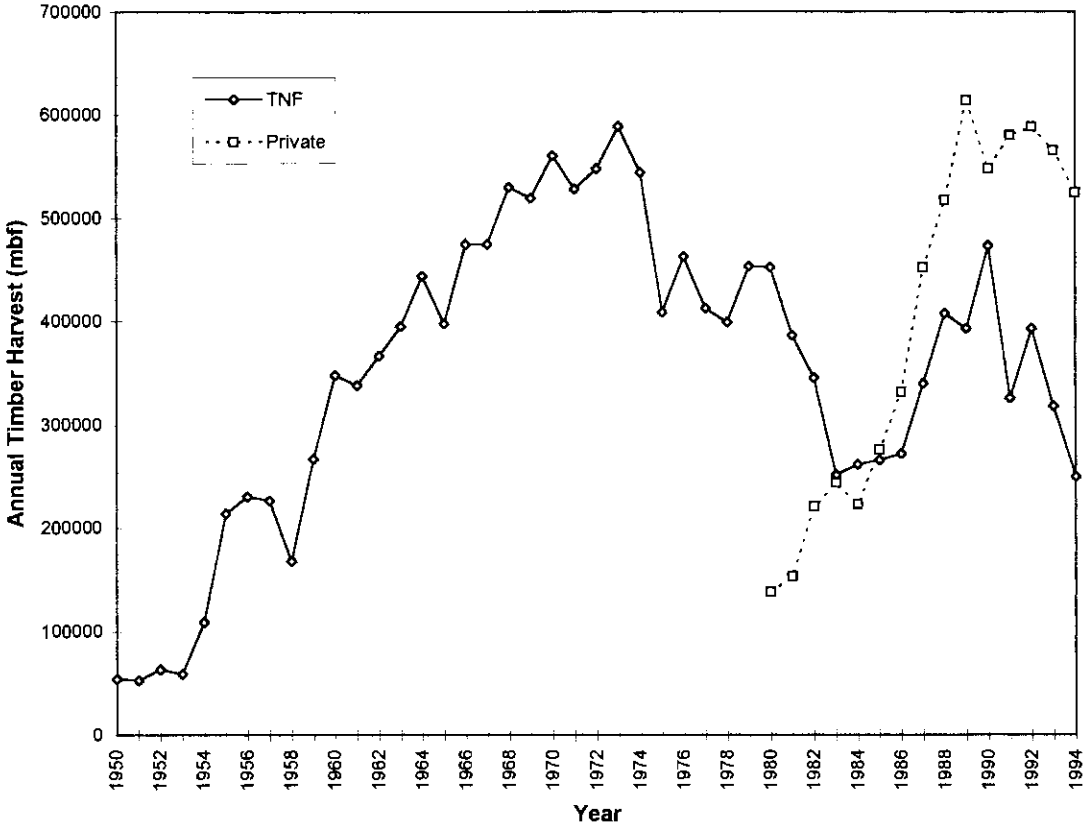

Figure 2. Annual timber harvest in southeast Alaska from the Tongass National Forest (TNF), and state and private lands from 1950 through 1994.

land from 1988 to 1994 was greater than the highest harvest on TNF lands at any point. The mill in Sitka closed in 1994 and a decline in timber harvest occurred on the TNF. The pulp mill in Ketchikan closed in 1997 further decreasing the demand for timber from the TNF.

History of Management Practices in Southeast Alaska

We separate management of riparian areas and stream habitat in the TNF into five intervals: 1950 to 1970; 1971 to 1982; 1983 to 1990; 1991 to 1996; and from 1997. The last period began with the adoption of the Tongass Land Management Plan (TLMP) by the U.S. Forest Service in May 1997. The intervals do not represent precise changes in management policies with the exception of 1991 when buffer strips 32.5m (100 ft) along fish streams were mandated (TTRA 1990). Changes throughout other periods were less apparent in administrative documentation, but the degree of protection to aquatic resources during timber harvest increased during each time interval.

1950-1970

From 1950 to 1970, logging generally proceeded up an entire stream basin with clear-cut logging from the streambank up to an elevation of about 324 m (800 ft). Although felling timber into streams and yarding across streams was prohibited, this was not strictly enforced. Trees were felled into creeks and removed during logging.[2] Logs were often dragged across streams or in some cases logs were dragged along streams. Tractor logging was often used in low gradient riparian areas and road crossings were made through streams. Heavy equipment was operated in streams and along stream banks. Log jams (large woody debris), both natural or resulting from logging operations, were systematically removed from streams either because they were perceived as blocks to migration or created pools that removed spawning areas. Evaluation of the effects of logging was generally accomplished by site inspections which frequently reported equivocal effects.[2] For example, "The tractor road channel may or may not be detrimental." Few guidelines for road layout or design that addressed the protection of aquatic habitat were available (U.S.D.A. Forest Service 1960).

[2]unpublished report. The effect of logging on twelve salmon streams in southeast Alaska. On file with the primary author.

1970-1982

The Logging and Fish Habitat Pamphlet of 1976 outlined improved forest practices for fish streams (U.S.D.A. Forest Service 1976). It identified the significance of small tributaries to juvenile salmonids, steelhead, and non-anadromous species such as Dolly Varden (*Salvelinus malma*) and cutthroat trout. Although the pamphlet recognized the importance of woody debris as cover for fish and cautioned against over-zealous debris removal, it did not recognize the significance of large wood on stream morphology and suggested that large accumulations were blocks to migration and affected spawning habitat. Extensive stream cleaning continued unabated from 1971 through 1982 (Bryant 1983). Examples of road construction, bridge and culvert crossing design, and construction timing to minimize the effect of sediment in streams were given in the Logging and Fish Habitat Pamphlet (U.S.D.A. Forest Service 1976). Significant improvements in aquatic habitat protection were implemented during this period in comparison to the previous 20 years, but streams in most watersheds that were logged during this period had no buffer strips and removal of large wood from most main stem and tributary streams was a common practice. Culverts were often installed that were too small for frequent floods in autumn and washed out. Others were improperly installed and were blocks to adult and juvenile salmonid movement (Bryant personal observation).

1983-1990

The 1979 Tongass Land Management Plan (TLMP 1979), the Alaska Regional Guide (U.S.D.A. Forest Service 1983), and a 1984 review of the Plan (U.S.D.A. Forest Service 1984) incorporated extensive sets of recommendations for stream habitat protection. The Plan in 1979 classified streams with anadromous fish as class I; streams with non-anadromous fish as class II; and tributaries or upstream sections of streams without fish that flow into fish bearing streams as class III. Each stream class was given a different level of protection with the highest afforded to class I streams. Riparian vegetation was retained to provide shade for temperature sensitive streams. Small tributary streams were recognized as habitat for resident trout and Dolly Varden. Riparian habitat was managed to provide large wood to streams that were identified as rearing habitat for identified

species. The recommendation for these streams was to retain all trees that were along the streambank and less than 30.5 cm in diameter at about 1.5 m above ground. Directional felling away from the stream channel was required as was full suspension of timber over streams (i.e. logs were completely elevated above the ground) during yarding operations. Logging in the riparian zone continued as a common practice. Where buffer strips were left, they were generally no wider than one tree height and usually were smaller, non-merchantable trees. In some cases, old-growth buffer strips were left along low gradient floodplain areas such as sections in the upper section of the South Fork of Staney Creek on Prince of Wales Island.

Retention of large wood in streams was included as a management practice in the 1986 Aquatic Habitat Management Handbook as a result of literature demonstrating the importance of large wood in the streams of the Pacific Northwest and Alaska (Swanson et al. 1976; Keller and Talley 1979; Bryant 1983; and Dolloff 1983 among others). Removal of woody debris was restricted to logging slash less than 4 cm diameter, although logging slash was to be prevented from entering the stream in the first place. The handbook specifically recommended against "removal of small debris jams in rearing streams". The large scale removal of woody debris prevalent before 1983 was largely discontinued throughout southeast Alaska (Bryant personal observation).

Significant improvements and specific recommendations for road layout, construction, and stream crossings were included in the Aquatic Habitat Management Handbook (U.S. Department of Agriculture 1986). Practices were incorporated to provide fish passage for both adult and juvenile salmonids and to minimize sediment production. These recommendations included information from an increasing number of reports on methods for forest road construction and stream crossing (Dane 1978; Anderson and Bryant 1980). Although the improved practices were applied and served to reduce effects of roads and stream crossing structures on aquatic habitat, implementation or effectiveness was not systematically monitored and numerous instances where procedures were either not followed or improperly applied occurred throughout the region. More significantly as roads were closed, many stream crossing structures were either not removed or improperly removed. As a result long term, chronic erosion and sediment transport sources were created.

1991-1996

The most obvious change in management in 1991 was the requirement for buffer strips that were at least 32.8 m (100 ft) wide where no trees would be cut along class I, and II streams that flowed into anadromous fish streams (Tongass Timber Reform Act 1990). It included provisions of the Aquatic Management Handbook (U.S. Department of Agriculture 1986). The "Best Management Practices" in the Soil and Water Conservation handbook (U.S. Department of the Agriculture 1993) were applied to protect riparian habitat and stream habitat in managed watersheds. Some significant additions in the Soil and Conservation Handbook were the recognition of riparian areas and floodplains, and specification of management practices to protect habitat beyond the mandated 32.8 m buffer zone. It states that "preferential consideration to riparian dependent resources will be given when conflicts among land use activities occur". The handbook also presents monitoring as an "essential" part of "Best Management Practices" and describes procedures and three stages of monitoring to evaluate management practices. Specific prescriptions are given to maintain channel integrity of all streams including non-fish streams. Guidelines are provided to reduce the potential for mass failures from road construction and location, road crossings, and log decks.

The set of management practices used from 1990 through 1996 were a significant improvement from those in place at the onset of industrial timber harvest and represented an accumulation of management and research information during that period. Throughout the evolution of "Best Management Practices" implementation was unevenly applied. In the absence of effective monitoring, detrimental streamside harvest practices such as debris removal, improper road and culvert installation and maintenance continued even after they were identified in various administrative directives. To the benefit of some watersheds, management practices exceeded those required by various directives. In many watersheds, buffer strips were extended beyond 32.8 m, particularly in floodplains. While the practices

used from 1991 through 1996 generally provided good protection to streams used by anadromous and non-anadromous salmonids, they did not address many of the important intermittent and higher gradient streams that are an integral part of the drainage systems in watersheds. Most of these systems are at elevations above 262.5 m (800 ft) and may not support perennial fish populations. However, they can have a significant influence on downstream habitat and fish populations (Heede 1972; Benda 1990; and Lamberti et al. 1991).

In 1994, the U.S. Congress directed the USDA Forest Service to review existing habitat protection for salmonid streams on the Tongass National Forest to determine their effectiveness (U.S. Department of Agriculture 1995). The report included an extensive literature review on Pacific salmon and their habitat in southeast Alaska and the Pacific Northwest, examined existing data and information on salmonid populations and their habitat in the Tongass National Forest, and reported results from a field evaluation of seven watersheds by a staff of Federal fisheries and watershed professionals (U.S. Department of Agriculture 1995). The assessment also included a pilot watershed analysis of three watersheds in southern, central, and northern southeast Alaska. Some of the key recommendations in the report were for increased protection of headwater streams, ephemeral streams, and non-fish bearing streams that are tributaries to resident and anadromous salmonid streams, site specific buffers on flood plains and alluvial streams, management of fish streams "to preserve biological integrity", and implementation of watershed analysis. The results and recommendations from the Anadromous Fish Habitat Assessment Report formed the cornerstone for the provisions for fish stream and watershed management in the Revision of the Tongass Land Management Plan (TLMP 1997).

1997

In May 1997, the revision of the Tongass Land Management Plan (TLMP) was completed and included increased protection for watersheds and associated riparian areas (TLMP 1997). A significant element of the TLMP is to consider fish habitat in the context of the whole watershed from the headwaters to the ocean during management planning. The plan directly addresses riparian areas and assigns management objectives that include maintaining natural conditions for fish and aquatic life, managing for biodiversity, and to maintain streambank and stream channel processes (TLMP 1997). In most floodplain stream reaches, buffer areas are required over the active floodplain. Changes to prescribed riparian buffers require a watershed analysis to assess "important riparian and aquatic habitat values and geomorphic processes within a watershed". While maintaining and reinforcing protection afforded to anadromous salmonid — class I streams, resident salmonid —class II streams, and non-fish —class III streams, TLMP identifies an additional group of streams—Class IV streams. These include intermittent, ephemeral, and small perennial channels with low flows or low sediment transport capabilities. Although these are not specifically addressed for riparian protection, buffer strips in V-notches of higher gradient class III streams, which may be ephemeral, are included. The management strategy of the standards and guides is to retain natural stream function and processes in watersheds that fall into management categories that will be used for timber harvest or other development activities.

History of Timber Harvest

Timber harvest in southeast Alaska was not located randomly throughout time or location (TLMP Revision 1997). The sites with easy access and high volumes of wood were harvested first. High volume timber sites were frequently located in the valley bottoms less than 244 m (800 ft) in elevation and along riparian corridors. More than 70% of the total timber volume on the TNF was harvested from 1950 through 1982, the period when minimum riparian protection was given to streams (Table 1). Of the 156,600 ha (387,000 acres) logged from 1950 through 1994, 38% (59,896 ha) were harvested from 1950 through 1970 (Table 1). Furthermore, timber harvest was concentrated on selected islands such as Chichagof, Kuiu, Prince of Wales, and Revillagigado islands (Figure 3). Other locations such as Admiralty Island, most of which has been designated as a National Monument, and Baranof Island which contains large wilderness areas, are less developed. They also contain extensive areas of non-commercial forests.

During the first 20 years of large scale timber harvest, most harvest occurred in the valley bottoms where low gradient streams are located. These

TABLE 1. Timber harvest of the Tongass National Forest by time period and by elevation.

Time period	Area in hectares and (percent harvested) during each time period			
		Elevation (m)		
	Total	<244	244-457	>457
1950-70	59.896	54027.6 (90.2)	5665.8 (9.5)	202.35 (0.34)
1971-1982	53.420	45164.5 (84.5)	8094.0 (15.2)	161.88 (0.30)
1983-1990	28.329	20639.7 (73.6)	6879.9 (24.5)	526.11 (1.88)
1991-1994	14.974	9308.1 (62.2)	5261.1 (35.1)	404.7 (2.70)

streams and their associated riparian habitats (i.e. beaver ponds) also are the most productive locations for anadromous salmonids. From 1950 to 1970, 90% of the total area harvested was less than 244 m in elevation (Table 1). As timber harvest progressed, proportionally more area above 244 m was harvested and by 1991 more than 37% of the timber harvest occurred above 244 m in elevation (Table 1). Less than 30% of the area was harvested while measures were in place that recognized small rearing streams and afforded increased protection to non-anadromous species. Less than 10% of the total area was harvested when buffer strips were an integral and required part of watershed management.

Mayheso Creek, located on Prince of Wales Island, was harvested during the 1950's. Harvest began at the bottom of the valley and continued up to 366 m (1,200 ft) in elevation during one entry (Figure 4). All merchantable timber along the riparian zone was harvested which was the general pattern of harvest for the first 20 years of timber harvest. In subsequent years, harvest patterns changed and by 1970 smaller patches (usually < 40 ha) were cut. The earlier harvest units tended to be in the valley bottoms. Most of the harvest units during 1974 through 1976 in the Corner Creek watershed were located along the stream, whereas, those cut in 1986 or later were located at higher elevations away from the main channel (Figure 5). A similar pattern occurred in the Hamilton Creek watershed with later units placed further upstream and on higher elevations (Figure 6). The concentration of intense timber harvest in the riparian zone before 1980 left a legacy of watersheds with little or no riparian protection.

The effect of time and elevation of timber harvest is reflected in the number of kilometers of stream that were within 100 m of a timber harvest unit. More than 75% of the length of anadromous streams were near or adjacent to a timber harvest unit that was cut from 1950 to 1982 (Table 2). As changes in forest management that moved harvest units away from the riparian zone were implemented, the amount of harvest in these areas decreased. The largest part of timber harvest occurred in the valley bottoms adjacent to streams before adequate management practices were in place to reduce even the most obvious detrimental effects of timber harvest on stream habitats.

The Persistence of Salmon

Despite the legacy of watersheds that were logged with poor riparian management practices throughout southeast Alaska, such as the Maybeso Creek watershed, few documented declines of salmon stocks are recorded (Baker et al. 1996). However, few watersheds in southeast Alaska have escapement records that are dependable before 1960 and no systematic or quantitative studies document juvenile or smolt numbers from watersheds before the onset of large scale logging in southeast Alaska (Halupka et al. in press).

Natural fluctuations at a region-wide scale caused by changes in marine survival, weather cycles, or other environmental factors as well as production of salmon from intact watersheds will obscure declines in a single watershed. The limitations of catch statistics as indicators of stock abundance are generally acknowledged (see Bisson et al. 1992), but the cycles of the commercial catch of coho salmon from 1893 through 1998 demonstrate the magnitude of region-wide variation that obscure trends

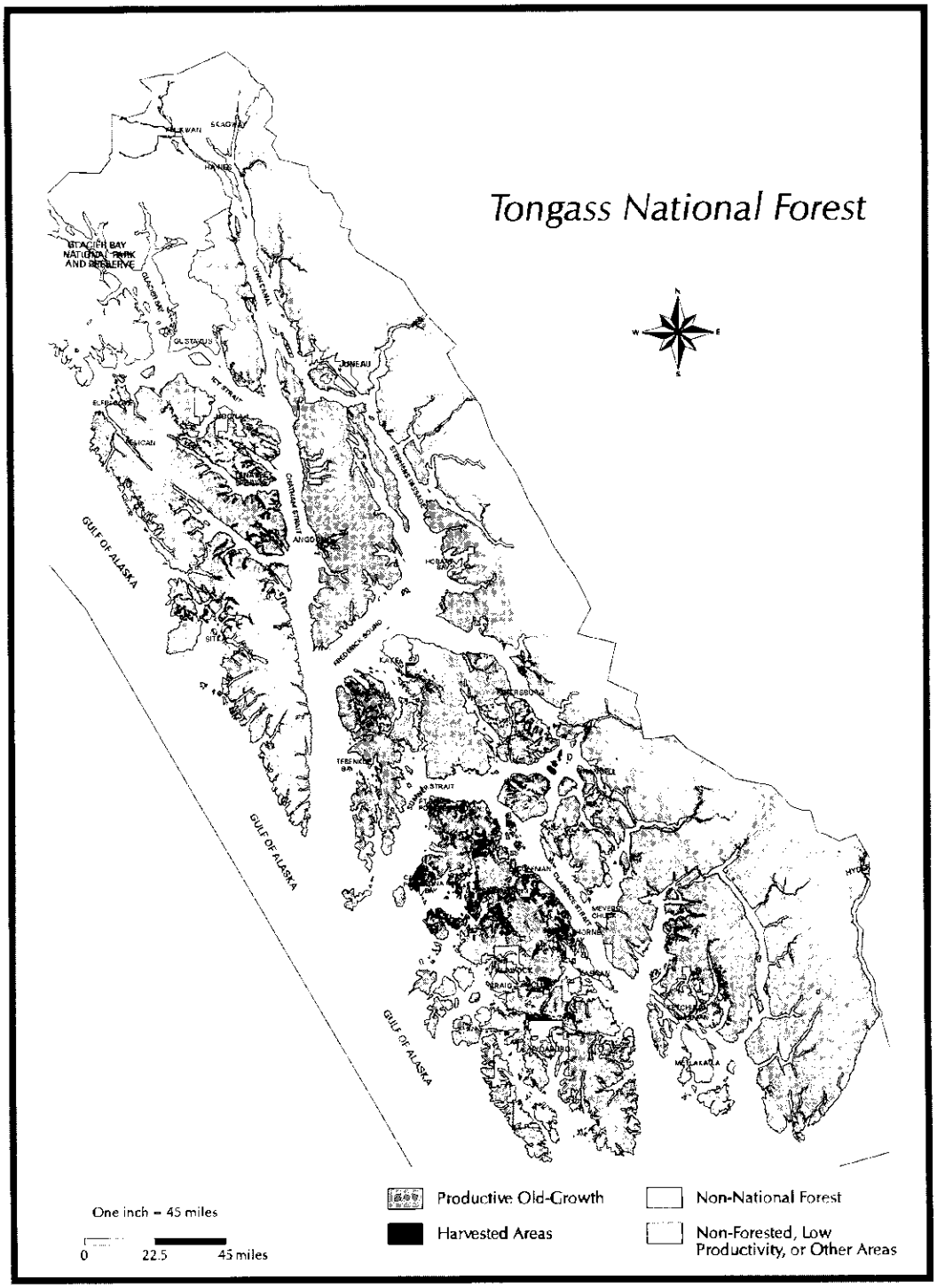

Figure 3. Distribution of commercial old-growth forest and timber harvest throughout southeast Alaska. "Productive Old-growth" was defined as forest cover of old-growth Sitka Spruce and Hemlock ≥ 22.8 cm (9 in) diameter breast height (dbh) and >150 years old at a volume of > 8 million board feet/acre. Timber harvest on non-National Forest land is not identified.

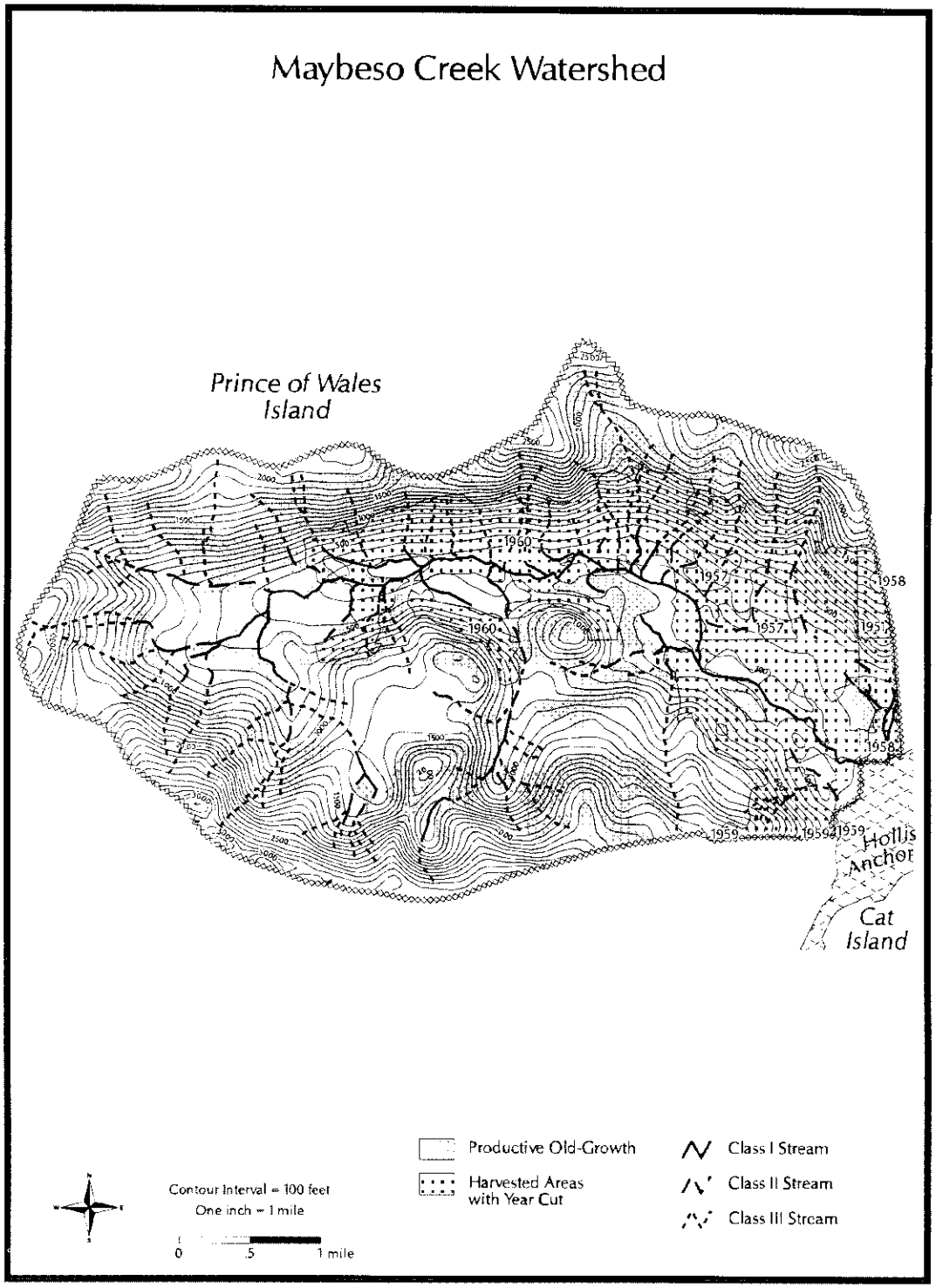

Figure 4. Timber harvest, dates of harvest, and remaining old-growth forest on the Maybeso Creek watershed, Prince of Wales Island.

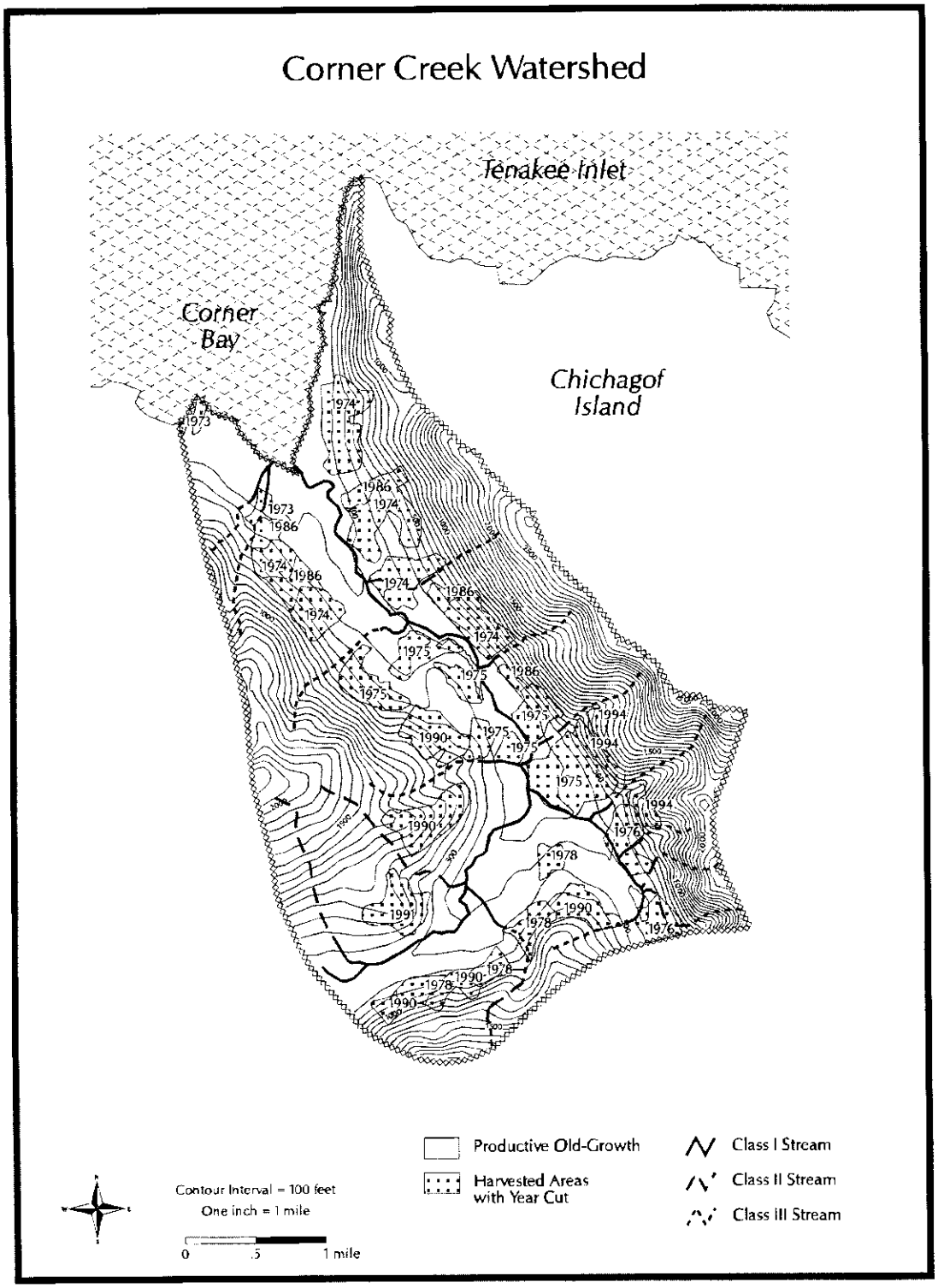

Figure 5. Timber harvest, dates of harvest, and remaining old-growth forest on the Corner Creek watershed, Chichagof Island.

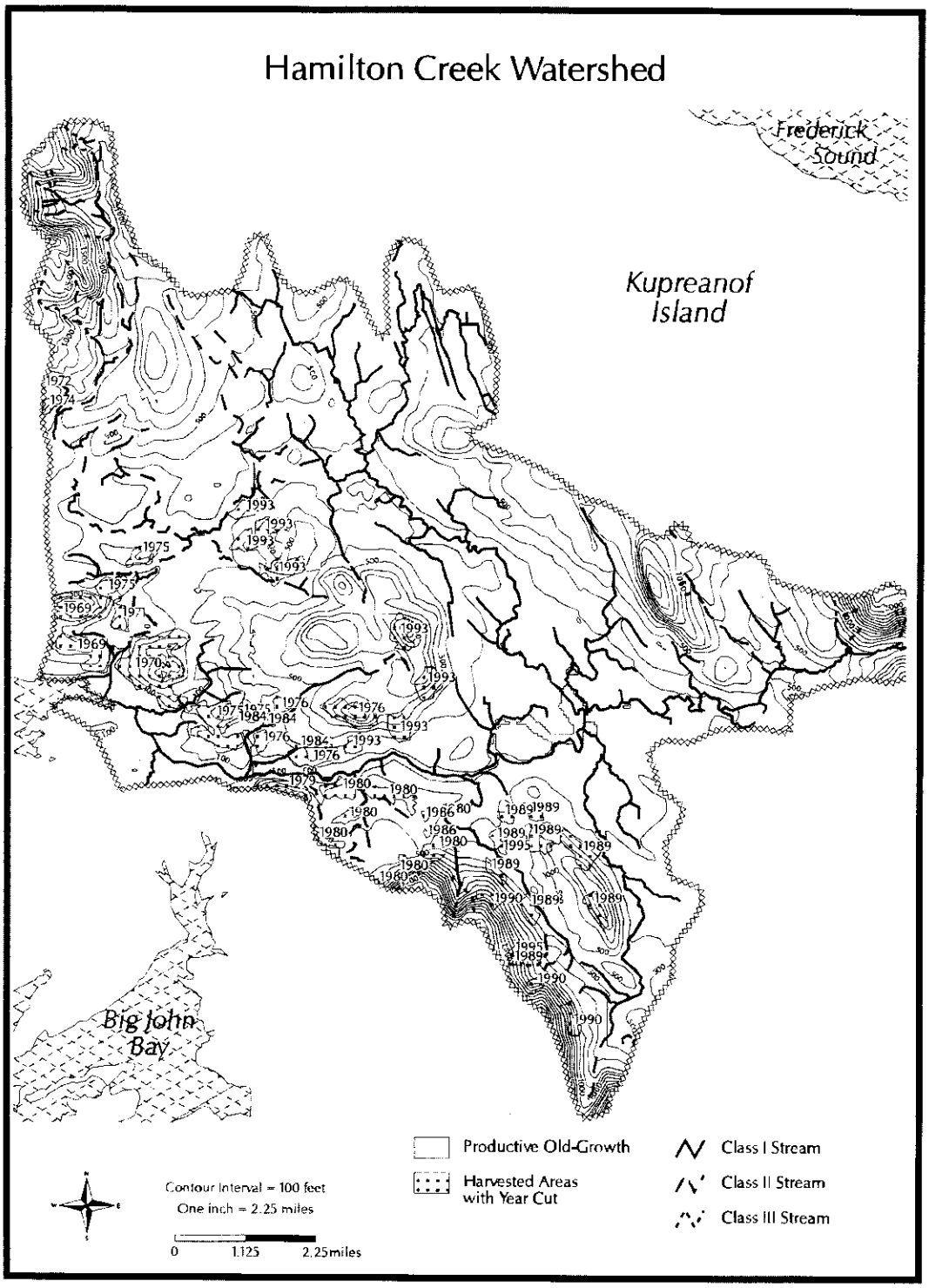

Figure 6. Timber harvest, dates of harvest, and remaining old-growth forest on the Hamilton River watershed, Kupreanof Island.

TABLE 2. Kilometers and (percent) of anadromous and non-anadromous streams within 100 m of a timber harvest unit.

Year Harvested	Anadromous (Class I) Streams	Resident Class II Streams	Total
1950-1970	880.3 (44)	326.8 (32)	1207.9 (40)
1971-1982	678.0 (34)	372.6 (38)	1051.2 (35)
1983-1990	114.4 (6)	127.0 (13)	241.5 (8)
1991-1995	319.8 (16)	189.1 (19)	509.2 (17)

for individual watersheds (Figure 7). As a result, any declines in salmon production in individual watersheds have gone undetected.

High marine survival rates during the past decade have been implicated in increased catch and escapement to all watersheds (Hofmeister et al. 1988; Shaul 1994; Beamish et al. 1997). Changes in marine survival will have a large effect on adult returns and increased marine survival will offset decreased freshwater survival as a result of habitat loss in individual watersheds. For example, estimates from Hugh Smith Lake in southeast Alaska were 51,789 smolt for 1983 and 23,499 smolt for 1985; the estimated marine survival rates for the two years were 7.4% and 19.1%, respectively (Shaul 1994). Adult returns, using these two survival estimates, were 3,832 fish in 1983 and 4,488 fish in 1985. The higher marine survival in 1985 produced a higher return from 50% fewer smolts. An increase in marine survival can easily mask a 50% loss of juvenile salmonid production in freshwater.

The effects of logging on salmon populations under various management prescriptions and particularly in watersheds with poor protection of riparian and stream habitat have been documented and reviewed in several sources (Salo and Cundy 1987; Meehan 1991; Hicks et al. 1991; Bisson et al.1992; Woodsmith and Buffington 1996). Habitat loss and reduced survival of anadromous salmonids have been documented in several studies of

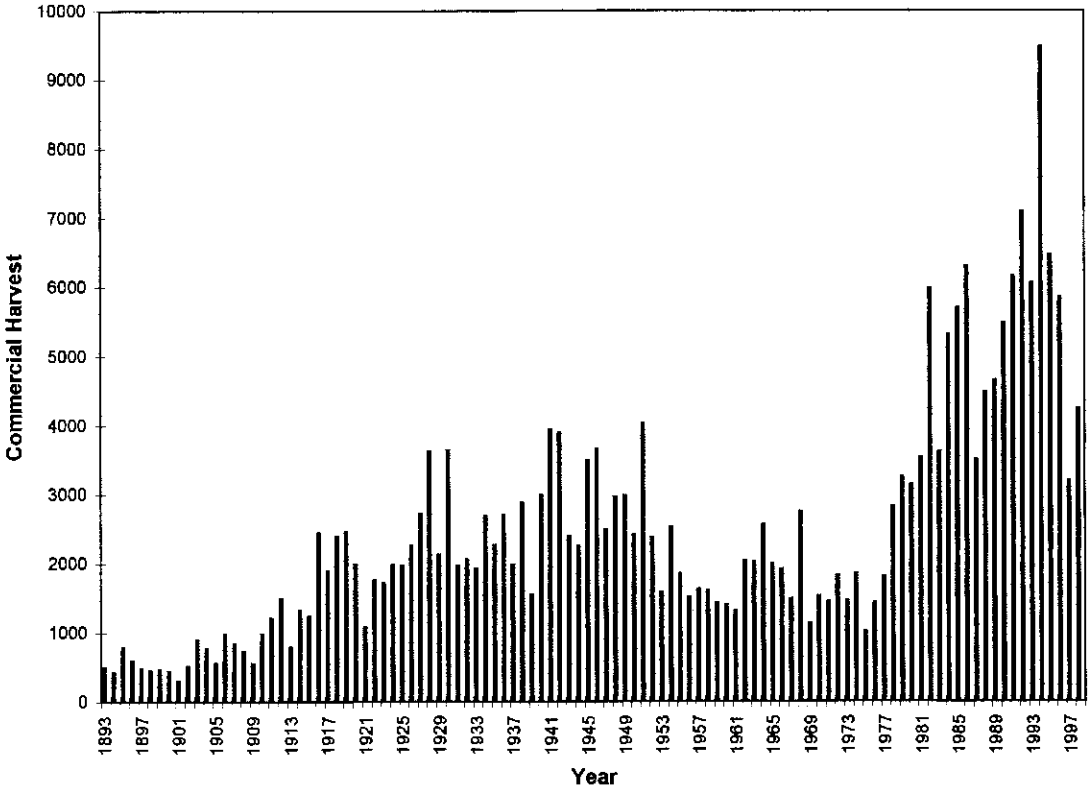

Figure 7. Annual commercial harvest of coho salmon in southeast Alaska from 1900 through 1992 (Rigby et al. 1991 and unpublished data Alaska Department of Fish and Game, Division of Commercial Fisheries, Juneau, AK).

watersheds in southeast Alaska that were logged before 1980 (Bryant 1985a; Thedinga et al. 1989; Montgomery et al.1995). The common theme among these studies is loss of stream channel complexity and simplification of habitat. However, even in intensively logged watersheds, populations of anadromous salmonids remain, but these populations are closely linked with complex habitats maintained by large wood (Dolloff 1983; Bryant 1985a). These complex habitats have been maintained by large wood that was present in streams before timber harvest. For example, in severely impacted watersheds, such as Maybeso Creek and the Harris River, large wood from old-growth trees that fell into the stream before logging provide the only anchors for large wood accumulations. The residence time of large trees in stream channels of southeast Alaska spans decades (Swanson et al. 1984; Murphy and Koski 1989) and the effects of their loss will be gradual and appear decades after the cessation of timber harvest in the watershed (Bryant 1980). A commonly accepted misconception is that stream recovery begins shortly after the cessation of activity in the watershed. In reality, the loss of habitat quality is likely to continue for more than 100 years after logging until riparian trees become large enough to maintain stream channel complexity (Bryant 1980; 1985a; Beechie et al. 1994).

Persistence of fish in some of the more severely affected watersheds can be attributed to escapements which fully seed available habitats, and the ability of some species of salmonids, coho salmon in particular, to exploit marginal habitats. The presence and expansion of beaver pond habitats in intensively logged watersheds may be an important factor maintaining coho salmon population in some of these watersheds (Bryant 1985b; Sampson 1994). Small, low gradient 2nd to 3rd order tributaries, often less than 2 meters in width, are common rearing habitat for juvenile coho salmon in southeast Alaska (Elliott and Hubbartt 1978). The size of the tributaries may allow retention of smaller wood debris (Bilby 1985). Small streams with greater amounts of large wood in intensively managed watersheds maintain higher densities of juvenile coho salmon than those with smaller amounts of wood and few pools (Dolloff 1983). The persistence of salmonids in larger watersheds is maintained by complexity that includes both instream structure, most of which is related to residual large wood, and heterogeneity of habitats within watersheds which includes a mixture of off-channel habitats, beaver ponds, and small tributaries.

On a region-wide scale, the existence and retention of intact watersheds may be a significant factor in maintaining the current status of salmon stocks in southeast Alaska. More than 75% of the Tongass National Forest is classified into Land Use Designations (LUD) I or II which include wilderness areas, National Monuments, and roadless areas (Tongass Land Management Revision 1997). This includes a substantial part of Admiralty Island and the Misty Fjord which are designated as National Monuments. Although large portions of the area in LUD I and LUD II categories are alpine, glacial, or areas of marginal economic value with respect to timber harvest, numerous highly productive watersheds in old-growth forests are also included in these designations. The watersheds in these protected landscapes represent the largest group of intact watersheds in the northern hemisphere. Other designations are LUD III which is primarily recreation use and LUD IV which allows intensive resource use such as timber harvest and mining.

The existence of these low impact land use designations (LUD I, III, and III) has resulted in a relatively high percent of anadromous streams (56%) —measured by length— that are located in watersheds that have not been affected by timber harvest (Figure 8). A total of 9,911 km (6,160 miles) (or 44%) of anadromous stream flow through watersheds that have been exposed to varying degrees of timber harvest. Of this group of watersheds, 6% of the stream length is located in watersheds that have had more than 20% of the area logged. These are likely to be the most severely affected stream systems. The largest percent of anadromous stream length is in watersheds with less than 20% of the area logged (Figure 8). While it is likely that habitat has been lost in these systems, especially in the most intensively harvested watersheds such as Maybeso Creek (Bryant 1980; 1985a), a decrease in the number of fish may be obscured by natural fluctuations and by production of fish from intact streams on a region-wide basis.

We propose the following hypothesis: 1) anadromous salmon production—measured by smolt output—will be more variable in intensively logged (>20% of area) watersheds than in intact

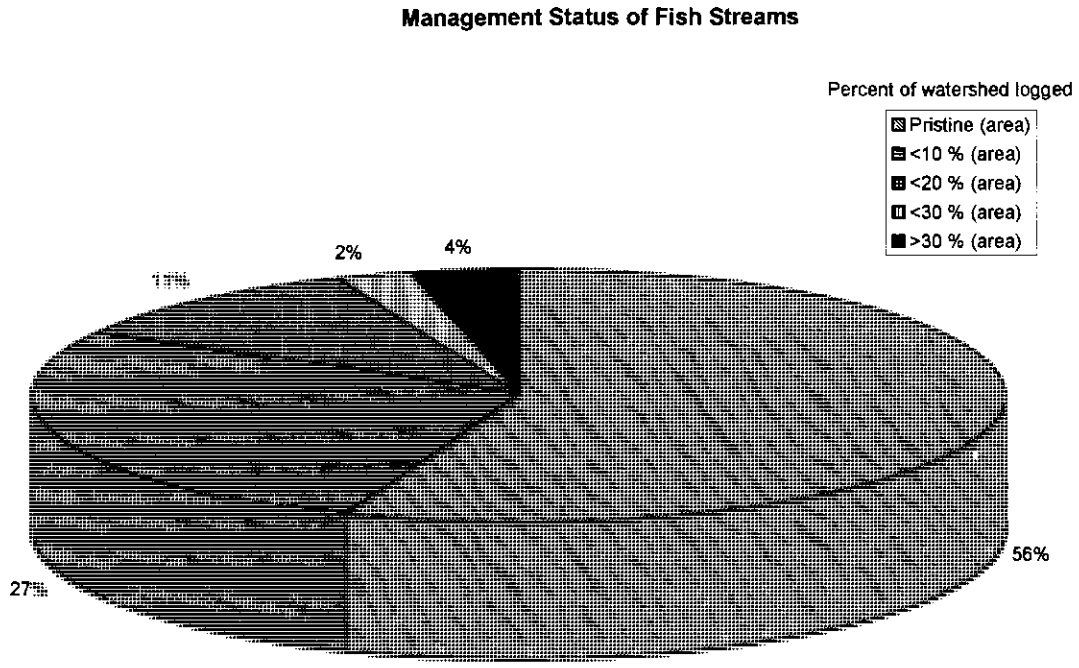

Figure 8. Relative length (%) of anadromous streams flowing through watersheds with varying percent of area logged in the Tongass National Forest.

watersheds; 2) decreases in marine survival, including natural and fishing mortality, will contribute to greater variability of smolt production in intensively managed watershed than in intact watersheds; and 3) environmental disturbances—droughts, severe floods, landslides—will have a proportionally greater effect on variability in intensively managed watersheds than in intact watersheds. The unifying theme in these hypothesis is the test of resiliency of watersheds and the prediction that intensively managed watershed will be less resilient than intact watersheds. The outcome is that salmon production in watersheds that have been intensively logged such as those during the first 30 years of industrial timber harvest in southeast Alaska may not be measurably different than many intact watersheds during benign environmental conditions and high ocean productivity, but are likely to be less resistant to adverse effects during periods of environmental stress. The streams in the unmanaged (i.e. pristine) watersheds are likely to be the most effective buffer for freshwater production of anadromous salmon region-wide.

Summary

Although "Best Management Practices" have substantially improved during the past 40 years of commercial timber harvest in southeast Alaska, most timber harvest on the Tongass National Forest occurred during the first 20 years of industrial logging and was disproportionately distributed in the valley bottoms before adequate riparian protection was in place. As a result, a legacy of watersheds with second-growth vegetation with limited potential for recruitment of large wood, as well as other adverse effects rising from poor timber harvest practices remains. While loss of salmonid production from these watersheds may be significant, it has been undocumented. However, more than 50% of the length of anadromous stream habitat remains in pristine watersheds of the Tongass National Forest. The number and distribution of intact watersheds in the TNF are critical elements for sustainable salmon populations in the face of habitat loss elsewhere in southeast Alaska and the Pacific Northwest.

The catastrophic declines in salmon stocks that have occurred in other areas of the Pacific Northwest are the result of multiple effects that include urban and agricultural development, timber harvest practices, and large dams such as those on the Columbia River which may have been exacerbated by cyclical decreases in marine productivity (Bisson et al. 1992). Furthermore, the patchwork of landowners along nearly all the watersheds of the Pacific Northwest complicates watershed management and restoration (Kohonen 1996). The multiple effects present in the Pacific Northwest are generally absent in southeast Alaska; therefore, the prospect of sustainable salmon stocks in southeast Alaska is good, but depends on a combinations of factors. The retention of existing intact watersheds is a key element. Improving riparian management in managed watersheds is important to prevent further loss of aquatic productivity. A systematic and well-planned and monitored watershed restoration effort is possible and could accelerate the recovery of damaged watersheds. A continuing conservative harvest strategy on salmon that accommodates changes in marine and freshwater conditions to allow spawning escapements that maintain freshwater productivity is essential.

Globally, few intact watersheds remain in the northern hemisphere and nearly all are in the northern regions of the hemisphere (Dynesius and Nilsson 1994; Arthington and Welcome 1995). Most of the remaining intact watersheds of the temperate rain forest occur in southeast Alaska. As the land base within southeast Alaska is further separated into old-growth forests and nonforest land, the relative amount of unmanaged old-growth forest becomes smaller. The relatively small proportion of old-growth forests supports some of the most productive salmonid habitats in North America, and a disproportionately large segment of the salmonid fishery value for the Pacific Northwest. Furthermore, these watersheds represent calibration sites for management of disturbed watersheds within the northern temperate zone and are of global significance.

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RESEARCH ARTICLE

Long-term exposure to more frequent disturbances increases baseline carbon in some ecosystems: Mapping and quantifying the disturbance frequency-ecosystem C relationship

Brian Buma[1]*, **Thomas Thompson**[2]

1 Department of Integrative Biology, University of Colorado, Denver, United States of America, **2** USDA Forest Service, Resource Monitoring and Assessment Program, PNW Research Station, Anchorage, AK, United States of America

* brian.buma@ucdenver.edu

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Abstract

Disturbance regimes have a major influence on the baseline carbon that characterizes any particular ecosystem. Often regimes result in lower average regional baseline C (compared to those same systems if the disturbance processes were lessened/removed). However, in infrequently disturbed systems the role of disturbance as a "background" process that influences broad-scale, baseline C levels is often neglected. Long-term chronosequences suggest disturbances in these systems may serve to increase regional biomass C stocks by maintaining productivity. However, that inference has not been tested spatially. Here, the large forested system of southeast Alaska, USA, is utilized to 1) estimate baseline regional C stocks, 2) test the fundamental disturbance-ecosystem C relationship, 3) estimate the cumulative impact of disturbances on baseline C. Using 1491 ground points with carbon measurements and a novel way of mapping disturbance regimes, the relationship between total biomass C, disturbance exposure, and climate was analyzed statistically. A spatial model was created to determine regional C and compare different disturbance scenarios. In this infrequently disturbed ecosystem, higher disturbance exposure is correlated with higher biomass C, supporting the hypothesis that disturbances maintain productivity at broad scales. The region is estimated to potentially contain a baseline 1.21–1.52 Pg biomass C (when unmanaged). Removal of wind and landslides from the model resulted in lower net C stocks (-2 to -19% reduction), though the effect was heterogeneous on finer scales. There removal of landslides alone had a larger effect then landslide and wind combined removal. The relationship between higher disturbance exposure and higher biomass within the broad ecosystem (which, on average, has a very low disturbance frequency) suggest that disturbances can serve maintain higher levels of productivity in infrequently disturbed but very C dense ecosystems. Carbon research in other systems, especially those where disturbances are infrequent relative to successional processes, should consider the role of disturbances in maintaining baseline ecosystem productivity.

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Introduction

Disturbance processes, like fires, windstorms, and landslides are ubiquitous and present in all ecosystems [1, 2] and critical to carbon cycling [3]. They trigger rapid change in resources availability, short term carbon loss [4], and reorganization of the ecological community [5]. Individual biomes and ecosystems generally have a characteristic suite of disturbance processes operating on varying spatial and temporal scales, and individual ecosystems can be maintained by characteristic disturbance regimes [6]. While historical systems may have had some sort of characteristic C stock value which incorporated the historical disturbance regime, e.g. [7], climate change is rapidly altering the drivers of disturbances, increasing or decreasing their frequency, intensity, and subsequent severity. Novel disturbance processes and new interactions between historical disturbances are also emerging [8]. As a result, understanding the fundamental relationship between disturbances and carbon is important to projecting the magnitude and even the direction of future changes in carbon stocks.

Knowledge of how disturbances alter ecosystem C stocks from short to long time scales has advanced considerably over past decades. Long-term data, paleoecological reconstructions of historical disturbances, e.g., [9], and real-time flux measurements [4] have allowed for precise understandings of C loss, recovery, and net change. However, most studies have been conducted in single disturbance scenarios. Our knowledge of multiple disturbances and their interactive effects on C stocks and trajectories is much more limited [8, 10].

Over relatively short time spans, and holding other factors constant, relative differences in C stocks are driven by the mean return interval of a disturbance and the time required to recover lost C from that disturbance [11, 12]. If disturbances are frequent (occur prior to the recovery of C lost in earlier events), the cumulative effect would be C stocks less than the "potential" maximum C stocks, e.g. [13]. Very long-term chronosequences, however, suggest that at millennial or longer return intervals the relationship between disturbances and baseline ecosystem C is reversed [14], where baseline means average C densities across multiple sites. Undisturbed ecosystems sometimes display retrogression, where soil nutrient availability declines [15] result in lowered ecosystem productivity and potentially biomass C [16], for example through paludification [17]. Disturbance events can also serve to increase nutrient availability by rapidly decomposing and releasing nutrients sequestered in live or dead biomass [18, 19] (but see [20]), disrupting impermeable soil layers [21, 22], exposing unweathered parent material [23], or providing openings for early successional, N-fixing species [24]. Overall, intermediate frequencies of disturbance may result in more productive forests [25] and potentially higher overall C than in the absence of disturbance events. So what is the fundamental role of disturbance frequency in ecosystem C balance?

The prediction of lowered C at both very high disturbance frequencies and very low disturbance frequencies results in the expectation of maximal "baseline" biomass for a given ecosystem occurring at relatively intermediate levels of disturbance frequencies when examined over very long periods of time [11, 16]. Time between disturbance events would be long enough (on average) that biomass stocks could re-accumulate after any given disturbance event, but not so long as to result in hydrologically driven productivity declines, nutrient limitations, or other retrogressive process.

Support for the "disturbance-baseline C hypothesis," that an intermediate level of disturbance promotes higher baseline biomass, is typically inferred from chronosequences [14, 16] or paleo ecological studies [11, 22]. An alternative method involves investigating biomass and ecosystem functions along an exposure gradient. Similar to a chronosequences varying time via space, this method varies disturbance frequency as a function of space.

Our goal here is to test the disturbance-baseline C hypothesis, and in particular the untested expectation that areas of very infrequent disturbance will have less baseline C than areas with intermediate frequency disturbance events, indicating that the disturbance process is influential in maintaining long-term productivity and higher biomass levels. It is difficult to generate long-term, spatially explicit records of disturbance frequencies. Many disturbance types interact directly with landscape composition and structure, shaping the relative frequency and disturbance intensity themselves (e.g., fire and vegetation feedbacks).

Site

The temperate rainforests of southeast Alaska avoid these difficulties. The disturbance regime is spatially stable with little confounding effects. Event size is small relative to landscape size and recovery rapid relative to return interval, so the assumption that recent large events are unlikely to be driving any observed pattern is reasonable [26, 27]. There is essentially no fire; the most recent evidence of widespread natural fire is >5000 years ago and limited to the southern portion of the region [28]. Insect and pest outbreaks are infrequent and minor, rarely resulting in mortality [29]. The primary disturbances are windstorms and mass movement, driven and constrained by topography. Yellow-cedar decline, a mass mortality event associated with climate warming [30, 31] is widespread but does not appear to be affecting net C stocks [32]. As a result, it is possible to test if long-term spatial differences in disturbance exposure (representing the relative frequency of disturbances within a given area, *sensu* [33]) drives spatial differences in regional ecosystem C stocks above and beyond the short-term fluctuations resulting from any given disturbance event and recovery.

Wind is the most common disturbance driver. The location of low-pressure systems and their movement is strongly constrained by the semi-permanent Aleutian low pressure system, the shape of the Gulf of Alaska, and the Coast Range mountains. This constrains storm force winds to the south-southeasterly aspects [34], and the incised landscape results in a heterogeneous distribution of exposed slopes. Dendrochronological reconstructions show that disturbance frequency corresponds well with exposure [35], a pattern exploited to study soil C dynamics [36], watershed-scale C distributions [37], and regional forest dynamics [32].

Landslides (and avalanches) are similarly spatially stable. Because the region has relatively shallow soils the most dominant type of landslides are shallow debris avalanches and flows that occur in predictable locations (associated with slope, drainage, and wind) and a several-century return interval [38]. For both processes, because the topography of the landscape has been essentially stable since the Last Ice Age, exposure to is assumed to be relatively consistent.

The climate is moderate along the entire coast due to its hyper-maritime nature. The species composition is consistent, dominated by intermixed conifer species, *Picea sitchensis* and *Tsuga heterophyla*, with other species occasionally found throughout: *T. mertensiana*, *Thuja plicata*, and *Cupressus nootkatensis*. *Alnus viridis* is the only major broadleaf species, and generally only found in recently disturbed locations. This consistent species assemblage limits confounding C estimates with changes in community composition. All species (except *T. plicata* found in the far south) range both further north and south of the region limiting concerns about significant range edge effects and shifting dominance patterns [39, 40]. Therefore the relationship between relative disturbance exposure and C stocks can be quantified spatially with minimal confounding influence from climate or variable vegetation composition.

Here we test the disturbance-baseline carbon hypothesis that higher disturbance rates are associated with higher C in infrequently disturbed systems and lower C in frequently disturbed areas. This was done by asking three primary questions:

1. Does long-term disturbance exposure drive fundamental differences in biomass, density, or basal area at the regional scale, above and beyond any influence of short-term variability driven by recent disturbance or stochastic mortality?

2. Does the type of disturbance change the magnitude of the effect?

3. What is the cumulative baseline non-soil ecosystem C in the region (excluding human management), and how does that compare to scenarios without disturbance?

Methods

USFS Forest Inventory and Analysis plots (FIA) is the most extensive forest survey in the region. FIA plots are a fixed area design, consisting of one central location and three surrounding subplots at 120^0, 240^0, and 360^0, at a 36 m center-to-center distance from the central point; all four subplots are 7.3m radius. Plots are placed approximately 5.3 km apart and visited on ~10 year intervals (10% of plots visited per year). Survey dates utilized here range from 2005 to 2017. Plots with any history of management, historical or contemporary, were discarded to focus on natural dynamics and total potential regional C in the absence of human intervention. In total, 1491 plots met this criterion. Of those, 484 are in non-forested locations (cover permanently <10%). For testing the disturbance exposure-carbon relationships, only the forested plots were utilized (n = 1007) unless otherwise noted. This eliminates non-informative low values of C in areas on glaciers and other alpine areas. The entire set of 1491 plots were used for regional C modeling to accurately capture low C values at high elevations.

Three forest structure metrics were assessed. The FIA program estimates dry biomass by measuring all live tree greater than 2.54 cm diameter at breast height (DBH) and dead trees >5" DBH; allometric equations are used to scale volume to dry biomass [41]. Where needed, this value was converted to C at a rate of 50% [42]. Trees per hectare (TPH) is a significant response variable because a decrease in per-tree biomass may be offset by an increase in density. Finally, while C content or biomass is generally the goal of ecosystem ecology studies in a global context, estimation of those values requires allometric equations which embody considerable uncertainty. Basal area (BA) is a variable which incorporates both tree diameter and tree density and may be more precise for plot to plot comparisons [43]. All three variables were assessed using identical methods.

Environmental variables

At each site, several topographic, bio-climatic, and disturbance exposure metrics were quantified. Elevation, slope, and transformed aspect [44] were derived from 1 arc-second ASTER GDEM2 data. Mean summer and winter precipitation and temperature, mean annual temperature, and the mean length of growing season (LOG), date of continuous freeze (DOF), and date of continuous thaw (DOT) were taken from the Scenarios Network for Arctic Planning (1960–1990 climate normal, 771m, [45]), then downscaled to 30m resolution via bilinear interpolation. Percent forest cover was from the Landsat Vegetation Continuous Fields (VCF) tree layer [46], which estimates the percentage of ground per 30m pixel covered by vegetation >5m in height via rescaling MODIS vegetation values with Landsat 5 TM and ETM+ data.

Disturbance exposure

Exposure represents the relative probability of a disturbance within a given spatial area (e.g., [47]). Wind exposure was modeled via methods described in [33] (the EXPOS model), modified [35] and applied [32] to the region. Briefly, the method assumes straight-line winds which interact with topography; high topographic barriers upwind can shelter downwind locations. Wind passes over those topographic obstacles and bends downwards at eight angles ranging

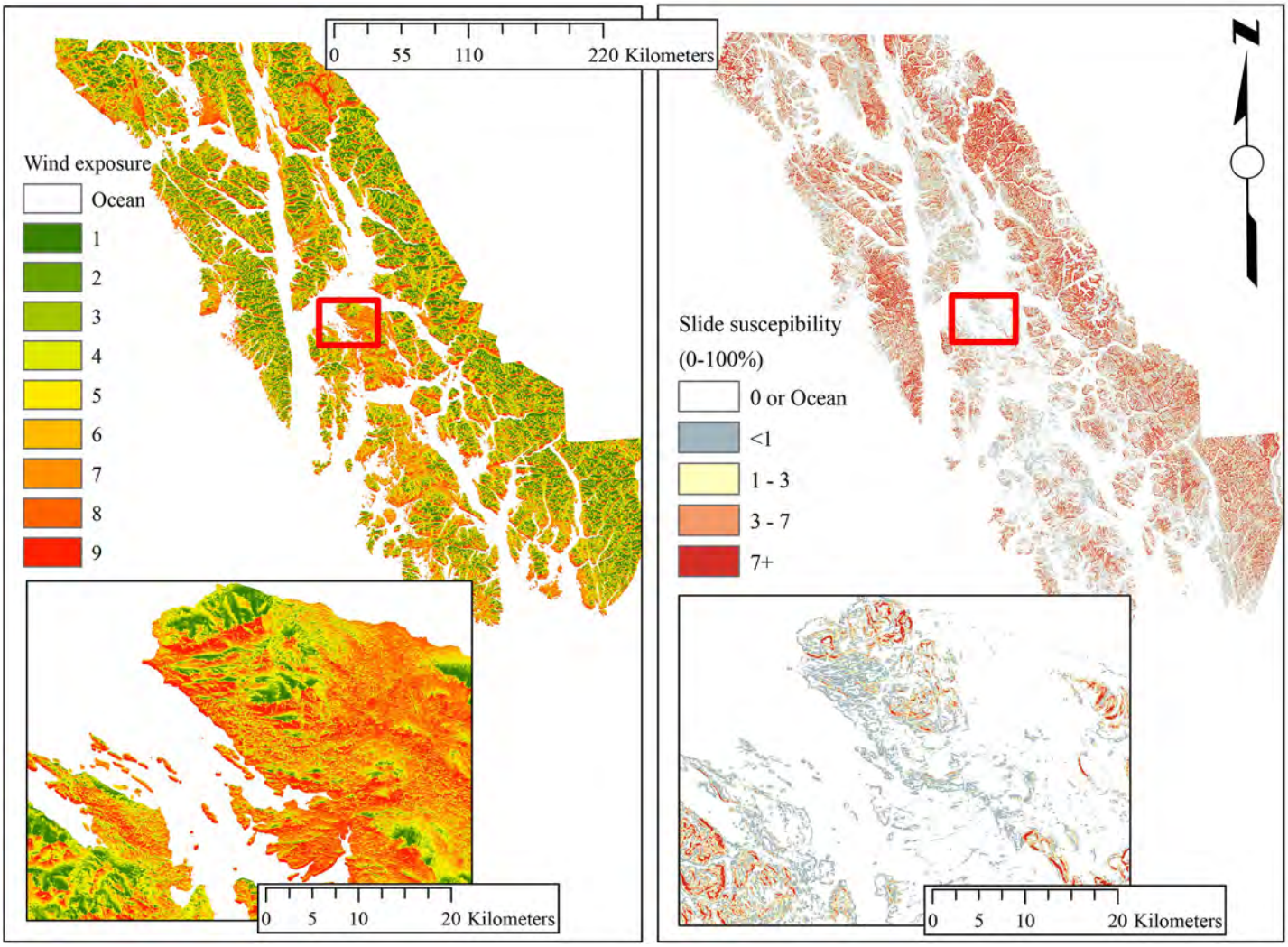

Fig 1. Disturbance exposure maps utilized in the investigation. Both are relative scales, from low to high. Wind exposure refers to storm force winds, see [35] for design and comparison to field data. Slide susceptibility is based on observed slides, see [48] for design. Inset maps are for illustrative purposes at a finer scale.

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from 1 – 14° in 2° increments. The degree of exposure is a result of how directly a location is exposed to storm winds. Using the observed distribution of wind directions [34], an average of three incoming wind directions (SW to SE) was created to determine average long-term exposure.

A recent assessment [48] modeled landslide likelihood, ranging from 0 (very unlikely) to 100 (has slid/likely to slide), based on observed slide locations in non-harvested forest throughout the central portion of the region. Significant variables include slope, local topographic position, contributing area, and exposure to wind during storms. The two exposure maps (Fig 1) were created at 30m resolution in R using the raster, sp, and rgdal packages.

Analysis

In these forests, stochastic mortality of large trees can cause very high variance at plot scales, as single large trees (>60m tall) may dominate a plot [43, 49]. The fundamental interest here is

the underlying "baseline" average, incorporating that natural variability. In other words, the effect of time-since-disturbance was intentionally not considered. Thus 1) the modeling and statistical focus is on relative differences *on average* compared to exposure and 2) all models were chosen and explicitly tuned to avoid overfitting plot-scale data (with its inherent time-since-disturbance variability) to regional projections.

Several methods were used because of the inherent variance in the data given the goals, but no test was considered as definitive on its own. This avoids the potential for Type 1 errors resulting from multiple methods to influence results. The statistical framework and all modeling choices were made *a priori*, without seeing the real data. All coding was done in R using dummy data by Buma, then the framework was applied to the actual FIA dataset by Thompson independently. This avoids unintentional bias or over-tuning in modeling building.

Direct correlation. To explore direct relationships between disturbance exposure and biomass at the plot scale, simple, single-error linear regressions were constructed between the disturbance exposure metrics (wind, landslides) and the response variables (biomass, TPH, BA). Assumptions were checked and data log transformed if necessary.

Disturbance importance, plot scale. Two stepwise regression models were conducted, one with, and one without, the disturbance variables (wind and landslide exposure). Backwards stepwise linear regression was done, starting with all the potential explanatory variables. When the best model (as evaluated via AIC reduction) was constructed for each condition (with and without considering disturbance), Chi-square difference tests were used to compared between the models with and without disturbance. Significant differences in residual deviance were interpreted as one model being significantly better.

Spatial modeling. Finally, random forest modeling was used to explore variable significance and create predictive maps of regional C. Random forests [50] are a machine learning technique built on classification and regression trees that accommodate non-linear interactions and correlated/non-independent predictor variables. To construct the random forests, first a random subset (80% of the data) were selected for model construction with 20% retained for testing.

The extreme topography, which stretches from the ocean to icefields, required a two-step modeling procedure to capture both variation *within* forested areas as well as variation *between* forested areas and non-forested areas. First, a random forest model for the forested areas only (>10% cover, n = 1007) was created, which focused on variation as a function of variables relevant to intraforest variability in the structural attributes. Then, to constrain biomass estimates in areas of non-forest or low forest cover (<10% tree cover), a second model incorporating the full suite of plots (n = 1451) was constructed, which results in a clear high elevation/ice cap vs. forest distinction. This output was applied to areas on the landscape where tree cover was <10% to create a continuous response surface.

Random forests have a well-known tendency to overestimate low values and underestimate high values due to a regression to the mean associated with bagging predictors. A correction was applied [51] using cubic smoothing splines.

Using four different biomass calculation methods, [52] determined that the aboveground component measured by the FIA data is between 46–58% of the total biomass C. Data was scaled by those factors for a high and low estimate ("scaling" factor in Table 1). To accommodate the steep topography, final values were scaled from raster pixel area to actual surface area by a cos(slope) raster.

Disturbance importance, regional scale. To estimate the significance of disturbances to regional C balance, we utilized a method conceptually similar to [53], which "turned off" fire in a global vegetation distribution model and calculated differences between with-fire and without-fire outputs. To estimate the cumulative effect of wind on the landscape, the model

Table 1. Estimated potential carbon content. Scaling refers to the assumed proportion of total ecosystem carbon in the modeled biomass.

Scenario	Scaling (%)	Estimated C (Pg)	Difference from actual (%)
Actual conditions	58	1.21	-
	46	1.52	-
No wind	58	1.19	-2
	46	1.50	-2
No slide	58	0.98	-19
	46	1.24	-18
Neither	58	1.02	-16
	46	1.29	-15

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was re-run with wind exposure set to 1 (the minimum) across the entire landscape. The difference between this "null" wind model and the biomass model was calculated via subtraction. The same was done for landslide likelihood (set to 0) and both wind and landslides simultaneously to create three hypothetical comparison scenarios–without wind, without landslides, and without both.

Results

There were direct correlations between exposure to both disturbance exposures and the forest variables considered, though with considerable variability. Higher wind exposure was significantly correlated with higher tree density, though with considerable variance ($F_{(1, 1005)} = 23.4$, $p < 0.001$, $r^2 = 0.02$), but not biomass or BA. Landslide exposure was significantly associated with all three metrics, in the direction of larger, more widely spaced individuals–higher landslide exposure was correlated with significantly higher biomass ($F_{(1, 1005)} = 137.6$, $p < 0.001$, $r^2 = 0.12$) and BA ($F_{(1,1005)} = 104.4$, $p < 0.001$, $r^2 = 0.09$), and significantly lower tree density ($F_{(1, 1005)} = 14.4$, $p < 0.001$, $r^2 = 0.01$).

Disturbance importance, plot scale

Models incorporating disturbance as a predictor were significantly improved over non-disturbance models. For dry biomass, model selection identified the candidate model which included landslide exposure as significantly better than the model without (Chi_2 test, $p < 0.001$). Wind was not retained in the stepwise variable selection process, and thus no significant improvement was seen for dry biomass when considering wind. For BA, both wind and landslide exposures were retained in the better model (Chi_2 test, $p <0.0001$). For TPH, both wind and landslide exposures were retained in the better model (Chi_2 test, $p < 0.0001$).

Spatial modeling

Modeled dry biomass was correlated with the 20% of observations retained for testing (simple linear regression, comparison between predicted and independently observed dataset; $F_{(1,200)} = 133.3$, $r^2 = 0.4$, $p < 0.001$). Basal area was similar to dry biomass ($F_{(1, 200)} = 97.4$, $r^2 = 0.33$, $p < 0.001$). Tree density observations were also significantly correlated with model predictions, though with high variance ($F_{(1, 200)} = 7.8$, $r^2 = 0.04$, $p < 0.006$).

After applying the spline correction and projecting over the region, the fit between predicted and observed values at each sampling point was strong for dry biomass ($F_{(1, 1005)} = 1509$, $p < 0.001$, $r^2 = 0.60$), tree density ($F_{(1,1005)} = 1311$, TPH, $p < 0.001$, $r^2 = 0.57$), and BA ($F_{(1,1005)} = 1863$, $p < 0.001$, $r^2 = 0.65$). Incorporating the entire dataset, including non-forested plots, further improved the overall predicted vs. observed fit for all variables ($F_{(1, 1489)}$ for all;

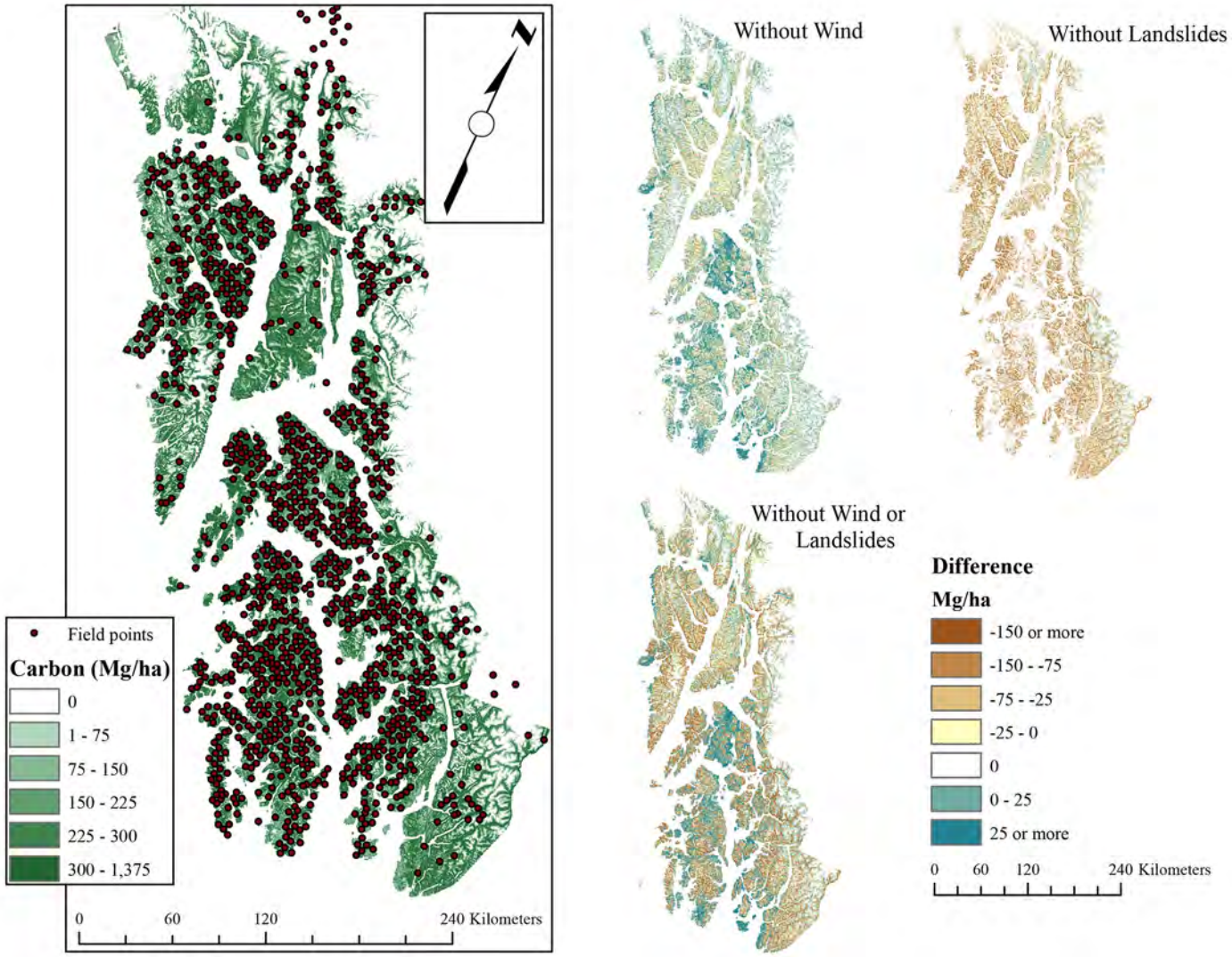

Fig 2. Modeled baseline C stocks and distribution of field points. Smaller regional maps show result of removing disturbance processes (via setting exposure to zero) and then differencing with modeled carbon stocks; negative values indicate more C when disturbances are included, positive values indicate less C.

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dry biomass: $r^2 = 0.69$; TPH: $r^2 = 0.67$; BA: $r^2 = 0.77$). The most important variables in predicting variation in biomass and BA within forested landscapes was observed forest cover, slope, elevation, and likelihood of landslide initiation. For tree density, the most important variables were observed forest cover, aspect, elevation, and mean winter precipitation.

Estimated regional carbon, basal area, and tree density and disturbance effects. Estimated potential biomass (living and dead) carbon ranged from 1.21–1.52 Pg (Table 1). Overall regional biomass did not change substantially (-2%) when the model was run assuming no wind disturbance. There were local differences, however, with many open, low angle areas predicted to have higher biomass (Fig 2). Removing the landslide component resulted in a substantial decline (-18 - -19%) in predicted biomass regionally, with a few areas of predicted increase in very steep topographical locations. Removing both disturbance processes resulted in a substantial decline (-15 - -16%) in regional carbon, but not as much as removing landslide alone. This suggests an interaction between wind and landslide exposure (Fig 2).

Basal area was higher in all non-disturbance scenarios, though to a lesser extent than C. In terms of individual stems, the removal of wind lowered estimated cumulative stem counts by a substantial margin, whereas removing landslides increased the number of stems. These two counteracting processes nearly canceled each other out when both disturbances were removed (Table 2). Together, these results suggest that wind exposure has little effect on carbon (potentially via the offsetting of lower biomass and higher tree density, though the multiple tests do not agree on strength); landslides have a stronger relationship. Though landslides appear to drive lower overall tree density (setting landslide exposure to zero results in higher cumulative stem values), their removal from the BA estimation reduces cumulative BA enough to also result in a significant decline in total biomass. Overall, both disturbances independently and combined result in higher biomass through changes to both BA and tree density.

Discussion

The goal of this work was to determine the significance of disturbances to regional C baselines in the most biomass-C dense forest biome on the planet, temperate rainforests. The region investigated, southeast Alaska, is known as a globally significant C storehouse [52, 54] though a high resolution, spatially explicit estimate of C stocks never been conducted. Spatial differences in disturbance exposure were correlated with baseline differences in biomass, tree density, and BA throughout the region, and those differences scaled up to the regional level. Generally, higher biomass was correlated with increasing disturbance exposure at the low end of the disturbance gradient, as hypothesized, even when accounting for the short-term variation imposed by random mortality. At the high end of the disturbance exposure gradient, very high wind exposures were not correlated with lower biomass C, but very high landslide exposures were associated with low (zero) estimates of biomass C. This is not surprising, as extremely steep slopes (e.g., $>60^0$) are highly unstable and generally not vegetated as a result. In all cases, removing disturbance as a factor in the models resulted in lower overall biomass C, sometimes substantially. This was also true for BA. Tree density was mixed, with wind driving higher densities and landslides lower densities. This suggests that regionally, disturbances are maintaining higher levels of production in vegetated landscapes than would exist in their absence.

Regional C stocks

Estimated potential regional biomass C stocks ranged from 1.21–1.52 Pg. This represents 5–7% of total US forest C (including public and private land, [41]). This is a potential C stock, not considering active or historical removal of C via logging. Logging has a history in the region; [52] estimated that between .01-.02 Pg were lost to logging between 1900–1995. Soil C, not included here, is a larger fraction of total ecosystem C than biomass. A regional modeling effort [55] estimated an average of 302 (+/- 146) Mg soil C ha^{-1}, for a total of 1.78 Pg for the same region. Together, they add to a potential ecosystem C of 2.99–3.30 Pg C for the study area.

Table 2. Estimated cumulative basal area and cumulative stems.

	Regional basal area (m^2)	Difference from actual (%)	Cumulative stems (individuals)	Difference from actual (%)
Actual conditions	2.40×10^8	-	7.49×10^9	-
No wind	2.27×10^8	-5	6.89×10^9	-8
No slide	2.14×10^8	-11	8.13×10^9	+9
Neither	2.12×10^8	-12	7.71×10^9	+3

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The estimate presented here align well with previous estimates in the general area, which ranged from 0.97–0.99 Pg of biomass C and 2.8 +/- 0.5 including soil C [52] and the FIA dataset, which estimated 0.863 Pg biomass C in forested areas only (>10% cover). The slightly lower value predicted by [52] is likely due to differences in design and slight differences in extent: This study focused on the potential natural C stocks for the region and thus excluded logging as a factor, whereas [52] incorporated logging. They also only estimated C stocks for the Tongass National Forest (68,000 km^2), whereas this study did the entire region (76,492 km^2). The FIA dataset is similarly estimating C over a smaller area (>10% only).

Wind. Across the landscape, the role of wind exposure was relatively minor compared to landslides, and the statistical results were mixed. In terms of a direct correlation, trees were denser in areas with higher wind exposure but that relationship did not extend to BA or biomass. The correlation with density, while significant, was very weak. No decline in biomass at high levels of wind exposure was found (using all 1491 plots to allow for the possibility of sparse/non-forest conditions due to wind; S1 Fig), suggesting that even in areas of highest exposure, the current wind regime does not drive significantly lower levels of baseline C at the regional scale. This implies that while wind disturbance may quite significant in terms of impact after discrete blowdown events [35], its influence at the regional scale is less important than other factors. In [35], at a finer scale (single island), exposure was calculated slightly differently: Wind exposure was incorporated into a broad, single index which included wind, soil stability, slope, and elevation. In the present study, those were treated as separate variables from wind exposure to allow for interactions between topography and the wind itself. The lack of a meaningful relationship between wind and biomass/BA (in terms of regional scale totals) suggests that the role of wind exposure in driving baseline differences in forest structure is more significant at finer, subregional scales and less important at broader scales (Fig 2). Generally, wind effect was strongest on lower slopes, where excluding wind resulted in slightly higher projected C; this was offset by the lower magnitude but widespread reduction in C on steeper slopes. The total effect is little net change in C upon removing wind exposure.

Landslides. In contrast to wind, landslide suitability was correlated more strongly with all structural variables and appeared to be more influential at the regional scale. In addition to potentially alleviating nutrient limitation (by exposing bedrock to weathering and shifting live biomass to the decomposing biomass pool), landslides also influence drainage and likely disrupt iron pan formation and Sphagnum introgression. Soil moisture is negatively correlated with forest structure (community composition and structure: [56]; biomass: [37]), as well as nutrient dynamics [57]. Most studies of the influence of landslide on soil properties have focused on fertility, with a decline in nutrient availability associated with the burial of surficial organic layers, e.g. [58], noted in some studies, or no difference noted between landslide deposits and undisturbed soils [59] immediately after a disturbance. However, others [60] noted that disturbances in areas where N-fixers are common post-disturbance community, as here, disturbances can progressively increase soil N availability. Over short timescales (<30 years), slower height growth on landslide scars relative to neighboring slopes was noted [61] in coastal forests to the south, however they did not distinguish between planted and naturally regenerating seedlings. They also noted that N-fixing species were a main component of the recovering ecosystem, potentially increasing long-term biomass, which is the focus here. In perhumid temperate rainforest systems, where drainage is highly correlated with productivity, changes in nutrient availability and drainage (associated with decreases in soil density in deposit zones) seem likely to encourage more productive forests in landslide susceptible locations [62]. On the other end of the disturbance exposure spectrum, a decline in biomass at high levels of landslide exposure is apparent (using all 1491 plots to allow for the possibility of sparse/non-forest conditions due to wind; S1 Fig). This is trivially true but nonetheless noted,

as cliffs and very steep slopes are consistently moving, precluding forest establishment in many cases.

It is also possible that the drivers of landslide likelihood (steeper slopes) are partially confounding the relationship. Slope is strong driver of biomass at local scales [37], and significantly positively correlated with biomass at this broad scale, though weakly (forested plots only; $p < 0.05$, $r^2 = 0.06$; square root transformed). That landslide susceptibility was retained in each independent test of the relationship (correlations, stepwise variable selection, and random forests) suggests that landslides themselves have different properties from purely slope-induced differences in drainage rates. At sub-regional scales, the effect of removing landslides was muted, with no change in areas where no landslides are possible (flat areas) and declines in C in steeper terrain (Fig 2). A second difficulty in landslide exposure is primarily related to landslide initiation rather than potential deposition. Landslides in SE Alaska are generally confined to steep slopes, even in their depositional area [38], however more intensive modeling of landslide movement downhill and across slopes is necessary to get a better estimate of exposed area.

Overall. The cumulative impact of both disturbances is a slight decrease in tree density and an increase in carbon and BA, based on modeled outcomes where disturbances were removed. The effect of the individual disturbance types partially offset each other at the regional scale, with wind correlated with higher tree densities in some locations countered by lower density, larger trees in other, more landslide exposed areas (Fig 2). In other words, the overall correlation of higher biomass with disturbance exposure is spatially heterogeneous when seen at finer scales. In the central portion of the region, where the topography is lower slope, disturbances (particularly wind) appear to be constraining biomass C, in that removing them from the model results in projected higher C values. In steeper locations, however, removing disturbances generally results in lower C, as those are the areas where landslides appear more significant. It should be noted that landslides are not independent of wind; there are higher landslide probabilities in wind exposed locations. This has been attributed to shaking of the trees during high rain/storm events [39, 48]. This interaction adds further nuance to the disturbance discussion and suggests that treating the individual disturbances separately also requires considering their interactions as a separate mechanistic component of the overall regime. In sum, even within an ecosystem type individual disturbance processes vary in their spatial impacts and therefore in their correlation with baseline C.

This correlation between higher disturbance rates and higher biomass is strikingly different from further south in the seasonal temperate rainforest (where fire is a factor and can impact broad-scale C for centuries; [63]) where disturbance rates maintain lower baseline C than would be expected without disturbance [64]. In that case, suppression of disturbances would be expected to raise C stocks, though unlikely to be sustainable. In this perhumid portion of the temperate rainforest, however, disturbance frequency and extents are some of the lowest of all forested ecoregions in North America (<0.5% per year, [27]). It appears that while there is some finer scale heterogeneity in the effects of increasing disturbance exposure (Fig 2), the net effect is higher biomass in areas of higher exposure, and as a result, higher predicted C stocks.

Limitations

Like most broad-scale regional studies, the results here are correlative rather than necessarily causal. A counter hypothesis would be that disturbance exposure is higher in areas that are also prone to higher biomass forests for other reasons. The mechanisms that would drive higher biomass in these exposed areas independent of disturbance would be likely be aspect (via solar exposure) and slope (via better drainage) for wind and landslide disturbances

respectively. Both aspect and slope were considered explicitly alongside the disturbance variables, and the disturbance exposure variables were retained in the statistical models. Second, aspect and slope are not perfectly correlated with wind and landslides; wind exposure is modified by upwind topography, so there are sheltered southeast facing slopes (for example). Similarly, there are steep slopes with relatively low landslide probability because local topography does not concentrate drainage. Further fine scale work would be valuable, especially dendrochronological or paleoecological studies which could link specific disturbance histories at a point to stand biomass metrics. To our knowledge, there are no long-term (1000+ year) studies on landslide succession, which would be valuable comparisons to existing long-term chronosequences (generally post-glacial, [16]). This is often challenging in this environment due to heart rot making precise dating impossible and the difficulty of ascribing the cause of stand initiation several thousand years later. Nonetheless, the consistent association of higher baseline biomass and higher exposures regardless of recent disturbance history, coupled with undisturbed chronosequences from the region (Glacier Bay) which show substantial biomass declines in undisturbed areas [14], suggests that productivity is enhanced by infrequent but not non-existent disturbance regimes.

Conclusions

The temperate rainforests of southeast Alaska are some of the most biomass-C rich forests in the world and can potentially contain C stores equivalent to 5–7% of the lower 48 forest C stocks in biomass pools alone. Higher C, contained in lower density but larger tree stands, was associated with higher exposure to infrequent disturbance (wind and landslide) processes; areas sheltered from those disturbances had lower values (lower BA, lower biomass). Landslides were more associated with higher biomass C than wind exposure, which was more associated with higher tree densities. The strength of the cumulative action of these two disturbance processes was heterogeneous in space, and at finer scales some areas had lower predicted biomass C. This is consistent with the hypothesis that in very infrequently disturbed systems, occasional mortality events result in more productive stands at the landscape and regional scale. While any disturbed location will lose C as a result of that event in the short term, the overall, broad-scale relationship in this high C density, infrequently disturbed system is higher baseline C in areas of higher wind and landslide exposure.

Supporting information

S1 Fig. Direct correlations between disturbance and biomass. Wind exposure and landslide exposure compared to the biomass variables: Tree density, biomass, and basal area. Red line represents a simple linear regression to show trends.
(PDF)

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Author Contributions

Conceptualization: Brian Buma.

Data curation: Brian Buma, Thomas Thompson.

Formal analysis: Brian Buma.

Funding acquisition: Brian Buma.

Investigation: Brian Buma, Thomas Thompson.

Methodology: Brian Buma.

Project administration: Brian Buma.

Resources: Brian Buma.

Software: Brian Buma.

Validation: Brian Buma, Thomas Thompson.

Visualization: Brian Buma.

Writing – original draft: Brian Buma.

Writing – review & editing: Brian Buma.

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Emergent freeze and fire disturbance dynamics in temperate rainforests

BRIAN BUMA,[1]* ENRIC BATLLORI,[2] SARAH BISBING,[3] ANDRES HOLZ,[4] SARI C. SAUNDERS,[5] ALLISON L. BIDLACK,[6] MEGAN K. CREUTZBURG,[7] DOMINICK A. DELLASALA,[8] DAVE GREGOVICH,[9] PAUL HENNON,[10] JOHN KRAPEK,[11] MAX A. MORITZ[12,13] AND KYLA ZARET[4]

[1]*Department of Integrative Biology, University of Colorado, Denver, 1151 Arapahoe St., Denver, Colorado 80204, USA (Email: brian.buma@ucdenver.edu);* [2]*Universitat Autònoma de Barcelona, Cerdanyola del Vallès, Spain;* [3]*Department of Natural Resources & Environmental Science, University of Nevada – Reno, Reno, Nevada;* [4]*Department of Geography, Portland State University, Portland, Oregon, USA;* [5]*Coast Area Research, BC Ministry of Forests, Lands, Natural Resource Operations, and Rural Development, Nanaimo, British Columbia, Canada;* [6]*Alaska Coastal Rainforest Center, University of Alaska Southeast, Juneau, Alaska;* [7]*Institute for Natural Resources, Oregon State University, Portland;* [8]*Geos Institute, Ashland, Oregon;* [9]*Alaska Department of Fish and Game, Wildlife Conservation Division, Douglas;* [10]*USDA Forest Service, PNW Research Station;* [11]*Juneau Greens, Juneau, Alaska;* [12]*Agriculture and Natural Resources Division, University of California Cooperative Extension; and* [13]*Bren School of Environmental Science & Management, University of California, Santa Barbara, California, USA*

Abstract The coastal temperate rainforests of South and North America are part of the most biomass dense forest biome on the planet. They are also subject to rapid climatic shifts and, subsequently, new disturbance processes – snow loss-driven mortality and the emergence of fire in historically non-fire-exposed areas. Here, we compare and contrast Southern and Northern Hemisphere coastal temperate rainforests of the Americas, two of the largest examples of the biome, via synthesis of current literature, future climate expectations and new downscaling of a global fire model. In terms of snow loss, a rapid decline in winter snow is leading to mass mortality of certain conifer species in the Northern Hemisphere rainforests. High-elevation Southern Hemisphere forests, which are beginning to see similar declines in snow, may be vulnerable in the future, especially bogs and high-water content soils. Southern Hemisphere forests are seeing the invasion of fire as an ecological force at mid-to-high latitudes, a shift not yet observed in the north but which may become more prominent with ongoing climate change. We suggest that research should focus on the flammability of seral vegetation and bogs under future climate scenarios in both regions. By comparing these two drivers of change across similar gradients in the Northern and Southern Hemispheres, this work points to the potential for emerging change in unexpected places in both regions. There is a clear benefit to conceptualising the coastal temperate rainforests of the Americas as two examples of the biome which can inform the other, as change is proceeding in similar directions but at different rates in each region.

Abstract in Spanish is available with online material.

Key words: climate change, coastal temperate rainforest, emerging disturbance regimes, fire, snow loss.

INTRODUCTION

Climate change is affecting global forests in multiple ways, often by altering the abiotic conditions forests experience. Direct effects include increasing water stress and associated drought-induced tree mortality (Adams *et al.* 2009; Holz *et al.* 2017), CO_2 fertilisation (Bolker *et al.* 1995) or lengthening of the growing season (Cleland *et al.* 2007), with these effects leading to altered productivity and/or range shifts (Krapek & Buma 2018). These changes may be punctuated, resulting from the crossing of climatic thresholds that drive major ecological changes related to species physiological tolerances (e.g. Allen *et al.* 2010). The existence of environmental thresholds may also cause shifts in or intensification of disturbance regimes (Brooks *et al.* 2004; Veblen *et al.* 2011; Buma 2015; Millar & Stephenson 2015), which can lead to sudden changes in ecosystem type when disturbance severity or frequency exceeds species, community and ecosystem tolerances (Buma & Wessman 2011). As a result, significant shifts in ecological relationships – such as the establishment of

*Corresponding author.
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novel functional relationships (Gilman *et al.* 2010) or the formation of no-analogue communities (Williams & Jackson 2007) – are widely anticipated.

Precipitation – including its phase, amount, intensity and timing – is one such climate factor expected to potentially drive significant, threshold-like change. The phase of precipitation (snow or rain) represents only a small shift in winter temperatures but results in a very large change in the physical environment that forests inhabit. For example, the loss of a winter snowpack results in decreased soil insulation in winter (Groffman *et al.* 2001), loss of nival habitat (Pauli *et al.* 2013), altered plant communities (Bannister *et al.* 2005) and reduced summer streamflows (e.g. Mote 2003), among other factors. Changes in the amount and timing of precipitation also have major impacts. Several researchers have focused on forest health in relation to precipitation-associated physiological stress (Anderegg *et al.* 2013), winter high flow/summer low flow events (Surfleet & Tullos 2013), and erosion and flooding severity and timing (Klos *et al.* 2014).

Fire is also a well-known driver of rapid threshold-like change. Fire is a major disturbance agent in most of the world's forests (Rundel 1981; Scott 2000), burning ~348 Mha annually (Bowman *et al.* 2009; Giglio *et al.* 2013) and influencing vegetation (He *et al.* 2016), soil (Certini 2014), and a host of other ecological aspects of forest ecosystems. Fire occurrence responds rapidly due to climate or anthropogenic activity (Doerr & Santín 2016). Because of its near ubiquity, fire can be difficult to disentangle from the baseline functioning of the forest (Pausas & Keeley 2009). When excluded via modelling, it is apparent that whole biome distributions likely result from the presence (or absence) of fire (Bond *et al.* 2005). Anticipating changes to fire regimes – especially the emergence of fire in areas where it was not historically present, or rare enough to not be an evolutionary force – is important for management, conservation and prediction of future ecosystem dynamics and functioning.

Both snow loss and fire regime shifts are major concerns to forest managers, conservationists, researchers and culture bearers in forests worldwide. Here, we illustrate how the loss of snow and the emergence of fire may act as important agents of change. We focus on coastal temperate rainforests, a globally important biome that holds an immense amount of carbon in relatively intact forests (Keith *et al.* 2009), where snow loss is proceeding faster than anywhere else globally and where the fire regime is expected to intensify or even emerge in areas where it was previously essentially absent.

Objectives

The objectives of this review are to (i) synthesise and describe the significance of crossing precipitation and fire thresholds to the ecology and functioning of this globally important forest biome; (ii) discuss the spatial pattern of potential change within the regions; and (iii) use the analysis of drivers of the new disturbances in this ecosystem to anticipate new dynamics in other temperate rainforest systems. We hope to draw attention to the benefits of considering the southern and northern coastal temperate rainforest as two regions which can inform each other via direct comparison.

COASTAL TEMPERATE RAINFORESTS: A FRONTIER OF CHANGE

Coastal temperate rainforests (hereafter CTRFs) are globally important as the most carbon-dense forested areas on the planet, containing upwards of 1867 tons C ha^{-1} (Australian CTRFs, Keith *et al.* 2009), a result of generally low rates of decomposition, low water stress, moderate climate and relatively long growing seasons. They provide a multitude of ecosystem services, from significant cultural resources to wildlife habitat, and function as the headwaters of globally significant fisheries (Brandt *et al.* 2014; Rodriguez-Echeverry *et al.* 2018). These regions are also associated with high endemic biodiversity (e.g. South American forests) in terms of nonvascular plants and lichens (DellaSala 2011) and relatively low levels of human development in many places.

Despite occurring across a wide range of latitudes (~30° north to south), CTRFs have relatively consistent, moderate climate conditions due to their close proximity to the ocean (Alaback 1991; DellaSala 2011; Fig. 1). The mild, consistent climatic conditions favour evergreen tree species, fine-scale, infrequent disturbances and generally older, late-successional forests over much of the landscape. Average annual temperatures range between 4 and 12°C, with annual precipitation from as low as 1.5 m to as high as 5 m or more in some areas (DellaSala 2011). The historic disturbance regime was dominated by relatively frequent tectonic activity, landslides, windstorms, fine-scale tree mortality and infrequent fires (Veblen & Alaback 1996; Buma & Barrett 2015; Holz *et al.* 2016). Although observed and projected absolute warming rates in CTRFs are not as high as in polar or high-latitude interior regions, warming temperatures are crossing key climatic and ecological thresholds (Veblen *et al.* 2011; Shanley *et al.* 2015), notably: (i) a phase change from snow to rain as mean winter temperatures cross the 0°C threshold and (ii) the emergence or increased role of fire on the landscape (e.g. see DellaSala *et al.* 2018).

The snow-to-rain transition is an abrupt physical threshold driven directly by temperature. This

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fundamental shift in hydrology influences a variety of ecosystem properties and underlying processes, including subnival habitat (Pauli *et al.* 2013) and snow disturbance dynamics (Hennon *et al.* 2016). Large areas of high-latitude CTRFs are at or near the 0°C isotherm during winter, meaning that precipitation usually falls as snow. This climatological location makes CTRF snow regimes especially vulnerable to a loss of days below freezing given even minimal warming (Meehl *et al.* 2004), as illustrated by the crossing of the snow-to-rain threshold already reported in portions of the biome (Buma 2018).

Historically, fire in CTRFs was very infrequent, although large in extent, at lower latitudes (~40°–50°) but very rare at higher latitudes (>55°) due to climatological constraints, especially the relatively wet summer periods and limited natural ignition sources (Veblen & Alaback 1996; Kitzberger *et al.* 2016). Areas with continuous human habitation had a more frequent fire regime associated with land management (Hoffman *et al.* 2016; Méndez *et al.* 2016), though many mid- to high-latitude locations had fire return intervals >1000–10 000 years (Veblen & Alaback 1996; Gavin *et al.* 2003). At lower latitudes, fire was rare but a significant driver of landscape pattern (e.g. Washington State, USA, Agee 1993; Gavin *et al.* 2007). Paleoecological records from the higher latitude portions of North America's CTRF (>54° N) have recorded essentially no charcoal since approximately 7500 years before present and no widespread fires since the Holocene Climatic Optimum (Baichtal *et al.* 2008). The anticipated general increase in fire activity at temperate latitudes worldwide, where fuel is abundant, is tied to lower moisture availability resulting from predicted higher temperatures, reduced precipitation and/or longer fire season

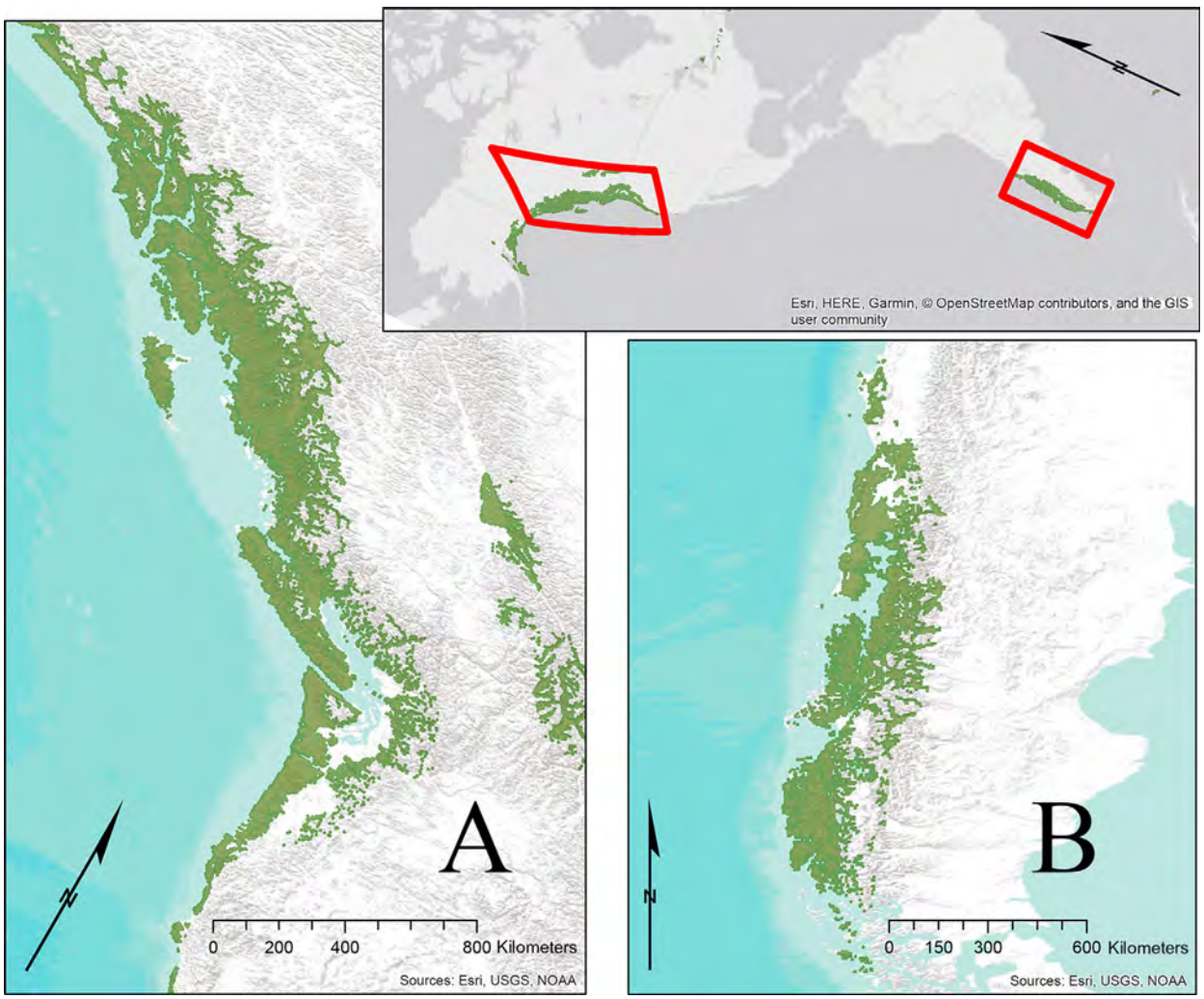

Fig. 1. The North Pacific coastal temperate rainforest (panel a) and the South Pacific CTR (panel b). The focus of the comparison is on the perhumid and seasonal zones of both forests. Biome map from DellaSala (2011). [Colour figure can be viewed at wileyonlinelibrary.com]

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(Westerling *et al.* 2006; Bowman *et al.* 2009; Moritz *et al.* 2012; Abatzoglou *et al.* 2017). In addition, increasing temperatures generate higher water deficits (via increased evapotranspiration) even under potentially increased rainfall at higher latitudes. The lack of historical fire exposure combined with a likely increase in future fire activity makes CTRFs potentially vulnerable to novel change.

In sum, the particular climate and disturbance 'state space' in CTRFs makes them an ideal biome for studying how climate change-induced shifts in winter precipitation phase and the emergence of fire have direct and indirect, long-lasting consequences for ecosystem structure and function. Coastal temperate rainforests can serve as model systems for developing predictions about future changes in other forested regions, which are expected to undergo similar snow-to-rain and fire regime shifts in the future (Bowman *et al.* 2014; Holz *et al.* 2016).

We take advantage of the fact that the two largest examples of the biome, the North Pacific coastal temperate rainforest (NPCTR) of North America and the South Pacific coastal temperate rainforests (SPCTR) of Chilean South America (27.3 and 12.5 million ha, respectively), together represent 50.7% of all global temperate rainforests (DellaSala 2011). We focus on the perhumid and seasonal portions, which straddle the snow–rain and fire-presence/absence thresholds (Fig. 1; Veblen & Alaback 1996). Parallel characteristics result from adjacency to the highly moderating, cool Pacific maritime environment and continuously wet conditions that favour dense, contiguous evergreen forests (needleleaf in the Northern Hemisphere and broadleaf in the Southern Hemisphere) and a large abundance of temperate, peat-accumulating wetlands in areas of poor drainage. Both have high carbon (C) stocks, with slightly more in the Northern than the Southern Hemisphere (NPCTR: 568–794 tons C ha^{-1}, SPCTR: 326–571 tons C ha^{-1}; studies synthesised in Keith *et al.* 2009). Both regions of the CTRF exhibit a gradient of human impacts, from significant development/land cover change at lower latitudes to more intact landscapes at higher latitudes (DellaSala 2011). While the higher latitude areas are certainly impacted by extensive historical logging, often targeting large trees and vulnerable portions of the ecosystem (flood plain/riparian zones; Albert & Schoen 2013) with important negative impacts on fish, wildlife and habitat (Beier *et al.* 2008a), they still retain relatively high value in regard to cultural resources and biodiversity among other ecosystem services (Brandt *et al.* 2014; Rodriguez-Echeverry *et al.* 2018). The long north–south latitudinal extent of these CTRFs means that climate change is asynchronous across this gradient, with poleward regions warming faster and equatorward regions experiencing more rapid precipitation changes (IPCC 2014).

In contrast to many other well-studied biomes experiencing rapid changes in precipitation regimes and fire dynamics (e.g. Serreze *et al.* 2000), CTRFs are useful study systems in that climate change impacts can be examined concurrently and comparatively in both the Northern and Southern Hemispheres. A threshold may be crossed earlier in one region than the other, providing insights into how the slower changing area might respond. Comparative studies can provide a more robust test for hypothesised mechanisms and an indication of potential climate change impacts on local climatology and ecosystem responses to these and to other disturbances (Alaback 1991). We take advantage of that cross-comparison in the following discussion.

LOSS OF SNOW AND EMERGING FREEZE DISTURBANCE

Among the most visible environmental changes associated with a warming climate in these two regions are the reduced depth, extent and persistence of snow as average winter temperatures cross the rain–snow threshold of 0°C (Fig. 2). In CTRFs, where precipitation is ample year-round, precipitation phase and subsequent routing of runoff play important roles in forest functioning (Bisbing *et al.* 2016); this change in phase may be a more significant shift than any absolute change in the overall amount or seasonality. Because the CTRF regions have historically occurred along the 0°C isotherm in winter, the number of snow-covered days, there is decreasing faster than in any other biome (Meehl *et al.* 2004). Currently, the rain–snow boundary bisects the NPCTR, starting at approximately 2000 m by 50° N and reaching sea level around 57° N (Shanley *et al.* 2015). The more maritime southern SPCTR in Patagonia is generally already above this threshold at low elevations, and snow is transient in those areas, though areas at higher latitudes and elevations do remain snow and ice covered. While, to our knowledge, no research has been done on long-term changes in snow persistence in those higher elevation areas of the SPCTR, satellite-based observations in the Andes just north of the region (from ~28° to 36.5° S) identified significant declines in duration of snow persistence, approaching 2–3% per year, with the rate of snow loss highest at higher latitudes (Saavedra *et al.* 2018). Additionally, various ice caps in the region have been retreating rapidly over the last several decades, attributed to warming and subsequent raising of the 0°C isotherm (Davies & Glasser 2012), and this is expected to continue over the next decades (Fig. 2). Thus, research of climate

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change effects on the rain–snow threshold in the NPCTR may be instructive to the less studied snow dynamic SPCTR.

Observed effects of snow loss

The crossing of the rain–snow threshold can have significant ecological effects, especially among species adapted to reliable winter snow environments. In the NPCTR, yellow-cedar (*Callitropsis nootkatensis*) is experiencing extensive mortality over 9° of latitude (>400 000 ha; Buma *et al.* 2017). This species has a competitive strategy of fine root growth in the early spring when supplies of nitrogen are abundant in upper soil layers (Hennon *et al.* 2016); however, fine root death can occur during subfreezing weather events when snow is no longer present (Hennon *et al.* 2016). Snow is an effective insulator for soils, buffering soil temperatures from atmospheric variability; a lack of snow generally leads to colder soils in winter months (Groffman *et al.* 2001) and an overall increase in soil temperature variability (Jungqvist *et al.* 2014). Even in a warming climate with less snow, sporadic cold weather events have persisted in portions of the NPCTR (Beier *et al.* 2008b; Buma 2018), driving continued tree mortality. Ongoing mortality is likely to lead to shifts in community composition to a smaller suite of species more tolerant of snow-free winter conditions (Oakes *et al.* 2014). But, tree mortality is not the only effect of increasing soil freezing. Mobilisation of contaminants (Mohanty

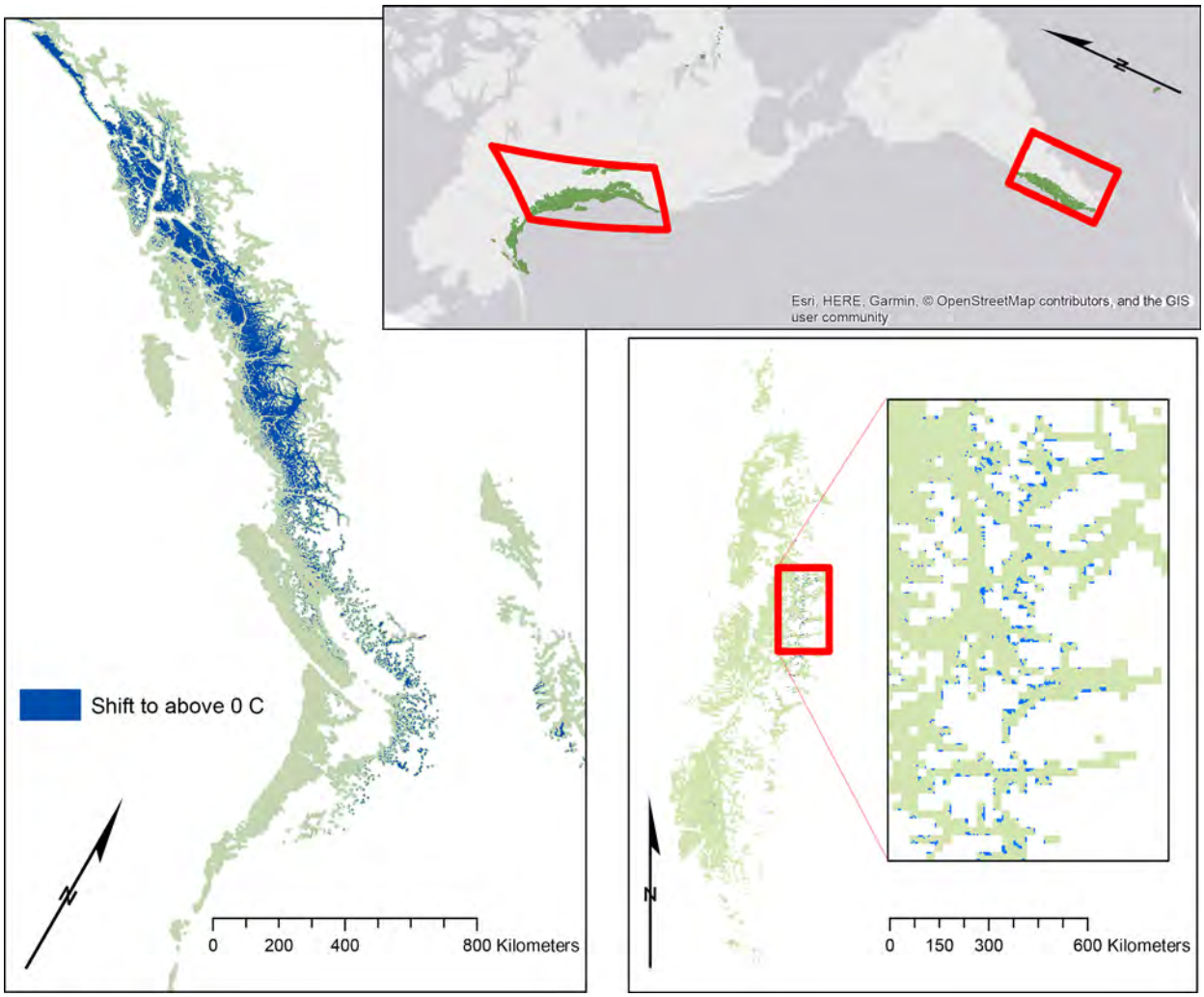

Fig. 2. Anticipated shifts in winter snow threshold in the South Pacific CTR (SPCTR) and North Pacific coastal temperate rainforest (NPCTR) using the HadGEM2-ES (RCP 8.5) climate model/emission scenario. No areas are expected to shift from above to below freezing. Due to substantial lower elevation/slope areas in the NPCTR, the spatial extent of change is large. In the SPCTR, shifts are likely along the higher altitudinal range edge throughout the forest, and the inset shows a subsection in detail to illustrate this pattern. Climate data from Hijmans *et al.* (2005) at 1-km resolution, HadGEM2-ES GCM. [Colour figure can be viewed at wileyonlinelibrary.com]

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et al. 2014) and microbial communities (Larsen *et al.* 2002) can drive significant changes to nutrient cycling (Fitzhugh *et al.* 2001; Urakawa *et al.* 2014). These changes, triggered by increasing freeze–thaw dynamics in soils, may have significant downstream effects as well.

There are multiple pathways by which snow loss might cause continued plant mortality in the NPCTR and at higher elevations in SPCTR forests. Climate change is altering the phenology of forest species, leading to increased risk of cold-related damage in a warmer world (Gu *et al.* 2008; Rigby & Porporato 2008). The general ecological strategy of early spring activity as a means to gain competitive advantage is widespread (Polgar & Primack 2011). Apart from root freezing, warmer temperatures speed plant development earlier each year, making them vulnerable to frost (Gu *et al.* 2008). Increased cold damage associated with climate warming and earlier spring phenological development is well recognised (Gu *et al.* 2008; Inouye 2008) and is typically associated with aboveground bud mortality. Broadscale damage to sensitive bud tissue has already been noted in a variety of locations (Inouye 2008). The threat of root freezing due to a lack of snow has not been generally quantified outside of NPCTR, but it is expected to be a significant factor in areas where snow cover will shift from continuous to transient in temperate zones (Bannister *et al.* 2005). The risk would be highest in areas of the landscape prone to shallow rooting habits (e.g. wetlands and bogs). The most vulnerable species are likely to be those adapted to early-onset seasonal growth that historically occurred under reliable cover of snow.

It is possible that subfreezing-induced damages could decline after the transitional period, when temperatures rise above the freeze–thaw boundary (Henry 2008, Buma 2018), but this is dependent on freezing probability corresponding to average temperatures as they have in the past. Given the topography of both regions, with significantly colder areas located in close geographic proximity to these ecosystems (on the eastern sides of the Andes in South America and Coast Mountains of the United States and Canada), it is unclear whether that relationship will hold (Beier *et al.* 2008b). Areas of historically thin snowpack and winter temperature slightly below freezing in the SPCTR should be monitored for plant stress, root mortality and other emerging dynamics suggested by the NPCTR decline.

THE EXPANSION OF FIRE

The combination of decreasing winter snowpack resulting from precipitation phase change, earlier snowmelt and thaw, and increasing spring–summer evaporative demand (e.g. vapour pressure deficit) over longer, rainfall-free growing seasons is likely to result in an increase in the prevalence of fire in certain portions of CTRFs (Westerling *et al.* 2006; Littell *et al.* 2010; Moritz *et al.* 2012). As noted, the historic importance of fire in CTRFs has been relatively low, ranging from a nearly nonexistent fire regime at higher latitudes (Gavin *et al.* 2003) to infrequent, though stand-replacing, fires at lower latitudes (Agee 1993; Veblen & Alaback 1996; Holz *et al.* 2012; Walsh *et al.* 2015; Hoffman *et al.* 2016; Fig. 3). Although there is a substantial amount of highly contiguous vegetation that may act as fuel for a fire in CTRFs, vegetation is typically too wet to burn, and natural ignitions are very infrequent, as lighting is rare and generally accompanied by rain. Overall, fire across both CTRF regions has historically been driven by the confluence of atmospheric circulation patterns cycling at multiple temporal scales (Whitlock *et al.* 2008; Littell *et al.* 2010; Holz *et al.* 2017) and human presence (Hoffman *et al.* 2016; Méndez *et al.* 2016).

Under climate change projections, the poleward portions (>55 °N and >50° S) of both the NPCTR and the SPCTR are expected to experience significant annual warming, lower snowpack, a potential increase in spring–summer drought (Veblen *et al.* 2011) and increasing moisture deficits (40–50% at ~60°N; Haufler 2010). As a result, the equatorward portions of CTRFs are expected to become more flammable throughout the fire season (e.g. summer; Fig. 4), leading to more flammable conditions (Littell *et al.* 2010; Sheehan *et al.* 2015). Additionally, warmer coastal ocean temperatures are likely to create the potential for more ignitions via increased lightning activity (Garreaud *et al.* 2014). In several of the drier and mountainous areas in the SPCTR, increases in lightning-set fires have been observed in recent decades (Veblen *et al.* 2011), and increases in the frequency of dry, warm periods have been linked to global climate warming and ozone depletion in Antarctica (Holz & Veblen 2011; Holz *et al.* 2017). Recent research has identified the southern SPCTR as an area where wildfires might 'invade' as soon as ~2039 if ignitions are provided (Moritz *et al.* 2012).

Downscaled fire projections

Unlike global data related to the winter snow–rain temperature threshold (e.g. Meehl *et al.* 2004; Buma *et al.* 2017), comparable, high spatial-resolution fire modelling studies do not currently exist for northern and southern CTRFs. For the purposes of this synthesis, we used the global modelling framework of Moritz *et al.* (2012) to evaluate changes in climate-driven probability of fire across these regions. This framework integrates global fire datasets (NASA MODIS missions) and environmental covariates

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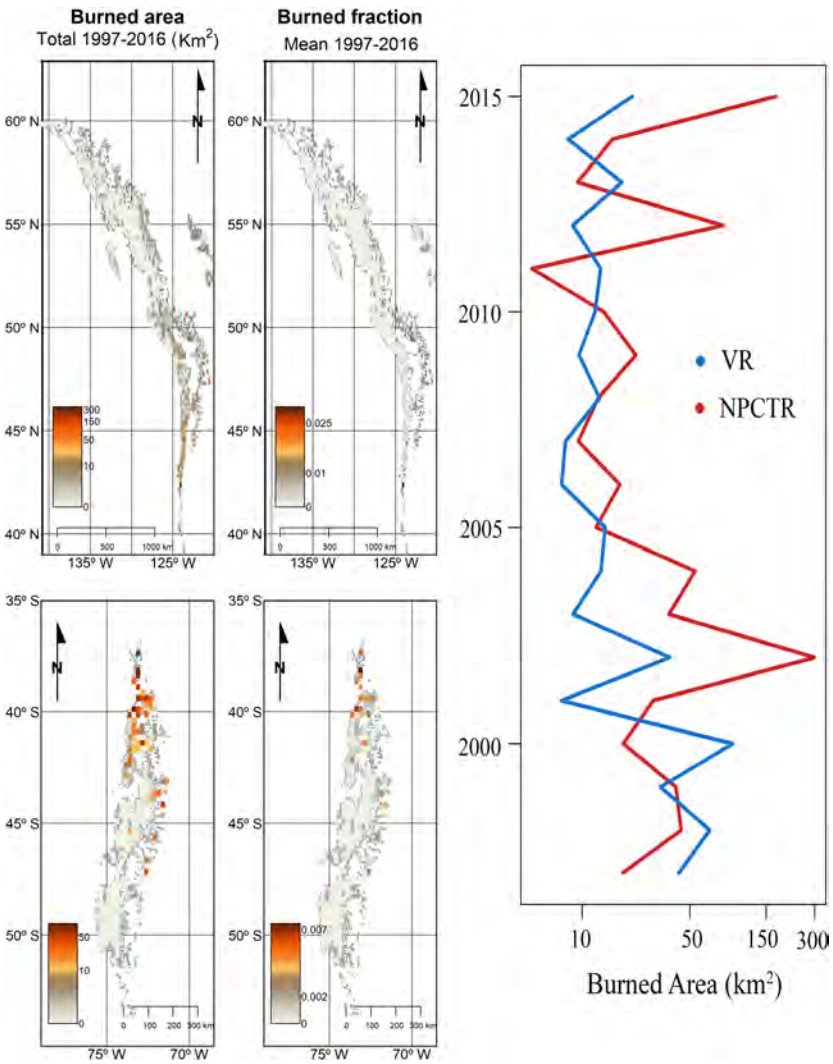

Fig. 3. Yearly burned area, cumulative burned area (per 0.25^0) and per cent burned (mean fraction of each pixel burned per 0.25^0) from 1997 to 2016 in the South Pacific CTR and North Pacific coastal temperate rainforest. Fire statistics come from the Global Fire Emissions Database version 4 (Giglio *et al.* 2013). [Colour figure can be viewed at wileyonlinelibrary.com]

representing fire-conducive climate conditions over the reference period 1971–2000 to determine fire–climate relationships and assess the likelihood of fire under ongoing climate change. We used the framework to build spatial statistical models of relative changes to fire probability that describe the long-term potential of fire occurrence over the period 2071–2100. Using methods based on Moritz *et al.* (2012), we used projected changes in temperature seasonality, precipitation of the driest month and annual precipitation to describe potential alterations in the probability of fire over 0.5° latitudinal bands across CTRFs (Fig. 5). Projected future climate data were obtained from the WorldClim CMIP5 HadGEM2-ES (RCP 8.5; Hijmans *et al.* 2005, updated to CMIP5 in 2017) for the study region (for full details, see Appendix S1–S3).

Because the scope of this synthesis is focused on relative changes, modelling was limited to the Hadley climate projection. The Hadley model performed the best in aggregate when compared to climate-precipitation values for five other GCM's at the seasonal level for the North Pacific forest region (SNAP 2009). To check the assumption that the climate–fire relationships quantified in Moritz *et al.* (2012) would remain valid with the Hadley climatic dataset, correlations between the WorldClim CMIP5 HadGEM2-ES data (used here) and CMIP3 data used in Moritz *et al.* (2012) were computed. We found very high Spearman rank coefficients for the variables used in the model (>0.9; Appendix S3) and considered the climate–fire relationships from the Moritz *et al.* (2012) ensemble fire model suitable to assess potential relative alterations in the CTRFs' likelihood of

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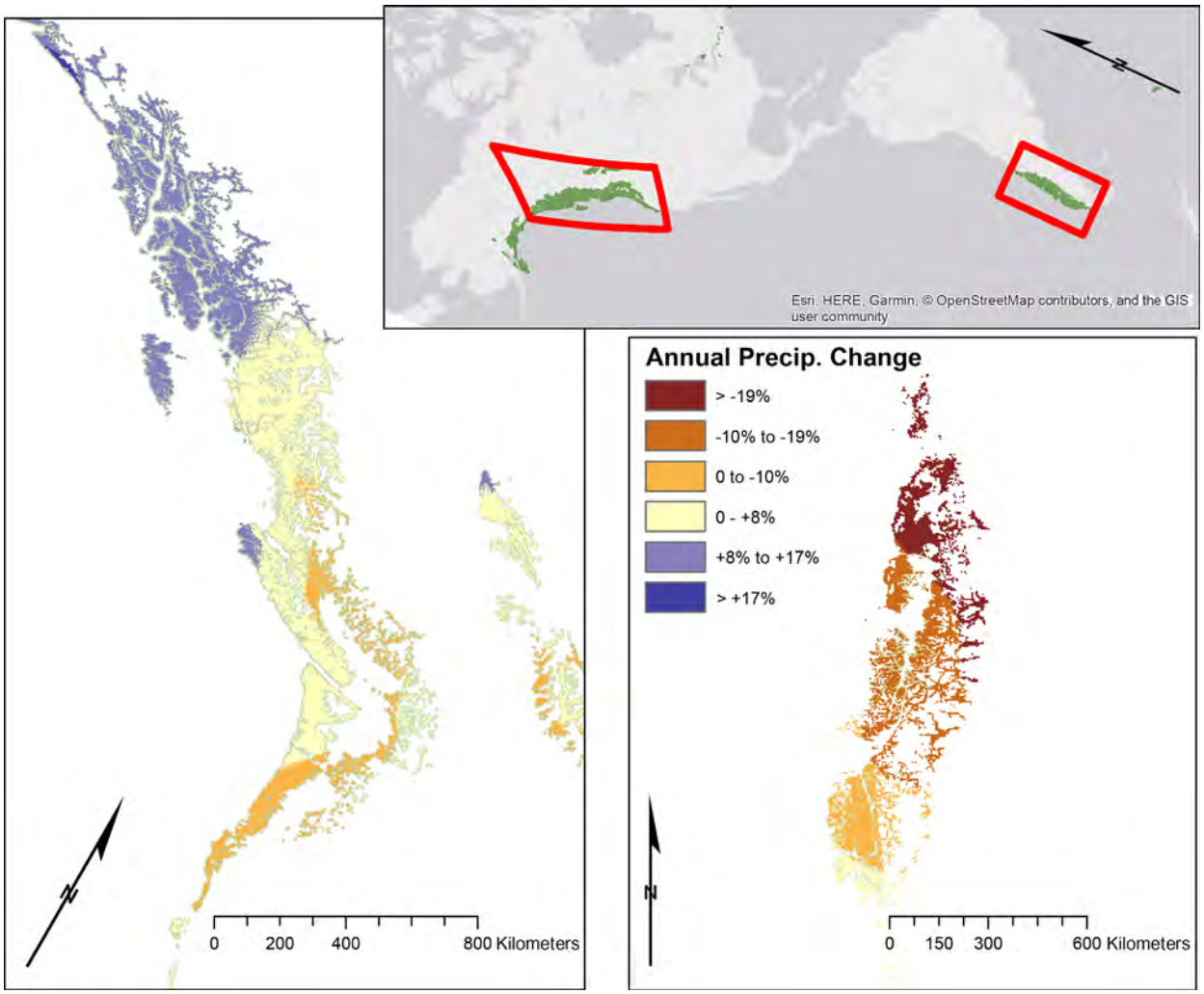

Fig. 4. Projected precipitation change. Relative change in summer precipitation (defined as warmest quarter of the year) by 2070 for the North Pacific coastal temperate rainforest (left) and the South Pacific CTR (right) using the HadGEM2-ES (RCP 8.5) climate model/emission scenario. Climate data from Hijmans *et al.* (2005) at 1-km resolution, HadGEM2-ES GCM (updated to CMIP5, see http://www.worldclim.org). [Colour figure can be viewed at wileyonlinelibrary.com]

fire for the period 2071–2100 driven by HadGEM2-ES climate projections. The purpose of this exercise was not to predict absolute changes in fire probability but rather to identify the portions of each region likely to see the largest relative change. For further details associated with this modelling methodology, see Appendix S1 and Moritz *et al.* (2012).

While the frequency of dry, warm conditions increases at low latitudes in both regions (where the majority of fire modelling work has been concentrated, for example Littell *et al.* 2010), the modelling results suggest that the central latitude portions of the biome in both hemispheres will see the largest relative increase in climate-driven fire probability. This is primarily due to projected changes in dry season precipitation (Fig. 5). The more equatorward portions of the biome will likely have higher rates of fire due to their higher baseline rates. However, a larger relative increase in fire activity indicates a more substantial departure from historical norms and is thus worth noting for future estimates of CTRF dynamics.

It should also be noted that these models are based on broadscale climatic trends and reflect general climate–fire relationships; the actual occurrence and behaviour of fires at regional and local scales is a result of finer scale weather patterns, topo-edaphic gradients and vegetation-fire feedbacks as well.

Effects of fire regime changes in the NPCTR

Fires have occurred historically in the southern and central portions of the NPCTR but with highly

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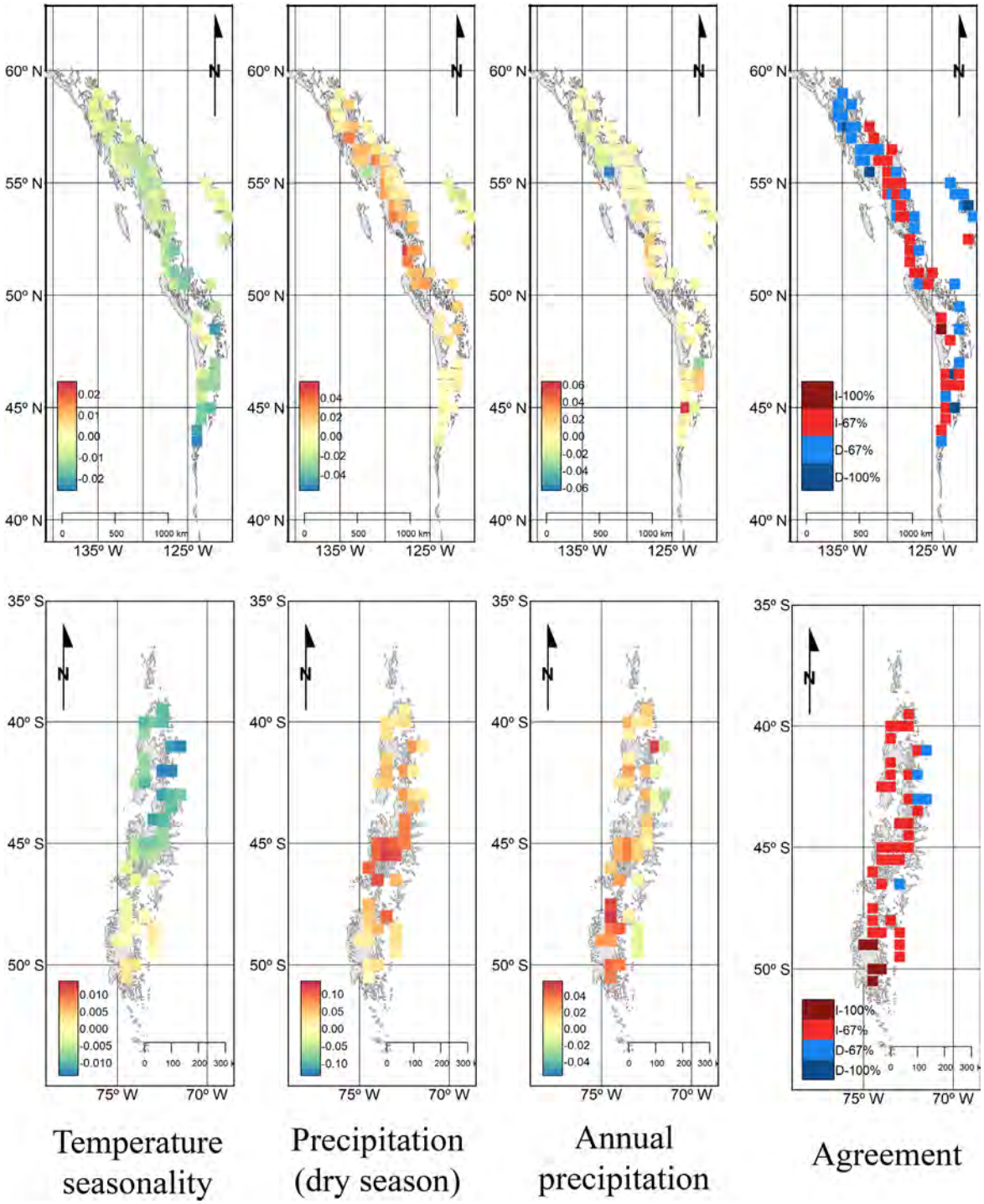

Fig. 5. Relative change in fire probability projected as a function of the major climate drivers in the North Pacific coastal temperate rainforest (NPCTR; top) and the South Pacific CTR (SPCTR; bottom) using the 2070 HadGEM2-ES (RCP 8.5) climate model/emission scenario. A negative (green or blue) value indicates the driver is expected to change in a direction that reduces relative fire probability, and a positive value (orange or red) indicates an increase in fire probability. Agreement between predictors: 100% indicates the three drivers had similar signs (i.e. all were positive), whereas 67% indicates that two-thirds were in agreement for either an increase (I) or a decrease (D) in fire probability. In general, there is strong agreement that fire frequency will increase throughout the SPCTR. Expectations are mixed in the NPCTR, but generally an increase is expected further north than historical fires. The extreme southern portion of the NPCTR is not modelled due to a lack of climate data at the proper scale; for climate locations used to create the probability graphs, see Fig. S1. Note differences of scale of axes and legends. [Colour figure can be viewed at wileyonlinelibrary.com]

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variable return intervals (300–3000+ years, Fig. 4) and spatial heterogeneity due to variable ignitions and climatic conditions (Hoffman *et al.* 2018). Thin-barked, non-serotinous species dominate CTRFs, and fire-adapted species are generally absent (Veblen & Alaback 1996). In the southern NPCTR (<55° N), where fire has an infrequent but more significant role (Agee 1993; Tepley *et al.* 2013; Whitlock *et al.* 2014), a few seral thick-barked species occur (e.g. Douglas-fir, *Pseudotsuga menziesii*) that can survive low-to-moderate fire intensities (Agee 1993).

Anticipating the effects of both increasing and emerging fire (at lower and higher latitudes, respectively) is critical (Littell *et al.* 2010). Theory suggests that in wet systems, where flammability declines as forest structure develops, the introduction of fire-conducive conditions can lead to a cycle of increasing fire extents and subsequent widespread ecological changes. This positive feedback occurs because increases in the spatial extent and connectivity of the more flammable, early seral vegetation after each fire event lead, subsequently, to more extensive fires. Given sufficient ignition opportunities and a climate conducive to periodic fire (Perry *et al.* 2012), rapid and persistent threshold-like changes can occur when forests at the landscape scale cross critical flammable-connectivity thresholds and any fire event is likely to spread over the majority of the landscape. Increased NPCTR flammability associated with early successional species can lead to positive fire-vegetation feedbacks (Agee & Huff 1987) due to highly flammable early successional fine fuels that dry rapidly even in the relatively short fire season common to the region's climate. Thus, the emergence or acceleration of fire regimes is a significant concern and changes projected here should be considered conservative estimates. Recent events in mesic forest stands in the southern portions of the NPCTR (e.g. Eagle Creek Fire in the Columbia Gorge Scenic Area in Oregon and Norse Peak Fire in Washington, from a human ignition) remind us that the transformation of these forests by fire is likely as a potential result of the warming regional climate. To this point, the historical and already-underway expansion of fire into the SPCTR is instructive for the NPCTR.

The expansion of fire in the SPCTR

The expansion of fire is well documented in the SPCTR. Historically, humans were the ignition source for most SPCTR fires (Holz *et al.* 2016), with ignitions occurring primarily at the warmest and driest equatorward edge. Since the 1970s, there has been an increase in lightning-ignited fires in the northern and central regions of the SPCTR (Veblen *et al.* 2011), attributed to environmental shifts associated with climate change (Thompson *et al.* 2011, IPCC 2014, Garreaud *et al.* 2014). As biomass is not limiting and the climate is becoming more conducive to fire, the occurrence of fire is increasingly ignition limited (Paritsis *et al.* 2013). Emerging fires tied to climate change and increasing variability in climate have already been transforming ecological composition, structure and function, particularly in fire-sensitive *Pilgerodendron* forests (Holz & Veblen 2009; Bannister *et al.* 2012). These species are mostly fire sensitive, and the landscape has been relatively nonflammable historically – but is becoming less so today.

In addition to climate change driving increases in fire probability, tree plantations in the SPCTR have the potential to increase fire risk. Pine and eucalypts plantations, like other plantations in the southern NPCTR (Zald & Dunn 2018), facilitate fire spread due to homogeneous patch structure and connectivity, which in turn can result in higher fire frequency and severity (McWethy *et al.* 2018; Paritsis *et al.* 2018). While many historical and recent large fires in Chile occurred just north of the CTRF region in the more Mediterranean central valley, there are plantations of Eucalypts in the SPCTR as far south as 42° on Isla Chiloé, and at the Patagonian dry forest/steppe ecotone lodgepole pine (*Pinus contorta* var. *latifolia*), a fire-associated species has established from plantations even further south (at least 45.5°; Taylor *et al.* 2017). There are concerns that invasion by fire-adapted species may alter the water balance, fuel type (Tng *et al.* 2012) and fuel structures (Cobar-Carranza *et al.* 2014) of the region.

When fires do occur, they can cause significant changes to forests and have the potential to initiate positive feedbacks that drive further increases in fire frequency (e.g. Paritsis *et al.* 2013; Taylor *et al.* 2017). In the SPCTR, invasion of shade intolerant *Sphagnum* species can result in subsequent waterlogging of the habitat due to an overall decline in evapotranspiration (Díaz *et al.* 2007). *Sphagnum* mosses, which dominate wetlands in the NPCTR as well, acidify substrates, outcompete tree seedlings, transform the plant community and potentially lock the system into an alternative stable state (Kitzberger *et al.* 2016; Zaret & Holz 2016). Preliminary results suggest that: (i) water table and substrate interact and best explain patterns in post-fire tree seedling abundance and plant community; (ii) small seedling abundance is best explained by water-table height and plant community type (Zaret & Holz 2016); and (iii) fine-fuel fibric peats are more likely to support a high-frequency, low-severity fire regime given an amenable climate for fire resulting in a positive feedback between fire and vegetation flammability (Holz 2009, Kitzberger *et al.* 2016).

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This observation of a rapid ecosystem shift triggered by the emergence of fire and explained by the expansion of more flammable, early seral vegetation coupled with increasingly favourable climates for fire is clearly instructive to the NPCTR. It also echoes observations from paleoecological reconstructions in temperate New Zealand (Perry *et al.* 2012) and recent events in Tasmanian temperate rainforests, where unprecedented fires in 2015 and 2016 burned fire-sensitive trees as well as vast tracks of peatland and alpine vegetation (~105 000 ha) during the driest season on record (Marris 2016). That the largest relative change in fire probability is not anticipated at the equatorward edge of the biome, but rather in the middle and at higher latitudes, is an unexpected finding and suggests research should consider emerging fire regimes beyond the drier, equatorward extents typically considered where adaptations to fire are minimal and resilience potentially low. The story of fire emergence clearly illustrates the value of linking the NPCTR and the SPCTR in a single analysis, as physical drivers are changing in similar directions, but at different times, in both areas – the emergence of fire in the south is a valuable case study for the forests in the north.

THE EMERGING BIOME EDGE

Much of the concern regarding climate change-driven mortality centres on changes at the trailing (lower latitude or elevation; Parmesan & Yohe 2003) or leading edges (higher latitude or higher elevation; Mason *et al.* 2015) of species and biome distributions. Generally, discussion in temperate regions has focused on increasing temperatures and declining precipitation on the southern boundary, as both are known to strongly structure species range edges and biome extents. At broad scales, species and biomes do generally track long-term climate conditions such as mean winter temperatures or annual water balance, though there may be lag after major climatic shifts (e.g. Ice Ages) due to slow migration rates (Krapek & Buma 2018). In the future, however, there is the potential for thresholds in species' tolerances to be crossed elsewhere within the current range of a forest ecosystem, because climate warming and precipitation changes are not synchronous and occur at different rates. 'Edges' of climatic tolerance may emerge within central portions of a range due to the intersection of climatic trends with important physical thresholds or biological tolerances.

The NPCTR and SPCTR regions demonstrate significant, climate change-driven changes occurring near the geographic middle of a biome – not just on the lower latitude portions. Loss of snow is causing the most significant ecological changes in the geographic middle of the NPCTR, where snow was reliably present but winter mean temperatures were near 0° (Buma *et al.* 2017). Higher elevations of the SPCTR and surrounding ecosystems are likely to be similarly susceptible in the future, as snow loss (although minimal in absolute terms) is reported in the central portion (Fig. 2). Further research on the role of snowpack changes in determining species ranges is a significant need (Pauli *et al.* 2013), either via direct mortality as in the case of yellow-cedar or via interactions with other stressors and disturbance agents (e.g. Poulos 2014). Similarly, the highest increase in relative fire likelihood is in the geographic middle latitudes of the biome in both hemispheres, where fire was historically rare and species are not well adapted to fire (this analysis, Fig. 5). Fire may emerge in unexpected locations, and the emergence – rather than simple intensification – is also a major research need, especially in landscapes where theory suggests rapid transformations due to seral changes in flammability.

Temperate rainforests, by virtue of their long latitudinal extent, exemplify these emerging edge phenomena at both the species and ecosystem levels. This suggests that climate change monitoring in a variety of regions should focus not only on leading or lagging edges, but also emerging edges driven by climatic shifts like precipitation phase.

CONCLUSIONS

The objectives of this synthesis and review were to draw attention to emerging disturbance phenomena in the coastal temperate rainforests of the Pacific Coast, discuss the causes and effects of those phenomena and utilise the cross-hemispheric comparison to enable more general predictions about change than can be done from single-system studies. This comparative examination of climate–ecosystem relationships across hemispheres of the CTRFs provides a framework within which to hypothesise the nature, geographic location and potential effects of emergent disturbances within similar systems.

The counter-intuitive nature of the processes being observed – root freezing due to warming and fire in wet forests – makes their prediction more difficult but the broad nature of the changes underway underlines the significance of these emergent trends. In particular, the sensitivity of forests to root freezing mortality in areas where snow will become transient should be investigated in other systems, especially those prone to late spring cold events. Emergence of fire, or increases in fire frequency and/or intensity, may constrain survival and self-replacement of dominant species, leading to long-lasting shifts in community composition or landscape structure, or the establishment of alternative ecosystem states. Both are

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resulting in community simplification by acting as filters – selective removal of freeze-susceptible trees and selection for more fire-tolerant species; longer term implications of this shift clearly need more research.

Temperate rainforests are undergoing novel change largely driven by climate change. Change in these systems is particularly important given the two regions we examined are well recognised for their global biodiversity importance and relative intactness, which provides opportunities to proactively respond to emerging conditions relative to highly disturbed areas. Their role as major storehouses of carbon at the global scale underlines the importance of these shifts. Finally, the use of coastal temperate rainforests as early indicators of change is valuable and can lead to predictive capabilities for similar functional groups and responses in forests elsewhere.

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SUPPORTING INFORMATION

Additional supporting information may/can be found online in the supporting information tab for this article.

Appendix S1. Fire modelling.

Appendix S2. Red dots depict the *x–y* coordinates used to extract the values of each bioclimatic variable in each region.

Appendix S3. Spearman rank correlation coefficient between WorldClim CMIP5 HadGEM2-ES (RCP8.5; Hijmans et al. 2005) and the CMIP3 climate projections used to build the ensemble fire models in the Moritz et al. (2012) global fire framework.

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JOURNAL ARTICLE

Using Radar to Estimate Populations and Assess Habitat Associations of Marbled Murrelets

Alan E. Burger

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USING RADAR TO ESTIMATE POPULATIONS AND ASSESS HABITAT ASSOCIATIONS OF MARBLED MURRELETS

ALAN E. BURGER,[1] Department of Biology, University of Victoria, Victoria, British Columbia, V8W 3N5, Canada

Abstract: I used high-frequency marine radar to count marbled murrelets (*Brachyramphus marmoratus*) entering 20 watersheds in Clayoquot Sound, Vancouver Island, British Columbia, Canada, in 1996–1998. My goal was to develop standard protocols for radar inventory and to explain landscape-level habitat associations of this threatened species. Dawn counts were consistently higher and less variable than dusk counts, but both sampling periods produced similar rankings of watersheds and proportionate numbers of murrelets. Most dawn surveys showed a unimodal pre-sunrise pulse of incoming murrelets, but a few dawn surveys showed post-sunrise pulses, likely caused by repeat visits by some birds. These post-sunrise pulses, although rare, inflated estimates of incoming murrelets and were avoided by restricting analyses to pre-sunrise counts. Dawn and dusk counts were higher on cloudy days (≥80% cloud cover) than on clear days, but among cloudy days there was no additional effect on counts caused by precipitation (thick fog or drizzle). Numbers of murrelets entering watersheds varied seasonally, reflecting the breeding chronology, but counts restricted to the core period covering incubation and chick-rearing (mid-May through mid-Jul) showed no significant seasonal effects. Counts varied among years at some stations, but when all stations were considered together, no significant inter-annual variation occurred. Murrelets sometimes flew over low ridges (200–600 m), taking shortcuts into watersheds or crossing from 1 watershed into another. I therefore adjusted the boundaries of some inland catchment areas (based on topography and likely flight paths) to match correctly counts made at the watershed mouths with the appropriate inland catchment area. Radar counts at 18 watersheds were significantly correlated with total watershed area, areas of mature (>140 year old) forest, and—most strongly—with areas of mature forest below 600 m. Logging produced negative impacts. Three of the 5 watersheds with extensive logging of low-elevation forest had fewer murrelets per area than unlogged watersheds or those that were <10% logged, but these differences disappeared once remaining low-elevation mature forests were considered. With the removal of old-growth forests, murrelets evidently moved elsewhere and did not pack into the remaining old-growth patches in higher densities.

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Key words: Alcidae, *Brachyramphus marmoratus*, British Columbia, census method, habitat, logging impacts, marbled murrelet, populations, radar.

The marbled murrelet is listed as threatened in both Canada and the United States. Loss of nesting habitat in old-growth coniferous forest is the primary reason for its decline (Ralph et al. 1995). Nests are cryptic and difficult to locate, and adults commute between interior nest sites primarily during dawn and dusk twilight. Consequently, it is impossible to assess stand and watershed populations from visual counts of nests or adults.

High-frequency radar (Cooper et al. 1991) has been found to be an effective tool for locating and counting murrelets as they cross from the ocean to the forest (Burger 1997, Cooper et al. 2001). No other method allows counts of murrelets entering large landscape units. Such counts are valuable in estimating local and regional populations, comparing watersheds for land-use management, assessing macrohabitat preferences of the murrelets, and tracking long-term changes in local populations. Radar inventory is, however, a new technique, and the variability in counts caused by flight behavior; weather; and diurnal, seasonal, and annual variations are poorly known (Cooper et al. 2001).

I address these issues using counts made at 20 watersheds over 3 breeding seasons. One goal of this article is to determine the most appropriate protocol for counting murrelets with radar. To this end, I examined diurnal, seasonal, and annual variations in murrelet counts; the effects of weather on radar counts and murrelet behavior; and the species and numbers of other birds likely to be confused with murrelets on radar. I also compared murrelet counts per watershed with landscape-level habitat characteristics of the watersheds and investigated the effects of clearcut logging on watershed populations.

STUDY AREA

I conducted my study from 1996–1998 in Clayoquot Sound, southwest Vancouver Island,

[1] E-mail: aburger@uvic.ca

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JOURNAL ARTICLE

Effect of Road Traffic on Two Amphibian Species of Differing Vagility

Laurie W. Carr and Lenore Fahrig

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Effect of Road Traffic on Two Amphibian Species of Differing Vagility

LAURIE W. CARR AND LENORE FAHRIG

Ottawa-Carleton Institute of Biology, Carleton University, 1125 Colonel By Drive, Ottawa, Ontario K1S 5B6, Canada, email lfahrig@ccs.carleton.ca

Abstract: *Vehicular traffic can be a major source of mortality for some species. Highly vagile organisms may be at a disadvantage in landscapes with roads because they are more likely to encounter roads and incur traffic mortality. To test this prediction, we assessed the population abundance of two anuran species of differing vagility, the leopard frog (*Rana pipiens*, more vagile) and the green frog (*Rana clamitans*, less vagile), at 30 breeding ponds. Traffic density, an index of the amount of potential traffic mortality, was measured in concentric circles radiating from the ponds out to 5 km. We conducted multiple linear regressions relating population abundance to traffic density, pond variables, and landscape habitat variables and found that leopard frog population density was negatively affected by traffic density within a radius of 1.5 km. There was no evidence that the presence of vehicular traffic affected green frog populations. These results suggest that traffic mortality can cause population declines and that more vagile species may be more vulnerable to road mortality than less vagile species.*

Efectos del Tráfico Carretero sobre Dos Especies de Anfibios de Diferente Movilidad

Resumen: *El tráfico vehicular puede ser une causa importante de mortalidad para algunas especies. Organismos altamente móviles pueden estar en desventaja en paisajes con carreteras, debido a que su probabilidad de encontrar carreteras e incurrir en mortalidades por tráfico es mayor. Para probar esta predicción, evaluamos la abundancia de dos especies de anuros de diferente movilidad, la rana leopardo (*Rana pipiens*, más móvil) y la rana verde (*Rana clamitans*, menos móvil), en tres estanques de reproducción. La densidad del tráfico como un índice de la cantidad potencial de mortalidad por tráfico fue medida en círculos concéntricos radiando de los estanques hacia afuera y hasta los 5 km. Llevamos a cabo regresiones lineales múltiples relacionando la abundancia poblacional con la densidad del tráfico, las variables del estanque y las variables del hábitat del paisaje. Encontramos que la población de rana leopardo estuvo negativamente afectada por la densidad del tráfico dentro de un radio de 1.5 km. No hubo evidencias de que la presencia de tráfico vehicular afectó a las poblaciones de rana verde. Estos resultados sugieren que la mortalidad por tráfico puede ocasionar declinaciones poblacionales y que las especies más móviles pueden ser más vulnerables a la mortalidad carretera que las especies menos móviles.*

Introduction

The presence of vehicular traffic and the continuous, linear nature of roads can be a significant physical barrier to movement for many species (small mammals: Oxley et al. 1974; Garland & Bradley 1984; Mader 1984; Merriam et al. 1989; snails: Baur & Baur 1990; arthropods: Mader et al. 1990) and a major source of mortality. For example, over 2 years, 32,000 amphibians, reptiles, birds, and mammals were found as roadkill on the 3.6-km Long Point causeway adjacent to Big Creek Wetland, Lake Erie, Canada (Ashley & Robinson 1996). Ehmann and Cogger (1985) estimate that road traffic kills 5.48 million reptiles and frogs in Australia each year. The loss of individuals due to traffic mortality can have an effect on two levels: reduced population sizes and reduced movement between complementary resources and conspecific populations (Carr et al. 2000). As barriers to movement, roads create smaller patches and increase patch isolation. Smaller

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46 Fed. Reg. 18026 (March 23, 1981)
As amended (1986)

COUNCIL ON ENVIRONMENTAL QUALITY

Executive Office of the President

Memorandum to Agencies:

Forty Most Asked Questions Concerning
CEQ's National Environmental Policy Act Regulations

SUMMARY: The Council on Environmental Quality, as part of its oversight of implementation of the National Environmental Policy Act, held meetings in the ten Federal regions with Federal, State, and local officials to discuss administration of the implementing regulations. The forty most asked questions were compiled in a memorandum to agencies for the information of relevant officials. In order efficiently to respond to public inquiries this memorandum is reprinted in this issue of the Federal Register.

Ref: 40 CFR Parts 1500 - 1508 (1987).

FOR FURTHER INFORMATION CONTACT:

General Counsel,
Council on Environmental Quality,
722 Jackson Place NW,
Washington, D.C. 20006;
(202)-395-5754.

March 16, 1981

MEMORANDUM FOR FEDERAL NEPA LIAISONS, FEDERAL, STATE, AND LOCAL OFFICIALS AND OTHER PERSONS INVOLVED IN THE NEPA PROCESS

Subject: Questions and Answers About the NEPA Regulations

During June and July of 1980 the Council on Environmental Quality, with the assistance and cooperation of EPA's EIS Coordinators from the ten EPA regions, held one-day meetings with federal, state and local officials in the ten EPA regional offices around the country. In addition, on July 10, 1980, CEQ conducted a similar meeting for the Washington, D.C. NEPA liaisons and persons involved in the NEPA process. At these meetings CEQ discussed (a) the results of its 1980 review of Draft EISs issued since the July 30, 1979 effective date of the NEPA regulations, (b) agency compliance with the Record of Decision requirements in Section 1505 of the NEPA regulations, and (c) CEQ's preliminary findings on how the scoping process is working. Participants at these meetings received copies of materials prepared by CEQ summarizing its oversight and findings.

These meetings also provided NEPA liaisons and other participants with an opportunity to ask questions about NEPA and the practical application of the NEPA regulations. A number of these questions were answered by CEQ representatives at the regional meetings. In response to the many requests from the agencies and other participants, CEQ has compiled forty of the most important or most frequently asked questions and their answers and reduced them to writing. The answers were prepared by the General Counsel of CEQ in consultation with the Office of Federal Activities of EPA. These answers, of course, do not impose any additional requirements beyond those of the NEPA regulations. This document does not represent new guidance under the NEPA regulations, but rather makes generally available to concerned agencies and private individuals the answers which CEQ has already given at the 1980 regional meetings. The answers also reflect the advice which the Council has given over the past two years to aid agency staff and consultants in their day-to-day application of NEPA and the regulations.

CEQ has also received numerous inquiries regarding the scoping process. CEQ hopes to issue written guidance on scoping later this year on the basis of its special study of scoping, which is nearing completion.

NICHOLAS C. YOST
General Counsel

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END NOTES

1a. **Range of Alternatives.** What is meant by "range of alternatives" as referred to in Sec. 1505.1(e)?

A. The phrase "range of alternatives" refers to the alternatives discussed in environmental documents. It includes all reasonable alternatives, which must be rigorously explored and objectively evaluated, as well as those other alternatives, which are eliminated from detailed study with a brief discussion of the reasons for eliminating them. *Section 1502.14.* A decisionmaker must not consider alternatives beyond the range of alternatives discussed in the relevant environmental documents. Moreover, a decisionmaker must, in fact, consider all the alternatives discussed in an EIS. *Section 1505.1(e).*

1b. **How many alternatives** have to be discussed when there is an infinite number of possible alternatives?

A. For some proposals there may exist a very large or even an infinite number of possible reasonable alternatives. For example, a proposal to designate wilderness areas within a National Forest could be said to involve an infinite number of alternatives from 0 to 100 percent of the forest. When there are potentially a very large number of alternatives, only a reasonable number of examples, covering the full spectrum of alternatives, must be analyzed and compared in the EIS. An appropriate series of alternatives might include dedicating 0, 10, 30, 50, 70, 90, or 100 percent of the Forest to wilderness. What constitutes a reasonable range of alternatives depends on the nature of the proposal and the facts in each case.

2a. **Alternatives Outside the Capability of Applicant or Jurisdiction of Agency.** If an EIS is prepared in connection with an application for a permit or other federal approval, must the EIS

rigorously analyze and discuss alternatives that are outside the capability of the applicant or can it be limited to reasonable alternatives that can be carried out by the applicant?

A. Section 1502.14 requires the EIS to examine all reasonable alternatives to the proposal. In determining the scope of alternatives to be considered, the emphasis is on what is "reasonable" rather than on whether the proponent or applicant likes or is itself capable of carrying out a particular alternative. Reasonable alternatives include those that are practical or feasible from the technical and economic standpoint and using common sense, rather than simply desirable from the standpoint of the applicant.

2b. Must the EIS analyze **alternatives outside the jurisdiction** or capability of the agency or beyond what Congress has authorized?

A. An alternative that is outside the legal jurisdiction of the lead agency must still be analyzed in the EIS if it is reasonable. A potential conflict with local or federal law does not necessarily render an alternative unreasonable, although such conflicts must be considered. *Section 1506.2(d).* Alternatives that are outside the scope of what Congress has approved or funded must still be evaluated in the EIS if they are reasonable, because the EIS may serve as the basis for modifying the Congressional approval or funding in light of NEPA's goals and policies. *Section 1500.1(a).*

3. **No-Action Alternative.** What does the "no action" alternative include? If an agency is under a court order or legislative command to act, must the EIS address the "no action" alternative?

A. Section 1502.14(d) requires the alternatives analysis in the EIS to "include the alternative of no action." There are two distinct interpretations of "no action" that must be considered, depending on the nature of the proposal being evaluated. The first situation might involve an action such as updating a land management plan where ongoing programs initiated under existing legislation and regulations will continue, even as new plans are developed. In these cases "no action" is "no change" from current management direction or level of management intensity. To construct an alternative that is based on no management at all would be a useless academic exercise. Therefore, the "no action" alternative may be thought of in terms of continuing with the present course of action until that action is changed. Consequently, projected impacts of alternative management schemes would be compared in the EIS to those impacts projected for the existing plan. In this case, alternatives would include management plans of both greater and lesser intensity, especially greater and lesser levels of resource development.

The second interpretation of "no action" is illustrated in instances involving federal decisions on proposals for projects. "No action" in such cases would mean the proposed activity would not take place, and the resulting environmental effects from taking no action would be compared with the effects of permitting the proposed activity or an alternative activity to go forward.

Where a choice of "no action" by the agency would result in predictable actions by others, this consequence of the "no action" alternative should be included in the analysis. For example, if

denial of permission to build a railroad to a facility would lead to construction of a road and increased truck traffic, the EIS should analyze this consequence of the "no action" alternative.

In light of the above, it is difficult to think of a situation where it would not be appropriate to address a "no action" alternative. Accordingly, the regulations require the analysis of the no action alternative even if the agency is under a court order or legislative command to act. This analysis provides a benchmark, enabling decisionmakers to compare the magnitude of environmental effects of the action alternatives. It is also an example of a reasonable alternative outside the jurisdiction of the agency which must be analyzed. *Section 1502.14(c).* See Question 2 above. Inclusion of such an analysis in the EIS is necessary to inform the Congress, the public, and the President as intended by NEPA. *Section 1500.1(a).*

4a. **Agency's Preferred Alternative.** What is the "agency's preferred alternative"?

A. The "agency's preferred alternative" is the alternative which the agency believes would fulfill its statutory mission and responsibilities, giving consideration to economic, environmental, technical and other factors. The concept of the "agency's preferred alternative" is different from the "environmentally preferable alternative," although in some cases one alternative may be both. See Question 6 below. It is identified so that agencies and the public can understand the lead agency's orientation.

4b. Does the **"preferred alternative"** have to be identified in the Draft EIS **and** the Final EIS or just in the Final EIS?

A. Section 1502.14(e) requires the section of the EIS on alternatives to "identify the agency's preferred alternative if one or more exists, in the draft statement, and identify such alternative in the final statement . . ." This means that if the agency has a preferred alternative at the Draft EIS stage, that alternative must be labeled or identified as such in the Draft EIS. If the responsible federal official in fact has no preferred alternative at the Draft EIS stage, a preferred alternative need not be identified there. By the time the Final EIS is filed, Section 1502.14(e) presumes the existence of a preferred alternative and requires its identification in the Final EIS "unless another law prohibits the expression of such a preference."

4c. Who recommends or determines the **"preferred alternative?"**

A. The lead agency's official with line responsibility for preparing the EIS and assuring its adequacy is responsible for identifying the agency's preferred alternative(s). The NEPA regulations do not dictate which official in an agency shall be responsible for preparation of EISs, but agencies can identify this official in their implementing procedures, pursuant to Section 1507.3.

Even though the agency's preferred alternative is identified by the EIS preparer in the EIS, the statement must be objectively prepared and not slanted to support the choice of the agency's preferred alternative over the other reasonable and feasible alternatives.

5a. **Proposed Action v. Preferred Alternative.** Is the "proposed action" the same thing as the "preferred alternative"?

A. The "proposed action" may be, but is not necessarily, the agency's "preferred alternative." The proposed action may be a proposal in its initial form before undergoing analysis in the EIS process. If the proposed action is [46 FR 18028] internally generated, such as preparing a land management plan, the proposed action might end up as the agency's preferred alternative. On the other hand the proposed action may be granting an application to a non- federal entity for a permit. The agency may or may not have a "preferred alternative" at the Draft EIS stage (see Question 4 above). In that case the agency may decide at the Final EIS stage, on the basis of the Draft EIS and the public and agency comments, that an alternative other than the proposed action is the agency's "preferred alternative."

5b. Is the analysis of the **"proposed action"** in an EIS to be treated differently from the analysis of alternatives?

A. The degree of analysis devoted to each alternative in the EIS is to be substantially similar to that devoted to the "proposed action." Section 1502.14 is titled "Alternatives including the proposed action" to reflect such comparable treatment. Section 1502.14(b) specifically requires "substantial treatment" in the EIS of each alternative including the proposed action. This regulation does not dictate an amount of information to be provided, but rather, prescribes a level of treatment, which may in turn require varying amounts of information, to enable a reviewer to evaluate and compare alternatives.

6a. **Environmentally Preferable Alternative.** What is the meaning of the term "environmentally preferable alternative" as used in the regulations with reference to Records of Decision? How is the term "environment" used in the phrase?

A. Section 1505.2(b) requires that, in cases where an EIS has been prepared, the Record of Decision (ROD) must identify all alternatives that were considered, ". . . specifying the alternative or alternatives which were considered to be environmentally preferable." The environmentally preferable alternative is the alternative that will promote the national environmental policy as expressed in NEPA's Section 101. Ordinarily, this means the alternative that causes the least damage to the biological and physical environment; it also means the alternative which best protects, preserves, and enhances historic, cultural, and natural resources.

The Council recognizes that the identification of the environmentally preferable alternative may involve difficult judgments, particularly when one environmental value must be balanced against another. The public and other agencies reviewing a Draft EIS can assist the lead agency to develop and determine environmentally preferable alternatives by providing their views in comments on the Draft EIS. Through the identification of the environmentally preferable alternative, the decisionmaker is clearly faced with a choice between that alternative and others, and must consider whether the decision accords with the Congressionally declared policies of the Act.

6b. **Who recommends or determines** what is environmentally preferable?

A. The agency EIS staff is encouraged to make recommendations of the environmentally preferable alternative(s) during EIS preparation. In any event the lead agency official responsible for the EIS is encouraged to identify the environmentally preferable alternative(s) in the EIS. In all cases, commentors from other agencies and the public are also encouraged to address this question. The agency must identify the environmentally preferable alternative in the ROD.

7. **Difference Between Sections of EIS on Alternatives and Environmental Consequences.** What is the difference between the sections in the EIS on "alternatives" and "environmental consequences"? How do you avoid duplicating the discussion of alternatives in preparing these two sections?

A. The "alternatives" section is the heart of the EIS. This section rigorously explores and objectively evaluates all reasonable alternatives including the proposed action. *Section 1502.14*. It should include relevant comparisons on environmental and other grounds. The "environmental consequences" section of the EIS discusses the specific environmental impacts or effects of each of the alternatives including the proposed action. *Section 1502.16*. In order to avoid duplication between these two sections, most of the "alternatives" section should be devoted to describing and comparing the alternatives. Discussion of the environmental impacts of these alternatives should be limited to a concise descriptive summary of such impacts in a comparative form, including charts or tables, thus sharply defining the issues and providing a clear basis for choice among options. *Section 1502.14*. The "environmental consequences" section should be devoted largely to a scientific analysis of the direct and indirect environmental effects of the proposed action and of each of the alternatives. It forms the analytic basis for the concise comparison in the "alternatives" section.

8. **Early Application of NEPA.** Section 1501.2(d) of the NEPA regulations requires agencies to provide for the early application of NEPA to cases where actions are planned by **private applicants** or **non-Federal entities** and are, at some stage, subject to federal approval of permits, loans, loan guarantees, insurance or other actions. What must and can agencies do to apply NEPA early in these cases?

A. Section 1501.2(d) requires federal agencies to take steps toward ensuring that private parties and state and local entities initiate environmental studies as soon as federal involvement in their proposals can be foreseen. This section is intended to ensure that environmental factors are considered at an early stage in the planning process and to avoid the situation where the applicant for a federal permit or approval has completed planning and eliminated all alternatives to the proposed action by the time the EIS process commences or before the EIS process has been completed.

Through early consultation, business applicants and approving agencies may gain better appreciation of each other's needs and foster a decisionmaking process which avoids later unexpected confrontations.

Federal agencies are required by Section 1507.3(b) to develop procedures to carry out Section 1501.2(d). The procedures should include an "outreach program", such as a means for

prospective applicants to conduct pre-application consultations with the lead and cooperating agencies. Applicants need to find out, in advance of project planning, what environmental studies or other information will be required, and what mitigation requirements are likely, in connection with the later federal NEPA process. Agencies should designate staff to advise potential applicants of the agency's NEPA information requirements and should publicize their pre-application procedures and information requirements in newsletters or other media used by potential applicants.

Complementing Section 1501.2(d), Section 1506.5(a) requires agencies to assist applicants by outlining the types of information required in those cases where the agency requires the applicant to submit environmental data for possible use by the agency in preparing an EIS.

Section 1506.5(b) allows agencies to authorize preparation of environmental assessments by applicants. Thus, the procedures should also include a means for anticipating and utilizing applicants' environmental studies or "early corporate environmental assessments" to fulfill some of the federal agency's NEPA obligations. However, in such cases the agency must still evaluate independently the environmental issues [46 FR 18029] and take responsibility for the environmental assessment.

These provisions are intended to encourage and enable private and other non-federal entities to build environmental considerations into their own planning processes in a way that facilitates the application of NEPA and avoids delay.

9. **Applicant Who Needs Other Permits.** To what extent must an agency inquire into whether an applicant for a federal permit, funding or other approval of a proposal will also need approval from another agency for the same proposal or some other related aspect of it?

A. Agencies must integrate the NEPA process into other planning at the earliest possible time to insure that planning and decisions reflect environmental values, to avoid delays later in the process, and to head off potential conflicts. Specifically, the agency must "provide for cases where actions are planned by . . . applicants," so that designated staff are available to advise potential applicants of studies or other information that will foreseeably be required for the later federal action; the agency shall consult with the applicant if the agency foresees its own involvement in the proposal; and it shall insure that the NEPA process commences at the earliest possible time. *Section 1501.2(d).* (See Question 8.)

The regulations emphasize agency cooperation early in the NEPA process. *Section 1501.6*. Section 1501.7 on "scoping" also provides that all affected Federal agencies are to be invited to participate in scoping the environmental issues and to identify the various environmental review and consultation requirements that may apply to the proposed action. Further, Section 1502.25(b) requires that the draft EIS list all the federal permits, licenses and other entitlements that are needed to implement the proposal.

These provisions create an affirmative obligation on federal agencies to inquire early, and to the maximum degree possible, to ascertain whether an applicant is or will be seeking other federal assistance or approval, or whether the applicant is waiting until a proposal has been substantially developed before requesting federal aid or approval.

Thus, a federal agency receiving a request for approval or assistance should determine whether the applicant has filed separate requests for federal approval or assistance with other federal agencies. Other federal agencies that are likely to become involved should then be contacted, and the NEPA process coordinated, to insure an early and comprehensive analysis of the direct and indirect effects of the proposal and any related actions. The agency should inform the applicant that action on its application may be delayed unless it submits all other federal applications (where feasible to do so), so that all the relevant agencies can work together on the scoping process and preparation of the EIS.

10a. **Limitations on Action During 30-Day Review Period for Final EIS.** What actions by agencies and/or applicants are allowed during EIS preparation and during the 30-day review period after publication of a final EIS?

A. No federal decision on the proposed action shall be made or recorded until at least 30 days after the publication by EPA of notice that the particular EIS has been filed with EPA. *Sections 1505.2 and 1506.10*. Section 1505.2 requires this decision to be stated in a public Record of Decision.

Until the agency issues its Record of Decision, no action by an agency or an applicant concerning the proposal shall be taken which would have an adverse environmental impact or limit the choice of reasonable alternatives. *Section 1506.1(a).* But this does not preclude preliminary planning or design work which is needed to support an application for permits or assistance. *Section 1506.1(d).*

When the impact statement in question is a program EIS, no major action concerning the program may be taken which may significantly affect the quality of the human environment, unless the particular action is justified independently of the program, is accompanied by its own adequate environmental impact statement and will not prejudice the ultimate decision on the program. *Section 1506.1(c).*

10b. Do these **limitations on action** (described in Question 10a) apply to **state or local agencies** that have statutorily delegated responsibility for preparation of environmental documents required by NEPA, for example, under the HUD Block Grant program?

A. Yes, these limitations do apply, without any variation from their application to federal agencies.

11. **Limitations on Actions by an Applicant During EIS Process.** What actions must a lead agency take during the NEPA process when it becomes aware that a non-federal applicant is about to take an action within the agency's jurisdiction that would either have an adverse environmental impact or limit the choice of reasonable alternatives (e.g., prematurely commit money or other resources towards the completion of the proposal)?

A. The federal agency must notify the applicant that the agency will take strong affirmative steps to insure that the objectives and procedures of NEPA are fulfilled. *Section 1506.1(b).* These steps could include seeking injunctive measures under NEPA, or the use of sanctions

available under either the agency's permitting authority or statutes setting forth the agency's statutory mission. For example, the agency might advise an applicant that if it takes such action the agency will not process its application.

12a. **Effective Date and Enforceability of the Regulations.** What actions are subject to the Council's new regulations, and what actions are grandfathered under the old guidelines?

A. The effective date of the Council's regulations was July 30, 1979 (except for certain HUD programs under the Housing and Community Development Act, 42 U.S.C. 5304(h), and certain state highway programs that qualify under Section 102(2)(D) of NEPA for which the regulations became effective on November 30, 1979). All the provisions of the regulations are binding as of that date, including those covering decisionmaking, public participation, referrals, limitations on actions, EIS supplements, etc. For example, a Record of Decision would be prepared even for decisions where the draft EIS was filed before July 30, 1979.

But in determining whether or not the new regulations apply to the preparation of a particular environmental document, the relevant factor is the date of filing of the draft of that document. Thus, the new regulations do not require the redrafting of an EIS or supplement if the draft EIS or supplement was filed before July 30, 1979. However, a supplement prepared after the effective date of the regulations for an EIS issued in final before the effective date of the regulations would be controlled by the regulations.

Even though agencies are not required to apply the regulations to an EIS or other document for which the draft was filed prior to July 30, 1979, the regulations encourage agencies to follow the regulations "to the fullest extent practicable," i.e., if it is feasible to do so, in preparing the final document. *Section 1506.12(a).*

12b. Are **projects authorized by Congress before** the effective date of the Council's regulations grandfathered?

A. No. The date of Congressional authorization for a project is not determinative of whether the Council's regulations or former Guidelines apply to the particular proposal. No incomplete projects or proposals of any kind are grandfathered in whole or in part. Only certain environmental documents, for which the draft was issued before the effective date of the regulations, are grandfathered and [46 FR 18030] subject to the Council's former Guidelines.

12c. **Can a violation of the regulations give rise to a cause of action?**

A. While a trivial violation of the regulations would not give rise to an independent cause of action, such a cause of action would arise from a substantial violation of the regulations. *Section 1500.3.*

13. **Use of Scoping Before Notice of Intent to Prepare EIS.** Can the scoping process be used in connection with preparation of an **environmental assessment,** i.e., before both the decision to proceed with an EIS and publication of a notice of intent?

A. Yes. Scoping can be a useful tool for discovering alternatives to a proposal, or significant

impacts that may have been overlooked. In cases where an environmental assessment is being prepared to help an agency decide whether to prepare an EIS, useful information might result from early participation by other agencies and the public in a scoping process.

The regulations state that the scoping process is to be preceded by a Notice of Intent (NOI) to prepare an EIS. But that is only the minimum requirement. Scoping may be initiated earlier, as long as there is appropriate public notice and enough information available on the proposal so that the public and relevant agencies can participate effectively.

However, scoping that is done before the assessment, and in aid of its preparation, cannot substitute for the normal scoping process after publication of the NOI, unless the earlier public notice stated clearly that this possibility was under consideration, and the NOI expressly provides that written comments on the scope of alternatives and impacts will still be considered.

14a. **Rights and Responsibilities of Lead and Cooperating Agencies.** What are the respective rights and responsibilities of lead and cooperating agencies? What letters and memoranda must be prepared?

A. After a lead agency has been designated (Sec. 1501.5), that agency has the responsibility to solicit cooperation from other federal agencies that have jurisdiction by law or special expertise on any environmental issue that should be addressed in the EIS being prepared. Where appropriate, the lead agency should seek the cooperation of state or local agencies of similar qualifications. When the proposal may affect an Indian reservation, the agency should consult with the Indian tribe. *Section 1508.5.* The request for cooperation should come at the earliest possible time in the NEPA process.

After discussions with the candidate cooperating agencies, the lead agency and the cooperating agencies are to determine by letter or by memorandum which agencies will undertake cooperating responsibilities. To the extent possible at this stage, responsibilities for specific issues should be assigned. The allocation of responsibilities will be completed during scoping. *Section 1501.7(a)(4).*

Cooperating agencies must assume responsibility for the development of information and the preparation of environmental analyses at the request of the lead agency. *Section 1501.6(b)(3).* Cooperating agencies are now required by Section 1501.6 to devote staff resources that were normally primarily used to critique or comment on the Draft EIS after its preparation, much earlier in the NEPA process -- primarily at the scoping and Draft EIS preparation stages. If a cooperating agency determines that its resource limitations preclude any involvement, or the degree of involvement (amount of work) requested by the lead agency, it must so inform the lead agency in writing and submit a copy of this correspondence to the Council. *Section 1501.6(c).*

In other words, the potential cooperating agency must decide early if it is able to devote any of its resources to a particular proposal. For this reason the regulation states that an agency may reply to a request for cooperation that "other program commitments preclude any involvement or the degree of involvement requested in the action that is the subject of the environmental

impact statement." (Emphasis added). The regulation refers to the "action," rather than to the EIS, to clarify that the agency is taking itself out of all phases of the federal action, not just draft EIS preparation. This means that the agency has determined that it cannot be involved in the later stages of EIS review and comment, as well as decisionmaking on the proposed action. For this reason, cooperating agencies with jurisdiction by law (those which have permitting or other approval authority) cannot opt out entirely of the duty to cooperate on the EIS. See also Question 15, relating specifically to the responsibility of EPA.

14b. How are **disputes resolved between lead and cooperating agencies** concerning the scope and level of detail of analysis and the quality of data in impact statements?

A. Such disputes are resolved by the agencies themselves. A lead agency, of course, has the ultimate responsibility for the content of an EIS. But it is supposed to use the environmental analysis and recommendations of cooperating agencies with jurisdiction by law or special expertise to the maximum extent possible, consistent with its own responsibilities as lead agency. *Section 1501.6(a)(2).*

If the lead agency leaves out a significant issue or ignores the advice and expertise of the cooperating agency, the EIS may be found later to be inadequate. Similarly, where cooperating agencies have their own decisions to make and they intend to adopt the environmental impact statement and base their decisions on it, one document should include all of the information necessary for the decisions by the cooperating agencies. Otherwise they may be forced to duplicate the EIS process by issuing a new, more complete EIS or Supplemental EIS, even though the original EIS could have sufficed if it had been properly done at the outset. Thus, both lead and cooperating agencies have a stake in producing a document of good quality. Cooperating agencies also have a duty to participate fully in the scoping process to ensure that the appropriate range of issues is determined early in the EIS process.

Because the EIS is not the Record of Decision, but instead constitutes the information and analysis on which to base a decision, disagreements about conclusions to be drawn from the EIS need not inhibit agencies from issuing a joint document, or adopting another agency's EIS, if the analysis is adequate. Thus, if each agency has its own "preferred alternative," both can be identified in the EIS. Similarly, a cooperating agency with jurisdiction by law may determine in its own ROD that alternative A is the environmentally preferable action, even though the lead agency has decided in its separate ROD that Alternative B is environmentally preferable.

14c. What are the specific responsibilities of federal and state **cooperating agencies to review draft EISs**?

A. Cooperating agencies (i.e., agencies with jurisdiction by law or special expertise) and agencies that are authorized to develop or enforce environmental standards, must comment on environmental impact statements within their jurisdiction, expertise or authority. *Sections 1503.2, 1508.5*. If a cooperating agency is satisfied that its views are adequately reflected in the environmental impact statement, it should simply comment accordingly. Conversely, if the cooperating agency determines that a draft EIS is incomplete, inadequate or inaccurate, or it has other comments, it should promptly make such comments, conforming to the requirements

of specificity in section 1503.3.

14d. How is the lead agency to treat the comments of another agency with jurisdiction by law or special expertise which has **failed or refused to cooperate or participate in scoping or EIS preparation**?

A. A lead agency has the responsibility to respond to all substantive comments raising significant issues regarding a draft EIS. *Section 1503.4*. However, cooperating agencies are generally under an obligation to raise issues or otherwise participate in the EIS process during scoping and EIS preparation if they reasonably can do so. In practical terms, if a cooperating agency fails to cooperate at the outset, such as during scoping, it will find that its comments at a later stage will not be as persuasive to the lead agency.

15. **Commenting Responsibilities of EPA.** Are EPA's responsibilities to review and comment on the environmental effects of agency proposals under **Section 309 of the Clean Air Act** independent of its responsibility as a cooperating agency?

A. Yes. EPA has an obligation under Section 309 of the Clean Air Act to review and comment in writing on the environmental impact of any matter relating to the authority of the Administrator contained in proposed legislation, federal construction projects, other federal actions requiring EISs, and new regulations. *42 U.S.C. Sec. 7609.* This obligation is independent of its role as a cooperating agency under the NEPA regulations.

16. **Third Party Contracts.** What is meant by the term "third party contracts" in connection with the preparation of an EIS? See Section 1506.5(c). When can "third party contracts" be used?

A. As used by EPA and other agencies, the term "third party contract" refers to the preparation of EISs by contractors paid by the applicant. In the case of an EIS for a National Pollution Discharge Elimination System (NPDES) permit, the applicant, aware in the early planning stages of the proposed project of the need for an EIS, contracts directly with a consulting firm for its preparation. See 40 C.F.R. 6.604(g). The "third party" is EPA which, under Section 1506.5(c), must select the consulting firm, even though the applicant pays for the cost of preparing the EIS. The consulting firm is responsible to EPA for preparing an EIS that meets the requirements of the NEPA regulations and EPA's NEPA procedures. It is in the applicant's interest that the EIS comply with the law so that EPA can take prompt action on the NPDES permit application. The "third party contract" method under EPA's NEPA procedures is purely voluntary, though most applicants have found it helpful in expediting compliance with NEPA.

If a federal agency uses "third party contracting," the applicant may undertake the necessary paperwork for the solicitation of a field of candidates under the agency's direction, so long as the agency complies with Section 1506.5(c). Federal procurement requirements do not apply to the agency because it incurs no obligations or costs under the contract, nor does the agency procure anything under the contract.

17a. **Disclosure Statement to Avoid Conflict of Interest.** If an EIS is prepared with the assistance of a consulting firm, the firm must execute a disclosure statement. What criteria

must the firm follow in determining whether it has any "financial or other interest in the outcome of the project" which would cause a conflict of interest?

A. Section 1506.5(c), which specifies that a consulting firm preparing an EIS must execute a disclosure statement, does not define "financial or other interest in the outcome of the project." The Council interprets this term broadly to cover any known benefits other than general enhancement of professional reputation. This includes any financial benefit such as a promise of future construction or design work on the project, as well as indirect benefits the consultant is aware of (e.g., if the project would aid proposals sponsored by the firm's other clients). For example, completion of a highway project may encourage construction of a shopping center or industrial park from which the consultant stands to benefit. If a consulting firm is aware that it has such an interest in the decision on the proposal, it should be disqualified from preparing the EIS, to preserve the objectivity and integrity of the NEPA process.

When a consulting firm has been involved in developing initial data and plans for the project, but does not have any financial or other interest in the outcome of the decision, it need not be disqualified from preparing the EIS. However, a disclosure statement in the draft EIS should clearly state the scope and extent of the firm's prior involvement to expose any potential conflicts of interest that may exist.

17b. If the firm in fact has no promise of future work or other interest in the outcome of the proposal, **may the firm later bid** in competition with others for future work on the project if the proposed action is approved?

A. Yes.

18. **Uncertainties About Indirect Effects of A Proposal.** How should uncertainties about indirect effects of a proposal be addressed, for example, in cases of disposal of federal lands, when the identity or plans of future landowners is unknown?

A. The EIS must identify all the indirect effects that are known, and make a good faith effort to explain the effects that are not known but are "reasonably foreseeable." *Section 1508.8(b).* In the example, if there is total uncertainty about the identity of future land owners or the nature of future land uses, then of course, the agency is not required to engage in speculation or contemplation about their future plans. But, in the ordinary course of business, people do make judgments based upon reasonably foreseeable occurrences. It will often be possible to consider the likely purchasers and the development trends in that area or similar areas in recent years; or the likelihood that the land will be used for an energy project, shopping center, subdivision, farm or factory. The agency has the responsibility to make an informed judgment, and to estimate future impacts on that basis, especially if trends are ascertainable or potential purchasers have made themselves known. The agency cannot ignore these uncertain, but probable, effects of its decisions.

19a. **Mitigation Measures.** What is the scope of mitigation measures that must be discussed?
A. The mitigation measures discussed in an EIS must cover the range of impacts of the proposal. The measures must include such things as design alternatives that would decrease

pollution emissions, construction impacts, esthetic intrusion, as well as relocation assistance, possible land use controls that could be enacted, and other possible efforts. Mitigation measures must be considered even for impacts that by themselves would not be considered "significant." Once the proposal itself is considered as a whole to have significant effects, all of its specific effects on the environment (whether or not "significant") must be considered, and mitigation measures must be developed where it is feasible to do so. *Sections 1502.14(f), 1502.16(h), 1508.14*.

19b. How should an EIS treat the subject of available mitigation measures that are (1) **outside the jurisdiction** of the lead or cooperating agencies, or (2) **unlikely** to be adopted or enforced by the responsible agency?

A. All relevant, reasonable mitigation measures that could improve the project are to be identified, even if they are outside the jurisdiction of the lead agency or the cooperating agencies, and thus would not be committed as part of the RODs of these agencies. *Sections 1502.16(h), 1505.2(c).* This will serve to [46 FR 18032] alert agencies or officials who can implement these extra measures, and will encourage them to do so. Because the EIS is the most comprehensive environmental document, it is an ideal vehicle in which to lay out not only the full range of environmental impacts but also the full spectrum of appropriate mitigation.

However, to ensure that environmental effects of a proposed action are fairly assessed, the probability of the mitigation measures being implemented must also be discussed. Thus the EIS and the Record of Decision should indicate the likelihood that such measures will be adopted or enforced by the responsible agencies. *Sections 1502.16(h), 1505.2*. If there is a history of nonenforcement or opposition to such measures, the EIS and Record of Decision should acknowledge such opposition or nonenforcement. If the necessary mitigation measures will not be ready for a long period of time, this fact, of course, should also be recognized.

20. **Worst Case Analysis**. [Withdrawn.]

21. **Combining Environmental and Planning Documents.** Where an EIS or an EA is combined with another project planning document (sometimes called **"piggybacking"**), to what degree may the EIS or EA refer to and rely upon information in the project document to satisfy NEPA's requirements?

A. Section 1502.25 of the regulations requires that draft EISs be prepared concurrently and integrated with environmental analyses and related surveys and studies required by other federal statutes. In addition, Section 1506.4 allows any environmental document prepared in compliance with NEPA to be combined with any other agency document to reduce duplication and paperwork. However, these provisions were not intended to authorize the preparation of a short summary or outline EIS, attached to a detailed project report or land use plan containing the required environmental impact data. In such circumstances, the reader would have to refer constantly to the detailed report to understand the environmental impacts and alternatives which should have been found in the EIS itself.

The EIS must stand on its own as an analytical document which fully informs decisionmakers and the public of the environmental effects of the proposal and those of the reasonable alternatives. *Section 1502.1*. But, as long as the EIS is clearly identified and is self-supporting, it can be physically included in or attached to the project report or land use plan, and may use attached report material as technical backup.

Forest Service environmental impact statements for forest management plans are handled in this manner. The EIS identifies the agency's preferred alternative, which is developed in detail as the proposed management plan. The detailed proposed plan accompanies the EIS through the review process, and the documents are appropriately cross-referenced. The proposed plan is useful for EIS readers as an example, to show how one choice of management options translates into effects on natural resources. This procedure permits initiation of the 90-day public review of proposed forest plans, which is required by the National Forest Management Act.

All the alternatives are discussed in the EIS, which can be read as an independent document. The details of the management plan are not repeated in the EIS, and vice versa. This is a reasonable functional separation of the documents: the EIS contains information relevant to the choice among alternatives; the plan is a detailed description of proposed management activities suitable for use by the land managers. This procedure provides for concurrent compliance with the public review requirements of both NEPA and the National Forest Management Act.

Under some circumstances, a project report or management plan may be totally merged with the EIS, and the one document labeled as both "EIS" and "management plan" or "project report." This may be reasonable where the documents are short, or where the EIS format and the regulations for clear, analytical EISs also satisfy the requirements for a project report.

22. **State and Federal Agencies as Joint Lead Agencies.** May state and federal agencies serve as joint lead agencies? If so, how do they resolve law, policy and resource conflicts under NEPA and the relevant state environmental policy act? How do they resolve differences in perspective where, for example, national and local needs may differ?

A. Under Section 1501.5(b), federal, state or local agencies, as long as they include at least one federal agency, may act as joint lead agencies to prepare an EIS. Section 1506.2 also strongly urges state and local agencies and the relevant federal agencies to cooperate fully with each other. This should cover joint research and studies, planning activities, public hearings, environmental assessments and the preparation of joint EISs under NEPA and the relevant "little NEPA" state laws, so that one document will satisfy both laws.

The regulations also recognize that certain inconsistencies may exist between the proposed federal action and any approved state or local plan or law. The joint document should discuss the extent to which the federal agency would reconcile its proposed action with such plan or law. *Section 1506.2(d).* (See Question 23).

Because there may be differences in perspective as well as conflicts among [46 FR 18033] federal, state and local goals for resources management, the Council has advised participating

agencies to adopt a flexible, cooperative approach. The joint EIS should reflect all of their interests and missions, clearly identified as such. The final document would then indicate how state and local interests have been accommodated, or would identify conflicts in goals (e.g., how a hydroelectric project, which might induce second home development, would require new land use controls). The EIS must contain a complete discussion of scope and purpose of the proposal, alternatives, and impacts so that the discussion is adequate to meet the needs of local, state and federal decisionmakers.

23a. **Conflicts of Federal Proposal With Land Use Plans, Policies or Controls.** How should an agency handle potential **conflicts** between a proposal and the objectives of Federal, state or local land use plans, policies and controls for the area concerned? See Sec. 1502.16(c).

A. The agency should first inquire of other agencies whether there are any potential conflicts. If there would be immediate conflicts, or if conflicts could arise in the future when the plans are finished (see Question 23(b) below), the EIS must acknowledge and describe the extent of those conflicts. If there are any possibilities of resolving the conflicts, these should be explained as well. The EIS should also evaluate the seriousness of the impact of the proposal on the land use plans and policies, and whether, or how much, the proposal will impair the effectiveness of land use control mechanisms for the area. Comments from officials of the affected area should be solicited early and should be carefully acknowledged and answered in the EIS.

23b. What constitutes a **"land use plan or policy"** for purposes of this discussion?

A. The term "land use plans," includes all types of formally adopted documents for land use planning, zoning and related regulatory requirements. Local general plans are included, even though they are subject to future change. Proposed plans should also be addressed if they have been formally proposed by the appropriate government body in a written form, and are being actively pursued by officials of the jurisdiction. Staged plans, which must go through phases of development such as the Water Resources Council's Level A, B and C planning process should also be included even though they are incomplete.

The term "policies" includes formally adopted statements of land use policy as embodied in laws or regulations. It also includes proposals for action such as the initiation of a planning process, or a formally adopted policy statement of the local, regional or state executive branch, even if it has not yet been formally adopted by the local, regional or state legislative body.

23c. What options are available for the decisionmaker when **conflicts with such plans** or policies are identified?

A. After identifying any potential land use conflicts, the decisionmaker must weigh the significance of the conflicts, among all the other environmental and non-environmental factors that must be considered in reaching a rational and balanced decision. Unless precluded by other law from causing or contributing to any inconsistency with the land use plans, policies or controls, the decisionmaker retains the authority to go forward with

the proposal, despite the potential conflict. In the Record of Decision, the decisionmaker must explain what the decision was, how it was made, and what mitigation measures are being imposed to lessen adverse environmental impacts of the proposal, among the other requirements of Section 1505.2. This provision would require the decisionmaker to explain any decision to override land use plans, policies or controls for the area.

24a. **Environmental Impact Statements on Policies, Plans or Programs.** When are EISs required on policies, plans or programs?

A. An EIS must be prepared if an agency proposes to implement a specific policy, to adopt a plan for a group of related actions, or to implement a specific statutory program or executive directive. *Section 1508.18*. In addition, the adoption of official policy in the form of rules, regulations and interpretations pursuant to the Administrative Procedure Act, treaties, conventions, or other formal documents establishing governmental or agency policy which will substantially alter agency programs, could require an EIS. *Section 1508.18*. In all cases, the policy, plan, or program must have the potential for significantly affecting the quality of the human environment in order to require an EIS. It should be noted that a proposal "may exist in fact as well as by agency declaration that one exists." *Section 1508.23*.

24b. When is an **area-wide or overview EIS** appropriate?

A. The preparation of an area-wide or overview EIS may be particularly useful when similar actions, viewed with other reasonably foreseeable or proposed agency actions, share common timing or geography. For example, when a variety of energy projects may be located in a single watershed, or when a series of new energy technologies may be developed through federal funding, the overview or area-wide EIS would serve as a valuable and necessary analysis of the affected environment and the potential cumulative impacts of the reasonably foreseeable actions under that program or within that geographical area.

24c. What is the function of **tiering** in such cases?

A. Tiering is a procedure which allows an agency to avoid duplication of paperwork through the incorporation by reference of the general discussions and relevant specific discussions from an environmental impact statement of broader scope into one of lesser scope or vice versa. In the example given in Question 24b, this would mean that an overview EIS would be prepared for all of the energy activities reasonably foreseeable in a particular geographic area or resulting from a particular development program. This impact statement would be followed by site-specific or project-specific EISs. The tiering process would make each EIS of greater use and meaning to the public as the plan or program develops, without duplication of the analysis prepared for the previous impact statement.

25a. **Appendices and Incorporation by Reference.** When is it appropriate to use appendices instead of including information in the body of an EIS?

A. The body of the EIS should be a succinct statement of all the information on environmental impacts and alternatives that the decisionmaker and the public need, in order to make the decision and to ascertain that every significant factor has been examined. The EIS must

explain or summarize methodologies of research and modeling, and the results of research that may have been conducted to analyze impacts and alternatives.

Lengthy technical discussions of modeling methodology, baseline studies, or other work are best reserved for the appendix. In other words, if only technically trained individuals are likely to understand a particular discussion then it should go in the appendix, and a plain language summary of the analysis and conclusions of that technical discussion should go in the text of the EIS.

The final statement must also contain the agency's responses to comments on the draft EIS. These responses will be primarily in the form of changes in the document itself, but specific answers to each significant comment should also be included. These specific responses may be placed in an appendix. If the comments are especially voluminous, summaries of the comments and responses will suffice. (See Question 29 regarding the level of detail required for responses to comments.)

25b. How does an **appendix** differ from **incorporation by reference?**

A. First, if at all possible, the appendix accompanies the EIS, whereas the material which is incorporated by reference does not accompany the EIS. Thus the appendix should contain information that reviewers will be likely to want to examine. The appendix should include material that pertains to preparation of a particular EIS. Research papers directly relevant to the proposal, lists of affected species, discussion of the methodology of models used in the analysis of impacts, extremely detailed responses to comments, or other information, would be placed in the appendix.

The appendix must be complete and available at the time the EIS is filed. Five copies of the appendix must be sent to EPA with five copies of the EIS for filing. If the appendix is too bulky to be circulated, it instead must be placed in conveniently accessible locations or furnished directly to commentors upon request. If it is not circulated with the EIS, the Notice of Availability published by EPA must so state, giving a telephone number to enable potential commentors to locate or request copies of the appendix promptly.

Material that is not directly related to preparation of the EIS should be incorporated by reference. This would include other EISs, research papers in the general literature, technical background papers or other material that someone with technical training could use to evaluate the analysis of the proposal. These must be made available, either by citing the literature, furnishing copies to central locations, or sending copies directly to commenters upon request.

Care must be taken in all cases to ensure that material incorporated by reference, and the occasional appendix that does not accompany the EIS, are in fact available for the full minimum public comment period.

26a. **Index and Keyword Index in EISs.** How detailed must an EIS index be?

A. The EIS index should have a level of detail sufficient to focus on areas of the EIS of

reasonable interest to any reader. It cannot be restricted to the most important topics. On the other hand, it need not identify every conceivable term or phrase in the EIS. If an agency believes that the reader is reasonably likely to be interested in a topic, it should be included.

26b. Is a **keyword index** required?

A. No. A keyword index is a relatively short list of descriptive terms that identifies the key concepts or subject areas in a document. For example it could consist of 20 terms which describe the most significant aspects of an EIS that a future researcher would need: type of proposal, type of impacts, type of environment, geographical area, sampling or modeling methodologies used. This technique permits the compilation of EIS data banks, by facilitating quick and inexpensive access to stored materials. While a keyword index is not required by the regulations, it could be a useful addition for several reasons. First, it can be useful as a quick index for reviewers of the EIS, helping to focus on areas of interest. Second, if an agency keeps a listing of the keyword indexes of the EISs it produces, the EIS preparers themselves will have quick access to similar research data and methodologies to aid their future EIS work. Third, a keyword index will be needed to make an EIS available to future researchers using EIS data banks that are being developed. Preparation of such an index now when the document is produced will save a later effort when the data banks become operational.

27a. **List of Preparers.** If a consultant is used in preparing an EIS, must the list of preparers identify members of the consulting firm as well as the agency NEPA staff who were primarily responsible?

A. Section 1502.17 requires identification of the names and qualifications of persons who were primarily responsible for preparing the EIS or significant background papers, including basic components of the statement. This means that members of a consulting firm preparing material that is to become part of the EIS must be identified. The EIS should identify these individuals even though the consultant's contribution may have been modified by the agency.

27b. Should agency staff involved in reviewing and editing the EIS also be included in the **list of preparers?**

A. Agency personnel who wrote basic components of the EIS or significant background papers must, of course, be identified. The EIS should also list the technical editors who reviewed or edited the statements.

27c. How much information should be included on each person listed?

A. The list of preparers should normally not exceed two pages. Therefore, agencies must determine which individuals had primary responsibility and need not identify individuals with minor involvement. The list of preparers should include a very brief identification of the individuals involved, their qualifications (expertise, professional disciplines) and the specific portion of the EIS for which they are responsible. This may be done in tabular form to cut down on length. A line or two for each person's qualifications should be sufficient.

28. **Advance or Xerox Copies of EIS.** May an agency file xerox copies of an EIS with EPA pending the completion of printing the document?

A. Xerox copies of an EIS may be filed with EPA prior to printing only if the xerox copies are simultaneously made available to other agencies and the public. Section 1506.9 of the regulations, which governs EIS filing, specifically requires Federal agencies to file EISs with EPA no earlier than the EIS is distributed to the public. However, this section does not prohibit xeroxing as a form of reproduction and distribution. When an agency chooses xeroxing as the reproduction method, the EIS must be clear and legible to permit ease of reading and ultimate microfiching of the EIS. Where color graphs are important to the EIS, they should be reproduced and circulated with the xeroxed copy.

29a. **Responses to Comments.** What response must an agency provide to a comment on a draft EIS which states that the EIS's methodology is inadequate or inadequately explained? For example, what level of detail must an agency include in its response to a simple postcard comment making such an allegation?

A. Appropriate responses to comments are described in Section 1503.4. Normally the responses should result in changes in the text of the EIS, not simply a separate answer at the back of the document. But, in addition, the agency must state what its response was, and if the agency decides that no substantive response to a comment is necessary, it must explain briefly why.

An agency is not under an obligation to issue a lengthy reiteration of its methodology for any portion of an EIS if the only comment addressing the methodology is a simple complaint that the EIS methodology is inadequate. But agencies must respond to comments, however brief, which are specific in their criticism of agency methodology. For example, if a commentor on an EIS said that an agency's air quality dispersion analysis or methodology was inadequate, and the agency had included a discussion of that analysis in the EIS, little if anything need be added in response to such a comment. However, if the commenter said that the dispersion analysis was inadequate because of its use of a certain computational technique, or that a dispersion analysis was inadequately explained because computational techniques were not included or referenced, then the agency would have to respond in a substantive and meaningful way to such a comment. If a number of comments are identical or very similar, agencies may group the comments and prepare a single answer for each group. Comments may be summarized if they are especially voluminous. The comments or summaries must be attached to the EIS regardless of whether the agency believes they merit individual discussion in the body of the final EIS.

29b. How must an agency respond to a comment on a draft EIS that raises a **new alternative not previously considered** in the draft EIS?

A. This question might arise in several possible situations. First, a commenter on a draft EIS may indicate that there is a possible alternative which, in the agency's view, is not a reasonable alternative. *Section 1502.14(a).* If that is the case, the agency must explain why the comment does not warrant further agency response, citing authorities or reasons that support the agency's position and, if appropriate, indicate those circumstances which would trigger agency

reappraisal or further response. *Section 1503.4(a).* For example, a commentor on a draft EIS on a coal fired power plant may suggest the alternative of using synthetic fuel. The agency may reject the alternative with a brief discussion (with authorities) of the unavailability of synthetic fuel within the time frame necessary to meet the need and purpose of the proposed facility.

A second possibility is that an agency may receive a comment indicating that a particular alternative, while reasonable, should be modified somewhat, for example, to achieve certain mitigation benefits, or for other reasons. If the modification is reasonable, the agency should include a discussion of it in the final EIS. For example, a commenter on a draft EIS on a proposal for a pumped storage power facility might suggest that the applicant's proposed alternative should be enhanced by the addition of certain reasonable mitigation measures, including the purchase and setaside of a wildlife preserve to substitute for the tract to be destroyed by the project. The modified alternative including the additional mitigation measures should be discussed by the agency in the final EIS.

A third slightly different possibility is that a comment on a draft EIS will raise an alternative which is a minor variation of one of the alternatives discussed in the draft EIS, but this variation was not given any consideration by the agency. In such a case, the agency should develop and evaluate the new alternative, if it is reasonable, in the final EIS. If it is qualitatively within the spectrum of alternatives that were discussed in the draft, a supplemental draft will not be needed. For example, a commenter on a draft EIS to designate a wilderness area within a National Forest might reasonably identify a specific tract of the forest, and urge that it be considered for designation. If the draft EIS considered designation of a range of alternative tracts which encompassed forest area of similar quality and quantity, no supplemental EIS would have to be prepared. The agency could fulfill its obligation by addressing that specific alternative in the final EIS.

As another example, an EIS on an urban housing project may analyze the alternatives of constructing 2,000, 4,000, or 6,000 units. A commentor on the draft EIS might urge the consideration of constructing 5,000 units utilizing a different configuration of buildings. This alternative is within the spectrum of alternatives already considered, and, therefore, could be addressed in the final EIS.

A fourth possibility is that a commenter points out an alternative which is not a variation of the proposal or of any alternative discussed in the draft impact statement, and is a reasonable alternative that warrants serious agency response. In such a case, the agency must issue a supplement to the draft EIS that discusses this new alternative. For example, a commenter on a draft EIS on a nuclear power plant might suggest that a reasonable alternative for meeting the projected need for power would be through peak load management and energy conservation programs. If the permitting agency has failed to consider that approach in the Draft EIS, and the approach cannot be dismissed by the agency as unreasonable, a supplement to the Draft EIS, which discusses that alternative, must be prepared. (If necessary, the same supplement should also discuss substantial changes in the proposed action or significant new circumstances or information, as required by Section 1502.9(c)(1) of the Council's regulations.)

If the new alternative was not raised by the commentor during scoping, but could have been,

commenters may find that they are unpersuasive in their efforts to have their suggested alternative analyzed in detail by the agency. However, if the new alternative is discovered or developed later, and it could not reasonably have been raised during the scoping process, then the agency must address it in a supplemental draft EIS. The agency is, in any case, ultimately responsible for preparing an adequate EIS that considers all alternatives.

30. **Adoption of EISs.** When a cooperating agency with jurisdiction by law intends to adopt a lead agency's EIS and it is not satisfied with the adequacy of the document, may the cooperating agency adopt only the part of the EIS with which it is satisfied? If so, would a cooperating agency with jurisdiction by law have to prepare a separate EIS or EIS supplement covering the areas of disagreement with the lead agency?

A. Generally, a cooperating agency may adopt a lead agency's EIS without recirculating it if it concludes that its NEPA requirements and its comments and suggestions have been satisfied. *Section 1506.3(a), (c).* If necessary, a cooperating agency may adopt only a portion of the lead agency's EIS and may reject that part of the EIS with which it disagrees, stating publicly why it did so. *Section 1506.3(a).*

A cooperating agency with jurisidiction by law (e.g., an agency with independent legal responsibilities with respect to the proposal) has an independent legal obligation to comply with NEPA. Therefore, if the cooperating agency determines that the EIS is wrong or inadequate, it must prepare a supplement to the EIS, replacing or adding any needed information, and must circulate the supplement as a draft for public and agency review and comment. A final supplemental EIS would be required before the agency could take action. The adopted portions of the lead agency EIS should be circulated with the supplement. *Section 1506.3(b).* A cooperating agency with jurisdiction by law will have to prepare its own Record of Decision for its action, in which it must explain how it reached its conclusions. Each agency should explain how and why its conclusions differ, if that is the case, from those of other agencies which issued their Records of Decision earlier. An agency that did not cooperate in preparation of an EIS may also adopt an EIS or portion thereof. But this would arise only in rare instances, because an agency adopting an EIS for use in its own decision normally would have been a cooperating agency. If the proposed action for which the EIS was prepared is substantially the same as the proposed action of the adopting agency, the EIS may be adopted as long as it is recirculated as a final EIS and the agency announces what it is doing. This would be followed by the 30-day review period and issuance of a Record of Decision by the adopting agency. If the proposed action by the adopting agency is not substantially the same as that in [46 FR 18036] the EIS (i.e., if an EIS on one action is being adapted for use in a decision on another action), the EIS would be treated as a draft and circulated for the normal public comment period and other procedures. *Section 1506.3(b).*

31a. **Application of Regulations to Independent Regulatory Agencies.** Do the Council's NEPA regulations apply to independent regulatory agencies like the Federal Energy Regulatory Commission (FERC) and the Nuclear Regulatory Commission?

A. The statutory requirements of NEPA's Section 102 apply to "all agencies of the federal government." The NEPA regulations implement the procedural provisions of NEPA as set

forth in NEPA's Section 102(2) for all agencies of the federal government. The NEPA regulations apply to independent regulatory agencies, however, they do not direct independent regulatory agencies or other agencies to make decisions in any particular way or in a way inconsistent with an agency's statutory charter. *Sections 1500.3, 1500.6, 1507.1, and 1507.3.*

31b. Can an Executive Branch agency like the Department of the Interior **adopt an EIS** prepared by an independent regulatory agency such as FERC?

A. If an independent regulatory agency such as FERC has prepared an EIS in connection with its approval of a proposed project, an Executive Branch agency (e.g., the Bureau of Land Management in the Department of the Interior) may, in accordance with Section 1506.3, adopt the EIS or a portion thereof for its use in considering the same proposal. In such a case the EIS must, to the satisfaction of the adopting agency, meet the standards for an adequate statement under the NEPA regulations (including scope and quality of analysis of alternatives) and must satisfy the adopting agency's comments and suggestions. If the independent regulatory agency fails to comply with the NEPA regulations, the cooperating or adopting agency may find that it is unable to adopt the EIS, thus forcing the preparation of a new EIS or EIS Supplement for the same action. The NEPA regulations were made applicable to all federal agencies in order to avoid this result, and to achieve uniform application and efficiency of the NEPA process.

32. **Supplements to Old EISs.** Under what circumstances do old EISs have to be supplemented before taking action on a proposal?

A. As a rule of thumb, if the proposal has not yet been implemented, or if the EIS concerns an ongoing program, EISs that are more than 5 years old should be carefully reexamined to determine if the criteria in Section 1502.9 compel preparation of an EIS supplement.

If an agency has made a substantial change in a proposed action that is relevant to environmental concerns, or if there are significant new circumstances or information relevant to environmental concerns and bearing on the proposed action or its impacts, a supplemental EIS must be prepared for an old EIS so that the agency has the best possible information to make any necessary substantive changes in its decisions regarding the proposal. *Section 1502.9(c).*

33a. **Referrals.** When must a referral of an interagency disagreement be made to the Council?

A. The Council's referral procedure is a pre-decision referral process for interagency disagreements. Hence, Section 1504.3 requires that a referring agency must deliver its referral to the Council not later than 25 days after publication by EPA of notice that the final EIS is available (unless the lead agency grants an extension of time under Section 1504.3(b)).

33b. May a **referral** be made after this issuance of a Record of Decision?

A. No, except for cases where agencies provide an internal appeal procedure which permits simultaneous filing of the final EIS and the record of decision (ROD). *Section 1506.10(b)(2).* Otherwise, as stated above, the process is a pre-decision referral process. Referrals must be

made within 25 days after the notice of availability of the final EIS, whereas the final decision (ROD) may not be made or filed until after 30 days from the notice of availability of the EIS. *Sections 1504.3(b), 1506.10(b).* If a lead agency has granted an extension of time for another agency to take action on a referral, the ROD may not be issued until the extension has expired.

34a. **Records of Decision.** Must Records of Decision (RODs) be made public? How should they be made available?

A. Under the regulations, agencies must prepare a "concise public record of decision," which contains the elements specified in Section 1505.2. This public record may be integrated into any other decision record prepared by the agency, or it may be separate if decision documents are not normally made public. The Record of Decision is intended by the Council to be an environmental document (even though it is not explicitly mentioned in the definition of "environmental document" in Section 1508.10). Therefore, it must be made available to the public through appropriate public notice as required by Section 1506.6(b). However, there is no specific requirement for publication of the ROD itself, either in the Federal Register or elsewhere.

34b. May the **summary section** in the final Environmental Impact Statement substitute for or constitute an agency's Record of Decision?

A. No. An environmental impact statement is supposed to inform the decisionmaker before the decision is made. *Sections 1502.1, 1505.2.* The Council's regulations provide for a 30-day period after notice is published that the final EIS has been filed with EPA before the agency may take final action. During that period, in addition to the agency's own internal final review, the public and other agencies can comment on the final EIS prior to the agency's final action on the proposal. In addition, the Council's regulations make clear that the requirements for the summary in an EIS are not the same as the requirements for a ROD. *Sections 1502.12 and 1505.2.*

34c. What provisions should **Records of Decision** contain pertaining to **mitigation and monitoring**?

A. Lead agencies "shall include appropriate conditions [including mitigation measures and monitoring and enforcement programs] in grants, permits or other approvals" and shall "condition funding of actions on mitigation." *Section 1505.3.* Any such measures that are adopted must be explained and committed in the ROD.

The reasonable alternative mitigation measures and monitoring programs should have been addressed in the draft and final EIS. The discussion of mitigation and monitoring in a Record of Decision must be more detailed than a general statement that mitigation is being required, but not so detailed as to duplicate discussion of mitigation in the EIS. The Record of Decision should contain a concise summary identification of the mitigation measures which the agency has committed itself to adopt.

The Record of Decision must also state whether all practicable mitigation measures have

been adopted, and if not, why not. *Section 1505.2(c).* The Record of Decision must identify the mitigation measures and monitoring and enforcement programs that have been selected and plainly indicate that they are adopted as part of the agency's decision. If the proposed action is the issuance of a permit or other approval, the specific details of the mitigation measures shall then be included as appropriate conditions in whatever grants, permits, funding or other approvals are being made by the federal agency. *Section 1505.3 (a), (b).* If the proposal is to be carried out by the [46 FR 18037] federal agency itself, the Record of Decision should delineate the mitigation and monitoring measures in sufficient detail to constitute an enforceable commitment, or incorporate by reference the portions of the EIS that do so.

34d. What is the **enforceability of a Record of Decision?**

A. Pursuant to generally recognized principles of federal administrative law, agencies will be held accountable for preparing Records of Decision that conform to the decisions actually made and for carrying out the actions set forth in the Records of Decision. This is based on the principle that an agency must comply with its own decisions and regulations once they are adopted. Thus, the terms of a Record of Decision are enforceable by agencies and private parties. A Record of Decision can be used to compel compliance with or execution of the mitigation measures identified therein.

35. **Time Required for the NEPA Process.** How long should the NEPA process take to complete?

A. When an EIS is required, the process obviously will take longer than when an EA is the only document prepared. But the Council's NEPA regulations encourage streamlined review, adoption of deadlines, elimination of duplicative work, eliciting suggested alternatives and other comments early through scoping, cooperation among agencies, and consultation with applicants during project planning. The Council has advised agencies that under the new NEPA regulations even large complex energy projects would require only about 12 months for the completion of the entire EIS process. For most major actions, this period is well within the planning time that is needed in any event, apart from NEPA.

The time required for the preparation of program EISs may be greater. The Council also recognizes that some projects will entail difficult long-term planning and/or the acquisition of certain data which of necessity will require more time for the preparation of the EIS. Indeed, some proposals should be given more time for the thoughtful preparation of an EIS and development of a decision which fulfills NEPA's substantive goals.

For cases in which only an environmental assessment will be prepared, the NEPA process should take no more than 3 months, and in many cases substantially less, as part of the normal analysis and approval process for the action.

36a. **Environmental Assessments (EA).** How long and detailed must an environmental assessment (EA) be?

A. The environmental assessment is a concise public document which has three defined

functions. (1) It briefly provides sufficient evidence and analysis for determining whether to prepare an EIS; (2) it aids an agency's compliance with NEPA when no EIS is necessary, i.e., it helps to identify better alternatives and mitigation measures; and (3) it facilitates preparation of an EIS when one is necessary. *Section 1508.9(a).*

Since the EA is a concise document, it should not contain long descriptions or detailed data which the agency may have gathered. Rather, it should contain a brief discussion of the need for the proposal, alternatives to the proposal, the environmental impacts of the proposed action and alternatives, and a list of agencies and persons consulted. *Section 1508.9(b).*

While the regulations do not contain page limits for EA's, the Council has generally advised agencies to keep the length of EAs to not more than approximately 10-15 pages. Some agencies expressly provide page guidelines (e.g., 10-15 pages in the case of the Army Corps). To avoid undue length, the EA may incorporate by reference background data to support its concise discussion of the proposal and relevant issues.

36b. Under what circumstances is a **lengthy EA** appropriate?

A. Agencies should avoid preparing lengthy EAs except in unusual cases, where a proposal is so complex that a concise document cannot meet the goals of Section 1508.9 and where it is extremely difficult to determine whether the proposal could have significant environmental effects. In most cases, however, a lengthy EA indicates that an EIS is needed.

37a. **Findings of No Significant Impact (FONSI).** What is the level of detail of information that must be included in a finding of no significant impact (FONSI)?

A. The FONSI is a document in which the agency briefly explains the reasons why an action will not have a significant effect on the human environment and, therefore, why an EIS will not be prepared. *Section 1508.13.* The finding itself need not be detailed, but must succinctly state the reasons for deciding that the action will have no significant environmental effects, and, if relevant, must show which factors were weighted most heavily in the determination. In addition to this statement, the FONSI must include, summarize, or attach and incorporate by reference, the environmental assessment.

37b. What are the criteria for deciding whether a **FONSI** should be made available for **public review** for 30 days before the agency's final determination whether to prepare an EIS?

A. Public review is necessary, for example, (a) if the proposal is a borderline case, i.e., when there is a reasonable argument for preparation of an EIS; (b) if it is an unusual case, a new kind of action, or a precedent setting case such as a first intrusion of even a minor development into a pristine area; (c) when there is either scientific or public controversy over the proposal; or (d) when it involves a proposal which is or is closely similar to one which normally requires preparation of an EIS. *Sections 1501.4(e)(2), 1508.27*. Agencies also must allow a period of public review of the FONSI if the proposed action would be located in a floodplain or wetland. E.O. 11988, Sec. 2(a)(4); E.O. 11990, Sec. 2(b).

38. **Public Availability of EAs v. FONSIs.** Must (EAs) and FONSIs be made public? If so,

how should this be done?

A. Yes, they must be available to the public. Section 1506.6 requires agencies to involve the public in implementing their NEPA procedures, and this includes public involvement in the preparation of EAs and FONSIs. These are public "environmental documents" under Section 1506.6(b), and, therefore, agencies must give public notice of their availability. A combination of methods may be used to give notice, and the methods should be tailored to the needs of particular cases. Thus, a Federal Register notice of availability of the documents, coupled with notices in national publications and mailed to interested national groups might be appropriate for proposals that are national in scope. Local newspaper notices may be more appropriate for regional or site-specific proposals.

The objective, however, is to notify all interested or affected parties. If this is not being achieved, then the methods should be reevaluated and changed. Repeated failure to reach the interested or affected public would be interpreted as a violation of the regulations.

39. **Mitigation Measures Imposed in EAs and FONSIs.** Can an EA and FONSI be used to impose enforceable mitigation measures, monitoring programs, or other requirements, even though there is no requirement in the regulations in such cases for a formal Record of Decision?

A. Yes. In cases where an environmental assessment is the appropriate environmental document, there still may be mitigation measures or alternatives that would be desirable to consider and adopt even though the impacts of the proposal will not be "significant." In such cases, the EA should include a discussion of these measures or alternatives to "assist [46 FR 18038] agency planning and decisionmaking" and to "aid an agency's compliance with [NEPA] when no environmental impact statement is necessary." *Section 1501.3(b), 1508.9(a)(2).* The appropriate mitigation measures can be imposed as enforceable permit conditions, or adopted as part of the agency final decision in the same manner mitigation measures are adopted in the formal Record of Decision that is required in EIS cases.

40. **Propriety of Issuing EA When Mitigation Reduces Impacts.** If an environmental assessment indicates that the environmental effects of a proposal are significant but that, with mitigation, those effects may be reduced to less than significant levels, may the agency make a finding of no significant impact rather than prepare an EIS? Is that a legitimate function of an EA and scoping?

[**N.B.:** Courts have disagreed with CEQ's position in Question 40. The 1987-88 CEQ Annual Report stated that CEQ intended to issue additional guidance on this topic. Ed. note.]

A. Mitigation measures may be relied upon to make a finding of no significant impact only if they are imposed by statute or regulation, or submitted by an applicant or agency as part of the original proposal. As a general rule, the regulations contemplate that agencies should use a broad approach in defining significance and should not rely on the possibility of mitigation as an excuse to avoid the EIS requirement. *Sections 1508.8, 1508.27.*

If a proposal appears to have adverse effects which would be significant, and certain

mitigation measures are then developed during the scoping or EA stages, the existence of such possible mitigation does not obviate the need for an EIS. Therefore, if scoping or the EA identifies certain mitigation possibilities without altering the nature of the overall proposal itself, the agency should continue the EIS process and submit the proposal, and the potential mitigation, for public and agency review and comment. This is essential to ensure that the final decision is based on all the relevant factors and that the full NEPA process will result in enforceable mitigation measures through the Record of Decision.

In some instances, where the proposal itself so integrates mitigation from the beginning that it is impossible to define the proposal without including the mitigation, the agency may then rely on the mitigation measures in determining that the overall effects would not be significant (e.g., where an application for a permit for a small hydro dam is based on a binding commitment to build fish ladders, to permit adequate down stream flow, and to replace any lost wetlands, wildlife habitat and recreational potential). In those instances, agencies should make the FONSI and EA available for 30 days of public comment before taking action. *Section 1501.4(e)(2).*

Similarly, scoping may result in a redefinition of the entire project, as a result of mitigation proposals. In that case, the agency may alter its previous decision to do an EIS, as long as the agency or applicant resubmits the entire proposal and the EA and FONSI are available for 30 days of review and comment. One example of this would be where the size and location of a proposed industrial park are changed to avoid affecting a nearby wetland area.

ENDNOTES

The first endnote appeared in the original Federal Register. The other endnotes are for information only.

1. References throughout the document are to the Council on Environmental Quality's Regulations For Implementing The Procedural Provisions of the National Environmental Policy Act. 40 CFR Parts 1500-1508.
2. [46 FR 18027] indicates that the subsequent text may be cited to 48 Fed. Reg. 18027 (1981). Ed Note.
3. Q20 Worst Case Analysis was withdrawn by final rule issued at 51 Fed. Reg. 15618 (Apr. 25. 1986); textual errors corrected 51 F.R. p. 16,846 (May 7, 1986). The preamble to this rule is published at ELR Admin. Mat. 35055.

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Supporting Online Material
www.sciencemag.org/cgi/content/full/333/6045/1019/DC1
Materials and Methods
Figs. S1 to S9
Tables S1 to S12
References (*29–49*)

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Rapid Range Shifts of Species Associated with High Levels of Climate Warming

I-Ching Chen,[1,2] Jane K. Hill,[1] Ralf Ohlemüller,[3] David B. Roy,[4] Chris D. Thomas[1]*

The distributions of many terrestrial organisms are currently shifting in latitude or elevation in response to changing climate. Using a meta-analysis, we estimated that the distributions of species have recently shifted to higher elevations at a median rate of 11.0 meters per decade, and to higher latitudes at a median rate of 16.9 kilometers per decade. These rates are approximately two and three times faster than previously reported. The distances moved by species are greatest in studies showing the highest levels of warming, with average latitudinal shifts being generally sufficient to track temperature changes. However, individual species vary greatly in their rates of change, suggesting that the range shift of each species depends on multiple internal species traits and external drivers of change. Rapid average shifts derive from a wide diversity of responses by individual species.

Threats to global biodiversity from climate change (*1-8*) make it important to identify the rates at which species have already responded to recent warming. There is strong evidence that species have changed the timing of their life cycles during the year and that this is linked to annual and longer-term variations in temperature (*9–12*). Many species have also shifted their geographic distributions toward higher latitudes and elevations (*13–17*), but this evidence has previously fallen short of demonstrating a direct link between temperature change and range shifts; that is, greater range shifts have not been demonstrated for regions with the highest levels of warming.

We undertook a meta-analysis of available studies of latitudinal (Europe, North America, and Chile) and elevational (Europe, North America, Malaysia, and Marion Island) range shifts for a range of taxonomic groups (*18*) (table S1). We considered $N = 23$ taxonomic group × geographic region combinations for latitude, incorporating 764 individual species responses, and $N = 31$ taxonomic group × region combinations for elevation, representing 1367 species responses. For the purpose of analysis, the mean shift across all species of a given taxonomic group, in a given region, was taken to represent a single value (for example, plants in Switzerland or birds in New York State; table S1) (*18*).

The latitudinal analysis revealed that species have moved away from the Equator at a median rate of 16.9 km decade^{-1} (mean = 17.6 km decade^{-1}, SE = 2.9, $N = 22$ species group × region combinations, one-sample t test versus zero shift, $t = 6.10$, $P < 0.0001$). Weighting each study by the √(number of species) in the group × region combination gave a mean rate of 16.6 km decade^{-1}. For elevation, there was a median shift to higher elevations of 11.0 m uphill decade^{-1} (mean = 12.2 m decade^{-1}, SE = 1.8, $N = 30$ species groups × regions, one-sample t test versus zero shift, $t = 7.04$, $P < 0.0001$). Weighting elevation studies by √(number of species) gave a mean rate of uphill movement of 11.1 m decade^{-1}.

A previous meta-analysis (*14*) of distribution changes analyzed individual species, rather than the averages of taxonomic groups × regions that we used, and also included data on latitudinal and elevational shifts in the same analysis (*18*). It concluded that ranges had shifted toward higher latitudes at 6.1 km decade^{-1} and to higher elevations at 6.1 m decade^{-1} (*14*), whereas the rates of range shift that we found were significantly greater [$N = 22$ species groups × regions, one-sample t test versus 6.1 km decade^{-1}, $t = 3.99$, $P = 0.0007$ for latitude; $N = 30$ groups × regions, one-sample t test versus 6.1 m decade^{-1}, $t = 3.49$, $P = 0.002$ for elevation (*18*)]. Our estimated mean rates are approximately three and two times higher than those in (*14*), for

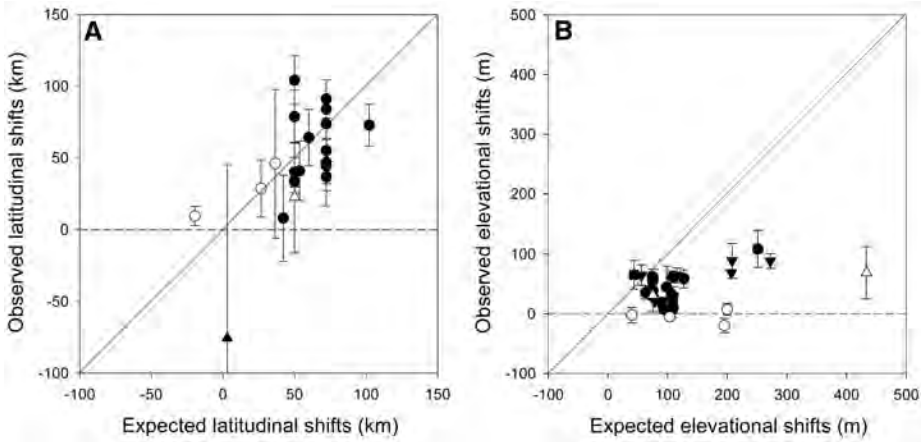

Fig. 1. Relationship between observed and expected range shifts in response to climate change, for (**A**) latitude and (**B**) elevation. Points represent the mean responses (±SE) of species in a particular taxonomic group, in a given region. Positive values indicate shifts toward the pole and to higher elevations. Diagonals represent 1:1 lines, where expected and observed responses are equal. Open circles, birds; open triangles, mammals; solid circles, arthropods; solid inverted triangles, plants; solid square, herptiles; solid diamond, fish; solid triangle, mollusks.

[1]Department of Biology, University of York, Wentworth Way, York YO10 5DD, UK. [2]Biodiversity Research Center, Academia Sinica, 128 Academia Road, Section 2, Nankang Taipei 115, Taiwan. [3]School of Biological and Biomedical Sciences, and Institute of Hazard, Risk and Resilience, Durham University, South Road, Durham DH1 3LE, UK. [4]Centre for Ecology & Hydrology, Crowmarsh Gifford, Wallingford, Oxfordshire, OX10 8BB, UK.

*To whom correspondence should be addressed. E-mail: chris.thomas@york.ac.uk

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latitude and elevation respectively, implying much greater responses of species to climate warming than previously reported (*18*). Most of the data we analyzed are from the temperate zone and from tropical mountains (table S1), where ecosystems are at least partly temperature-limited; different rates of change might be observed in moisture-limited ecosystems (*19*).

Published studies have shown nonrandom latitudinal and elevational changes (*1, 7, 13–17*) but have not previously demonstrated a statistical linkage between range shifts and levels of warming. We found that observed latitudinal and elevational shifts (the latter more weakly) have been significantly greater in studies with higher levels of warming (mean latitudinal shift versus average temperature increase; $N = 23$ species groups × regions, Pearson correlation coefficient (r) = 0.59, $P = 0.003$; mean elevational shift versus temperature increase; $N = 31$, $r = 0.37$, $P = 0.042$). Temperature gradients differ across the world, so a given level of warming leads to different expected range shifts of species in different regions (*20*), assuming that species track climate changes. To estimate the expected shifts, we calculated the distances in latitude (kilometers) and elevation (meters) that species in a given region would have been required to move to track temperature changes and thus to experience the same average temperature at the end of the recording period as encountered at the start (*18*) (table S1). We found that both observed latitudinal and elevation range shifts were correlated with predicted distances (Fig. 1A, $N = 20$ species groups × regions, $r = 0.65$, $P = 0.002$ for latitude; Fig. 1B, $N = 30$ groups × regions, $r = 0.39$, $P = 0.035$ for elevation), so our analyses directly link terrestrial range shifts to regional and study differences in the warming experienced.

Despite reports that many species lag behind climate change (*21–23*), nearly as many studies of observed latitudinal changes fall above as below the observed = expected line in Fig. 1A (9 points above, 11 below; $\chi^2 = 0.20$, 1 df, $P = 0.65$), suggesting that mean latitudinal shifts are not consistently lagging behind the climate. The lag in elevation response (Fig. 1B; 2 points above the 1:1 line, 28 below; $\chi^2 = 22.53$, 1 df, $P < 0.001$) is equally surprising because the required distances to track climate are much shorter than for latitudinal shifts (*20*). Real and apparent elevation lags may arise if suitable new conditions at higher elevations occur only in locations that cannot be reached easily (for example, on other mountain peaks), or they may reflect the topographic and microclimatic complexity of mountainous terrain [for example, cooler locations may be on poleward-facing slopes rather than higher (*24*)]; the need for finer-resolution analyses (*25*); and additional topographic, climatic, geological, and ecological constraints [for example, causing declines in cloud forest species (*26–28*)].

Taxonomic differences are not consistent predictors of recent response rates. For example, birds seem to have responded least in terms of elevational shifts but had a slightly greater than expected latitudinal shift (Fig. 1). Much greater variation is associated with differences among species within a taxonomic group than between taxonomic groups (Fig. 2 and table S2). For latitudinal studies, on average 22% (average of $N = 23$ species groups × regions) of the species actually shifted in the opposite direction to that expected. Similarly, 25% of species shifted downhill rather than to higher elevations (average of $N = 29$ species groups × regions). Thus, despite an overall significant shift toward higher latitudes and elevations, which is greatest where the climate has warmed the most, and despite around three-quarters of species shifting poleward and to higher elevations, we found that species have exhibited a high diversity of range shifts in recent decades.

At least three processes are likely to generate the high diversity of range shifts among species: time delays in species' responses, individualistic physiological constraints, and alternative and interacting drivers of change. Species may lag behind climate change if they are habitat specialists or immobile species that cannot colonize across fragmented landscapes (*17*, *21–23*), or if they possess other traits associated with low extinction or colonization rates (*29*). Species may also show individualistic physiological responses to different aspects of the climate, such as different sensitivities to maximum and minimum temperatures at critical times of their life cycles. These sensitivities will combine with variable wait times for different novel climatic extremes to take place (*30*). Species are also affected to different extents by nonclimatic factors and by multispecies interactions, which themselves depend on a diversity of environmental drivers (*21*, *28*). For example, a species might retreat toward the Equator at its poleward margin if it contracts with habitat loss faster than it expands through climate warming; whereas the poleward range margin of a species that thrives in novel agricultural landscapes may spread at a rate exceeding that expected, were warming the sole driver.

We found that rates of latitudinal and elevational shifts are substantially greater than reported

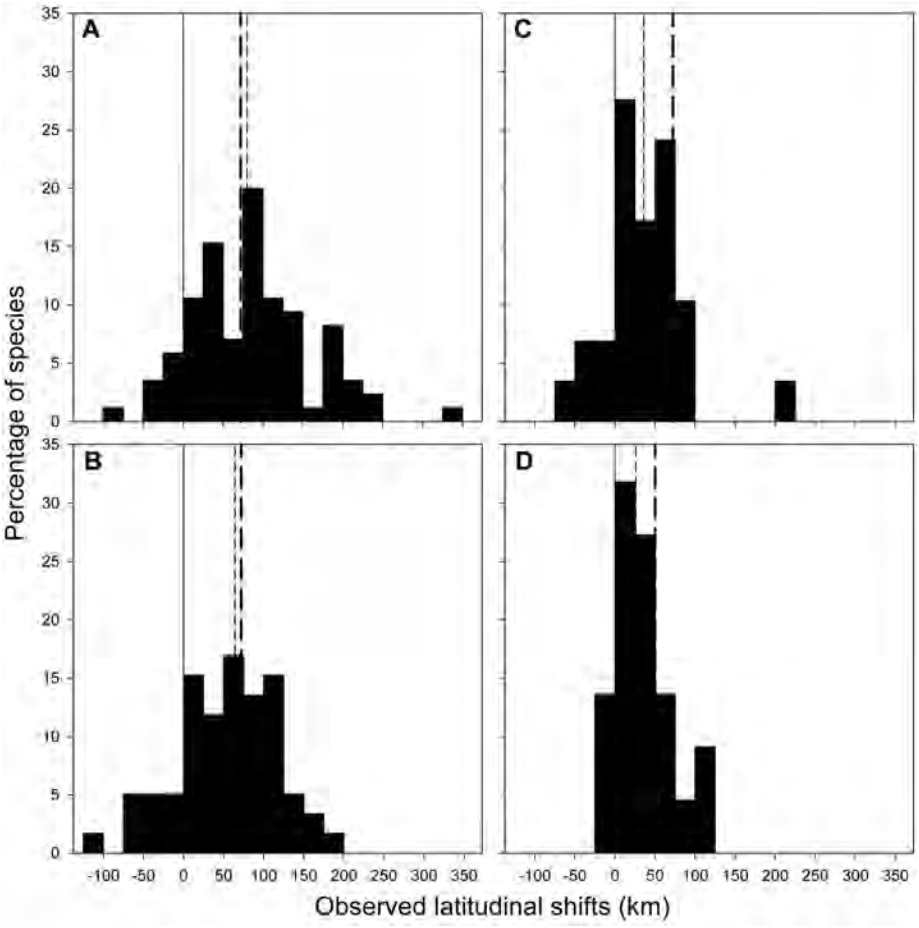

Fig. 2. Observed latitudinal shifts of the northern range boundaries of species within four exemplar taxonomic groups, studied over 25 years in Britain. (**A**) Spiders (85 species), (**B**) ground beetles (59 species), (**C**) butterflies (29 species), and (**D**) grasshoppers and allies (22 species). Positive latitudinal shifts indicate movement toward the north (pole); negative values indicate shifts toward the south (Equator). The solid line shows zero shift, the short-dashed line indicates the median observed shift, and the long-dashed line indicates the predicted range shift.

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in a previous meta-analysis, and increase with the level of warming. We conclude that average rates of latitudinal distribution change match those expected on the basis of average temperature change, but that variation is so great within taxonomic groups that more detailed physiological, ecological and environmental data are required to provide specific prognoses for individual species.

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Supporting Online Material

www.sciencemag.org/cgi/content/full/333/6045/1024/DC1
Materials and Methods
Tables S1 and S2
References (*31*–*51*)

1 April 2011; accepted 6 July 2011
10.1126/science.1206432

Aneuploidy Drives Genomic Instability in Yeast

Jason M. Sheltzer,[1] Heidi M. Blank,[1] Sarah J. Pfau,[1] Yoshie Tange,[2] Benson M. George,[1] Timothy J. Humpton,[1] Ilana L. Brito,[3] Yasushi Hiraoka,[2,4] Osami Niwa,[5] Angelika Amon[1]*

Aneuploidy decreases cellular fitness, yet it is also associated with cancer, a disease of enhanced proliferative capacity. To investigate one mechanism by which aneuploidy could contribute to tumorigenesis, we examined the effects of aneuploidy on genomic stability. We analyzed 13 budding yeast strains that carry extra copies of single chromosomes and found that all aneuploid strains exhibited one or more forms of genomic instability. Most strains displayed increased chromosome loss and mitotic recombination, as well as defective DNA damage repair. Aneuploid fission yeast strains also exhibited defects in mitotic recombination. Aneuploidy-induced genomic instability could facilitate the development of genetic alterations that drive malignant growth in cancer.

Whole-chromosome aneuploidy—or a karyotype that is not a multiple of the haploid complement—is found in greater than 90% of human tumors and may contribute to cancer development (*1*, *2*). It has been suggested that aneuploidy increases genomic instability, which could accelerate the acquisition of growth-promoting genetic alterations (*1*, *3*). However, whereas aneuploidy is a result of genomic instability, there is at present limited evidence as to whether genomic instability can be a consequence of aneuploidy itself. To test this possibility directly, we assayed chromosome segregation fidelity in 13 haploid strains of *Saccharomyces cerevisiae* that carry additional copies of single yeast chromosomes (*4*). These aneuploid strains (henceforth disomes) display impaired proliferation and sensitivity to conditions that interfere with protein homeostasis (*4*, *5*). We measured the segregation fidelity of a yeast artificial chromosome (YAC) containing human DNA and found that the rate of chromosome missegregation was increased in 9 out of 13 disomic strains relative to a euploid control (Fig. 1A). The increase ranged from 1.7-fold to 3.3-fold, comparable to the fold increase observed in strains lacking the kinetochore components Chl4 or Mcm21. Consistent with chromosome segregation defects, 8 out of 13 disomic strains displayed impaired proliferation on plates containing the microtubule poison benomyl, including a majority of the strains that had increased rates of YAC loss (Fig. 1B).

Chromosome missegregation can result from defects in chromosome attachment to the mitotic spindle or from problems in DNA replication or repair. Defects in any of these processes delay mitosis by stabilizing the anaphase inhibitor Pds1 (securin) (*6*). Five out of five disomes (disomes V, VIII, XI, XV, and XVI) exhibited delayed degradation of Pds1 relative to wild type after release from a pheromone-induced G_1 arrest (Fig. 1C and fig. S1). Defective chromosome biorientation delays anaphase through the mitotic checkpoint component Mad2 (*6*). Deletion of *MAD2* had no effect on Pds1 persistence in four disomes, but eliminated this persistence in disome V cells (fig. S1). Disome V also delayed Pds1 degradation after release from a mitotic arrest induced by the microtubule poison nocodazole, which demonstrated that this strain exhibits a biorientation defect. Disome XVI, which displayed Mad2-independent stabilization of Pds1, recovered from nocodazole with wild-type kinetics (fig. S2). Thus, Pds1 persistence results predominantly from Mad2-independent defects in genome replication and/or repair (see below).

We next investigated whether aneuploidy could affect the rate of forward mutation. Disomes V, VIII, X, and XIV displayed an increased mutation rate at two independent loci, whereas disome IV displayed an increased mutation rate at *CAN1* but not at *URA3* (Fig. 2A). The fold increase ranged from 2.2-fold to 7.1-fold, less than the 9.5-fold and 12-fold increases observed in a recombination-deficient *rad51*Δ mutant and a mismatch repair–deficient *msh2*Δ mutant, respectively. Additionally, in an assay for microsatellite instability, we found that disomes VIII and XVI displayed increased instability in a poly(GT) tract (fig. S3), which demonstrated that aneuploidy can enhance both simple sequence instability and forward mutagenesis.

To define the mechanism underlying the increased mutation rate in aneuploid cells, we

[1]David H. Koch Institute for Integrative Cancer Research and Howard Hughes Medical Institute (HHMI), Massachusetts Institute of Technology, Cambridge, MA 02139, USA. [2]Graduate School of Frontier Biosciences, Osaka University 1-3 Yamadaoka, Suita 565-0871, Japan. [3]Department of Ecology, Evolution and Environmental Biology, Columbia University, New York, NY 10027, USA. [4]Kobe Advanced ICT Research Center, National Institute of Information and Communications Technology 588-2 Iwaoka, Iwaoka-cho, Nishi-ku, Kobe 651-2492, Japan. [5]The Rockefeller University, 1230 York Avenue, New York, NY 10065, USA.

*To whom correspondence should be addressed. E-mail: angelika@mit.edu

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Science

Rapid Range Shifts of Species Associated with High Levels of Climate Warming

I-Ching Chen, Jane K. Hill, Ralf Ohlemüller, David B. Roy and Chris D. Thomas

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City of Gustavus
P.O. Box 1
Gustavus, AK 99826
Phone: (907) 697-2451

November 20, 2018

Tongass National Forest
Earl Stewart, Forester Supervisor
648 Mission Street
Ketchikan, Alaska 99901

Subject: **Roadless Rule Exemption Proposal for Alaska**

Reference: City of Gustavus Resolution CY 19-31 Commenting on the USFS Roadless Rule Proposal

The City of Gustavus and the community generally have long advocated thoughtful forest management and protection for the Tongass National Forest, and particularly for lands in our proximity in the Icy Strait region. In our view, timber should be made available for regional use, but on a small, sustainable scale, and in ways that fully protect other values. We are proud to say that our efforts, in concert with other Alaskans, have resulted in a framework of protection and use that makes our region an example of sustainability. One does not have to go far to find examples of more destructive practices.

The State of Alaska is now proposing to the US Forest service that a form of this protection, the Roadless Rule, be revised in ways that could reopen certain places along Icy Strait to large-scale logging. We oppose that. The attached Resolution # CY 19-31 provides the City's general rationale in that regard. The purpose of this letter is to spell out our history with, and values we attach to, areas and issues of particular concern.

HOMESHORE (Point Couverden)

This area's name refers to its long use and residency by local people, which speaks to its values in a host of ways. Large-scale logging in this area at the northeastern margin of Icy Strait began over 30 years ago. Gustavus opposed the scale and subsidization of the logging and roading that has gone on there but has been more supportive of the small-scale sales granted to local mills in recent years. The initial road system was terminated near Point Couverden, with the understanding that it might someday be extended northeastward into what is classified under the present Tongass Land Use Management Plan (TLMP) as Old-Growth Habitat. The City would oppose that extension on the basis that the area has already been too heavily exploited.

POINT ADOLPHUS/MUD BAY/IDAHO INLET

Gustavus advocated successfully for designation of this area as "legislative LUD II" in the Tongass Timber Reform Act, and later for its inclusion in TLMP and as a Roadless Area. It includes numerous fish streams, anchorages, deer hunting areas, campsites, berry patches, and summer tourist visitation sites of great value to our community and others along Icy Strait. In our view, this area (and all LUD II areas) have been given formal protection by Congress and should certainly not be considered for exclusion from roadless designation.

CHICKEN CREEK

This large watershed across Icy Strait from Gustavus contains the most extensive stands of high-volume timber left in our region. The creek system is host to large runs of pink, chum, and especially coho salmon. Its shores are of value to locals for recreation, deer hunting, and, in spring, wild plant gathering. Its value as old-growth habitat, already high, is raised in importance because the lands to eastward have been so heavily logged and roaded by Sealaska Corporation. The community worked hard and successfully to stop a large timber sale there about twenty years ago, and to get the watershed designated in TLMP as Old-Growth Habitat. However, this designation is strictly administrative. At present, it is also protected by the Roadless Rule, and we feel strongly it should stay that way.

CLIMATE CHANGE/ECOSYSTEM SERVICES

Climate change is a fact. The urgency of slowing and eventually stopping the drivers for climate change have been made all the more clear in the Oct., 2018 Intergovernmental Panel on Climate Change Special Report *Global Warming of 1.5°C*. As noted in the City's resolution, the amount of stored carbon in our intact old-growth forest and soils of the Tongass National Forest represents one of the greatest carbon stores in the world. Intact roadless areas on the Tongass are essential for maintaining America's resilience and slowing down climate change throughout the world, and they should not be diminished. Additionally, the ecosystem services—clean water, clean air, fish and animal habitat, etc.—that the Tongass provides 24/7 at no cost to taxpayers must be a major consideration in the management of our forest. The Roadless Rule serves this purpose admirably.

City of Gustavus
P.O. Box 1
Gustavus, AK 99826
Phone: (907) 697-2451

CLOSING REMARKS

Many of us in Gustavus are dismayed that the Forest Service, despite overwhelming public support for the Roadless Rule, is working to relax that rule. In doing so, the Forest Service is severely jeopardizing its relationship with the communities it is supposed to serve. People need to be confident that forest management decisions have a sound scientific basis, not—as is the case with the attempt to relax the roadless rule—the whims of politics or ideology. The damage to the Forest Service's credibility, even under the best of circumstances, may take decades to repair.

On behalf of the City Council of Gustavus, I urge that the USFS 2001 Roadless Rule should remain intact for Alaska and the Tongass National Forest Roadless Areas should retain the full protection of the existing rule.

Sincerely,

Calvin Casipit
Mayor, City of Gustavus, Alaska

Enclosure: (1) City of Gustavus, Alaska Resolution CY19-31
cc: Dave Schmid, Acting Regional Forester

CITY OF GUSTAVUS
RESOLUTION CY 19-31

A RESOLUTION CITY OF GUSTAVUS COMMENTING ON THE USFS ROADLESS RULE

WHEREAS, The City of Gustavus is a distinctive community that prospers by and through protection of its natural resources; and

WHEREAS, the sustainability and resilience of our Gustavus economy and our lifestyles depend upon nurturing and sustaining the natural resources of our forests, rivers, and ocean, locally and regionally in Southeast Alaska; and

WHEREAS, many Gustavus residents obtain their livelihood from environmentally dependent tourism and fishing activities; and

WHEREAS, the Roadless Rule was first adopted in 2001 to protect over 50 million acres of our nation's most essential and intact roadless areas in National Forests throughout the United States; and

WHEREAS, a USFS scoping meeting on the future of the Roadless Rule in Alaska held recently in Gustavus was well attended by Gustavus residents; and

WHEREAS, roadless areas on Tongass represent some of the most spectacular and unique roadless areas anywhere in the national forest system and support hunting, fishing, customary and traditional uses, unparalleled outdoor recreation opportunities, and opportunities for businesses; and

WHEREAS, the amount of stored carbon in our intact old-growth forest and soils of the Tongass National Forest represents one of the highest carbon stores in the world, and the conservation of intact inventoried roadless areas on the Tongass is essential for maintaining America's resilience and slowing down climate change throughout the world; and

WHEREAS, the Tongass provides a wide range of ecosystem services beyond carbon sequestration to residents and visitors alike on a 24/7 basis naturally and at no cost to the taxpayer; and

WHEREAS, increased logging of mature forest in the existing roadless areas of the Tongass will seriously impact the ability of the forest to provide these vital ecosystem services; and

WHEREAS, the community of Gustavus in the 1980s successfully advocated for wilderness status for Icy Strait islands and for the designation of the Chicken Creek watershed and much of northwest Chichagof Island as roadless wildlands; and

WHEREAS, the community of Gustavus has opposed the large-scale roading and logging at Homeshore (Point Couverden); and

WHEREAS, according to the State of Alaska's own economic experts, Tongass timber is uncompetitive because of fundamental, permanent changes that have occurred in global timber markets, high labor costs, distance from markets, and less-expensive substitutes; and

WHEREAS, timber from Tongass forest lands has always been sold at a significant loss to the US taxpayers; and

WHEREAS, 374 timber-related jobs in Southeast Alaska in 2017 represented only 0.78 percent of total Southeast Alaska employment; and

WHEREAS, if the USFS and the State of Alaska are interested in increasing employment on the Tongass, there are numerous opportunities to employ forest workers to restore forest ecosystems damaged by past logging activities;

THEREFORE BE IT RESOLVED, that, given the serious and long-lasting impacts of any reduction in current Roadless Rule protections on the Tongass, the City of Gustavus strongly supports lasting protection for all inventoried roadless areas on the Tongass National Forest as provided for in the 2001 Roadless Rule. There should be no exemption from the Roadless Rule for Alaska and there should be no Alaska-specific roadless rule; and

BE IT FURTHER RESOLVED, that the City of Gustavus urges that sustainable timber production opportunities be identified on federal, state, and private forest lands, particularly focusing on second-growth timber, outside the existing roadless areas, in support of a domestic wood products industry suitable in scale to the limited demand for products from this region; and

BE IT FINALLY RESOLVED, that the City of Gustavus urges the USFS to invest in programs and employment opportunities for Southeast residents to restore the health and productivity of Tongass forests and streams, as a means of supporting vital ecosystem services such as carbon sequestration, small-scale timber production, fish and wildlife populations, natural vistas, visitor amenities, subsistence, hydrological resources, and more.

The City of Gustavus submits the attached letter with expanded and specific comments for our local area.

PASSED and **APPROVED** by the Gustavus City Council this 19th day of November, 2018.

Calvin Casipit, Mayor

Karen Platt, City Clerk

City of Pelican

BOX 737 - PELICAN, ALASKA 99832 - PHONE: 735-2202/2203 - FAX: 735-2258 - EMAIL: cityhall@pelicancity.org - WEBSITE: www.pelican.net

EQUAL OPPORTUNITY EMPLOYER

CITY OF PELICAN RESOLUTION 2019 - 7

A RESOLUTION BY THE PELICAN CITY COUNCIL COMMENTING ON THE PROPOSED ALASKA ROADLESS RULE

WHEREAS, the City of Pelican is a unique community that prospers by and through the protection of its natural resources; and,

WHEREAS, the sustainability of the future of Pelican's economy and the lifestyles of our citizens depend on nurturing and sustaining the natural resources supported by our forests, rivers, and ocean, both locally and regionally in Southeast Alaska; and,

WHEREAS, many Pelican residents obtain their livelihoods from economic activities including commercial fishing and tourism, which are highly dependent on a pristine and productive natural setting; and,

WHEREAS, salmon populations are facing increased challenges from changing ocean conditions, climate change, ocean acidification, rising water temperatures in streams, and ocean warming events in the Pacific Ocean, and freshwater habitat development in the contiguous United States; and,

WHEREAS, Southeast Alaska is the most visited region in Alaska, with two thirds of all visitors to the State coming to this region and therefore acting as a significant contribution; and

WHEREAS, the Roadless Rule was first adopted in 2001 to protect 58 million acres of our nation's most essential and intact roadless areas in the National Forest system; and,

WHEREAS, the State of Alaska has been against the inclusion of the Tongass in the Roadless Rule since it was promulgated in 2001, despite support for the rule from a wide variety of constituents living in Southeast Alaska; and,

WHEREAS, during the scoping process for an Alaska Specific Roadless Rule in 2018, communities throughout Southeast Alaska support of the 2001 Roadless Rule, as well as the heavy majority of the official scoping comments received strongly support maintaining the 2001 Roadless Rule on the Tongass; and,

WHEREAS, the State of Alaska invested time, staff, and money to convene a stakeholder-led Citizen Advisory Committee in 2018 that provided recommendations and perspectives from different interest groups in Southeast Alaska on what a workable compromise for an Alaska Roadless Rule could look like; and,

WHEREAS, the pursuit of a full exemption from the 2001 Roadless Rule is highly divisive among local populations and stakeholders and increased conflict on the Tongass will tarnish the reputation of the region nationwide, with negative impacts for the tourism industry; and,

WHEREAS, roadless areas on the Tongass are essential to Southeast Alaska's way of life and represent some of the most spectacular and unique roadless areas anywhere in the National Forest System and support hunting, fishing, customary and traditional uses, subsistence, outdoor recreation opportunities, and opportunities for businesses; and,

WHEREAS, the amount of carbon stored in the intact old growth forests and soils of the Tongass National Forest represent one of the highest carbon stores in the world, and the conservation of intact roadless areas and old growth forests on the Tongass are essential for maintaining local and national climate resilience and slowing down climate change throughout the world; and,

WHEREAS, the Tongass provides a wide range of ecosystem services beyond carbon sequestration to residents and visitors alike on a 24/7 basis naturally and at no cost to the taxpayer, and;

WHEREAS, increased logging of old growth forests in the existing roadless areas of the Tongass seriously impact the ability of the forest to provide these vital ecosystem services, such as clean water, clean air, stream temperature regulation, and healthy salmon and wildlife habitat; and,

WHEREAS, increased logging of old growth forests in existing roadless areas would incur a significant cost to taxpayers, as new reports estimate that the subsidization of industrial-scale logging on the Tongass has cost taxpayers over $600 million over the past twenty years; and,

WHEREAS, according to the State of Alaska's own economic experts and commissioned reports, Tongass timber is uncompetitive due to fundamental, permanent changes that have occurred in global markets, high labor costs, distance from markets, and the availability of less-expensive substitutes; and,

WHEREAS, the commercial fishing industry and visitor industries that provide the economic backbone of Southeast Alaska depend on pristine, scenic vistas and healthy, productive watersheds; and,

WHEREAS, opening up roadless areas for future development industrial-scale logging will actively work against the Tongass Transition and the 2016 Tongass Land Management Plan Amendment, which was created after years of collaborative work between diverse groups of stakeholders;

THEREFORE, BE IT RESOLVED, that given long-lasting, fiscally irresponsible, and environmentally damaging impacts of any reduction in current Roadless Rule protections on the Tongass, the City of Pelican strongly supports lasting protection for all inventoried roadless areas on the Tongass National Forest as provided for in the 2001 Roadless Rule. There should be no exemption from the Roadless Rule for Alaska and there should be no Alaska-specific roadless rule; and,

BE IT FURTHER RESOLVED, that the economic livelihoods of City of Pelican residents would be irreparably harmed by the effects of a full exemption from the Roadless Rule on the Tongass, which would lead to increased climate change, increased threats to fish habitat, and development occurring in pristine natural areas that showcase the tourism potential and natural beauty of the Tongass National Forest; and,

BE IT FINALLY RESOLVED, that the City of Pelican encourages the USFS to invest in programs, workforce development, and employment opportunities for Southeast Alaskan residents to restore the health and productivity of the Tongass National Forest's degraded watersheds and streams, as a means of supporting vital ecosystem services such as carbon sequestration, small-scale timber production, fish and wildlife populations, natural vistas, visitor amenities, subsistence, hydrological resources, and more.

Passed and Approved by the Pelican City Council this 18th day of November, 2019.

Signed: _____
Walt Weller, Mayor

Attest: _____
Lattieca Stewart, City Clerk

CITY OF TENAKEE SPRINGS

Dan Kennedy
MAYOR
cityts@gmail.com

P.O. Box 52
Tenakee Springs, Alaska 99841
Phone 907-736-2207
Fax 907-736-2249

USDA Forest Service
Attn: Alaska Roadless Rule
P.O. Box 21628
Juneau, Alaska 99802

November 21, 2019

Gentlefolk:

I am submitting these comments and accompanying Resolution with the full support of the City Council of Tenakee Springs, Alaska.

The City of Tenakee Springs wishes to retain the existing Roadless Rule and supports selection of Alternative 1.

The City of Tenakee Springs has a long history of support for retaining the essential wild character of Tenakee Inlet and has supported the Roadless Rule since its earliest days. At both the local meetings with the Forest Service regarding proposed changes, 25-30 people attended and were 100% in favor of retaining the existing Roadless Rule. This may not sound like many people, but depending on the season, it represents between 30 and 60% of our adult population. It is obvious that we as a community, and across a diverse political spectrum, want to maintain Tenakee Inlet's wild character. This is not merely to protect our subsistence lifestyles. It is also essential to our economy which relies heavily on fishing and guiding.

Again, The City of Tenakee Springs supports Alternative 1, the no action alternative that retains the Roadless Rule as it currently exists for the Tongass National Forest.

Sincerely,

Dan Kennedy

Dan Kennedy
Mayor

City of Tenakee Springs
RESOLUTION 2020-07

In the Council
November 21, 2019

Introduced by
Council President

A RESOLUTION OF THE CITY OF TENAKEE SPRINGS, ALASKA EXPRESSING THE CITY OF TENAKEE SPRINGS' SUPPORT FOR ALTERNATIVE 1 OF THE DRAFT ENVIRONMENTAL IMPACT STATEMENT FOR THE MANAGEMENT OF INVENTORIED ROADLESS AREAS WITHIN THE TONGASS NATIONAL FOREST.

WHEREAS, the community of Tenakee Springs depends on fishing, guiding, and tourism as its main economic drivers; and

WHEREAS, all these activities depend on maintaining the wild character of the lands surrounding Tenakee Inlet; and

WHEREAS, in 2001, the U.S. Department of Agriculture (USDA) finalized a management plan for the Tongass National Forest, which includes the National Roadless Area Conservation Rule (Roadless Rule) in response to disappearing wildlands and a road maintenance backlog of $8.4 billion; and

WHEREAS, the intent of the Roadless Rule was to protect the social, economic, and ecological values and characteristics of inventoried roadless areas from road construction and reconstruction and certain timber harvest activities; and

WHEREAS, at the time of promulgation in 2001, the Roadless Rule had the most extensive public involvement of any rulemaking process in the history of federal rulemaking, generating over 1.6 million comments, the vast majority of which supported the creation of a strong national policy protecting roadless areas; and

WHEREAS, inventoried roadless areas provide large, relatively undisturbed blocks of important habitat for a variety of terrestrial and aquatic wildlife and plants; contribute to healthy watersheds and clean drinking water; and provide extensive opportunities for outdoor recreation and tourism; and

WHEREAS, the Roadless Rule protects some of the last remaining tracts of old-growth forest habitat in the Tongass that contribute to the absorption of global warming pollution and protect native animal species that rely on them; and

WHEREAS, roadless areas support subsistence lifestyles through hunting and fishing, and support outdoor recreation, commercial fisheries, and tourism industries; and

WHEREAS, the Roadless Rule contains exceptions that allow roadbuilding for community access, hydropower projects, utility connectors, and other economic development projects when they serve a legitimate public interest; and

WHEREAS, protection of these roadless areas in the Tongass National Forest is of local and national importance; and

WHEREAS, in 2018, the State of Alaska petitioned the USDA to craft a state-specific roadless rule and requested that a full exemption be pursued from the 2001 Roadless Rule; and

WHEREAS, in 2018 the USDA accepted the State of Alaska's petition and initiated a National Environmental Policy Act review, an environmental impact statement (EIS), and public rulemaking process to address the management of inventoried roadless areas within Alaska, and to evaluate the exemption set forth in the petition; and

WHEREAS, the great majority of comments received by the USDA following its notice of intent of proposed rulemaking opposed changing the Roadless Rule for Alaska; and

WHEREAS, all the citizens of Tenakee Springs who testified publicly at Forest Service hearings express unequivocal support for maintaining the Roadless Rules as they now stand; and

WHEREAS, the draft EIS provides an analysis of six alternatives related to roadless management in Alaska, ranging from no action to a full exemption from the Roadless Rule on the Tongass National Forest; and

WHEREAS, the USDA has identified Alternative 6, which is a full exemption, as the preferred alternative at this time; and

WHEREAS, the City of Tenakee Springs opposes Alternative 6 and feels that Alternative 1, which requests no action and would maintain the 2001 Roadless Rule on the Tongass National Forest, best protects public interests while also protecting the unique ecosystem of the Tongass; and

WHEREAS, the USDA is seeking public comment on the draft EIS from October 18, 2019 to December 17, 2019 and will issue its decision regarding and Alaska Roadless Rule in June of 2020;

NOW THEREFORE BE IT RESOLVED: That the City Council of the City of Tenakee Springs expresses its support for Alternative 1 and its opposition to Alternative 6, of the draft environmental impact statement for the management of inventoried roadless areas within the Tongass National Forest.

ADOPTED 6 Ayes, 1 Absent **THIS** 21st **DAY OF** November **2019**

Dan Kennedy
City Council President
Ex officio MAYOR

ATTEST:

Mary Almy
City Clerk

Evolution, 55(7), 2001, pp. 1408–1418

RAPID EVOLUTION IN THE *NEBRIA GREGARIA* GROUP (COLEOPTERA: CARABIDAE) AND THE PALEOGEOGRAPHY OF THE QUEEN CHARLOTTE ISLANDS

T. E. CLARKE,[1,2] D. B. LEVIN,[1,3] D. H. KAVANAUGH,[4,5] AND T. E. REIMCHEN[1,6,7]
[1]*Department of Biology, University of Victoria, P.O. Box 3020, Victoria, British Columbia, V8W 3N5, Canada*
[2]*E-mail: tec8435@ksu.edu*
[3]*E-mail: dlevin@uvic.ca*
[4]*Department of Entomology, California Academy of Sciences, Golden Gate Park, San Francisco, California 94118*
[5]*E-mail: dkavanaugh@calacademy.org*
[6]*E-mail: reimchen@uvic.ca*

Abstract.—Morphological differentiation in the ground beetles of the *Nebria gregaria* group, found on the Queen Charlotte Islands, has been used as support for the glacial refugium proposed for the northwest coast of North America. Two members of this species group, *N. charlottae* and *N. louiseae,* are restricted to cobble beaches in this archipelago. A third, *N. haida,* is found only in alpine regions of the archipelago and the adjacent mainland. The remaining two species of the *gregaria* group, *N. lituyae* and *N. gregaria,* show highly restricted distributions in the mountains of the Alaska panhandle and on the beaches of the Aleutian Islands, respectively. To determine the relationships of the five species, we conducted phylogenetic analyses on nucleotide sequence data obtained from five regions of the mitochondrial DNA. In total, 1835 bp were analyzed. The results suggest that one species, *N. lituyae,* does not belong in the *gregaria* group, and that only seven mutations separated the two most divergent of the four remaining species. We also conducted random amplified polymorphic DNA fingerprinting analyses on genomic DNA extracted from the five species. Analyses of genetic diversity revealed a lack of molecular differentiation among the Queen Charlotte species, suggesting that these populations may be postglacial in origin and that together *N. gregaria, N. charlottae, N. louiseae,* and *N. haida* might represent local variations of a single species. These results are consistent with conclusions derived for the morphological and genetical differentiation among *Gasterosteus* populations in the archipelago.

Key words.—Carabidae, glacial refugium, mitochondrial DNA, molecular evolution, phylogenetic analysis, Queen Charlotte Islands, random amplified polymorphic DNA.

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The glacial cycles of the late Pleistocene have defined the current biogeography of North America, affecting both the composition and distribution of the animal and plant species. The regular glacial advances and retreats have resulted in the movement of species across vast distances and the reorganization of many ecological communities in their wake. Glacial advances have occurred with a period of approximately every 100,000 years with the most recent (Wisconsin) ice age beginning approximately 70,000 years ago, reaching a maximum extent during the Frasier glaciation between 25,000 and 15,000 years ago. (Liu 1992). At the height of the Frasier glaciation ice sheets occupied much of the northern half of the continent, leaving only a scattering of localities along the coast uncovered (Blaise et al. 1990). These ice-free refugia existing along the edge of the ice sheets are known for their role in maintaining species diversity through the glaciations by sheltering species and subspecies that would otherwise have risked extirpation. Refugia have also contributed to the biogeography of North America by allowing these species to rapidly colonize adjacent land in the wake of the glacial retreat (Pielou 1991). Whether coastal refugia have contributed to speciation, however, is still a topic of debate. In the South American and African tropics, the periodic fluctuations in global temperature that have characterized the late Pleistocene have promoted speciation through the creation of ecological refugia—isolating populations in fragmented forest and grassland ecosystems (Haffer 1977; Colinvaux 1998; Danin 1999) or on mountains and highlands during periods of global warming (Fjeldsa and Lovett 1997; Roy 1997; Smith et al. 1997). The physical refugia generated by the ice sheets along the coast of northern North America may have played a similar role in promoting speciation by maintaining populations in isolation from each other for periods of several tens of thousands of years.

On the northwest coast of North America, glacial refugia may have existed on Kodiak Island (Karlstrom and Ball 1969), the Alaskan panhandle (Kavanaugh 1988), and the Queen Charlotte Islands (Foster 1965), with the evidence based primarily on taxa that are either endemic to the refugia or display disjunct distributions. For the Queen Charlotte Islands, the largest of the hypothesized western refugia, the assemblage of endemic, dispersing, and disjunct taxa has included mammals (Foster 1965), birds (Cowan 1989), insects (Hamilton 1982; Ferguson 1987; Kavanaugh 1989), plants (Schofield 1989; Taylor 1989), and lichens (Brodo 1995). Although the glacial geology of the islands has suggested no refugia other than possible coastal beaches and mountaintop nunataks (Brown and Nasmith 1962), recent data suggest a possible refugial source area in the now submerged continental shelf separating the archipelago from the continent, as lowered sea levels may have exposed wide stretches of this terrain during the glacial period (Barrie et al. 1993; Josenhans et al. 1995).

One criterion used in evaluating evidence for glacial refugia is the extent of morphological differentiation or endemism in local biota. Similarity between the source populations on the continent and those on the adjacent islands suggests recent, presumably postglacial colonization, where-

[7] Corresponding author.

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as marked dissimilarity suggests long-term separation. Yet the latter possibility pivots on the broad assumption that morphological change accrues slowly within isolated populations and that only the most minor of differences between populations were likely to have occurred in postglacial time (McCabe and Cowan 1945; Foster 1965). Under a more rigorous evaluation, however, the basis for this assumption would appear to be unfounded. Numerous examples of rapid postglacial morphological change have been documented, including the repeated evolution of benthic and limnetic morphs of fish (McPhail 1992) and size differences in insular rodent fauna (Angerbjorn 1986). On the Queen Charlotte Archipelago, studies on threespine stickleback demonstrate that the extreme population variation in body form and armor is an adaptive response to differences in the predator regimes among localities (Moodie and Reimchen 1976; Reimchen et al. 1985; Reimchen 1994), although there is evidence for deep phylogenetic branching among populations (O'Reilly et al. 1993; Deagle et al. 1996). Similarly, a molecular study of west coast black bears demonstrated that the morphologically distinct Queen Charlotte Islands subspecies (*Ursus americanus carlottae*) is very similar genetically to the coastal mainland bear population, indicating that the island and coastal subspecies have diverged very recently and that the morphological differences are likely an adaptation to differences in their respective habitats (Byun et al. 1997, 1999). The evidence that morphology can change rapidly, and has done so repeatedly through the postglacial period, necessitates a reexamination of the endemic Queen Charlotte Islands and western coastal taxa that have been used to support evidence for glacial refugia during the Fraser glaciation.

The ground beetles of the *Nebria gregaria* group, distributed intermittently along the northwestern coast of North America from the Queen Charlotte Islands to the Aleutians, are considered to be a taxon for which the repeated isolation of populations in glacial refugia has played an important role in the promotion of speciation (Kavanaugh 1992). Composed of five morphologically similar species, the members of the *Nebria gregaria* group display a remarkable degree of endemism to locations associated with hypothesized glacial refugia. The three Queen Charlotte Island species consist of *N. charlottae,* which is restricted to cobble beaches on Graham Island; *N. louiseae,* which is found in similar habitats on Moresby Island and many of the smaller islands on the southeast of the archipelago (Kavanaugh 1989); and *N. haida,* found only in alpine locations on Graham and Moresby Islands and on a mountaintop north of Prince Rupert, British Columbia (Kavanaugh 1992). Their closest relatives are *N. lituyae,* found in alpine areas in the Alaska panhandle, and *N. gregaria,* which occupies beaches on a number of the Aleutian Islands. Together, these five species are characterized by similar morphology, flightlessness, and a preference for low-temperature rocky habitats that contrasts with the riparian habitat and fully winged morphology of their wide-ranging sister taxa, *N. sahlbergii, N. acuta,* and *N. arkansana.* The five species of the *N. gregaria* group are believed to originated from a winged, riparian ancestor in the early Pleistocene and to have radiated in glacial refugia through the mid- to late-Pleistocene (Kavanaugh 1989). The earliest division within the group is thought to have been associated with habitat, grouping the alpine-dwelling species *N. lituyae* and *N. haida* into one lineage and the beach-dwelling species *N. gregaria, N. charlottae,* and *N. louiseae* into a second. Later periods of refugial isolation acted to produce the five species, with the most recent division between *N. gregaria* and *N. louiseae* believed to have occurred as a product of the Fraser glaciation (Fig. 1; Kavanaugh 1979a, 1989, 1992).

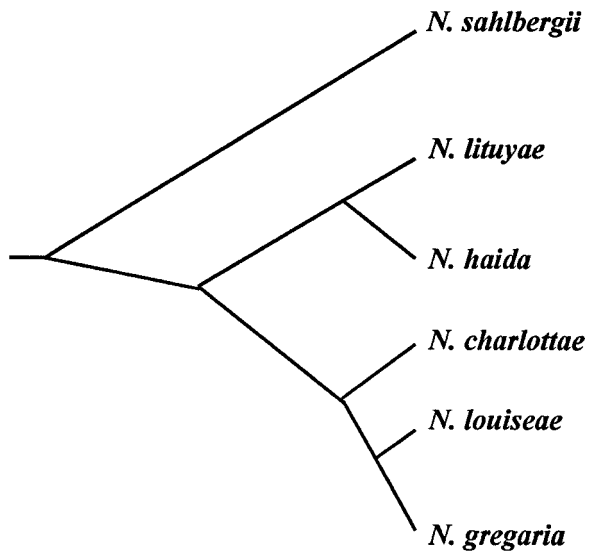

FIG. 1. Phylogeny of the *Nebria gregaria* infragroup (Kavanaugh 1989).

Evidence for morphological stasis and the extended periods of time required for reproductive isolation within the predacious ground beetles are based on data derived from paleoentomological studies of insect fossils from the high-arctic tundra, which show highly conserved morphology (Matthews 1980; Ashworth 1996). As such, the marginal morphological differentiation in the *gregaria* group is thought to represent multiple phases of speciation and stasis, beginning in the Early or Middle Pleistocene (Kavanaugh 1989). Yet, in view of the rapid morphological change observed in other Queen Charlotte taxa, we test here whether the diversification of the *gregaria* group is a recent event, either due to a rapid recolonization of northwest North America by a single *gregaria* species following the last glacial retreat, or the subdivision of a widespread ancestral species into Queen Charlotte, southwestern Alaskan, and Aleutian refugial populations during the most recent glacial advance. With no fossil evidence of long-term occupation of the Queen Charlotte Islands by the *gregaria* group and little probability that any such evidence could be found, given the extreme glacial conditions on the Pacific Northwest, we have used molecular markers to examine the phylogeny, age, and species status of these beetles to determine whether their biogeography indicates survival in a west coast refugium during the Fraser glaciation.

MATERIALS AND METHODS

Collections

Specimens of *Nebria* for use in DNA sequence analysis were collected into 80% ethanol from sites on the Queen

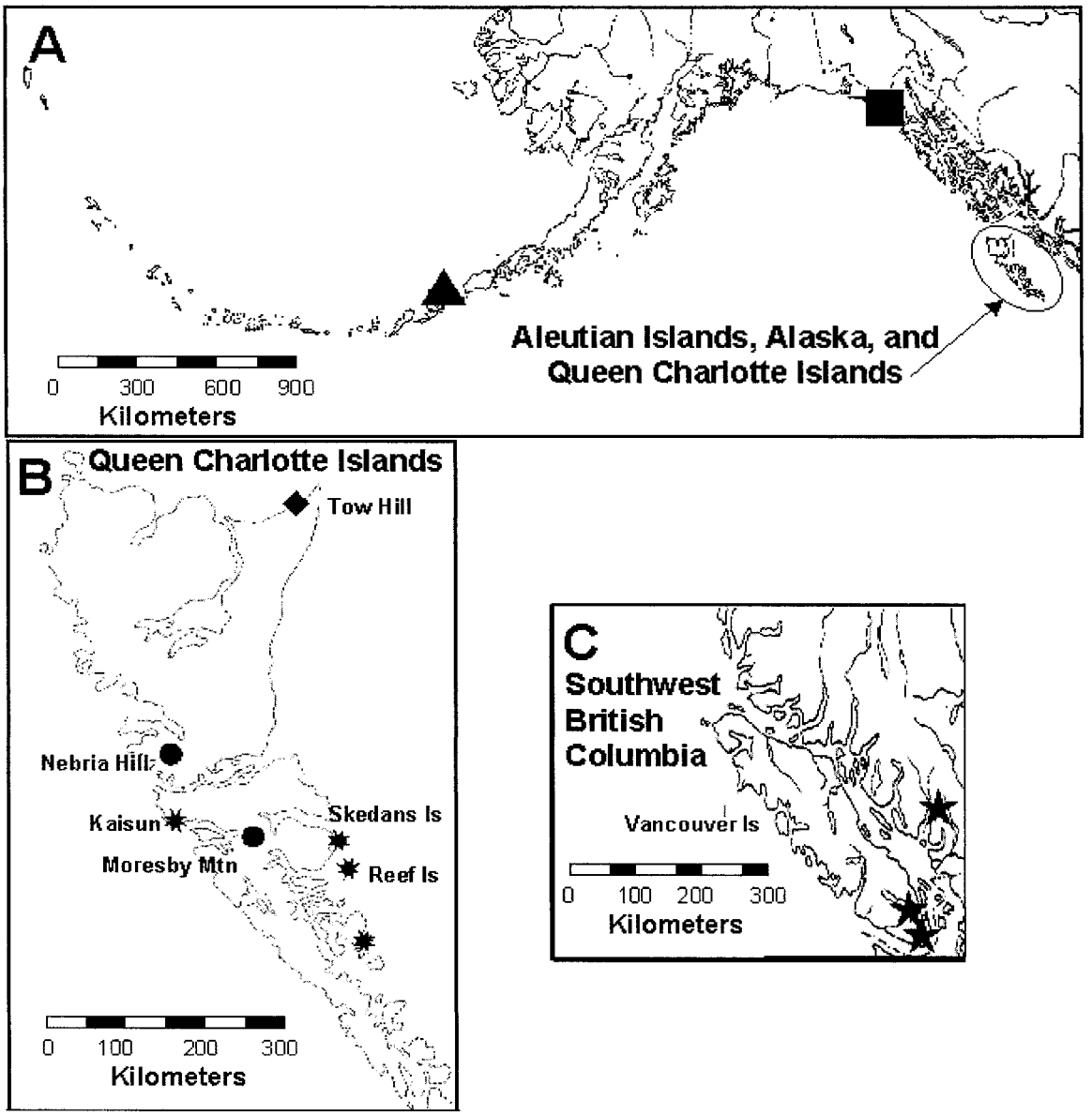

FIG. 2. Sites from which *Nebria* were obtained in western North America. (A) Map of collection sites of *Nebria gregaria* (▲) and *Nebria lituyae* (■) in the Aleutian Islands and Alaska, and the location of the Queen Charlotte Islands; (B) Collection sites of *Nebria charlottae* (♦), *Nebria haida* (●), and *Nebria louisea* (✹) on the Queen Charlotte Islands; (C) Collection sites of *Nebria sahlbergii sahlbergii* (★) in southwestern British Columbia, including Vancouver Island.

Charlotte Islands and the west coast of British Columbia (Fig. 2A). *Nebria charlottae* was obtained from cobble beaches at the base of Tow Hill and surrounding Estrado Lagoon, near Masset township, Graham Island (Fig. 2B). *Nebria louiseae* was collected from Kaisun village, Moresby Island, and from cobble beaches at Skedans and Reef Islands to the east of Moresby (Fig. 2B). *Nebria haida* was obtained from above the treeline on Nebria Hill, located south of Shields Bay, Graham Island, and from Mount Moresby, on north Moresby Island (Fig. 2B). *Nebria sahlbergii sahlbergii* and *N. gebleri* were both collected from stream banks in Manning Park, British Columbia (Fig. 2C). *Nebria diversa* was obtained from North Beach, Graham Island, and Cape Scott, Vancouver Island (Fig. 2B, C). One specimen each of *N. gregaria* collected from the Aleutian Islands and *N. lituyae* from the mountains near Juneau, Alaska (Fig. 2A) were obtained from the collection at the California Academy of Sciences. The *N. gregaria* specimen had been killed and stored in ethyl acetate before pinning, whereas the *N. lituyae* specimen was preserved in silica gel. Three specimens from each of the Tow Hill, Kaisun Village, Skedans Island, Reef Island, and Nebria Hill beetle populations as well as single specimens of *N. charlottae* from Masset, *N. louiseae* from Skedans Village site, *N. lituyae* and *N. gregaria* were selected for DNA extraction and nucleotide sequence analysis. Three specimens of *N. sahlbergii*, two of *N. gebleri*, and two each of the Queen

TABLE 1. Gene locus, direction of amplification, and nucleotide sequence of primers used for polymerase chain reaction amplification of *Nebria* mitochondrial DNA sequences.

NDI forward primer	5′-GCATCACAAAAGGCTGAGGA-3′
NDI reverse primer	5′-ACATGATCTGAGTTGAAACC-3′
COI forward primer (mtd-6)	5′-GGAGGATTTGGAAATTGATTAGTTCC-3′
COI reverse primer (mtd-9)	5′-CCCGGTAAAATTAAAATATAAACTTC-3′
COIIa forward primer (mtd-16)	5′-ATTGGACATCAATGATATTGA-3′
COIIa reverse primer (mtd-18)	5′-CCACAAATTTCTGAACATTGACCA-3′
COIIb forward primer (mtd-13)	5′-AATATGGCAGATTAGTGCA-3′
COIIb reverse primer (mtd-15)	5′-TCATAAGTTCARTATCATTG-3′
Cyt *b* forward primer (mtd-26)	5′-TATGTACTACCATGAGGACAAATATC-3′
Cyt *b* reverse primer (mtd-28)	5′-ATTACACCTCCTAATTTATTAGGAAT-3′

Charlotte and Cape Scott *N. diversa* were also processed for use as outgroups.

DNA Extraction

Beetle DNA used in polymerase chain reactions (PCRs) was extracted using either the nondestructive DNA extraction protocol of Phillips and Simon (1995) or the QIAamp tissue kit following the manufacturer's protocol (Qiagen, Mississauga, Ontario, Canada). The former method involves perforating dried beetles with a sterilized size 1 inset pin and extracting DNA with a solution of 8% dodecyltrimethyl-ammonium bromide (DTAB), 1.5 M NaCl, 100 mM Tris-HCl (pH 9.0), 50 mM EDTA at 68°C for 12 h, followed by chloroform extraction and DNA precipitation with 5% cetyltrimethylammonium bromide (CTAB) and 0.4 M NaCl. Pellets were resuspended in 1.2 M NaCl and reprecipitated with ethanol, followed by suspension in TE buffer (10 mM Tris-HCl; 1 mM EDTA, pH 7.5).

Sequence Analysis

Five regions of the mitochondrial DNA (mtDNA) were selected to be amplified, from total DNA extracted from the beetles, using the PCR. The regions of mtDNA amplifed were located inside the NADH subunit I gene (NDI region), within the cytochrome oxidase subunit I gene (COI), the cytochrome *b* gene (Cyt *b*), and within two sections of the cytochrome oxidase II gene (COIIa and COIIb; Table 1). PCR amplifications of fragments NDI and COI for beetles from Tow Hill, Kaisun Village, Nebria Hill, as well as all outgroup species were performed in 50-μl volumes containing 50–100 ng of beetle DNA, 83 ng of each primer, 200 μM of each deoxyribonucleotide triphosphate (dNTP), 15 mM Tris-HCl (pH 8.0), 2.5 mM $MgCl_2$, 60 mM KCl, and either 10 units of Taq DNA polymerase or 10 units of Pfu DNA polymerase, with 35 cycles of 94°C for 45 sec, 47°C for 60 sec, and 72°C for 60 sec. Fragments were ligated using T4 ligase (New England Biolabs, Beverly, CA) into the TA cloning vector pCR2.1 (Invitrogen, Carlsbad, CA). PCR amplifications of all other fragments were performed in 50-μl volumes containing Pfu polymerase (Stratagene, La Jolla, CA) and either $<$ 10 ng of beetle DNA (*N. gregaria*) or 50–100 ng of beetle DNA (all others), 83 ng of each primer, 200 μM of each dNTP, 20 mM Tris-HCl (pH 8.8), 10 mM KCl, 6 mM $(NH_4)_2SO_4$, 1.5 mM $MgCl_2$, 0.1% Triton X-100. Fragments were ligated into *Eco*RV (New England Biolabs) cut pBluescript (Stratagene) plasmid using T4 DNA ligase.

Sequencing was performed using the Sanger dideoxy sequencing method (Sanger et al. 1977) on Automated Laser Fluorescence sequencing machines (Pharmacia, Uppsala, Sweden) with fluorescene-labeled dideoxy nucleotides. To avoid sequence errors introduced by the PCR process, between three to five different clones were sequenced from fragments amplified with Taq polymerase. Because Pfu polymerase exhibits a much higher degree of fidelity then Taq polymerase, only two to three different clones were sequenced from fragments amplified with Pfu polymerase.

Individual sequences were assembled using the Seqman program in the Lasergene package (DNASTAR, Madison, WI). Homologous sequences from each beetle were aligned to each other using the Clustal V alignment option of the Megalign program (also from the Lasergene package). Genetic distance relationships between beetles were obtained from the output of the Megalign program, and cladistic analysis of the sequence data was performed using the PAUP 3.1 program (Swofford 1993). The species *N. gebleri* was used to root the trees (Kavanaugh 1978). Sequences are stored in Genbank under accession codes AF093761–AF093770, AF095196–AF09514, AF095468–AF095477, and AF096699–AF096708.

Random Amplified Polymorphic DNA Fingerprinting Analyses

The beetles used in the random amplified polymorphic DNA (RAPD) analysis were all selected from ethanol-collected material from the Queen Charlotte Islands, with the exception of the silica-gel-preserved *N. lituyae* specimen supplied by D. H. Kavanaugh (Table 2).

DNA for RAPD analysis was extracted from the beetles using the QIAamp tissue kit following the manufacturer's protocol (Qiagen). The concentration and purity of the DNA was tested using a spectrometer and by running samples on agarose gels. Aliquots of the extracted DNA were diluted with ddH_2O to a concentration of 6 ng/μl for use in the RAPD reactions. Ready-to-go RAPD analysis beads (Amersham Pharmacia Biotech, Piscataway, NJ) were used to perform the RAPD reactions. Each reaction contained 6 ng of genomic DNA, 25 pmol of a 10-mer oligonucleotide primer, and a RAPD bead containing of AmpiTAQ and Stoffel fragment thermostable DNA polymerases, 400 μM of each dNTP 0.1 g/ml BSA, 3 mM $MgCl_2$, 30 mM KCl, and 10 mM Tris pH 8.3 in a total reaction volume of 25 μl. The nucleotide sequences of primers used for the RAPD reactions were: 5′-

GGTGCGGGAA-3′, 5′-GTTTCGCTCC-3′, 5′-GTAGACCCGT-3′, and 5′-AAGAGCCCGT-3′. Reactions were overlaid with mineral oil to prevent evaporation. The thermocycler program used for the RAPD reactions was as follows: one cycle at 95°C for 5 min, followed by 45 cycles of 95°C for 1 min, 36°C for 1 min, and 72°C for 2 min. Bands were separated using 2% agarose gels, stained with ethidium bromide, and scored by hand from photographs and computer images taken of the gels. Each beetle sample was tested using each of four different primers in separate reactions. Because banding pattern produced by the RAPD reaction has been demonstrated to be sensitive to the reaction conditions (Innis and Gelfand 1990), RAPD reactions were standardized through the use of a manufactured kit containing preformulated amounts of enzyme and buffers and by using only high-purity extractions of DNA. Replicate RAPD reactions and blank control reactions were used to eliminate artifactual banding.

Rooted neighbor joining trees were constructed for the 47 samples using the Jaccard coefficient option of the RAPDistance package (ver. 1.04; Armstrong et al. 1994) to calculate genetic distance. The Jaccard coefficient calculates distance based on the number of shared bands divided by the sum of shared bands and mismatches. Alternative methods of determining genetic distance were calculated using two variants of the Jaccard coefficient, the Nei and Li coefficient, which doubles the value of shared bands to emphasize genetic similarities among taxa, and the Sokal and Sneath coefficient, which doubles the value of mismatches to emphasize differences between taxa. The average heterozygosity of each population was calculated using Nei's (1978) formula for estimating heterozygosity from small sample sizes. An analysis of molecular variance using the WINAMOVA (ver. 1.04; Excoffier 1992; Excoffier et al. 1992) was used to compare variation between the six different populations as well as among *N. sahlbergii sahlbergii* and the other five populations together.

Results

Mitochondrial DNA Sequence Analyses

Five fragments were amplified and sequenced from the mitochondrial DNA of the *Nebria* beetles for a total of 1835 bp, of which 214 bp were variable and 47 bp were phylogenetically informative. No differences in sequence could be found between beetles taken from the same population, with the exception of *N. louiseae,* for which three different haplotypes were discovered (designated KV, SKA, and SKB). KV and SKB haplotypes were restricted to populations from Kaisun Village and the islands east of Moresby Island, respectively, whereas the SKA haplotype was found at both sites.

Twenty-four base-pair substitutions occurred among the species of the *gregaria* species group, 135 between the *gregaria* species group and the species *N. diversa,* and 106 between the *gregaria* species group and *N. gebleri* (of 1363 bp sequenced for *N. gebleri*). Within the *gregaria* species group seven of the protein coding base-pair substitutions occurred at the first codon position and the remaining 15 occurred at the third codon position, with two mutations taking place in intergenic regions. The transition:transversion ratio (Ti:Tv) within the *gregaria* species group was 20:4. In contrast, 22 first codon position, nine second codon position, and 96 third codon position mutations occurred between *N. diversa* and the *gregaria* species group, with eight substitutions taking place in nonprotein coding regions of the DNA. Eighteen first, 13 second, and 64 third codon position substitutions occurred between *N. gebleri* and the *gregaria* species group. The Ti:Tv ratio was 78:57 between *N. diversa* and the *gregaria* species group and 53:51 between *N. gebleri* and the *gregaria* species.

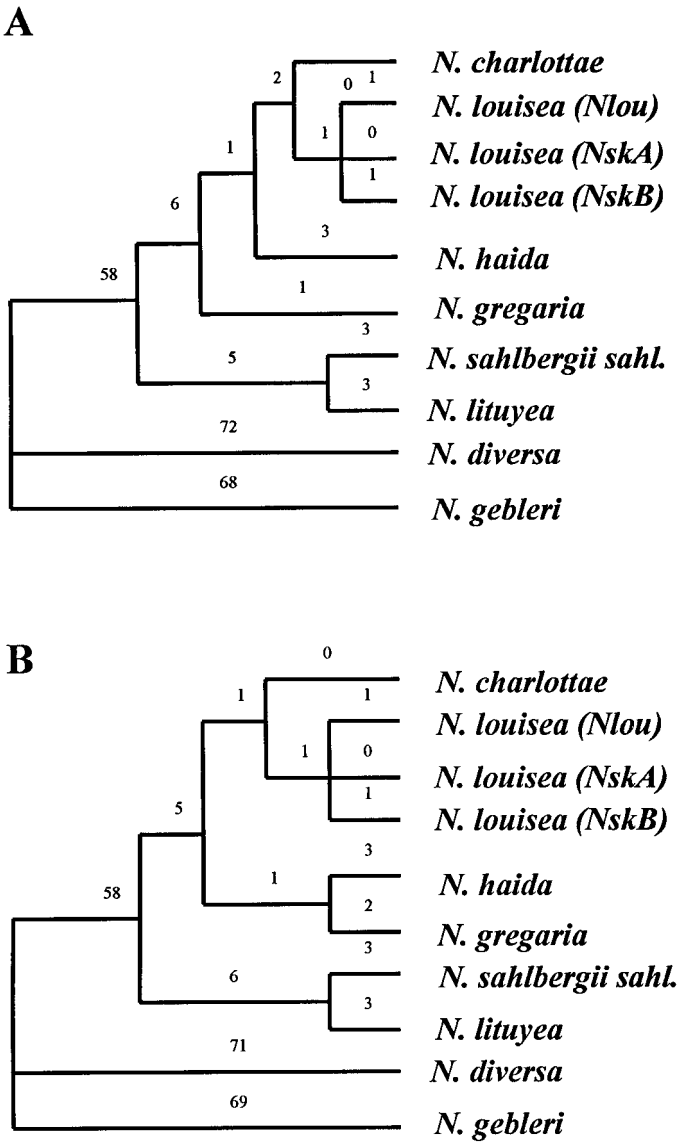

FIG. 3. The two most parsimonious trees resulting from phylogenetic analyses of *Nebria* species using mitochondrial DNA sequences. Numbers above the branches indicate the number of synapomorphies supporting each clade.

Cladistic analyses using the branch-and-bound algorithm of the total dataset produced two most parsimonious trees, each of 225 steps (Fig. 3). Both trees produced a trichotomy at the basal node, consisting of *N. gebleri, N. diversa,* and a clade containing all other species (the *N. gregaria* species

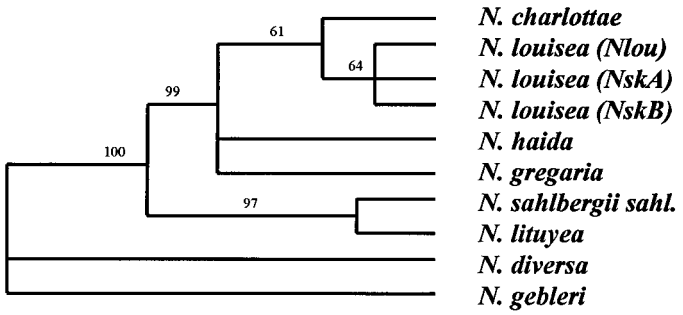

FIG. 4. Bootstrap tree of the total *Nebria* mitochondrial DNA sequence dataset using 1000 replicates. Numbers above the branches indicate the number of synapomorphies supporting each clade.

group, sensu lato). Within the *gregaria* group, both trees aligned the species *N. lituyae* as sister to *N. sahlbergii,* thus removing it from the *gregaria* group. The two trees differed from each other only in the placement of *N. gregaria,* either as a sister taxon to all of the Queen Charlotte Islands *Nebria* or as the sister taxon to *N. haida.* Very little difference existed between the members of the *gregaria* group (exclusive of *N. lituyae*). In both trees, no more than two character changes separated any of the *N. louiseae* haplotypes from *N. charlottae,* whereas the greatest distance between any two of the *gregaria* group taxa (excluding *N. lituyae*) involved seven mutations. Both trees had a consistency index of 0.978 and a retention index of 0.921.

Homoplastic base changes were concentrated between *N. gebleri* and the *N. gregaria* species group in tree A and between *N. gebleri* + *N. diversa* and the *gregaria* species group in tree B. The degree of sequence divergence between members within the *gregaria* group, exclusive of *N. lituyae,* ranged from 0.05% to 0.38%, whereas sequence divergence between the two represented lineages of the *gregaria* species group reached a maximum of 0.98%, between either of *N. sahlbergii* or *N. lituyae* and *N. haida.*

A bootstrap analysis of the data, using 1000 replicates and the branch-and-bound algorithm, supported the clade formed by *N. sahlbergii sahlbergii* and *N. lituyae,* the clade formed by the remaining four members of the *gregaria* group, and the sister relationship between the two groups of taxa (Fig. 4). Within the latter clade, the group formed by *N. charlottae* and *N. louiseae* was maintained as a distinct clade from *N. haida* and *N. gregaria* in 61% of the replicates.

Random Amplified Polymorphic DNA Analyses

In total, 77 bands were produced using RAPDs (Table 2). Across the 47 samples, only one band was uniformly present, although 12 other bands each occurred in 45 or more of the samples. Eleven bands were unique, each occurring in only a single individual. The populations with the greatest degree of fixation for bands were the eastern *N. louiseae* and *N. charlottae* populations, each with fixation rates exceeding 70% and average heterozygosity of less then 0.090. The Kaisun Village *N. louiseae* showed a greater degree of heterozygosity, with only 60.0% of the bands present in all individuals and an average heterozygosity of 0.132. *Nebria haida,* when considered as a single population, contained the greatest heterozygosity, with only 30% of the bands fixed across all 12 individuals and an average heterozygosity of 0.210, comparable to the outgroup species *N. sahlbergii sahlbergii.*

The neighbor-joining tree calculated using the Jaccard coefficient of genetic distance groups the five specimens of *N. sahlbergii sahlbergii* together to form an outgroup cluster with a genetic distance of 0.166 (Fig. 5) from the cluster containing the members of the *gregaria* group. Members of the Kaisun Village population of *N. louiseae, N. charlottae,* and the eastern *N. louiseae* population each formed distinct clusters, whereas *N. haida* formed a single cluster with *N. lituyae* as sister group. Moresby Island and Graham Island populations of *N. haida* could not be differentiated. The four populations of the *gregaria* group from the Queen Charlotte Islands were linked by a simple hierarchy of descent with *N. haida* and *N. lituyae* forming the basal group from which the Kaisun Village population was derived. The Kaisun Village population, in turn, represented the ancestor from which the sister clusters of *N. charlottae* and the eastern *N. louiseae* were derived.

The statistical significance of the neighbor-joining tree was assessed using a permutation tail probability test. Twenty random trees were generated from the RAPD data of the *Nebria* populations collected on the Queen Charlotte Islands,

TABLE 2. Summary of random amplified polymorphic DNA statistics for *Nebria* specimens from Queen Charlotte Island and mainland localities. *N,* population size; FB, number of fixed bands; VB, number of variable bands; SB, number of single bands; AB, number of absent bands; %UFB, percentage of unfixed bands; AH, average heterozygosity.

Population	*N*	FB	VB(2+)	SB	AB	%UFB	AH
Eastern *N. louiseae* (A + B)	10	33	10	3	31	28.3%	0.082
Reef Island (A)	5	36	6	4	31	21.7%	0.089
Skedans Isle (B)	5	38	4	2	33	13.6%	0.053
Tow Hill	10	35	8	1	33	20.5%	0.089
Kaisun Village	9	29	13	6	28	40.0%	0.132
N. haida (total)	12	17	31	8	21	69.6%	0.210
N. haida (Graham Island)	8	19	26	8	24	64.2%	0.217
N. haida (Moresby Island)	4	29	11	6	31	37.0%	0.210
N. lituyae	1	37	n/a	n/a	40	n/a	n/a
N. sahlbergi sahlbergi	5	20	9	8	40	46.0%	0.192
All *Nebria*	47	1	65	11	n/a	98.7%	0.200

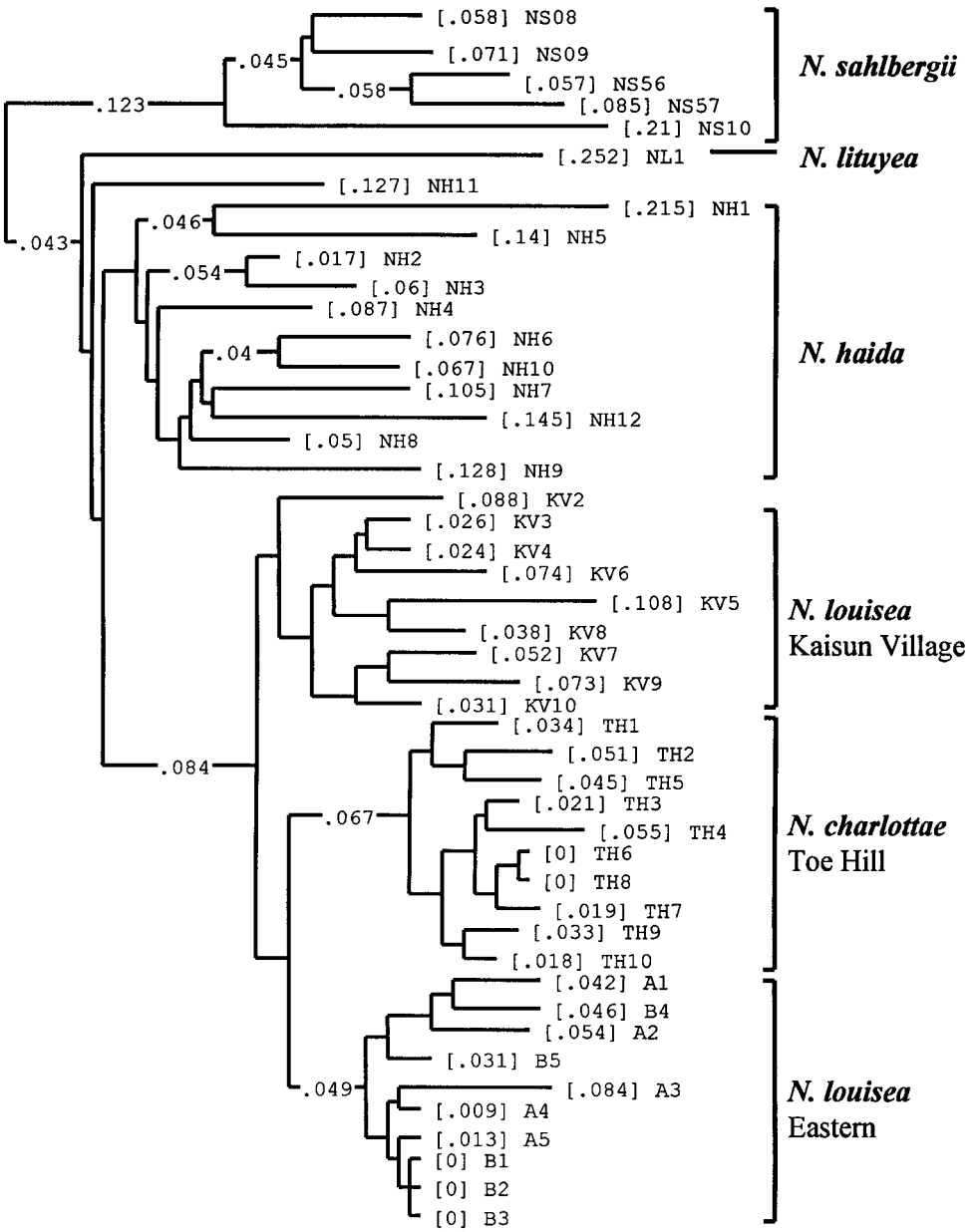

FIG. 5. Neighbor-joining tree from RAPD analysis data using the Jacard coefficients of relatedness.

having a mean length of 6.60 and a PTP value of 0.06 (equivalent to a standard deviation), compared to a length of 4.05 for the original tree. The mean length of the neighbor-joining tree exceeding the mean length of randomly generated trees by 41.94 standard deviations.

An analysis of molecular variance (AMOVA) using the genetic distances calculated from the three coefficients divided the variation fairly evenly, with between-population variance accounting for 53.4% and within-population variance comprising 46.6% of the total. The PHI-statistic represents the correlation of random haplotypes within populations relative to that of pairs of haplotypes drawn randomly from the whole species and is essentially equal to the variance among populations. PHI equals 0.534 for the Jaccard distance matrix, with a 99% level of significance.

Nebria charlottae and the eastern *N. louiseae* populations displayed high degrees of genetic uniformity, with mean values of genetic distance to a common ancestor of 0.028 for eastern *N. louiseae* and 0.027 for *N. charlottae.* The Kaisun Village population displayed a mean genetic distance value of 0.057, whereas mean values of 0.101 and 0.096 were obtained for *N. haida* and *N. sahlbergii sahlbergii*, respectively. *Nebria lituyae* was represented by a single long branch with a length of 0.252.

Neighbor-joining trees calculated using the Nei and Li or Sokal and Sneath coefficients and their associated statistics (data not shown) did not differ significantly from those of the Jaccard neighbor-joining tree, with the one exception of the Sokal and Sneath neighbor-joining tree placing the specimen of *N. lituyae* within the species *N. haida.*

Discussion

The importance of geological complexity in the promotion of speciation has long been recognized in surveys of biological diversity (Cracraft 1985). The presence of mountainous or otherwise subdivided terrain coupled with the movement of climatic zones produced by glacial cycles provides a mechanism for the fragmentation of populations that can lead to allopatric speciation (Haffer 1969, 1977; Prance 1982). Pleistocene species radiations have been documented in biodiversity studies of African birds and plants (Fjeldsa and Lovett 1997), the biogeographical analysis of montane butterfly subspecies (Hammond 1990), and molecular studies of birds (Roy 1997) and butterflies (Britten and Brussard 1992; Brower 1996).

In contrast, the reigning paradigm of evolution and speciation within the predaceous ground beetles is that of long-term stasis. The age of most extant species is believed to extend through the Pleistocene into the late Tertiary, and very few examples of either speciation or significant phyletic evolution have been documented (Coope 1970; Matthews 1980; Elias 1994; Ashworth 1996). Although early hypotheses of beetle evolution postulated that the climatic upheavals of the Middle to Late Pleistocene had the effect of promoting speciation within the Carabidae, particularly within the forest- and tundra-dwelling genera of the northern Holarctic (e.g., Ball 1966), numerous studies of fossil coleopterans from northern temperate and arctic locations in Europe and North America were able to conclusively show that almost all of the beetle fauna of the Early Pleistocene and Late Pliocene that had been examined could be matched to extant species (e.g., Coope 1961, 1962; Matthews 1974). Rather than promoting speciation and phyletic evolution, the Pleistocene glacial period appeared to have resulted in massive range shifts for much of the ground beetle fauna of the Northern Hemisphere, with species tracking their optimal latitudes in response to climate change. The only phyletic evolution acknowledged for most ground beetle species was the adaptation to changes in day-length as populations migrated into new latitudes (Ashworth 1996).

However, the fossil surveys upon which theories of ground beetle evolution were based were not without bias. Most surveys of fossils had been performed in relatively stable lowland areas with little geological complexity and the majority of the insects studied typically occupied wide ranges during both glacial maxima and interglacials (Ashworth 1996). Despite the emphasis of paleoentomology on this narrow subset of possible habitats, the conclusions of these studies had an effect on phylogenetic hypotheses for ground beetles that occupied very different conditions, with assumptions on the rate of morphological change for other lineages modified to fit the evidence of slow, gradual evolution.

The strongest influence of the static ground beetle theory is seen in those phylogenetic hypotheses dealing with the speciation of alpine and subalpine beetles. The importance of montane habitat and climatic change to the process of speciation in Carabid beetles has been recognized in the two most comprehensive zoogeographical analyses of montane Nearctic ground beetle genera, *Harpalus* (Noonan 1990, 1992) and *Nebria* (Kavanaugh 1978, 1979b, 1988). However, most examples of speciation within these lineages have been taken to be old events predating the Pleistocene, based on the degree of morphological and ecological change between sister taxa. The few Pleistocene speciation events (in terms of both morphology and habitat) that have been proposed for the *Nebria* are thought to have occurred between only the most similar species (Kavanaugh 1979a), whereas no speciation events are thought to have occurred within the last 1.6 million years in the genus *Harpalus* (Noonan 1990).

The original phylogeny of the *gregaria* group proposed that the five species of the group survived the Fraser glaciation in a series of refugia centred on the Queen Charlotte Islands, the Alexander Archipelago, and the Aleutian Islands (Kavanaugh 1989). Evidence for the refugial survival of these species comes from their restricted distributions, loss of wings, adaptation to a low-temperature environment, and a lack of potential source populations in either the Beringia refugium or south of the ice sheets. The molecular sequence data presented in this study, however, suggest that four of the five members of the *gregaria* group are incipient species that have radiated from a single common ancestor, and one species (*N. lituyae*) does not belong within the *gregaria* group. However, the position of *N. lituyae* is inferred from a single specimen.

The low level of sequence divergence among the members of this species group indicates that the radiation of these species and populations has occurred relatively recently, however, dating the actual speciation events is complicated by a lack of a reliable molecular clock. Estimates of the rate of sequence divergence between species for arthropod mitochondrial DNA varies from 1.2% to 2.3% per million years (Brower 1994; Su et al. 1996a,b; Knowlton and Weigt 1998) based on a variety of techniques not limited to sequence data and using taxa that have diverged between 3 million and 20 million years ago. Among the few studies that have examined more recently derived insects (within the last 1 million to 3 million years), rates of sequence divergence between taxa have been higher, approaching 5% per million years (Vogler and DeSalle 1993; Juan et al. 1996). Calibrations of the molecular clock for vertebrates has demonstrated that recently derived taxa contain predominantly noncoding mutations (e.g., the third ''wobble'' base of each codon; Li et al. 1987), and that among these unconserved regions of the DNA the rate of sequence divergence can approach 10% per million years (Irwin et al. 1991). Given the wide range of possible estimates of the molecular clock, speciation dates for the *gregaria* infragroup have been estimated using the division between the infragroup and *N. sahlbergii* as a calibration.

If this divergence is assumed to have occurred during the glacial period preceding the Wisconsin, approximately 150,000 years ago (Liu 1992), this would give the *gregaria* species group a rate of sequence divergence of 5.7% per million years and indicate that the component lineages of this species group branched from each other between 20,000 and 50,000 years ago. Extending the age of separation between *N. sahlbergii* and the *gregaria* infragroup to the beginning of the Late Pleistocene, approximately 400,000 years ago, would still place the radiation of the *gregaria* infragroup within the Wisconsin glacial period. To allow the radiation of the *gregaria* infragroup to have occurred before the Wis-

consin, the division between the *gregaria* infragroup and *N. sahlbergii* would need to have occurred approximately 900,000 years ago and would require a molecular clock of 1% per million years, which is far slower then any other insect taxa and not in keeping with the low level of sequence divergence.

RAPD analyses provide a measure of the variation between populations at the genomic level (Williams et al. 1990). There is a nested hierarchy of populations within the species of the Queen Charlotte Islands with both Moresby and Graham Island populations of *N. haida* clustering as a single genetically diverse unit, sister to a paraphyletic assemblage consisting of *N. charlottae* and distinct western and eastern populations of *N. louiseae.* The pattern of genetic diversity in the Queen Charlotte Island species indicates that the genetic variation is not strongly subdivided among the populations, and that much of the existing variation is associated with the species *N. haida,* the expected outcome of a recent radiation of the species group. The single specimen of *N. lituyae* displayed a DNA banding pattern that was similar to several individuals within the *N. haida* populations.

Scenarios that postulate long periods of isolation for each of the populations (e.g., a preglacial differentiation of the four populations) probably would result in much greater partitioning of variation among populations and some form of genetic break between Moresby and Graham Island populations of *N. haida.* In contrast, recent colonization of the Queen Charlotte Islands would reduce both mitochondrial and nuclear diversity through the process of the leading-edge effect during northward migration and concentrate uch of the remaining diversity in the beach populations (Hewitt 1993; Soltis et al. 1997). Given that most other *Nebria* species have responded in the past to warming climates by migrating into the mountains, following the altitude of optimal temperature (Kavanaugh 1979b), the most probable scenario to explain the molecular data would be that the ancestral *gregaria* species ascended into the alpine regions of the Queen Charlotte Islands following deglaciation, leaving behind small scattered populations in the lowland regions that would become the modern species *N. louiseae* and *N. charlottae.*

Where did the *Nebria* of the Queen Charlotte Islands come from? New evidence supports the existence of exposed terrain in the Hecate Strait during the height of the Fraser glaciation. Geologically, both the offshore banks of the Queen Charlotte Islands and coastal regions in the vicinity of Lituya Bay are believed to have had ice-free areas during the course of the Wisconsin glacial period (Barrie et al. 1993; Josenhans et al. 1995; Mann and Hamilton 1995), the product of changing ocean levels and crustal uplift in response to the weight of the mainland ice sheets. Remnants of vegetation discovered in an exposed cliff face on eastern Graham Island indicate that the deglaciation of the Queen Charlotte Islands began at least 1000 years earlier than the adjacent mainland and point toward a nearby reservoir of herbaceous and arboreal plants that would have supplied colonists for the Graham Island site (Warner et al. 1982). Molecular evidence for distinct west coast lineages of black bear (Byun et al. 1997), pine marten and weasel (Byun et al. 1999), as well as genetically disjunct populations of west coast plants (Soltis et al. 1997) are consistent with proposals of a biologically complex refugium located in the now submerged region between the Queen Charlotte Islands and Vancouver Island. The presence of *N. haida* on the mainland uplands would suggest that the common ancestor of the Queen Charlotte Island *Nebria* was also present in the putative refugia located on the inner coast.

This observation of rapid radiation within the *gregaria* infragroup has implications for the understanding of the rate and process of speciation and diversification among predacious ground beetles. The conservative nature of carabid systematics has reflected an understanding of ground beetle evolution based on insects that had extensive ecologically stable habitats with few physical boundaries and little in the way of opportunities for allopatric speciation (Ashworth 1996). Our study, however, has focused on a small group of beetles for which isolation, combined with strong selection pressure for environmental adaptation, appears to have promoted rapid speciation and diversification. The morphology and ecological preferences of the sister taxa to the *gregaria* infragroup would indicate that the common ancestor to the *gregaria* infragroup was macropterous and occupied riparian habitats without a particular altitude preference (Kavanaugh 1978). Based on the rate of molecular evolution proposed here, the species of the *gregaria* infragroup would therefore have acquired brachypterous wing morphology and cold-tolerance within a span of only 20,000 to 50,000 years. While this example of rapid change (in morphology, behavior, and physiology) in response to environmental pressures may represent an extreme brought about by the unique conditions of the west coast refugium, many genera within the predacious ground beetles occupy habitats that would have undergone repeated fragmentation during the cyclical climate changes of the Pleistocene period. The molecular reexamination of these more typical representatives of the family Carabidae will provide invaluable insight into the role played by the Pleistocene climate change in the diversification of one of the largest families of insects.

The results of this study also suggest that a revision of the taxonomy of *N. gregaria* group may be required. MtDNA analyses indicated that *N. haida, N. louiseae,* and *N. charlottae* are closely related. RAPD analyses also revealed close association among these three species. These two genetic targets combined with high morphological similarity among the three species raises questions concerning their taxonomic designations. Based on mtDNA sequence data, *N. lituyae* may be more closely related to *N. sahlbergii* than to other species of the *gregaria* group and, if so, should be excluded from the *gregaria* group. However, this suggestion is based on data derived from a single specimen of *N. lituyae,* and, clearly, more data must be gathered. *Nebria louiseae* and *N. charlottae* may not represent distinct taxonomic units but rather variants of a single beach-dwelling species, *N. charlottae.* Whether even *N. haida* can be considered a distinct species appears debatable, given the close genetic relationship and slight morphological differences between the alpine and beach-dwelling populations. Because only minor differences in morphology and biology and the physical separation of alpine from cobble beach habitat serves to differentiate *N. haida* as a distinct taxonomic unit, we feel that this species may be only at an incipient stage of speciation, and might more appropriately be accorded subspecific status within the

single species, *N. charlottae*. However, we must obtain additional molecular data for other genes, including nuclear genes, and from additional taxa to test this initial evidence further before formally proposing the taxonomic changes suggested by our results.

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Article

Terrestrial Condition Assessment for National Forests of the USDA Forest Service in the Continental US

David Cleland [1], Keith Reynolds [2,*], Robert Vaughan [3], Barbara Schrader [4], Harbin Li [5] and Larry Laing [5]

1 USDA Forest Service, National Forest System, Rhinelander, WI 54501, USA; dcleland@fs.fed.us
2 USDA Forest Service, Pacific Northwest Research Station, Corvallis, OR 97331, USA
3 Redcastle Resources, Inc., Salt Lake City, UT 84103, USA; robertvaughan@fs.fed.us
4 USDA Forest Service, Alaska Region, Juneau, AK 99801, USA; bschrader@fs.fed.us
5 USDA Forest Service, National Forest System, Washington, DC 20250, USA; hli@fs.fed.us (H.L.); larrylaing@fs.fed.us (L.L.)
* Correspondence: kreynolds@fs.fed.us; Tel.: +1-541-750-7434

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Abstract: The terrestrial condition assessment (TCA) evaluates effects of uncharacteristic stressors and disturbance agents on land-type associations (LTAs) to identify restoration opportunities on national forest system (NFS) lands in the United States. A team of agency scientists and managers, representing a broad array of natural resource disciplines, developed a logic structure for the TCA to identify appropriate data sources to support analyses. Primary national data sources included observed insect- and pathogen-induced mortality, key critical loads for soil and the atmosphere, long term seasonal departures in temperature and precipitation, road densities, uncharacteristic wildfires, historical fire regime departure, wildfire potential, insect and pathogen risk, and vegetation departure from natural range of variability. The TCA was implemented with the ecosystem management decision support (EMDS) system, a spatial decision support system for landscape analysis and planning. EMDS uses logic models to interpret data, synthesizes information over successive layers of logic topics, and draws inferences about the ecological integrity of LTAs as an initial step to identifying high priority LTAs for landscape restoration on NFS lands. Results from the analysis showed that about 74 percent of NFS lands had moderate or better overall ecological integrity. Major impacts to ecological integrity included risk of mortality due to insects and disease, extent of current mortality, extent of areas with high and very high wildfire hazard potential, uncharacteristically severe wildfire, and elevated temperatures. In the discussion, we consider implications for agency performance reporting on restoration activities, and subsequent possible steps, including strategic and tactical planning for restoration. The objective of the paper is to describe the TCA framework with results from a national scale application on NFS lands.

Keywords: ecological integrity; stressors; disturbance agents; spatial decision support; restoration; assessment

1. Introduction

National forests and grasslands, under the management of the U.S. Department of Agriculture Forest Service (USFS), have been experiencing unprecedented impacts due to uncharacteristic stressors and disturbance agents over the past few decades. The U.S. burns twice as many acres as three decades ago [1], fire seasons on average have been extended by 78 days in the western United States [2], and the largest insect and disease infestation on record globally is occurring in the western United States and Canada [3]. Multiple stressors are responsible for these problems, in particular warming

temperatures, over-stocking and altered fuel complexes in fire dependent ecosystems due to fire suppression, and invasive species.

The USFS has conducted restoration related activities for decades, however the need for reestablishing and retaining resilience of national forest system (NFS) lands to achieve sustainable management has never been greater. The imminent risk of insects and disease, uncharacteristically high rates of mortality that have already occurred, and the extensive areas with high or very high wildfire hazard potential are major concerns of the USFS. Deleterious effects of elevated temperature and reduced precipitation, particularly in the west, uncharacteristically severe or frequent wildfire, fragmentation of habitat due to roads, and the effects of air pollution or invasive species are also adversely impacting NFS lands.

As a consequence, the USFS has made restoration a major priority within the agency. Policy [4], collaborative landscape restoration projects, and on-the-ground activities have emerged in response to restoration needs, with 1.9 million ha treated for restoration needs in 2014 alone. The team conducting the terrestrial condition assessment (TCA) was commissioned to develop a comprehensive assessment of resource conditions and stressors that may warrant restoration consideration to assist in identifying terrestrial restoration opportunities and improve the agency's transparency and accountability for terrestrial restoration investments. The TCA was chartered by the sustainable land management board of directors, composed of Washington office leadership from NFS, research, and state and private forestry branches of the USFS.

The TCA was designed to complement the watershed condition framework (WCF), a national effort to evaluate the status of watersheds across all NFS lands [5]. The TCA and WCF share goals of assessing resource conditions, but the focus and approach differ. The WCF focuses on conditions and stressors affecting water quality and quantity, and aquatic organisms and their habitat, uses watersheds as analytical and reporting units, and is based primarily on expert opinion in a paneling process that scores indicators of the ecological integrity of watersheds. The TCA addresses terrestrial outcomes, uses landscape-scale analytical and reporting units, is data-driven with existing national data sets, and provides an assessment of ecological integrity based on data interpretation and analyses.

The TCA assesses conditions and processes affecting the ecological integrity of landscape ecosystems on NFS lands. The concept of ecological integrity has evolved over the years [6–9]. It is commonly accepted that an ecosystem has integrity when its dominant ecological characteristics (composition, structure, function) occur within their natural ranges of variation, and can withstand and recover from perturbations caused by natural environmental processes or human activities [9–11]. Thus, the key elements of ecological integrity should include intactness (in terms of natural ranges of variation of all key indicators), biodiversity and species viability, ecosystem structure, ecological processes, and stressors.

In North America, ecological integrity has been mapped across national parks in Canada by the Canadian park service [9]. The Canadian approach includes ecological, species diversity, and human development measures, organized into biodiversity, ecosystem processes, and stressor categories. The National Park Service and NatureServe have developed a preliminary ecological integrity assessment framework intended to introduce concepts and methods to managers and to highlight their potential use [12]. The system recommends use of NatureServe's ecological systems as a coarse filter of biodiversity, but also employs measures of vulnerable species assemblages and their habitats, and species-level measures of the vulnerability of individual plant and animal species. Threats and stressors including human development, resource extraction, roads, pollution, and climate change are included in the assessment. The TCA estimates the ecological integrity of landscape ecosystems by comparing current conditions and processes to reference conditions and processes, but also includes indicators of uncharacteristic biological and environmental stressors, including air pollution and road density.

The TCA is a mid-scale evaluation of conditions and stressors occurring across NFS lands, utilizing the landtype association tier of the national hierarchical framework of Ecological Units [13] as

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averaging 8000 hectares in size. An ultimate goal is to understand the resilience of landscape ecosystems to stressors, as well as the extent and magnitude of various stressors themselves. However, methods and data for quantifying resilience are lacking [14]. Given our present inability to measure ecological resilience, we use estimates of ecological integrity as a proxy and as a means of addressing escalating degradative ecological changes. The primary goals of the TCA are to assist land managers in identifying restoration needs at a national scale, and provide the tools necessary for regional and local applications including science delivery, data access, and guidance on analytical procedures. Secondary goals are to support restoration prioritization activities, and provide a baseline from which restoration and maintenance activities can be tracked and effects on ecological integrity documented. The objective of the paper is to describe the TCA framework with results from a national scale application on NFS lands.

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2. Materials and Methods

A team of scientists and resource specialists from NFS, Research and Development, and State and Private Forestry branches of the USFS conceived and designed the TCA. The team addressed questions related to current restoration investments, resource conditions warranting investments, appropriate scale and units of analysis, selection of measureable indicators, data availability and acquisition, and computational methods.

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2.1. Study Area and Analysis Units

The TCA included all administrative units (National Forests and National Grasslands) of the NFS of the USFS in the continental United States (Figure 1). The total land area of the NFS is distributed across 112 administrative units (labeled "Forests" in the figure), and covers approximately 86 million ha.

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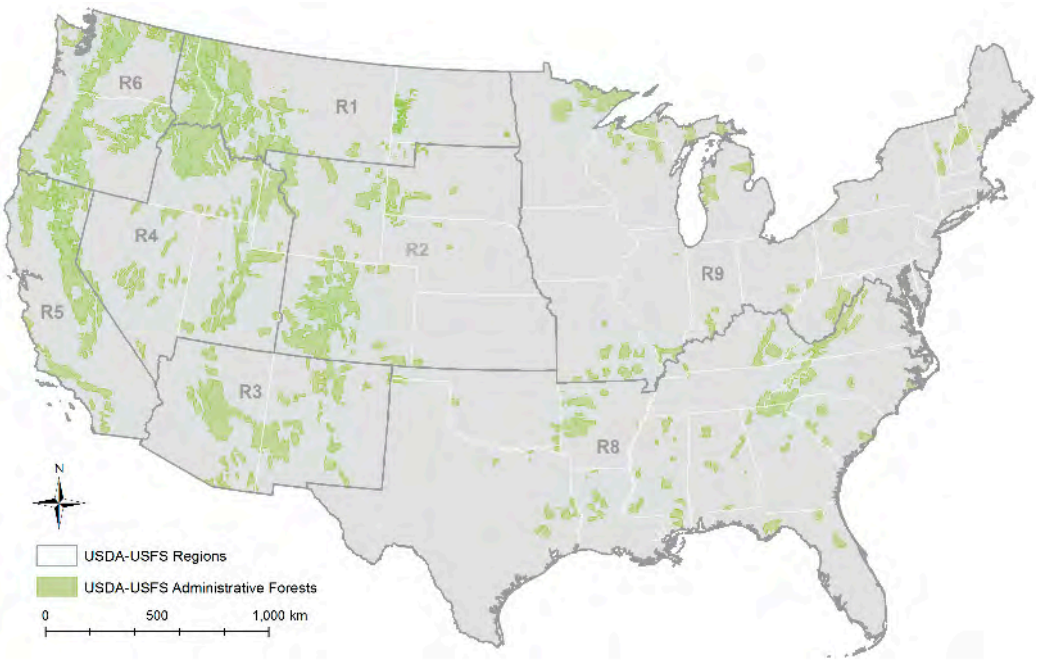

Figure 1. Study area of the Terrestrial Condition Assessment in the continental United States.

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Landscape units used in TCA were a combination of landtype associations or generalizations of LANDFIRE's biophysical settings. LTAs are the landscape-level units in the national hierarchical framework of ecological units [13,14], and are based on patterns of surficial or bedrock geology, lithology, topography, soils and vegetation. LTAs were used in the analysis when these were available for an NFS region. Otherwise, the generalized biophysical settings were used as close approximations to LTAs.

The objective of using LTAs was to reduce the variability primarily in dominant vegetation as well as natural disturbance regimes. The study area included a total of 10,213 such landscape units. Hereafter, we refer to the analysis units as LTAs.

2.2. Data Sources

Data supporting the TCA were drawn from a variety of sources (Table 1). Estimates for each LTA were derived by zonal statistics using the appropriate input raster indicator dataset and LTA. This methodology was chosen specifically to reduce the overall variance within each estimate but also account for different resolution input indicator datasets. Detailed metadata on the metrics supporting each indicator are included in the supplementary materials .

Table 1. Data sources for metrics used in indicators of the Terrestrial Condition Assessment (TCA).

Indicator	Metric [1]	Data Source
Tree mortality	Mortality due to Insects and Pathogens Data unit: Binary of presence or absence (Ordinal) TCA metric: percent area	National forest pest conditions database produced by USFS forest health technology enterprise team (FHTET) https://foresthealth.fs.usda.gov/portal Raster data at the resolution of 240 m
Terrestrial invasive species	Local Data; Occurrence	NRIS TESP Data are incomplete and not available yet, so this data source is not included in the analysis, although the model includes a placeholder for it.
Road density	Highway road density Paved road density Light duty road Density Unimproved road density Data unit: mi/sq. mi. (Numeric) TCA metric: mi/sq. mi. (Numeric)	USFS FSTOPO transportation dataset developed by USFS geospatial technology and applications center (GTAC) http://data.fs.usda.gov/geodata/vector/index.php Vector line features
Climate exposure	**Temperature**: Mean seasonal temperatures Spring, summer, fall, winter Data unit: Degrees Fahrenheit (Numeric) TCA metric: Degrees F difference **Precipitation**: Total seasonal precipitations Spring, summer, fall, winter % precipitations Spring, summer, fall, winter Data unit: Inches (Numeric) TCA metric: Inches difference	PRISM Climatological Data produced by PRISM Climate Group of Oregon State University with **P**arameter elevation **R**egression on **I**ndependent **S**lopes **M**odel http://prism.oregonstate.edu/ Raster data mostly at the resolution of 4 km
Air pollution	Terrestrial acidification (Exceedance, CAL); Data unit: Ranks of good, moderate, or poor (Ordinal) TCA metric: Ranks of good, moderate, or poor Terrestrial eutrophication (N) Data unit: kg/ha/yr (Numeric) TCA metric: kg/ha/yr (Numeric)	Terrestrial acidification database produced by USFS southern global change program, using the simple mass balance equation (SMBE) http://fsweb.wo.fs.fed.us/wfw/airquality/criticalloads.html Raster data are at the resolution of 1 km^2 Terrestrial eutrophication database generated by EPA's community multiscale air quality (CMAQ) modeling system https://www.epa.gov/air-research/community-multi-scale-air-quality-cmaq-modeling-system-air-quality-management Raster data are at the resolution of 120 m (resampled from 12 km)

Table 1. *Cont.*

Indicator	Metric [1]	Data Source
Catastrophic disturbance	Uncharacteristic fire severity Uncharacteristic fire frequency Data unit: Binary of uncharacteristic and other (Ordinal) TCA metric: Percent area	Database of uncharacteristically severe wildfires derived from (1) Monitoring trends in burn severity (MTBS) data by USGS and USFS and (2) LANDFIRE data of percent low severity fire and percent mixed-severity Fire http://mtbs.gov https://landfire.gov/fireregime.php Raster data at the resolution of 30 m Database of uncharacteristically frequent fire derived from a combination of (1) MTBS as the current condition and (2) Mean fire return interval (MFRI) of LANDFIRE as the reference condition.
Wildfire potential	Uncharacteristic fuel buildup Data unit: Binary of high risk or other TCA metric: Percent area	Wildfire hazard potential (WHP) database produced by USFS Fire Modeling Institute http://www.firelab.org/project/wildfire-hazard-potential Raster data at the resolution of 270 m
Insect and pathogen risk	Potential uncharacteristic mortality Data unit: Binary of presence or absence (Ordinal) TCA metric: Percent area	National insect and disease risk map (NIDRM) produced by USFS forest health protection (FHP) http://www.fs.fed.us/foresthealth/technology/nidrm.shtmlRaster data at the resolution of 270 m
Vegetation departure	Vegetation departure index Data unit: 0–100% (Numeric) TCA metric: Mean	Vegetation departure index (VDEP) produced by LANDFIRE http://www.landfire.gov Raster data at the resolution of 30 m
Ecological process departure	Missed Fire Cycle TCA metric: Mean	Mean fire return interval (MFRI) produced by LANDFIRE http://www.landfire.gov

[1] Metrics represent measurable quantities. Indicators may have one or more metrics.

2.3. Overview of EMDS Framework

EMDS is a spatially enabled decision-support framework for integrated landscape evaluation and planning [15]. We describe EMDS as a decision support framework because the data sources, scales of analysis, and models employed in EMDS applications are all user defined. As a result, the system has been applied to a wide variety of decision support problems since 1997 [16,17]. At version 5.5, the system provides decision support for landscape-level analyses through logic and decision engines integrated with the ArcGIS® 10.x geographic information system (GIS, Environmental Systems Research Institute, Redlands, CA, USA), as well as QGIS [18] and MapWindow [19]. The NetWeaver logic engine (Rules of Thumb, Inc., North East, PA) evaluates landscape data against a formal logic specification (e.g., a knowledge base in the strict sense) designed in NetWeaver Developer® [20], to derive logic-based interpretations of ecosystem conditions such as ecosystem integrity. EMDS 5.5 implements the decision engines of three decision support applications. Criterium DecisionPlus® (CDP, InfoHarvest, Seattle, WA, USA) implements the analytical hierarchy process (AHP) [21,22], and can be used for both strategic and tactical planning. GeNIe® (BayesFusion, LLC, Pittsburg, PA, USA) implements Bayesian networks and influence diagrams, while VisiRule® (Logic Programming Associates, Ltd, London, UK) implements Prolog-based decision trees. Both GeNIe and VisiRule are perhaps most applicable to tactical planning in the EMDS context, although strategic applications are also possible. The terms strategic and tactical planning have various interpretations, depending on context. In the particular context of spatial decision support, strategic planning in EMDS is concerned with which management units are the highest priority for management activities, whereas tactical planning is concerned with selecting the highest priority management actions in specific landscape features.

In the present study, our analysis is limited to logic-based processing to assess ecological integrity of LTAs. However, we have introduced the decision engines in this section because their functionality is pertinent to subsequent steps in the larger decision support process that is considered in the later Discussion section.

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2.4. NetWeaver Logic Design for TCA

NetWeaver models are implemented as a network of networks (Figure 2). For example, evaluation of the logic network, terrestrial condition, is directly dependent on the evaluation of the two networks, disturbance agents and vegetation condition, at the next lower level of the network outline. Conversely, we can describe the relation as disturbance agents and vegetation condition are logically antecedent to terrestrial condition. Similarly, biotic agents and abiotic agents are the direct logical antecedents of disturbance agents.

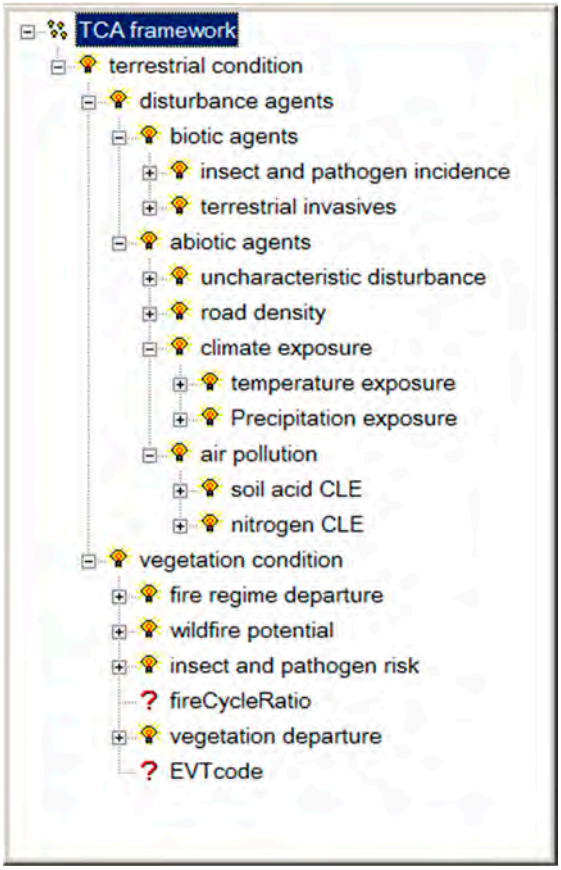

Figure 2. Outline of NetWeaver logic structure for the Terrestrial Condition Assessment. The top level, TCA framework, is simply a container for logic networks. Each item in the outline is a logic network, except fireCycleRatio and EVTcode, which are data inputs used to control the flow of logic processing under the network, vegetation condition. Networks listed under vegetation condition represent the latter's logical antecedents (e.g., the evaluation of vegetation conditions depends on fire regime departure, etc.). See the accompanying text for additional explanation of network concepts in NetWeaver.

Apart from its name, each logic network that makes up a logic model has four other important attributes. Each network:

1. Evaluates a proposition about the topic represented by the network, which is contained in a comment field;
2. Has a logical specification composed of its immediate logical antecedents and one or more logic operators that determine how the antecedents contribute to the proposition;
3. Has a measure of the strength of evidence for the proposition provided by its antecedents;
4. Has one or more documentation attributes that describe important aspects of the network (e.g., most networks have an explanation attribute at a minimum).

In the interest of space, the TCA logic outline (Figure 2) is not shown fully expanded. In particular, the model includes additional levels of logic under the networks such as uncharacteristic disturbance, road density, and climate exposure. Comprehensive HTML documentation on the NetWeaver logic for TCA can be found in the supplementary materials section below.

As suggested by the phrase, "a network of networks", NetWeaver models are structurally recursive, such that higher level networks are composed of antecedent networks. This structural recursion terminates with elementary networks that evaluate data. Within each elementary network, data are interpreted with fuzzy logic by comparing observed data values to fuzzy membership functions that translate the observed value into a measure of the strength of evidence for the parent elementary network [20]. Metrics for strength of evidence are propagated upward through the logic structure. Within each network, the strength of evidence metrics contributed by the antecedent networks in the logic specification of their dependent network are logically synthesized by fuzzy logic operators such as AND, OR, and Union [20].

Threshold values presented in Table 2 were established based on a review of the literature, consultation with subject matter experts, and examination of data distributions to ensure model sensitivity. For example, thresholds for indicator 1, extent of insect and disease caused mortality in the past five years, are set at 5% for full evidence of high integrity and 25% for no evidence of high integrity. The 2013–2027 national insect and disease forest risk assessment [23] uses a natural annual background rate of 0.89% for evaluating mortality at a national scale. Based on this literature, the TCA evaluated mortality occurring within the past five years, and considered rates of 5% or less to be natural. The 25% or greater value represents systems that are experiencing mortality at five or more times the natural background rate, affecting ten percent of LTA's nationally. Values between 5% and 25% are ramped and evaluated continuously between these thresholds, such that 7% mortality is very close to full evidence and 23% very close to no evidence.

For readers who may not be familiar with fuzzy logic theory, here, we provide a brief comparison to probability theory, and related issues around confidence limits. Whereas probability theory is concerned with uncertainty in the sense of uncertainty about the likelihood of events, fuzzy logic is concerned with a fundamentally different concept of uncertainty, referred to as linguistic (or lexical) uncertainty [24–26], which originates in the imprecision of human thought and communication. For example, the concept of a warm day is linguistically imprecise (hence fuzzy). Fuzzy logic, or more generally fuzzy math and fuzzy set theory, is actually a precise mathematics for handling imprecise information [24–26]. Fuzzy membership functions, introduced above, are a way of expressing linguistic uncertainty in terms of set theory (e.g., to what degree is an observation a member of some fuzzy set?). The metric for strength of evidence, discussed in the context of NetWeaver, is simply another way of describing degree of set membership, and thus uncertainty. Finally, most readers will have had some training in probabilistic uncertainty, so there is an expectation that our results should include confidence limits on the fuzzy metrics presented in maps of the Results section. There are two compelling reasons why confidence limits are not treated in NetWeaver outputs in EMDS. First, conceptually, doing so would conflate two fundamentally different measures of uncertainty (e.g., probabilistic and linguistic). Second, as a practical matter, the computation of confidence intervals would require solving a convolution integral [27] for the roughly 40 inputs on each of about 10,000 observations, and this assumes that one has error estimates for each of the 400,000 observations, and that the computational algorithm could account for the nonlinearities intrinsic to the logic at runtime. This last reason is a compelling counterargument.

Table 2. Indicators, metrics, and thresholds values used in the Terrestrial Condition Assessment.

Indicator Number	TCA Indicator	Associated Metrics	Threshold for No Evidence [1]	Threshold for Full Evidence [2]	Unit
1	Tree mortality	Mortality due to insects and pathogens	25.0	5.0	% Land-type associations (LTA) area
3	Road density	Highway	0.3	0.1	mile/square mile
		Paved roads	0.3	0.1	mile/square mile
		Light duty roads	1.5	0.5	mile/square mile
		Unimproved roads	2.5	1.0	mile/square mile
4	Climate exposure	Spring temperature	2.0	0.0	°F changed
		Summer temperature	2.0	0.0	°F changed
		Fall temperature	2.0	0.0	°F changed
		Winter temperature	2.0	0.0	°F changed
		Spring precipitation	−1.0	0.0	inch changed
		Summer precipitation	−1.0	0.0	inch changed
		Fall precipitation	−1.0	0.0	inch changed
		Winter precipitation	−1.0	0.0	inch changed
		Spring precipitation (%)	−10.0	0.0	% changed
		Summer precipitation (%)	−10.0	0.0	% changed
		Fall precipitation (%)	−10.0	0.0	% changed
		Winter precipitation (%)	−10.0	0.0	% changed
5	Air pollution	Terrestrial acidification	poor	good	rank
		Terrestrial eutrophication (N)	10.0	1.6	kg/ha/yr
6	Catastrophic disturbance	Uncharacteristic fire severity	5.0	0.0	% LTA area
		Uncharacteristic fire frequency	1.0	1.5	dimensionless
7	Wildfire potential	Uncharacteristic fuel buildup	66.0	20.0	% LTA area
8	Insect and pathogen risk	Potential uncharacteristic mortality	50.0	10.0	% LTA area
9	Vegetation departure	Vegetation departure index	67.0	43.0	% area departed
10	Ecological process departure	Missed fire cycle	35.0	200.0	year departed

[1] Value at which the fuzzy membership function interpreting the associated metric provides no evidence for a suitable condition; [2] Value at which the fuzzy membership function interpreting the associated metric provides full evidence for a suitable condition.

2.5. TCA Analysis in EMDS System

The TCA analysis to assess the ecological integrity of LTAs on NFS lands was implemented in the ArcMap (ESRI) version of the EMDS system. All metrics needed for the assessment (Table 2) were initially obtained or developed as separate GIS layers. Zonal statistics procedures, available in ArcMap, were used to attribute each metric to the LTA polygons. The TCA analysis for the full set of 10,213 LTAs in the continental U.S. was performed with the NetWeaver model (Section 2.4). Within EMDS, the basic products of a NetWeaver analysis are maps displaying the strength of evidence associated with the proposition for each logic topic (Figure 2). In the case of indicators evaluated in terms of multiple metrics (Table 2), map products assessing strength of evidence also were produced for each individual metric. The final ArcMap document (e.g., mxd file), including all map products of the TCA, is available in the supplementary materials section. After completing the full national TCA assessment for the continental U.S., results were parsed to each NFS Region and National Forest for subsequent use by these units. Within the overall scheme of the TCA process, it was envisioned that Regions and Forests could modify data inputs and NetWeaver logic as needed to improve the relevance of analytical products at the latter smaller spatial extents. Customizing the TCA for other spatial extents is addressed further in the Discussion.

3. Results

At a national scale, 55% of national forests and grasslands are in very good or good condition, whereas 26% are in poor or very poor condition (Table 3, Figure 3). Overall TCA condition ratings are based on simultaneous consideration of nine indicators and the twenty six metrics used to characterize indicators (Table 2). The importance of indicators varies geographically, and interpretations

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Table 3. Frequency and areal distributions of the LTAs among the five overall ecological integrity ratings at the national scale.

Terrestrial Condition [1]	Frequency	Hectares	Percent of National Forest System Lands
Very Good	1618	15,862,119	18.41
Good	3962	31,896,498	37.02
Moderate	1736	15,942,480	18.50
Poor	1226	9,785,574	11.36
Very Poor	1491	12,669,501	14.71

[1] Classes used for classification of terrestrial condition in this table, and subsequent tables and figures, represent equal intervals of the National average strength of evidence, with very good being ≥0.60, good condition being <0.60 and ≥0.20, etc.

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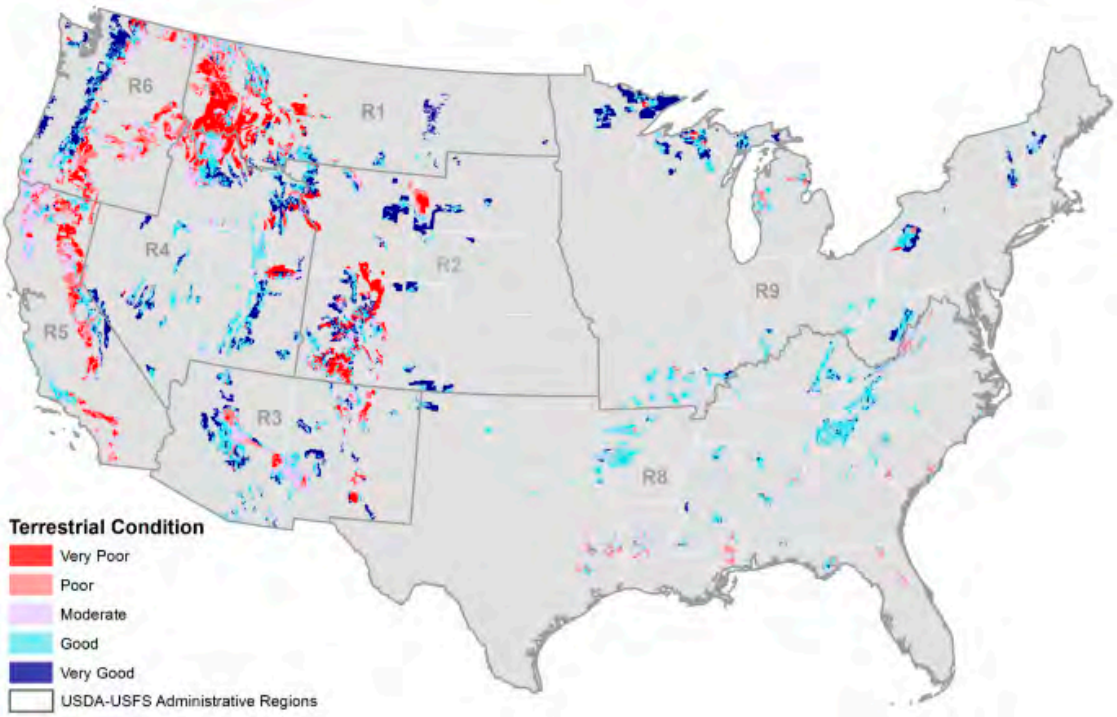

Figure 3. Overall ratings of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service administrative lands.

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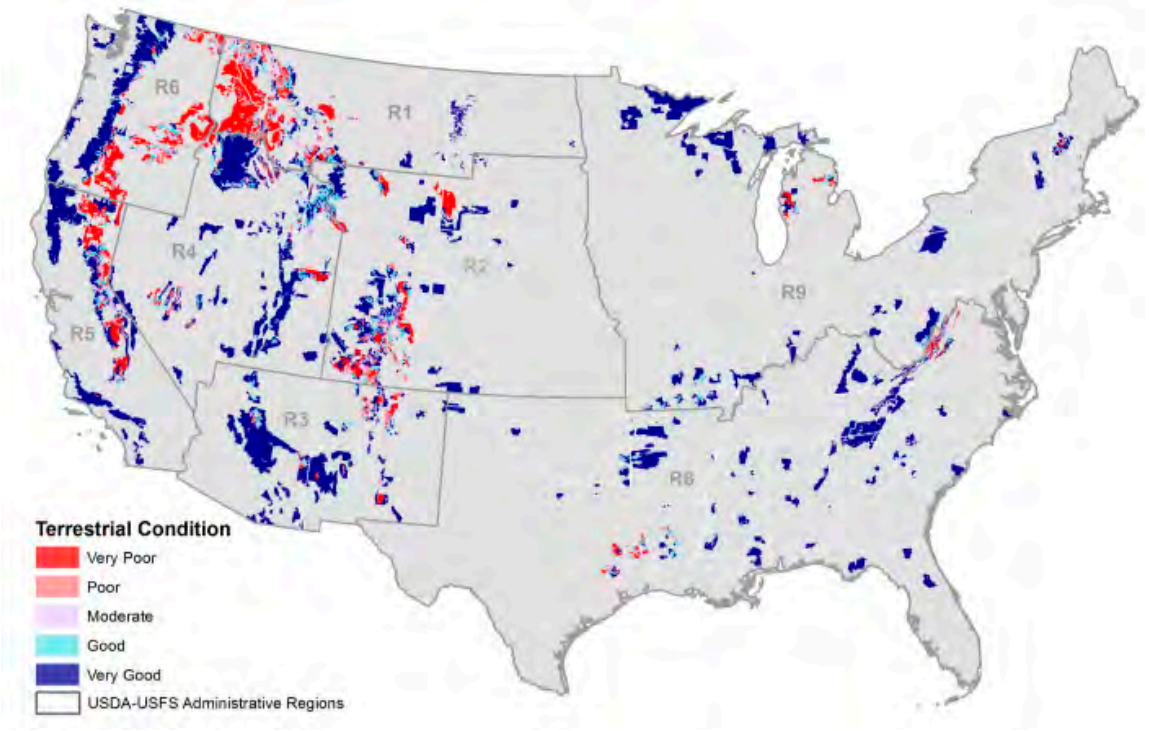

Figure 4. Ratings of the insect and disease risk metric of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands.

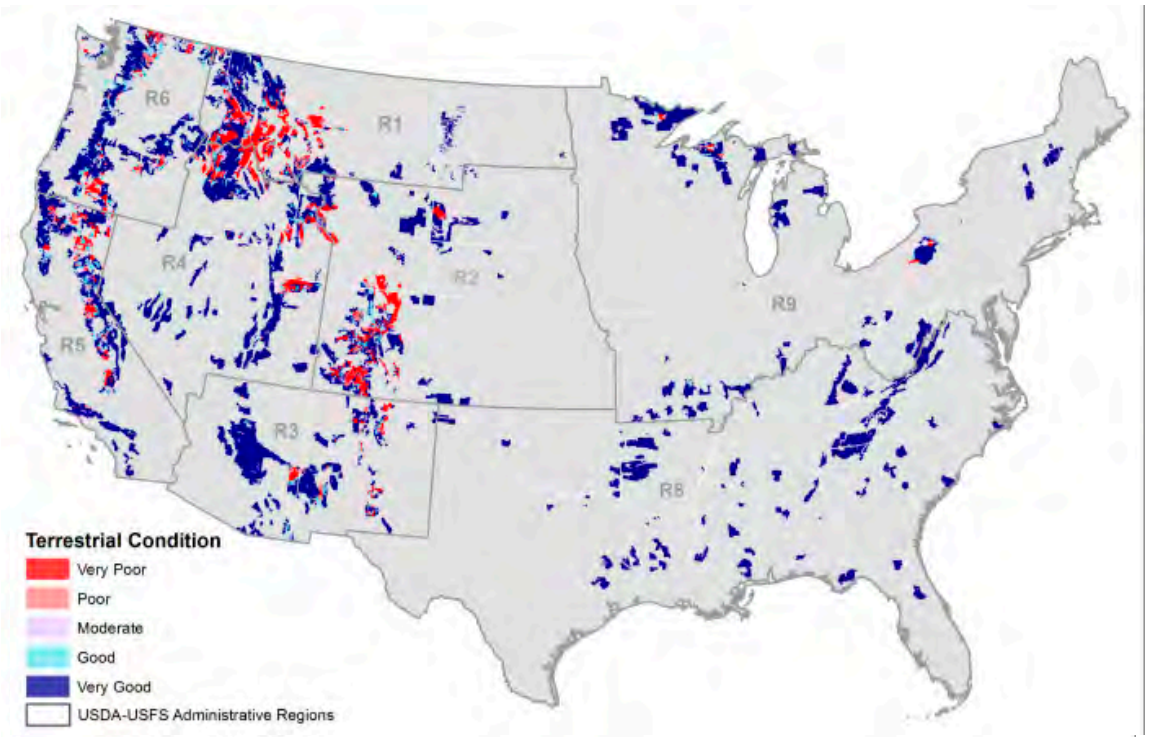

Figure 5. Ratings of the unhealthy tree mortality metric of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands. Tree mortality is based on 2010–2015 surveys of current mortality and excessive defoliation due to insect and disease outbreaks.

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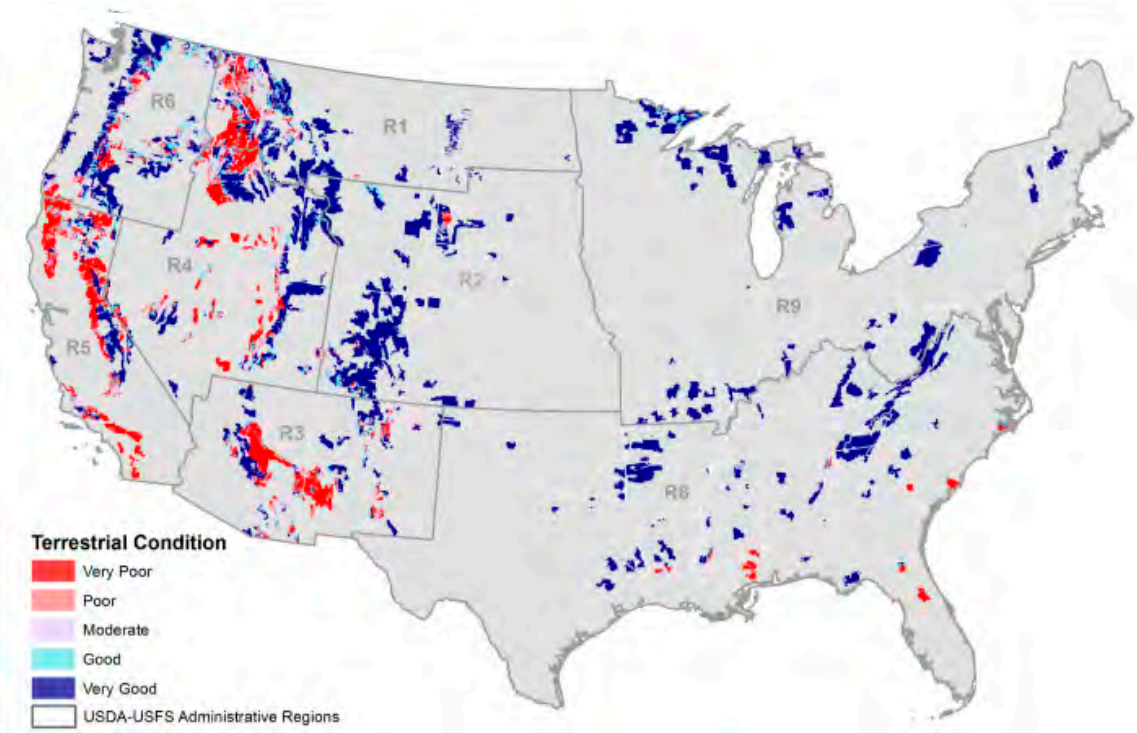

Figure 6. Ratings of the high and very high wildfire potential hazard metric of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands.

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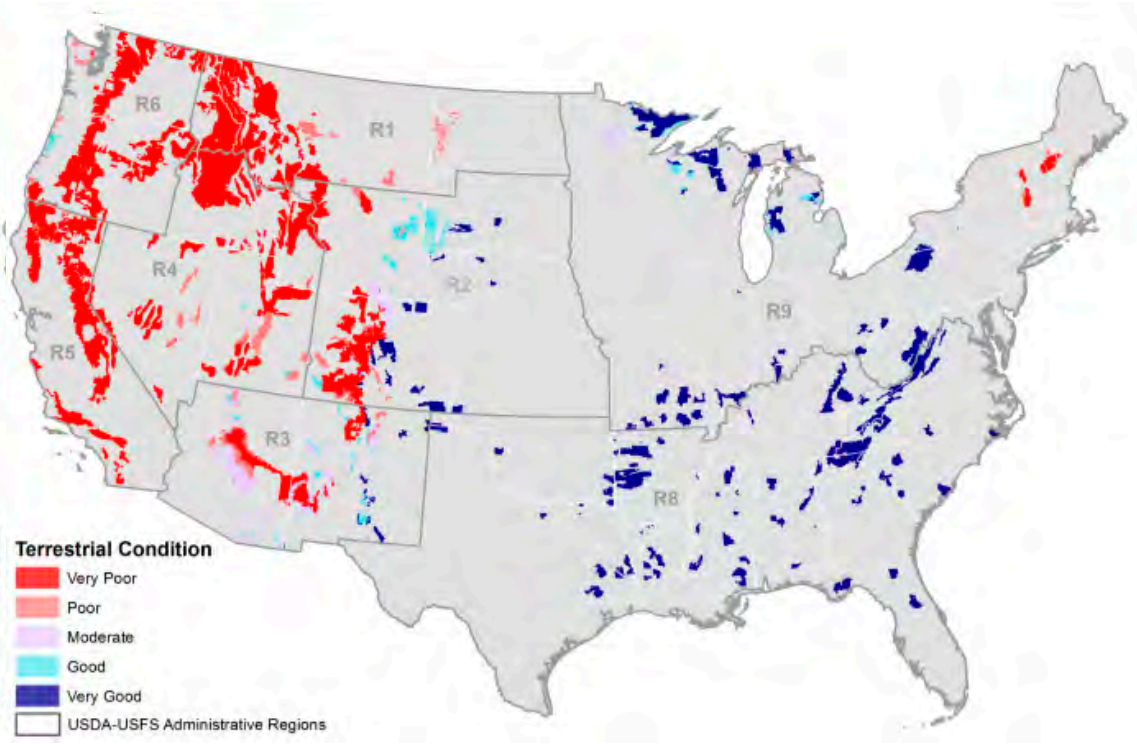

Figure 7. Ratings of the winter temperature shift metric of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands.

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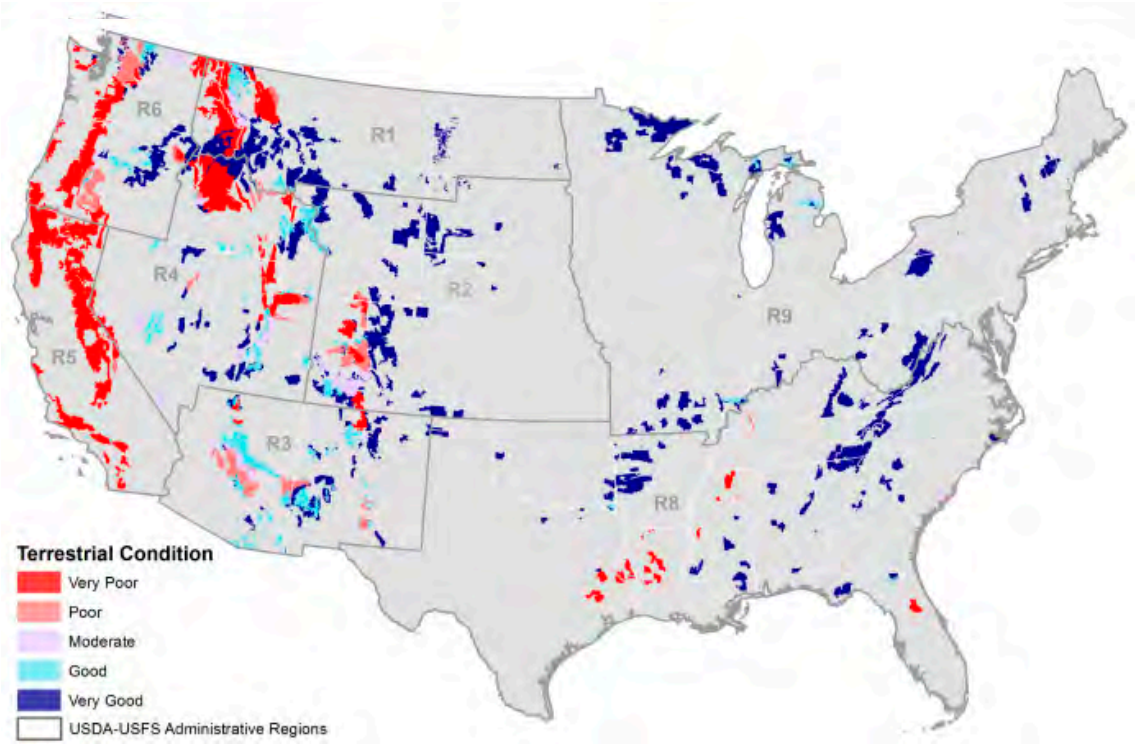

Figure 8. Ratings of the winter precipitation shift metric of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands.
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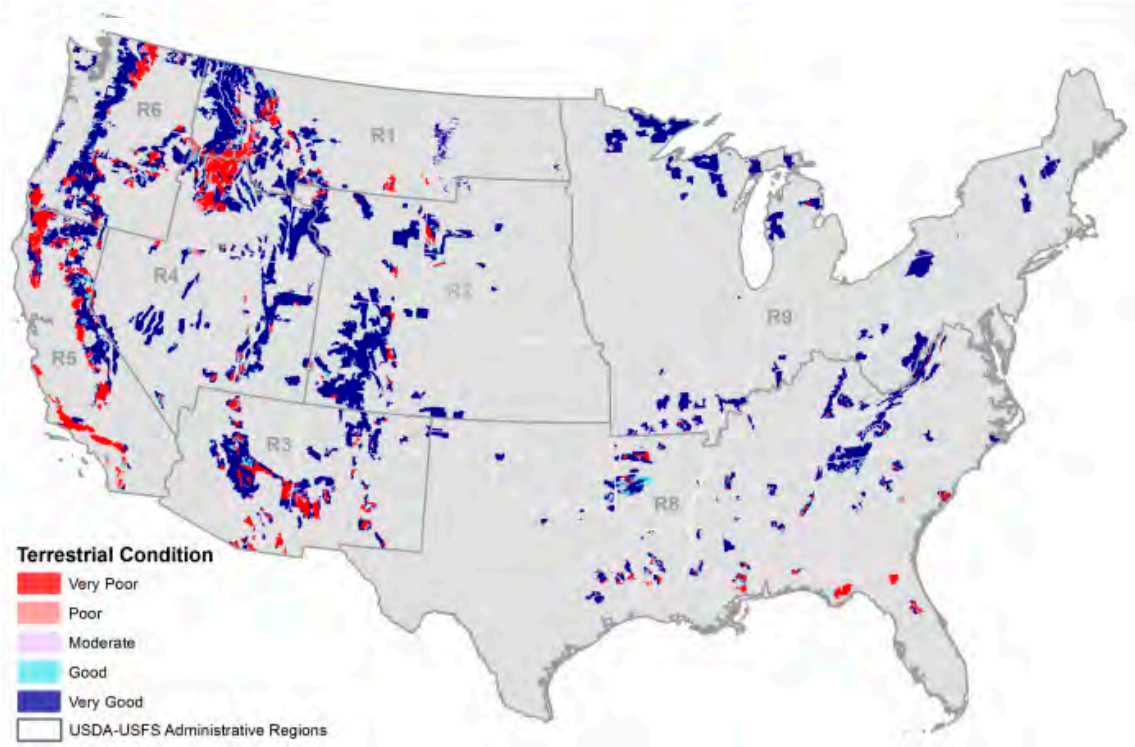

Figure 9. Ratings of the uncharacteristic wildfire indicator of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands.
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Figure 9. Ratings of the uncharacteristic wildfire indicator of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands.

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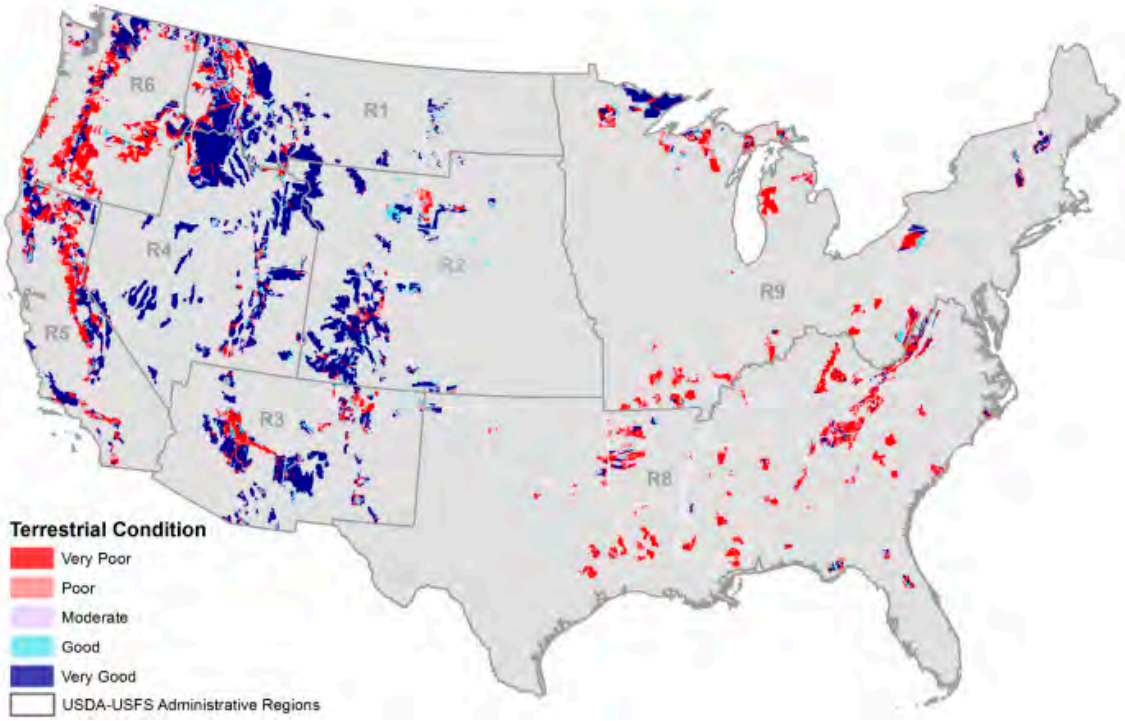

Figure 10. Ratings of the total road density metric of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands.

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In general, conditions in the eastern United States are better than much of the west, largely because of low fire hazard potential and very limited extent of current mortality. The primary stressors affecting National Forests in the east include high road densities (Figure 10), air pollution (Figure 11), and vegetation departure from reference conditions (Figure 12). National Forests in the southeast are also being impacted by uncharacteristically severe or frequent fires and reduced spring and fall precipitation.

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Poor and very poor conditions are concentrated in the western United States. Of the 64.8 million hectares occurring within western national forests' proclamation boundaries, 20 percent or 13.1 million hectares are at imminent risk of uncharacteristic mortality due to insects and disease, 9.4 percent or 6.3 million hectares have experienced mortality in the past five years, and 33 percent or 21.2 million hectares have high or very high wildfire hazard potential. Stressors of uncharacteristically severe wildfire and climate exposure (elevated temperatures, particularly winter temperatures, and reduced precipitation) are severe in the west but almost nonexistent in the east.

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At a regional scale (see Figure 1 for boundaries of the USFS regions), Regions 1, 2, 5, and 6 have very large percentages of very poor and poor conditions (Table 4). Region 1 has extensive areas with high insect and disease risk (37% of the Region), high and very high wildfire potential (34% of the Region), and high mortality occurring within the past five years (16% of the Region). Region 2 has extensive areas with high insect and disease risk (20% of the Region) and high recent mortality (13.4% of the Region). Region 5 has extensive areas with high and very high wildlife potential (53% of the Region), high insect and disease risk (18% of the Region), high recent mortality (8.3% of the Region), and high road densities. Region 6 has high insect and disease risk (23% of the Region). All western Regions are experiencing stress due to elevated temperatures and to a lesser degree reduced precipitation. Of greatest concern are increases in winter temperature (Figure 7) and decreases in winter precipitation (Figure 8).

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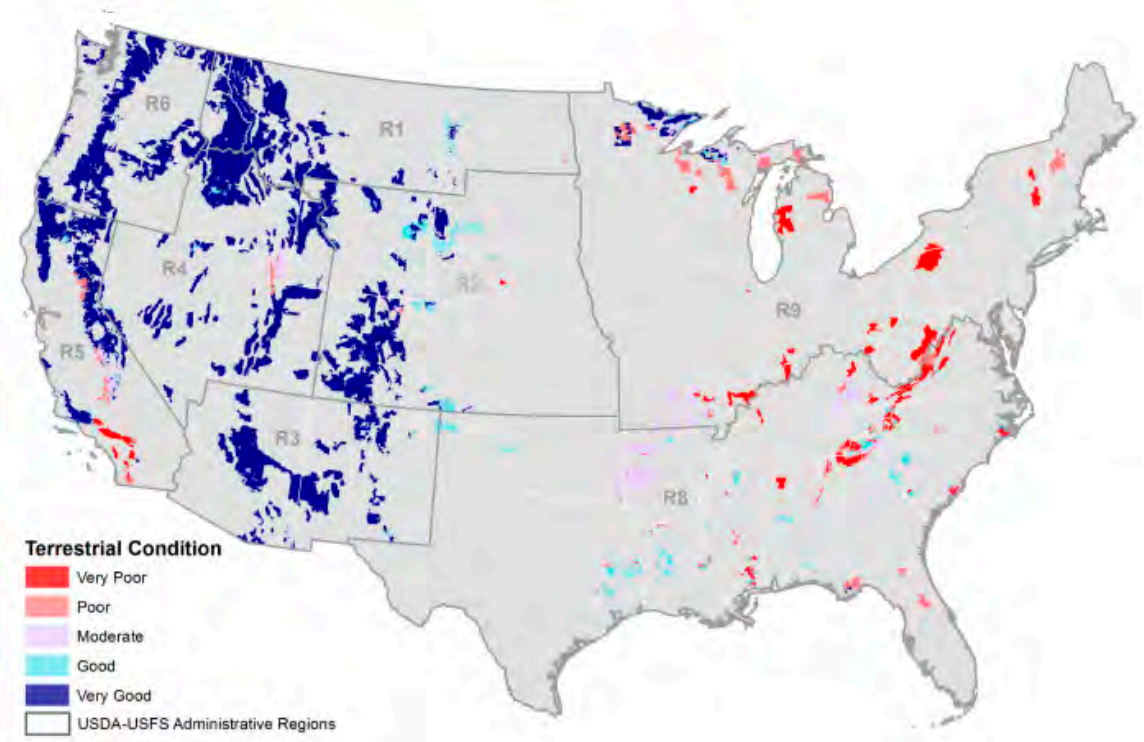

Figure 11. Ratings of the air pollution indicator of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands.

Figure 11. Ratings of the air pollution indicator of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands.

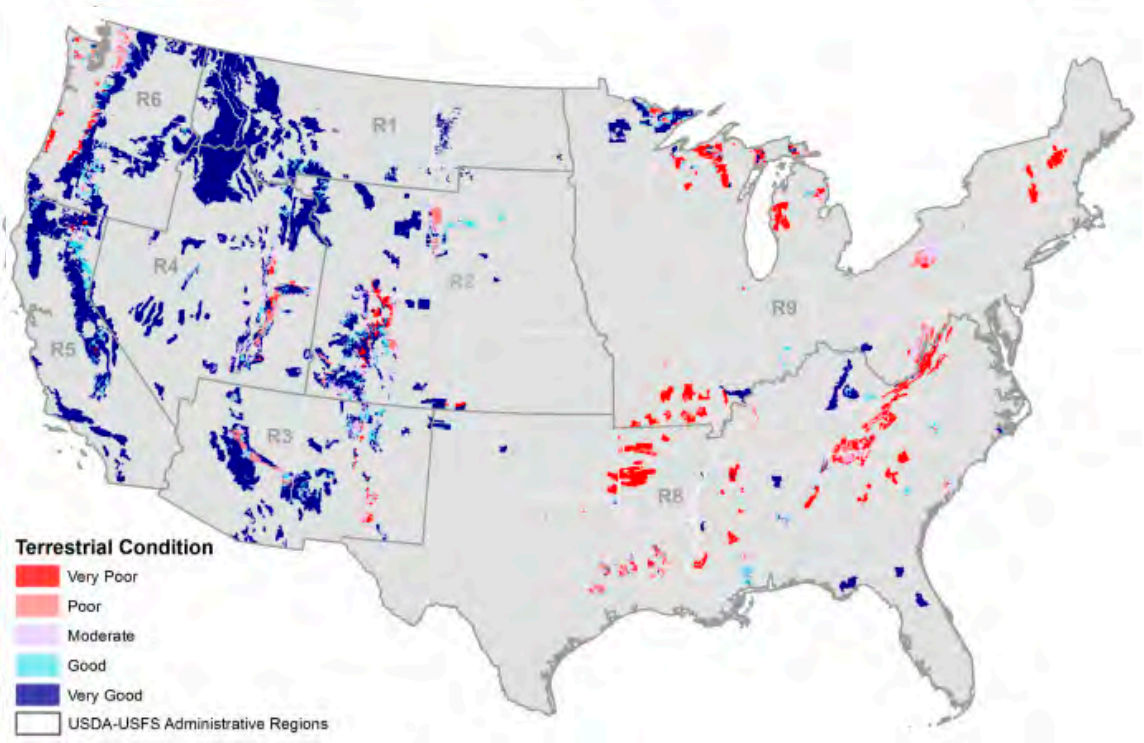

Figure 12. Ratings of the vegetation departure metric of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands.

Figure 12. Ratings of the vegetation departure metric of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands.
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Table 4. Frequency and areal distributions of the LTAs among the five overall ecological integrity ratings at the regional scale.

Region	Terrestrial Condition	Hectares	Percent Regional	Region	Terrestrial Condition	Hectares	Percent Regional
1	Very Good	1,065,169	9.46	5	Very Good	448,620	4.74
1	Good	2,547,906	22.62	5	Good	1,894,951	20.01
1	Moderate	1,711,470	15.20	5	Moderate	3,263,593	34.46
1	Poor	1,497,393	13.30	5	Poor	2,162,259	22.83
1	Very Poor	4,439,859	39.42	5	Very Poor	1,702,486	17.97
2	Very Good	3,540,636	31.51	6	Very Good	2,295,182	20.36
2	Good	3,213,151	28.60	6	Good	2,573,660	22.83
2	Moderate	1,157,275	10.30	6	Moderate	2,514,413	22.30
2	Poor	874,659	7.79	6	Poor	2,337,587	20.73
2	Very Poor	2,449,247	21.80	6	Very Poor	1,554,411	13.79
3	Very Good	1,838,627	19.76	8	Very Good	550,467	5.36
3	Good	3,777,133	40.59	8	Good	6,794,328	66.13
3	Moderate	2,236,714	24.03	8	Moderate	1,860,729	18.11
3	Poor	834,357	8.97	8	Poor	1,060,174	10.32
3	Very Poor	619,630	6.66	8	Very Poor	8379	0.08
4	Very Good	2,805,318	20.08	9	Very Good	3,318,101	35.44
4	Good	6,056,478	43.35	9	Good	5,038,891	53.82
4	Moderate	2,741,347	19.62	9	Moderate	456,939	4.88
4	Poor	634,118	4.54	9	Poor	385,028	4.11
4	Very Poor	1,732,351	12.40	9	Very Poor	163,139	1.74

4. Discussion

Our results show a marked contrast between the eastern and western U.S., especially with respect to climate differences. The western United States has been subject to uncharacteristically severe wildfire in past decades largely due to a century of fire suppression, past logging, and climate exposure [28]. Fire suppression has resulted in increased tree densities and associated moisture demand, and increased fuel loads relative to historical or pre-European settlement forest conditions [29]. Increased winter temperatures reduce snowpack and water storage [30], and also reduce cold-induced mortality of damaging insects and diseases [31]. Increased temperatures during the growing season reduces fuel moisture, aggravating conditions promoting uncharacteristic wildfire [1,2], and increases the extent to which trees are stressed and less able to resist adverse effects of insect and disease.

Recent shifts in temperature affect western national forests far more than those in the east. The TCA used an increase of 1.11 °C (2 °F) as a threshold to identify LTAs undergoing recent severe temperature stress. Based on that threshold, 48% of NFS lands in the west are experiencing severe winter temperature stress in contrast to the less than one percent in the east. Spring, summer, and fall severe temperature stress affected 6.8%, 27.1%, and 24.8% of western national forests, respectively. Conversely, spring, summer, and fall severe temperature stress affected less than one percent of eastern national forests. Interactions leading to poor conditions, including altered landscape patterns, fuel complexes, incidence of insect and disease caused mortality, and climate-induced stress, are therefore manifest in the western United States far more so than the east.

We have presented the TCA framework used to complete a national level assessment of ecological integrity based on uncharacteristic stressors, conditions, and disturbance agents for national forest system lands in the United States. Results, data, and guidance on analytical procedures have been produced for agency applications, including a web map viewer and web-based information delivery system. Applications for performance accounting are being developed at a national scale. Regional applications that include use of the TCA in addition to regional data and assessments are being initiated. Local applications in land management planning are taking place on select national forests involved in the planning revision process. Moving beyond a national product to support regional and local applications of the TCA is one of the next phases of the project.

4.1. Customizing TCA for NFS Regions and National Forests

As mentioned in Section 2.5, the TCA assessment for the continental U.S. was intended as a starting point or template from which USFS Regions and National Forests could customize the assessment to make it more relevant to their local contexts. Customization of the analysis presented here can be done in at least four distinct ways:

1. National data presented in this study could be replaced with local data sources if local data sources were believed to be more accurate or more appropriate for the local context.
2. Thresholds used to define fuzzy membership functions that interpret the TCA metrics (Table 2) could be revised to better reflect local conditions. A good example in this context is the interpretation of road densities with respect to their effects on wildlife habitat fragmentation.
3. The national TCA logic includes several metrics related to uncharacteristic disturbances (indicator 6, Figure 2) including the spatial extent of mine impacts, landslides, blowdown, and flooding. Although logic topics and metrics associated with these impacts were designed into the NetWeaver logic model, they are turned off in the national analysis that we have presented because national data for these effects are not available. However, regions and forests could turn on one or more of these logic topics to include in their local assessments if they were considered important. Within the national TCA template, Region and Forest staff have two options for accounting for these ecosystem impacts: use of continuous measures (e.g., measured spatial extent) or use of ordinal rankings provided by specialists (see the HTML NetWeaver logic documentation included in the supplementary materials at the end of the paper).
4. Finally, the basic logic structure of the national TCA template is easily edited in NetWeaver by Region and Forest staffs to customize the logic for local contexts. For example, some logic topics in the national TCA template may not be considered relevant in some local contexts, in which case they can be turned off. In addition, the combination of logic operations used to synthesize evidence for logical premises of a particular logic topic might be edited by changing logic operators, or reorganizing the logic structure of premises to alter how a set of premises contribute to the strength of evidence for their parent topic.

The ability to customize the national TCA template for local application as described above creates some tension between assessments conducted at the different spatial extents of national, Region, and Forest. On the one hand, the national template was intended to promote, as far as practicable, consistency in how TCA assessments are conducted across spatial extents. On the other hand, an excessive emphasis on consistency across spatial extents has the potential to seriously compromise the utility of assessment products at more local extents. As a result, the NFS may need to consider an explicit governance process that balances the competing interests of national consistency and local relevance.

4.2. Additional Steps in Decision Support for Ecosystem Restoration and Maintenance

The results presented in this article evaluate the ecological integrity of LTAs on NFS lands in the continental US. However, in important respects, the analysis only represents the first step in a complete decision support process for the restoration of ecological integrity. In particular, the assessment characterizes terrestrial condition, which is an important foundation for a planning process, but it does not provide explicit support for implementing strategic and tactical planning decisions needed to meet restoration goals of the agency. As we discussed in Section 2.3, the EMDS framework includes a collection of decision engines that provide additional support for strategic and tactical decisions. In this section, we discuss how the associated decision support systems can be brought to bear to support management decisions for restoration and maintenance of LTAs.

The decision engine of CDP has been used for design of strategic multi-criteria decision models in EMDS since 2002 [15]. Whereas NetWeaver solutions describe the state of the system, strategic decision models assist resource managers with identifying which landscape units that are a high priority for

management actions, by not only considering the state of the system, but by accounting for logistical considerations that are of practical importance to managers. Logistical considerations include such issues as feasibility, efficacy, cost, performance, consequences, social acceptability, etc., of potential management actions.

Whereas strategic decision models address the question of which landscape units are the highest priority for management, tactical decision models address the question of which management actions are the highest priority for any particular landscape unit, considering the biophysical context (or other contextual information) of the landscape unit. In other words, the strategic question concerns where, while the tactical question concerns what. Reynolds et al. [32] recently experimented with a CDP solution for tactical planning, however we believe that tactical decision models based on GeNIe and VisiRule may be more effective in tactical decisions, primarily because these systems can model more complex problems than CDP. For example, GeNIe supports sophisticated probabilistic reasoning based on Bayesian inference [33], and VisiRule, although providing a simple graphic interface, is supported by a powerful Prolog engine that allows very complex reasoning.

Reynolds et al. [32] also illustrated a variety of analytical sequences for decision support involving assessment and strategic and tactical planning, but more generally the architecture of EMDS was extensively re-engineered at version 5.0 to support the concept of workflows, by which any of the EMDS analytical components described above can be invoked in any sequence(s) (or series of sequences) needed to support spatial analysis and planning. EMDS currently supports Microsoft Windows Workflow for creating, running, and monitoring scientific workflows and Workflow NET. Interoperability of components is realized by data sharing, by which upstream analytical products are shared with any downstream analytical steps in an analysis sequence. In order to further extend interoperability in the workflow environment, EMDS now also implements Java script, R, and Python languages as tools for spatial data transformation.

5. Conclusions

The USDA Forest Service has recently completed an assessment of the ecological integrity of landscape ecosystems (LTAs) across its land base using the TCA framework. Results are beginning to be applied in national, regional, and local resource planning and management activities. Prospects for improving national performance accounting, and for developing strategic and tactical decision support systems that include social and economic considerations are being evaluated.

Supplementary Materials: The following are available online at 1. TCA metadata for data inputs to logic model https://www.cloudvault.usda.gov/index.php/s/5YHcvq213tFZQ3D; 2. Complete documentation of the NetWeaver logic in HTML https://www.cloudvault.usda.gov/index.php/s/rsHuooVYIk2xJLE; 3. ArcMap document with maps of all NetWeaver outputs for the TCA project https://www.cloudvault.usda.gov/index.php/s/11u5oCMxGvYAlhq.

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Author Contributions: All authors participated in the design and implementation of the TCA application; David Cleland and Robert Vaughan analyzed the data; Robert Vaughan did the geoprocessing to prepare data for EMDS; Keith Reynolds designed the NetWeaver logic and ran the analysis in EMDS; David Cleland, Barbara Schrader, and Larry Laing provided overall project oversight; David Cleland, Keith Reynolds, and Harbin Li wrote the paper.

Conflicts of Interest: The authors declare no conflict of interest.

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RESEARCH ARTICLE

Implications of introgression for wildlife translocations: the case of North American martens

Jocelyn P. Colella[1] · Robert E. Wilson[2] · Sandra L. Talbot[2] · Joseph A. Cook[1]

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Abstract

The evolutionary consequences of natural introgression provide a rare opportunity to retrospectively evaluate how the introduction of exotics or genetic rescue efforts may impact endemic faunas. Phylogeographic structure among mainland, endemic insular, and introduced North American marten (*Martes americana* and *M. caurina*) populations have been shaped by a complex history of natural, post-glacial population expansion followed by a series of anthropogenic introductions. In some cases, both natural colonization and translocations facilitated secondary contact, offering a series of replicated experiments that demonstrate how introgression, in these cases following isolation (insular and refugial), shapes genetic diversity. We test whether genetic exchange is occurring between North American marten species using mitochondrial genomes and ten nuclear loci. We present evidence of biased nuclear introgression from *M. caurina* into *M. americana* across two natural hybrid zones (insular and mainland) and found no remnant evidence of *M. caurina* on islands that received *M. americana* translocations, suggesting prior absence, potential extirpation, or genetic swamping of *M. caurina* from these islands. Our results highlight the importance of understanding phylogeographic variation prior to identifying source populations for wildlife translocations and caution the use of genetic rescue for North American marten populations. Although previously managed as a single species, these two species show substantial genetic divergence. When the two are placed into contact, they exhibit unidirectional, asymmetric introgression with potentially negative consequences for *M. caurina*, underscoring the value of mindful consideration of introgression in wildlife management.

Keywords Asymmetric introgression · Genetics · Hybridization · Introductions · Islands · Marten

Introduction

Hybridization can influence the evolutionary history of closely related species, by stimulating or impeding divergence (Barton 2005) through diverse outcomes ranging from heterosis (Shull 1948) and reinforcement (Dobzhansky 1937) to homogenization and genetic swamping (Rhymer and Simberloff 1996; Todesco et al. 2016). Because geographic isolation is a major factor limiting gene flow, wildlife translocations can have serious conservation implications for otherwise allopatric populations and species if they're not yet completely reproductively isolated (Rodriguez et al. 2011; Milián-García et al. 2015). For example, in mammals, anthropogenic reintroductions of the red wolf (*Canis rufus*) into its historical range in the eastern United States resulted in extensive introgression with both gray wolves (*C. lupus*) and coyotes (*C. latrans*), jeopardizing the protective status of these species and forever shaping their genetic architecture (Gese et al. 2015; vonHoldt et al. 2016). Captive breeding and reintroduction programs of black-footed ferrets (*Mustela nigripes*) have similarly worked to balance the effects of genetic rescue against the potential for genetic swamping or the loss of native genetic diversity (Wisely et al. 2008). Natural hybridization as a consequence of shifting geographic ranges can also inhibit divergence by promoting genetic assimilation (Rhymer and Simberloff 1996), so an understanding of the outcomes of introgression

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✉ Jocelyn P. Colella
jcolella.jc@gmail.com

[1] Department of Biology and Museum of Southwestern Biology, 1 University of New Mexico, MSC03-2020, Albuquerque, NM 87131, USA

[2] U.S. Geological Survey, Alaska Science Center, 4210 University Drive, Anchorage, AK 99508, USA

events, whether human-manipulated or natural, is critical to effective management of species.

The Pacific marten, *Martes caurina*, is endemic to Admiralty and Kuiu islands in Southeast Alaska's Alexander Archipelago, Graham and Moresby islands in British Columbia's Haida Gwaii Archipelago, and along the Pacific Northwest coast, south to California and eastward into the southern Rockies (Stone and Cook 2002; Fig. 1). In contrast, *M. americana* is widely distributed throughout northern North America, from interior Alaska to the east coast. Hybridization between genetically (Stone et al. 2002; Small et al. 2003; Dawson et al. 2017) and morphologically (Merriam 1890; Dawson and Cook 2012) distinct North American martens (*Martes americana* and *M. caurina*) occurs naturally where these species are parapatric, but is also a hypothesized outcome of intentional translocations of these species. As commercially valuable furbearers, the maintenance of robust marten populations has often been the focus of management actions, including at least 51 translocations in the mid-1900s (Powell et al. 2012). All but one of those manipulations occurred without understanding existing phylogeographic structure or even the recognition of two species. Recent reevaluation of supplemented populations of *M. americana* in Wisconsin, including populations that received individuals originally from populations in Colorado (*M. caurina*), found no evidence of *M. caurina* persistence (Grauer et al. 2017). That finding suggests that competition or biased introgression impact *M. caurina* survivorship when introduced into *M. americana* populations and calls for a more complete understanding of introgression between these species. Further, the smaller, often fragmented distribution of *M. caurina* relative to *M. americana* suggests that *M. caurina* may experience higher levels of inbreeding and island effects (Frankham 1998). As such, genetic rescue may be considered a potential management option to maintain commercially viable marten populations, but the potential outcomes of those actions should be carefully considered.

The deep biogeographic history of North American martens remains unresolved, but there are two competing hypotheses based on the geographic distributions and patterns of divergence: the two species may be the result of a single eastward colonization event across the Bering Land Bridge or multiple exchanges (Kurtén and Anderson 1980). The two species hypothetically diverged in at least two allopatric glacial refugia during the last glacial maximum (Stone et al. 2002), both located south of the ice-sheets, with *M. caurina* isolated in the west and *M. americana* in the east, as proposed for several other species (Swenson and Howard 2005; Colbeck et al. 2008). While *M. americana* expansion followed glacial recession east to west across the

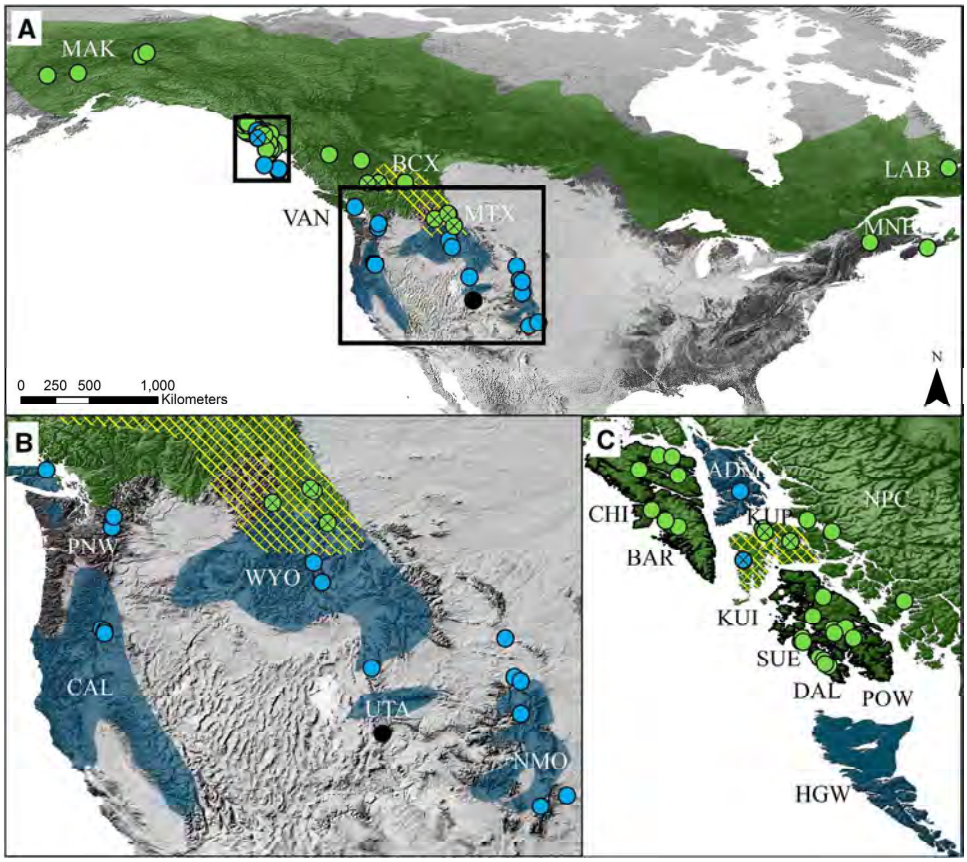

Fig. 1 Map of North American marten distributions (*M. americana* range in green, *M. caurina* in blue) with natural hybrid zones denoted by yellow hatching and islands that received translocations of *M. americana* outlined in heavy black (CHI, BAR, POW). Three letter localities codes correspond to population abbreviations in Table 1. Each dot represents a sampled specimen, colored by mitochondrial species assignment: *M. americana* in green, *M. caurina* in blue, with putative hybrids colored based on mtDNA assignment and identified by a yellow 'X'

northern latitudes (Stone and Cook 2002), two evolutionary scenarios were advanced for the disjunct, more southerly distribution of *M. caurina*: (I) isolation in both a southwestern refugium and a North Pacific Coastal (NPC) refugium (Carrara et al. 2007) or (II) a single southwestern refugium followed by northward colonization along a coastal migration route into the NPC. Although no *Martes* fossil evidence exists along the NPC during the Last Glacial Maximum, the possibility of persistence there remains viable (Heaton et al. 1996; Heaton and Grady 2003). Natural hybrid zones formed in response to post-glacial range expansion and secondary contact has been hypothesized along the Rocky Mountains in western Montana (Wright 1953) and on Kuiu Island in Southeast Alaska (Dawson et al. 2017) based on the presence of mixed mitochondrial haplotypes and intermediate morphological phenotypes. Contrasting the evolutionary consequences of introgression, natural or human-induced, provides an opportunity to evaluate how introductions impact local faunas, especially those with limited distributions such as island endemics.

We test for introgression and a directional bias in gene flow between North American martens to evaluate the role of historical wildlife translocations on genetic diversity and assess the use of genetic rescue as a viable management option in this system. We hypothesize that introgression will be occurring in two proposed contact zones, one insular (Kuiu Island) and one mainland (Northern Rocky Mountains), and potentially on islands that received translocations of *M. americana*. If there is a selective or demographic imbalance between the species, we anticipate there will be asymmetric introgression. Secondly, we examine the consequences of introgression on genetic diversity and species integrity by expanding geographic and genetic coverage from previous studies, through the addition of complete mitochondrial genomes (mitogenomes) and ten nuclear loci. The use of biparentally inherited nuclear loci is required to test for the presence of admixture (i.e., hybridization), which previously was only inferred from the presence of shared mitochondrial haplotypes. Last, we quantify interspecific divergence, refine the geographic limits of hybrid zones, and examine the implications of introgression for conservation and management.

Materials and methods

Extraction, amplification, and sequencing

Liver tissue samples that are broadly representative of both 'pure' species, natural hybrid populations (Kuiu Island and the Northern Rocky Mountains), and translocated islands (Prince of Wales, Baranof, and Chichagof islands) were loaned from the Museum of Southwestern Biology (MSB) and University of Alaska Museum of the North (UAM; Online Appendix 1). Populations were defined based on geography, ecology (e.g., dispersal; Johnson et al. 2009), and genetics (Dawson et al. 2017). We examined sequences from 90 individuals. Specimens were collected from 1972 to 2010. Additional specimen data are available at arctos.database.museum online (see Online Appendix 1 and Colella et al. 2018 for specimen details). DNA was extracted via standard salt extraction protocol (samples with mitogenomes) following Medrano et al. (1990) with modifications described in Sonsthagen et al. (2004) or E.Z.N.A. Tissue DNA kits (samples without mitogenomes sequenced; Omega bio-tek Inc., Norcross, Georgia, USA). Sable (*Martes zibellina*—a Eurasian relative) and other outgroup tissues (*Gulo gulo, Martes pennanti, Neovison vison, Mustela frenata*) were loaned from the MSB and extracted, amplified, and Sanger sequenced in parallel. Mitogenomes were amplified in 5 fragments using parallel-tagged amplicon sequencing (Supplementary Material 1). Ten PCR-amplified nuclear loci (nuDNA; Supplementary Material 1) were pooled with mitogenomes for sequencing. Library preparation followed the Nextera XT DNA Sample Preparation Guide (Illumina 2012, San Diego, California, USA) for sequencing on an Illumina MiSeq Small Genome Analyzer at the U.S. Geological Survey Alaska Science Center (Anchorage, Alaska, USA). We used a 2 × 250 bp Reagent Kit with 10% PhiX. For samples without mitogenomes, mitochondrial (mtDNA) gene cytochrome *c* oxidase-1 (*COX1*) was amplified with same ten nuclear loci (Supplementary Materials; Supplementary Material 1), purified (ExoSapIT; Affymetrix, Santa Clara, California, USA), cycle sequenced (BigDye v3.1; ThermoFisher Scientific, Waltham, Massachusetts, USA), and sequenced on an ABI 3100 (Molecular Biology Facility, University of New Mexico, Albuquerque, USA) to increase the number of specimens examined. Individual samples were recovered by identifying individual barcodes and adaptors and quality score trimming was performed with default settings using MiSeq Reporter software (Illumina, San Diego CA, USA). Paired-end reads were assembled and mapped to their respective GenBank reference sequences (Supplementary Materials) in CLC Genomics v7.5 (https://www.qiagenbioinformatics.com/). For nuclear loci, ambiguities were called when two alleles were present at a 0.3 frequency threshold.

Final analyses were based on 53 marten mitogenomes (16,604 bp, variable number of tandem repeats removed) and extracted mitogenome coding sequence (CDS; 11,429 bp) and 81 additional mitochondrial *COX1* sequences, for samples without sequenced mitogenomes. Complete mitogenomes and mitogenome CDS's were analyzed separately, as the CDS is expected to be evolving under negative selection to maintain functionality, while non-coding regions may exhibit greater diversity as a consequence of neutral

evolution. We analyzed a total of 1,308 nuclear sequences (alleles) representing 10 loci: adiponectin (*AdipoQ*, N = 122 alleles), agouti signaling protein (*ASIP*, N = 152), nicotinic cholinergic receptor alpha 1 (*CHRNA*, N = 154), feline sarcoma proto-oncogene (*FES*, N = 156), growth hormone receptor (*GHR*, N = 158), serotonin receptor (*HTR1B*, N = 140), inter-photoreceptor retinoid-binding protein (*IRBP*, N = 154), melanocortin receptor 4 (*MC4R*, N = 62), rhodopsin (*RHO*, N = 136), and Wilms Tumor (*WT*, N = 74) (Supplementary Material 1). Available GenBank sequences were also included (Online Appendix 1), but these individuals were represented by only a single locus. Details on sample names, museum accession numbers, and GenBank accession numbers are provided in Colella et al. (2018): https://doi.org/10.5066/P9V1L0SI.

Sanger sequences were inspected (Sequencher v 5.4.1; Gene Codes Corporation, Ann Arbor, Michigan, USA) and aligned (Geneious, Kearse et al. 2012; MUSCLE, Edgar 2004). Nuclear heterozygotes were phased (Stephens et al. 2001: 5 runs, 1000 step burn-in, 1000 iterations). Support values > 0.9 were accepted, remaining ambiguities were coded as missing.

Phylogenetic methods

We determined the best-fit model of evolution (jModeltest2, Darriba et al. 2012; Supplementary Material 1) for each nuclear locus and the mitochondrial genome (complete and CDS only). Models were used to infer the phylogenetic relationships among North American *Martes* using both maximum-likelihood (RAxML, Stamatakis 2014; 10,000 iterations) and Bayesian techniques (MrBayes v3.2.5, Ronquist and Huelsenbeck 2003; 2 runs, 4 chains, 20 million generations, 25% burn-in, sampling every 10,000 generations) to build gene trees for each locus, with and without outgroups. Phylogenies were visualized in FigTree v1.4.2 (Rambaut 2012). Median-joining haplotype networks were inferred with the R package pegas (Paradis 2010).

To evaluate species relationships, we inferred a consensus species tree based on maximum-likelihood gene trees for nuDNA alone and for both nuclear and mitochondrial loci using the summary method: MP-EST (Liu et al. 2010). Potentially admixed samples (identified using the methods below) were excluded from species tree analyses. We tested two hypotheses: a single species of North American marten versus two species. To evaluate consistency across methodologies, we also used the multispecies coalescent method in StarBeast (Heled and Drummond 2010). Convergence was assessed in Tracer v1.6 (Rambaut et al. 2014) based on shape of relative-rate multipliers and effective sample size (ESS) values > 200. TreeAnnotator (Drummond et al. 2012; 25% burn-in) was used to determine the maximum clade credibility consensus tree. Runs were combined using LogCombiner v1.6.1 (Drummond et al. 2012) and visualized in FigTree v1.4.2 (Rambaut 2012).

Population genetic and demographic analyses

To characterize the distribution of genetic diversity across the landscape, we calculated nucleotide diversity (π), the number of haplotypes (1 bp difference for nuDNA; pegas), number of variable sites, and pairwise differences (Geneious) for each locus (mitochondrial and nuclear) and population. CollapseTypes (Chesters 2013) defined mitogenome haplotypes based on 5, 10, 15, and 20 bp differences for complete and CDS mitogenomes, while pegas identified nuDNA haplotypes based on 1 bp difference. Genetic distances were calculated in MEGA v6.06 (Tamura et al. 2013) for concatenated nuclear loci alone, for nuclear and mitochondrial data combined, and for mitochondrial data alone to provide a metric of population differentiation: between species, between pooled natural hybrid populations, and between independent localities within each species. Distances were calculated under a Poisson model with uniform substitution rates among sites, gaps and missing data ignored, and 500 bootstrap replicates.

Because nuclear loci deviated from Hardy–Weinberg equilibrium (HWE; Supplementary Material 2), the use of F-statistics (e.g., Wright's F_{ST}) to test for significant variance in allele frequencies among groups, or χ^2 tests of allelic differentiation to test for population structure, should not be applied indiscriminately, as both assume HWE. We therefore first conducted χ^2 tests of genotypic differentiation on nuclear loci, using GenePop '007 (Raymond and Rousset 1995; Rousset 2008), to determine if the distribution of nuclear alleles differed between species and the samples identified as hybrids. Comparisons of results of this analysis with results of tests for significant differences in the variance in allele frequencies between groups, conducted using ARLEQUIN v. 3.5 (Excoffier and Lischer 2010) were generally concordant (Supplementary Materials 3 and 4). We therefore conducted analyses of molecular variance (AMOVA), also in ARLEQUIN, to test for genetic differentiation between and among populations to understand how gene flow may impact the distribution of diversity across the landscape. We used the find.clusters function (adgenet R package; Jombart 2008) on all nuclear loci to run a discriminate principal component analysis (DAPC, Jombart and Ahmed 2011) to calculate the optimal number of clusters (K) in our dataset. We ran this analysis using the optimal number of principal components, as determined by the optim.a.score function (Jombart and Collins 2015), in an effort to avoid overfitting and increase stability of membership probabilities. DAPC offers an uninformed test of divergence between the two species and can identify potentially admixed individuals. DAPC was run on all available data

and again on samples with < 25% missing data. Because non-recombining mtDNA violates the assumption of independent inheritance, it was excluded from DAPC analysis. Lower BIC scores indicated the more optimal K values, but a range of K values can be biologically meaningful. Individuals with < 25% missing data and < 95% assignment to a cluster were considered putative 'hybrids' and excluded from population analyses.

Fu's F_S (Fu 1997; DNAsp; Librado and Rozas 2009), Tajima's D (Tajima 1989), and R_2 (Ramos-Onsins and Rozas 2002; pegas) were used to detect recent population expansion for each locus (ARLEQUIN; 10,000 simulations, $\alpha < 0.05$). Given the broad geographic distribution of *M. americana*, we expected to see signatures of expansion that contrast those in *M. caurina*. Because demography can profoundly influence the outcome of introgression, we generated Extended Bayesian Skyline Plots (EBSPs; Heled and Drummond 2008) in Beast v2.3.2 (2 runs, 300 million generations, sampling every 50,000 generations; Bouckaert et al. 2014) to estimate the historical demographic distributions for each *Martes* species, using default priors and a mitogenome rate of 3.8×10^{-8} substitutions/site/year (Hope et al. 2014). Nuclear rates were initiated at 2.2×10^{-9} substitutions/site/yr. Tracer v1.6 assessed convergence, as above.

Introgression analyses

We used two methods to detect introgression in North American *Martes*. INTROGRESS (Gompert and Buerkle 2010) was used for nuDNA only to estimate hybrid indices (Buerkle 2005) for samples collected in known contact zones (KUI, BCX, MTX) and those identified as potentially admixed by DAPC, regardless of collection locality. INTROGRESS was run on all loci examined, and again excluding individuals with > 25% missing data. Sequences were coded by genotype and grouped into populations: 'pure' species (*M. americana* or *M. caurina*) or admixed. We then used the parametric method (mtDNA haploid, nuDNA co-dominant; Gompert and Buerkle 2009) to estimate genomic clines for each locus and contrast empirical clines against neutral expectations to determine if the directionality or intensity of gene flow deviates from expectations.

To estimate gene flow (m) and effective population sizes (N_e), we used a 2-population model of isolation with migration (Nielsen and Wakeley 2001; Hey and Nielsen 2007; Hey 2010) in IMa2p (Sethuraman and Hey 2016) excluding individuals missing > 25% of the loci examined. IMa2p estimated N_e (ancestral and contemporary populations/species) and two migration rates (m) scaled to the mutation rate per locus per year (μ). We analyzed two datasets with admixed individuals excluded: (1) complete mitogenomes and (2) nuclear loci, treated independently. Because IMa2p assumes no recombination within a locus, IMgc (Woerner et al. 2007) truncated nuclear loci to the largest non-recombining fragment. Inheritance scalers were 0.25 for mtDNA and 1.0 for nuDNA. Priors were determined by testing and narrowing wide estimates. Final runs for both datasets included a geometric heating scheme (–ha 0.99 –hb 0.98) for 20 million iterations after a 1 million burn-in, sampling every 100 steps. We ran 3 IMa2p iterations with different random seeds to confirm convergence (ESS > 100; Hey 2005).

Results

We examined 1444 DNA sequences (1345 mtDNA [52 mitogenome, 82 *COX1* (30 in addition to those extracted from mitogenome sequences)] and 1310 nuDNA) across 90 specimens representing six 'pure' americana localities, 6 'pure' *M. caurina* localities, four naturally introgressed populations, and three translocated *M. americana* islands (Table 1).

Phylogenetic inference

Mitochondrial gene trees support the presence of two species (*M. americana* and *M. caurina;* mitogenomes Supplementary Materials 5 and *COX1* Supplementary Materials 6) in North America, with varying nuclear support (Gene tree, Supplementary Materials 7 and haplotype networks, 8). Topologies were consistent across methods, but demonstrated limited intraspecific structure and occasionally showed North American *Martes* as paraphyletic with respect to Eurasian sable. Therefore, we included mink (*Neovison vison*) as an alternative outgroup. Sable nuDNA sequences showed elevated similarity with *M. caurina* (e.g., identical *ASIP, FES, WT* haplotypes). Nuclear loci *WT* and *CHRNA* were sorted between *M. caurina* and *M. americana* and 11 fixed differences in *COX1* haplotypes separate these species (Fig. 2; Supplementary Material 8).

Chi square tests of genotypic differentiation and tests for differences in the variance of allele frequency found significant differences between the two species for nuclear (8/10) loci (Supplementary Material 3). Estimates of evolutionary distance over sequence pairs between groups show significant differentiation between *M. americana, M. caurina*, and a single, pooled hybrid population (Supplementary Material 4), with slightly less divergence between hybrids and *M. americana*, compared to hybrids and *M. caurina*. All populations within each species were also significantly differentiated in estimates of evolutionary divergence (Supplementary Material 9). AMOVA tests also showed significant genetic differentiation between species (8 loci), between island/mainland populations within each species (9 loci), and between populations of both species (*M. americana*, 7 loci; *M. caurina*, 5 loci;

Table 1 Number of sequences (*n*) examined and the number of haplotypes (*H*) evaluated for each locus, by species and locality. Haplotypes are defined based on a 1 bp difference, except for mitogenomes which are based on a 20 bp difference threshold. For mitogenomes, additional, but incomplete sequences that were incomplete and therefore not haplotyped are noted in parentheses and islands that received translocations of *M. americana* are denoted with an asterisk

Locus		MT		*COX1*		*AdiQ*		*ASIP*		*CHRNA*		*FES*		*GHR*		*HTR1B*		*IRBP*		*MC4R*		*RHO*		*WT*		Sum
Populations	*Abbr.*	*n*	*H*	*n*	*H*	*n*	*H*	*n*	*H*	*n*	*H*	*n*	*H*	*n*	*H*	*n*	*H*	*n*	*H*	*n*	*H*	*n*	*H*	*n*	*H*	*N*
Martes americana		27	8	34	5	66	2	66	2	64	4	64	4	68	3	60	3	70	10	22	2	58	2	26	3	625
Mainland		11	5	12	5	26	2	24	2	24	4	24	4	26	3	24	1	28	8	8	1	22	2	10	2	239
Mainland Canada	LAB	1	1	1	1	4	1	2	2	2	2	2	2	2	1	2	1	2	2	2	1	–	–	2	1	22
Mainland Alaska	MAK	5 (4)	4	9	4	18	2	18	2	18	3	18	4	18	3	18	1	18	6	6	1	16	1	8	2	165
Continental U.S.	MLU	–	–	–	–	–	–	–	–	–	–	–	–	2	1	–	–	2	1	–	–	2	1	–	–	6
North Pacific Coast	NPC	1	1	2	2	4	1	4	2	4	3	4	2	4	2	4	1	6	4	–	–	4	1	–	–	37
Island		16	3	22	2	40	2	42	2	40	3	40	4	42	3	36	3	42	5	14	2	36	2	16	2	386
Baranof Is., AK*	BAR	2	1	3	1	4	1	4	2	4	2	6	3	6	3	4	1	6	3	2	1	6	1	6	1	53
Chichagof Is., AK*	CHI	–	1	3	2	6	1	6	2	4	2	4	2	4	2	2	1	6	3	4	2	–	–	4	2	43
Dall Is., AK	DAL	7	1	8	1	14	2	16	1	16	3	14	3	16	3	14	2	14	1	4	1	14	1	2	1	139
Prince of Wales Is., AK*	POW	2 (2)	1	5	1	10	2	10	2	10	3	10	3	10	3	10	3	10	3	–	–	10	1	–	–	85
Suemez Is., AK	SUE	3	2	3	1	6	2	6	1	6	1	6	2	6	3	6	1	6	2	4	2	6	1	4	1	62
Hybrid localities		14	7	22	4	24	3	38	3	38	7	44	4	40	4	38	6	38	4	22	3	38	2	26	8	382
Mainland		9	3	11	4	22	3	24	3	24	4	24	3	24	3	24	4	24	4	12	2	20		18	6	236
Northern Rocky Mts.	BCX	4 (2)	3	6	1	10	2	12	3	12	4	12	3	12	3	12	2	12	4	6	2	10	2	10	5	114
Central Rocky Mts.	MTX	2 (1)	1	5	4	12	3	12	1	12	3	12	2	12	2	12	3	12	3	6	1	10	1	8	3	113
Island		5	5	11	3	2	1	14	1	14	5	20	3	16	4	14	4	14	3	10	2	18		8	5	146
Kuiu Is., AK	KUI	3	3	6	2	–	–	6	1	4	3	10	3	6	3	6	3	4	3	2	1	8	1	4	2	59
Kupreanof Is., AK	KUP	2	2	5	2	2	1	8	1	10	4	10	2	10	3	8	2	10	3	8	2	10	1	4	4	87
Martes caurina		12	8	25	6	32	3	48	3	52	4	48	5	50	3	42	6	48	4	18	2	40	1	22	2	437
Mainland		6	4	15	6	18	3	28	2	32	3	28	4	30	3	26	5	28	3	10	1	22	1	10	2	253
New Mexico, Colorado	NMC	2	2	3	3	4	2	6	1	8	3	6	4	6	2	2	1	6	1	–	–	4	1	–	–	47
Pacific Northwest	PNW	3	2	8	3	10	2	16	2	16	1	16	1	16	2	16	4	14	3	10	1	12	1	4	2	141
Wyoming	WYO	1	1	4	2	4	3	6	1	8	1	6	1	8	1	8	3	8	1	–	–	6	1	6	1	65
Island		6	4	10	1	14	1	20	2	20	2	20	2	20	2	16	3	20	3	8	2	18	1	12	1	184
Admiralty Is., AK	ADM	1	1	3	1	2	1	6	1	6	1	6	1	6	2	4	2	6	1	4	1	6	1	4	1	54
Queen Charlotte Is., BC	QCI	3	2	5	1	8	1	10	2	10	2	10	1	10	1	8	2	10	3	–	–	10	1	4	1	85
Vancouver Is., BC	VAN	1	1	2	1	4	1	4	1	4	1	4	2	4	1	4	2	4	1	4	2	2	1	4	1	41
Total *n* examined		52		82		122		152		154		156		158		140		156		62		136		74		1444
Total *H* examined			20		12		3		5		9		7		7		10		12		4		3		9	101

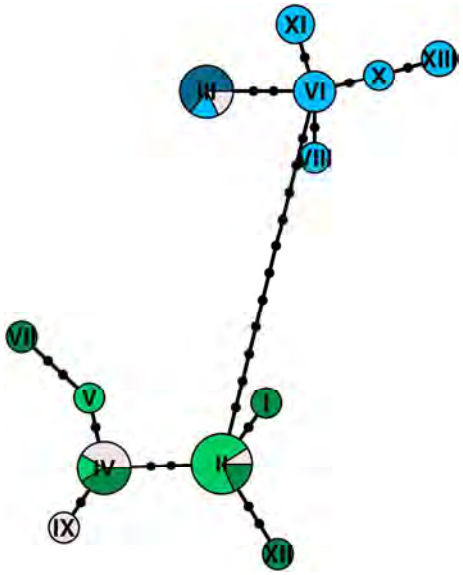

Fig. 2 Haplotype network for mitochondrial gene *COX1* showing 11 bp differences (1 dot = 1 bp change) separating North American *Martes* species. *Martes americana* haplotypes are in green and *M. caurina* in blue, with darker shades indicating samples collected from islands and lighter shades indicating samples collected from mainland localities. Individuals collected in contact zones are shown in grey

Supplementary Material 10). *CHRNA* and *WT* had the majority of variation spread among species, among island/mainland populations of each species, and among *M. caurina* populations.

Summary species-tree methods (MP-EST; Fig. 3), which used mink as an outgroup, support monophyly of North American *Martes* and suggest each species is an independent operational taxonomic unit (ESS > 200). The multispecies coalescent method (StarBeast; Fig. 3) also supports two independent marten species in North America.

Population genetic and demographic inference

The number of nuclear haplotypes sampled ranged from 1 to 12, with *IRBP* (12) and *WT* (9) having the highest haplotype diversity (Table 1). Nucleotide diversity varied by locus and population, but was generally higher for 'pure' *M. americana* populations relative to 'pure' *M. caurina* populations (Supplementary Material 11). Eleven nuclear haplotypes were restricted to naturally admixed populations, 12 to *M. caurina* populations (8 mainland, 4 island), and 14 to *M. americana* populations (10 mainland, 3 island, 1 both; Supplementary Material 7 and 8). We identified 20 complete mitogenome haplotypes (8 *M. caurina*, 2 contact-zone, 10 *M. americana*) and 10 CDS haplotypes (3 *M. caurina*, 1 *M. caurina*/contact-zone, 1 contact-zone, 1 *M. americana*/contact-zone, 4 *M. americana*) based on a 20 bp difference threshold (Supplementary Material 12). Average pairwise distance between the species' complete mitogenome haplotypes was 4.4%, with > 6% differentiation between North American *Martes* and sable (6.1% *M. caurina*, 6.3% *M. americana*). Nine mitochondrial haplotypes are shared between *M. americana* and natural hybrid populations, while only one is shared between *M. caurina* and hybrid populations.

Low BIC values in the DAPC cluster analysis (Supplementary Material 13) suggest that 2–5 K-clusters provide useful summaries of the data (Supplementary Materials 14–16). The greatest BIC delta (Evanno et al. 2005) identifies a 2-population model (K2) as most useful for describing variation in North American *Martes* (Jombart and Collins

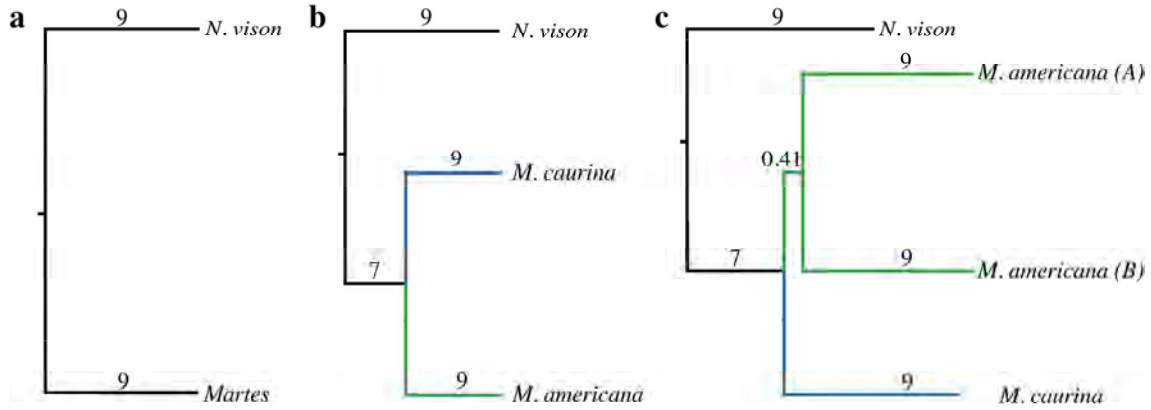

Fig. 3 Consensus MP-EST species trees show equivalent support for a single monophyletic *Martes* species (**a**) and two North American *Martes* species (**b**), corresponding to *M. americana* (green) and *M. caurina* (blue), with intermediate support for intraspecific substructure within *M. americana* (**c**). Branch lengths are in coalescent units, a branch length of '9' indicates that all gene tree triples supported the same topology. This provides strong support for two monophyletic *Martes* species in North America, but the actual branch length cannot be estimated due to the lack of topological variance among gene tree triples (Liu et al. 2010)

2015). The K2 model shows a clean, almost non-overlapping split between the two North American *Martes* species, while K3 shows the two species plus an intermediate, potentially hybrid population (Fig. 4). Under the K2 model, nine individuals were identified as potentially admixed, including three from Kuiu Island. However, four samples identified as potentially admixed had > 25% missing data and were therefore presumed to be inaccurately identified as admixed due to incomplete data. Among these false positives were three GenBank samples represented by only a single locus. The remaining samples were considered to be putative hybrids (N = 5; Supplementary Material 17). K3, K4, and K5 models found few samples that remained unsorted (< 95% posterior probability; Supplementary Material 15) between the clusters; However, because these partitioning schemes recognize a hybrid-like or intermediate cluster, independent from the two parental species clusters, this result is unsurprising. K4 and K5 exhibit little geographic partitioning, likely as a consequence of the rapid expansion of *M. americana* and historic translocations.

Surprisingly, INTROGRESS and DAPC (K3) both identified MSB197298 from mainland Alaska as potentially admixed. Also, contrary to expectations, DAPC classified two samples collected in the American Southwest (NMC = New Mexico-Colorado region) as *M. americana* and three as *M. caurina*; however, the two *M. americana* samples were recovered from GenBank and represented by only a single locus. Another sample with incomplete data (UAM54568) from Chichagof Island was identified as potentially admixed under a K2 model. Given the degree of missing data for these three samples, caution should be exercised in the interpretation of their results and further sampling in these regions and an expanded set of loci is necessary.

EBSPs for *M. caurina* weakly support constant N_e through time and were similar for *M. americana*, but followed by expansion beginning around 30 Kya (Supplementary Material 18). EBSPs showed limited informative variation in the *M. caurina* dataset, possibly as a consequence of inbreeding depression, reduced N_e, or too few samples, causing *M. caurina* N_e distributions to simply reflect priors. Similar contemporary N_e were estimated for both *M. americana* (30 k) and *M. caurina* (32 k) based on mtDNA. IMa2p estimates of N_e were also similar: 44,313 (HPD95% CI: 23,773–85,267) for *M. americana* and 35,779 (14,145–173,864) for *M. caurina* mtDNA (Fig. 5; Supplementary Material 19). For nuDNA, N_e was 9,948 (4757–19,463) and 13,840 (6487–28,114) for *M. americana* and *M. caurina*, respectively (Fig. 5; Supplementary Material 19). Although the use of alternative mutation rates does alter N_e estimates, the overall patterns remain the same. We suspect the actual mitochondrial mutation rate is slower than that predicted by Hope et al. (2014) and our data suggests that a slower mutation significantly lowers the mean N_e (Supplementary Data 19–20). This may explain why our nuDNA N_e estimate is lower than that for mtDNA. Contemporary population sizes were slightly larger than ancestral based on mtDNA, suggesting expansion following

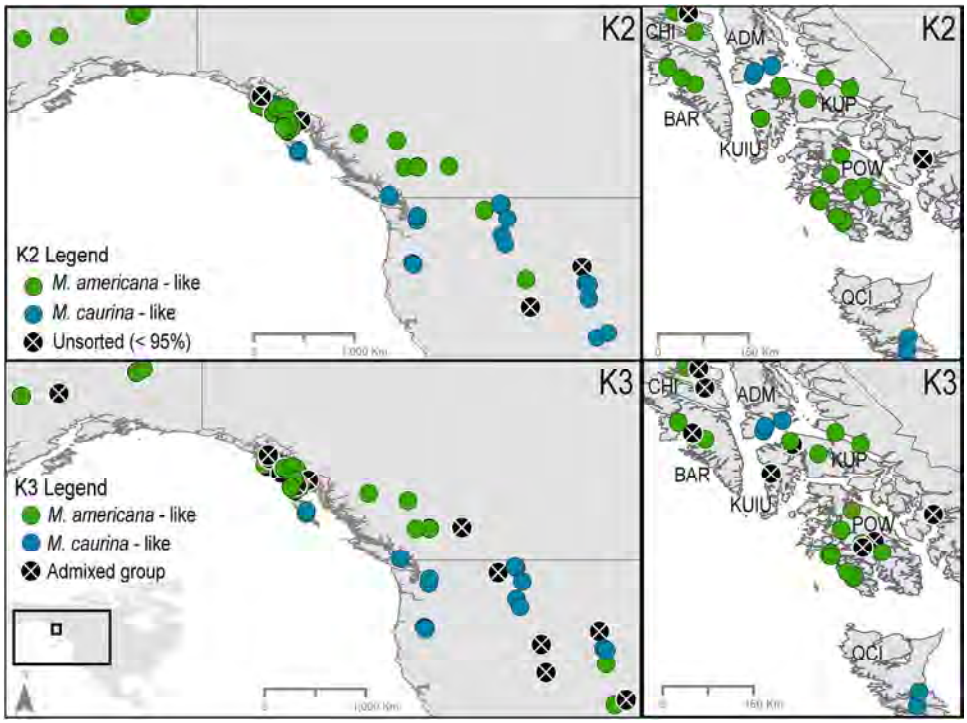

Fig. 4 Geographic distribution of DAPC results for 2 and 3 clusters (K), corresponding to Supplementary Materials 14 and 15. K2 (top row) shows a clean, almost non-overlapping split between the two North American *Martes* species, with the *M. americana*-like cluster shown in green, the *M. caurina*-like cluster in blue, and individuals sorted with < 95% posterior probabilities in black which a white 'X', indicating that they are potentially admixed. K3 (bottom row) parses the two *Martes* species (again, green and blue respectively) plus an intermediate, potentially hybrid population (Fig. 4), labeled as black with a white 'X'. For K3, admixed individuals include all specimens sorted into group '3' and those that were not confidently (< 95%) sorted into either parental species (N = 2; Supplementary Material 15)

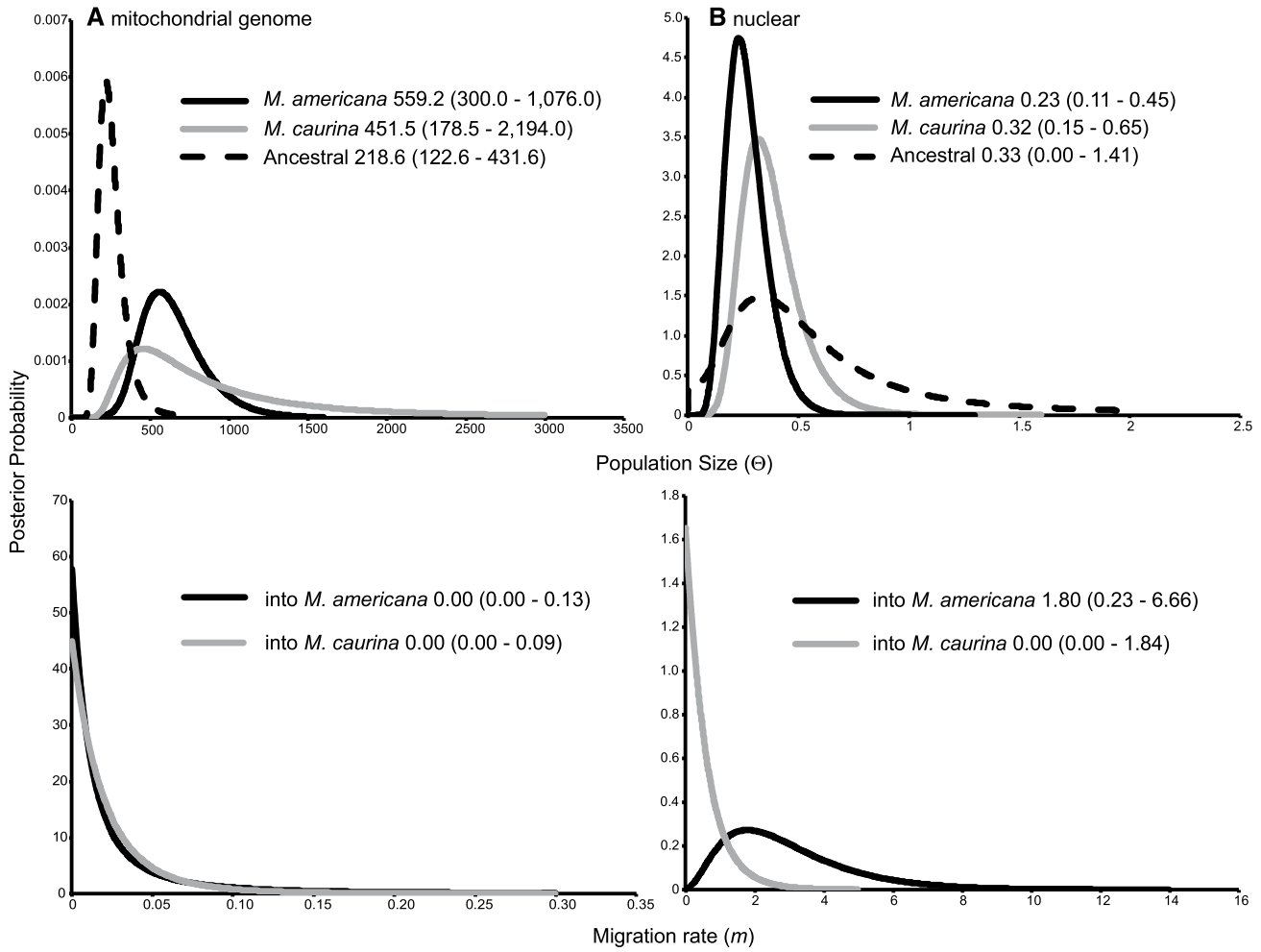

Fig. 5 IMa2p estimates of population size (Θ) and migration rate (*m*) between *M. americana* and *M. caurina* with peak values and 95% highest posterior density indicated

divergence. For nuDNA, posterior distributions for contemporary and ancestral population sizes broadly overlapped suggesting constant population sizes (Fig. 5). Expansion statistics showed inconsistent patterns across genes, lineages, populations, and species (Supplementary Material 21).

Inference of introgression

INTROGRESS identified significant asymmetrical gene flow from *M. caurina* into *M. americana* in one nuclear locus (*AdipoQ*) (Supplementary Material 22). *WT* exhibited a steep cline, associated with decreased rates of introgression relative to neutral expectations, while *FES* and *GHR* exhibited shallow clines associated with increased rate of introgression; however, these relationships were non-significant. Cline shapes are expected to vary between nuclear loci, as each marker examined is independently inherited with a distinct coalescent and, therefore, introgression history. We identified 12 individuals with intermediate hybrid indices (*HI*; Supplementary Material 23), including 4 from Kuiu Island and 7 from the mainland Rocky Mountain hybrid zone (3 British Columbia, 4 Montana). Admixed individuals identified by INTROGRESS were consistent with those identified through DAPC analyses, with the exception of four additional specimens that had > 25% missing data (e.g., AF498196, AF498220, MSB268541, UAM54568) which were not labeled as admixed by INTROGRESS and one individual from the mainland contact zone (MSB197115), however the range of this individual's *HI* 95% confidence interval still suggests the possibility of introgression.

IMa2p migration rates revealed restricted gene flow between species in both mitochondrial and nuclear datasets (Fig. 5). Migration posterior distributions support the hypothesis of no mitochondrial or nuclear gene flow from *M. americana* into *M. caurina*. However, this hypothesis (no gene flow) was rejected for *M. caurina* into *M. americana* (posterior = 1.80; 0.23–6.66) with an estimated migration rate (2 N m; 2N is the effective number of gene copies and

m is the rate at which genes are supplanted by incoming copies) of 0.28 (0.05–0.71; Fig. 5).

Discussion

Increasing recognition of the ubiquity of introgression in vertebrates through genomic investigations has led to a greater appreciation of the varied evolutionary consequences of gene flow and highlights the importance of incorporating gene flow estimates, on a case-by-case basis, into conservation strategies aimed to protect the natural processes influencing the generation and persistence of biodiversity (Haig and Allendorf 2006). In parallel, phylogeographic investigations of natural hybrid zones provide a framework to interpret how translocations may impact genetic diversity. Such perspectives may also be extended to sympatric species that share similar biogeographic histories with dynamic patterns of introgression.

Dynamic history of introgression

We identified hybridization in two zones of contact between North American martens: one on Kuiu Island in Southeast Alaska and another in the northern Rocky Mountains stretching from western Montana north into British Columbia, that were previously hypothesized with mtDNA (Fig. 1). Outside of these zones, *Martes* species remain genetically distinct, with the exception of potentially remnant signatures of historical admixture in mainland Alaska and through the Rocky Mountains into New Mexico/Colorado (Fig. 4; Supplementary Material 17). These remnants are geographically distant from contemporary hybrid zones and may reflect past episodes of introgression or incomplete lineage sorting correlated with cycles of range expansion and contraction. In Alaska, however, the lack of genetic or LGM fossil evidence of *M. caurina* suggests that Pacific marten were either recent colonizers, extirpated from the region, or share ancestral gene flow with Eurasian sable, as hypothesized by Kurtén and Anderson (1980). Elevated genetic similarity between sable and Pacific marten points toward a dynamic history of introgression throughout *Martes* that should be explored with expanded sampling and genomic-level analyses.

Of the 12 admixed specimens identified by intermediate *HI* (Supplementary Material 23), all but one (from Kuiu Island) have *M. americana* mtDNA. This skew suggests that hybrid offspring are primarily the result of female *M. americana* and male *M. caurina* crosses, a bias that could be driven by genetic incompatibilities (BDM, Schilthuizen et al. 2011), demography (e.g., mate availability, unequal N_e), or asymmetrical mating preferences. *Martes caurina* males may disproportionally contact more *M. americana* females (desperation hypothesis; Hubbs 1955) or *M. americana* females may be less choosy and more prone to mate with a non-conspecific (e.g., *Mus*; Teeter et al. 2008). Biased introgression is further supported by the asymmetrical movement of nuclear genes from *M. caurina* into *M. americana*. Consequentially, continued directionally-biased introgression may slowly remove *M. caurina* alleles from the combined gene pools. In cases like this, where hybridization acts as a sink for one species, reinforcement (Dobzhansky 1937) may expedite divergence.

Although introgression does not inherently compromise species boundaries, extended gene flow and slow emergence of isolating barriers may facilitate homogenization and the loss of biodiversity (Rhymer and Simberloff 1996). Even the peripheral fusion of smaller, endemic *M. caurina* populations with *M. americana* could result in local extirpation of the former. If *M. caurina* were present on North Pacific islands prior to translocations and reproduced with introduced *M. americana*, this signal should be evident in descendant genomes. Under a K2 model, the two marten species are diagnostically sorted and we detected a weak signature of introgression (K2; Fig. 4; Supplementary Materials 14–16) in a single Chichagof individual with incomplete data and a coastal Southeast Alaskan sample. However, K3 identified two clusters corresponding to each species and a third 'admixed' or intermediate cluster consistent with hypothesized hybrid zones (Kuiu/Kupreanof Islands and the Rocky Mountains). Additionally, DAPC (K3) was the only analysis that identified potentially admixed individuals on all translocated islands—Prince of Wales, Baranof, and Chichagof (Fig. 4)—as Pauli et al. (2015) hypothesized for Prince of Wales Island marten. Although parasitological evidence also supports the historical presence of *M. caurina* on several NPC islands (Hoberg et al. 2012), our inconclusive findings could indicate (I) that *M. caurina* were not present prior to introductions, (II) that the sampled genes provide insufficient resolution to detect *M. caurina*, or (III) that *M. caurina* alleles decayed and eventually disappeared following introductions and subsequent introgressive hybridization. Genetic signatures of introgression can be lost in 4 generations of backcrossing (20 years for marten) (Lavretsky et al. 2016) and swamping may be accelerated by asymmetrical gene flow or a demographic imbalance. Therefore, if *M. caurina* was swamped by introduced *M. americana*, there may be no signature of admixture remaining in these populations or the signature is so diluted that it requires whole genome-level analyses to detect it. Wildlife translocations that lack knowledge of existing genetic structure or accurate taxonomy can have cascading effects, particularly on endemic populations (coastal islands and sky islands of the American Southwest; Soulé 1987; Lande 1993). Although recolonization can similarly alter the genetic composition of populations (Stewart et al. 2017), the islands that received *M. americana* translocations are isolated from mainland

populations by 10–70 kilometers of ocean, suggesting that natural colonization would be extremely unlikely. Even if martens were capable of such oceanic dispersal, colonization by multiple individuals, with representatives of each sex, seems unlikely without human intervention. In these cases, human-aided translocations may have triggered the extirpation of remnant *M. caurina* from several NPC islands, but this warrants further investigation with expanded sampling.

In North America, natural marten hybrid populations (Kuiu Island and northern Rocky Mts.) account for 15% of observed mitochondrial (3/20 haplotypes restricted to hybrid zones) and nuclear haplotype diversity (10/66 total nuclear haplotypes). The geographic distribution of mixed haplotypes suggests that genes flow into hybrid zones, but not out. That is, hybrid zones effectively act as evolutionary sinks. The loss of *M. caurina* due to homogenization or genetic swamping would reduce North American *Martes* mitochondrial diversity by half and eliminate 13 unique nuclear variants; however, the long-term contribution of these rare variants to *Martes* evolution remains to be explored. Low hybrid zone variation combined with asymmetrical gene flow suggests that hybridization may lead to genetic swamping of *M. caurina* by *M. americana*. Given our results, and with the goal of sustaining biodiversity, introductions of *M. americana* onto islands would likely not result in the genetic rescue of *M. caurina*. Mainland to island translocations may further jeopardize the persistence of *M. caurina* alleles by increasing the frequency of hybridization beyond natural zones of introgression. Cook et al. (2006) suggest that each insular marten population (*M. caurina* or *M. americana*) be managed independently.

Taxonomy, introgression, and management

New perspectives on *Martes* taxonomy (Stone et al. 2002; Small et al. 2003; Dawson et al. 2017), including our expanded genetic analyses, are consistent with their original classification as two species (Turton 1806; Merriam 1890); however, these mesocarnivores have been managed as a single species for the last 70 years, including numerous translocation efforts. The demographic history of *M. americana* (Supplementary Materials 17) suggests a stable and constant population size throughout history, followed by recent expansion near the conclusion of the LGM (~21 Kya). This pattern is consistent with the hypothesized expansion from a single eastern glacial refugium, as proposed by previous research (Stone and Cook 2002). The lack of signal in the *M. caurina* EBSP (Supplementary Material 17) may indicate a historical bottleneck event, sufficiently severe to permanently alter the genetic signature. Although not diagnostic, a bottleneck for *M. caurina* is plausible considering their current disjunct and fragmented distribution. If hybridization is common and hybrid offspring are significantly less fit than parental genotypes, we expect population growth rates, particularly for *M. caurina*, to drop in areas of secondary contact (Todesco et al. 2016). Recognition of two marten species in North American has significant implications for the reinterpretation of diverse aspects of marten biology, ranging from evaluating critical habitat and occupancy models (Slauson et al. 2012) to understanding the evolution and biogeographic histories of their parasite communities and pathogens (Hoberg et al. 2012). Due to the widespread extirpation of martens across North America (Krohn 2012), translocations of marten to repopulate northern and western forests have been extensive, and include instances where both species were simultaneously introduced to a region (Kohn and Eckstein 1987; Williams et al. 2007). Genetic reevaluation of one such mosaic population, up to 65 years after introductions occurred, failed to uncover evidence of *M. caurina* persistence (Grauer et al. 2017), consistent with our results showing that hybridization results in the directional loss of *M. caurina* alleles. This potential for genetic swamping of *M. caurina* by *M. americana* may be exacerbated by a demographic imbalance and reinforces the importance of carefully selecting source populations, based on a firm grasp of phylogeographic variation and taxonomy. The revised taxonomic distinction proposed here has profound management implications for island endemic *M. caurina*, where several populations may be impacted by the many translocations of *M. americana* over the past century.

The extended history of marten translocations offers a cautionary tale to natural resource managers. Introductions that result in asymmetric introgression may lead to the extirpation of endemic populations or native species. Natural hybrid zones offer an experimental model for predicting the outcome of anthropogenically-mediated introgression in cases of genetic rescue or population supplementation. Additionally, proposed timber sales on Kuiu Island (USDA 2007a, b) and elsewhere in the Alexander Archipelago, may further impact the genetic structure of endemic and indicator species (e.g., inbreeding, Frankham 2005; loss of critical habitat; Hanski 2005; fragmentation; Keyghobadi 2007). At a broader scale, industrial timber harvests, catastrophic fires, and other perturbations have impacted western forest species assemblages, increasing the potential for instances of maladaptive hybridization (Anderson 1948; Todesco et al. 2016; Linnell et al. 2018). As such, caution should be exercised in both source population selection and the prescription of genetic rescue (or gene tweaking) as a means of conservation (Thomas et al. 2013), particularly in these at-risk, disturbed areas. One of the more powerful resources available to natural resource managers is the availability of geographically broad and temporally deep biological sample archives in museums. These biobanks provide objective measures of how populations vary through time by providing baseline or historical conditions. For example, expansion of

the insular marten hybrid zone (or at least diffusion of *M. caurina* alleles) from Kuiu to Kupreanof island, detected here, was only evident in tissue samples collected after the publication of Stone et al. (2002), potentially highlighting the importance of continued sampling through time to build archives that span multiple years and permit rigorous investigation into these dynamic mixing zones.

Biased introgression in a natural *Martes* hybrid zone on Kuiu Island in southeast Alaska, extending into neighboring Kupreanof Island, may have placed insular endemic *M. caurina* at an increased risk of extirpation, particularly in light of ongoing industrial-scale habitat modification on these islands. The genetic histories of North American martens function as a warning to wildlife managers, highlighting the importance of understanding evolutionary relationships within and among managed species and mindful source population selection prior to introductions, as introgression can have profound impacts on the genetic diversity and persistence of existing populations.

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Whole-genome analysis of *Mustela erminea* finds that pulsed hybridization impacts evolution at high latitudes

Jocelyn P. Colella[1], Tianying Lan[2,3], Stephan C. Schuster[4], Sandra L. Talbot[5], Joseph A. Cook[1] & Charlotte Lindqvist[2,3]

At high latitudes, climatic shifts hypothetically initiate recurrent episodes of divergence by isolating populations in glacial refugia—ice-free regions that enable terrestrial species persistence. Upon glacial recession, populations subsequently expand and often come into contact with other independently diverging populations, resulting in gene flow. To understand how recurrent periods of isolation and contact may have impacted evolution at high latitudes, we investigated introgression dynamics in the stoat (*Mustela erminea*), a Holarctic mammalian carnivore, using whole-genome sequences. We identify two spatio-temporally distinct episodes of introgression coincident with large-scale climatic shifts: contemporary introgression in a mainland contact zone and ancient contact ~200 km south of the contemporary zone, in the archipelagos along North America's North Pacific Coast. Repeated episodes of gene flow highlight the central role of cyclic climates in structuring high-latitude diversity, through refugial divergence and introgressive hybridization. When introgression is followed by allopatric isolation (e.g., insularization) it may ultimately expedite divergence.

[1] Division of Mammals, Museum of Southwestern Biology and Department of Biology, 1 University of New Mexico, MSC03-2020, Albuquerque, NM 87131, USA. [2] Department of Biological Sciences, University at Buffalo (SUNY), 109 Cooke Hall, Buffalo, NY 14260, USA. [3] School of Biological Sciences, Nanyang Technological University, 60 Nanyang Dr., Singapore 637551, Singapore. [4] Singapore Centre on Environmental Life Sciences Engineering, Nanyang Technological University, 60 Nanyang Dr, Singapore 637551, Singapore. [5] U.S. Geological Survey Alaska Science Center, 4210 University Dr., Anchorage, AK 99508, USA. Correspondence and requests for materials should be addressed to J.P.C. (email: jcolella.jc@gmail.com) or to C.L. (email: cl243@buffalo.edu)

Glacial refugia are fragmented pockets of unglaciated land where terrestrial species have persisted and diverged through ice ages, and which have significantly structured biological diversity at high latitudes[1–3]. Three primary macro-refugial locations are hypothesized to have enabled the persistence of temperate terrestrial North American species through Pleistocene glacial cycles (>24 glacial/interglacial cycles; 2.6 Mya–11.7 kya). These include an expansive Northern Beringian refugium shared with Asia[4] and two Southern refugia, separated into East and West by the Rocky Mountains or Great Plains/Mississippi drainage[5,6]. A fourth, smaller refugium (or series of refugia), hypothesized to exist on areas of exposed continental shelf during the last glacial maximum near contemporary North Pacific Coastal archipelagos[7–10], may have provided another isolated sanctuary for long-term coastal persistence of terrestrial species. Although this refugium has received varying degrees of empirical support, its existence is critical to understanding dynamics of intercontinental biotic exchange and it may have played a primary role in the early peopling of the Americas[11].

Phylogeography of high-latitude *Mustela erminea*, the stoat or ermine (Family: Mustelidae) delineates four genetically distinct mitochondrial lineages geographically corresponding to these four refugia[12]. Although glacial isolation appears to have impacted the long-term evolution of this mammalian meso-carnivore, the effects of glacial recession and subsequent secondary contact between refugial lineages remain unexplored. Therefore, we investigated the consequences of cyclic climates and pulsed introgression on genomic diversity in this circumboreal species complex. During inter-glacial warming, divergent lineages expanded from refugial centers until they reached environmental or biological barriers[13]. In some cases, admixture occurred upon secondary contact with other lineages. If *M. erminea* experienced serial bouts of climate-mediated introgression, a repeated signature of introgressive hybridization should be evident in the genomes of refugial descendants[14].

Contact zones are windows into evolutionary processes[15] now being explored on a genomic scale. We define introgressive hybridization as interbreeding and the movement of alleles between two genetically distinct lineages. Although often difficult to parse introgression from incomplete lineage sorting in closely related taxa or lineages[16], investigations of divergent lineages, such as those within the *M. erminea* complex, provide insight into the genomic consequences of introgression. For example, genetic swamping or homogenization can prevent divergence between mixing lineages, while Bateson–Dobzhansky–Muller[17–19] and mitonuclear[20] incompatibilities can lead to hybrid breakdown, thereby promoting divergence through reinforcement[21,22]. Although our understanding of hybridization emerges largely from research on plants[23,24], the genomic era has uncovered substantial reticulate evolution (e.g., introgressive hybridization) in a growing number of animal species[25,26], including humans[27]. As histories of divergence and speciation-with-gene flow continue to be discovered (e.g.,[28,29]), the implications of hybridization for the conservation of evolutionarily distinct units appear profound[30].

To test whether genomic structure in stoats reflects refugial origins and to explore the consequences of contact among lineages, we generated ten whole-genome sequences of representatives from each refugial clade. With expanded genomic coverage, we test for signatures of refugial divergence and introgression, characterize the timing of introgression events relative to climatic oscillations, and infer the impact of introgressive hybridization on divergence across the complex landscape of northwestern North America.

Results

Refugial origins and phylogeography. To test our hypothesis that the four mitochondrial stoat lineages originated as the result of isolation and divergence in each of the four major North American refugia, we generated whole-genome sequences for ten *M. erminea* drawn from each refugial lineage and two spatially disjunct contact zones (the interior border of Alaska-Yukon Territory and the NPC; Fig. 1). We contrasted mitochondrial and nuclear phylogenies against geographic clade distributions, estimated pairwise diversity metrics (pairwise mitochondrial and nuclear divergence, F_{ST}[31], relatedness[32]), and used ADMIXTURE[33] analyses to define populations.

Our results showed strong mitochondrial (Fig. 1; Supplementary Fig. 1) and nuclear (Figs. 1–3; Supplementary Fig. 2; Supplementary Table 1) support for four stoat clades geographically coincident with North American refugia. We identified deep divergence of a highly distinctive West clade, a group previously lacking strong nuclear support[12] (Figs. 1, 2a and c; Supplementary Fig. 2; Supplementary Tables 1 and 2). We detected substantial genetic divergence between mainland stoat lineages (e.g., 21% average pairwise difference across >3 million autosomal SNPs between Beringia and East clades), comparable to estimates between Beringia stoats and the outgroup species, *M. putorius* (domestic ferret, 20%), calling into question whether this widespread mustelid is a single species (Supplementary Table 1). Centrality of the Vermont sample is consistent with widespread expansion of the East lineage from a single eastern refugium followed by intra-clade divergence (Fig. 2b). Similar refugial signatures have been detected in other North American species (e.g., martens[34] and black bears[35]). Additional structure within the East clade (Fig. 2b) may reflect microrefugial diversification[36]. Geographic proximity (Fig. 1) is not predictive of genetic similarity (Fig. 2b; e.g., REV and BC), consistent with patterns also uncovered in primary stoat prey species such as voles[37] and lemmings[38].

The sister relationship between NPC and Beringia stoats, combined with substantial genetic divergence (nuclear >10%; mitochondrial >4%; Supplementary Table 1) between these clades and their current geographic distributions, are consistent with the Coastal Refugium Hypothesis[7–10]. Ancestry estimation[33] identified autonomous East and West populations but a single combined Beringia/NPC population (K = 3; Fig. 2c), again highlighting the paleoendemic ancestry of the NPC Island lineage. Pairwise mitochondrial divergence is also particularly elevated for the NPC Island lineage, bordering the species-level divergence threshold suggested for distinct mammal species[39].

Detecting introgression and determining source populations. Evidence of recurrent introgressive hybridization, whether as a single or recurrent event, should be evident in the genomes of refugial descendants. We hypothesized that signatures of introgression should be present in individuals collected within the Alaska-Yukon Border hybrid zone and expected that stoats from Beringia and East clades are the most likely source populations based on contemporary clade distributions. In addition to contrasting mitochondrial and nuclear phylogenies for discordance, we performed a series of tests for introgression including ADMIXTURE[33] and *f3*-statistics (Supplementary Table 3)[40,41] to assess shared genetic drift among all possible combinations of two source populations[42].

We identified mitonuclear discordance for two individuals collected near the Alaska-Yukon border (herein SYT and YTAK, representing Southern and Northern localities along the contact zone, respectively), which can indicate a history of genomic

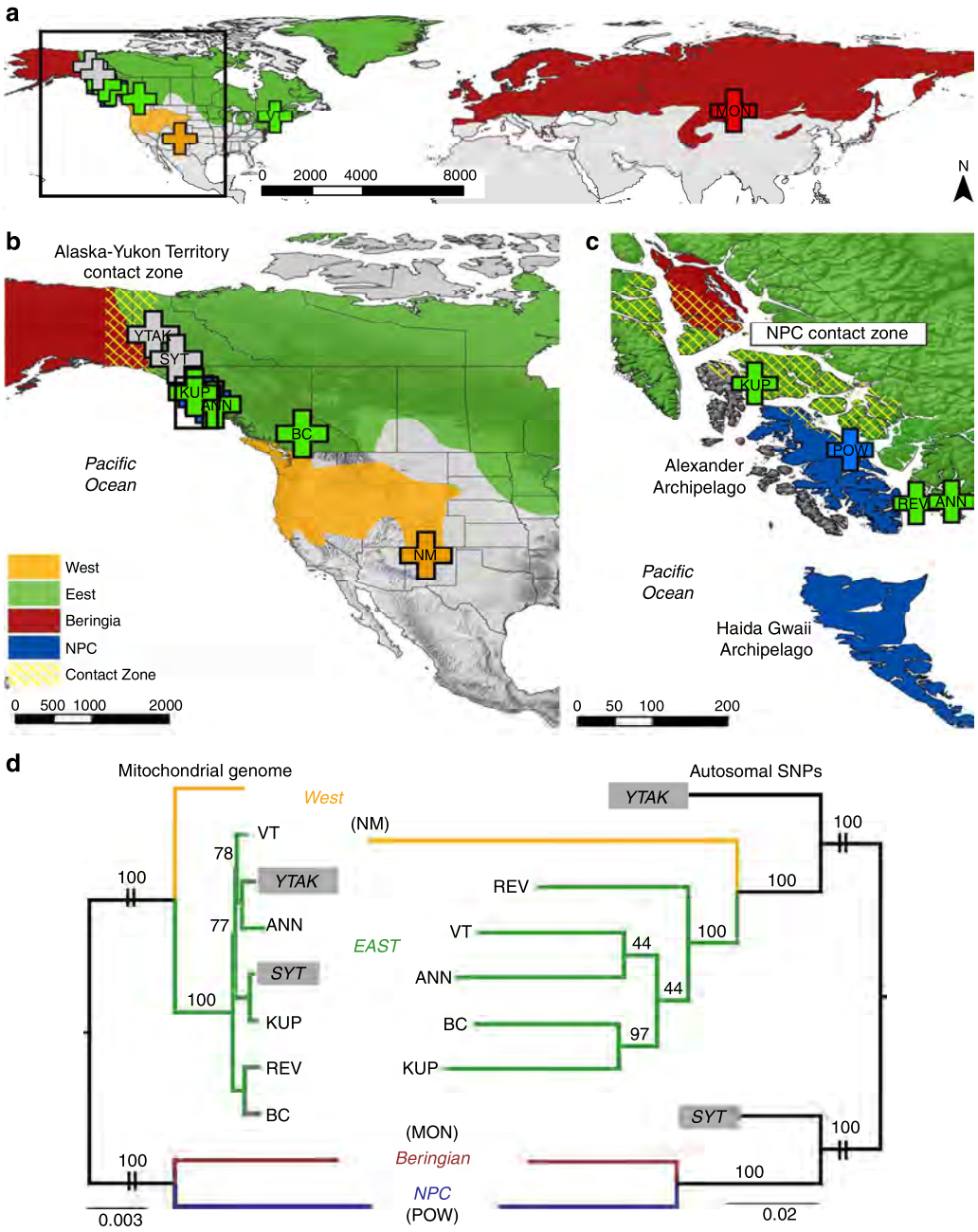

Fig. 1 *Mustela erminea* clade geographic distributions and phylogenetic relationships. **a–c** Stoat clade distributions (clade names reflect refugial origins) based on amplicon mitochondrial haplotype distributions[12] and IUCN (International Union for Conservation of Nature, www.iucn.org) range information. Genomic samples are labeled with locality abbreviations: ANN (Annette Island, Alaska, USA), BC (southern British Columbia, Canada), KUP (Kupreanof Island, Alaska, USA), MON (Mongolia), NM (New Mexico, USA), POW (Prince of Wales Island, Alaska, USA), REV (Revillagigedo Island, Alaska, USA), SYT (Southern Yukon Territory, Canada), YTAK (northern Yukon-Alaska border), VT (Vermont, USA). Contact zones are denoted with yellow hatching, based on the presence of mixed mitochondrial haplotypes[12]. Scale bars are in kilometers. **a** Global sampling scheme. **b** North American sampling localities. **c** Sampling within the northern North Pacific Coast (NPC) archipelagos. **d** Mid-point rooted maximum-likelihood phylogenies, scaled by genetic distance. **d** (left) Complete mitochondrial genomes and **d** (right) autosomal SNPs for ten *M. erminea* individuals. Phylogenies demonstrate strong support for four major refugial clades and mitonuclear discordance for hybrid samples (SYT, YTAK). Deep divergence between the NPC Island (POW) and Beringia (MON) clades is evident in the long branches in both phylogenies

introgression or incomplete lineage sorting. ADMIXTURE identified the same two individuals as admixed between East (K3: SYT 66%, YTAK 74%) and Beringia (K3: SYT 34%, YTAK 19%) clades or as a single, but distinct population, consistent with shared source populations (Fig. 2c). Both Alaska-Yukon samples were positioned between their hypothesized source populations in our PCA results (Fig. 2a and b), with SYT closer to Beringia and YTAK closer to the East clade, further supporting their mixed ancestry and suggested asymmetric backcrossing distributions. Lastly, *f2* statistics, a measure of distance between populations, were also lowest between the putative Alaska-Yukon hybrids and their suspected source populations, further supporting their derivation from the same source populations (Supplementary Table 4).

Table 1 Source populations and admixture date estimates for each hybrid sample

Target	Source1	Source2	*f3*-statistics			MixMapper date estimates		
			f3	Err	*Z*	α	Mixed drift	Years ($N_e = 375k$)
NPC	East	Beringian	−0.05	0.009	−5.28	0.48–0.59	0.00–0.00	0
SYT	East	Beringian	−0.17	0.003	−48.66	0.52–0.78	0.00–0.00	0
YTAK	East	Beringian	−0.15	0.003	−47.41	0.48–0.73	0.00–0.23	0–393,996.5

Admixture date estimates in years based on an effective population size (N_e) of 375k (ref. [12]) for the North Pacific Coast island (NPC) and the two Alaska-Yukon hybrids (SYT from Southern Yukon and YTAK from the northern Alaska-Yukon border). Err indicates the standard error and *Z* indicates the *Z*-score

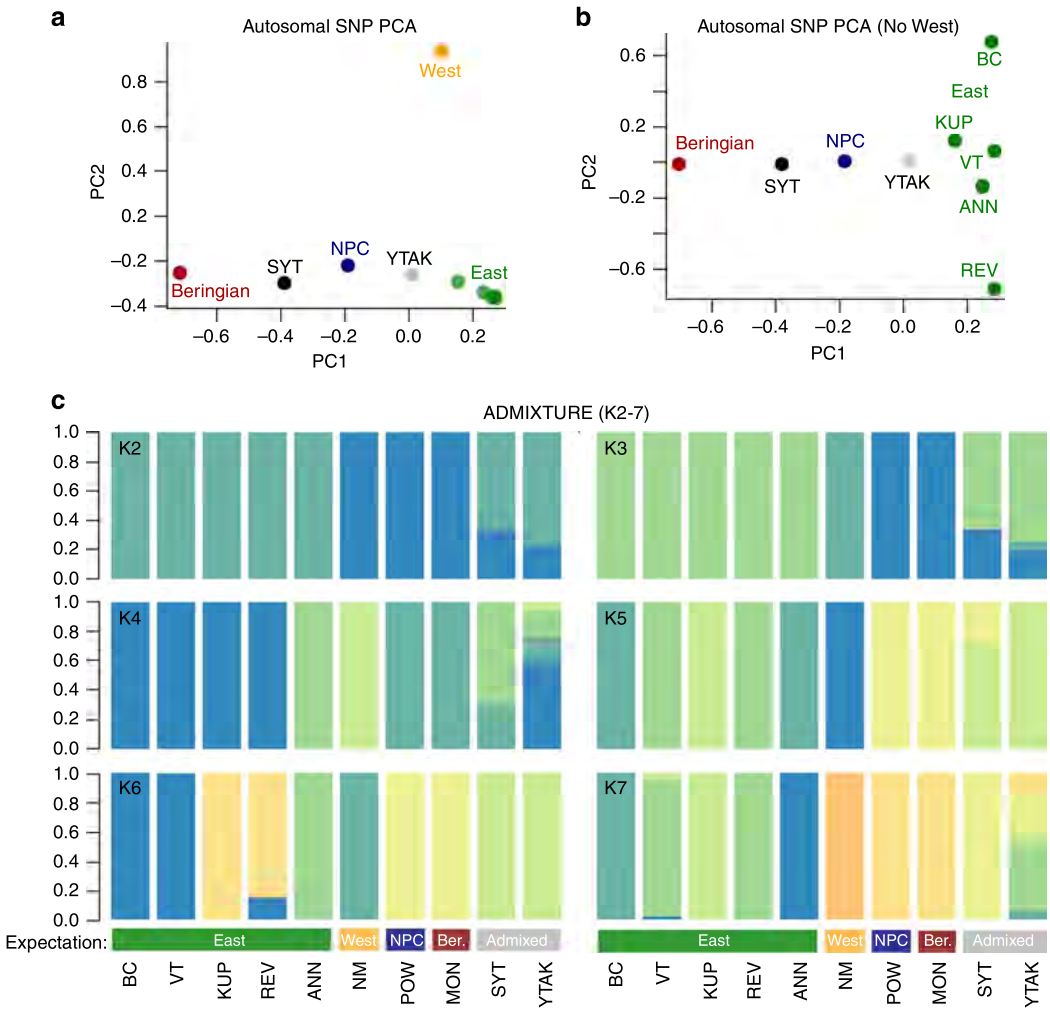

Fig. 2 Principal component analysis and ADMIXTURE results. **a**, **b** SNP principal component analysis (PCA). All samples identified as admixed by *f3*-statistics (NPC, SYT, YTAK) lie intermediate to Beringia and East clades, their purported source populations. **a** PCA with all samples demonstrates substantial nuclear genetic differentiation of the West clade. PC1 explains 20.7% of the total variation, and PC2 explains 14.1%. **b** PCA with the West population removed demonstrates additional structure within the East clade, centered on Vermont (VT). Hybrid samples are intermediate to their source populations. PC1 explains 29.8% of the total variation and PC2 11.8%. **c** ADMIXTURE plots for K2–7. K = 3 best matches our expectation based on the geographic distributions of refugial clades and is consistent with admixed ancestry for SYT and YTAK samples: (1) East samples as a distinct group, (2) a West population, and (3) a combined Beringia-NPC population with two hybrids admixed from these groups. K = 6 has the lowest cross-validation score for our data and shows an identical pattern to K3 but with additional substructure within the East clade, indicative of microrefugial substructure near Appalachia and consistent with PCA results suggesting East expansion from a single refugium

Surprisingly, NPC Island stoats also tested positive for admixture, with Beringia-East parentage (Table 1; Fig. 2a and b). These island stoats have been isolated off the NPC (Prince of Wales Island and the neighboring Haida Gwaii archipelago of British Columbia, Canada [not sequenced here]) for at least 10,000 years[12]; therefore, we suspect they are derived from an ancient hybridization event that occurred during an earlier interglacial period, prior to the contemporary insularization of this clade. We found no evidence of admixture, however, in the Kupreanof Island sample.

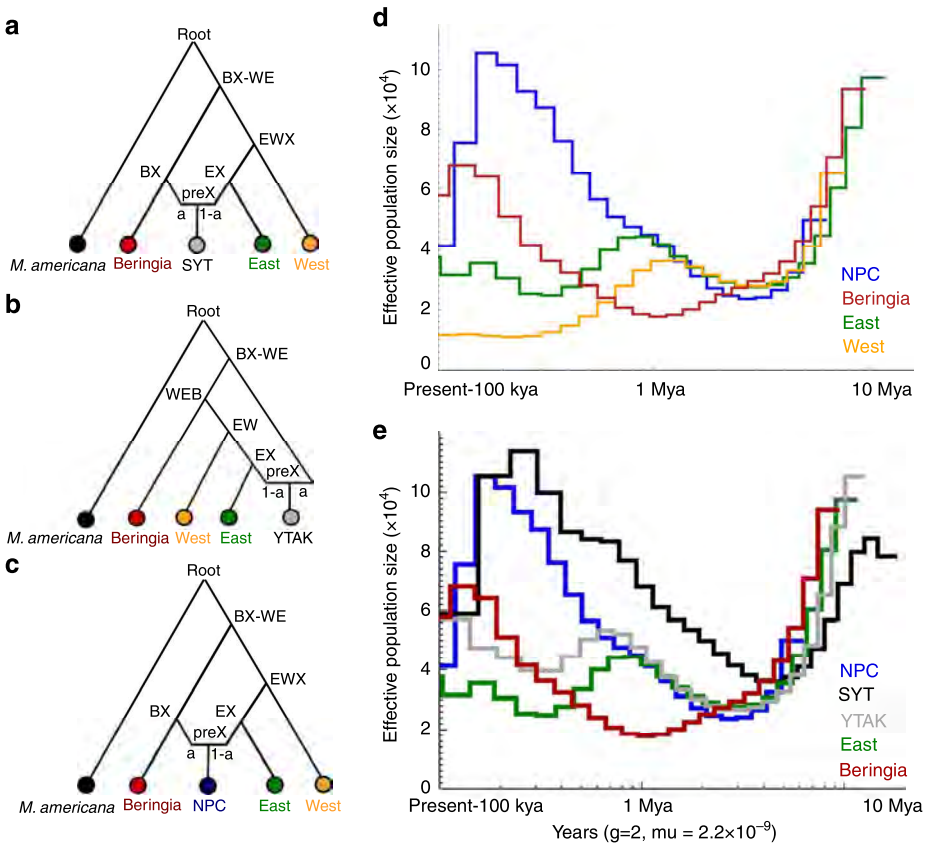

Fig. 3 Phylogenetic placement of admixed samples and historical demography. **a–c** Optimal phylogenetic placement of hybrid samples onto our non-admixed RAxML topology (see also Supplementary Fig. 5) based on *f4*-statistics fitting in Admixture Graph. **a** SYT originates from ancestors of Beringia and East populations; **b** YTAK is admixed from East and an unsampled ancestor of all *M. erminea* lineages; **c** NPC stems from ancestors of Beringia and East clades. (**d** distributions of effective population sizes (N_e) through time from pairwise sequential Markovian coalescent (PSMC) analysis scaled by a general mammalian mutation rate (2.2×10^{-9}) and a 2-year generation time. **d** PSMC plots for a representative from each refugial population (East, West, Beringia, NPC). **e** PSMC plots for hybrid samples (YTAK, SYT, NPC) and their source populations (Beringia, East). Hybrid distributions mimic that of their most-backcrossed source (f_4) with artefactual N_e inflation, as expected for admixed demographic histories

Testing for asymmetric introgression. Biased or asymmetric introgression may indicate a selective advantage, the existence of genetic incompatibilities, or a demographic imbalance. Alternatively, equal contributions from both parental source populations may suggest that there are minimal genetic incompatibilities between hybridizing stoat clades, as we would expect within a single species. For the NPC Island clade, biased allelic overlap with one source population may shed light on interglacial colonization dynamics, as the island may have been colonized by both Beringia and East ancestors who subsequently interbred. We used *F4*-statistics, similar to *D*-statistics[43,44] and ABBA/BABA[45], to highlight differential backcrossing histories among lineages and used admixture graph fitting[46] to estimate the topological placement of admixed samples relative to their potential sources.

First, our *f4* results highlight the influence of repeated glaciation in North America as reflected in multiple episodes of gene flow between lineages originating in independent refugia. Allele sharing (*f4*) reflected phylogenetic relationships, with stepwise genetic distance from the ancestral Beringia clade increasing with greater geographic distance from Beringia (Fig. 1; Supplementary Tables 5 and 6), a pattern consistent with more frequent or prolonged contact between geographically proximal refugial lineages.

Admixture graph fitting identified SYT as the result of Beringia and East admixture, while YTAK was produced from an ancestral admixture event between East and a shared ancestor of all stoat lineages (Fig. 3a and b; Supplementary Table 7). The vast majority of *f4* comparisons (Supplementary Tables 5 and 6) were highly significant (*Z*-score >5), reflecting substantial intra-specific divergence within *M. erminea*. Biased allelic overlap suggested that YTAK shares more alleles with the East source, whereas SYT is more backcrossed with the Beringia source (Table 2). Differentially backcrossed hybrid offspring within this broad, mainland contact zone suggests that hybridization between East

Table 2 *f4*-statistics for hybrid samples (*X*) with *M. putorius* outgroup (*W*)

W	*X*	*Y*	*Z*	*f4*	*Z*-scores
OG	NPC	East	Beringia	0.30	31.79
OG	NPC	West	East	0.32	76.99
OG	NPC	West	Beringia	0.51	70.42
OG	SYT	East	Beringia	0.13	9.29
OG	SYT	West	East	0.37	86.48
OG	SYT	West	Beringia	0.41	42.17
OG	YTAK	Beringia	East	0.28	27.12
OG	YTAK	West	East	0.40	90.67
OG	YTAK	West	Beringia	0.08	10.18

F4(W,X:Y,Z) results for each hybrid sample (*NPC* North Pacific Coast island, *SYT* Southern Alaska-Yukon border, and *YTAK* northern Alaska-Yukon border) relative to all possible parental source populations (East, West, Beringia)
All *Z*-scores are positive for ease of interpretation and significant (>5). Rotating the order of a pair (e.g., YZ to ZY) switches the sign +/− of both the *f4*-statistic and the *Z*-score but does not change the relationships

and Beringia lineages may have minimal negative fitness consequences (e.g., reduced fertility, viability)[47]. These preliminary inferences require additional sequencing of hybrids from the interior Alaska-Yukon hybrid zone to characterize the specific genomic regions that may be crossing semi-permeable lineage boundaries and to rigorously test for asymmetric introgression, which could be driven by selection (e.g, genetic incompatibilities, Darwin's corollary to Haldane's rule[48]), drift, or demographic parameters. If gene flow is neutral, extended periods of contact may promote fusion between East and Beringian lineages. However, the contemporary persistence of both Old World Beringian and New World East lineages in North America suggests that gene flow during brief periods of interglacial contact through the Pleistocene were insufficient to homogenize refugial divergence.

Biased allele sharing between NPC and Beringia (Table 2) is consistent with phylogenetic relationships (Fig. 1d) and three possible evolutionary histories: (1) differential introgression (selection) favoring Beringia alleles early in the NPC Island clade's origin; (2) extended or more frequent connection between Beringia and NPC refugia throughout the Pleistocene; or (3) stochasticity, potentially compounded by founder effects or bottlenecks.

Timing introgressive events. Given the pattern of North American glaciation[49], the current geographic isolation of the NPC Island clade, and the limited dispersal abilities of insular *M. erminea*[50], we suspect that hybridization is not ongoing on NPC islands. In contrast, based on the current distribution of mitochondrial diversity[12] and our understanding of postglacial population expansion, we expect that stoats from East and Beringia clades are actively interbreeding along the Alaska-Yukon border. To approximate the timing of admixture events, we converted drift unit branch lengths (D), output from MixMapper[51], to absolute time (years) using the formula $D \approx 1-e^{-t/2Ne}$ (solved for t generations)[52].

The estimated time of admixture for Alaska-Yukon hybrids did not differ from zero (Table 1), suggesting that the Alaska-Yukon Territory border is an active hybrid zone between the leading-edge of Beringia and East stoat clades. In contrast, NPC admixture is estimated at up to 394 kya (Table 1; Supplementary Table 8), prior to the Wisconsin glacial (30–80kya[53,54]) and Sangamonian interglacial (80–140kya[53,54]). The weaker NPC hybrid signal relative to contemporary Alaska-Yukon hybrids (*f3* *Z*-score), the insular isolation of this clade for at least the past 10,000 years[12,55], and the short generation time of stoats (2 years[56]) suggest that NPC Island admixture is more ancient than that occurring in the mainland Alaska-Yukon zone. We hypothesize that the hybrid origin of the NPC clade predated a period of glaciation and refugial isolation, but this point requires further exploration with expanded sampling and tests of linkage disequilibrium. If NPC stoats continue to evolve in isolation, our results suggest that continued divergence in allopatry may ultimately lead to speciation.

Based on our results, NPC islands were colonized either by admixed individuals or independent invasions of both parental lineages (East, Beringia) that subsequently hybridized. Introgressive hybridization, preceding divergence in isolation, may have disproportionately elevated the evolutionary distance between NPC Island stoats and other lineages (Figs. 1, 2, and 3d, e; Supplementary Tables 1–4). Isolation of a genetically recombinant population, at least through the Last Glacial Maximum, in a coastal refugium reinforced an evolutionary trajectory independent from either source population (Supplementary Fig. 3). Further, low amplicon variation in NPC stoats[12] suggests that island founder effects or a bottleneck[5] occurred among NPC stoats in the coastal refugium[57]. Drift may impact small populations and may have further accelerated NPC stoat differentiation.

Historical demography. We expected glacial cycling (periods of isolation/contact) to have impacted the historical demography of each refugial stoat clade differentially. Further, we hypothesized that individuals with a history of introgression will exhibit elevated effective population sizes (N_e), with a distribution most similar to that of their most-backcrossed source population (East for YTAK, and Beringia for SYT). Last, we expected cladogenesis to coincide with refugial isolation during glacial periods and demographic troughs, while N_e is expected to rise during interglacials as a consequence of population expansion.

Stoat demographic histories (PSMC[58]; Fig. 3d, e) showed clade-level responses to cyclic glaciation, highlighting the central role of large-scale climatic shifts in shaping diversity at high latitudes through episodic isolation and contact. Overall low, but relatively stable N_e in East and West clades reflected similar demographic responses to New World glacial cycling, where the West refugium was likely smaller than the East, as reflected in smaller N_e (Fig. 3d). Unlike Nearctic populations, Western Beringia experienced substantial expansion historically, corresponding to reduced glaciation in eastern Eurasia and the larger size of Beringia compared to the NPC refugium. Dates of cladogenesis mirrored estimates from the calibrated mitochondrial genome phylogeny (Supplementary Fig. 1), with inter-clade divergence beginning around 2 Mya when Beringia split from the other stoat lineages, followed by West (1.3 Mya) divergence, and most recently Island/East (0.9 Mya) cladogenesis (Fig. 3d). However, the hybrid ancestry of the NPC island sample suggest that cladogenesis estimates based on N_e may be unreliable. As expected, when hybrid distributions are juxtaposed against their parental populations (Fig. 3e), they mirror that of their most-backcrossed source population, but with an upward shifted N_e[59]. If the increase in hybrid N_e occurs at the time of hybridization between parental populations[59], SYT and YTAK admixture may be older than predicted based on admixture date estimates alone.

Combined with diversity metrics (Supplementary Fig. 2; Supplementary Tables 1–4), highly distinct demographic histories suggest taxonomic revision of *M. erminea* may be warranted, pending wider geographic sampling. This is a particularly compelling example of cryptic diversification in carnivores when contrasted against weakly differentiated PSMC demographic histories uncovered in some canid[60] (wolf, dingo, domestic dog) and felid[61] (tiger, leopard, cheetah) genomes.

Discussion

We document multiple temporally and spatially offset bouts of introgressive contact between divergent stoat lineages, corresponding to distinct phases of glacial cycling. Therefore, we predict that other higher latitude species with shared biogeographic histories may also exhibit genomic signatures of repeated introgression corresponding to cyclic climates. This process of recurrent introgressive hybridization between geographically proximal lineages, followed by refugial or insular isolation and divergence, may contribute to elevated levels of island endemism, particularly for continental archipelagos adjacent to glacially dynamic mainland areas.

When backcrossing occurs into both parental populations, as in the Alaska-Yukon hybrid zone, introgression may be stochastic or selectively neutral. Consequently, extended periods of contact may lead to homogenization[18,62] and result in a net loss of regional diversity[63], such that the divergent lineages evident today

(East, Beringia) may be ephemeral[64]. The narrow Alaska-Yukon hybrid zone and strong East-Beringia genomic differentiation, however, suggests that such peripheral hybrid populations are ephemeral and minimally contribute to long-term evolution of high latitude diversity. If introgression events are temporary, genomic evidence of serial hybridization offers a model for detecting refugia and tracking colonization routes among high latitude species. Here, we provide yet another example of introgressive hybridization in mammals, highlighting both the prevalence and varied role of introgression in mammalian evolution. Our results demonstrate that, for geographically widespread species, the long-term evolutionary role of introgression often may be inconsequential relative to larger biogeographical processes, such as island vicariance, refugial isolation, or rapid environmental change.

In some situations, however, a population bottleneck in a small refugium or in response to insularization can enable the temporary persistence of less fit recombinant genotypes[65]. Even a temporary reduction in fitness can enable traversal between fitness-landscape optima and promulgate alternative evolutionary trajectories, which can permanently impact the genomic structure of high latitude populations and potentially lead to speciation[65,66]. Genomic evidence of hybridization, followed by divergence in geographic isolation, is consistent with a hybrid origin for the NPC lineage. Though still incipient in this case, the process of introgression followed by allopatry may lead to hybrid speciation[67]. Hybridization can catalyze the evolution of reproductive barriers[68] or expedite divergence through the rapid generation of novel genetic recombinants. Although the mechanisms underlying this special case of divergence with gene flow remain elusive, a plausible precondition for speciation is the separation of lineages into allopatric ranges, which may enhance the genetic divergence of a recombinant population, even if introgression is not directly implicated in speciation. Therefore, NPC stoats potentially highlight the integral role of post-hybridization allopatric divergence in the evolutionary histories of island and refugial taxa, but expanded insular sampling is required to fully test this mechanism of divergence.

It has long been argued that pre-zygotic isolation is reinforced when diverging groups return to parapatry and hybridize[18,20], although this idea is not without controversy. Under this scenario, hybridization between incipient species can result in unfit hybrids and natural selection might act against hybridization, reinforcing pre-zygotic isolation mechanisms, eventually leading to speciation and avoiding maladaptive hybridization[18,69]. In this fashion, continuing and progressive reinforcement of pre-zygotic mechanisms would ultimately result in speciation and complete isolation between lineages[18,70], if hybridization and selective pressures on both lineages are symmetrical[71]. With an active and plausibly ongoing hybrid zone, along with more ancient island admixture, stoats provide a unique opportunity to investigate the impact of symmetrical versus asymmetrical introgression using genomics.

With multiple waves of mainland colonizers corresponding to climatic cycles and potentially adding novel genetic elements to insular genomic architectures, the process of repeated, climate-mediated hybridization may have played a role in diversification across other land-bridge island systems. Irish stoats, for example, exhibit elevated genetic diversity compared to neighboring populations in Great Britain, despite persisting on a smaller island farther from a continental source[13,72]. In this parallel system, the English Channel separating Britain from the European mainland has been serially exposed through Pleistocene glacial cycles[73], effectively homogenizing populations across the Channel. On the other hand, the Irish Sea, nearly twice the depth of the English Channel, further separates Ireland from Britain, presenting a formidable barrier to gene flow that allowed only occasional influx of novel alleles from mainland or British immigrants during extreme glacial events. This mechanism of post-hybridization allopatric divergence is not restricted to land bridge or continental archipelagos. Oceanic island populations of Darwin's storied finches on the Galapagos Islands also were impacted by climate-mediated pulsed hybridization that was coincident with changes in sea level due to glaciation[74].

Admixture in this insular endemic (NPC Island stoats, subspecies: *M. e. haidarum*) poses a particular challenge for the U.S. Endangered Species Act, which currently denies protection for hybrid taxa[30,75]. *Mustela erminea haidarum* is federally protected in Canada on the Haida Gwaii Islands of British Columbia[56], but also occurs on Prince of Wales Island, Alaska, where it has no protection. This study is congruent with previous work noting that this island-restricted stoat subspecies is genetically[12,76] and morphologically[77] distinctive, but levels of management protection at the federal level differ profoundly between Canada and the United States.

Genomic signatures of repeated admixture and refugial isolation among North American stoats highlight the episodic role of Late Quaternary climate cycles in generating and structuring high latitude diversity. We demonstrate two instances of gene flow between stoat lineages coincident with interglacial periods: one contemporaneous along the interior Alaska-Yukon border and a second, more ancient event, occurring along the NPC that left a signature that persisted through multiple glacial cycles. Each hybridization event exemplifies an antagonistic outcome of hybridization (e.g., homogenization versus diversity generation) within a single species complex, highlighting the multifaceted role of introgression in mammalian evolution and the varied temporal and geographic scales at which this process unfolds. At high latitudes, where dramatic range shifts in response to glacial cycling afforded recurring opportunities for contact and isolation, ephemeral hybridization has repeatedly shaped biological diversity[2,78,79].

Methods

Sequencing, assembly and post-processing. Contact zones and populations were defined based on expanded mitochondrial sampling[12]. Ten whole-genome sequences for representatives from each refugial stoat clade, and with additional sampling in suspected contact zones (Fig. 1), were generated on an Illumina HiSeq 2000 (paired-end reads [100 bp]), three samples in four Illumina lanes and the remaining samples in one lane each (Supplementary Table 9). Liver subsamples were obtained post-mortem from the University of New Mexico's Museum of Southwestern Biology in compliance with ethical regulations (IACUC #16-200526-MC). We used a DNeasy Blood and Tissue Kit (QIAGEN, USA) extraction. Next-generation sequencing libraries were prepared using the Illumina TruSeq DNA Sample Prep Kit. Our genome assembly pipeline followed Lan et al.[79] with reads examined using FastQC[80] and adapter sequences and sex chromosomes removed (Trimmomatic v0.33 (ref. [81])). Quality scales were compared across samples and rescaled to Illumina 1.8 + phred + 33 quality score, when necessary, to maintain consistency (Picard v1.9: http://picard.sourceforge.net; SAMTools[82]). Reads were mapped to the domestic ferret genome[83] (*Mustela putorius furo*) using the Burrows-Wheeler aligner[84] (BWA). An additional BWA iteration extracted mitochondrial genomes. Final depth of coverage ranged from 10 to 61× (Supplementary Table 9). PCR duplicates were removed using the MarkDuplicates tools from the Picard suite. Nuclear and mitochondrial genome consensus sequences were called using mpileup in SAMtools. SNPs were called using HaplotypeCaller in the Genomic Analysis Toolkit[85] (GATK) for all stoats and again against each of two outgroups (*M. putorius* and *Martes americana*) and filtered by minimum depth (minDP = 2), genotype quality (minGQ = 30), minimum minor allele frequency (MAF = 0.1), and scaffold size and location (1 Mb), and then private alleles and indels were removed using VCFtools[86] (Supplementary Table 10). We intentionally selected an MAF of 0.1 to remove singletons (e.g., individual-specific, rare mutations), and thereby any potential sequencing errors, which are more common in low-coverage genomes. Although singletons can be informative when analyzing within population variation, our analyses focus on shared alleles (e.g., *f*-statistics) to identify introgression and backcrossing histories, therefore singletons, since they are not shared, do not provide information about allelic overlap among populations. Our minDP (2) was set as one-third of the coverage of our lowest coverage samples (e.g., roughly one-third of 8×) as recommended by the PSMC manual.

Also, 2 is the minimum requirement for detecting heterozygotes. Format conversions (vcf, ped, bed) were conducted using PLINK[87]. Missing data were removed (--max-missing) based on analysis specifications. SNPs were spaced (1 per 50 bp window) to account for linkage disequilibrium (e.g., 25) and sorted into 46 'pseudo-chromosomes' to enable the application of human-specific analyses on a non-model system with 40 chromosomes.

Phylogenetic methods. jModeltest[88,89] estimated the most appropriate model of evolution for mitochondrial genomes and autosomal SNPs, with and without outgroup sequences (*M. putorius* and *Martes americana*). We generated phylogenies for the entire mitochondrial genome (calibrated/uncalibrated) and autosomal nuclear SNPs (uncalibrated) using RAxML[90] (GTRCAT model, 10,000 generations, random starting seed) and Beast2[91] (1 million generations, 4 chains, 2 runs, sampling frequency of 1000). It should be noted that phylogeny inference using highly variable data (e.g., SNPs) can induce acquisition bias resulting in longer branch lengths[92], however, other divergence metrics (e.g., PCA, relatedness, F_{ST}) provide additional evidence of substantial divergence between refugial stoat clades. The oldest known *M. erminea* fossil[93] (1.8 My) was used as a log-normal fossil calibration (mean = 5.16, sd = 0.25, offset = 180) to estimate divergence times in Beast2 (Supplementary Fig. 1) following the TreeThinkers tutorial (http://treethinkers.org/tutorials/divergence-time-estimation-using-beast/). A log-normal distribution places the highest probability on ages that are somewhat older than the fossil date, with non-zero probability to infinity. Log-normal distributional priors were selected to place 1.8 Mya at the median and shape priors encompassed late Pliocene to early Pleistocene, similar to Harding and Smith[94]. Fossil-calibrated divergence estimates separate two pairs of stoat clades (Beringia-NPC and East–West) at 4.47 Mya (±1.21 My, Pliocene), with the Beringia-NPC split dated within the early Pleistocene at 2.1 Mya (±0.6 My), over 1 million years before East–West cladogenesis (1 Mya ± 0.32; Supplementary Fig. 1). Phylogenies were visualized in FigTree v1.2.2 (http://tree.bio.ed.ac.uk/software/figtree/).

Traditional phylogenetic approaches were contrasted against TreeMix ancestral graphs modeling historic gene flow (migration events) among populations[95]. To accommodate genetic exchange between populations of the same species, TreeMix uses genome-wide allele frequency data to infer historical relationships among populations to maximize the composite likelihood of the sample covariance matrix (W_{hat}), while modeling both cladogenesis and migration. TreeMix iterates over a standard bifurcating topology to identify a series of migration edges that most increase the likelihood score. TreeMix was run without default sample size corrections (-noss), because multiple populations were comprised of a single individual. We limited the number of migration events (-m) to three, to examine only the most influential events among the few groups examined. TreeMix simulations with *M. putorius* as the outgroup identified ancient geneflow between Beringia stoat ancestors and *M. putorius*. Simulations were re-run with *M. americana* as an outgroup to assess whether long-branch attraction between distantly diverged *M. erminea* and *M. americana* (>11Mya[96]) impacts migration inference in TreeMix, but results were consistent, regardless of the outgroup used. After TreeMix identified gene flow between *M. erminea* and *M. putorius* (Supplementary Fig. 4), we only show Admixture Graph results with *M. americana* as outgroup.

PCAs were generated in SNPrelate[97]. Diversity statistics (relatedness2, Supplementary Fig. 2; nucleotide diversity, Supplementary Table 11; F_{ST}, Supplementary Table 2) were calculated using VCFtools. For ADMIXTURE[33] analysis, sites with >80% missing data were removed and the lowest cross-validation (cv = 10- and 5-fold) score identified the most appropriate number of populations (K = 6) by iteratively leaving a sample out and reexamining the partitioning of genetic structure among the remaining samples.

Introgression analyses. *F*-statistics were run in AdmixTools[41] and MixMapper[51] using *M. americana* as an outgroup after TreeMix identified ancient gene flow between *M. erminea* and *M. putorius* (Supplementary Fig. 4). *f2* statistics [f_2(Pop$_1$, Pop$_2$)] quantify drift through summary statistics (e.g., allele frequencies, heterozygosity, covariance, and the probability of two lineages coalescing), with larger values indicating greater divergence[42] (Supplementary Table 4). *f2* statistics were generated from the compute_moment_stats and compute_most_additive_trees functions in MixMapper with 1000 bootstraps and SNP blocks of 50 (1 per 50 bp). *f3*-statistics explicitly test for admixture (AdmixTools, 3PopTest) and considered all permutations where Source$_1$, Source$_2$, and the Target samples came from different populations. Populations (e.g., clades) were defined by phylogenetic relationships (Fig. 1) and ADMIXTURE results (Fig. 2c). A positive *f3* value does not necessarily indicate the absence of admixture[42]. Small sample sizes and the absence of a linkage map for *M. erminea* prevents linkage disequilibrium-based estimates of N_e and refined dating of admixture events.

To parse the backcrossing history of each hybrid sample, *f4*-statistics were used. We used block-jackknifing to accommodate non-independence between loci[42]. Although *f*-statistics alone cannot deduce the direction of gene flow in a system, admixture graph fitting can test whether a proposed evolutionary model fits the data well[45,51]. Admixture Graph[46] fit hybrid individuals into a non-admixed tree topology (Supplementary Fig. 5a) based on *f4* results (Supplementary Tables 5 and 6). We ran an additional Admixture Graph simulation on the optimal (lowest minimal error) 4-taxa non-admixed backbone topology identified by Admixture Graph (Supplementary Fig. 5b; Supplementary Table 12) and again for Alaska-Yukon hybrids onto a 5-taxon backbone (Supplementary Fig. 5c) to evaluate how inferred relationships changed (Supplementary Fig. 6). We compared individuals from each population against members of alternative populations (e.g., f_4 (Outgroup, NPC; East, Beringia)). We also compared hybrid individuals (as identified by *f3*-statistics) against individuals from 'pure' populations (e.g., f_4(Outgroup, Hybrid; East, Beringia)) and admixed populations (e.g., f_4 (East, SYT; Beringia; YTAK)) to decipher the backcrossing histories of hybrid samples and characterize patterns of gene flow across populations. *f4*-statistics are negative (*Z*-score ≤−5) if there is more allelic overlap between X and Y than between X and Z since the evolutionary split between Y and Z, and positive (*Z*-score ≥5) if there has been more recent gene flow between X and Z than between X and Y. *f4* results are in Table 2, Supplementary Tables 5 and 6, and Supplementary Fig. 7. Dawson et al.'s[12] N_e Model 2 using multi-locus Approximate Bayesian Computation approximates mean clade N_e at: 250k for Beringia, 50k for NPC, 125k for East, and 75k for West; therefore, the combined N_e for NPC, East, and Beringia populations is 375k. *D* is tightly correlated with N_e and additional efforts to refine historical population sizes (Pleistocene, Pliocene) should improve the resolution of our admixture date estimates. Generational output was converted to years using a *M. erminea* generation time of 2 years[56].

Historical demography. PSMC[58] was used on consensus genomic sequence data (using both complete and down-sampled sequence data) to characterize historical demography (Fig. 3d, e) by examining heterozygosity densities in a 100 bp sliding windows across the genome. PSMC was run twice for each individual, once utilizing all mapped sequence data (coverage 10–61×) and again on data down-sampled to 10× coverage (matching the lowest coverage sample) using the DownsampleSam tool (Picard). Results were scaled by a general mammalian mutation rate (2.2×10^{-9} per base pair per year[98]) and stoat generation time, resulting in distributions of N_e through time (Fig. 3d, e). One hundred PSMC bootstrap replicates were performed and plotted for both full-coverage and down-sampled data to confirm consistent distributional shapes and enable comparison across individuals, as PSMC is sensitive to variation in coverage depth (ideal coverage 18× (ref. [99])).

Code availability. Our custom python spacing script is available on GitHub at https://github.com/jpcolella/Ermine_WGS_2018.

Data Availability. Genomic reads are available in NCBI's Sequence Read Archive, accession: SRP138989. Assembled FASTA files are available on Dryad (https://doi.org/10.5061/dryad.bv2g720).

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Author contributions

J.P.C., S.L.T., J.A.C., and C.L conceived the project. J.P.C. assembled the genomes, conducted all analyses, and wrote initial manuscript draft with scientific and editorial input from T.L., S.L.T., J.A.C., and C.L. T.L. designed the bioinformatics pipeline and provided computational support throughout the analysis process. S.C.S. conducted whole-genome sequencing at Penn State University's Huck Institutes of the Life Sciences. S.L.T. provided substantial conceptual and editorial support. J.A.C. collected and provided tissue samples from the field via the Museum of Southwestern Biology. J.A.C guided sampling efforts based on previous research and provided substantial conceptual support in manuscript preparation and revisions. C.L. led the team and provided access to computational resources.

Additional information

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Statement on DEIS Prince of Wales Landscape Level Analysis

11 June 2018

I, Joseph Cook, am providing these comments to the Draft Environmental Impact Statement for the Prince of Wales Landscape Level Analysis Project (the DEIS). The DEIS explains that the Forest Service (FS) is evaluating three alternatives for old-growth forest logging volumes (115, 235, and 75 million board-feet [MMBF]) from Prince of Wales Island, Alaska. According to FS agency staff, the 235 MMBF alternative would require logging virtually every operable stand of old-growth forest on Prince of Wales Island that is currently allowed by the Tongass Land Management Plan (TLMP).

I am a Professor of Biology at the University of New Mexico (and formerly of the University of Alaska and Idaho State University). Over the past 27 years, my postdoctoral associates (n=5), graduate students (n=19), including the two collaborators co-signed here. Jocelyn Colella, is currently a doctoral researcher investigating the genetics of high-latitude mammals relevant to wildlife management and Natalie Dawson is an Assistant Professor in the Ecosystem and Conservation Science Department at the University of Montana, where her research has focused on the conservation of endemic and island mammals. Together, we have studied the systematics, biogeographic histories, parasite communities, and ecology of the mammals on the Tongass National Forest. That research has resulted in more than 60 peer-reviewed publications related to the wildlife of the North Pacific Coast. Those papers have been heavily cited in the scientific literature (> 4000 citations). Importantly, the current proposed timber sales (e.g., Prince of Wales and Kuiu islands) directly target areas that our studies indicate are the most sensitive habitat areas with regard to endemic mammals (e.g., ermine, *Mustela erminea celenda*; northern flying squirrel, *Glaucomys sabrinus griseifrons*; Alexander Archipelago wolf, *Canis lupus ligoni*; Pacific marten, *Martes caurina*).

These comments focus on the DEIS's analysis of the impacts of the proposed old-growth forest logging on Prince of Wales Island. We are very concerned that the USDA Forest Service has not used the best available science to direct current and planned future activities in the Tongass National Forest over the past few decades, including the currently proposed Prince of Wales Project. In light of these concerns, we recommend that the Forest Service not proceed with old growth timber sales as the limited old growth habitat that remains on Prince of Wales Island is critically important to the ecological integrity and evolutionary potential of endemic wildlife (and other organisms) found on the Tongass.

The Tongass National Forest is the largest forest managed by the USDA and it should be administered to sustain the wildlife, including wild mammal populations, for the long-term benefit of all Americans. Because this forest is distributed across a series of > 1,000 oceanic islands ranging in size from 213 km^2 (Yakobi Island) to 6,674 km^2 (Prince of Wales Island), the management of its native wildlife populations is complex and, as such, it should be managed

differently than most of the continental forests managed by the National Forest system. The Forest Service's current management, however, fails to account for this very basic understanding of scientific processes.

The best available science from islands systems across the globe has repeatedly demonstrated that island ecosystems are extremely vulnerable to human disturbance because they frequently support flora and fauna, or species that are found nowhere else on Earth (i.e., endemic to only these regions). In particular, the endemic flora and fauna on islands are highly susceptible to extirpation (local extinction) and ultimately extinction. Globally, more than 50% of all documented vertebrate extinctions in the last 400 years have occurred on islands. These are landforms that must be managed very carefully because mistakes have implications that cannot be reversed (e.g., extinction).

The issue of endemic conservation on the Tongass and the need to manage this forest using island biogeographic principles has been acknowledged by multiple independent scientific reviews of the Tongass Land Management Plan (TLMP) and in numerous publications (see references below), yet the USDA Forest Service continues to fail to adequately manage these public resources using approaches that rely on best available scientific information.

In the case of Prince of Wales Island, which has experienced some of the most intensive logging (and associated habitat conversion and fragmentation) over the past 60 years, the growing body of scientific literature shows that this island in particular also has a higher number of endemic organisms than most other islands in the Alexander Archipelago. This high rate of endemism likely reflects a long history of isolation (and subsequent divergence or differentiation) for the wildlife populations of Prince of Wales Island (Dawson et al. 2007), resulting in unique and diverse populations only present on this island. This high rate of endemism coincides with a long history of some of the highest rates of deforestation (clear cut logging of old growth forests) on the forest. Indeed, deforestation has been heaviest on the very island with the most documented mammalian endemics.

Additionally, our comments in this letter highlight several key findings in the best available scientific literature that the Forest Service must address in the Final Environmental Impact Statement for the Prince of Wales Landscape Level Analysis Project (the FEIS).

Project Level Surveys

In 1997, the Forest Service adopted a forest plan that required the Forest Service to conduct surveys of endemic mammals for projects that would significantly alter landscape composition on islands. Specifically, the agency was to "conduct surveys on larger islands if there is a high likelihood that endemic taxa are present that may be affected by the proposed project." Survey rigor was supposed to match the vulnerability of endemics and the level of existing and proposed forest fragmentation (i.e., more fragmented islands have more rigorous surveys prior to the occurrence of proposed habitat modification [i.e., logging or road building]). To date, these efforts have been inadequate based on our long association with the Tongass. In many cases, specimen-based inventories, followed by analyses of how distinctive (endemic) the mammals are, were never or minimally conducted within proposed timber sales.

These concerns were voiced to the USDA Forest Service in public fora (e.g., 2006 CSR meetings in Ketchikan), but ignored. Forest Service personnel have publicly acknowledged that surveys were not conducted and that they needed to focus additional attention on this problem. They did not. For more than 2 decades, we have stressed the need for rigorous, baseline inventories for all islands subject to proposed logging in the Tongass National Forest prior to the advancement of those activities, but the agency rejected all of those calls. For most old-growth associated organisms, the Tongass is attempting to manage with incomplete knowledge of identity, distributions, or populations status.

Like its predecessor, the 2016 Amended TLMP contains similar provisions governing endemic terrestrial mammals: "The objective is to maintain habitat to support viable populations and improve knowledge of habitat relationships of rare or endemic terrestrial mammals that may represent unique populations with restricted ranges." In assessing project-level effects (e.g., assessing the impact of the POW DEIS), the 2016 TLMP directs the agency to "[u]se existing information on the distribution of endemic mammals to assess project-level effects", but cautions that "[i]f existing information is lacking, surveys for endemic mammals may be necessary prior to any project that proposes to substantially alter vegetative cover."

Given the concerns outlined above, and the lack of historical surveying (and subsequent analyses), the Forest Service lacks sufficient information to assess project level effects of the proposed Prince of Wales Project. Further, those specimen-based inventories should focus not only on endemic mammals (which have been one of the primary groups to date), but also other sets of organisms, including amphibians, birds, plants, parasites, and arthropods to best inform managers of the current status of wildlife in these areas. We do not believe that the Tongass National Forest has followed its own requirements to conduct surveys for endemic mammals in the proposed Prince of Wales Island timber sales as mandated. As a result, the DEIS's analysis of the impacts on endemic terrestrial mammals (especially ermine, flying squirrels, and wolves) is unsubstantiated and inadequate.

Best Available Science

The 2008 Amended TLMP and 2016 Amended TLMP also ignored the best available science for wildlife on the Tongass National Forest. The best available science stems from a series of scientific investigations that illustrate the importance of island systems in generating and maintaining biological diversity. Endemism is a consequence of "island life" and the elevated presence of endemic organisms on the Tongass is consistent with biological theory.

Unfortunately, the Forest Service has failed to address management concerns related to logging of old-growth (high volume) timber forests for endemic organisms. Experts have voiced these concerns directly to the Forest Service (e.g., at the 2006 CSR meetings in Ketchikan) and in the published literature (Cook et al. 2006; MacDonald and Cook 2007), explaining that wildlife populations in this region are structured by the special (i.e., insular) characteristics found across the islands of the Tongass with each island harboring unique traits (i.e., genetic diversity, morphological distinctiveness, or adaptation to specific island conditions). To date, the only viability analysis the Forest Service ever conducted (as part of the 1997 TLMP planning effort) lumped all endemic species (rodents, carnivores, bats, etc.) into one category, rendering the analysis all but useless in understanding the habitat quality and quantity that must be retained to

ensure the viability of endemic species. Each endemic species has its own suite of environmental and ecological conditions under which it thrives. Therefore, the link between each of these independent entities and their associated habitat (which can differ dramatically across islands) should be assessed on a case-by-case basis. This analysis, of course, also fails to account for all of the new scientific discoveries that have developed since 1997.

The current Tongass Forest Service Standards and Guidelines have been poorly implemented for this archipelago and specifically for Prince of Wales Island. The Forest Plan (Appendix H, pages H-155-156) states that use of "biogeographic provinces takes into account island archipelago" issues and that at "fine filter" level, species-specific standards will be fully considered. We are biogeographers, but we do not fully understand this statement or how it has been applied by the FS. At a minimum, it is likely untrue that such an approach has been applied (and to-date, it certainly has been inadequate).

Indeed, the Forest Service recently acknowledged that it has "increasing" viability concerns for endemic populations: "There are roughly 24 mammal species or subspecies considered endemic to Southeast Alaska (Smith et al. 2005)…The long-term viability of these endemic populations is unknown, but of increasing concern since island endemics are extremely susceptible to extinction because of [their] restricted ranges, specific habitat requirements, and sensitivity to human activities such as species introductions (http://msb.unm.edu/isles/)."

The proposed old-growth logging on Prince of Wales Island is yet another example of this problem. The DEIS fails to account for the best available science regarding several important concepts that will need to be addressed in the EIS:

1. Individual islands often support distinctive forms (Darwin 1859), hence, consequences of Tongass commercial activities must be addressed on an individual island and species basis. This has not been fully addressed in the DEIS.
2. The Tongass is an island archipelago and, as such, it is highly susceptible to a host of impacts to wildlife that are amplified in island systems (Cook et al. 2006). In addition to loss of habitat and fragmentation, greater human access through additional road construction often means increased potential for invasion by exotic invasive species and pathogens. One only has to travel to Hawaii, where the introduction of invasive species has resulted in the permanent loss of many species, to recognize the problems inherent to wildlife management in island archipelagos. Human impacts, such as habitat conversion (e.g., logging), but also the introduction of exotic species (e.g., Courchamp et al. 2003), are the greatest drivers of global extinctions (Davies et al. 2006).
3. Consistent with island biogeography theory, island archipelagos have higher rates of extinction than nearby mainland areas (Frankham 1998, Mills et al. 2004). There is a large body of peer-reviewed scientific evidence from the Alexander Archipelago and elsewhere (e.g., Hawaiian archipelago) that recognizes extinction as a possible outcome of habitat manipulation (e.g., logging) and consideration of this should be a central tenet of responsible habitat management (i.e., planning timber sale projects) on islands in the Tongass.
4. The impacts of climate warming on island populations are predicted to be dire in some archipelagos, as movement by wildlife to better conditions is geographically limited in these systems.

5. Island archipelagos, like the Tongass, contribute unique elements (i.e., endemics) comprising the biological diversity on our planet (Myers 2000). As such, islands are key regions for sustaining the evolutionary processes related to diversification. Endemism, the evolutionary result of organisms being confined to islands over the long-term, is reflected in many species across the Hawaiian archipelago (i.e., Hawaiian honeycreepers, Hawaiian hoary bats, Hawaiian ducks). Similarly, Cook and MacDonald (2001) found that "the islands of the North Pacific Coast of North America…include about one-half of all known mammals endemic to North American islands north of Mexico". The Alexander Archipelago (most of the Tongass National Forest) contains a significant portion of endemic mammals for the entire continent and based on our information to date, Prince of Wales Island is a hotspot for endemics.
6. Related to that point, available science illustrates that many populations vary genetically and morphologically from island to island on the Alexander Archipelago, so the wildlife of individual islands can be distinctive (e.g., Lucid and Cook 2004, Hope et al. 2016). Therefore, a particular species' requirements and potential endemicity as well as individual island characteristics need to be accommodated in proposed logging of old-growth habitat.
7. Available science indicates that genetic information (DNA based studies) can reveal a population's distinctiveness (endemism) and susceptibility to extinction, as has been hypothesized for the Pacific martens (*Martes caurina*) on Kuiu Island and flying squirrels (*Glaucomys sabrinus*) on Prince of Wales Island, both of which show very low levels of genetic diversity. Low genetic diversity (i.e., variation) means these populations have fewer options to respond or adapt to change in the future. This is one reason island populations are so susceptible to changing conditions. But we still know little about most organisms on this archipelago.
8. Available science indicates that anthropogenic activities (such as logging) on specific islands can drastically change the composition of the biota found on those islands (e.g., Blondel 2008), hence, consequences of Tongass activities must be addressed on an individual island basis prior to proposed habitat modification.

Endemic Mammals on Prince of Wales Island (a few examples)

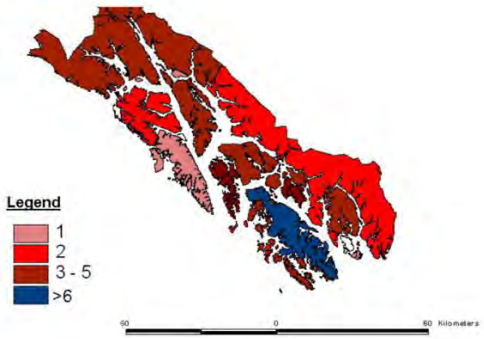

FIG 7. Map of endemic mammals across Alexander Archipelago (relative densities based on number of endemic lineages found on each island). High concentrations of endemics suggest that Prince of Wales Island is a hot spot of biodiversity.

(from Dawson et al. 2007)

Prince of Wales Ermine, *Mustela erminea celenda*
The DEIS also fails to account for the best available information regarding this endemic mammal. As a result, the Forest Service has failed to adequately assess the project-level effects of the proposed logging. Ongoing work on the Prince of Wales ermine (Dawson et al. 2014; Colella et al. 2018, Colella unpublished data) highlights the distinct evolutionary origin and unique genetic properties of this mammal that, to the best of our knowledge, is only found on Prince of Wales Island, and potentially a very few nearby islands (not yet fully assessed). Our genetic data suggest the Prince of Wales ermine, currently recognized as subspecies *Mustela erminea celenda*, is distinctive, but closely related to the subspecies *Mustela erminea haidarum*, which occurs on a few islands of the Haida Gwaii Archipelago (Queen Charlotte Islands), ~70 km south. The Haida Gwaii subspecies is currently listed under the Canadian Federal Species at Risk Act (SARA) and COSEWIC (Committee on the Status of Endangered Wildlife in Canada; S2--Imperiled or Rare) and is subject to protections and prohibitions under the British Columbia Wildlife Act and all harvesting of the subspecies is prohibited.

The Forest Service lacks understanding of the distribution, habitat needs, or viability requirements of the Prince of Wales ermine and this knowledge gap has not been fully acknowledged in the POW DEIS. The Forest Service's "condition-based" analysis means the agency is refusing to explain when and where it will be logging and building roads. The lack of information regarding the spatial and temporal scale of the logging and road construction renders the analysis in the DEIS severely lacking. It is not possible to determine the direct, indirect, or cumulative impacts to wildlife habitat or connectivity that could result from the logging. Given the Forest Service's lack of knowledge regarding this species and its habitat requirements on Prince of Wales Island and the fact that the Prince of Wales Project encompasses 1.8 million acres of National Forest System land, the agency must conduct a population survey, as prescribed by the 2016 Amended TLMP, to assess the impacts of the logging on the Prince of Wales ermine. Given the intensity of the historical logging on Prince of Wales, these survey efforts should be extensive, as the "extent and rigor of surveys will be commensurate with the degree of existing and proposed forest fragmentation, and potential risk to endemic mammals that may be present".

Examples of Other Endemics Potentially Impacted by Proposed Logging

Prince of Wales Flying Squirrel (Glaucomys sabrinus griseifrons)
Our genetic assessment of this subspecies (Bidlack and Cook, 2001; 2002) showed that this subspecies is distinctive from other flying squirrel populations to the east. This old growth associated species likely will be heavily impacted by the proposed logging action.

Alexander Archipelago Wolf (Canis lupus ligoni)
The morphological distinctiveness of this wolf was recognized years ago when the coastal wolf populations were described as a distinctive subspecies. Subsequently, our research group (Weckworth et al. 2005, 2010, 2011, 2015) and others have shown that this wolf subspecies is genetically distinctive from other wolves on the continent and populations on Prince of Wales Island have dropped to perilously low numbers in recent years. Although wolves are thought to be resilient to population fluctuations, the low numbers seen on Prince of Wales recently suggest the potential for negative effects due to loss of genetic variability. Impacts to prey (deer) and

denning site loss from historic logging on Prince of Wales and increased access for legal and illegal trapping have been implicated in the decline of these wolves. Further logging will only exacerbate these problems for these endemic wolves. Our experience with another subspecies of wolf in New Mexico (*Canis lupus baileyi*) over the past 3 decades suggests that it can be very expensive and difficult to restore wolf populations to the wild.

The Forest Service's reliance on habitat thresholds for endemic mammals has no basis in the contemporary science of conservation biology of island endemics. The DEIS's suggestion that so long as 20-50% of the old-growth habitat remains, then the impacts on any given species are minor is unsubstantiated based on the available science. Critically, the distribution of these forests and their connectivity can have dramatic effects on the survivorship of a species. Particularly for mammals, some of which (e.g., wolf) have large home range territories, the geography of the proposed logging on POW is essential to evaluating the impact of the plan on native faunas.

In addition to the survey, the Forest Service should assess the impacts of the proposed logging "relative to the distinctiveness of the taxa, population status, degree of isolation, island size, and habitat associations relative to the proposed management activity". "Where distinct taxa are located," the Forest Service is directed to "design projects to provide for their long-term persistence on the island". The DEIS fails to explain and analyze any of these considerations with regard to Prince of Wales ermine (or other endemic terrestrial mammals).

In sum, the Prince of Wales Project, if approved, represents an unacceptable risk to the long-term persistence of endemic mammals on Prince of Wales Island. The agency's scant analysis also calls into the question the agency's ability to maintain sufficient old-growth habitat to ensure the viability of these endemic species.

Here, we have focused primarily on the impact of the proposed POW old-growth forest logging on mammals, but it should be noted that preliminary work on other organisms (e.g., Sikes and Stockbridge 2013) also reflects the insular nature of Prince of Wales Island. In very preliminary surveys, other nearby islands such as Sukkwan also harbor endemics (Greiman et al. 2013), as seen through the identification of brand new species. Importantly these discoveries are consistent with what we know about island biology and island conservation. Certainly, these discoveries suggest the need for a much closer look at the impact of logging on all endemics (as mandated by federal courts after the 1996 TLMP).

In conclusion, we hope the Tongass National Forest will perform a more rigorous assessment of the impacts of the proposed logging on the wildlife on Prince of Wales Island before it approves this timber sale project. Such an assessment should explicitly address the highly endemic nature of this island and more broadly the globally distinctive and ecologically important region.

Sincerely,

Dr. Joseph Cook
Museum of Southwestern Biology

Biology Department
University of New Mexico
Albuquerque, NM 87131

Jocelyn Colella

Jocelyn Colella, M.S., PhD Candidate
Museum of Southwestern Biology
University of New Mexico
Biology Department
University of New Mexico
Albuquerque, NM 87131

Dr. Natalie Dawson
Professor, Department of Ecosystem and Conservation Science
University of Montana
Missoula, MT

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Review

Conservation of highly fragmented systems: The north temperate Alexander Archipelago

Joseph A. Cook, Natalie G. Dawson, Stephen O. MacDonald*

Biology Department and Museum of Southwestern Biology, MSC03 2020, 1 University of New Mexico, Albuquerque, NM 87131, United States

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ABSTRACT

The Alexander Archipelago of Southeast Alaska encompasses over 2000 named islands that extend along 600 km of the North Pacific Coast. This review summarizes recent research on mammals of this largely unexplored region. Field inventories followed by preliminary molecular genetic analyses of selected mammals demonstrate substantial spatial structure consistent with both the dynamic geologic history of this boreal region and the highly insular landscape. New views of taxonomic diversity, biogeographic history, and contemporary population connectivity lay a framework for managing and conserving this complex biome. First, repeated Pleistocene glacial advances along the coast fragmented species, leaving clear genetic signatures and a strongly diversified fauna. Organisms recolonized the coast following deglaciation from multiple northern (Beringia), southern (West Coast and Continental) or North Pacific Coastal refugia. Several species are composed of multiple, genetically distinctive lineages (in some cases, incipient or new species) due to independent colonization histories from distinct, divergent source populations. Second, the insular landscape of the Alexander Archipelago has produced highly endemic populations. These centers of endemism should be thoughtfully managed as hotspots of lineage diversity. Until a better understanding of connectivity among these divergent populations is developed, each island should be considered an independent biological unit. Finally, industrial logging, mining, human encroachment, tourism, wildlife consumption, and invasive species should be stringently monitored and regulated with respect to impact on island endemics and ecosystems. A new conservation paradigm for the Tongass National Forest should be developed that is built around the recognition of the complexity of this incomparable island archipelago of the North Pacific. In particular, recognition of high diversity and endemism should be central to management plans.

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* *Corresponding author:* Tel.: +1 505 277 1358; fax: +1 505 277 1351.
E-mail address: cookjose@unm.edu (J.A. Cook).
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"No excursion that I know of may be made into any other American wilderness where so marvelous an abundance of noble, newborn scenery is so charmingly brought to view as on the trip through the Alexander Archipelago... so numerous are the islands that they seem to be sown broadcast." John Muir, 1915

1. Introduction

Investigations of island systems around the world, such as the Galapagos, have provided key insights to fundamental questions in evolution, ecology and conservation. Of the 50 top "hotspots for biodiversity" defined by Conservation International, 70% are on islands. Islands are experiencing some of the highest rates of change in the world, but many island systems do not have proper management strategies, and have fallen victim to problems related to anthropogenic disturbance. Those impacts have led to massive loss of organisms, species extinctions, and disruption of ecosystems (Vitousek et al., 1995; Quammen, 1996). Unfortunately, our baseline understanding of what, where, and why with regard to biotic diversity in many archipelagos remains rudimentary, crippling our ability to assess and respond to perturbations. Knowledge of the relatively few island systems that have been thoroughly investigated has provided key insights in evolution and ecology, including fine-tuning our ability to predict the consequences of perturbations elsewhere.

Intensive studies of island systems also demonstrate that insular faunas are typically distinctive (endemic) and that

Table 1 – Lineage diversity (reciprocally monophyletic clades) has been assessed for 15 Southeast Alaska mammals based on mtDNA and nucDNA variation

Species	mtDNA *n*	# mtDNA lineages SE Alaska	nucDNA *n*	# nucDNA Lineages SE Alaska	References
Black bear	74	2	289	2	Byun et al. (1997), Stone and Cook (2000), Peacock (2004)
Brown bear	317	2	261	1	Talbot and Shields (1996), Paetkau et al. (1998), Barnes et al. (2002)
Wolf	325	1 (E[a])	221	1	Weckworth et al. (2005); unpublished manuscript
Wolverine	159	1 (E)	–	–	Tomasik and Cook (2005)
American marten	462	1	301	1	Small et al. (2003), Stone and Cook (2002), Stone et al. (2002)
Coastal marten	218	2 (2E)	112	2 (2E)	Small et al. (2003), Stone and Cook (2000), Stone et al. (2002)
Ermine	210	3 (1E)	–	–	Fleming and Cook (2002)
Northern flying squirrel	118	2 (1E)	233	2	Demboski et al. (1998), Bidlack and Cook (2001), Bidlack and Cook (2002)
Keen's mouse	257	2	–	–	Lucid and Cook (2004)
Tundra vole	214	1	63	1	Galbreath and Cook (2004)
Long-tailed vole	111	2	–	–	Conroy and Cook (2000)
Southern red-backed vole	449	1	15	1	Runck (2001), Runck and Cook (2005)
Northern red-backed vole	38	1	8	1	Runck (2001), Cook et al. (2004)
Montane shrew	70	2	–	–	Demboski and Cook (2001)
Cinereus shrew	46	1	–	–	Demboski and Cook (2003)

Sample sizes are reported for each species for the two classes (mitochondrial and nuclear) of molecular studies. E represents a reciprocally monophyletic lineage.

endemics are highly susceptible to extirpation and eventually extinction. Insular endemics are prone to extinction (Reid and Miller, 1989; Burkey, 1995) because they are vulnerable to habitat loss and fragmentation, introduction of exotics, and over exploitation (harvesting). Worldwide, more than 60% of documented vertebrate extinctions have occurred on islands (Diamond, 1989; Olson, 1989; Case et al., 1992; Steadman, 1995).

Fragmentation characterizes the expansive archipelagos of the North Pacific coastal landscape of North America. These systems pose significant challenges for natural resource management because of their complexity, but the challenges have scarcely been articulated, and rarely incorporated into management plans. Recent investigations point to a complex set of factors that drive species richness (Conroy et al., 1999), distribution (MacDonald and Cook, 1996),

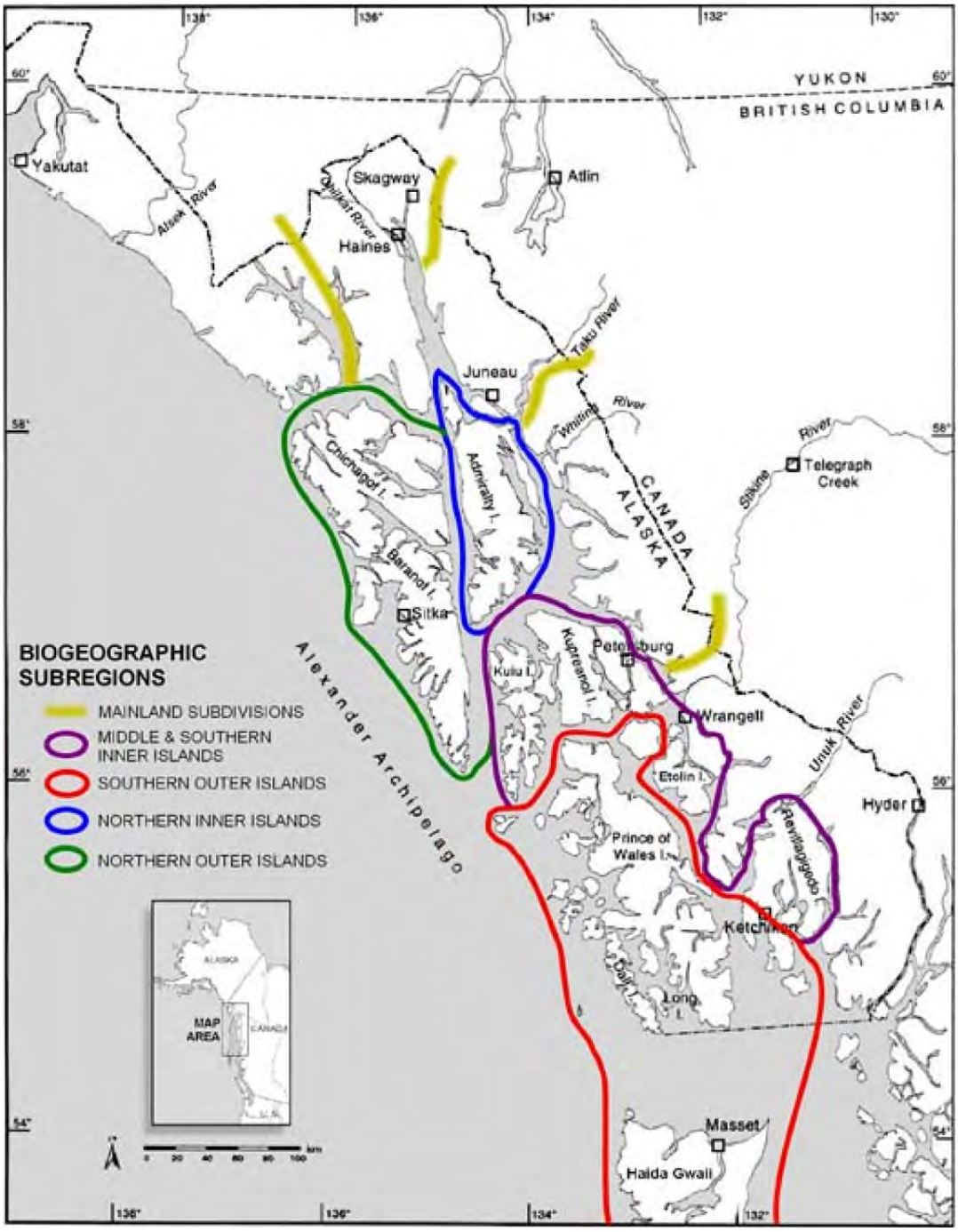

Fig. 1 – Southeast Alaska encompasses distinctive and isolated mammalian subregions along the North Pacific Coast of North America. Proposed biogeographic subregions are modified slightly from MacDonald and Cook (1996) on the basis of recent molecular phylogeographic studies that show close genetic relationships between some mammals on the Prince of Wales Island complex and the Queen Charlotte Islands (Bidlack and Cook, 2001; Fleming and Cook, 2002).

endemism (Cook and MacDonald, 2001; Cook et al., 2001), historical biogeography (Weckworth et al., 2005) and contemporary connectivity (Small et al., 2003) on these North Pacific islands. These factors will need to be addressed if these ecosystems, and associated high primary productivity and biomass (Alaback, 1995), are to remain intact and healthy.

Basic research efforts focused on mammals over the past 15 years have resulted in a series of publications (Table 1) that provide a preliminary view of diversity along the North Pacific Coast. We primarily restrict our focus to the coastal region along the panhandle of Alaska, a region that encompasses the Alexander Archipelago (Fig. 1); however, there are a number of parallels that can be drawn with other North Pacific archipelagos, especially the nearby Haida Gwaii islands of British Columbia. The Alexander Archipelago is one of the largest temperate archipelagos worldwide with more than 2000 named islands (including 4 of the 10 largest islands in the US). We call attention to unique challenges associated with managing biotic diversity on the archipelago and highlight the urgency of incorporating an "island" perspective into resource plans for the region.

Little attention has been paid to impending threats to the endemic fauna of archipelagos of the North Pacific Coast (Nagorsen, 2004), primarily because limited documentation has been available on the status of insular species (Cook and MacDonald, 2001). The few studies published on island organisms along the coast of nearby British Columbia (Nagorsen, 1994; Shank, 1999) have documented extinctions (*Rangifer tarandus dawsoni* last seen on Haida Gwaii in 1908) or endangerment (Vancouver Island marmot, *Marmota vancouverensis*, 20 adult individuals in 2000) of insular populations and species (COSEWIC, 2005). In general, however, management plans for public lands in British Columbia (Paquet et al., 2004) and Southeast Alaska (e.g., Tongass Land Management Plan, 1997) still have not sufficiently addressed the "island" issue.

In this review, we summarize the major points emerging from field inventories and molecular (phylogeographic) studies of a variety of mammals across the Alexander Archipelago. We then raise issues related to effective management of this north temperate island system in an effort to stimulate discussion (and action) related to current and future disturbances.

2. Mammalian research

2.1. Field inventories

Small mammals were sampled on over 100 islands and at numerous sites along the mainland of Southeast Alaska using standardized inventory methods (MacDonald and Cook, 1996, unpublished manuscript; Cook et al., 2005) each July and August between 1991 and 1999. These materials represent geographically extensive and site intensive collections. Each mammal was assigned a GPS locality, unique field identifier, and all tissues, parasites, and other subsamples were linked to the original voucher specimen (skeletal preparations, as whole bodied spirit-preserved or as dried study skins). Inventory crews preserved tissues (heart, liver, kidney, spleen, and lung), and embryos in liquid nitrogen. State and federal agency personnel and private individuals provided large series of specimens, especially material from larger game or furbearer species. Three species, the fisher (*Martes pennanti*), cougar (*Puma concolor*), and heather vole (*Phenacomys intermedius*), were documented as new to Alaska during that time (MacDonald and Cook, 1996; MacDonald et al., 2004). All materials were archived at the University of Alaska Museum of the North in Fairbanks and are available for further analyses (http://arctos.database.museum).

2.2. Phylogeographic analyses

Molecular genetic investigations of mammals were conducted on selected species in our laboratories or elsewhere. Those investigations focused on species that vary considerably in body size, life history traits, ecology, and evolutionary history, but used comparable sampling design and similar molecular markers (Table 1). A mitochondrial gene (usually cytochrome *b*) was sequenced for all. In several cases, nuclear genes were examined to test the validity of the mitochondrial perspective. Herein, we summarize the common themes related to conservation that have emerged across those analyses.

Morphological analyses provided the taxonomic framework that guided our molecular studies across the Alexander Archipelago. Subsequent molecular analyses refined our knowledge of how to manage the archipelago based on two key concepts in island conservation: endemism and connectivity. First, we discuss the historical assembly of the mammal fauna of the Alexander Archipelago; a history based on differential invasion of the region through multiple colonization corridors (Fig. 2) during the Quaternary and divergent evolution leading to endemism within the archipelago. Second, we address contemporary issues facing the biota of the archipelago based on this underlying dynamic history, and the heavy human imprint now evident throughout the system.

2.3. Deeper history: development of a regional fauna

Within North America, Southeast Alaska was recognized as a distinctive biogeographic unit over 100 years ago when Nelson (1887) coined the term "Sitkan District." Subsequent investigators (Swarth, 1911, 1936) followed this lead. These early explorers provided a baseline for understanding biotic diversity in the region, which is largely isolated from the remainder of continental North America by the coastal mountains. That physiographic barrier, transected by relatively few large rivers, apparently played a large role in structuring diversity.

The geologic history of the North Pacific Coast has been dynamic and principally dominated over the past two million years by large-scale climatic oscillations (Mann and Hamilton, 1995). During the Pleistocene, Southeast Alaska repeatedly was covered by glaciers (Mann and Hamilton, 1995; Carrara et al., 2003), and glacial dynamics shaped species assemblages (Klein, 1965; Heaton et al., 1996; Conroy et al., 1999). The massive Cordilleran Ice Sheet blanketed most of the region until about 12,000 years ago when ice retreated eastward into the Coast Range (Mann, 1986; Mann and

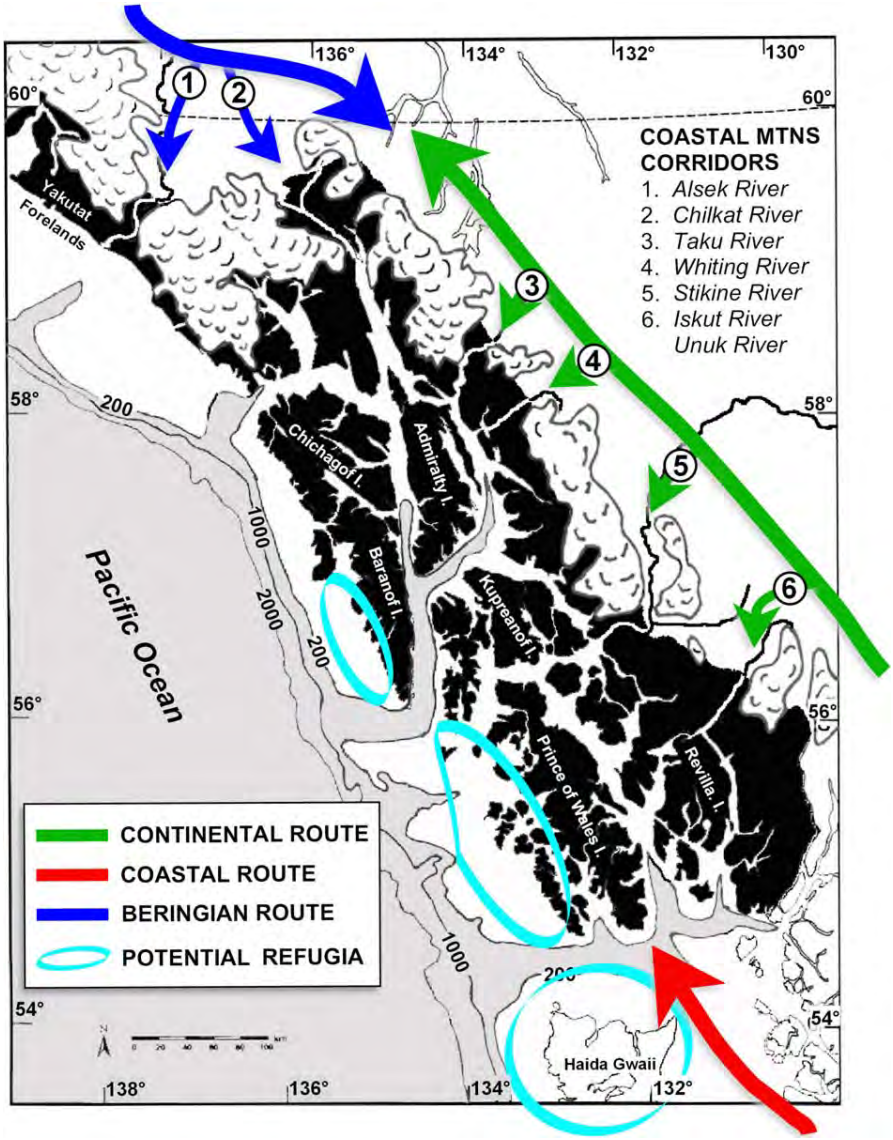

Fig. 2 – Map indicating the possible recolonization routes into Southeast Alaska following deglaciation of the Late Wisconsin advance (based on phylogeographic studies in Table 1). Locations of possible refugia are based on presence of paleoendemic lineages (e.g., Fleming and Cook, 2002) or glacial ice (Carrara et al., 2003).

Hamilton, 1995). Because this essentially created a " *tabula rasa*" situation, most extant species have recolonized the region during the Holocene. It may appear, then, that the Holocene sets the temporal stage for interpreting evolutionary change, endemism and the development of ecological communities within the terrestrial biota of Southeast Alaska. Molecular genetic studies and a growing fossil record suggest, however, that evolutionary changes also have accumulated and persisted over a series of glacial advances and retreats, thus producing a much deeper time frame for development of a divergent and highly endemic fauna.

Molecular studies of Southeast Alaska mammals generally fall into two broad categories (Cook et al., 2001) based on whether a particular "species" is represented in the region by a single deep (i.e., reciprocally monophyletic) lineage or by multiple deep lineages (Table 1). A single lineage plausibly indicates that this species colonized coastal Southeast Alaska from a single source, while species with multiple lineages originated from multiple locations.

2.3.1. *Single lineage species*

A suite of species (Table 1) apparently recolonized or expanded (Lessa et al., 2003) into the region from a single refugium, originating from the north (i.e., the Beringian Refugium with species such as wolverine *Gulo gulo*, northern red-backed vole *Myodes rutilus*, tundra vole *Microtus oeconomus*, arctic ground squirrel *Spermophilus parryii*), the south (i.e., southern coastal refugium; cinereus shrew *Sorex cinereus*, Keen's mouse *Peromyscus keeni*, wolf *Canis lupus*) or the east (i.e., southern continental refugium; northern flying squirrel *Glaucomys sabrinus*, moose *Alces alces*, southern red-backed vole *Myodes gapperi*).

2.3.2. Multiple lineage species

Other species are represented in the region by more than one lineage (Table 1). These presumably reflect multiple episodes of independent colonization that originated from multiple source populations or refugia. Those lineages, in some cases, are evolutionarily significant units (Moritz, 1994) or incipient species that likely began to diverge in distinct refugia located outside of this glaciated region before recolonizing. In several cases, the depth of genetic divergence between the lineages suggest that they began diverging in the mid-Pleistocene and have accumulated differences through repeated cycles of glacial advance (and presumably repeated recolonizations). Most of these species (marten *Martes americana*, montane shrew *Sorex monticolus*, long-tailed vole *Microtus longicaudus*, black bear *Ursus americanus*) are represented by lineages from southern coastal (termed Coastal) and southern continental (Continental) refugia. Following deglaciation, these "multiple lineage" species moved into the region along more than one route with individuals representing each of the distinctive lineages recolonizing Southeast Alaska independently. Brown bear (*Ursus arctos*) and ermine (*Mustela erminea*) also have multiple distinctive lineages, but they are special cases as each has a very old lineage (paleoendemic) that apparently is now restricted only to this region of the North Pacific Coast.

2.3.3. Endemics or tabula not so "rasula"

Brown bear and ermine are considered paleoendemics because these distinctive lineages likely arose in situ and have not spread, or at least persisted, elsewhere. The hypothesis that a North Pacific Coastal refugium (or series of refugia) existed has been supported in the last decade (Heaton et al., 1996; Fleming and Cook, 2002). A series of papers focused on the fauna and flora of the Haida Gwaii (Queen Charlotte Islands) also have supported the Coastal Refugium hypothesis (e.g., Byun et al., 1999; Burg et al., 2005; but see Demboski et al., 1999). The existence of refugia along the coast has significant implications for a "coastal route" of exchange of organisms (including humans) between Far East Asia and the lower latitudes of western North America (Rogers et al., 1991; Dalton, 2005). New fossil discoveries (Heaton and Grady, 2003) and distinctive molecular genetic signatures that are endemic to the region (e.g., ermine; Fleming and Cook, 2002) are consistent with the Coastal Refugium hypothesis. These paleoendemics push our temporal perspective of the diversification of this fauna much deeper than the Holocene (*contra* Klein, 1965) and suggest that glaciation did not wipe the slate clean for all species.

Summaries of species richness and endemism based on taxonomy concluded that the North Pacific Coast is one of the regions of highest taxonomic endemism for mammals in North America (Cook and MacDonald, 2001). For example, traditionally four island and one mainland subspecies of ermine were described as endemic to Southeast Alaska. Variation among DNA sequences of ermine from across North America and Eurasia partitioned into three distinct lineages of ermine worldwide. All three occur in Southeast Alaska (Fig. 3), but they do not strictly correspond to the previously described endemic subspecies. One lineage has a very limited distribution along the North Pacific Coast and is found only on the Prince of Wales Island (POW) complex in Southeast Alaska (two subspecies) and nearby Haida Gwaii (another subspecies). The federal government of Canada has listed the Queen Charlotte ermine (*M.e. haidarum*; a member of this endemic island lineage) as threatened (COSEWIC, 2005), but there is no legal protection for this taxon in the United States. The restricted distribution of this lineage and its level of divergence from others suggest that these island populations may be derived from relicts that persisted in a refugium on the North Pacific Coast during the Wisconsin Glaciation. Further work is needed to carefully reconcile the differences between the subspecies classifications, based on morphologic studies, and the distinctive genetic lineages uncovered by DNA studies. The Prince of Wales complex of islands supports a large number of other endemic vertebrates including divergent forms of Keen's mouse, northern flying squirrel (Bidlack and Cook, 2002), spruce grouse (*Falcipennis canadensis isleibi*; Dickerman and Gustafson, 1996), and chum salmon (*Oncorhynchus keta*; Kondzela et al., 1994). This complex also hosts highly disjunct populations of subalpine fir (*Abies lasiocarpa*; Carrara et al., 2003).

About 23% of the 107 mammalian species or subspecies known from Southeast Alaska are endemic to the region (MacDonald and Cook, 1996). Molecular analyses are providing an opportunity to assess the depth of divergence, spatial extent, and taxonomic validity of these endemics, thereby building a foundation for understanding impacts, such as deforestation, on insular forms. Because endemism is highly likely in classes of organisms other than mammals, there is an elevated potential for extinction or extirpation of a variety of plants and animals. Hence, molecular studies now limited principally to mammals should be expanded to other taxa.

2.3.4. Differential arrival in the region

Progressive eastward retreat of ice during the Holocene is mirrored in the molecular genetic structure of the colonizing lineages. Recolonization routes northward along the Pacific Coast (presumably including areas of continental shelf now submerged) would have been available earlier than routes that crossed the Coast Range. Corridors from the east apparently were obstructed by the relatively late recession of the Cordilleran Ice Sheet in the Coast Range (Small et al., 2003). Coastal access into the region from the northwest remains largely blocked by tidewater glaciers to this day. Hence, representatives of coastal lineages apparently arrived in the region much earlier than the continental lineages. Coastal lineages consistently show deeper divergence when compared with nearby continental populations and coastal lineages typically are found on islands. In contrast, continental lineages are minimally diverged, appear to have recently colonized the region (likely through one of the large river corridors or mountain passes that transect the Coast Range), and generally are found along the mainland and a few nearshore islands (e.g., continental lineage of *Martes americana*). Wolverine, tundra voles, northern red-backed voles, and the Beringian lineage of ermine colonized from the north and generally show very low levels of diversification likely reflecting their late arrival into the region. Klein (1965) and Darimont et al. (2005) indicated that moose entered Southeast Alaska and coastal British Columbia within the last 100 years. Southeast Alaska populations may be

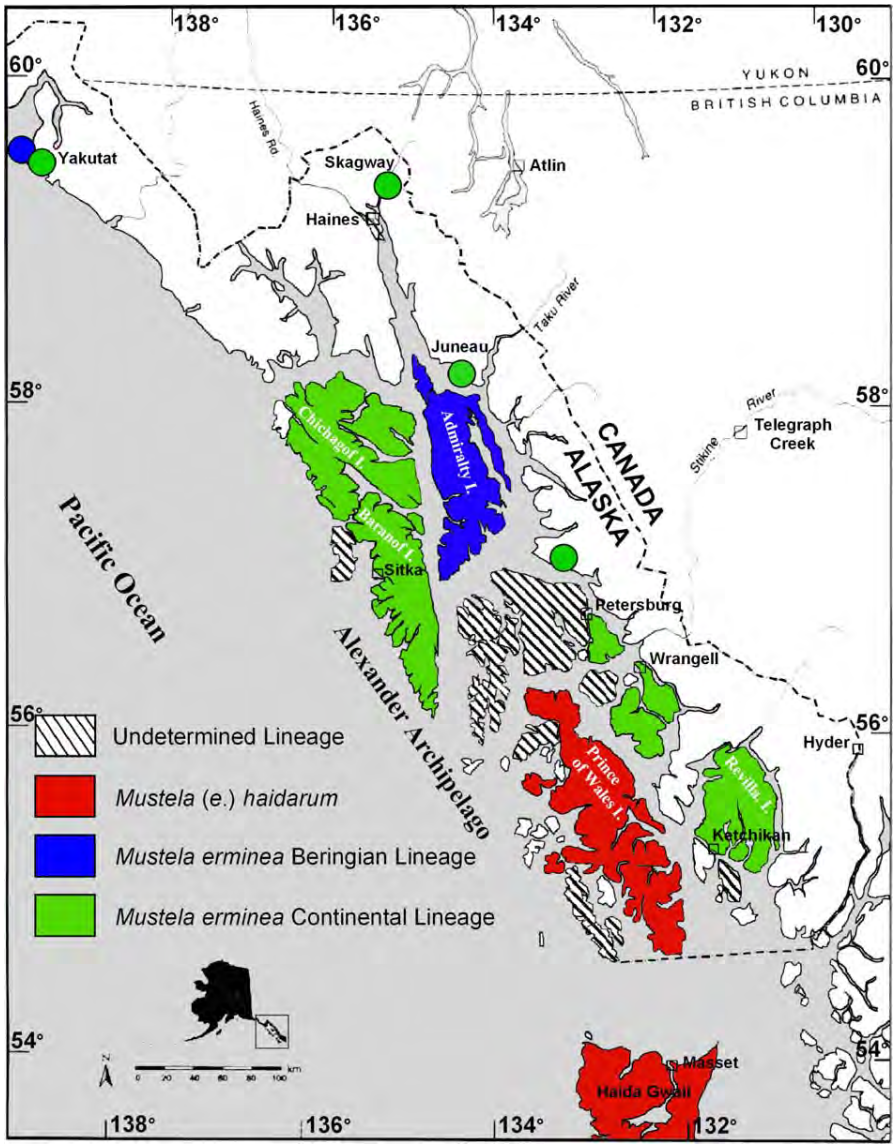

Fig. 3 – Three ermine clades are known worldwide and all three occur in Southeast Alaska. One lineage is endemic to only the POW island complex and the nearby Haida Gwaii islands of British Columbia (Fleming and Cook, 2002; Dawson unpublished data). Undetermined island records are those ermine specimens that have not been sequenced, so their lineage identity has not been determined yet. The mainland distribution is poorly documented, so only sites where lineages have been identified are shown.

the result of two immigrations, one from populations in nearby British Columbia and the other from northerly (Beringian) sources (Hundertmark et al., in press).

2.3.5. *Post-Pleistocene contact*

In several instances, distinctive lineages have come into secondary contact (contact zones) in Southeast Alaska (black bears, Peacock, 2004 and Stone and Cook, 2000; red-backed voles, Runck and Cook, 2005; marten, Small et al., 2003). Careful analysis of the dynamics of these zones has not been completed but may provide insight into the processes of diversification, reinforcement, and ultimately speciation. For example, we suspect that the two lineages of marten found in Southeast Alaska actually represent two distinct species, *Martes americana* and *M. caurina*, as previously proposed by Merriam (1890) based on morphological differences. If so, *M. caurina* has an extremely limited distribution in Southeast Alaska (Kuiu and Admiralty islands) and globally (northern California to Admiralty Island). Genetic analysis of marten shows deep phylogeographic subdivision that likely reflects incipient speciation. Using a suite of microsatellite markers and sequences of the mitochondrial cytochrome *b* gene, Stone and collaborators (Stone and Cook, 2000; Stone et al., 2002; Small et al., 2003) confirmed significant genetic divergence between *caurina* and *americana*. Hybridization between *caurina* and *americana* individuals was documented in two regions of sympatry (Kuiu Island in Southeast Alaska and southern Montana).

In summary, a number of species along the North Pacific Coast are comprised of multiple distinctive lineages that likely reflect a history of multiple colonization events from diverse geographic sources. In some cases, populations that we previously considered to represent a single species (e.g. montane shrew, Demboski and Cook, 2001) are instead composites of two species with very distinctive evolutionary histories. Both species should be recognized. In addition, populations of other species (although represented by a single lineage in Southeast Alaska) may be distinct from populations outside the region. Weckworth et al. (2005, submitted) documented unique genetic diversity in coastal wolves and concluded that these populations had colonized early in the Holocene from the south (southern source populations have since been extirpated) and subsequently were isolated from other North American populations. Limited dispersal eastward into interior British Columbia may be occurring, but there is no indication of gene flow into Southeast Alaska at this time. Knowledge of these historic routes of colonization may provide a framework for future restoration of some populations and species.

2.4. *Contemporary perspective: islands influence local diversity*

Elevated levels of intraspecific diversity due to the dynamic colonization history of the region described above are now heavily influenced or structured by the contemporary insular landscape. Some managers in Southeast Alaska have asked: Do we need to consider the special attributes that are associated with an "island lifestyle" given that the archipelago is likely less than 10,000 years old? The inventory of insular mammals stimulated a series of analyses aimed at addressing this question; analyses that were initiated at scales ranging from ecological communities down to molecular variability within populations.

2.4.1. *Island communities*

Consistent with classic island biogeography theory (MacArthur and Wilson, 1967), size and distance from the mainland have played a prominent role in shaping diversity across the Alexander Archipelago (Table 2). Larger islands (e.g., Revillagigedo) close to the mainland have higher species richness, while small distant islands (e.g., Forrester) have lower species richness (MacDonald and Cook, 1996). Conroy et al. (1999) used species lists for each island to explore factors responsible for community assembly of mammals across the archipelago. They demonstrated that mammalian communities on smaller islands were nested within communities on larger islands. Conroy et al. (1999) concluded that, across all species, colonization ability (not extinction probability) has been the most important determinant of current community composition on islands. That study, however, was unable to consider the hidden diversity (i.e., multiple lineages) that has since been documented for a number of these species. Reanalysis based on a more accurate depiction of richness uncovered by the molecular genetic studies (unpublished manuscript) indicates that lineages that are endemic to islands are not nested; those older forms instead appear to be structured by extinction processes.

Despite the overall nested pattern found in the archipelago (Conroy et al., 1999), species assemblages vary spatially (i.e., from island to island) and temporally (Heaton and Grady, 2003). For example, *M. longicaudus* is widespread among islands throughout the archipelago, while other species of *Microtus* have very limited distributions (e.g., *Microtus pennsylvanicus* on Admiralty Island). Historically, brown and black bears coexisted on Prince of Wales Island, but in the last 10,000 years brown bears have ceased to occupy this island (Heaton and Grady, 2003). Now, black bears occupy islands south of Frederick Sound, while brown bears occur on northern islands and the two species are sympatric only on the mainland.

2.4.2. *Linkages among islands*

Attempting to manage 2000 independent islands is an overwhelming task, but this complex landscape should not be managed as a single continuous unit. Substructure within the archipelago may exist, allowing the identification of groups of islands that are tightly connected (now or in the past, >10,000 ybp when sea levels were lower). MacDonald and Cook (1996) refined earlier attempts (e.g., Swarth, 1936) to simplify this complexity by proposing five biogeographic subregions based on the presence of endemics and unique combinations of native species. These subregions are bounded by significant biogeographic barriers and generally parallel Swarth's original review of the region (1911, 1936). We have attempted to refine these subregions using molecular phylogeography (Fig. 1). The challenge is to delineate areas with high levels of connectivity as these could form the basis for management units grounded in biology. Molecular studies

Table 2 – Island size and mammalian species richness varies across major islands of the Alexander Archipelago (MacDonald and Cook, 1996, unpublished data)

Island	Area (km^2)	Species richness
Prince of Wales	6675	15
Chichagof	5388	11
Admiralty	4362	14
Baranof	4064	10
Revillagigedo	2965	24
Kupreanof	2813	19
Kuiu	1962	16
Etolin	870	17
Dall	655	12
Wrangell	560	24
Mitkof	546	22
Zarembo	478	11
Kosciusko	437	12
Kruzof	435	7
Annette	392	7
Heceta	181	10
Sukkwan	167	8
Suemez	153	10
Duke	149	6
Long	115	9
Baker	115	3
Noyes	97	3
San Fernando	89	5
Lulu	78	3

are well suited to testing the significance of these barriers and validity of subregions. Indeed our very preliminary studies have revealed common genetic signatures (or linkages) across island groups for a variety of taxa. The endemic Prince of Wales flying squirrel, for example, has diverged from mainland squirrels (Demboski et al., 1998; Bidlack and Cook, 2001), but little to no differentiation was found among island populations within the Prince of Wales Complex (e.g., Prince of Wales and islands to the west like Heceta, Suemez, Dall). Such a pattern suggests high historic or contemporary levels of connectivity among these islands. Other species that share this close connectivity across the POW islands include Keen's mouse (Lucid and Cook, 2004) and ermine (Fleming and Cook, 2002).

A number of factors should be assessed with regard to connectivity. The role of purported linkages in facilitating gene exchange has varied considerably through time, so signals of contemporary and historic gene flow must be teased apart. Differences in life history characteristics among species and between sexes within a species also should be recognized when assessing probability of movement among islands (e.g., Paetkau et al., 1998). Large carnivores and herbivores are more vagile and likely moving among islands much more often (populations are more highly connected) than smaller mammals such as shrews and mice. Preliminary molecular perspectives (Table 1) are consistent with this trend. Differential connectivity across these groups provides a more precise view of the severity of particular barriers. With regard to assessing barriers throughout the archipelago, nearly ubiquitous species such as Sitka black-tailed deer (*Odocoileus hemionus sitkensis*) or Keen's mouse are good candidates for identifying levels of connectivity across the entire archipelago, but species with more limited distributions are also informative. A solid understanding of the relative timing of arrival into the region is also essential to interpreting connectivity for particular species. That is, wolverine or moose might be capable of dispersing to all islands in the archipelago, but because of their relatively recent arrival in the region they are still limited primarily to the mainland and nearshore islands (Tomasik and Cook, 2005; Hundertmark et al., in press) or islands tightly linked to the mainland such as Mitkof, Kupreanof and Kuiu.

Furthermore, faunal exchange among islands may not be symmetrical, with some islands acting as source populations while others act as sink populations. Kuiu Island has one of the highest black bear densities worldwide (e.g., 1.5 bear/km^2) and a larger number of migrants per generation move from Kuiu to Kupreanof Island than vice versa (Peacock, 2004). Unequal rates of immigration illustrate the distinctive microevolutionary dynamics that populations in close proximity, but on different islands, may be experiencing. In addition, Peacock (2004) noted little exchange between Kuiu and Prince of Wales islands, with each appearing to be discrete populations. Given the increased harvest pressures on black bears in Southeast Alaska (and in particular on these two islands), these new perspectives on connectivity should guide future management decisions related to potentially unsustainable harvests.

2.4.3. *Islands and genetic variability*

Lucid and Cook (2004) assessed levels of genetic variability across 23 island populations of the widespread Keen's mouse. Populations of this ubiquitous species showed highest levels of variability on the mainland and larger islands while lowest variability was recorded on the smallest and most distant islands. Other island endemics show extremely low levels of variability (Prince of Wales flying squirrel and *caurina* marten). Hence, insularization reduced species richness on these islands and also left a characteristic signature of lower genetic variability on the few species examined (Table 1). The generality of this finding to other Alexander Archipelago organisms should be explored. Lower genetic variability decreases the likelihood that a species will be able to respond to novel environmental challenges and thereby increases the probability of local extirpation (Frankham, 1995). Nonetheless, there is much we have to learn with regard to the relationship between genetic variability and the process of extinction (Aguilar et al., 2004). Considerable interest also is building in the new fields of landscape and community genetics (Whitham et al., 2003).

Islands have sculpted diversity at levels scaling from molecular variability through ecological structure and community composition in the Alexander Archipelago. It is precisely this "island effect" on diversity that may be profoundly impacted by humans in the coming decades as has been documented in other archipelagos worldwide (Quammen, 1996). Much more detailed sampling of the islands will allow the development of comprehensive archives of specimens that will help establish baseline (or historic conditions) for assessing change across the archipelago. When inventories are followed by molecular studies, new perspectives on the status of wildlife populations will emerge that are likely to contribute to a variety of future management questions across the Alexander Archipelago.

2.5. *Threats to island systems*

Habitat conversion, mining, increasing human settlement, tourism, sport and subsistence hunting and trapping, and species introductions should be more stringently monitored and regulated with respect to impact on island endemics. In addition, climate warming is predicted to substantially increase extinction risk for populations and species with limited ability to disperse (Thomas et al., 2004), such as those found on islands. A growing number of management plans have been established for archipelagos worldwide and these threats are the subject of numerous scientific papers, books, and action plans (e.g., Golumbia, 2000; Sherley, 2000; McNeely et al., 2001; Wittenberg and Cock, 2001).

2.5.1. *Differential impacts across islands*

The relatively few morphological (Eger, 1990; Dickerman and Gustafson, 1996) and molecular studies (Table 1) completed on Alexander Archipelago organisms in the past few decades have tagged the POW island complex as an important center of endemism (Fig. 4b). Perturbations on these islands need to be carefully regulated and monitored, yet it is unfortunate that most of these islands (e.g., Prince of Wales, Heceta, Suemez, Long, Tuxekan) already have been (or are projected to be) heavily roaded and deforested (Fig. 4a,c). Prince of Wales Island, in particular, is likely to experience large increases in human occupation (and associated impacts) in the coming

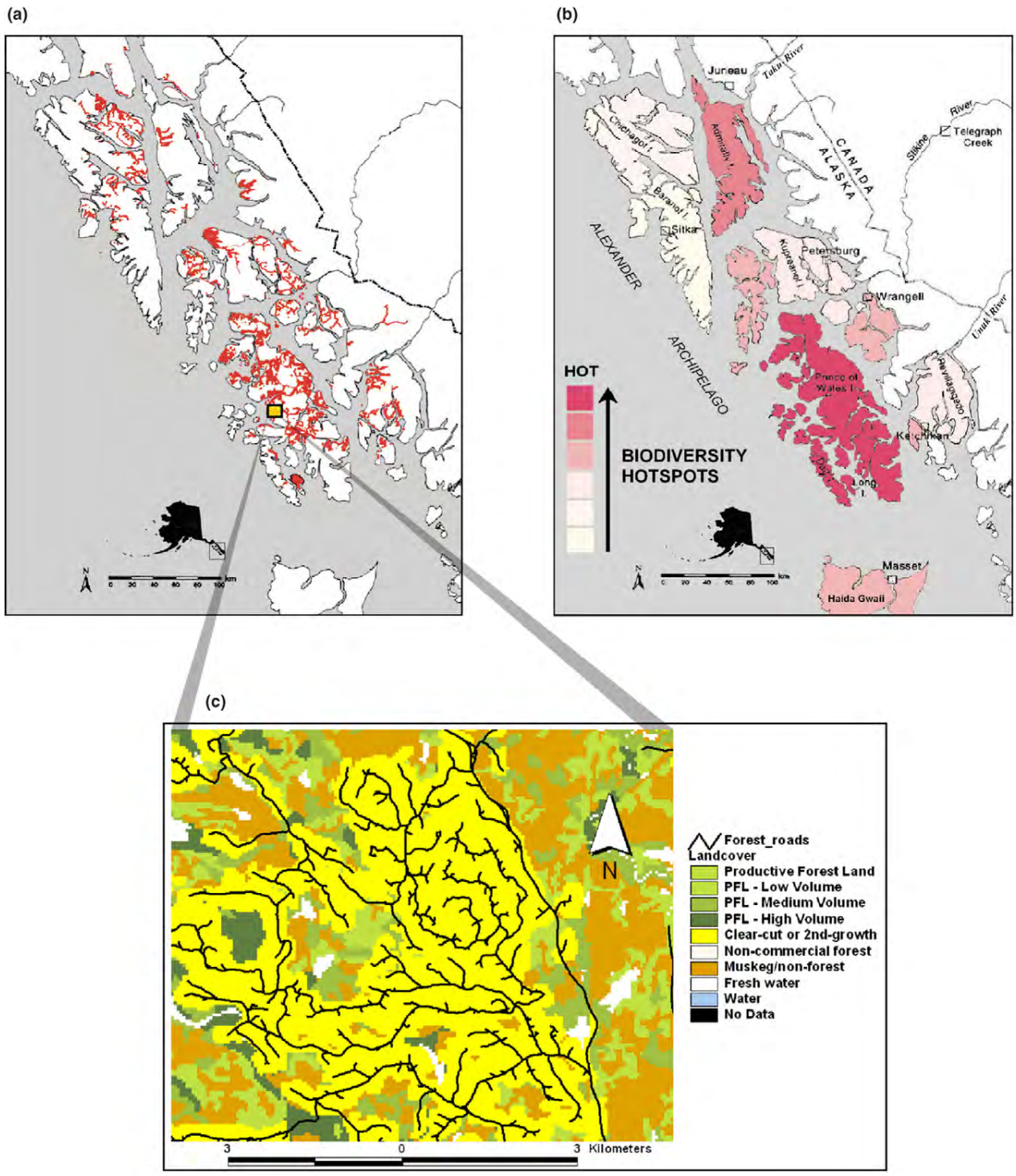

Fig. 4 – (a) Areas of concentrated clear-cut logging (now secondary-growth vegetation) and roads in Southeast Alaska since 1950 are highlighted in red (*Source*. www.inforain.org/tongass). (b) Areas of greatest conservation concern (diversity hotspots) in the Alexander Archipelago, Southeast Alaska, based on the distribution of eight endemic mammal lineages. Note the correspondence between most heavily impacted areas and highest concentration of endemics (e.g. POW island complex). (c) Clear-cut logging practices in Southeast Alaska result in a high density of roads and highly fragmented forests.

decade due in part to increased access across the island. Prince of Wales Island now has over 3000 km of roads, nearly half of all timber-associated roads within the Tongass National Forest. Roads can severely fragment populations and increase human access to remote areas, thus increasing the probability of over exploitation of some species (Person et al., 1996). New ferry terminals on northern Prince of Wales Island (Coffman Cove) and southern Mitkof Island (Blind

Slough) are opening in 2006 and will stimulate additional traffic and development across these islands.

2.5.2. *Invasive and exotic species decimate insular faunas*

A significant threat to island ecosystems worldwide is the introduction of nonnative species (Golumbia, 2000). For example, steep declines in diversity have been documented on tropical islands of the Pacific, such as Guam, other Mariana Islands, and the Hawaiian Archipelago (Fritts and Rodda, 1998). Introduction of exotic species such as rats, mongooses, and brown tree snakes is the prime cause of the demise of these faunas, and these ill-fated events have led to severe restrictions on commerce and development on these islands. Although tropical islands are routinely cited as the classic examples of highly threatened insular biotas, North Pacific archipelagos have also experienced significant reductions in diversity and loss of ecosystem function. Nearly 50 years ago, Murie (1959) noted the devastation experienced by nesting waterfowl on the Aleutian Islands after foxes and rats were introduced. In the last decade, the US Fish and Wildlife Service has instituted an aggressive (and expensive) campaign to eradicate those exotics from over 30 islands (S. Ebbert, pers. comm.). The Alaska Maritime Refuge also helped coordinate proactive efforts on St. George and St. Paul (Pribilof Islands) in 1993 to stymie the possibility that rats (*Rattus* ssp.) might establish populations on these productive islands. Closer to the Alexander Archipelago, the introduction of rats (*Rattus norvegicus*, *R. rattus*), raccoons (*Procyon lotor*) and red squirrels (*Tamiasciurus hudsonicus*) to Haida Gwaii (Queen Charlotte Islands) has had a profound impact on seabirds, songbirds, and other native species (Bertram and Nagorsen, 1995; Hartman and Eastman, 1999; Golumbia, 2000; Martin et al., 2001).

Of the two distinctive forms of marten on the Alexander Archipelago, molecular signatures suggest that *caurina* is the original island form (now only on Kuiu and Admiralty). These northern insular populations of *caurina* exhibited higher differentiation and lower variability relative to northern populations of *americana*. Greater divergence among *caurina* populations may reflect longer isolation and persistence in coastal forest habitat that was fragmented by rising sea level in the early Holocene. The endemic *caurina* lineage apparently has been displaced by the *americana* form, a more recent colonizer of the coast. Indeed, we suspect that *americana* has naturally colonized just a few near-shore islands, but the ongoing displacement (or genetic swamping) of *caurina* by *americana* has likely been hastened through human-mediated introduction of *americana* to Chichagof, Baranof, Prince of Wales and a number of other islands in the last century. Unfortunately, we have too few museum records from these islands to accurately determine the historic range of *caurina* or *americana* across the archipelago. Specimens from several key islands (e.g., Dall, Annette, Heceta, Tuxekan, Zarembo) have not yet been characterized. The impact of industrial logging, fur trapping, and translocations on marten and especially the insular populations of *M. caurina* should be carefully monitored.

There are examples of introductions of 18 other mammal species and three amphibians to Southeast Alaska (unpublished data) and these include exotics like raccoons to islands in Sea Otter Sound and Baranof Island, elk introduced onto Etolin Island (and now spread to Zarembo, Prince of Wales, and other nearby islands and surrounding mainland), and foxes to numerous islands (Bailey, 1993). Increasing human populations and associated pets will negatively impact wildlife populations, such as the transmission of pathogens to wild carnivores (Thorne and Williams, 1988). All of these species have devastated natural systems and cost millions of dollars in control efforts elsewhere when they have been introduced to islands (Bailey, 1993; Martin and Daufresne, 1999; Golumbia, 2000; Burbridge and Manley, 2002; Blackburn et al., 2004).

Over 20 species of exotic plants have been identified as invasive within southeast Alaska (Huette and Bella (in press)) and most thrive in disturbed habitat. Prince of Wales Island is experiencing an influx of domestic exotic plants (e.g., Scotch Broom *Cytisus scoparius*), that have rapidly invaded the island along roads. A comprehensive species introduction task force should be constituted to establish more effective regulations and ensure their implementation. The Alexander Archipelago has already experienced a number of intentional and unintentional introductions (unpublished data). The easiest and cheapest way to avoid significant impacts from invasive species is prevention.

2.5.3. *Maintaining connectivity among islands*

MacDonald and Cook (1999, manuscript) proposed several areas within Southeast Alaska that may function as linkages, but a comprehensive GIS review that takes into account factors such as oceanic currents and spatial reconstructions of past sea level fluctuation and its impact on connectivity needs to be completed. Management plans should prioritize the protection of sites that may facilitate connectivity among islands (not just within islands) by establishing logging (and other disturbance) buffers for suspected linkages and wildlife corridors for natural movement of organisms between islands.

2.6. *Maintaining the ecological integrity of the Alexander Archipelago*

The Alexander Archipelago is a thriving and productive north temperate island system undergoing significant environmental change. Limited information on island endemic mammals (morphological descriptions and molecular phylogeography) portends comparable or perhaps higher levels of divergence in other organisms. Inventory programs coupled with molecular perspectives of selected species have been used across the globe in the restoration and conservation of important managed landscapes. One example is helping managers choose appropriate source populations for efforts to supplement declining population (Florida panther *Puma concolor coryi*; Maehr and Lacy, 2002). Molecular techniques have allowed managers to supplement declining populations based on knowledge of evolutionary histories, and hence ensure genetic integrity of species (Matthee and Robinson, 1999). Such a molecular framework would have demonstrated the folly of the introduction of mainland *americana* marten onto several islands in the Alexander Archipelago. Active management of marten and marten habitat over the last 50

years has likely hastened the extirpation of *caurina* marten in the region, but the overall impact is unknown (MacDonald and Cook, 1996). In addition, few studies have addressed basic ecological attributes of these insular forms, but seminal ecological studies of coastal and insular mammals should help guide future management and restoration efforts (e.g., McCabe and Cowan, 1945; Darimont et al., 2004). Still, managers cannot reproduce the process of natural selection that has been shaping these coastal populations since their arrival, so it is critical that key components of this endemic fauna are not lost.

2.7. *Management recommendations and research needs*

Sophisticated technologies have significantly enhanced wildlife management over the past few decades. Given the complexity of issues facing resource managers along Alaska's southeastern coast and particularly on this vast archipelago, a diverse set of approaches is needed to effectively administer wildlife management plans and monitor potential threats to these native biotas. Several steps can be taken immediately that will lessen costs associated with impacts and facilitate swift responses to these perturbations in the future.

First, a serious discussion by state and federal management agencies, Native American and other local communities, politicians, NGOs, and other interested groups should focus on establishing a comprehensive management plan aimed at preserving the endemic biota and natural ecosystems of the Alexander Archipelago. Other regions of the planet are grappling with similar issues and are developing guiding principles for management of island systems.

Second, there is a pressing need to continue inventorying the biotic diversity of these islands, but inventories should be specimen based and include more than just vertebrates. Building spatially and temporally deep archives of insular wild populations on the Tongass is critical to many kinds of investigations that will form the basis for careful management (Chapman, 2005) and broad, integrated investigations (Cook et al., 2005; Thompson, 2005). In many situations worldwide, the lack of available baseline materials has become the primary limitation to effective application of new technologies.

Finally, inventories should be followed with surveys of molecular diversity and analyses of morphological variation so that centers of endemism and important sites for sustaining connectivity can be identified. Historical geologic events, combined with the insular nature of the region have contributed to the development of a highly structured, diverse, and endemic biota. Hence, islands are distinct based not only on their geographic isolation, but also due to the dynamic interplay between glacial advance, isostatic rebound and sea level recalibration. Molecular perspectives are powerful allies in our quest to understand island systems as they provide the bridge between deeper history and contemporary population dynamics and conservation biology (Riddle, 1996). This integrated approach is elucidating the region's rich historical complexity, highlighting contemporary insularity, and charting a course for addressing imminent conservation concerns on the Tongass and on other archipelagos worldwide.

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Should endemism be a focus of conservation efforts along the North Pacific Coast of North America?

J.A. Cook [a,*], S.O. MacDonald [a,b]

[a] University of Alaska Museum, 907 Yukon Drive, Fairbanks, AK 99775-6960, USA
[b] PO Box 58, Gila, NM 88038, USA

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Abstract

Most documented extinctions of vertebrates in the last 400 years have been island endemics. In this paper, we focus on the need to develop a historical framework to establish conservation priorities for insular faunas and, in particular, to test the validity of nominal endemics. We use the example of the islands of the North Pacific Coast (NPC) of North America, a region that includes approximately one-half of all mammals endemic to North American islands north of Mexico (seven species and 67 subspecies). Few of these endemics have been re-evaluated since their original descriptions, although many of these islands have been heavily impacted by habitat conversion, species introductions, over-exploitation, and secondary ripple effects. Evidence from molecular genetics and paleontology suggests that many taxa arrived in the region since the last glacial advance. Some of these nominal endemics show minimal differentiation, while others comprise multiple evolutionary lineages. The NPC may also have played an important role in the in situ diversification of some taxa (paleoendemics) during the Pleistocene. Evidence, such as new fossil discoveries and numerous described endemics, lends support for refugia hypotheses. However, these ideas cannot be tested until a better understanding of geographic variation and the evolutionary relationships of the fauna and flora of the North Pacific Coast is developed. This framework would provide significant insight into the dynamic biogeographic history of the region and help prioritize conservation efforts. © 2000 Elsevier Science Ltd. All rights reserved.

Keywords: Endemism; Mammals; North Pacific Coast; Alaska; British Columbia

1. Introduction

Finding solutions to the accelerating loss of our planet's rich biological heritage is a daunting challenge for conservation biology. A disproportionate percentage of documented loss during the past 400 years has occurred on islands, with some estimates suggesting that worldwide, 63% of mammalian, 95% of reptilian, and 80% of avian extinctions have been insular endemics (Diamond, 1989; Olson, 1989; Reid and Miller, 1989; Case et al., 1992; Steadman, 1995). Insular taxa are prone to extinction because they are vulnerable to demographic stochasticity, random climatic events, and anthropogenic disturbance (Shaffer, 1987; Burkey, 1995). Because humans continue to alter and fragment habitat, introduce exotics, and exploit native species, significant numbers of extinctions in the next few decades will continue to occur on islands (Diamond, 1989; Reid and Miller, 1989).

It is, therefore, surprising that little attention has focused on potential threats to island biotas along the North Pacific Coast (NPC) of North America (Fig. 1), particularly those taxa that are found only there (i.e. endemic). NPC extends from the Olympic Peninsula of Washington north and west to the Bering Straits off Alaska. Islands in NPC vary from nearshore archipelagos in British Columbia and southcoastal Alaska to the more isolated islands of the Bering Sea, supporting seven endemic species and 67 endemic subspecies in six mammalian orders (Fig. 2). These diverse taxa comprise about half of the total number of island endemic mammals for North America north of Mexico (Hall, 1981, 1984; updated by Nagorsen, 1990; Wilson and Reeder, 1993). In addition, three species and 51 subspecies of mammals are endemic to NPC, but are not exclusively

* Corresponding author. Biological Sciences, Idaho State University, Pocatello, ID 83209-8007, USA. Tel.: +1-208-282-4082; fax: +1-208-282-4570.

E-mail addresses: cookjose@isu.edu (J.A. Cook), somacdonald@gilanet.com (S.O. MacDonald).

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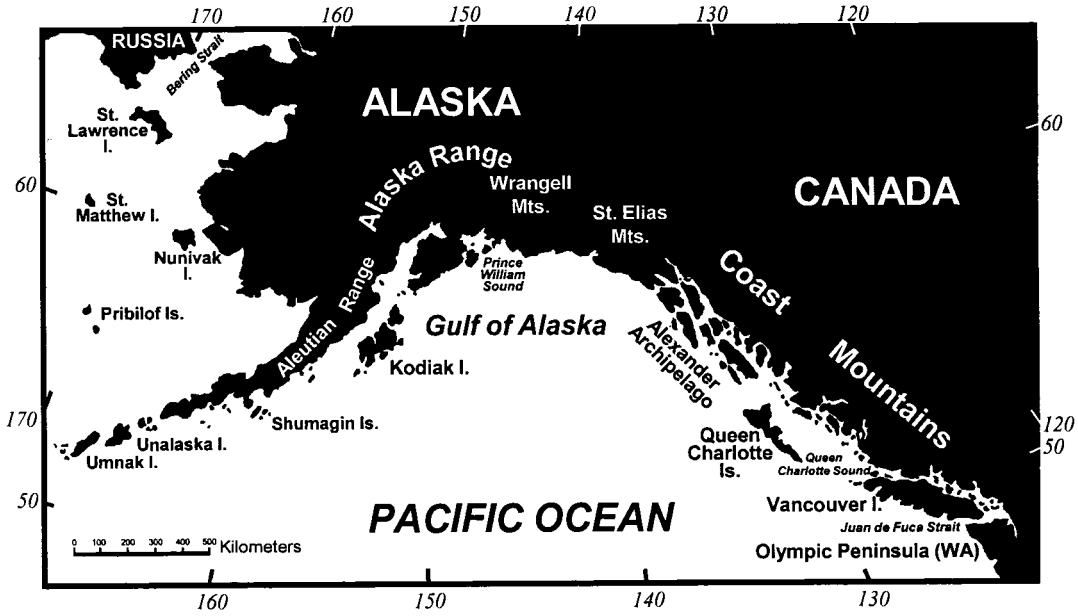

Fig. 1. Location map of the North Pacific Coast (NPC) of North America. The extreme western islands of the Aleutian Archipelago are not shown. The coastal region is naturally defined as the region west of the coastal ranges which extends from Washington through British Columbia to Alaska.

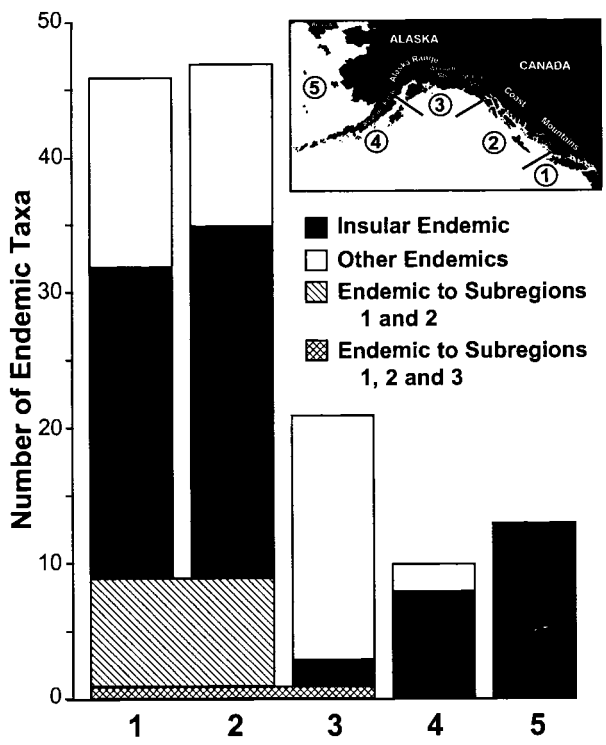

Fig. 2. Number of insular and non-insular mammalian taxa that are endemic to five subregions along the North Pacific Coast (NPC) of North America. Five subregions are delineated to simplify the interpretation of endemism across this large region. Subregions are defined based on physiographic features which define distinctive breaks in spatial clusters of nominal endemics (from maps in Hall, 1981). These breaks apparently do not reflect the underlying phylogeographic structure of NPC (Cook et al., 2001).

insular. The highest concentration of endemics is found along the coasts of British Columbia and southeastern Alaska (Fig. 2) which are largely isolated from the remainder of North America by coastal mountain ranges. With few exceptions (e.g. Rausch and Rausch, 1995), most NPC endemics have not been re-evaluated since their initial descriptions decades ago (Nagorsen, 1990; MacDonald and Cook, 1996); a significant issue when dealing with current conservation law such as the Endangered Species Act.

The failure to focus on NPC endemics has fostered a perception that few taxa in the region are of imminent conservation concern (e.g. Schoen and West, 1994; Bunnell and Chan-McLeod, 1997). In British Columbia, only one island endemic mammal, the Vancouver Island marmot, *Marmota vancouverensis*, is listed nationally and provincially as endangered (Nagorsen, 1994; Shank, 1999). Six additional island taxa are on the provincial list but as unofficial 'candidates' only. One subspecies, the Queen Charlotte caribou, *Rangifer tarandus dawsoni*, is now considered extinct. In Alaska, no terrestrial mammals are currently listed as threatened or endangered by the United States Fish and Wildlife Service (Federal Register, 1997), although NPC island taxa have or are expected to experience significant impacts (e.g. Hafner et al., 1998). Examples of human-caused extirpation early in this century include several island populations of beaver, *Castor canadensis*, and marten, *Martes americana* (MacDonald and Cook, 1996).

Endemism has been defined by two primary attributes: taxonomic validity and spatial distribution (Anderson, 1994). Assessing the temporal scale of divergence

provides a third perspective on endemism. These three aspects have not been defined for most NPC endemics.

1.1. Taxonomic validity

Because nomenclature shapes our perceptions of how the biological world is organized (Avise, 1994), taxonomic categories are the foundation for legal actions and subsequent management efforts. The limitations of these classifications have been under-appreciated for NPC endemics. The descriptions of most NPC endemic mammals were based on relatively few specimens (e.g. Osgood, 1905; Jackson, 1928), so the large number of NPC endemics may simply be the result of overzealous 'taxonomic splitting' in vogue during the earlier part of this century (Engstrom et al., 1994). For example, the Admiralty Island beaver, *Castor canadensis phaeus*, was described on the basis of only three adult specimens (Heller, 1909). If these purported endemics are invalid (and a number of them undoubtedly are), the legally mandated task of conserving diversity in this complex region may be less daunting.

1.2. Spatial scale

The geographic distribution of an endemic should be documented carefully. Without an analysis of geographic variation based on extensive sampling, the spatial scale of conservation efforts cannot be defined. With few exceptions (e.g. Waits et al., 1998), the geographic ranges of NPC endemics are based on collections from few localities (Nagorsen, 1994; MacDonald and Cook, 1996). In the Alexander Archipelago, limited inventories of mammals (only 22 of the >2000 islands) had formed the basis for more than 80 years of federal management (including the 6.9 million ha Tongass National Forest (MacDonald and Cook, 1996)).

1.3. Temporal scale

The time (depth) of divergence between populations or taxa is a third attribute of endemism. Myers and Giller (1988) distinguished paleoendemic (i.e. an older, relictual lineage that once had a larger distribution) from neoendemic (i.e. a newer lineage that is the result of recent arrival and in situ differentiation). Conroy et al. (1999) suggested that the majority of mammal species on the Alexander Archipelago are neoendemics. Others (e.g. Byun et al., 1997) have provided evidence of paleoendemism on the Queen Charlotte Islands (but see Demboski et al., 1999). All endemic taxa currently recognized should be re-evaluated and tested with molecular markers that would provide a relative estimate of divergence times; such approaches should be central to the prioritization of conservation efforts for endemics (Dizon et al., 1992). Within a particular clade, lineages (or taxa) with a longer history of divergence (e.g. paleoendemics) should be accorded higher conservation protection (Vane-Wright et al., 1991; Crozier, 1992). Characterizing the temporal dimension of endemics may, in turn, assist in efforts to maintain the evolutionary potential of the biota (Brooks et al., 1992; Avise, 1994).

1.4. Phylogeography and endemism

Phylogeographic studies (Avise, 1994; Wagner and Funk, 1995) aimed at diagnosing evolutionarily significant units (Moritz, 1994) or evolutionary species (Mayden and Wood, 1995) provide a foundation for conservation efforts. Phylogeographic studies can help define: (1) the taxonomic identity of endemics; (2) the spatial and temporal (neoendemic vs. paleoendemic) scales of concern; (3) overlapping or common tracts across endemics ('hot spots'); and (4) unique ecological (community) associations (Brooks et al., 1992). Once studies of individual taxa have been completed, concordant historical patterns can be identified (Cook et al., 2001). Comparative phylogeography (Avise, 1994; Riddle, 1996; Zink, 1996; Moritz and Bermingham, 1998) has the potential to provide the historical framework for decisions specifically related to management of endemics, as well as to conservation biology in general. For example, da Silva and Patton (1998) studied 15 species of neotropical rodents in Amazonia and identified geographical areas of high divergence and zones of concordant phylogeographic breaks. Some of these areas were completely unexpected and call attention to the need to re-evaluate conservation planning priorities previously identified for the Amazon Basin (Rylands, 1990). New approaches have also been advanced for quantitatively assessing congruent phylogenies and identifying evolutionarily distinct communities (e.g. Page, 1996; Moritz and Faith, 1998).

A phylogeographic approach has not previously been applied to NPC. Beyond the original taxonomic descriptions of NPC endemics, few investigations have assessed their status or further explored geographic variation. Conservation plans for the NPC have seldom recognized the limitations of existing taxonomic classifications. Because the region supports a large number of nominal endemics that may be impacted by a variety of anthropogenic stresses, we suggest that NPC may be a good case study for assessing the value of applying the historical perspectives derived from phylogeographic studies to conservation and management efforts.

2. Historical evidence for endemism in the NPC

New fossil records (Heaton and Grady, 1991, 1993; Heaton, 1995; Nagorsen et al., 1995, 1996; Heaton et

al., 1996) indicate that the NPC has a dynamic history. Mammals colonized recently de-glaciated areas (in some cases, perhaps, prior to the formation of island archipelagos by rising sea levels). Some fossil and sub-fossil taxa documented from newly discovered cave deposits in the Alexander Archipelago indicate the relatively recent turnover (sensu Vrba, 1992) of faunas since the last interstadial or warming period. The composition of ecological communities has been dynamic through the last 30,000 years in the region. For example, species typical of more northern latitudes (e.g. lemmings and arctic fox) occurred in the southern archipelagos during the cooler periods of full glacial advance.

Molecular genetic studies of NPC taxa (Hogan et al., 1993; Demboski et al., 1998b; Conroy and Cook, 2000) are providing alternative perspectives on these purported endemics. Patterns emerging from these investigations suggest a complex biogeographic history (Cook et al., 2001). Some taxa (e.g. tundra voles, *Microtus oeconomus*) show minimal amounts of genetic variation (based on allozymes or mitochondrial gene sequences) throughout the region; these populations appear to be the result of recent colonization of de-glaciated areas (Lance and Cook, 1998). This species is represented by seven nominal subspecies that are endemic to islands in NPC, yet the genetic data suggest that some of these may represent neoendemics or perhaps invalid taxonomic units.

In other cases, deep breaks suggest that the existing taxonomy underestimates the evolutionary diversity present in a species (Cook et al., 2001); failure to recognize and conserve these distinctive lineages may lead to serious management mistakes (Daugherty et al., 1990). Substantial levels of intraspecific differentiation suggest that glacial refugia may have played a role in the differentiation of NPC taxa, although some of the deeper genetic splits may have been initiated prior to the Pleistocene glacial advances. Potential refugia include Beringia (to the north), at least two areas south of the continental ice sheets (west coastal and central), and possible refugia along the NPC (Heusser, 1989; Rogers et al., 1991). These preliminary phylogeographic patterns indicate the need for a serious re-evaluation of the taxonomic framework currently employed in management efforts.

Documentation of the spatial and temporal elements of diversity and the genetic relationships of taxa would provide blueprints for managing intact ecosystems, in addition to the future rehabilitation of those already impacted. With two distinctive lineages of marten in NPC (Stone, 2000), we now realize that sound and effective management of this species could be compromised without a reasonable idea of the evolutionary history and the geographic structure of genetic variation. Marten were transplanted to several NPC islands in this century (Burris and McKnight, 1973) without prior assessment of native populations across the Alexander Archipelago. Preliminary data suggest that the americana lineage may be replacing the native caurina lineage on Kuiu Island (Stone, unpublished). Phylogeographic studies would provide a basis for selecting source populations, if needed, to restore species to particular islands or ecosystems. Such a framework would also facilitate the interpretation of causality for dynamic range shifts (e.g. habitat conversion, climate change) including the recent colonization of the NPC by mountain lion (*Puma concolor*), fisher (*Martes pennanti*), and moose (*Alces alces*) (MacDonald and Cook, 1996).

3. Threats to NPC endemics

3.1. Habitat destruction

Of the four threats to insular populations outlined by Diamond (1989), habitat destruction may be the most significant cause of endangerment on the southern NPC archipelagos that support temperate rainforest. On the Tongass National Forest in Southeast Alaska, for example, annual timber harvests of up to 267 million board feet will be allowed until 2007 (Tongass Land Management Plan, 1997). This rapid rate of habitat change may impact the fauna associated with the remaining old-growth forests (e.g. Demboski et al., 1998a), forest stands on beach fringes, and estuaries. Clear-cut harvesting on many southern NPC islands and the mainland (Fig. 1; subregions 1–3) has fragmented the old-growth forest into extensive, unproductive tracts of 100-year-old-or-less thickets of second-growth closed-canopied forests. The effects of these activities on endemic populations of mammals are largely undocumented; however, the few studies relating species diversity to timber management in the region suggest the potential for significant impacts (Dellasala et al., 1996; Parker et al., 1996).

The high incidence of endemism noted for Prince of Wales Island and surrounding islands (MacDonald and Cook, 1996) coincides with intense logging activities on the Tongass National Forest and adjoining private lands. For example, the ermine subspecies, *Mustela erminea celenda*, is endemic to Long, Prince of Wales, and Dall islands. Assessment of mitochondrial DNA sequences from those populations indicates that they are distinctive from nearby mainland populations (Cook et al., 2001; Fleming and Cook, unpublished). Similarly, the endemic flying squirrel, *Glaucomys sabrinus griseifrons*, from the Prince of Wales complex, is also genetically distinct from mainland populations (Demboski et al., 1998b).

3.2. Effects of species introductions

The negative effects on endemics caused by the introduction of exotic species are well documented (Nilsson,

1983; Diamond, 1984; MacPhee and Marx, 1997; Fritts and Rodda, 1998). A long history of species introductions has lead to the subsequent decline of native fauna on NPC islands (Burris and McKnight, 1973; Bailey, 1993; Gaston, 1994; Bertram and Nagorsen, 1995). For example, arctic fox (*Alopex lagopus*) and red fox (*Vulpes vulpes*) were introduced for commercial fur harvesting to over 455 NPC islands (Bailey, 1993). Russians began these introductions in 1750, resulting in the demise of many insular populations of birds (Murie, 1959; Bailey, 1993). Elk, *Cervus elaphus*, were transplanted to Etolin Island in the Alexander Archipelago in 1987 and have increased in number, recently spreading to nearby islands (MacDonald and Cook, 1996). Mountain goats, *Oreamnos americanus*, were to be introduced to the alpine regions of Montague Island in Prince William Sound in 1993, near the type locality of an endemic marmot, *Marmota caligata sheldoni*, that has not been seen (or surveyed) in more than 90 years (Cook, 1998).

3.3. Over-exploitation

A third potential cause of endangerment to NPC endemics is direct killing by hunting and harvesting. Hunted species with restricted distributions, small population sizes and densities, large home ranges, low rates of population increase, poor dispersal abilities, and/or specialized niche requirements may be particularly vulnerable (Primack, 1993). Person et al. (1996) reported a significant positive correlation between the linear kilometers of logging roads and the number of wolves killed by hunting or trapping on Prince of Wales Island. Declines in island populations of this endemic subspecies of wolf, *Canis lupus ligoni*, may result from increased access due to this expanding network of roads. Concerns for the viability of island populations of other game or fur-bearing species such as brown bear, marten, river otter, *Lontra canadensis*, and mountain goat, among others, have been raised recently for this region (Suring et al., 1992; Kiester and Eckhardt, 1994).

3.4. Secondary ripple effects

The extirpation of an endemic may lead to a cascade of secondary extinctions of other species. For example, keystone species like beaver apparently were over-harvested on some islands (e.g. Chichagof and Baranof) in the early 1900s (MacDonald and Cook, 1996). Similarly, loss of insular populations of flying squirrels or red-backed voles, *Clethrionomys gapperi*, may impact forest regeneration because these species appear to be key to dispersing fungal spores (Maser et al., 1986; Terwilliger and Pastor, 1999). These spores are known to form ectomycorrhizal symbioses that are critical to forest health, including the invasion of meadows by conifers. Furthermore, most of these island faunas have low species richness, so the loss of a small herbivore that functions to cycle nutrients and feed predators may significantly alter ecological processes.

4. Conclusions

Conservation plans for the region generally have not acknowledged potential threats to the endemic fauna. These threats should be evaluated and prioritized based on a re-examination of the validity of purported endemics including estimates of levels of genetic divergence. A growing number of phylogeographic studies (e.g. Karl and Bowen, 1999) reveal that current taxonomic designations for some species of high conservation concern may not accurately depict existing genetic structure. For the NPC, limited human and financial resources for conservation efforts may not be efficiently allocated if spent on nominal endemics that show minimal levels of genetic differentiation. Preliminary surveys of NPC mammals have discovered substantial levels of divergence for some endemics, while other purported endemics are not genetically distinctive (Cook et al., 2001).

The effective conservation of NPC endemics will require collaborative efforts and diverse perspectives from the various approaches now encompassed by the field of conservation biology. A solid taxonomic framework that adequately reflects the geographic structure of genetic variation found in the organisms of this dynamic region will provide a necessary foundation for these conservation studies (Vane-Wright et al., 1991; Rojas, 1992; Vane-Wright, 1996). The limitations and uncertainty of available information on endemics must be incorporated into management decisions, particularly those with a high probability of impact on insular populations (May, 1990).

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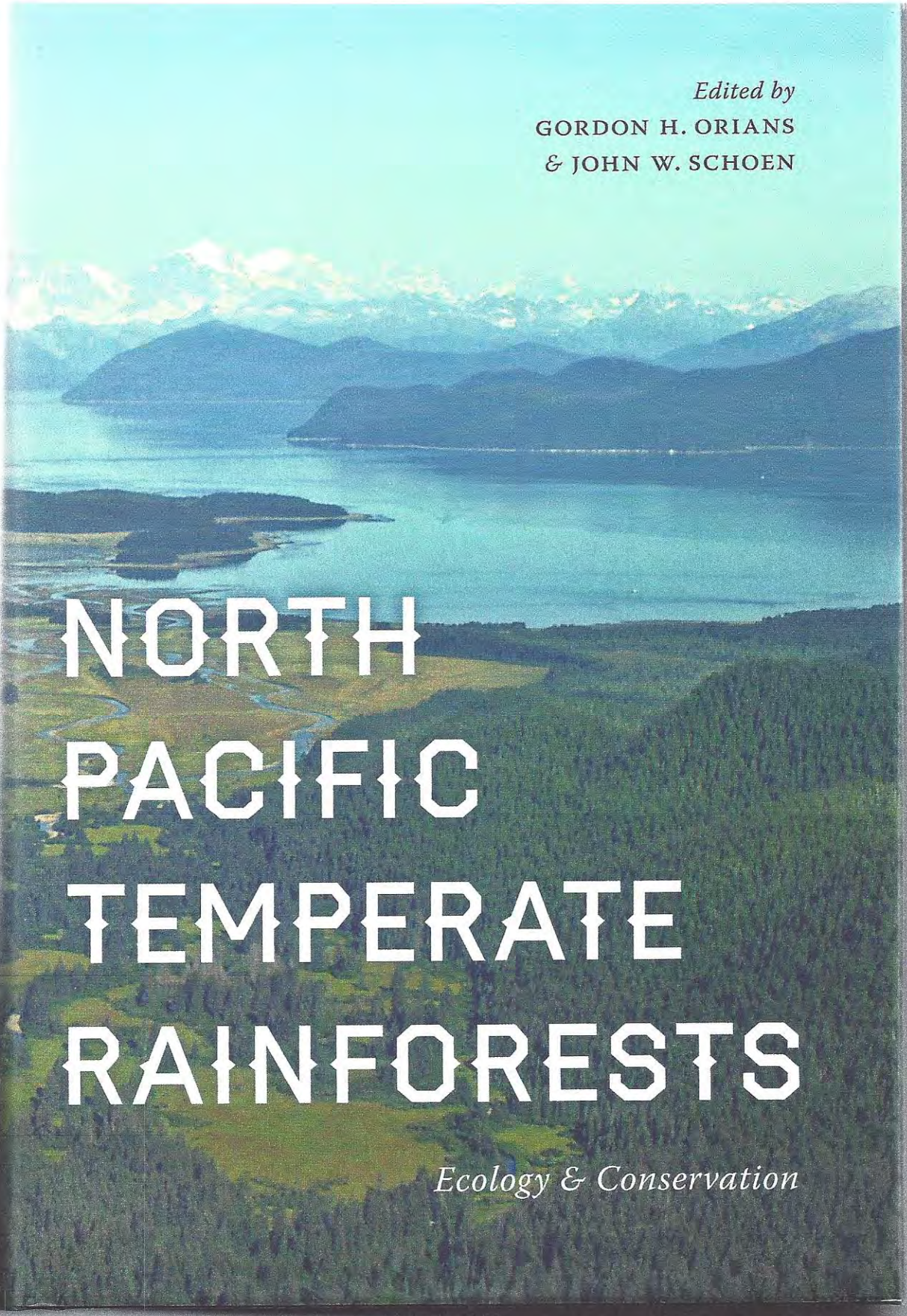

Edited by
GORDON H. ORIANS
& JOHN W. SCHOEN

NORTH PACIFIC TEMPERATE RAINFORESTS

Ecology & Conservation

2

Island Life

COMING TO GRIPS WITH THE INSULAR NATURE OF SOUTHEAST ALASKA AND ADJOINING COASTAL BRITISH COLUMBIA

Joseph A. Cook and Stephen O. MacDonald

> *Island environments have been generally misunderstood, misinterpreted, and mismanaged. . . . Islands are not simply miniature continents and . . . continental solutions do not simply need to be scaled down in order to be successful.*
>
> —P. D. Nunn, "Through a Mist on the Ocean: Human Understanding of Island Environments"

INTRODUCTION

Southeast Alaska (fig. 2.01), with its remarkable temperate rainforest, extends from Icy Bay southward to the international border at Dixon Entrance. Across this 800 km expanse, more than 1,000 named islands of the Alexander Archipelago extend along the coastline. The adjoining mainland is narrow and bordered to the east by the Coast Mountains and towering Wrangell-St. Elias ranges. These mountains rise to over 5,400 m in elevation, are capped by alpine glaciers and extensive ice fields, and form a substantial barrier to the movement of organisms into the coastal region. Hence, two physiographic features, the precipitous coastal mountains and the highly fragmented Alexander Archipelago, overwhelmingly contribute to the current geographic structure of this terrestrial biota. Although southeast Alaska is the primary focus of this chapter, to the south the coast of British Columbia and archipelago of Haida Gwaii (Queen Charlotte

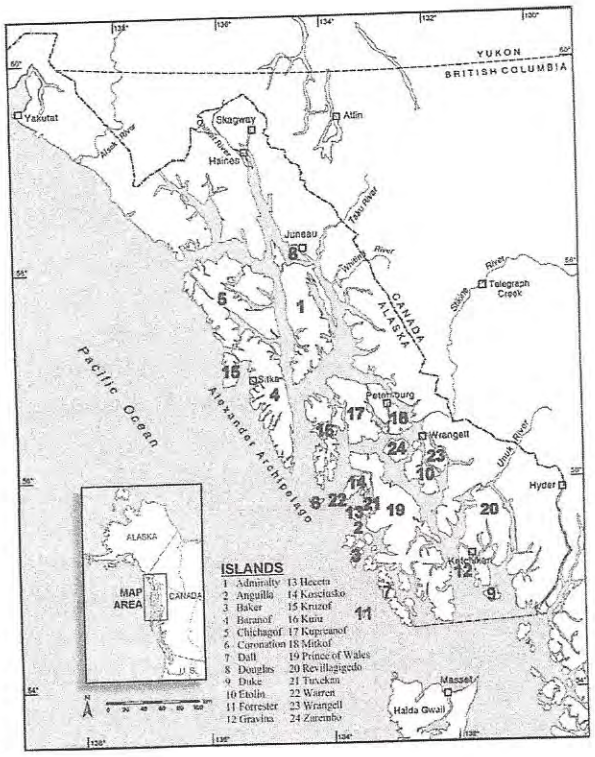

FIGURE 2.01. Map of southeast Alaska with major islands identified (see table 2–2).

Islands) are equally significant components of the North Pacific temperate rainforest. The dominant influence of geography throughout the entire region suggests that a comprehensive view of the coastal biome will require critical appraisal of the role of physiographic features in species existence and persistence, community assembly, and ecosystem functioning. At both ecological and evolutionary time scales, a host of environmental perturbations combined with topographic changes due to glacial rebound and sea level fluctuation characterize the Alexander Archipelago and Haida Gwaii. A fundamental understanding of these crucial processes is critical for sound management under the concept of adaptive management (Walters 1997). Biological processes throughout this region are shaped by the isolation of islands and dynamic biogeographic history that included multiple waves of colonization along the coast following the retreat of the glaciers. Moreover, recognition of the peculiar processes associated with island life provides a powerful framework for devising effective strategies for management agencies such as the Tongass National Forest, the largest national forest in the United States.

DEEPER HISTORY: REGIONAL COLONIZATION FOLLOWING DEGLACIATION

The geologic history of the Pacific Northwest coast was dynamic and heavily influenced in the Quaternary by large-scale climatic change (Heusser 1960; Mann and Hamilton 1995; Hebda and Whitlock 1997; Barrie *et al.* 2005). During the Late Pleistocene (fig. 2.02), most of southeast Alaska and coastal British Columbia was covered by the Cordilleran ice sheet (Mann and Hamilton 1995; Carrara *et al.* 2003, 2007; Mandryk *et al.* 2001; Hetherington *et al.* 2004; Baichtal *et al.* 2008; Baichtal and Carlson 2010), hence glacial dynamics played a primary role in shaping historic and contemporary biotic communities (Klein 1965; Heaton *et al.* 1996; Conroy *et al.* 1999; Burles *et al.* 2004; Reimchen and Byun 2005; Wigen 2005; MacDonald and Cook 2007). Two ice-adapted species (arctic fox, *Vulpes lagopus*; and ringed seal, *Pusa hispida*), present during the height of the last glaciation (28–16.5 thousand years BP), no longer occur in the region (Heaton and Grady 2003). Phylogeographic studies, combined with insight from fossils recovered from the extensive karst system on Prince of Wales Island (and more recently on Haida Gwaii: Ramsey *et al.* 2004; Fedje and Mathewes 2005; Erlandson *et al.* 2008), however, suggest that other terrestrial species also persisted nearby in coastal ice-free refugia during glacial advances. About 12,000 years ago, ice retreated eastward into the coastal mountains, exposing a clean slate that allowed widespread recolonization of the deglaciated territory during the Holocene. Species colonized from a variety of distinct sources and they arrived at different times. Species distributions, community composition, and other fundamental attributes of this highly productive coastal region, therefore, have been dynamic due to extensive glacial cycling and associated sea level changes that characterized the last several thousand millennia (fig. 2.02).

Because some of the oldest lineages occur on the more western islands of the Alexander Archipelago and on Haida Gwaii, extant organisms may have persisted in glacial refugia on the outer continental shelf, or they may have first colonized this region of the coast along a tidewater route. Strong molecular genetic signals suggest that a suite of periglacial species persisted in coastal refugia during the last glacial maximum and then recolonized outer islands early in the Holocene. Later in the Holocene, corridors through the Coast Mountains were opened from the interior and allowed the westward colonization of the coastal mainland and eventually the

FIGURE 2.02. Map of the extent of glaciation in northwestern North America at the last glacial maximum (LGM), about 20,000 years before the present. Lowered sea levels create the possibility of the existence of glacial refugia to the west of contemporary exposed land.

nearshore islands of the archipelagos. This backdrop is essential to interpreting distributions and variable rates and patterns of evolutionary change so far uncovered in the region (Cook *et al.* 2006, MacDonald and Cook 2007). Nelson (1887) recognized that the biota of this portion of the North Pacific coast was distinctive from the remainder of North America (his "Sitkan District"). Later, Swarth (1936) and Klein (1965) provided overviews of vertebrate diversity and concluded that the region was isolated from the remainder of North America by the coastal range. A series of subsequent studies have corroborated that view.

PATTERNS OF INTRASPECIFIC VARIATION PROVIDE INSIGHT INTO MANAGEMENT UNITS

The DNA of contemporary coastal organisms (i.e., genetic structure of populations) has a tell-tale signature of this dynamic biogeographic his-

tory and provides a rigorous framework for implementing future efforts to sustain ecosystem processes and conserve the biota. Molecular genetic investigations of a variety of mammals have identified common patterns of geographic differentiation and genetic structure of populations that reflect common responses to large scale geologic events in southeast Alaska (Cook *et al.* 2006). Some species originated from a single refugium and expanded into southeast Alaska (e.g., Lessa *et al.* 2003; Frances 2008). Single origin species (summarized in Cook *et al.* 2006) arrived from the northern Beringian refugium (e.g., moose, *Alces alces gigas*; wolverine, *Gulo gulo*; northern red-backed vole, *Myodes rutilus*; root vole, *Microtus oeconomus*; arctic ground squirrel, *Spermophilus parryii*; collared pika, *Ochotona collaris*; and brown lemming, *Lemmus trimucronatus*); from a southern coastal refugium (e.g., cinereus shrew, *Sorex cinereus*; northwestern deermouse, *Peromyscus keeni*; wolf, *Canis lupus*; Sitka black-tailed deer, *Odocoileus hemionus sitkensis*; mountain goat, *Oreamnos americanus*; and Pacific marten, *Martes caurina*); or from eastern (continental) refugia located south of the Laurentide ice sheet (e.g., northern flying squirrel, *Glaucomys sabrinus*; southern red-backed vole, *Myodes gapperi*; and meadow vole, *Microtus pennsylvanicus*).

In contrast, several mammal species now found in southeast Alaska are represented by more than one lineage. Multiple lineages indicate multiple colonization events from independent source populations or refugia. Substantial genetic divergence within several species (e.g., dusky shrew, *Sorex monticolus*; long-tailed vole, *Microtus longicaudus*; and black bear, *Ursus americanus*) indicates that diversification was initiated in the mid-Pleistocene with differences persisting through repeated cycles of glacial advance and retreat. Most of these species are represented by lineages originating from a combination of southern coastal (termed Coastal) and southern continental (Continental) refugia (Cook *et al.* 2001, 2006).

OVERVIEW OF ISLAND BIOGEOGRAPHY

Island biogeography, essentially the study of the distribution and dynamics of insular species, provides a theoretical and operational foundation for the design of research and conservation plans in southeast Alaska and along the North Pacific coast of British Columbia. Over the past 150 years, studies of island systems played a fundamental role in the formation of ecological and evolutionary theory (Darwin 1859; Wallace 1902; MacAr-

thur and Wilson 1967), including advances in our understanding of processes related to colonization, extinction, and speciation. Today, new technology combined with intensive studies of organisms on a series of island systems worldwide are proving detailed and often novel views of the history and contemporary dynamics of species (Ricklefs and Bermingham 2008; Trewick and Cowie 2008). To date, the Alexander Archipelago has played a limited role in the published literature on evolution, ecology, and conservation biology, although this system is one of the largest temperate archipelagos worldwide, with 7 of the 15 largest islands in US possession. In this regard, Haida Gwaii has fared better with a series of seminal papers and books focused on the fauna and flora of that archipelago (e.g., Scudder and Gessler 1989; Burles *et al.* 2004; Fedje and Mathewes 2005). Thus far, however, only a small proportion of the more than 7,000 islands that constitute the four major island groups of southeast Alaska and coastal British Columbia has ever been rigorously surveyed. Despite this, a large number of endemic taxa of both animals and plants have been described from these islands (e.g., 50 mammals), suggesting that substantial new diversity will be discovered in the future. There remains much to be learned about the region, and it could emerge as a model system for testing fundamental hypotheses related to environmental and biological change.

CONTINENTAL ARCHIPELAGO?

Alfred Russel Wallace (1902) identified a fundamental dichotomy between islands based on their geologic origin by proposing that archipelagos could be classified as either oceanic or continental (also known as land bridge). Oceanic islands are formed over oceanic plates and have never been connected to continental landmasses. These islands typically are devoid of life at the outset but gradually accumulate species from distant mainland source populations. In contrast, continental (land bridge) islands are found on the continental shelf and were directly connected to the mainland during the Pleistocene ice advances when sea level dropped as much as 130 m (Whittaker and Fernández-Palacios 2007).

Because of their proximity to the mainland, land bridge islands are more heavily dominated by repeated colonization than are distant island archipelagoes, resulting in a relatively high diversity of lineages. Some groups also have diversified within the islands, providing an opportunity to study both intrinsic and extrinsic processes responsible for the buildup of diversity.

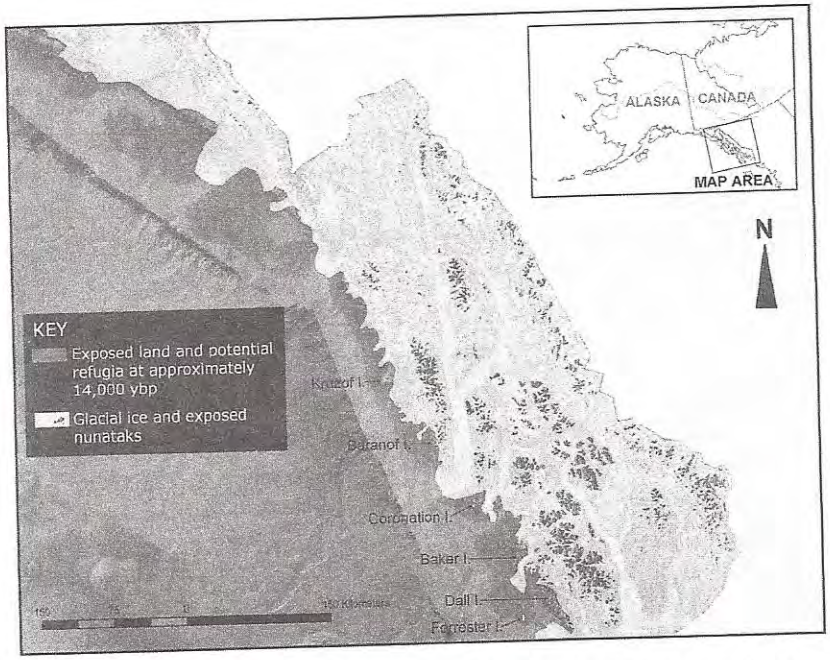

FIGURE 2.03. Predicted extent of glacial advance and ice-free refugia in southeast Alaska about 14,000 years before the present. Map is modified from Carrara *et al.* (2007), Baichtal and Carlson (2010), and James Baichtal (unpublished data).

Although technically considered land bridge islands, the Alexander Archipelago differs in two fundamental ways: the Clean Slate Hypothesis and the Coastal Refugia Hypothesis. These can only be understood within the context of the geologic history of the Pacific Northwest coast.

During the last glacial maximum about 18,000 years ago, cordilleran ice overrode most of this coastal region, essentially erasing terrestrial biota except in glacial refugia. Following the retreat of the Cordilleran ice sheet, the region was essentially a clean slate with regard to terrestrial life, so the contemporary biota is derived from organisms that colonized the North Pacific coastal rainforest from elsewhere. This clean slate scenario suggests that subsequent assembly of species on the islands of the Alexander Archipelago more closely resembles dynamics on oceanic islands rather than land bridge islands (Whittaker and Fernández-Palacios 2007). This cataclysmic event, which was likely repeated on multiple occasions through the Pleistocene, did not encompass all available terrestrial areas (fig. 2.03), since falling sea level exposed substantial sections of the continental shelf

to the west (Carrara *et al.* 2007). Therefore, hypothesized sources for colonizers included not only multiple mainland locations, but also coastal refugia to the west of the archipelago. These distinct sources likely supported populations that were diverged from the mainland sources.

EVOLUTIONARY EXPECTATIONS OF ISLAND LIFE

Potential for Rapid Morphological Evolution

Millien (2006) showed that rates of morphological evolution are much higher for island mammal populations than for mainland populations, with significant changes evident over time scales as short as a few decades. Cardini *et al.* (2007) further investigated Vancouver Island marmots (*Marmota vancouverensis*) and reported limited genetic differentiation from mainland populations of hoary marmots (*M. caligata*), but substantial morphological differences. They concluded that the Vancouver Island marmot is a morphospecies; essentially the other side of the coin of the many "cryptic" species that have been recently uncovered via molecular genetic studies (Bickford *et al.* 2007). Cardini *et al.* (2007) suggested rapid morphologic variation of this island population (Millien 2006), noting that insular populations may be distinctive due not only to temporally deep separation, but also due to peculiarities that may arise rapidly from island life (e.g., changes in size, fecundity, population density, behavior, survival, aggressiveness, and so forth). Rapid morphological change on a Pacific Northwest island drives home the point that each insular population is potentially distinctive. Differences may arise not only from deeper temporal evolution, but also from changes that occur at ecological time scales. Similarly, dwarfism of the now-extinct Dawson caribou (*Rangifer tarandus dawsoni*) of Haida Gwaii is hypothesized to have occurred within less than 1,500 years (Wigen 2005).

Potential for Rapid Molecular Evolution

Lucid and Cook (2004) assessed levels of genetic variability across 23 island populations of the northwestern deermouse in the Alexander Archipelago. Populations of this ubiquitous species showed highest levels of variability on the mainland and on larger islands while lowest levels of variability were recorded on the smallest and most distant islands. Other island endemics show extremely low levels of variability (e.g., Prince of

Wales Island flying squirrel and the western pine marten). Hence, insular faunas show not only reduced species richness; individual species also have a characteristic signature of lower genetic variability which may increase the probability of local extirpation (Frankham 1995). Lower genetic variability decreases the ability of organisms to meet novel environmental challenges (Sgro *et al.* 2010), but the relationship between genetic variability and the process of extinction is complex (Aguilar *et al.* 2004). The few mammals examined on the Alexander Archipelago to date exhibit low genetic variability (e.g., Bidlack and Cook 2002; Small *et al.* 2003; Cook *et al.* 2006; Dawson 2008), so the generality of this trend and its implications for persistence of other North Pacific insular organisms should be examined. Populations with limited ability to disperse with changing conditions, such as island populations, are at much higher risk of loss (Thomas *et al.* 2004).

ENDEMISM ON ISLANDS IN SOUTHEAST ALASKA AND COASTAL BRITISH COLUMBIA

There are likely a number of parallels between the Alexander Archipelago and other glaciated nearshore archipelagos of midlatitudes, such as Haida Gwaii (Reimchen and Byun 2005), the Japanese Archipelago (Millien-Parra and Jaeger 1999; Conservation International 2012), and the British Isles (Vincent 1990). Natural resource managers should identify collaborative projects with research and management initiatives underway in other archipelagos to meet the peculiar challenges associated with managing insular biotas (e.g., Gaston *et al.* 2008; BC Integrated Land Management Bureau 2007, 2009). For example, limited effort has focused on threats to the endemic fauna and flora of the Pacific Northwest coast (Cook *et al.* 2001), primarily because the status of these insular biotas remains poorly documented (Cook and MacDonald 2001; Nagorsen 2004). Few intensive surveys of island organisms have been completed along the coast of British Columbia (Nagorsen 1994, 2004; Shank 1999). For those groups that have been surveyed, such as small mammals (McCabe and Cowan 1945, Foster 1965) and birds (Cannings 1998), there is a tremendous opportunity for specimen-based resurveys to provide an extraordinary assessment of change in these ecosystems on a decadal basis. Much of British Columbia's coastal region has never been properly surveyed for amphibians and reptiles (Matsuda *et al.* 2006). Knowledge of the vascular plants for many

TABLE 2.1. Extinctions of recent mammals from islands in southeast Alaska and coastal British Columbia (AA, Alexander Archipelago; HG, Haida Gwaii; VAN, Vancouver Island; LGM, last glacial maximum).

	ISLANDS	TIME PERIOD PRESENT	REMARKS
RODENTS			
hoary marmot	AA	pre-LGM→LGM[1]	
brown lemming	AA	pre-LGM[1]	
western heather vole	AA	pre-LGM[1]	(*Phenacomys intermedius*)
CARNIVORES			
arctic fox	AA	pre-LGM→ early postglacial[1]	
red fox	AA	LGM[1]	(*Vulpes vulpes*)
brown bear	AA	pre-LGM; early postglacial→early Holocene[1]	
	HG	early postglacial[1]	
Pacific marten	AA	unknown	former presence on Island inferred from genetic signature of parasite[2]
cf. wolverine	AA	early postglacial	based on molar fragment from a cave on Prince of Wales Island[3]

islands also remains limited or nonexistent (Roemer 1994). Nonvascular plants—the bryophytes (Schofield 1994; Ryan 1996), lichens (Goward 1994), and fungi (Redhead 1994)—remain poorly documented. Terrestrial and freshwater invertebrates, perhaps the richest of the region's biodiversity, are the least known (Cannings 1994; Scudder 1996; Forsyth 2004; Ovaska *et al.* 2010).

Studies based on fossil or molecular genetic evidence have documented extinctions on islands in southeast Alaska and coastal British Columbia (table 2.1). Other studies provided evidence of high endangerment of insular species (e.g., Vancouver Island marmot, <50 individuals; see Bryant 2002). In British Columbia, two-thirds of 74 endemic taxa are restricted to

TABLE 2.1. *(continued)*

	ISLANDS	TIME PERIOD PRESENT	REMARKS
UNGULATES			
Caribou	AA	pre-LGM; early post-glacial→early Holocene[3]	
	HG	pre-LGM; present up until early 1900s early postglacial→recent[1]	
mountain goat	VAN	early postglacial[4]	
	AA	pre-LGM	horn core and molars from Prince of Wales Island tentatively identified as either this species or saiga[1]

1 Fedji *et al.* 2004
2 Koehler 2006; Hoberg *et al.*, in press
3 Heaton and Grady 2003
4 Nagorsen and Keddie 2000

islands, among which over half (56%) are forest dwelling (Bunnell *et al.* 2005). Less than 45% of endemic taxa, however, are listed provincially or nationally (4 of 6 endemic plants are listed; 29 of 35 endemic mammals are not listed) (Bunnell 2005). Management plans in British Columbia and southeast Alaska (e.g., USDA Forest Service 2008a; BC Integrated Land Management Bureau 2007, 2009) still are not tuned to the special attributes of island life (Paquet *et al.* 2004; DellaSala, Moola, *et al.* 2011).

TEMPORAL SCALE OF ENDEMISM

The Coastal Refugia Hypothesis has been accumulating support over the past four decades (Foster 1965; Rogers *et al.* 1991; Heaton *et al.* 1996; Byun *et al.* 1999), and two summary volumes have focused on this issue as it relates to the natural history of Haida Gwaii (Scudder and Gessler 1989; Fedje and Mathewes 2005). A mammalian example that potentially reflects the signature of persistence in a coastal refuge is the discovery of three distinctive lineages of ermine that were detected with molecular techniques in southeast Alaska and adjoining British Columbia (fig. 2.04). Two of these

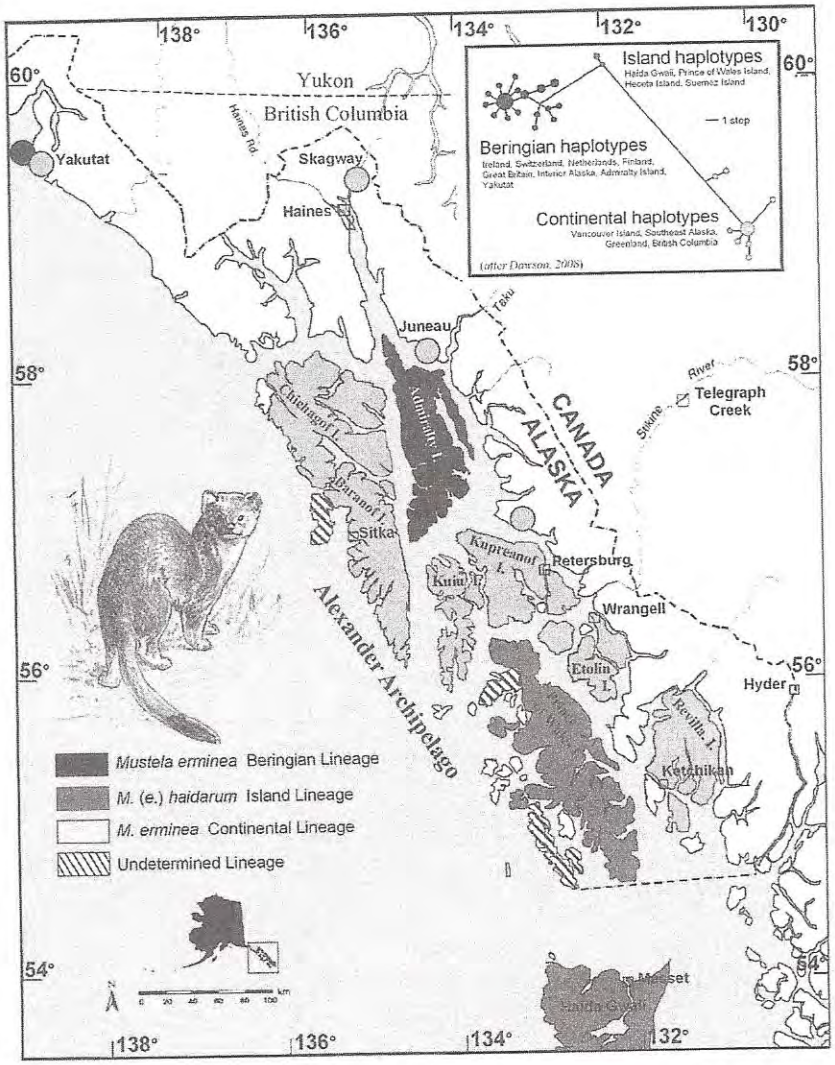

FIGURE 2.04. Three clades of ermine have been identified in southeast Alaska. The Beringian clade is widely distributed from Europe, through Asia, and into far northwestern North America (and Admiralty Island). The Continental clade is distributed elsewhere in northern and temperate North America with the exception of Haida Gwaii and the Prince of Wales archipelagos, where the third (Island) clade is endemic (Fleming and Cook 2002; Dawson 2008).

lineages also occur widely outside of the coastal region, whereas the third lineage is restricted to Prince of Wales Island, nearby outer islands, and on Haida Gwaii (Fleming and Cook 2002). This island-restricted ermine represents a periglacial relict or paleoendemic that persisted in the region since before the last glacial maximum (LGM). Examples such as this have led to a significant reassessment of the temporal horizon over which elements of this coastal fauna diversified. These paleoendemics also have important implications for the human colonization of North America along a coastal route. Because the LGM apparently did not wipe the slate completely clean along the entire coast, the existence of Pacific Northwest coastal refugia raises the possibility of a suite of other highly divergent coastal taxa that have not yet been detected, and these may require special management status under laws such as the Endangered Species Act.

A review of endemic mammals for North America north of Mexico concluded that the Pacific Northwest coast supports high levels of endemism (Cook and MacDonald 2001; Reimchen and Byun 2005). Subsequent molecular phylogeographic studies largely supported this finding (Cook *et al.* 2006). Indeed, endemic lineages of the northwestern deermouse, northern flying squirrel (Bidlack and Cook 2002), spruce grouse (*Falcipennis canadensis isleibi*) (Dickerman and Gustafson 1996; Barry and Tallmon 2010), chum salmon (*Oncorhynchus keta*) (Kondzela *et al.* 1994), and subalpine fir (*Abies lasiocarpa*) (Carrara *et al.* 2003) have also been identified on the Prince of Wales Archipelago. Likely numerous other endemic organisms occur there that have yet to be discovered, but the baseline samples necessary to complete those analyses are lacking.

ISLAND COMMUNITIES

Consistent with classic island biogeography theory (MacArthur and Wilson 1967), size and distance from the mainland have played a prominent role in shaping diversity across the Alexander Archipelago (Table 2.2). Larger islands (e.g., Revillagigedo, on which Ketchikan is located) close to the mainland have higher species richness, while small distant islands (e.g., Forrester, to the west of Prince of Wales) have lower species richness (MacDonald and Cook 1996). Nonetheless, when the factors of size, distance, and greatest elevation for each of 20 islands are regressed against species richness, only island distance is significant (fig. 2.05). Conroy *et al.* (1999) used species lists for each island to explore factors responsible for

TABLE 2.2. List of 24 major islands in the Alexander Archipelago. Map numbers refer to figure 2.01. Island area, distance to the mainland, maximum elevation, and species richness of mammals are reported for each island.

MAP NUMBER	ISLAND	AREA (HA)	DISTANCE TO MAINLAND (KM)	MAXIMUM ELEVATION (M)	NUMBER OF NATIVE SPECIES
1	Admiralty	431,309	5	1,418	14
2	Anguilla	1,210	9	198	8
3	Baker	13,512	13	697	9
4	Baranof	424,016	6	1,644	9
5	Chichagof	545,317	5	1,192	9
6	Coronation	9,120	11	598	7
7	Dall	65,869	7	949	12
8	Douglas	20,274	1	1,018	20
9	Duke	15,538	5	503	8
10	Etolin	88,995	2	1,135	18
11	Forrester	1,013	34	409	3
12	Gravina	23,307	3	833	14
13	Heceta	18,916	9	761	11
14	Kosciusko	48,259	7	947	12
15	Kruzof	44,680	7	976	9
16	Kuiu	193,455	3	1,080	19
17	Kupreanof	282,415	2	1,091	20
18	Mitkof	54,753	1	1,012	26
19	Prince of Wales	578,202	6	1,219	16
20	Revillagigedo	302,659	2	1,401	24
21	Tuxekan	8,523	7	320	11
22	Warren	5,067	11	710	8
23	Wrangell	56,948	1	991	26
24	Zarembo	47,263	5	777	13

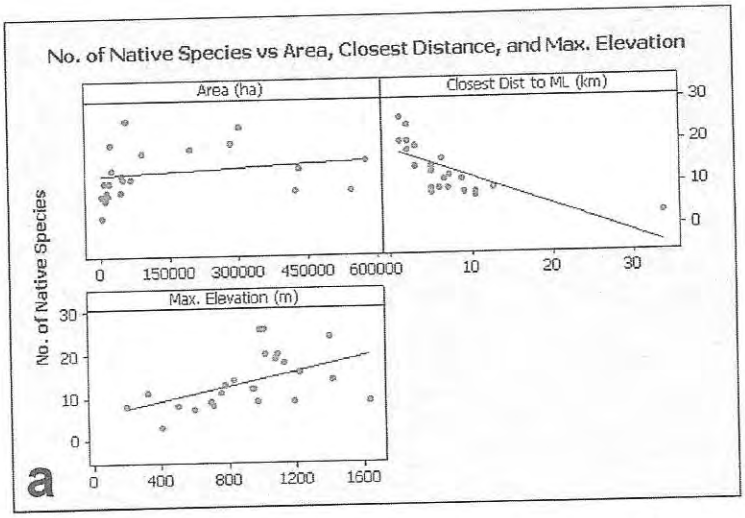

FIGURE 2.05. Plots of species richness for mammals by island area (ha), island maximum elevation (m), and closest distance to the mainland (km) for the 24 islands reported in table 2.1. Only island distance from the mainland is a significant ($p < 0.05$) factor in explaining differences in species richness. Island distance remains the only significant factor when the outlier (Forrester Island) is removed from the regression analyses (not shown).

community assembly of mammals across the archipelago and showed that mammalian communities on smaller islands were nested within communities on larger islands. They concluded that, across all species, colonization ability (not extinction probability) has been the most important determinant of current community composition on islands, a result that is consistent with the history of heavy glaciations. That study, however, was unable to consider endemism that has since been documented based on molecular analyses. Reanalysis based on estimates of lineage richness rather than species richness (unpublished data) indicates that endemic lineages are not nested; those older forms instead appear to be structured by extinction processes (i.e., island size rather than island distance is important in explaining their presence).

Despite the overall nested pattern found in the archipelago (Conroy *et al.* 1999), species assemblages vary spatially (i.e., from island to island) and temporally (Heaton and Grady 2003). For example, cinereus shrews

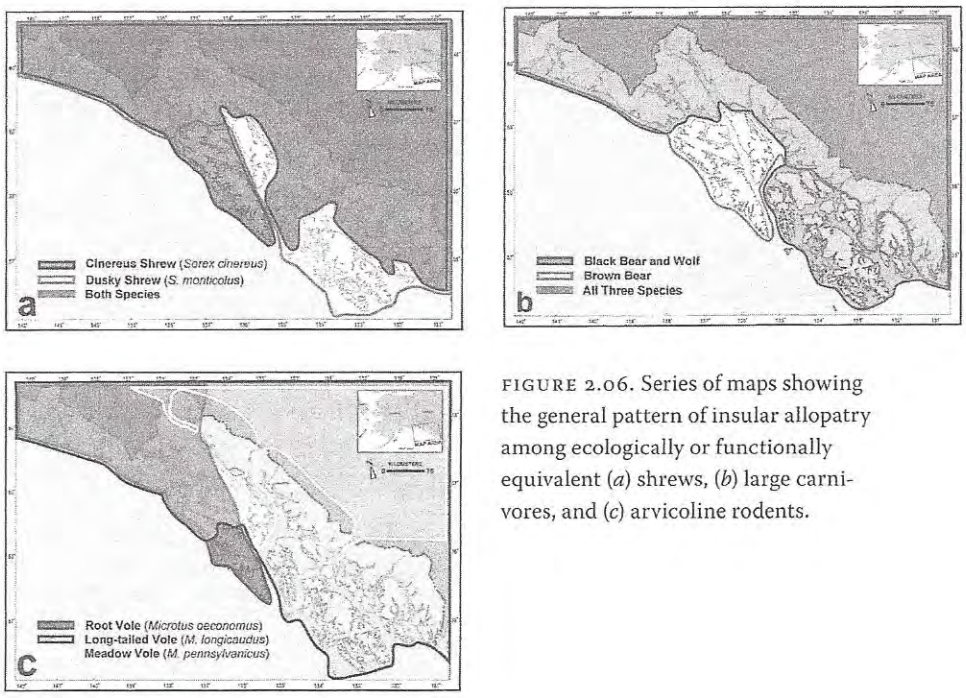

FIGURE 2.06. Series of maps showing the general pattern of insular allopatry among ecologically or functionally equivalent (*a*) shrews, (*b*) large carnivores, and (*c*) arvicoline rodents.

are found on Baranof and Chichagof islands, but not Admiralty or Prince of Wales islands (fig. 2.06a), while dusky shrews show the opposite pattern of occurrence. Both species are sympatric on the mainland, on nearshore islands (e.g., Revillagigedo, Wrangell) and on the peninsula-like islands of Mitkof, Kupreanof, and Kuiu (fig. 2.06b). Historically, brown and black bears coexisted on Prince of Wales Island and Haida Gwaii, but in the last 10,000 years brown bears have ceased to occupy these islands (Heaton and Grady 2003; Wigen 2005). Now, black bears (and wolves) occupy islands south of Frederick Sound, whereas brown bears (and no wolves) occur on northern islands of the Alexander Archipelago. All three of these top predators are sympatric on the mainland. Patterns of exclusion also are seen in rodents of the genus *Microtus* (voles) (fig. 2.06c).

INTERISLAND CONNECTIVITY

MacDonald and Cook (2007) identified the need to document linkages or connections between islands that facilitate metapopulation structure and

movement of organisms within the Alexander Archipelago. Linkage sites should be priorities for conservation because gene flow is a critical process with benefits ranging from persistence of small populations to introduction of traits necessary to adapt to fluctuating environments (Templeton *et al.* 2001; Manel *et al.* 2003). These hypothesized connections among islands (MacDonald and Cook 2007) should be assessed, and likely they could be improved as they fail to account for potentially crucial factors, such as oceanic currents or similarity of habitat on either end of the crossing. Reconstruction of sea level fluctuations and the related influence of underwater topography on historic connections among islands also should factor into our understanding of linkages. Sites that may facilitate connectivity among islands should be identified and protected by establishing disturbance buffers and associated wildlife corridors for movement of organisms between islands. Proposed connections should be viewed as hypotheses that can be tested; they also serve to raise awareness of the complex interactions and metapopulation dynamics that may occur among islands. Molecular studies across a number of species have provided preliminary tests of linkages (Cook *et al.* 2006; MacDonald and Cook 2007). The endemic Prince of Wales Island flying squirrel (*Glaucomys sabrinus griseifrons*), for example, has diverged from mainland squirrels (Demboski *et al.* 1998; Bidlack and Cook 2001), but not from other populations of squirrels within the Prince of Wales Archipelago (e.g., Prince of Wales, Heceta, Suemez, Dall; see Bidlack and Cook 2002). Such a pattern, also shared in northwestern deermice, ermine, and potentially southern and northern dusky shrews (Demboski and Cook 2001), suggests high historic or contemporary levels of connectivity among islands of the Prince of Wales Archipelago (Fleming and Cook 2002; Lucid and Cook 2004).

ISLANDS WITHIN ISLANDS

Temporal and spatial scale must be considered as management actions are initiated that further fragment habitats. In addition to the dynamic geologic history and natural fragmentation of the archipelago, industrial logging has recently produced a patchwork of habitats within many islands. Maintaining connectivity or corridors among patches to help mitigate important processes such as dispersal and gene flow should be a goal of management plans along the Pacific Northwest coast. Nevertheless,

hypotheses of how landscape corridors will reduce the negative effects of fragmentation are largely untested (Damschen *et al.* 2006; Dixon *et al.* 2006) and predictions are especially uncertain in the face of climate change (Thomas *et al.* 2004). Highly vagile birds, carnivores, and ungulates may disperse among islands more easily than flightless arthropods and small mammals. Preliminary molecular perspectives on mammals (Cook *et al.* 2006) indicate differential connectivity across taxonomic groups. With regard to assessing barriers throughout the archipelago, ubiquitous species are good candidates for identifying overall levels of connectivity, while less vagile species with more limited distributions or narrow habitat requirements may provide finer-scale views of connectivity.

Faunal exchange among islands may be asymmetrical, with some islands acting as source populations while others act as sink populations. Kuiu Island has had high black bear densities (e.g., 1.5 bears/km^2), and a larger number of migrant bears per generation move from Kuiu to Kupreanof Island than vice versa (Peacock 2004; Peacock *et al.* 2007). Unequal rates of immigration and emigration illustrate the distinctive microevolutionary dynamics that populations in close proximity, but on different islands, may be experiencing, as is the case with the Kermode bear (*Ursus americanus kermodei*) among the islands of the northern British Columbia coast (Klinka and Reimchen 2009). Peacock (2004) also recorded little exchange of black bears between Kuiu and Prince of Wales islands. With heavy harvest pressures on these two islands, these new perspectives on connectivity in black bears provide an important framework for maintaining viable populations. Furthermore, the consequences of habitat fragmentation within islands on complex landscapes such as that found on Kuiu Island has been little studied.

CONSERVATION OF ISLAND ECOSYSTEMS

Logging, mining, human sprawl, tourism, hunting and trapping, and introductions of exotic and domesticated species present significant challenges to the maintenance of viable and well-distributed wildlife populations. Each of these impacts may be magnified in insular endemics (Steadman 2006; Whittaker and Fernández-Palacios 2007). All are present, to varying degrees, in southeast Alaska and coastal British Columbia. Several islands have experienced significant impacts from one or more of these threats

(e.g., central Prince of Wales, Long, Revillagigedo, Kuiu, Heceta), and the future will likely see a shift from one set of impacts to others. For example, logging has substantially converted old-growth forest habitat on northern Chichagof and central Prince of Wales islands to even-aged second-growth plantations. Increased accessibility and forest fragmentation resulting from the associated high-density network of roads has significantly impacted some wildlife species (Person *et al.* 1996). In most cases, these challenges have not been rigorously evaluated or monitored. Now added to this mix of potential threats is anthropogenic climate change, which is predicted to substantially increase extinction risk for populations and species with limited ability to disperse (Thomas *et al.* 2004), such as many of those found on islands. Inability of insular species to respond (e.g., disperse to new islands) as conditions change raises the possibility that anthropogenic climate change could become a major cause of extirpation of species from islands in southeast Alaska or even extinction for the insular endemic forms.

Inventory programs should be tied to permanent archives (i.e., museums) and research laboratories that would be interested in investigations of selected species. An example is the recent series of papers on northwestern martens that established a molecular framework for understanding geographic variation in these economically important furbearers. This framework, if it had been available earlier, would have demonstrated the folly of the introduction of mainland marten (*Martes americana)* onto several islands in the Alexander Archipelago where the other species (*M. caurina*) may have been endemic (Hoberg *et al.*, in press). Active management of marten and marten habitat over the last 50 years has likely hastened the extirpation of Pacific marten (*M. caurina*) on several islands in the region through introduction of the exotic sister species combined with a substantial reduction in old-growth forest by industrial-scale logging on several islands (e.g., Kuiu Island), but the overall impact remains unknown (MacDonald and Cook 1996, 2007). Koehler *et al.* (2009) uncovered significant phylogeographic structure in a nematode that is parasitic on marten across a number of locations in southeast Alaska. Molecular genetic variation in these worms from marten on Chichagof correspond to two independent introductions of American marten (*M. americana*) from the southeast mainland and Anchorage area, as noted by historic records of introductions of this species. In addition, Koehler *et al.* (2009) found an

endemic lineage of worm on Chichagof that is more closely allied with worms from Pacific marten on Admiralty Island and Haida Gwaii. This signal from the parasite is consistent with the recent extirpation of *M. caurina* on Chichagof Island, perhaps hastened by the introduction of *M. americana* (Hoberg *et al.*, in press).

MANAGEMENT RECOMMENDATIONS AND RESEARCH NEEDS

Preliminary investigations of mammals clearly indicate inadequacies in our current understanding of diversity. Outdated taxonomic designations still are used for characterizing diversity and fail to identify forms unique to the region (i.e., endemics). These investigations also show that past logging efforts are strongly correlated with the occurrence of hot spots of endemism (Cook *et al.* 2006). In the future, a well-annotated set of natural history collections could provide the basis for assessing the status and distribution of Pacific Northwest coastal endemics. Assessing biotic change begins with modern inventory studies that are conducted hand in hand with long-term monitoring programs in an effort to develop rigorous databases related to the status of Tongass wildlife and ecosystems. These data should be accessible via the internet and linked to other online databases, GIS applications, research programs, and management initiatives (fig. 2.07). A baseline for wildlife and their associated parasites, especially the pathogens that may impact wildlife (and human) health, is essential to evaluate emerging threats on federally managed lands. The relationship between climate change and emerging pathogens, for example, has recently been explored in high-latitude ungulate populations (Kutz *et al.* 2005; Jenkins *et al.* 2006; Hoberg *et al.* 2008). Because these large mammals play a critical role in subsistence economies, efforts to monitor their health are essential to persistence of northern indigenous cultures. The list of potential studies is primarily limited by the availability of specimens and the creativity of investigators.

WHAT IS NEEDED NOW?

1. Interagency, intergovernmental (i.e., for southeast Alaska, the USDA Forest Service, US Fish and Wildlife Service, Alaska Department of Fish and Game, Native Sovereign Nations, National Marine Fisheries,

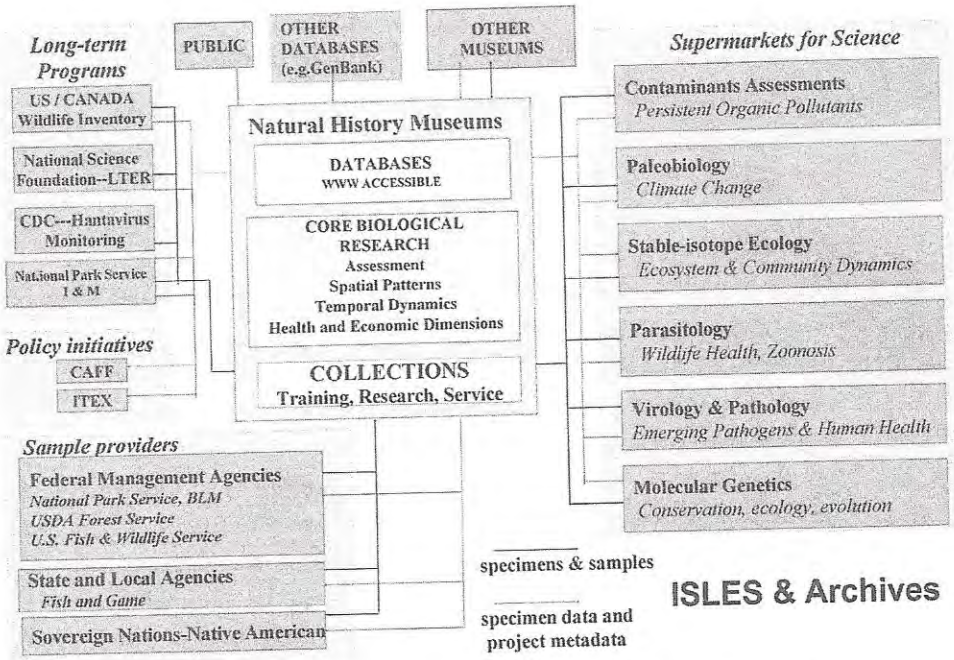

FIGURE 2.07. Schematic of a multiagency effort on the Tongass National Forest called Island Surveys to Learn about Endemic Species (ISLES). Examples of other collaborators or users of the archival observatory include Long Term Ecological Research (LTER) sites, Conservation of Arctic Fauna and Flora (CAFF), International Tundra Experiment (ITEX), Bureau of Land Management (BLM), and Centers for Disease Control (CDC). http://www.msb.unm.edu/mammals/ISLES_website_final_2009/isles_home.html.

National Park Service, universities, and natural history museums), and new international agreements and specialized working groups should focus on establishing a comprehensive management plan aimed at preserving the endemic biota and natural ecosystems of areas such as the Alexander Archipelago and Haida Gwaii. Substantial physical and personnel infrastructure already exists, but coordinated action on island issues is lacking. Because agencies are mandated to manage resources wisely, this infrastructure could be used to implement island-centered management plans. Resource managers are grappling with a number of issues in archipelagos (e.g., Golumbia 2000; Sherley 2000; McNeely *et al.* 2001; Wittenberg and Cock 2001; BC Integrated Land Management Bureau 2007), but they have yet to develop guiding principles that fully address management goals and objectives unique

to island systems. An international conference focused on island management along the North Pacific coast would be one productive outcome of such a working group.

2. Rigorous, integrated inventories should be conducted on all islands over 500 ha. There is a pressing need for specimen-based, all-taxa inventories of these islands. Work on Haida Gwaii provides examples of taxonomic groups that could also be the focus of more intensive studies in southeast Alaska. Prior work along the Pacific Northwest coast opens productive opportunities to engage international collaborators and expand seminal studies on mosses (Schofield 1989), flowering plants (Calder and Taylor 1968), ground dwelling beetles (Kavanaugh 1992; Clarke *et al.* 2007), birds (Martin *et al.* 1995; Topp and Winker 2008), and fish (Moodie and Reimchen 1973; Reimchen 1989, 1994). Spatially extensive and site-intensive archives of insular organisms are critical to stimulating investigations that will form the basis for careful management (Chapman 2005). Sophisticated and varied technologies (e.g., stable isotopes, molecular genetics, biotoxin assays) have transformed wildlife management recently and can be employed through truly integrated investigations (Hoberg *et al.* 2003; Reimchen *et al.* 2003; Wilson *et al.* 2009). Specimen archives are fundamental to the development of multidisciplinary views of diversity, because all of these studies can be tied together through individual voucher specimens.

3. Monitoring efforts should be designed to provide information necessary to assess environmental change. Elements to be monitored should include species that (a) are most susceptible to impacts such as fragmentation (e.g., endemics), (b) are introduced and exotic to the region, (c) span multiple trophic levels, (d) have complex lifecycles, (e) are sensitive to climate change and have limited ability to move, or (f) are key to human subsistence.

4. Partnerships to investigate and monitor the region should be actively sought and then built, perhaps modeled after the long-running and highly successful program by the Laskeek Bay Conservation Society on Haida Gwaii (http://www.laskeekbay.org). Subsistence hunters, trappers, resource agencies, rural K–12 schools, citizen scientists, and academia are examples of potential partners that could have powerful and last-

ing impacts on resource management in southeast Alaska and coastal British Columbia.

5. Introduction of exotic species should be thoroughly regulated, since there are a growing number of introduced species in the region (MacDonald and Cook 2007; Gaston *et al.* 2008). This is a looming issue for resource managers and one that already has been experienced in southeast Alaska (Heutte and Bella 2003; Schrader and Hennon 2005; McClory and Gotthardt 2008) and Haida Gwaii (Golumbia 2000). The best way to manage introduced species is to implement laws that will help to slow their entry. If exotics become established, detect and eradicate them quickly, or manage them at acceptably low densities to prevent significant negative effects (Simberloff 2002).

Given the complexity of issues facing resource managers working in the extensive island systems of southeast Alaska and coastal British Columbia, a diverse set of approaches is needed to effectively administer wildlife management plans and monitor potential threats to native biotas, including strong, coordinated efforts that include local communities and that cut across agencies and international boundaries. Several steps can be taken immediately that will lessen costs associated with impacts and facilitate swift responses to perturbations in the future, but lack of available baseline materials has become the primary limitation to understanding the magnitude and complexity of change and ultimately applying effective new technologies. Now is the time to develop proactive strategies, rather than waiting until our efforts are entirely consumed by restoration actions.

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Marbled Murrelet
(Brachyramphus marmoratus)

Paul Cotter and Matthew Kirchhoff

The marbled murrelet, *Brachyramphus marmoratus*, is a small seabird (length 24-25 cm; mass 190-270 g; Nelson 1997) belonging to the auk family (Alcidae). It is found coastally from northern California to the western Aleutian Islands, although the vast majority of birds occupy the central portions of this range in Southeast Alaska (Nelson 1997, Agler et al. 1998). The bird's marbled gray-brown breeding plumage contrasts sharply with its distinct black and white winter (basic) plumage (Fig 1). Like other auks, marbled murrelets are expert divers and underwater swimmers. They mostly feed on small schooling fish, such as Pacific herring *(Clupea barengus)*, capelin *(Mallotus villosus)*, and Pacific sand lance *(Ammodytes hexapterus)* (Nelson 1997). Like other seabirds, the marbled murrelet visits land only during the breeding season. Unlike nearly all other seabirds, however, this species nests as solitary pairs, most often in mature conifers within 30 km from the coast (Burger 2004). Little is known about the general ecology, life history, and habitat requirements of this species in Southeastern Alaska (Southeast) (DeGange 1996). The majority of breeding season information comes from the Pacific Northwest and British Columbia—areas that have witnessed significant murrelet population declines in recent decades (Nelson 1997, Burger 2002).

FIG 1. Marbled murrelet on the water winter plumage. (Milo Burcham)

STATUS IN SOUTHEASTERN ALASKA

Distribution

Marbled murrelets are abundant and widely distributed throughout the waters of Southeast (Agler et al. 1998). Their at-sea distribution in summer finds them using inside sheltered waters, and often within several miles (km) of the shoreline. Agler et al. (1998) and Lindell (2005) reported a number of areas within Southeast where murrelets appeared to be abundant during surveys. It is not clear whether those represent areas with consistently high marbled murrelet numbers, or only transitory aggregations. Preferred inland nesting areas have not been documented in Southeast.

Abundance and Trend

Obtaining a reliable estimate of the Southeast marbled murrelet population is difficult. Secretive nesting behavior, low detect ability on nests, high risk of repeated counts of individuals on the water, and inter-observer variability all contribute to the difficulty in establishing population estimates for marbled murrelets (Agler et al. 1998, Whitworth et al. 2000). Large aggregations, commonly observed throughout the year, may shift during very short time frames, adding to the difficulty in accurately identifying populations of murrelets at sea (DeGange 1996).

Summer estimates of the Southeast population of marbled murrelets range from approximately 100,000 (Piatt and Ford 1993) to nearly 900,000 (upper CI, Agler et al. 1998). The population in 1996 was conservatively estimated to be in the low hundreds of thousands (DeGange 1996).

Population trend information for marbled murrelets is hampered by lack of long-term consistent monitoring across the region. However, there is good agreement with rates of decline estimated from Icy Strait and Glacier Bay (-12.7 vs. -11.8%), and these estimates are supported by observed trends from a region wide survey (-11.5%) (Piatt et al. 2007). Taken together, these efforts suggest a region-wide population decline in Marbled Murrelets of 79% over the past 20 years (Piatt et al. 2007). Multiple factors are likely contributing to the observed decline, including loss of old-growth habitat, changing ocean conditions, mortality from oil pollution and fishing nets, and increased natural predation (Piatt et al. 2007).

Special Taxonomic Considerations

No subspecies of marbled murrelet is recognized in North America. A closely related species, the long-billed murrelet *(B. perdix)*, is found along the coast of Asia (American Ornithologists' Union 1997).

Special Management and Conservation Designations

Because of the apparent reliance of the marbled murrelet on old-growth forests for nesting habitat (Fig 2) and the observed population declines over much of its range (Beissinger 1995, Nelson 1997, Burger 2002), the marbled murrelet has attracted the attention of wildlife agencies and conservation groups in western North America. In California, Oregon, Washington, and in Canada, marbled murrelets are considered threatened under the U.S. and Canadian endangered species acts (this status, however, is under review and may soon change). A recent decision by the Assistant Secretary of the Interior to unify these at-risk populations with the relatively abundant Canadian and Alaskan populations reverses an earlier conclusion by the U.S. Fish and Wildlife Service (USFWS) that the lower 48 population of marbled murrelets deserved separate protection. The USFWS recently completed a 5-year status review of the entire North American population of marbled murrelets (McShane et al. 2004).

In Alaska, the USFWS lists the marbled murrelet as a species of management concern. The U.S. Forest Service (USFS) considers it a species of concern in the Tongass National Forest (2002), and Audubon Alaska

FIG 2. Adult marbled murrelet on a nest in the top of an old-growth hemlock on Baranof Island in southeastern Alaska. (Jeff Hughes)

(2005) has placed the marbled murrelet on its WatchList because of the potential loss of breeding habitat in Southeast.

In the 1997 Tongass National Forest Land and Resource Management Plan, the USFS noted a dearth of data on the marbled murrelet in Southeast. In the plan, the USFS recommended coordination between multiple state and federal agencies to increase understanding of murrelet biology. In addition, the USFS suggested the following: a 600-ft (200-m) no cut buffer zone be maintained around identified murrelet nests, minimization of activity within the buffer zone, and monitoring of nests during the nesting period. Although made in the spirit of conservation, the recommendations are inadequate. Marbled murrelet nests are extremely difficult to find, and conservation measures that rely on known nest locations are futile. A more effective conservation strategy would be that used in the Pacific Northwest, where a buffer of 0.5 mi (0.8 km) radius is placed around any stand that is "occupied" by marbled murrelets (Raphael 2006). A stand is defined as occupied if murrelets are observed flying beneath or within the canopy, or are observed circling tightly above a stand.

In April, 2006, a panel of species experts on forest-nesting birds (USFS 2006) recommended consideration of 7 conservation measures for marbled murrelets in Southeast:

1. Maintain a system of Old Growth Reserves to conserve viable, well-distributed populations of marbled murrelets across the Forest.
2. Move or enlarge the Old-Growth Reserves to include important habitat identified by the marbled

murrelet habitat suitability model, and mapped using MARXAN.
3. Conduct bi-annual at-sea surveys of marbled murrelets. Coordinate these efforts with other agencies.
4. Remove the current standard and guideline that creates buffers around "found nests." Replace it with one that protects "occupied stands." Conduct appropriate field surveys during timber sale layout to identify these stands.
5. Consider deferring timber harvesting in rare, large-tree stands between 0.3 and 19 mi (0.5 and 30 km) from the shore. Allow exceptions where rigorous field surveys show a stand is unoccupied.
6. Promote and help fund research into marbled murrelet population trends, habitat requirements and limiting factors in Southeast.
7. Convene an independent task force of marbled murrelet researchers from Alaska and British Columbia. Regularly revise the marbled murrelet habitat model and amend standards and guidelines to ensure they reflect best available science

Significance to the Region and Tongass National Forest

Southeast represents just 10-15% of the species linear range, but it supports 60-70% of the global marbled murrelet population (USFS, 2006). Marbled murrelets contribute to the rich natural heritage of the Tongass National Forest. Nowhere in the world are these distinctive birds found in densities as high as those occurring in Southeast. The marbled murrelet is an important part of the complex ecology of both forest and nearshore marine environments, the 2 most striking attributes of the Tongass National Forest.

HABITAT RELATIONSHIPS

In summarizing the known terrestrial habitat characteristics of marbled murrelets in Southeast, DeGange (1996) stated, "Limited research has been undertaken in Southeast Alaska to assess terrestrial habitat characteristics, use, and suitability. This topic remains the single largest data gap for this species in this portion of its range in North America." In its supplemental environmental impact statement (SEIS) (2003) to the Tongass National Forest Land and Resource Management Plan (1997), the USFS lists important habitat for marbled murrelets as: "productive old growth within 31 mi (50 km) of the ocean, and at lower elevation in heads of bays." Conservation options stated in the SEIS are to "maintain productive old growth in heads of bays, emphasizing those near aquatic or terrestrial concentration areas." The beach fringe area (within 0.3 mi [0.5 km]) is not used by marbled murrelets for nesting, presumably because of the higher densities of avian predators found there (Burger 2004).

Specific requirements for nesting habitat of marbled murrelets in Southeast are presumably similar to requirements of the bird in British Columbia. Key microhabitat characteristics for marbled murrelet nest sites in British Columbia were listed by Burger (2004, citing Hamer and Nelson 1995, Nelson 1997, and Burger 2002) as:

1. Sufficient height to allow stall landings and jump-off departures;
2. Openings in the canopy for unobstructed flight access;
3. Sufficient platform diameter to provide a nest site and landing pad;
4. Soft substrate to provide a nest cup; and
5. Overhead cover to provide shelter and reduce predation by predators.

These requirements explain the overwhelming preference for marbled murrelets nesting in trees. There are records of ground nesting birds as well, especially in the treeless northern edge of the species range (Ford and Brown 1994, Marks and Kuletz 2001), although the proportion of the population that ground nests is relatively small (Piatt and Naslund, 1995).

Burger (2004) summarized suitable nesting habitat for breeding marbled murrelets as "old seral stage coniferous forest providing large trees with suitable platforms (limbs or deformities > 6 in [15 cm] in diameter), and a variable canopy structure allowing access to platforms." Several studies that used radar to track movements of marbled murrelets found that watershed populations were directly proportional to available old-growth forest (Burger 2002).

DeGange (1996) compared nest site characteristics of all 6 confirmed marbled murrelet nests (4 tree nests, 2 ground nests) found in Southeast. In all cases, stands were classified as "old growth, uneven age," canopy cover was dense, and nest material was moss. All tree nests in Southeast were in broken-topped hemlock in declining health (including 1 nest in mountain hemlock *[Tsuga mertensiana]* (Quinlan and Hughes 1990) and three nests were in western hemlock [*Tsuga heterophylla*]). Whether these recorded nesting sites are representative of nesting habitat for the entire

Southeast population of marbled murrelets is unknown. Based on examination of over 200 nests in British Columbia, Burger (2004) concluded it was unlikely that murrelets select particular tree species, but that certain species are more likely to provide the microhabitat requirements listed above.

In British Columbia, murrelet nesting habitat occurred over a broad range of elevations but the tendency was for nesting at elevations below 2,600 ft (800 m), with less suitable conditions at 2,952 to 4,920 ft (900-1500m), and areas above 4,920 ft (1500 m) unlikely to be used (Burger 2004, Zharikov et. al. 2006). Those thresholds are probably unduly high for Southeast, where temperatures are cooler and the tree line is lower than in British Columbia.

Marbled murrelets do nest on steep slopes, and in some studies, nest success was correlated with slope (Bradley 2002). At two study areas on the coast of British Columbia, murrelets preferred steep slopes with the mean slope of nest locations greater than 30 degrees (Zharikov et. al. 2006, Huettmann, unpublished data). Other studies have shown negative or neutral effects of slope on rates of occupied detections and measures of nest habitat quality (Burger 2002). Aspect does not appear to have a strong effect on the placement or success of nests (Burger 2004).

Marbled murrelets in Southeast and British Columbia travel substantial distances between foraging areas and nesting habitats (Newman et al. 2005, Huettmann, unpublished data). Whitworth et al. (2000) followed six individuals radio-tagged in Auke Bay (north of Juneau) during the breeding season (24 June–17 July). On 20 occasions, the same individual was located at both coastal and at-sea sites on the same day. Distance between these locations varied considerably up to a maximum of 77 mi (124 km). Generally, these six birds were moving between inland sites at Auke Bay and western Icy Strait/Glacier Bay, a known seabird foraging area. Therefore, they can be dependent on near-shore waters and still travel long distances in a day. However, they are traveling these long distances to access rich foraging sites, not to access inland forests. Hull et al. (2001) concluded that, in British Columbia, the distance of nest from water and nest success was not related.

Generally, marbled murrelets are more numerous in nearshore environments (about 1.3 mi [2 km]) than offshore. Whether at-sea murrelet distribution is dictated by bathymetric features is unknown (DeGange 1996), although it may be related to up-wellings and tide-rips which create food-rich feeding zones. At-sea habitat use in Southeast can be influenced by a variety of temporal variables, including tide, time of day, and season (Speckman et al. 2000).

Habitat Capability Model

To evaluate marbled murrelet habitat values within watersheds and compare watershed values within biogeographic provinces, a model for nesting habitat capability was developed by a group of interagency experts. Habitat values were rated by using data on murrelet habitat selection from studies in adjacent regions, including Prince William Sound, British Columbia, and the Pacific Northwest. The model estimates relative value of nesting habitats for marbled murrelets during summer based on forest stand age, tree size, distance from shoreline, and slope. Further details of the model (including habitat coefficients) are presented in Chapter 2. The nesting habitat values of watersheds to murrelets are ranked within each biogeographic province, an approach that avoids watershed comparisons between provinces (Appendix B).

IMPLICATIONS FOR CONSERVATION

It is widely believed that logging old-growth coniferous forests has contributed to substantial declines in marbled murrelet populations in the Pacific Northwest (Nelson 1997, Burger 2002). Because murrelet nests are difficult to locate, recommended practices prescribed by the USFS to minimize impacts on known nest sites are neither practical nor adequate. Widespread harvest of old-growth forest throughout the Tongass and adjacent private lands poses a significant risk to marbled murrelets nesting in old growth. Direct impacts of logging on murrelet nesting habitat can be assumed. Buffer zones are recommended by the USFS, but the effectiveness of buffers is unknown and more significantly few nests have ever been found and not a single buffer prescribed.

Because of short (100-yr) timber rotation cycles, clear-cut logging will permanently reduce murrelet nesting habitat in Southeast. Hundred-year timber rotations result in the permanent conversion of old-growth forest to even-aged second growth stands. Timber harvest rotations of less than approximately 250–300 years eliminate complex, old-growth forest structures, including large limbs with abundant moss, which are necessary for murrelet nesting platforms.

Trees in younger forests are much smaller and lack the large branches and robust epiphyte growth required for successful nesting sites. The more open canopy and vertical complexity of old growth also allows murrelets better flying conditions for landing on nest sites than the more densely stocked, closed-canopy second growth. In Southeast, stands of old, very large trees, characterized by large mean diameters and low stems per acre (Caouette and DeGayner 2005) are of importance to nesting marbled murrelets (USFS 2006) (Fig 3).

Fragmentation of nesting habitat, and the resulting decrease in forest patch size, can negatively affect murrelet populations. As forest patch size decreases, forest-edge habitat and predator access increases. Some avian predators of murrelets, especially corvids (i.e., ravens, crows, jays), are known to increase both with forest fragmentation and proximity to human activity (Burger 2002). Studies in British Columbia and Oregon revealed that nest success was higher at nesting sites farther from forest edges. Nest predation rates were also higher near forest edges (Burger 2002). Fragmented forests differ from contiguous forests in temperature, solar radiation, and wind penetration. The influence of these factors on murrelets through direct (such as thermal stress) and indirect (such as epiphyte growth patterns) mechanisms has been suggested but requires further study (Burger 2002). Alternatively, Zharikov et. al. (2006) suggest that murrelets may continue nesting in fragmented old-growth patches greater than 25 acres (10 hectares).

Marbled murrelets spend most of the year in nearshore environments and are susceptible to any threats posed there. Pollutant discharge (e.g., oil or fuel spills) may affect murrelets directly or through food web interactions (Carter and Kuletz 1995).

Commercial fishing nets pose a threat to murrelets foraging in shallow waters (Piatt and Naslund 1995). Mortality of murrelets in fisheries relying on other gear types is considered to be less problematic. Gillnet mortality of Marbled murrelets is a special concern in areas where murrelet aggregations and increased fishing effort overlap. The biggest overlaps in Southeast likely occur along Sumner Strait and the mainland shores of Stephens Passage (Carter et al. 1995). Murrelet bycatch, combined with nesting habitat loss, can be very detrimental to local populations. Long-lived species with low fecundity, common life history traits of seabirds, are especially sensitive to adult mortality.

Conservation of Southeast marbled murrelets depends on accurate population estimates, reliable population trend data, and a feasible monitoring protocol (DeGange 1996, Speckman et al. 2000). Several methods have been used or suggested to monitor marbled murrelets (Piatt and Ford 1993; Agler et al. 1998; Speckman et al. 2000, Burger 2001, USFS 2001). The Alaska Department of Fish and Game is currently conducting a study to compare different survey methods, including boat-based line and strip transects, aerial strip transects, shore-based flyway counts, and ornithological radar.

Ensuring productive, sustainable populations requires maintaining adequate habitat. Recent research in the lower 48 states and British Columbia has established the need to maintain undisturbed tracts of

FIG 3. Contrasting photos of old-growth forest canopy (left) and second-forest canopy (right). Old growth trees have large limb structures with abundant moss which provide nest platforms for murrelets. The canopy gaps in old growth may also allow better flight access to nest sites. In contrast, second growth has smaller limbs with less moss and a more closed canopy making flight to and from nest more difficult. (John Schoen)

old-growth forest to prevent further declines in marbled murrelet populations. Forest management that reduces and fragments large-tree old growth will likely have negative impacts on murrelet populations (Burger 2002).

Clearly, increasing the understanding of the life history of Southeast marbled murrelets and establishing basic requirements for nesting habitat will be important for developing conservation goals and management recommendations. Although several models for habitat suitability and habitat capability have been generated for the ranges of murrelets elsewhere, it is essential that regional models also be developed to account for differences in vegetation characteristics, climate, biogeography, and landscape conditions across the murrelet range in Southeast.

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Mammal invaders on islands: impact, control and control impact

FRANCK COURCHAMP[1], JEAN-LOUIS CHAPUIS[2] and MICHEL PASCAL[3]

[1] *Université Paris-Sud – Bat.* 362-91405 *Orsay Cedex, France. Tel*: 0033 16915 5685. *Fax*: 0033 16915 5696 (e-mail: franck.courchamp@ese.u-psud.fr)
[2] *UMR* 6553, *Rennes* 1, *Muséum National d'Histoire Naturelle – Département Ecologie et gestion de la Biodiversité* (*USM* 0304) 36, *rue Geoffroy Saint-Hilaire* – 75005 *Paris, France*
[3] *INRA – Equipe Faune Sauvage et Biologie de la Conservation – Station SCRIBE – Campus de Beaulieu* – 35 042 *Rennes Cedex, France*

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ABSTRACT

The invasion of ecosystems by exotic species is currently viewed as one of the most important sources of biodiversity loss. The largest part of this loss occurs on islands, where indigenous species have often evolved in the absence of strong competition, herbivory, parasitism or predation. As a result, introduced species thrive in those optimal insular ecosystems affecting their plant food, competitors or animal prey. As islands are characterised by a high rate of endemism, the impacted populations often correspond to local subspecies or even unique species. One of the most important taxa concerning biological invasions on islands is mammals. A small number of mammal species is responsible for most of the damage to invaded insular ecosystems: rats, cats, goats, rabbits, pigs and a few others. The effect of alien invasive species may be simple or very complex, especially since a large array of invasive species, mammals and others, can be present simultaneously and interact among themselves as well as with the indigenous species. In most cases, introduced species generally have a strong impact and they often are responsible for the impoverishment of the local flora and fauna. The best response to these effects is almost always to control the alien population, either by regularly reducing their numbers, or better still, by eradicating the population as a whole from the island. Several types of methods are currently used: physical (trapping, shooting), chemical (poisoning) and biological (e.g. directed use of diseases). Each has its own set of advantages and disadvantages, depending on the mammal species targeted. The best strategy is almost always to combine several methods. Whatever the strategy used, its long-term success is critically dependent on solid support from several different areas, including financial support, staff commitment, and public support, to name only a few. In many cases, the elimination of the alien invasive species is followed by a rapid and often spectacular recovery of the impacted local populations. However, in other cases, the removal of the alien is not sufficient for the damaged ecosystem to revert to its former state, and complementary actions, such as species re-introduction, are required. A third situation may be widespread: the sudden removal of the alien species may generate a further disequilibrium, resulting in further or greater damage to the ecosystem. Given the numerous and complex population interactions among island species, it is difficult to predict the outcome of the removal of key species, such as a top predator. This justifies careful pre-control study and preparation prior to initiating the eradication of an alien species, in order to avoid an ecological catastrophe. In addition, long-term monitoring of the post-eradication ecosystem is crucial to assess success and prevent reinvasion.

Key words: alien species, biological invasions, introduced species, island restoration.

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I. INTRODUCTION

(1) Biological invasions: a cause of major change

The invasion of species into naive ecosystems, a process known as biological invasion, has historically been overlooked not only by many scientists, but also by politicians and economists. It is now, however, considered the second most important cause of biodiversity loss, after habitat destruction and fragmentation (Vitousek *et al.*, 1997*b*). Thanks to human migration and commerce, the number of species colonising new habitats has increased by orders of magnitude in the last two centuries (di Castri, 1989; in Mack *et al.*, 2000). Not only do human activities promote biological invasions by being very effective dispersal agents, but they also facilitate the establishment of new immigrants through husbandry activities that protect small and vulnerable immigrant populations until they become larger and more resistant (Mack *et al.*, 2000). It has been estimated that the current rate of species extinction is between 50 and several hundred times greater than was estimated previously on the basis of geological data (Wall *et al.*, 2001). The ecological homogenisation caused by biological invasions is mainly due to the relatively recent increase in human travel around the globe leading to the erosion of geographical barriers and reduced isolation among ecosystems. For tens of millions of years, species not capable of long-range dispersal have diverged behind their sea barriers on separate land habitats (Holdgate, 1986). Recent accelerated influxes of previously separated species by human dispersal activities are now eliminating such isolation. Biological invasions currently constitute one of the main causes of major global ecological change (Vitousek *et al.*, 1997*a*), and could

very well influence other agents of global change (Mack *et al.*, 2000). For the purpose of this review, we distinguish between natural and human-induced movement of species into ecosystems, with a focus on the latter.

(2) Islands: fragile and threatened ecosystems

Most introductions of species into new habitats are failures (deVos & Petrides, 1967). For example, of the 150 or so species of birds introduced to Hawaii, and of the 145 introduced to New Zealand, 'only' 30 and 36 have become established, respectively (Roots, 1976; Veitch & Clout, 2001). This relates to what has been termed the '10's rule': approximately 10% of introductions succeed and approximately 10% of these will be significantly ecologically disruptive (Williamson, 1996). However, with the sheer number of previous attempts, a small proportion of successful introductions translates into a very large number of cases, which concern virtually all major islands. In addition, some species, whose ecological equivalents are naturally lacking in the native ecosystem, may have a disproportionately greater impact and do not follow the 10% rule. For example, of the 55 mammal species introduced to New Zealand, 32 have become established, far more than the 10% (Veitch & Clout, 2001). A little more than a decade ago, Ebenhard recorded 644 mammal introductions on islands (Ebenhard, 1988). Few islands have escaped the problems caused by species introductions, and islands are the recipients of 80% of documented bird and mammal introductions (Ebenhard, 1988). Over 80% of all islands have been invaded by rats, *Rattus* sp. (Atkinson, 1985), and at least 65 major island groups have also been invaded by cats, *Felis catus* (Atkinson, 1989).

Oceanic islands are home to plant and animal communities with relatively little diversification, simplified trophic webs and high rates of endemism (Chapuis *et al.*, 1995). These three points partially explain the high susceptibility of island ecosystems to disturbances, and the observed high rate of extinction of species. Most of contemporary worldwide extinction events have indeed occurred or are currently occurring in insular ecosystems. For example, more than 90% of the 30 species of reptiles and amphibians reported as extinct since the 17th century were island forms (Honnegger, 1981). Similarly, 93% of 176 species or subspecies of bird extinctions that have occurred during this period, as well as 81% of the 65 mammal species, also occurred on islands (King, 1985; Ceballos & Brown, 1995). These include only the species that are known to have disappeared: the vertebrate fauna of mainland ecosystems is in many cases better known than for insular ecosystems predating human arrival, hence the extinction rates could be much higher. Similarly, it is likely that the smaller number of reptile and amphibian extinctions reported above only reflects the smaller number of studies on these taxa. Few data are available concerning extinct insular species of plants and of invertebrates, but the figure is undoubtedly proportional. The actual number of species extinctions is thus likely to be far greater than the currently accepted figures.

The successful introduction of alien species is the major cause of ecosystem perturbation and biodiversity losses on islands (Atkinson, 1985; Moors & Atkinson, 1984; Lever, 1994; Williamson, 1996). For example, predation by introduced animals has been a major cause of 42% of island bird extinctions in the past, and is a major factor endangering 40% of currently threatened island bird species (King, 1985). There is thus an urgent need to understand and counteract the processes by which biological invasions threaten biodiversity on insular ecosystems. For research on threatened species, resources and time are limited and are usually focused on a single species; there will never be sufficient time and resources to manage each taxon individually (Fitzgerald & Gibb, 2001; Saunders & Norton, 2001). Emerging from this is the concept of biodiversity hotspots: areas featuring exceptional concentrations of endemic species and experiencing exceptional loss of habitat (e.g. Reid, 1998; N. Myers *et al.*, 2000). Islands or island groups such as New Caledonia, New Zealand, the Caribbean, Polynesia/Micronesia, Madagascar and the Philippines have been defined as biodiversity hotspots (N. Myers *et al.*, 2000). No less than nine of the 25 hotspots defined by N. Myers *et al.* (2000) are entirely or mainly made up of islands, and almost all tropical islands fall into one or another hotspot (N. Myers *et al.*, 2000). Conservation of these insular ecosystems is not only among the most urgent, it might also be among the most feasible: in most cases, the biodiversity crisis taking place on islands is singularly the result of biological invasions, which is easier to fight in a closed and often remote area as compared to the same surface area on a continent.

(3) Nature and purpose of this review

Introduced mammals have reportedly caused more problems than any other vertebrate group (Ebenhard, 1988; Lever, 1994). Mammals also are responsible for the best documented and most spectacular of ecological disturbances resulting from biological invasions (Holdgate, 1967). This is certainly due in part to the general lack of naturally occurring terrestrial mammals on most

remote islands (Atkinson, 2001). By themselves, introduced mammal species constitute a large subject, and thus we will focus only on them in this review, although we recognise that other taxa, including plants and insects, are also notoriously devastating invaders (e.g. Pimentel *et al.*, 2001).

In this paper, we will attempt to review the effects that can follow the introduction of one or several mammal species onto an island. In addition, we will outline the most utilised methods of control of introduced mammal populations. We will then discuss the main possible impacts, both positive and negative, of mammal control. Because exhaustiveness is not possible on this subject, we will instead provide an overview of the main effects of mammal introductions, of the main methods of their control and of the main consequences of such control, illustrating most cases with typical examples. By the nature of studies on biological invasions (often triggered by conservation problems following invasion events), and by our choice of giving striking examples of the effects of such invasions, this review may be biased towards reporting negative effects of introduced species.

Following the Invasive Species Specialist Group's (ISSG) definition of terms (I.U.C.N., 2001), alien, foreign, exogenous and exotic species will be used synonymously to qualify species occurring outside of their natural range of potential dispersal, by opposition to native, indigenous or autochthonous species. Alien invasive species will be used for an alien species that becomes established and is an agent of change in the invaded ecosystem, threatening the native biological diversity. Introduced species refers to a movement (unintentional or intentional) by human agency of a species outside its historically known natural range.

II. ECOLOGICAL CONSEQUENCES OF INTRODUCTIONS

(1) Impact assessment

The impact of alien species can be considered at five different levels: effects on individuals, on genetics, on population dynamics, on community composition and functioning and on ecosystem processes (Parker *et al.*, 1999). On islands, the successful establishment of introduced species can lead to dramatic changes at all five levels. As we will detail below, the presence of introduced species often causes significant changes of the abundance and/or spatial structure of the autochthonous species it interacts directly or indirectly with, and thereby can affect significantly the global functioning of the ecosystems concerned (Williamson, 1996, 1999).

It is generally very difficult to assess the impact of an introduced species on the ecosystem it has invaded (Parker *et al.*, 1999). In most cases, data are not available to compare communities before and after the invasion. The recovery of native flora and fauna following exotic mammal eradication is generally a good illustration of their real impact on invaded ecosystems (Newman, 1994; Taylor, Kaiser & Drever, 2000; Towns, Daugherty & Cree, 2001). An alternative approach is possible in the case of archipelagos, where several combinations of introduced species can occur on nearby islands. Such islands are similar in most biotic and abiotic aspects, which allows the estimation of single species effects (e.g. comparing islands with and without introduced goats, *Capra hircus*) or of combinations of introduced species (e.g. islands with rats and cats *versus* islands with cats only, islands with rats only and islands with no introduced species). Unfortunately, studies on the effects of alien control programmes based on comparison between areas with or without alien removal (or comparisons before/after removal) cannot always be considered as unconditional evidence. Indeed, the communities may not restore themselves to their former states, due to the deep modifications caused by the alien prior to its removal, to the presence of other aliens, or to other factors. Even evidence based on diet of invaders and correlated with shifts of abundance of native species can be rightly criticised, since a causative process is next to impossible to demonstrate in this context. In the absence of such evidence, there has even been doubt concerning the actual impact of species considered by most conservationists as the most notoriously harmful invading species (Norman, 1975). As stated by Ebenhard, 'The decision whether an introduction has had an ecological effect is often hard to make on the basis of published information only, and hence is open to criticism by people with first-hand experience of some particular situation' (Ebenhard, 1988). There is however a very long list of circumstantial evidence of the effects of introduced species, which, taken together, is a body of 'circumstantial proof' (Peabody, 1961) that can overpower even the slightest doubt about the dramatic impact of biological invasions on oceanic islands.

According to Ebenhard (1988), introduced animal species may affect native species in one or more of six different ways. They may: (*i*) affect plant populations and species living in the habitat structured by these plants; (*ii*) be predators of native prey; (*iii*) induce interference or resource exploitation competition; (*iv*) spread micro- and macroparasites into native populations; (*v*) induce genetic changes to native species through hybridisation; (*vi*) act as prey to native predators. The last two categories are probably relatively

unimportant as far as islands are concerned. Effects through competition or release of pathogens are probably much more important, but there are few examples in the literature, as it is extremely difficult to demonstrate unambiguously such effects. The examples given in this review will be mostly restricted to the results of exploitation of native species by introduced species, as well as resulting indirect effects (habitat destruction and competition with other species).

(2) Reasons for high impact

Not only are most alien invasive species highly adaptable (e.g. Apps, 1986; van Aarde, 1986; Ebenhard, 1988, Flux, 1993), but they also encounter favourable conditions in the new environment, compared to their native environment. In fact, alien invasions can be facilitated by three factors: more (or better) resources, less (or less efficient) natural enemies, and advantageous physical environment (Shea & Chesson, 2002). Because of their long evolution in isolation from other species, insular species have not evolved traits to provide an adapted response to efficient interference and exploitation (van Aarde & Skinner, 1981; Moors & Atkinson, 1984; Atkinson, 1985, 2001; Burger & Gochfeld, 1994). For example, defensive traits against intense herbivory such as high fecundity, compensated growth, production of toxins or unpalatable substances, or physical deterrents such as spines, are generally absent in plants on oceanic islands (Atkinson, 1989). Rather, plant communities of oceanic islands are mainly composed of species that are highly palatable and vulnerable (Holdgate, 1967). Similarly, small vertebrates may not have learned to hide their sounds, odours, or their shelters from predators, and once discovered, they may not flee or defend their young. Flightless birds are a good example of this: dodos in Mauritius, moas and kiwis in New Zealand, kagus in New Caledonia. In addition to being behaviourally maladapted, these prey species lack the life-history traits associated with many continental species, such as an earlier age of first reproduction and high fecundity [some examples are given for New Zealand shorebirds in Dowding & Murphy (2001)]. These species constitute a major (sometimes unique) and easy trophic resource for the few founders of introduced predators. Introduced cats, rats and dogs *Canis familiaris*, have all participated in the extermination of the flightless birds of Tristan da Cunha, Amsterdam, Ascension and Saint Paul islands, among others (Lever, 1994). The main islands mentioned in the text are indicated on the map shown in Fig. 1.

In addition to a potentially great abundance of resources on islands, the lack of natural enemies also favours the successful establishment of introduced species. Introduced species generally suffer from a smaller subset of parasites that are often less virulent than on the mainland (Dobson, 1988), for two key reasons. First, the limited number of hosts involved in introduction processes results in fewer pathogenic parasites being transmitted to the new environment. Second, many introduced parasitic species may be lost because either the new environment is unsuitable, key species are lacking that prevent the completion of their life cycle, or because the density of the introduced population is initially too low to allow the maintenance of the parasite population (Keeling & Grenfell, 1997). Where potential insular predators exist, they are often relatively inefficient in preying upon introduced species. After more than a century, the skua *Catharacta skua* still does not prey very efficiently on rabbits *Oryctolagus cuniculus*, introduced to the Kerguelen islands, preying mostly on young and sick individuals (Moncorps *et al.*, 1998).

With an abundance of resources and a lack of pressure from natural enemies, invasive animals often build up numbers in an abnormally short period after introduction. Initially confronted with a closed (and often small) geographic area, and freed from the normal biotic pressures found on the mainland, introduced species often increase in population size beyond the limit of sustainability of the new environment (in terms of food resources as well as shelters). As a result, when these resources are depleted a population crash may follow a marked peak (Nugent, Fraser & Sweetapple, 2001). For example, the 29 reindeer *Rangifer tarandus*, introduced onto St Matthew Island initially responded so well to the high quality and quantity of forage that they increased in less than 20 years up to 6000 individuals. The huge impact of such a large population on a restricted environment led to a severe impoverishment of the flora, resulting in a dramatic population crash, down to 50 individuals (Klein, 1968). Such a reduction of the population can lead to an increase of the vegetation, which then may allow another build up of the herbivore population, sometimes leading to another crash. In 1957, a pair of mouflons (*Ovis musimon*) was introduced on a small island (650 ha) of the Kerguelen Archipelago. From those two individuals, a population developed that numbered around 700 individuals 20 years later. Between 1977 and 1995, at least five population crashes occurred, mainly because of insufficient food availability at high density. Each crash, concerning 400–450 individuals, was followed by a recovery of the flora, despite an increasing degradation of the environment by erosion, which in turn led to a rebuilding of the population. This cycle of events was broken in 1995,

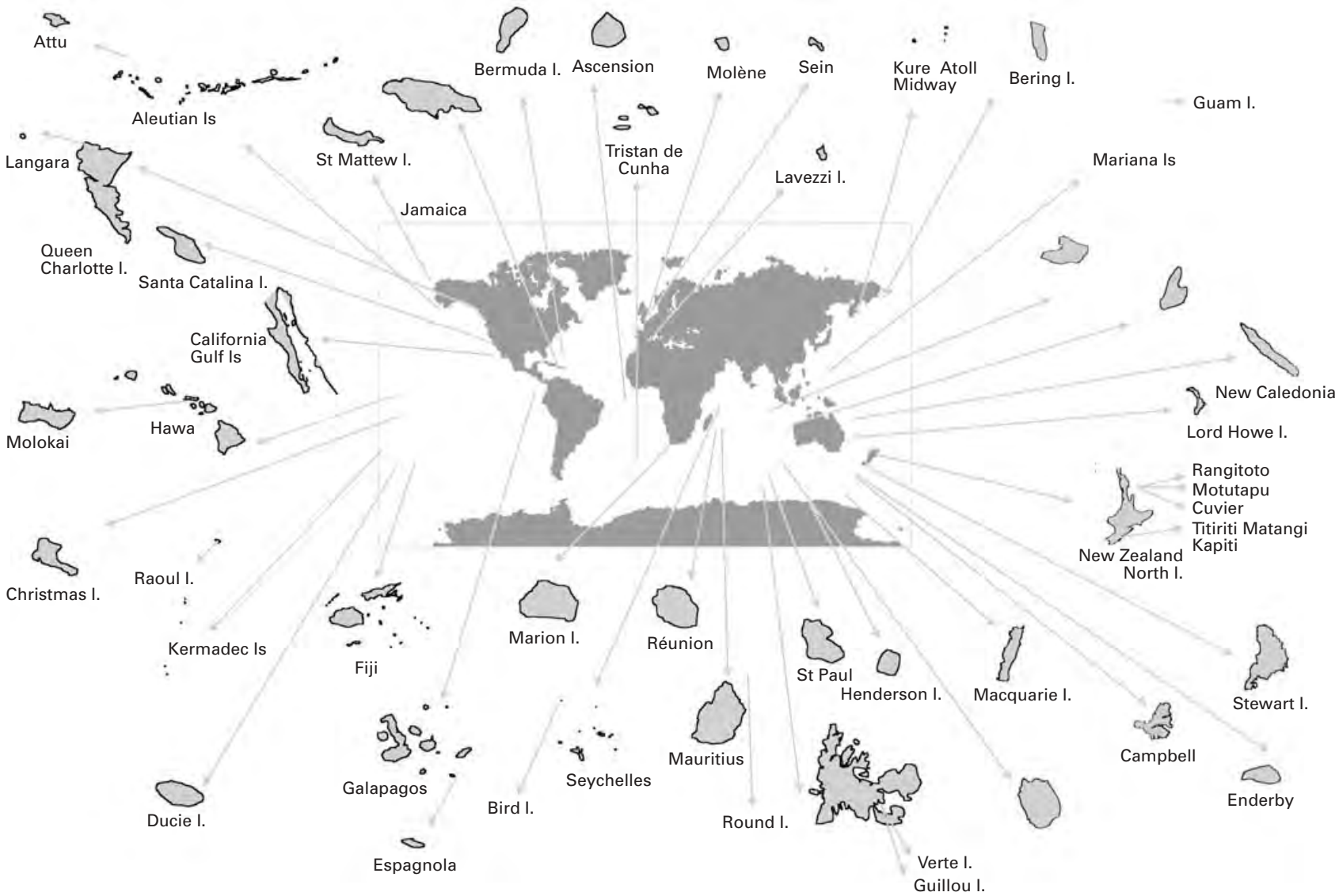

Fig. 1. World location of main islands mentioned in the text.

the date of the last population crash, since when the population has been kept small by a yearly shooting campaign (Boussès, Réale & Chapuis, 1994; Chapuis, Boussès & Barnaud, 1994*a*; Réale, Boussès & Chapuis, 1996).

Alternatively, the damage to the environment may be such that the habitat or the resources themselves can be totally destroyed, also resulting in the extinction of the introduced species. In the long term, such a natural extinction of introduced species (more likely to occur on small islands with simple ecosystems) may be more general than currently witnessed. For example, it has been reported that for the 607 islands where the fate of introduced rabbits is known, the population died out naturally in more than 10% of cases (Flux, 1993). However, following disappearance or removal of exotics, recolonisation by local species is not always possible, owing to the very high endemism of oceanic islands and the lack of neighbouring populations that can act as recolonisation sources, or because the alien invasive species caused irreversible modifications of the environment.

(3) Simple ecological effects

(*a*) *Simple effects due to single species introductions*

Ecological effects of introduced species may be better appreciated if one can disentangle the different effects of different species from the effects of species combinations. Because trophic webs on oceanic islands are often very simplified, with little ecological or taxonomical redundancy (Holdgate, 1967), dramatic increases of introduced species may lead to important imbalances in the whole ecosystem. The decline of endemic species that are victims of interference or exploitation from introduced species is often the first consequence, and may then lead to further damage at the ecosystem level. Three types of ecological effects can be distinguished: (*i*) shift of relative species abundances, without affecting the total number of species; (*ii*) disappearance of some species from parts of the habitat; (*iii*) extinction of some species. Some ecological effects, habitat changes or shifts of native abundance of plant or animal species caused by predation or competition have been reported in at least 40% of all mammal introductions (Ebenhard, 1988). These dramatic results are better appreciated when compared to the effects of bird introductions, which have resulted in observed ecological effects in only 5% of all cases (Ebenhard, 1988). The figure for mammals is much higher if only introductions onto oceanic islands are considered.

Ebenhard (1988) also calculated that 93 out of 95 introductions of strict herbivore species caused habitat change on oceanic islands, and the number of plant species affected by introduced herbivores is probably innumerable. Because there is little published information on the indirect effects of plant community changes on invertebrates, it is difficult to assess the direct and indirect effects of mammal introduction on local invertebrates. It is likely that they are considerable.

A single introduced species can also affect, directly and indirectly, a whole range of indigenous species, thereby affecting the whole invaded ecosystem. Nine Indian mongooses (*Herpestes auropunctatus*), introduced to Jamaica from Calcutta in 1872, seem to have been the source of all the populations on 29 Caribbean and four Hawaiian islands before the end of the 19th century (Lever, 1994). These mongooses eradicated many invertebrates, reptiles, amphibians, birds and mammals, mostly in the West Indies (Westermann, 1953; Nellis & Everard, 1983; Lever, 1994). Another good example of this is given by domestic goats, generally seen as one of the most destructive feral mammals to native vegetation (Turbott, 1948; Parkes, 1990; Keegan, Coblentz & Winchell, 1994). Introduced goats have a very large spectrum diet: they consumed at least 48 species of vascular plants on Raoul Island, mostly indigenous species, and several of them rare and endangered (Parkes, 1984). Within 35 years of overgrazing, over-browsing and trampling, introduced goats reduced the number of plant species from 143 to only 70 on Great Island, New Zealand (Turbott, 1948). Through direct competition, or by severely reducing the vegetation cover, goats introduced in the Galapagos archipelago also indirectly threatened several species of birds and reptiles (including the local race of the giant tortoise; Hamman, 1975). On Santa Catalina Island, they also almost wiped out the native sagebrush *Artemisia californica*, and the resulting reduced coverage seriously threatened the California quaii *Lophortyx californica*, as well as several native rodents; consequently, several species of reptilian and mammalian predators have decreased as well (Coblentz, 1978).

(*b*) *Effects of resource exploitation*

The main impact type is the result of exploitation of the invaded ecosystem resources by browsers, grazers and predators. In this section, we review the effects of consumption of native species by herbivores, omnivores and carnivores, focussing on three emblematic species. Other examples could easily have been used, as many other mammals have notoriously devastated populations and communities. Most of man's commensal species, and many domestic animals have been introduced onto islands. Some have been introduced

inadvertently; for example, they went ashore from ships by themselves or they came from newly established human colonies (e.g. rats, mice *Mus musculus*, cats, dogs). Nevertheless, in contrast to plants and invertebrates, the majority of mammals were deliberately introduced, sometimes repeatedly and tenaciously. These were generally introduced either to control the spread of previously introduced rodents (e.g. cats, mongooses), for fur or stock farming or hunting (e.g. cattle *Bos taurus*, goats, pigs *Sus domestica*, wild boar *S. scrofa*, horses *Equus caballus*, donkeys *E. asinus*, foxes *Alopex lagopus* and *Vulpes vulpes*, possums, reindeer, mouflon, deer), or to provide a durable source of food for shipwrecked mariners (e.g. rabbits, goats, sheep *Ovis aries*, and the brown and Arctic hares *Lepus europaeus* and *L. arcticus*). A large part of research on biological invasion is focussed on characterising the biological attributes of successful invading species (Crawley, 1986; Williamson, 1996; Mack *et al.*, 2000; Kolar & Lodge, 2001; Sakai *et al.*, 2001). Some species are particularly more successful in this regard, both because they have been more often introduced and because such introductions have been more often followed by establishment. Among these, some are well known to conservation biologists because their establishment is almost invariably followed by rapid and important changes in the original communities. The most important introduced mammals, both in terms of numbers of introductions and resultant damages, are the rabbit, the cat, three commensal species of rats, the house mouse, the goat, the pig and cattle. Apart from cattle, all are in the ISSG list of 100 of the worst alien invasive species (http://www.issg.org/database/welcome/). Many of these invasive species have had a major impact, as illustrated by the number of species allegedly extirpated by their introduction. For example, (Jackson, 1977 cited in Ebenhard, 1988) estimated that 61 bird taxa have been exterminated by introduced predators, with cats (33 extinctions), rats (14 extinctions) and mongooses (nine extinctions) being the most important predators.

Specialists on several of these species share similar strong terms to describe the impact of these mammals when introduced into naive ecosystems:

'The domestic goat undoubtedly has been the most destructive feral mammal to native vegetation, particularly as it travels over all types of terrain and consumes all kinds of browse and herbaceous material' (deVos, Manville & Van Gelder, 1956).

'With the exception of man, the rat is the mammal that has undoubtedly had the greatest impact on the environment and economy of most Pacific islands …' (Storer, 1962).

'Of all the introduced pests, the rabbit is by far the worst. Its acclimatization in Australia was the greatest single tragedy that the economy and the native animals have ever suffered' (Frith, 1979).

'Cats are now ubiquitous on inhabited islands, and frequently present on those uninhabited. Probably no other alien predator has had such an universally damaging effect on seabirds' (Moors & Atkinson, 1984).

'The avifauna of some islands have suffered drastically from predation by rats. (…) On a few islands, the proportion of bird species that have declined or become extinct following the introduction of *R. rattus* is so great than the term catastrophe is appropriate' (Atkinson, 1985).

'Rats … have been implicated in the greatest number of extinctions due to predators (54 percent … Of the three species which have shared the blame for this record of predation, the Black Rat or Roof Rat (*Rattus rattus*) has been the most serious problem …' (King, 1985).

'Introduced predators have accounted for about half of island bird extinctions. Rats and cats are the most notorious killers and island birds the most notorious victims, in this regard' (Diamond, 1989*a*).

'… there can be little doubt that the mongoose in the West Indies … has helped to endanger or exterminate more species of mammals, birds and reptiles within a limited area than any other animal deliberately introduced by man anywhere in the world' (Lever, 1994).

This list of infamy could go on. Instead, we now provide a short view of the effects of the introduction of three of these species, in order to illustrate the effects of an herbivore, an omnivore and a predator.

(*i*) *Effects of herbivores: the rabbit as an example*

Because their impact is focused at the base of trophic webs, introduced herbivores can have a wide-ranging impact on invaded ecosystems. One of the most documented introduced mammal species, often referred to for its dramatic impact on endemic plant species, is the rabbit. This herbivore has been introduced (most of the time voluntarily) to more than 800 islands so far (Flux & Fullagar, 1992). Rabbits have a high ecological adaptability and introductions easily succeed (Flux, 1993). They can adapt to harsh conditions, eat a wide range of plant species (Thompson, 1953; Chapuis, 1981, 1990; Homolka, 1988), and have an exceptional growth rate for their body mass (Smith & Quin, 1996). The very rapid increase of their populations in ecosystems where indigenous grazers are in general much less numerous and competitive, leads to a dramatic impoverishment of the vegetation, both quantitatively and qualitatively

(Gillham, 1955; Costin & Moore, 1960; Holdgate & Wace, 1961; Norman, 1967; Selkirk *et al.*, 1983; Chapuis *et al.*, 1994*a*), and results in severe impacts on associated indigenous fauna.

Overgrazing by rabbits constitutes a reduction of the plant cover for terrestrial nesting birds, which often affects their reproductive success, as was seen on Meeuw Island (Gillham, 1963) or on the Kerguelen archipelago (Weimerskirch, Zotier & Jouventin, 1989). It also increases direct competition for food among birds depending directly (e.g. granivorous) or indirectly (e.g. insectivorous) on the terrestrial vegetation (Gillham, 1963). Overgrazing may lead to the exposure of larger proportions of soil surface to frost heaving, rain and wind erosion (Scott, 1988), resulting in dramatic denudation of the soil. Rabbits also compete directly for nesting burrows (or places to dig them) with many burrowing seabird species (e.g. Young, 1981). Furthermore, this burrowing can accelerate erosion (Norman, 1967; Chapuis, 1995). It is also believed that as much as 10% of Hooker's sea lion pups die annually by being trapped and suffocated in rabbit burrows on Enderby and Rose Islands (Sanson & Dingwall, 1995). Rabbit overgrazing also prevents any large-scale regeneration following natural catastrophes such as fire (Norman, 1967) or cyclones (Kirk & Racey, 1992). Rabbits may also have significant direct effects on other species, such as disturbance (Gillham, 1963), which could lead to colony desertion by seabirds. Lastly, and as we will see later on, rabbits often constitute an alternative prey for indigenous (e.g. skuas) or introduced predators (e.g. cats). By increasing the number of predators, or by allowing them to persist when and where local prey is scarce, they contribute to the decline of indigenous prey (Brothers & Copson, 1988). By one or several of these factors, rabbits are believed to have been ultimately responsible for the decline or extinction of several reptile and bird species (King, 1985; North, Bullock & Dulloo, 1994; Smith & Quin, 1996). One classic example concerns the extreme damage caused to Round Island, in Mauritius. Helped also by feral goats, introduced rabbits reduced the community of several endemic palm trees, which normally protected the island from hurricane damage, to 5–15 individuals depending on the species, with several having completely disappeared (Bullock, 1977). Of course, not only the plant species were directly affected, but also the animals depending on them disappeared as well, such as four species of endemic reptiles. As a result, this 151 ha island possessed the highest density of threatened plant and animal species in the world, before conservation efforts concentrated on eradicating both goats and rabbits (North & Bullock, 1986; North *et al.*, 1994).

Another example concerns Laysan Island in the Hawaiian chain, where rabbits alone were responsible for eliminating 26 species of plants between 1903 and 1923, a rate of loss exceeding one species per year (Christophersen & Caum, 1931 cited, in Atkinson, 1989). On that island, they have also led to the elimination of three bird species. A fourth, the Laysan duck, increased from seven individuals by one hundred-fold after the rabbit's removal (Warner, 1963). Rabbits are also believed to have been responsible for the decline or extinction of several other vertebrate species by the impact they have had on their habitat.

(*ii*) *Effects of omnivores: rats as an example*

The other main category of mammal pest invaders concerns omnivorous species. These have often been more present in the minds of conservation biologists because of their more direct and visible effect on indigenous wildlife. Three species of rats have become unequalled worldwide threats for small insular prey: the Kiore, Polynesian or Pacific rat *Rattus exulans*, the Black, Roof or Ship rat *R. rattus* and the Brown or Norwegian rat *R. norvegicus*. These three commensals are known to have colonised at least 82% of the 123 major island groups (Atkinson, 1985). Because they show important ecological differences, the three species have had a dissimilar impact, both quantitatively and qualitatively on insular wildlife (Atkinson, 1985; Lever, 1994).

The ecological impact of the Pacific rat is difficult to assess, but is it said to be the least harmful of all three species on vertebrates (although more destructive on insect and plant communities). Nevertheless, in addition to its impact on the native plant communities, this partly arboreal and partly terrestrial rodent (Atkinson, 1977) is believed to be responsible for the extinction of many invertebrate, amphibian, reptile and bird species (Atkinson, 1978; Crook, 1973). For example, on Kure Atoll, these rats are suspected of killing the translocated Laysan albatrosses (*Diomeda immutabilis*, Fig. 2), as well as sooty terns (*Sterna fuscata*), Noddy terns *Anous stolidus* and Bonin petrels *Pterodroma hypoleuca* (Kepler, 1967). Similarly, the impact of this rat on Murphy's petrel seems so great (100% of chicks killed within four days of hatching) on Henderson Islands, that the colony is believed to be subsisting only through immigration from the species' largest breeding site world-wide, located on two nearby islands, Oeno and Ducie (Lever, 1994).

The two other rat species have a much larger impact on the fauna of the islands they have invaded. The Norwegian rat has largely terrestrial habits, putting birds with similar habits at the highest risk, in particular seabirds: eggs, chicks or even brooding adults are regularly killed, often resulting in local extinction. Because it

Fig. 2. Introduced Polynesian rat attacking a Laysan Albatross behaviourally non-adapted to terrestrial predation. Photograph courtesy of Cameron Kepler.

is arboreal and therefore affects forest birds as well, the Black rat probably has had the largest impact of all three species (Lever, 1994). A well-known example of the important impact of black rats on native vertebrate communities is their establishment around 1964 on Big South Cape Island, New Zealand. There, they caused the local loss of three endemic birds, and the complete extinction of two other bird species and of one species of bat, in less than two years (Bell, 1978). The populations of many other native species were depleted, although not to complete extirpation. Another example illustrating the dramatic impact of introduced mammals on biodiversity concerns the emblematic tuatara *Sphenodon punctatus*, last of the Sphenodontida, an order of reptile present 225–80 million years ago in different parts of the world. This endangered species is endemic to New Zealand, but has been extirpated from the mainland and now occurs only on a few small islands. They are, however, not safe in those few restricted areas: the invasion of Whenuakura Island by Norway rats in 1982 led to the elimination of the entire local population of more than 130 tuatara (Newman, 1988). Rats have had an impact on most remaining tuatara populations as well as on other rare reptile species (Newman, 1988; Newman & McFadden, 1990; Cree, Daugherty & Hay, 1995; Towns *et al.*, 2001).

The impact of rats on the vegetation of the islands they have invaded is also extremely important. Rats eat leaves, seeds and fruits, flowers, bark and stems of many endangered species. By hindering regeneration and killing sapling and small adult trees, they modify entire plant communities, and thereby affect the associated fauna (Clark, 1981). By interference and/or exploitation, black rats that now occur on all vegetated habitat of the seven main islands of the Galapagos are said to be responsible for the extinction of four species of endemic rice rats *Oryzomys* sp. (Brosset, 1963). Black rats have an effect on the regeneration of maritime forest (Campbell, 1978) because they are major consumers of inland podocarp seeds, and also because they compete with and kill many species of birds that disperse seeds (Clark, 1981). Inversely, although examples are rare, introduced rodents may favour seed dispersion: black and Pacific rats are better seed dispersers than the native bats on Tonga (Drake & McConkey, 2001).

The Pacific rat has been documented killing at least 15 different bird species, predation by the black rat has been described in at least 39 bird species, while the brown rat has been recorded killing at least 53 bird species (Atkinson, 1985). Evidence of predation on small vertebrates by rats on remote islands is more difficult to obtain, therefore these numbers surely are an underestimation of the actual number of bird species killed by each rat species. Arthropods, snails, amphibians and reptiles are less studied than birds, and few data are available concerning the number of species under threat by introduced rats, but here again, there is little doubt that the number is very large (e.g. Clark, 1981; McKenzie, 1993).

As mentioned above, the impact of an introduced species on a remote oceanic island is very difficult to prove and even more difficult to quantify. Norman (1975) even suggests that the impact of introduced rats may have been overstated in some cases. That author argues that evidence of bird population decreases is often circumstantial, and that few data are available to conclude that rats are solely responsible for some bird extinction events. Norman (1975) suggests that the majority of the impact of introduced predators has been imputed to rats while the impact of cats on the same island may have been much higher. Although most specialists concur on the nearly invariable significant impact of introduced rats, it is true that untangling the effects of several introduced predators, such as rats and cats, is difficult at best. Cats are indeed a major problem on oceanic islands and are often present on islands where rats have been introduced.

(*iii*) *Effects of predators: the cat as an example*

Because they were the main means of controlling rodents on ships, domestic cats travelled around the world with explorers, sealers, whalers and other seafarers, and thus gained easy access to most remote oceanic islands (Todd, 1977). On those islands, in addition to all the cats that escaped ashore during a stopover, and those

that stayed with colonies or lighthouse keepers, many cats were in fact introduced (as were mongooses) in an attempt to control growing populations of rodents near newly established human populations. Domestic cats are very adaptable (Konecny, 1983; Apps, 1986; van Aarde, 1986) and have survived in the most inhospitable conditions on many remote oceanic islands, both inhabited and uninhabited (Fitzgerald, 1988). They are found on most major island groups, from subantarctic islands to arid islands with no fresh water (Derenne, 1976; Pascal, 1980; Tabor, 1983; Atkinson, 1989).

Despite sometimes-harsh climatic conditions, these ecosystems were very favourable for domestic cats, being virtually exempt of natural enemies and often full of defenceless prey (as well as introduced prey such as rodents and rabbits, which may allow cats to survive the season when migratory birds are absent). The cat is an opportunistic predator, and its diet on oceanic islands may include a large proportion of reptiles (e.g. Konecny, 1987; Bamford, 1995), birds (e.g. Fitzgerald, 1988; Rodriguez-Estrella *et al.*, 1996), or mammals (Derenne & Mougin, 1976; Jones, 1977; Pascal, 1980), according to the prey's relative abundance in different seasons and regions.

As a consequence of the wide diet of this very efficient predator and the lack of adaptation of the local prey, introduced cats are known to be the direct cause of severe reductions or extinctions of numerous populations of insular vertebrate species (Iverson, 1978; Taylor, 1979*b*; Moors & Atkinson, 1984; King, 1985). As is the case for rats, to enumerate the exhaustive list of (suspected) victims of domestic cats would be too long and depressing. As Lever (1994) points out in the case of impacts of cats on birds, 'the list of species they have helped to exterminate or endanger reads like a roll-call of avian disaster'. Here we provide a few famous examples to help to illustrate the huge impact that a few individual cats introduced onto an island can have on the local fauna some decades later.

Cats have often been introduced in small numbers, but their ease in proliferation in such favourable conditions makes then a fearsome threat for local animal communities. For example, within 70 years of their introduction to the Fijian islands, cats increased so much in numbers on Viti Levu that they were hunted for food and sport (Gibbons, 1984). It has been estimated that 25 years after the introduction of five cats on Macquarie Island, the established population of 2000 cats killed nearly half a million burrowing petrels per year (van Aarde, 1980). Similarly, in the Kerguelen Archipelago, 30 years after the introduction of a cat and her three kittens, the population of 3500 cats destroyed 1.2 million birds per year (Pascal, 1980). Some predators kill more than their immediate need: foxes store surplus birds and eggs for use in winter when few birds are present, and single larders have been found with more than 100 seabirds of several species (Bailey, 1993). This may obviously unbalance the predator–prey dynamics if the prey population is not adapted to such behaviour. On small islands, the impact of such a population may result in the extinction of the prey species, especially if alternative prey are present to sustain the predator population (see Fig. 3).

In a few decades, cats introduced onto Herekopare islands destroyed all but a few thousand of the 400 000 birds living there, extirpating no less than six entire species of land birds, and large colonies of seabirds (Fitzgerald & Veitch, 1985). In addition to many local extinctions, introduced domestic cats are said to have caused more than half of the avian species extinctions due to introduced predators (Jackson, 1977 cited in Ebenhard, 1988). Here again, it is not unreasonable to suggest that since birds are far more studied than other animal taxa on oceanic islands, the lack of published data on the extinction by cats of indigenous invertebrates, amphibians and reptiles probably does not accurately reflect the actual widespread occurrence of such events.

(*iv*) *Other mammals*

Many other species have caused significant conservation problems all over the world, especially goats (Turbott, 1948; Parkes, 1984, 1990, 1993; Rudge, 1984), pigs (Hone, 1990, 1992, 1995; Sterner & Barrett, 1991), small Indian mongooses (Seaman, 1952), mice (Rowe-Rowe, Green & Crafford, 1989; Crafford, 1990; Newman, 1994; Le Roux *et al.*, 2002), foxes (arctic *Alopex lagopus* and red *Vulpes vulpes*, West & Rudd, 1983; Bailey, 1992, 1993) and domestic dogs (Iverson, 1978). For example, feral dogs in the vicinity of humans are reputed to hunt principally for 'sport', and can exceed other alien predators in exterminating species. A prime illustration of this is the individual dog that is said to have systematically hunted and killed more than half of the remaining 900 New Zealand brown kiwis, *Apteryx australis*, in less than three weeks (Taborsky, 1988; Diamond, 1989*b*).

Some mammals can have an important impact on multiple levels such as in the case of the brushtail possum *Trichosurus vulpecula* in New Zealand (Clout, 1999). The primary conservation impact of this invading marsupial is through the damage they cause to native forest through browsing. Possums also feed on flowers and fruits, reducing fruit crops of native plants and hence both forest regeneration and the food supplies of several native birds, which in turn may fail to reproduce

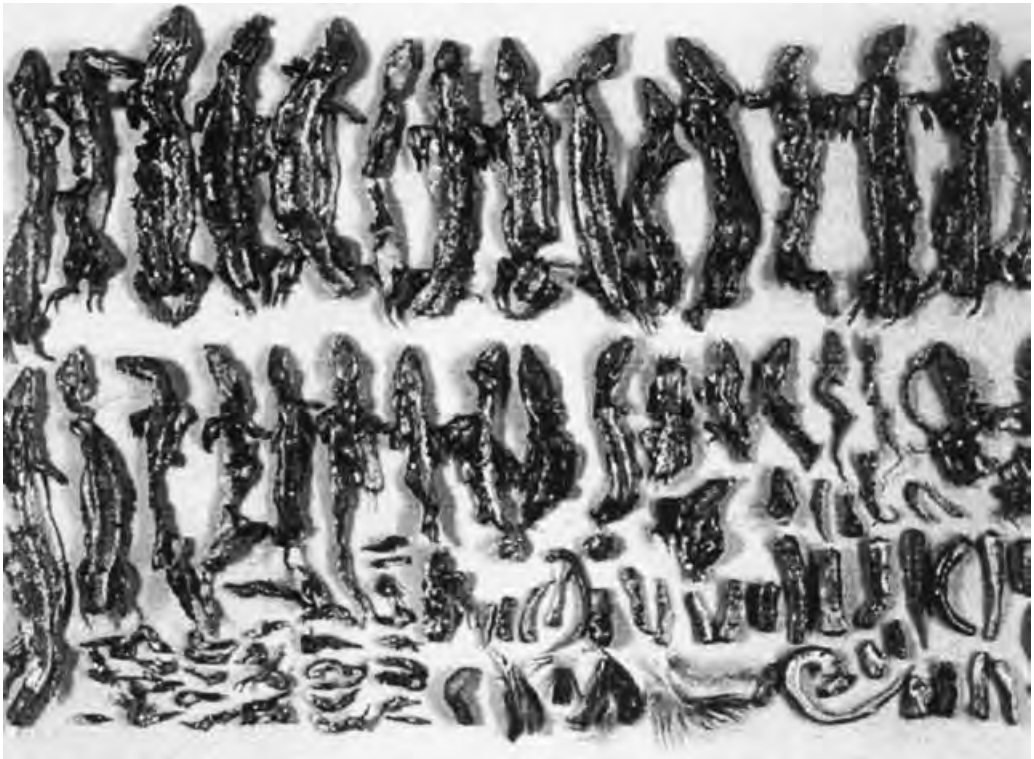

Fig. 3. Stomach content of a single feral cat caught in New Zealand, with at least 34 native skinks *Leiolopisma* spp. Photograph courtesy of John Dowding.

and even survive. They also compete for hollows with endemic hole-nesting species, such as kiwi (*Apteryx* spp.). Although mostly herbivorous, possums have recently been shown preying on the eggs and chicks of birds, including those of threatened species (Brown *et al.*, 1998). Finally, they also have an economic impact, as they transmit the bovine tuberculosis virus to cattle and deer (Clout & Sarre, 1997). Because of their multiple ecological and economic effects as an invasive species, the introduction of possums to New Zealand is now recognized as 'an unmitigated disaster for the natural ecosystems and native biota of this archipelago' (Clout, 1999).

In addition to the above, several authors list mammal species reported to have been introduced onto islands (deVos *et al.*, 1956; Holdgate, 1967; Ebenhard, 1988; Lever, 1994). There have been at least 80 mammal species introduced since 1700 (Atkinson, 1989), but some of them cause more damage than others. The other introduced mammals include mustelids (stoat *Mustela erminea*, weasel *M. nivalis*, ferret *M. furro*, North American mink *M. vison*, polecat *M. putorius*, Japanese weasel *M. sibirica itatsi*), wallabies (dama *Macropus eugenii*, parma *M. parma*, and brush-tailed rock *Petrogale penicillata*) and deer (axis *Cervus axis*, fallow *C. dama*, red *C. elaphus*, rusa *C. timorensis*, while-tailed *Odocoileus virginianus* and reindeer). Some species are introduced less often but may nonetheless be important, for example the Virginia opossum *Didelphis marsupialis*, the house shrew *Crocidura russula*, the musk shrew *Suncus murinus*, the crab-eating macaque *Macaca fascicularis*, the green monkey *Cercopithecus aethiops* and the water buffalo *Bubalus arno*.

(*c*) *Effects of competition and parasitism*

Apart from the classically studied direct effects of herbivores and predators, which we have detailed above, introductions may be deleterious for native populations through two other processes, both understudied but potentially very important: competition and spread of parasites.

Although competition is rather rare, when it occurs it often affects the abundance of the native species (Ebenhard, 1988). Competition can be of two types. It can be called interference competition, as is the case with grey-bellied squirrels (*Callosciurus caniceps*) introduced on the island of Oshima, which chase local birds away from the flowers they feed on and pollinate (Lever, 1994). It can also be a resource exploitation competition, a process both more likely to occur and more difficult to demonstrate. Examples of such interactions include the competition of rabbits and seabirds for nesting burrows

(Cooper & Brooke, 1982) or the competition of introduced ungulates (which often overgraze) for plants that constitute the basis of diet or shelter for native species (see Lever, 1994 for examples). These relationships can be complex, because some birds may benefit from rabbit burrowing, or from the vegetation opening. Direct competition for food with introduced goats is considered one of the direct reasons for the decline or even extinction of several populations of birds and reptiles, among which is the Galapagos giant tortoise *Geochelone elephantus* (Daly & Goriup, 1987). Obviously, strict herbivores are more likely to get involved in competitive relationships with native species than are omnivores or predators, especially since terrestrial predators are scarce on oceanic islands. However, some examples concern the competition for food with native predators, notably in the Kerguelen islands between cats and skuas: where cats are present, there are not enough petrels for the skua to reproduce, despite the presence of alternative prey such as mice and rabbits (Chapuis *et al.*, 2001).

The spread of new parasites with invading animals is probably the least studied of the major effects of species introductions on native ecosystems and only a few examples exist concerning mammal pathogens. These include the introduction of unidentified pathogens by black rats on Christmas Island, that seems to have led to the extinction of the endemic bulldog rat (*Rattus nativitatis*; Day, 1981) and the introduction by the Northern red-backed vole (*Clethrionomys rutilus*) to Bering Islands of the cestod *Echinococcus multilocularis* which affected native foxes (deVos *et al.*, 1956).

(*d*) *Simple ecological effects due to multi-species introductions*

Unfortunately, there are many well-known examples of coincident introduction of several species, leading to more complex, and often detrimental, effects on the original trophic webs. For example, the construction of a tourist hotel on Caicos Islands led within three years to the near extirpation of the 5500 endemic West Indian rock iguanas *Cyclura carinata* that were hunted by introduced cats and dogs (Iverson, 1978). On Stewart Island, cats and three species of rats have exterminated five species of birds, three of which were endemic (Karl & Best, 1982). Similarly, in the Marianas Islands, Rota Island numbered no more than 121 adult trees of the endangered Micronesian *Serianthes nelsonii*, while the neighbour Guam had only one, these trees being mostly threatened by introduced red deer, pigs and granivorous insects (Wiles *et al.*, 1996).

In many cases, several invaders belonging to distinct trophic levels have been introduced into naive ecosystems, leading to some of the most complete cases of community disruptions. The effects of mammal introductions on Lord Howe Island, off Australia, are a good example of this. The endemic forest of *Howea forsterana* now contains very few small individuals, due to the combination of grazing by goats and seed consuming by rats (Pickard, 1982). In addition, in less than 20 years, rats eradicated at least five endemic bird species (which account for more than 40 % of all the indigenous species of land birds) and three endemic invertebrates on this island. They also extirpated all the seabird colonies and greatly reduced the numbers of lizards, land snails, and other invertebrates of the island (Recher & Clark, 1974; Atkinson, 1985). These two introduced mammals have had many other indirect effects, including changes to the structure of the habitat and removal of native species. One can easily imagine how profound changes can be following the introduction of even more mammal species, as is the case for many other islands. In addition to increased effects owing to increased numbers of actors, there are many cases of interactions between introduced species having a synergistic effect in the damage suffered by the local species. These are what we term complex interactions.

(4) Complex interactions

On most islands, several introduced mammal species interact to modify trophic webs greatly. Most islands that have been inhabited by humans, even for a short time, also shelter a great number of alien species, many of which are mammals. A great number of alien mammals were introduced, even on the most remote islands, as long as they were in a useful location for ships, or worse, had a permanent colony, often deliberately. For example, the Kerguelen Archipelago is one of the most remote island groups, in the subantarctic zone of the Indian Ocean, 4000 km off the coast of Africa and 3000 km away from Australia, Nevertheless, rats, mice, cats and dogs rapidly followed sealers and whalers. In addition, rabbits, pigs and sheep were introduced to constitute self-maintaining reserves of food, while American mink were introduced for their fur, reindeer to serve as draught animals, and Corsican mouflon for hunting purposes (Pascal, 1983). Apart from pigs, dogs and American mink, all these species are still currently present in the wild in these islands (Chapuis *et al.*, 1994 *a*). Islands that are inhabited or that have been heavily used for their resources may be even more heavily impacted. For example, the islands of Hawaii now shelter one of the most complete arrays of alien species, with many mammals, including the three species of commensal rats, mice, cats, mongooses, dogs, pigs, goats, cattle, sheep, rabbits, axis deer and even brush-tailed rock

wallabies. The very complex interactions between indigenous and one or several introduced species make it very difficult to characterise the impact of introduced species on the indigenous flora and fauna.

The simplest example of interaction is when a previously introduced species facilitates the invasion of a second invasive species, usually by modifying the habitat. For example, some exotic birds need the forest to be modified by browsing herbivores and are unable to colonise undisturbed native forest in the absence of alien mammals. After the removal of introduced browsers (and predators) from Cuvier Island, the indigenous forest was able to regenerate, and the exotic bird species disappeared (Diamond & Veitch, 1981). A comparison of Pacific islands with and without introduced ungulates suggests that many island species of plant can 'resist' alien plant invasions in the absence of ungulates (Merlin & Juvik, 1992, cited in Cabin *et al.*, 2000).

There are also many cases of direct interactions between several established alien species. For example, if a predator is introduced into a trophic web where two prey species are competing for a resource, the complexity of its effects may increase if one of the prey is preferred over the other. An exacerbation of this is when both a prey species and its predator are introduced into a naive ecosystem. In this case, the introduced prey is well adapted to the predation and has an indirect competitive advantage over the native prey. Worse, the introduced prey's adaptation to predation may be such that they are less favoured (i.e. more difficult to catch), but at the same time this allows an increase of the predator population without suffering from it (because only the population surplus is killed), leading to hyperpredation (see below). In this context, it is interesting to compare the effect of introduced cats on seabird communities, with and without alternative prey.

Impacts of the presence of alternative prey are well illustrated by the example of rabbits, whose effects on endemic vertebrate species can be more complex than the ones presented above, especially when predators have also been introduced. In this case, rabbits are only a secondary prey item in months when seabirds are present, but appear to enable predators (mostly domestic cats) to subsist over winter, when seabirds are absent or rare (Pascal, 1980; Brothers, Skira & Copson, 1985; Chapuis *et al.*, 1994*a*). They also allow predator populations to reach remote colonies or populations of indigenous prey on islands with heterogeneous indigenous prey distribution (Brothers & Copson, 1988). In addition, they can exacerbate the predation pressure on indigenous species through what has been termed hyperpredation (Smith & Quin, 1996). This process is well illustrated by the example of Macquarie Island, where predation by introduced cats caused the decline of burrow-nesting petrels (Brothers, 1984) and the extinction of an endemic parakeet *Cyanoramphus novaezelandiae erythrotis* and a banded rail *Rallus philippensis* (Taylor, 1979*a*). Cats were introduced to this island 60 years before the introduction of rabbits. However, the dramatic impact of cat predation on bird populations dates back to just ten years following the introduction of rabbits (Taylor, 1979*a*). It is believed that the rabbit population allowed not only the maintenance of the cat population in winter (when seabirds are absent from the island), but also a significant increase, therefore resulting in increased predation pressure on the land bird species. The rabbit population, which was more adapted to predation by cats supported such an increase. It was, however, fatal to the bird population, which on its own could support a small cat population, but was extirpated by an over-sized cat population that no longer depended on the presence of birds to survive (Courchamp, Langlais & Sugihara, 1999*b*; Courchamp, Langlais & Sugihara, 2000). Since a great many islands harbour both populations of introduced predators and their natural prey, this hyperpredation process could be a widespread cause of further threat to small indigenous vertebrate populations.

A fascinating example of complex interactions through single species introduction comes from the California Gulf Islands. There, an endemic fox species, *Urocyon littoralis*, has dangerously declined more than 90% over the past decade. In parallel, its only competitor, an endemic skunk *Spilogale gracilis amphiala*, has been increasing. The reason for this decline was far from obvious. Careful studies showed that skunks were not responsible: they are largely dominated by foxes in a competitive relationship, and their increase was a consequence of the fox decline. Pathogens were not the cause either, and prey was plentiful and indeed increasing, following the relaxation of fox predation. The only remaining explanation was a top predator whose presence, or effect, was new to the ecosystem (since foxes were known to be the top predators on these islands, and their decline was recent). Researchers on these islands pointed out that the decline in foxes was linked to a coincident increase in golden eagle (*Aquila chrysaetos*) predation (Roemer, 1999; Roemer *et al.*, 2001). A combination of field data and mathematical modelling suggested a hyperpredation process, engendered by the introduced population of pigs (Roemer, Donlan & Courchamp, 2002). On these islands, golden eagles are not able to survive as a population on the local prey only, but piglets now constitute year round sources of abundant food, which allows eagles to breed. Pigs have thus facilitated the colonisation of the islands by eagles;

while the pig population does not suffer too much from predation (only the surplus of piglets is taken by eagles), they represented a threat to foxes. The endangerment of this endemic fox is directly linked to a significant increase of an endemic skunk, its natural competitor, and is caused by a predator whose survival is threatened in other places, thus complicating conservation measures.

Some other interesting complex effects of mammal introductions include the displacement of native species from preferred habitat by introduced predators, or habitat destruction by introduced herbivores, both of which lead to increased exposure to predators. Both have been documented. On Bermuda islands, introduced rats have forced cahows (*Pterodroma cahow*) to nest in suboptimal habitat where chicks are threatened by the white-tailed tropicbird *Phaethon lepturus* (Jackson, 1977, cited in Ebenhard, 1988). In the Galapagos Islands, goat browsing has destroyed the cover of land iguanas *Conolophus pallidus* and thereby increased their susceptibility to predation by Galapagos hawks, *Buteo galapagoensis* (Dowling, 1964; Gibbons, 1984).

There are also mutualist interactions between introduced species, which can act in synergy to inflict damage to local communities. An interesting illustration concerns the facilitation of access of introduced brush-tailed possums to forests. In the northern forests of New Zealand, introduced deer destroy the understorey, which causes the forests to become drier. This makes the habitat more suitable for introduced possums. The possums in turn cause a partial elimination of the tree canopy, which facilitates understorey regeneration, and thereby benefits red deer (Wallis & James, 1972).

These examples illustrate the near-impossibility of accurately predicting the ecological consequences of relationships among several introduced and native species. Nevertheless, conservationists generally agree about the need to control mammals introduced on oceanic islands. We will see that such complex interactions necessitate a deeper understanding of the system in order to predict properly the result of drastic management actions such as the removal of one species from the ecosystem.

III. CONTROL: TRADITIONAL *VERSUS* BIOLOGICAL METHODS

The first and best course of action is to prevent species introduction in the first place (I.U.C.N., 2001). Unfortunately, a great number of introductions already exist, forcing a response after the fact. There are three strategies to alleviate problems caused by introduced species: exclusion, control and eradication (see, e.g. Bomford & O'Brien, 1995). All three involve a reduction of the numbers of the animals causing problems. Exclusion is only a local solution, as it concerns a spatially delimited zone of effort from which the alien species will be removed. Control can have two meanings: it can be a general term of action against an alien species ranging from simple reduction up to eradication, and it can more specifically mean lowering of the introduced population numbers, that is mitigation or reduction. In this review, we use control as a general term, and use reduction or mitigation for partial population elimination. Control in the sense of mitigation is the reduction of the size of the pest population, down to acceptable levels, in ecological or economic terms. As such it is opposed to eradication (see below). Since it is not complete, such a removal strategy involves constant and/or repeated actions, to keep the population at low density after the initial decline. Although it is generally more feasible than eradication, the gains achieved by control are temporary. Many pest mammals have a density-dependent reproductive rate with higher reproductive levels at low density, leading to faster recovery rates for controlled populations. For example, after an 80% reduction, a goat population returned to previous levels within four years, the doubling time of a controlled goat population being around 20 months (Parkes, 1984).

Eradication is the complete removal of all the individuals of the population, down to the last potentially reproducing individual, or the reduction of their population density below sustainable levels (J. H. Myers *et al.*, 2000). This is generally, although not always, the best strategy for islands, but it is often limited by its high cost, logistically as well as economically. There are a number of factors that make some islands more difficult to free from alien species, or some alien species more difficult to remove from islands. Some factors are related to the island, for example area, ecosystem accessibility (field ruggedness, ecosystem penetrability, mooring possibilities, …) and isolation (distance to continent or other source). Some other factors are related to the alien species, for example dispersal abilities, diet width and ecological plasticity. Some species are said to be particularly robust and difficult to eliminate from large remote surfaces. In addition, when pushed down to a very low density, the last individuals of the populations of most species are generally very difficult to remove, for two reasons. First, the few remaining individuals are simply more difficult to find, and second, because of the density dependence that characterises most populations of invaders, these last individuals will quickly reproduce, which is a force that the control programme will have to fight against. An amazing example concerns the last

remaining mule deer on San Clemente Island, USA. During a pig eradication programme in 1991, hunting dogs turned up an old male mule deer, which was shot. This was the first sighting of a deer on the island since the Navy's Natural Resources Office 'completed' deer eradication in 1977. Of course, being a single male, eradication had been essentially achieved, but this illustrates well how even large animals can remain undetected for quite a long time (14 years in this case; R. Brand Phillips, personal communication). For the pig eradication programme on Santiago Island, Galapagos, 450 times more effort was required to remove a pig in 2000 than in 1998; the actual cost per pig removed increased by approximately 2000% between 1997 and 2000 (Patry, 2001).

However, unless the island has a high likelihood of repeated invasions, the eradication should have to be implemented only once, as opposed to regulative control. It may be initially more expensive, but will be more cost effective than repeated (or continuous) population reductions because it is permanent (Veitch & Bell, 1990; Rice, 1991). For example, the cost of rat mitigation for the Mediterranean Lavezzi islands was estimated at around 3000 euros per year while its eradication cost five times that amount (M. Pascal, unpublished data). Although the larger amount is more difficult to obtain, it is more cost-efficient in the long term when compared to a 5+ year population reduction programme. In addition, if poisoning is one of the chosen action methods, eradication requires lesser amounts to be used than would be necessary for sustained control efforts. Similarly, fewer animals will have to be killed during an eradication campaign than would be for a long-term mitigation, which may be more acceptable to animal rights campaigners.

(1) Traditional methods

We will here compare two main types of control: traditional and biological. Each method comes with its own set of advantages and disadvantages and often these depend on the species to be controlled, resulting in some control methods being very well adapted to a given species. We provide in Table 1 a summary of method types, with the associated main advantages, disadvantages and typical target species.

Rather than describe all existing traditional control methods for introduced mammals, we will now provide a few typical examples of the techniques often used to control the main mammal invaders. Table 1 summarises the main advantages and disadvantages for each type of control method (physical, chemical and biological).

(*a*) *Fencing*

Because of their large size, ungulates are among the easiest mammals to exclude using fences. This solution is most appropriate when the area to be controlled is too large for an eradication programme, or if the herd is not to be entirely destroyed. An example of this is the feral cattle of Amsterdam Island (55 km^2) in the Indian Ocean (Micol & Jouventin, 1995). In 1871, a farmer gave up his livestock attempt on this island because of the harsh conditions, abandoning five cattle. These individuals adapted well enough, and bred rapidly to form a population reaching 2000 individuals in 1988. Owing to the grazing and trampling of this large population, as well as several major fires since the eighteenth century (1792, 1853, 1899, 1974), the native vegetation regressed in favour of introduced species. From the 1980s, introduced cattle started to colonise high elevation zones, which are especially sensitive to trampling and which shelter a relic population of Amsterdam albatross (*Diomedea amsterdamensis*). This species was discovered in the early 1980s and at that time numbered only 12 breeding pairs (Jouventin & Roux, 1983). Despite their impact, eradication of the cattle was not initiated because they represented one of the very few feral herds of this species anywhere in the world, a large herd that had evolved in total isolation for more than a century. The restoration strategy adopted was to divide the island into two sectors with a fence, and the cattle were eliminated from the main and most sensitive part of the island while the herd was regulated on a smaller sector with no birds. Fences are also used to exclude smaller animals, such as foxes, cats, rabbits, rats or even mice, in conjunction with other control methods, to clear the fenced area of the unwanted species. An impressive example is the Karori Wildlife Sanctuary, near Wellington, New Zealand (Saunders & Norton, 2001). It is a mammal-proof enclosure of 252 ha from which all introduced animals (including black rats, mice, possums, stoats and ferrets) have been eliminated in order to restore the local hardwood forest and to reintroduce extirpated species.

(*b*) *Shooting*

Large mammals will evidently be more easily shot, and therefore hunting has historically been used to control them. This is also linked to the associated gain in meat, and sometimes to the recreational aspect of hunting. The main disadvantage of this method is the poor accessibility of many oceanic islands, which makes it costly and logistically difficult to maintain enough manpower for a sufficient task force. This problem may be at one or both of two scales: accessibility to the island itself, and accessibility to some parts of the island. For example, it

Table 1. *Advantages and disadvantages of different control methods*

Method type	Typical target species	Advantages	Disadvantages
Physical methods			
Shooting	Ungulates	Very efficient for large animals Very selective Environmentally clean More ethical method (but usual targets have a high public profile)	Requires good accessibility for hunters in the field Requires experienced and dedicated staff May be costly and logistically difficult to maintain enough manpower for a sufficient amount of time Requires public information and sensibilisation May require special authorisation for guns May imply additional techniques: dogs, Judas technique, helicopters, etc.
Trapping	Small carnivores, possums	Efficient for small populations of small and medium-sized species in accessible and small areas Can be selective Environmentally clean	Certain traps may pose ethical problems Requires a limited infested area Requires a limited population Requires good accessibility to carry traps on the field May be expensive to buy sufficient traps May require good trapping experience to be efficient Requires attractive baits for target species that are non attractive (or repulsive) for non-targets May be selective (age or sex) Rarely achieves eradication unless used with another method
Chemical methods			
Poisoning	Rodents, carnivores, small herbivores	Very efficient for small rodents	Requires special authorisation May affect non-target species (protection measures may be expensive and not fully efficient) Implies complete coverage of the whole infested area Targets can develop bait aversion Implies the use of adequate baits and often of bait stations May cause secondary poisoning (accumulation in animal tissues) Some toxins are only slowly degraded Can be costly, especially for large islands Requires public information and sensibilisation
Biological methods			
Predator introduction	Rodents, small herbivores	Environmentally clean Very low cost	Great risk of specificity loss, causing further ecological disequilibria Historical reason for the existence of alien cat and mongoose populations Add a species and its parasites to the ecosystem

Table 1. (*Cont.*)

Method type	Typical target species	Advantages	Disadvantages
Competitor introduction	Carnivores	Can be very efficient if well planned	Released individuals must to be removed or die off without reproducing Very little information available Add a species and its parasites to the ecosystem
Pathogen introduction	Carnivores, small herbivores	Environmentally clean Very low cost Can be very specific	Poor efficiency at low density Ethical problems (may induce suffering) Sanitary risks Requires good knowledge of host-parasite system Requires absence of risk for non-target species Potential for host adaptation (evolution of immunity)
Virus vectored immuno-contraception	Rodents, carnivores, small herbivores	Most ethical method Low cost (when self-dissemination) Can be very specific Environmentally clean	Not yet operational Concerns for loss of control Very slow Irreversible process Low public acceptance of release of genetically engineered organisms Potential for development of host resistance

takes more than twenty days round trip by ship to access the Kerguelen islands, with a limited number of trips each year. In addition, the main area of this archipelago is extremely rugged: it has an area of approximately 6500 km^2 and a coastal length of more than 1000 km (i.e. between three and four times more than a perfect circle of the same area). Most parts of the island are reachable only by foot, requiring several days of trekking. This precludes the use of many control methods, and it seriously hinders others such as hunting. Yet, when accessibility can be dealt with, hunting is one of the most efficient ways of controlling or even eradicating a herd of ungulates, especially if large incentives are made available for hunters (Gosling & Baker, 1989). Using helicopters, it would be possible to eliminate the reindeer population from Kerguelen, but this would incur a high cost.

The most difficult part of large mammal control by hunting is the localisation of the last few individuals, since the very low density may seriously hinder the elimination of these few survivors, thereby threatening the whole control programme. Two main methods are used to overcome this problem. In many cases, dogs can be trained to assist hunters with rifles, and their help in detecting isolated individuals may make the difference (Cowan, 1992). Another, very clever method, especially designed to eradicate goats, is what has been called the Judas goat method (Parkes, 1984, 1990; Keegan *et al.*, 1994). It consists of fitting one individual, preferably a female in oestrus, with a radio collar. Goats are highly gregarious animals, and this female will be very efficient at finding isolated groups throughout the island. At regular times, the radio signal will be used to localise the female, and the hunters will have little difficulty in reaching the group, either by foot or by helicopter. Once in view, the whole group will be shot, except for the 'Judas', which will be spared so that it can find another group, thus involuntarily leading the hunters to it. When the 'Judas' is regularly found alone, the population is considered eliminated and that last goat is removed.

(*c*) *Trapping*

Trapping has historically been used for medium-sized mammals, such as small carnivores and large rodents, because they are less easy to shoot, but still valuable to collect either for fur or for meat. Trapping is mostly

used for rodents, cats, mustelids, mongooses, rabbits and hares, but only for population reduction, since their eradication is not feasible by trapping alone (except for carnivores). Trapping has also been very efficient for possums (Cowan, 1992). Leg-hold traps are more rarely used now for ethical reasons, rather live door-traps are routinely used. However, in some cases leg-hold trapping may be the only efficient method, especially for cats (Veitch, 2001). The control methods based on trapping are very similar to those based on poisons: both will generally need to cover a substantial area and to develop attractive baits. However, traps may be of limited use if the area to control is too large, with limited accessibility, if the population is too large, or if the animals are trap shy. The eradication of a small population of cats can be considered by trapping on a small island if the terrain is not too irregular, but populations on large and/or rugged islands cannot be eradicated, since it is simply impossible to carry and set enough traps on very large or poorly accessible areas. However, traps do have the advantage of selectivity, because they can be designed to exclude or reduce accidental capture of native animals.

(*d*) *Poisoning*

Poisoning is the other major control method for medium and small mammals. Poisoning programmes often attempt to avoid non-target species kills with baits targeted for the pest species. However, discrimination can be difficult to achieve when native and alien species are taxonomically or ecologically close, and the impact of poisoned bait on native species can sometimes be important. Losses of non-target individuals to poisoning must thus be assessed with respect to losses caused by the uncontrolled introduced species (I.U.C.N., 2001). In fact, research consistently suggests that the harmful effects of introduced mammals are greater (and longer lasting) than those of the toxin used to remove these mammals. Even if many non-target individuals die from poisoning, populations of these individuals often recover rapidly once the pressure from the introduced species is removed (Merton, 1987). As an example, Pukekos (*Porphyrio porphyrio*) have re-populated Tiritiri Matangi and Motuihe Islands, New Zealand, after an initial reduction of around 90% due to a poisoning campaign with brodifacoum (Dowding, Murphy & Veitch, 1999). In some cases, elimination of non-target species through poisoning has been deemed unavoidable, and the eradication programme has been designed so that the native species could be reintroduced, once the alien species was eliminated. This was the case for the burrowing bettong *Bettongia lesueur*, which was eliminated during the poisoning of black rats on Boodie Island, off Australia. However, this had been foreseen by the programme leaders, who thus designed their programme such that once the rat was eradicated, bettongs could be reintroduced from a nearby island used as a reservoir (Morris, 2001).

Another possibility is to capture at-risk non-target endemics in order to protect them from poison during the eradication programme and then to release them once the invasive population is eradicated and the poison removed or degraded. This has been done for several programmes in New Zealand. In 1996 Pacific and Norway rats were eradicated from Kapiti Island (2000 ha), New Zealand (Empson & Miskelly, 1999). During the operation three pairs of takahe *Porphyrio mantelli*, a giant flightless rail, were held in captivity on the island for just over two months. In addition approximately 200 weka, *Gallirallus australis*, a large flightless rail, were removed from the island, 50 of which were held in pens in a mainland reserve and the remainder released in a remote former part of the species mainland range. The captive birds were held for around three months then returned to the island. During the period 1996–2000, the eradication of five introduced mammal species (feral cat, rabbit, black rat, Norway rat and house mouse), was attempted on four inhabited islands in the Seychelles. Since no rat-free island was available to which native animals at risk from poisoning might be transferred, it was necessary to maintain approximately 560 individuals of three threatened animal species in captivity for the three months duration of the eradication programme. Very low mortality was detected during captivity, and all captive animals were released in their former, now safe habitat, once the rats were eradicated (Merton *et al.*, 2001). The brown rat eradication programme for Fregate Island, Seychelles, included the capture and captivity of the entire population of the critically endangered Seychelles Magpie-robin *Copsychus sechellarum* (accounting for 50% of the world's population) and over 300 Seychelles Fodies *Foudia sechellarum* in 20 large aviaries for several months (Millet & Shah, 2001).

Paralleling the increased interest in mammal pest control and island restoration, there has been some technical progress in the development of mammal attractants and lures (e.g. Clapperton *et al.*, 1994; Saunders & Harris, 2000) and further research is currently under way. There has also been much progress to prevent aversion development in animals exposed to previously used poison (Cook, 1999; Hickling, Henderson & Thomas, 1999). Much effort has also been devoted to the development of baits that do not attract non-target species, by diminishing both hazard and exposure (e.g. McDonald *et al.*, 1999; Morgan, 1999).

Several toxins are routinely used for the control of small introduced mammals, mostly in Australia and New Zealand: sodium monofluoroacetate (or 1080), pindone, cholecalciferal, brodifacoum, cyanide, strychnine. Of these, 1080 and brodifacoum are currently the most widely used, with brodifacoum, an anticoagulant, being the most common used toxin (Innes & Barker, 1999). For example, brodifacoum was used in 28 of 33 mammal eradication programmes undertaken by the New Zealand Department of Conservation (DOC) in the last decade (Innes & Barker, 1999). Although 1080 is usually rapidly eliminated from live animals and rapidly broken down by microbial activity in baits, water and soil, it is still highly toxic to a wide range of animals and may pose problems for non-target species (Innes & Barker, 1999). By contrast, brodifacoum shows a greater persistence: it accumulates in vertebrate tissues, it is insoluble in water and only slowly broken down by microbial action (Dowding *et al.*, 1999; Eason *et al.*, 1999). It is however less toxic for non-target species (Godfrey, Reid & McAllum, 1981). Although it is often known for the control of brushtail possums in New Zealand, brodifacoum has also been used in successful mammal eradication programmes on islands for both rats (Buckle & Fenn, 1992; Taylor & Thomas, 1993; Brown, 1997; Empson & Miskelly, 1999) and rabbits (Godfrey *et al.*, 1981; Merton, 1987).

During control programmes, poisoned animals (alive or dead) may be eaten by carnivores or scavengers, potentially resulting in secondary poisoning. This may provide conservation managers with an efficient multi-species tool for controlling mammalian pests on off-shore islands where their re-invasion is unlikely (Gillies & Pierce, 1999). Indeed, it is now known that mammal predator numbers can be effectively reduced by secondary poisoning with both brodifacoum and 1080 following rat control (McIlroy & McIlroy, 1992; Heyward & Norbury, 1998; Gillies & Pierce, 1999; Murphy *et al.*, 1999), to such a point that deliberate secondary poisoning has been suggested as a potentially cost-effective predator eradication tool (Alterio, 1996).

There is also a risk that pest species surviving a poison operation aimed at other pest species in the same area may subsequently develop an aversion to the poisoned baits and become exceedingly resistant to further toxin-based control (Hickling *et al.*, 1999). This not only can induce a learned aversion to the chemical used in the poisoned paste but also can cause animals to exhibit 'enhanced neophobia', as seen in possums following rabbit control (Hickling *et al.*, 1999). All these points highlight the need for developing medium and long-term strategies for the use of poisoning for mammal pest control.

(*e*) *Advantages and disadvantages of traditional methods*

Despite improved efficiency, especially concerning the recent progress for the eradication of rodents from larger islands, traditional methods remain logistically difficult, are costly in material, manpower and time, and generally have little specificity. These methods can have a low cost-efficiency on large and/or poorly accessible islands. This has led several biodiversity managers to look for alternative strategies, among which biological methods have an interesting potential (see Table 1).

(2) Biological methods

Biological control is the control of pest species by an enhancement of a different species that decreases the reproduction or survival of the target pest species. Several categories of natural enemies are used in this context, but predators and pathogens have historically been the most used for mammals.

(*a*) *Decreasing the survival of alien mammals*

The main biological control strategy is the introduction of natural enemies of the target species, with the aim of decreasing the survival of the target species. Parasitism, predation and competition are the three possible processes involved, although the former is the most common: the natural enemies most used for mammal pests are microparasites (viruses and bacteria). Concerning pathogen introductions, the best-known example is the myxoma virus, introduced in Australia and then on several islands to control rabbit populations (Brothers *et al.*, 1982; Chapuis, Chantal & Bijlenga, 1994*b*; Fenner & Ross, 1994; Chekchak *et al.*, 2000). Despite problems linked with virulence and immunity evolutions in large populations, this virus was successful at removing entire populations in many cases (Flux, 1993). More recently, the introduction of the rabbit haemorrhagic disease (RHD) in New Zealand and Australia has led to mass mortality in rabbit populations, but these were not scientifically planned eradications for conservation biology purposes and are extremely contentious. Regarding RHD in Australia, the disease has now been established in Australia for over five years and has reduced the abundance of rabbits across Australia, most spectacularly in arid, inland Australia (Brian Cooke, personal communication). Another example is the eradication of feral cats from Marion island, using the feline panleucopenia virus (van Rensburg, Skinner & van Aarde, 1987). In this case, a combination of biological control (the virus) and traditional control (hunting) was required to complete the eradication of cats, as neither technique alone would have been successful.

Generally, biological control by predator introduction has been disastrous. The presence of mongoose and or domestic cats on most islands is the result of failed attempts to control rodents, rabbits or snakes. Most of the time, these mammalian predators ignored the target prey and instead turned to native prey, which were often easier to locate and/or kill because they evolved in the absence of these terrestrial predators (Atkinson, 2001). The consequences of predator introductions for biological control of rodents are sometimes more subtle or unpredictable. For example, in the West Indies, rat control has often been attempted by mongoose introduction. But mongooses are typically terrestrial, and their presence often caused the rats to shift habitats in favour of arboreal niches (Seaman, 1952). As a result, not only have rats become less available to mongooses, which started preying on many species of ground birds, but also the introduction of mongooses led to an increased predation on tree nesting birds by the now more arboreal rats.

One of the rare attempts at controlling a mammal pest through competitive processes concerns a successful programme to eradicate arctic foxes from the Aleutian islands. This programme used the competitive superiority of red foxes: some sterilised individuals were introduced, and then removed once the arctic foxes were extirpated (West & Rudd, 1983; Bailey, 1992, 1993). This original method was highly efficient and, although very costly, certainly deserves more interest.

(*b*) *Decreasing the reproduction of alien mammals*

The common goal of control methods, either traditional or biological, is to decrease the survival of individuals of the target species. In some species with a short generation time and high reproduction rate (typically rodents and most other pest species), decreasing survival may not be as efficient as decreasing reproduction. During the last decade, a significant impetus of research effort has been devoted to the development of techniques lowering the fertility of mammal species. Notably, as an alternative to increasing the death rate, immunocontraception aims to reduce birth rates (Tyndale-Biscoe, 1994). Immunocontraception is a process by which the immune system of an individual is induced to attack its own reproductive cells, leading to sterility (Tyndale-Biscoe, 1994). This is achieved by infecting individuals with a protein derived from the follicular layers, which activates the production of antibodies against its own gametes, thereby blocking fertilisation (Bradley, Hinds & Bird, 1997). Depending on the size of the animal and the level of control desired for its population, infection is achieved by injection, as for large mammals (Kirkpatrick *et al.*, 1997) or by bait, typically for small carnivores (Bradley *et al.*, 1997). More recently, attention has been focussed on new methods that would allow infection through living vectors, typically genetically engineered viruses, bacteria or macroparasites. Virus-vectored immunocontraception (VVIC), for example, utilises a species-specific virus to disseminate this vaccine through a pest population by placing the gene encoding the reproductive protein into the genome of the virus (Tyndale-Biscoe, 1994). This potentially powerful new technique would be used mainly for rodents and small herbivores, such as rabbits and possums (Cowan, 1996; Rodger, 1997; Smith, Walmsley & Polkinghorne, 1997), but also could be very efficient for small carnivores as well (Bradley *et al.*, 1997; Pech *et al.*, 1997; Verdier *et al.*, 1999; Courchamp & Cornell, 2000).

Control by VVIC presents numerous advantages over traditional control methods based on increases in mortality. Because it does not result in animal suffering, the general public, ethics committees, and animal rights organisations are more likely to accept this method (Loague, 1993; Cowan, 1996). Moreover, unlike chemical control, this method is less harmful to the environment. It also shares the advantages of biological control over traditional methods: being self-disseminating, VVIC can be used to control large areas (even where accessibility is limited), for a minimal cost (Chambers, Singleton & Hood, 1997). Finally, it seems to be the most host-specific of current methods (Tyndale-Biscoe, 1994). Although genetic engineering of a pathogen is a prerequisite, preliminary studies are very encouraging (Moodie, 1995), and suggest a possibility of a durable sterilisation of the host without modifying its social behaviour. The main disadvantages of this emerging method include the irreversibility of the process once the vector is released (Nettles, 1997), the potential for development of host resistance, the need for the engineering of a genetically modified vector (Bradley *et al.*, 1997), a slow response time for definitive results as well as for monitoring progress (McCallum, 1996) and a low public acceptance of the release of genetically engineered organisms.

We cannot overemphasise that biodiversity conservation planners must be cautious and responsible with respect to the use of control through release of pathogens in general, and of genetically modified pathogens in particular. VVIC methods should only be used once they are deemed truly reliable and only in conditions of maximum security for the wild or domestic non-target populations inside and outside the target ecosystem. In this context, the application of VVIC methods, if

the local biotic and abiotic conditions are favourable, should be first (and perhaps only) used in remote uninhabited islands, where potential unforeseen effects could be naturally and more easily circumvented (Courchamp & Cornell, 2000).

(*c*) *Advantages and disadvantages of biological methods*

The main advantages of biological control are a potentially very good specificity, and a higher cost efficiency since it corresponds to a self-disseminating control method, or what could be characterised as 'release and watch' effort. This can be crucial when large areas, low accessibility or low density prevent the use of traditional methods, as is often the case on islands (Courchamp & Sugihara, 1999).

However, there are also a number of disadvantages with this type of method, which have precluded the generalisation of its use. One drawback is the poor specificity of some of the species that have been used to control pest species. Although it could be argued that most failed attempts at introducing natural enemies to control exotic mammals could have been foreseen with a minimal scientific pre-release survey, the history of island restoration is dominated by catastrophic failures.

It is important to stress that biological control meets with growing success as programmes are scientifically established and implemented. The main criticisms of biological control concern programmes that have been conducted without adequate preliminary study of the ecosystem (see below). One must keep in mind that biological control, at least in the case of remote oceanic islands, is nothing less than the introduction of an alien species in order to solve a problem of a previous alien introduction. History has shown us that often more harm is generated from such interventions. In the case of attempted control of rodents in sugarcane fields in Jamaica, cane growers introduced ants (*Formica omnivora*), which did not reduce rat numbers but soon became a problem themselves. To remove rats and ants together, it was then decided to introduce venomous Marine toads (*Bufo marinus*). But toads still did not control rats, and became a pest themselves. Finally, small Indian mongooses were introduced to control rats and toads. Mongooses failed to control either, and began preying on native birds, posing new threats to wildlife (Lever, 1994).

The other main disadvantage of biological control is the lack of control once the natural enemy population is released. The advantage of a self-disseminating system may become a problem if the control method is not working properly, which it rarely does. Having no control over the released population also implies difficulties for the monitoring of the efficiency of the operation. Introduced species and their new environment interact in ways that are hard to predict even if we know all the details, which is seldom the case. Since we know little in most cases, accurate predictions are extremely difficult and entail unknown risks to every single introduction, including those for restoration purposes. This has led some authors to describe scientifically controlled introductions as 'a serious fallacy' (Ebenhard, 1988).

The third important problem in using biological methods to reduce mammal pest species is of an ethical nature. Because pathogens are more often used for the biocontrol of mammals, there are more concerns about safety and ethics than in classical biocontrol programmes in agricultural ecosystems. Most pathogens, for example, induce diseases that cause suffering before killing the individual host, which can be difficult to justify. Even though some animal rights association representatives agree that the death of a few individuals by control is more acceptable than the loss of hundreds of individuals, either by direct or indirect actions of invasive species, they rarely accept methods that induce suffering. The use of pathogens also generates concerns for the security of other populations and other species, including humans.

The risk of the control organism making its way back to the ecosystem from which the pest originated is a potential disaster. As has been shown with the introduction of the myxoma virus in France to reduce rabbit numbers, and more recently with the introduction of RHD in New Zealand for the same goal, it may be difficult to control fully the intentional introduction of micro-organisms by persons who are not fully aware of (and/or interested by) the potential ecological effects of such actions. It is particularly relevant to the case of the possums invading New Zealand, and native to Australia. Biological control of alien possums has been considered (Barlow, 1994; Cowan, 1996), but the risk such a programme represents for the native Australian population is a major drawback.

It is also often stated that biological control is more efficient at controlling than eradicating populations. One argument is that because of a long co-evolution, and the associated evolutionary 'arms race' of both the target species and its natural enemy (generally the host and its pathogen), the natural enemy has lost the ability to eliminate completely the host population (for it would disappear too). One point that is not taken into account in this reasoning is that the conditions may be very different from those where the two species have co-evolved. In particular, the pathogen may rarely, if ever, have had to cope with a single population in complete

isolation. It is conceivable that many pathogens have maintained characteristics that cannot eliminate a host population in contact with others, but which would eliminate a hypothetical population with no immigrants. Similarly, because of the founder effect, the genotype diversity of alien host populations may be reduced relative to mainland populations, and thus may not encompass enough variability to allow immunity of a fraction of the host population. A mammal population that would normally survive in presence of a virus may not do so if no immune individuals are present. Another argument is that some pathogens can be used in conditions that are not encountered naturally, for example if the virus is genetically modified.

A general question has been whether the advantages of biological control outweigh its risks (Simberloff & Stiling, 1996*a*, *b*; Thomas & Willis, 1998). Without entering a long debate that lies outside the purpose of this review, it is worth emphasising two points. Firstly, even though it is true that there have been more failures or semi-successes than complete successes with biological control, one should bear in mind that most of the programmes involving biological control have been carried out at a time when careful feasibility and impact assessments were not the rule. Thus past failures mostly reflect that these programmes were not optimised, but they do not rule out the possibility of using the same tools to get better results. It is conceivable that the small number of studies concerning biological control of vertebrates is due more to a lack of confidence and of effort than to a lack of potential (Wood, 1985). Secondly, even in the days when scientists and conservation managers did not benefit as we do from the hard learned lessons of initial failures or partial success, there were nevertheless interesting cases of impressive achievements through biological control. For example, the eradication of the cats from Marion Island would not have succeeded without the initial elimination of some 80% of the population by the introduction of feline panleucopenia virus. Biological control thus remains a potentially powerful method that merits future investigations.

Despite all the positive and negative examples given above, it should be recognised that neither biological control nor traditional methods are a universal panacea. Each method has specific disadvantages and each particular case of invasion should be managed with an adapted and specific plan of action. The most appropriate strategy will be more often the simultaneous use of biological, chemical and mechanical control methods. It is commonly accepted that a combination of strategies, that is, integrated pest management, is the best response to mammal introductions.

(3) Strategies

The dynamics of invasion can be divided into four stages: (*i*) the transport of organisms to a new location; (*ii*) the establishment of the invading species at this location; (*iii*) the consequences of the growth in numbers of the invading species for other species in the invaded community (interspecific interactions); (*iv*) the spatial spread of the invading species from the initial successful population (Hastings, 1996; Shea & Chesson, 2002). Although the best option is to act at stage (*i*) (I.U.C.N., 2001), many organisms have already been introduced and the strategies for invasive species control will differ according to the stage of invasion.

(*a*) *Eradication of established populations*

Many authors have insisted on investing time in optimising the strategy used to reduce or eradicate an alien species from an island. In particular, to ensure the best chances of success, an investigation of the historical, ecological, social and economic situation of the island must be completed. Moreover, the aim of the programme must be clearly defined in order to identify the available options and their respective likelihood of success.

Applying methods that were successful elsewhere, even if the same alien species is concerned or other aspects seem identical, is not necessarily a guarantee of success. Each case must be thoroughly studied independently, with careful planning of the protocol and how to institute it.

Many authors have given rather precise guidelines based on their experience with certain species, and much relevant information can be found either in published studies or from the national agencies that led or financed the programmes (e.g. the DOC). There are, however, a number of basic points that can be cited as important for the success of a programme. To be successful a control programme requires (but is not guaranteed by): (1) good planning (on all aspects, from logistics to ecology); (2) a good pre-control study of the situation (see below); (3) careful choice of the selected method or suite of methods (including good timing); (4) sufficient and lasting financial and political support to complete eradication (even if it takes time); (5) public support (e.g. Veitch & Clout, 2001).

It is outside the scope of this review to expand on each of these points. We provide a summary of the most important recommendations for a control programme (Fig. 4), but each aspect certainly deserves more detailed exposition. As an example, a few words can be said concerning the timing of the programme. This point is crucial for the success of the programme on several

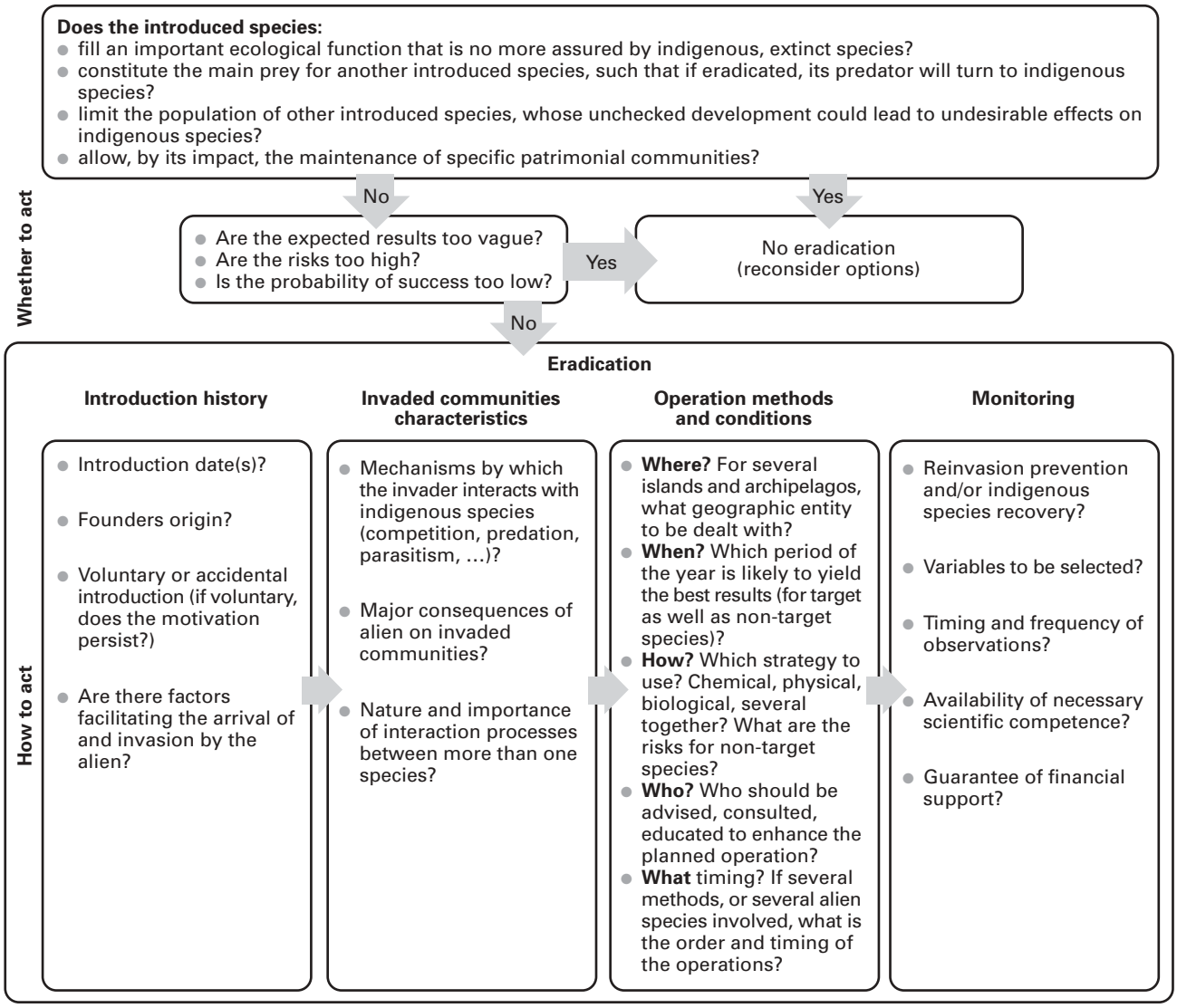

Fig. 4. Decision diagram for the eradication of mammal species from islands.

grounds, depending on the type of alien population i.e. whether it is an established or new incursion. For the former, the timing of the operation is of major importance because biotic and abiotic conditions play a role in the population dynamics of alien invasive and indigenous species (Innes & Barker, 1999). These conditions (climate, fluctuating dynamics of other species, etc …) can promote or hinder the growth of the target population, and it is better to have these forces acting with the protocol rather than against it. For example, it may make a difference to launch an eradication programme on a species when the food resource of that species (either plant or prey) is lacking, e.g. in winter or during migratory bird absence (Chapuis *et al.*, 2001). It may be also judicious to act when the target species is more vulnerable because it has less cover (e.g. vegetation cover is lacking in winter on many islands) or because it has a restricted home range (e.g. arctic foxes avoid the inner parts of the Aleutian islands in winter, restricting their movement to a coastal strip, and they are therefore more readily removed from those islands during this period, Steve Ebbert, personal communication). If several introduced mammals are present, careful thought must be dedicated to the strategy to be implemented. There is no currently admitted prioritisation scheme, but any such system would have to consider, among others, the known effect(s) of the alien invasives, the probability and rate of spread of the alien invasives and probability and cost of achievable eradication, and the interactions among the alien invasive species as well as with native communities. It is, however, our deep conviction that often all

should be done to remove each introduced species, and that prioritisation schemes should equate to eradication timing not to a choice of which species to remove and which to leave. We describe in the fourth part of this review why and how the timing of such programmes is crucial for the restoration of the ecosystem.

(*b*) *Eradication of new incursions*

Eradication of new incursions also needs good planning, careful selection of methods, financial backing, and public support, but the reaction time is fundamentally different. Indeed, one needs to react rapidly when an invasion is first documented, because it almost invariably starts with few individuals in a localised place and this is easier to deal with than a large and expanding (or even established) population. Launching a control operation as close as possible to the beginning of the invasion, and if possible before the species is established, is by far the best option for success. Thus, the best approach is to have in place a strategy/policy that allows for rapid action. In this regard, it is crucial that a legal mechanism should exist that allows action to be taken swiftly by an appropriate authority or institution that is prepared for such situations. Similarly, public support must exist beforehand for the fact that rapid action may be needed at some stage. Accordingly, public consultation should be focussed on a general strategy for emergencies, rather than on a particular case. The 1999 exemplary case of the mussel-invaded marina near Darwin (Kaiser, 1999), immediately dealt with by Australia's Northern Territory government and the CSIRO was a great success and provides a model example for quick decision and rapid response in future cases.

IV. CONSEQUENCES OF ALIEN MAMMAL CONTROL

(1) An increased number of successful control programmes

Control or eradication programmes have been increasing in number in the last few decades. This is partly because of a shift in ecologists' interests, with an increased interest in conservation biology and restoration ecology (Young, 2000) in general, and by biological invasions and alien species in particular (there were more published articles with 'Biological Invasion' as key words for the first trimester of 2002 than for all years between 1981 and 1990). It is also because new protocols have allowed successful eradication of mammals from islands in conditions that were previously judged impossible to overcome (Dingwall, Atkinson & Hay, 1978). For example, it was admitted as late as 1976 that islands of more than 1 ha could not be freed of rats (Atkinson, 2001). However, improved protocols led to their eradication from the 3253 ha Langara Island (Bertram & Nagorsen, 1995; Drever & Harestad, 1998; Taylor *et al.*, 2000). This was made possible by both the availability of the 'one shot' anticoagulants in baits that are attractive to rodents and the development of a successful technique of dispensing poison baits from fixed stations spaced such that at least one station lies within the home range of each rat (Taylor & Thomas, 1989; Taylor & Thomas, 1993; Clout, 1999). This has been recently replaced by aerial poisoning using the Global Positioning System (GPS) to distribute baits accurately and evenly, a method less time-consuming than fixed bait stations. This innovative method has created a new impetus both to research on control protocols and to eradication campaigns on new (large) threatened insular ecosystems. Similarly, the use of the GPS and Geographic Information System can assist hunters in covering more efficiently the ground without leaving huge gaps and in coordinating their movements with extreme precision, thereby much improving overall hunting effectiveness (Patry, 2001).

Eradication programmes are hard to quantify in terms of their success or failure. Data are difficult to trace for many parts of the world and such programmes are not systematically reported in the classical scientific literature. This is especially the case concerning failures, which are more rarely reported. One good counter example of this is New Zealand, for which such data have been carefully recorded. The total number of completed mammal eradications was 153 at the end of 2001, with a further 24 ongoing operations, as compared with three stopped and nine failed programmes (C. R. Veitch, personal communication). In total, failures or discontinuation of programmes account for a mere 7.3% of programmes not currently in progress. In total, these eradication programmes concern 144 islands, three quarters of which are now successfully completed, 2.8% were stopped midway in the operation, and 7% were considered failures (C. R. Veitch, personal communication).

Table 2 shows the largest islands from which 14 of the most notorious mammal island invaders have been successfully eradicated. This Table may lack completed programmes that are larger that those mentioned: not all programmes are presented in classical scientific literature, and, in most cases programmes need time in order for the managers to claim with success confidence. For example, rabbits have very probably been eradicated from the 8 km^2 St Paul Island, France, but the programme has only recently been completed, and several

Table 2. *Largest islands where eradication was achieved for 14 introduced mammals.* [1]*Eradication on part of the island, managed population on the rest;* [2]*Terres Australes et Antarctiques Françaises;* [3]*Galapagos National Park Services;* [4]*Charles Darwin Foundation;* [5]*23 years of control and four years of eradication*

Mammal species	Island and country	Size (km^2)	Method	Supervising institution	Estimated number eradicated	Date of completion	Operation duration (years)
Arctic fox	Attu, Alaska, USA	905.8	Shooting & trapping	US Fish & Wildl. Service	373	1999	?
Black rat	St Paul, F	8	Poison	TAAF	8–12 000	1999	3
Brushtail possum	Rangitoto-Motutapu, NZ	38.5	Poison, trapping, dogs	NZ Dpt Cons. (DOC)	21 000	1997	8
Brush-tailed rock wallaby	Rangitoto-Motutapu, NZ	38.5	Poison, trapping, dogs	NZ DOC	12 500	1997	8
Cat	Marion, SA	190	Biocontrol, trapping & shooting	S. Afr. Dpt Envir. Affairs	2790 + 1124	1990	4
Cattle	Amsterdam, Indian Ocean, F	55[1]	Shooting	TAAF	1059	1989	2
Goat	San Clemente, USA	148	Shooting, trapping	US Navy	30 000	1992	19
Mouse	Enderby, NZ	7.1	Poison	NZ DOC	?	1995	3
Norway rat	Langara, Can	32.5	Poison	Can. Wildl. Service	3000	1995	< 1
Pig	Santiago, Galapagos, Equator	584.6	Shooting, dogs, poison	GNPS[2] CDF[3]	19 210	2000	27[4]
Pacific rat	Kapiti, NZ	19.7	Poison	NZ DOC	?	1996	< 1
Rabbit	Enderby, NZ	7.1	Poison	NZ DOC	?	1995	3
Red fox	Dolphin, Aust	32.8	Poison	Aust. Dpt Cons. & Land Managt.	30	1980	10
Sheep	Campbell, NZ	112.2	Shooting	NZ DOC	?	1991	21

additional years of checking for recovering populations will be required before it is classified as a definitive success (T. Micol, personal communication). Similarly, the Norway rat may have been eradicated from the 113 km^2 Campbell Island, NZ, which would make it the world's largest successful rat eradication project. As is the case for St Paul Island, time is needed to assess the real state of the population and the population cannot be officially declared eradicated until the island has been confirmed rat-free after two years. It seems however that this major operation, which included an aerial poisoning campaign by the DOC, involving the airdrop of 120 tonnes of anticoagulant baits, is already a great success. A forthcoming project also aims to eradicate goats, from the 2100 km^2 of Isabela Island, Galapagos (Kaiser, 2001), which would make it the largest eradication project involving this species. A further problem of this table is the eradication of mammals living on small portions of very large islands. For example, the feral goats eradicated from the 460 km^2 Auckland Island, NZ, actually lived in only approximately 40 km^2 (Chimera, Coleman & Parkes, 1995). This poses problem for the definition of islands. In this review, we do not consider Great Britain, Australia or New Zealand as islands, although they are geographically defined as such. Therefore, we did not include cases such as the coypu, *Myocastor coypus*, eradicated from England, considered one of the most impressive achievements to date in this context (Gosling & Baker, 1989).

Some countries have been leaders both in research and implementation, most notably New Zealand and Australia. This is probably because they are countries with many islands, with a flora and fauna that evolved in the absence of many mammals (there is an absence of indigenous terrestrial mammals in NZ; Atkinson, 2001), and thus they have suffered more from species

Table 3. *Effort of mammal eradication for a few island groups. Data were obtained from Ian McFadden and John Parkes (NZ Is), Keith Morris (Western Australian Is), Steve Ebbert and Vernon Byrd (Aleutian Is), Fred Kraus (Hawaiian Is), Michel Pascal (French Is), Bernie Tershy and Josh Donlan (Mexican Is). *French islands include islands in different parts of the world: Brittany, Mediterranean, Caribbean, sub-Antarctic and Indian Ocean*

	Number of islands	Number of eradications	Species	Methods	Mean size (km^2) [min, max]
NZ Is	107	153	Mouse, cat, rabbit, brushtail possum, wallaby, ship rat, Norway rat, Pacific rat, stoat, ferret, pig, goat, cattle, sheep, red deer	Poison, shooting, dogs, traps	3.07 [1, 11 216]
W. Aus. Is	45	57	Mouse, Norway rat, fox, cat, rabbit, goat	Poison, shooting, traps	5.28 [0.01, 42.67]
Aleutian Is	34	36	Fox, cattle	Shooting, poison, traps, biological control	25.05 [0.25, 91.74]
Hawaiian Is	9	13	Cattle, sheep, goat, rabbit, Polynesian rat, ship rat	Poison, shooting, traps	50.55 [0.04, 18.85]
French Is*	39	46	Rabbit, goat, cattle, ship rat, Norway rat, mongoose	Poison, shooting, traps	27.3 [0.01, 55]
Mexican Is	24	32	Cat, rabbit, goat, sheep, mouse, ship rat, Norway rat	Poison, shooting, dogs, traps	3.3 [0.02, 20]

invasions than less isolated countries. Also undeniably important is the political will, the social maturity on environment-related matters and the economic ability to act to preserve the native fauna and flora. These three points unfortunately are not always present together in most other countries. The leadership of New Zealand and Australia in the restoration and management of invaded ecosystems is eloquently reflected in hard data showing the number of islands subject to mammal eradication programmes (see Table 3).

According to J. H. Myers *et al.* (2000), six factors contribute to successful eradication. First, financial support must be sufficient both in quantity and in duration. Second, all the necessary authority must be invested in the group in charge of the programme to ensure the protocol can be enacted. Third, the life history, reproductive biology, behaviour and dispersal ability of the invading species must make it susceptible to control procedures (implying that these parameters must be known before the implementation of the programme). Fourth, the programme should take into account the necessity of stopping the influx of individuals into the ecosystem. Elimination of mammals from islands might in this context be more successful because the probability of reintroduction can be easily reduced. Fifth, detection of the target species must be possible even at low densities, so that eradication can be best achieved and new introductions can be recognised before the species becomes widespread. Finally, complementary restoration actions might have to be taken to accompany the elimination of the pest species if it had assumed an important role in the ecosystem and if its removal can trigger further problems (Towns, 1997).

(2) On the definition of successful programmes

The increased number of successful control programmes must not mask the fact that in too many cases the so-called success is simply equated to achieving elimination of the target species, which cannot always be rightly called a success. As the term 'impact' requires definition (Parker *et al.*, 1999), it is also important to consider the meaning of 'successful' in the context of

island restoration. The removal of the alien invasive species might be sufficient to restore the ecosystem if the invader did not have sufficient time to inflict irreversible damage to the normal functioning of the ecosystem. Often, the introduced mammal species will have had a quantitative impact on species it interacts with, and after its elimination, natural ecological processes will drive the interacting indigenous species back to their former equilibrium. However, unexpected changes may arise from the sudden removal of the alien species. Introduced species may have had an irreversible impact that requires further restoration actions. The simplest case is when an entire population or species has been eliminated from the considered islands and potential source populations are too far away to allow a rapid natural recolonisation. Another case is when the habitat has been degraded to a point where regeneration cannot be made rapidly enough to avoid the collapse of whole communities. In addition, the introduced species may have changed the local biotic and abiotic conditions in such a way that its sudden removal cannot facilitate the reestablishment of previous states, and sometimes may even lead to further threats to the native species. Dramatic changes to islands following the eradication of introduced species have been sometimes reported but less often quantified. The number of cases where an introduced species removal has led to even greater risks for the protected species is unfortunately high, and cases are still occurring with current restoration programmes. These situations, and recommended actions which help to reduce their occurrence, are discussed in the next section.

(3) Positive and negative effects of mammal eradication

There are a number of basic recommendations that can be made concerning the restoration of invaded islands (see Fig. 4, but also Parkes, 1990; Veitch & Bell, 1990; Towns & Ballantine, 1993; Towns, 1997; Atkinson, 2001; Saunders & Norton, 2001, for examples). Because removing a species from an ecosystem, even an alien species, can have diverse consequences, both desired and undesired, it is crucial to be able to quantify and to predict these effects. Indeed, quantification of the desired effects can lead to improvement of control methods as well as a better justification of the control programme for biodiversity conservation. Adequate knowledge can also help predict and thus prevent the undesired, and previously unexpected, effects.

Towards this end, one crucial point is to assess fully and precisely the current state of the ecosystem in order to establish a zero state qualification and quantification. As indicated in Fig. 4, it is necessary to assess the current state of invaded ecosystems prior to drastic interventions such as species removal, even if it is an introduced species (Thomas & Willis, 1998). This step is essential to allow a rigorous estimation of the effects of the invading species and of the expected effects following its removal. It is also fundamental to implement the best control strategies qualitatively as well as quantitatively (e.g. see Choquenot & Parkes, 2001), according to local conditions. Finally, it is indispensable to predict, and thereby avoid, the potential side effects of alien invasive removal. Researchers and managers are increasingly aware of community level interactions because pest control can have unforeseen repercussions such as mesopredator (or competitor) release and prey switching (Murphy & Bradfield, 1992; Dowding & Murphy, 2001). From some of the long-term pre-removal studies that have been conducted, it has been concluded that 10 years of study prior to an eradication may be required for post-eradication observations to be compared adequately (R. Veitch, personal communication). However, in many cases it is difficult to wait that long before acting on the invading population.

It may sometimes prove essential, however, in order to avoid the 'surprise effects' following eradications that are often mentioned by programme managers. The associated catastrophic chain reactions emphasise how crucial pre-control studies are for successful ecosystem restoration. Unexpected and undesired secondary effects are in general more likely to occur when ecosystems contain more than one invading species (that is, the great majority of islands), when invading species are in the late stages of invasion and when they have eliminated native species and replaced them functionally (Zavaleta, Hobbs & Mooney, 2001). In general, secondary effects following the sudden removal of alien species are very damaging to the ecosystem, and careful monitoring of the communities prior to any control action has the potential to prevent such catastrophic chain reactions. As we have seen in detail, exotics interact with native species as well as among themselves, creating a complex pattern of direct and indirect effects that can be extremely difficult to comprehend, let alone to predict. For example, the removal of one exotic species can favour the expansion of other exotics that were held in check by the removed species. This has been called the Sisyphus effect (Mack & Lonsdale, 2002).

The removal of herbivorous aliens such as rabbits and goats can lead to a release of exotic plants that, in the absence of browsing, are more competitive than native plants, leading to an explosion of such weeds. There are very few examples of islands on which the presence of feral goats or other browsers is more

positive than negative (Daly & Goriup, 1987). But 'the immediate removal of introduced browsing mammals … may lead to unexpected and even undesired results' (Taylor, 1968). For example, a recent project was undertaken to remove goats and pigs from the Sarigan Island, one of the Mariana Islands, in order to stop and reverse the loss of forest and accompanying erosion, and thereby to protect the endangered native fauna (Kessler, 2001). In only two months, nearly 1000 animals were shot from this 500 ha tropical island, allowing rapid recovery of both plants and monitored vertebrates. The project was successful in reversing the trend of forest loss, and improving the habitat for local birds and bats. However, the removal of alien mammals has allowed the introduced vine *Operculina ventricosa* to thrive and spread, to such a point that part of the island is now covered by an almost uninterrupted carpet of vine, with unknown consequences for the future of the whole ecosystem (see Fig. 5). The introduced mammals had previously held the vine at a low density such that minimal pre-operation monitoring had not identified this threat.

A similar example, concerning predatory relationships, is what has been called the mesopredator release effect (Soulé *et al.*, 1988; Courchamp, Langlais & Sugihara, 1999*a*). This process predicts that once top predators are suppressed, a burst in growth of intermediate predators, or mesopredators, may follow that leads to prey extinction. A classic illustration of this is the removal of cats from islands where rats are also present: the elimination of feral cat populations from such ecosystems could lead to a severe negative impact on the endemic species, through a rapid growth of rodent populations following the removal of their predators. Attempted reduction of the cat population of Amsterdam Island is alleged to have caused a compensating increase in the number of rats and mice, and so has been abandoned (Holdgate & Wace, 1961). It is axiomatic that removal of cats (as well as similar predators) from islands must be accompanied by a control of their introduced prey (herbivores as well as omnivores or carnivores).

However, outcomes of changes of these already perturbed trophic webs are not intuitive and intervention as dramatic as species eradication should, when possible, be preceded by careful empirical and theoretical studies of the whole ecosystem. Sometimes, the presence of a few individuals of a species that may appear of minor importance can mask powerful interspecific interactions. Thus, avoiding unexpected changes may not be as simple as removing the most obvious introduced species: in this case, removing both cats and rats might not be sufficient. On Bird Island, in the Seychelles, eradication of the introduced rat population led to an explosion of the exotic crazy ant *Anoplolepis longipes*, now threatening colonies of Sooty terns and the endemic skink *Mabuya seychellensis* (Feare, 1999). Invertebrates are often overlooked in insular restoration programmes, perhaps because they are not as conspicuous as mammals, and may not be considered as harmful. However they are species anchoring the base of trophic pyramids. This implies that in some cases, anything but the most finely tuned intervention may lead to more damage than benefit to biodiversity, begging the question as to whether intervention should be done at all. Nevertheless, we can still gain from unwitting mistakes made in the past, since all results contribute to an understanding of island ecology and can be used in future conservation actions on other islands.

The other major aspect, also often overlooked in many control programmes, is post-eradication monitoring. This long-term surveillance is crucial for many aspects. First, it allows an assessment of the success of the proximate goal of the programme: durable elimination of the target species, with no further invasion. It also can quantify the ultimate goal of this elimination: the reversal of a decline of affected native species, their increase, or their recolonisation of the ecosystem. This part is not to be neglected: it may be surprising to learn that assessment of whether expected positive effects (restoration of initial ecological conditions) have been reached is still not systematically included in restoration programmes. Historically, the lack of assessment in control programmes may have led to a slower realisation that alien invasive elimination does not systematically lead to ecosystem restoration. Third, post-eradication monitoring allows the early detection of the increasingly described 'unexpected changes' that can be disastrous in closed and non-redundant ecosystems.

The before-after-control-impact study design, also called BACI, is used to assess the impact of some event on variables that measure the state of an ecosystem. The design involves repeated measures over time, made at one or more control sites and one or more impacted sites, both before and after the time of the event that may cause an impact (Manly, 2000).

The need for long-term monitoring is well illustrated by the famous case of Round Island, Mauritius, a unique ecosystem endangered by introduced species. This 151 ha island is an important refuge for several endemic plants and reptiles that have disappeared from the mainland of Mauritius (Bullock, 1977). The effects of goats and rabbits introduced in the early 19th century were so large that the main habitats were greatly modified and several native species were threatened with

Fig. 5. Landscape of Sarigan Island, before (1996) and after goat control (1999). In the presence of the goats, the destruction of the vegetation is very important (the scale is given by the goat in the centre, shown by the black arrow). Two years after goat eradication, the introduced vine has spread, eventually covering and asphyxiating the native vegetation that had already begun to recover. Photograph courtesy of Curt Kessler.

extinction (Bullock, 1986; North & Bullock, 1986). By 1986, rabbits and goats were eradicated from the island (Merton, 1987), and a programme was implemented to monitor the changes in the extent and composition of key elements of the biota, such as vegetation, invertebrates and reptiles (North *et al.*, 1994). Long-term monitoring now shows that a satisfactory regeneration and recolonisation of plant species formerly threatened by alien grazers is somewhat impeded by a new threat on native plants that is linked to the colonisation and rapid spread of several alien invasive plants (North *et al.*, 1994; Bullock *et al.*, 2001).

V. CONCLUSIONS

(1) There are many other aspects of the impact of mammal invaders and their control that are also important to discuss, and we hope that this review will trigger discussions on complementary elements of this theme. As a conclusion, two main points need to be especially stressed. First, despite the rather pessimistic image of the current situation depicted here, there is a growing number of successful control programmes, both because there are more programmes, indicating that researchers and conservationists are becoming more aware, interested and involved in the problems of biological invasions, and because programmes are better planned, with lessons learned from past experiences. Thus, some optimism is not unjustified and we foresee a sustained increase of successful restorations of insular ecosystems. The best strategy will always depend on the specific conditions and will often involve integrated pest management, combining the advantages of mechanical, chemical and biological methods, while avoiding their respective disadvantages through careful planning. As scientific and managing tools improve and become more available, we hope that other countries will follow the example of New Zealand and Australia with respect to their cogent political, financial and scientific resoluteness in controlling biological invasions and their effects.

(2) Second, if one lesson only were to be learned from past failures or semi-successes, it is that a restoration programme cannot be limited merely to eradication. A thorough pre-eradication assessment and long-term post-eradication monitoring (not limited to those communities directly linked to the eradicated species, as there can be unexpected indirect consequences) are both necessary. Only in this way will we ensure the best chances of success of full and durable ecosystem restoration. Moreover, these techniques will allow us to benefit fully from past experiences and enrich the knowledge of conservation biology as well as more general areas of modern ecology.

(3) We wish to conclude on the potential of insular ecosystem restoration programmes as sources of valuable knowledge for fundamental research. Experimentation in the epidemiology of biological invasions has been advocated as a way to obtain definitive synthesis, generalisation and prediction (Mack *et al.*, 2000). Future and ongoing eradication programmes also provide excellent research opportunities for ecologists to study the roles of species in communities, the impact of non-indigenous species, and the behaviour and population dynamics of exotics for which eradication is being considered (J. H. Myers *et al.*, 2000; Sakai *et al.*, 2001). In general, we advocate that eradication projects be viewed as ecological experiments in which the modification of trophic webs (by addition and subtraction of species) can expose community processes. This contributes to our understanding of biological invasion, but also more generally to many areas of population biology and general ecology (Shea & Chesson, 2002). As such, each alien control programme should be designed and implemented as a scientific experiment, with hypotheses, rigorous protocols, and control of expected (and unexpected) results (Pascal & Chapuis, 2000). This will allow researchers in conservation biology to establish a wider and stronger theoretical framework, which is much needed in this discipline (Caughley, 1994), and will also allow conservation managers to benefit from ensuing generalisations.

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Craig Tribal Association
P.O. Box 828
Craig, Alaska 99921
Tel: 907-826-3996
Fax: 907-826-3997

CTA Resolution 2019-26

TITLE: AUTHORIZATION OF TRIBAL SUPPORT FOR THE APPLICATION OF THE 2001 ROADLESS AREA CONSERVATION RULE ('ROADLESS RULE') ON THE TONGASS NATIONAL FOREST and THE AUTHORIZATION OF STRONG SUPPORT FOR LASTING PROTECTION AND IMPLEMENTATION OF THE ROADLESS RULE.

WHEREAS, the Craig Tribal Association a federally recognized Indian tribe, organized pursuant to the authority of Section 16 of the Act of Congress of June 18, 1934, (48 Stat.984), as amended by the acts of Congress, June 15th, 1935, (49 Stat.378 and May 1, 1936, (49 Stat. 1250); and,

WHEREAS, the aforementioned tribes represent the majority of tribal residents on Prince of Wales Island and are empowered under their respective Constitutions and By-Laws to preserve, protect, and promote the history, culture, and traditions of the Tribal citizens they represent, through the exercise of their powers and authority to make rules and regulations; and,

WHEREAS, protection of the inherent right to harvest and use traditional and customary foods require careful stewardship and protection of the environmental and natural resources; and,

WHEREAS, the care of the lands now classified as "inventoried roadless areas" in the Tongass National Forest stretches for millennia; and

WHEREAS, these lands not only provide our Alaska Native people with food, they essentially define who are and where we come from; and

WHEREAS, inventoried roadless areas contain many sites sacred to Tribal citizens and other Alaska Natives who use these roadless areas for spiritual and religious practices and other customary uses and activities; and,

WHEREAS, inventoried roadless areas conserve natural biodiversity and serve as a bulwark against the spread of invasive species; and,

WHEREAS, inventoried roadless areas protect healthy watersheds to ensure a clean drinking water supply, and help ensure the continued protection of fish and wildlife habitat that is essential to the pursuit of our social, nutritional, and ecological values and our traditional way of life; and,

WHEREAS, the Tongass National Forest represents one of the highest carbon stores in the world and the conservation of the remaining intact inventoried roadless areas on Prince of

Wales Island and the wider Tongass National Forest is essential for maintaining our local ability to adapt and mitigate to climate change; and,

WHEREAS, the Roadless Rule prevents the disturbance of soils and wasteful construction of damaging roads in inventoried roadless areas that negatively impact fish habitat and winter deer habitat and therefore the subsistence harvest opportunities available to our Tribal members; and,

WHEREAS, according to the State of Alaska's own economic experts, Tongass timber is uncompetitive because of permanent and fundamental changes in global timber markets, high labor costs, distance from markets, and less expensive substitutes; and,

WHEREAS, continuing to spend millions of taxpayer dollars to build roads in inventoried Tongass roadless areas makes no economic sense, particularly given the agency's enormous road maintenance backlog; and

WHEREAS, there are already more than 5,000 miles of roads across the entire Tongass National Forest, which fragment valuable wildlife habitat, threaten salmon by blocking fish passage, and serve as the primary source of sediment into fish streams; and,

WHEREAS, over 1,000 miles of roads are found on Prince of Wales Island alone, and the Island has suffered the brunt of the industrial-scale logging industry since its inception in the 1950s; and,

WHEREAS, Prince of Wales Island and surrounding islands that comprise the traditional territories of the aforementioned tribes continue to be the location of the majority of large-scale timber sales on the Tongass National Forest, including the recent authorization of some of the largest old growth timber sales that Southeast Alaska has experienced in years, known as the Prince of Wales Landscape Level Assessment and the Big Thorne Project; and,

WHEREAS, these large-scale timber sales on our traditional territories have been pursued with the Roadless Rule in effect on the Tongass National Forest, to the detriment of subsistence opportunities for community members and wildlife habitat and populations; and,

WHEREAS, the presence of the extensive logging road system on Prince of Wales Island has been continuously used as justification for the development of the logging industry on this island, as opposed to elsewhere in the Tongass National Forest, which compounds the negative effects that these logging practices have on subsistence resources; and,

WHEREAS, tribal members on Prince of Wales have repeatedly testified in public forums including the federally-convened Southeast Subsistence Regional Advisory Council, which was established by provisions in the Alaska National Interest Lands Conservation Act, regarding the concerns related to the effects of industrial-scale logging on wildlife habitat and subsistence harvest opportunities for tribal members;

WHEREAS, at hearings held across Southeast Alaska in 2000 on the proposed Roadless Rule, nearly 60 percent of the Southeast Alaskans supported including the Tongass National Forest in the 2001 Roadless Area Conservation Rule, and support for the rule has only grown to upwards of 80 percent to date; and,

WHEREAS, the State of Alaska petitioned the United States Department of Agriculture (USDA) to exempt Tongass inventoried roadless areas from the Roadless Rule on January 19, 2018;

WHEREAS, for all the above reasons, the State's petition severely mischaracterizes the actual extent and impacts of the Roadless Rule on Southeast Alaska; and,

WHEREAS, the State's petition ignores the fact that the USDA narrowly tailored the Roadless Rule to limit only two activities in roadless areas, road construction and commercial logging; and,

WHEREAS, the State of Alaska's petition ignores the Rule's established exceptions, including Federal Aid Highway projects connecting communities, access to mining claims, and logging incidental to otherwise permitted activities, including utility corridors and hydropower projects; and,

WHEREAS, on August 2, 2018 the Forest Service signed a Memorandum of Understanding (MOU) with the State of Alaska to develop an Alaska-specific Roadless Rule that addresses management of inventoried roadless areas on the Tongass National Forest; and,

WHEREAS, three days before the MOU was signed and six months after the State filed its petition, the Forest Service informed the tribe by letter dated 30 July 2018 that the U.S. Secretary of Agriculture Perdue and State of Alaska Governor Walker had reached an agreement to prepare an Alaska-specific rule that would replace the 2001 Roadless Rule and "provide for activities needed to further the State of Alaska's economic development while conserving roadless areas for future generations;" and,

WHEREAS, the July 30th letters served as in invitation from Acting Regional Forester, David E. Schmid to "Tribal Leader[s]" inviting "government-to-government consultation in advance of a formal public comment period and an opportunity for your Tribe[s] to participate as a cooperative agency," because of our "expertise on subsistence and potential impacts to specific communities within Alaska" during preparation of an environmental impact statement (EIS) to evaluate the proposed Alaska-specific Roadless Rule for the Tongass National Forest; and,

WHEREAS, the U.S. Department of Agriculture, which includes the Forest Service, published a Notice of Intent (NOI) published on August 30, 2018, announcing an intent to prepare an environmental impact statement (EIS) and initiate a public rulemaking process; and,

WHEREAS, representatives from *three of the federally recognized Tribes* participated in the Alaska Roadless Rule Public Scoping Meeting conducted by the Forest Service in Craig on September 18th, 2019 and voiced significant concern regarding tribal consultation efforts and the overall timeline of the Rulemaking process; and,

WHEREAS, the aforementioned Tribes are unified in our criticism of the imposition of an arbitrarily rushed timeline that appears to be solely for political purposes and is not respectful of the inherent controversy and substantial implications that this Rulemaking process will have on the integrity of our lands and the ability of our Tribal governments to fulfill obligations and protect the wellbeing of our Tribal citizens; and,

WHERAS, the aforementioned Tribes are skeptical of the motivations given by the State of Alaska regarding the necessity of pursuing a Tongass-specific Roadless Rule,

WHEREAS, the aforementioned tribes are concerned that the effects of the Roadless Rulemaking process will be concentrated on Prince of Wales Island instead of spreading across the region, due to the extensive logging infrastructure that is already in place as well as the vast road system on the Island; and,

WHEREAS, it is critical that Prince of Wales Island continue its recovery from industrial scale logging, which has contributed to the significantly reduced deer harvest experienced by Tribal citizens over the past three years; and,

WHEREAS, Prince of Wales Island and the surrounding smaller islands are important to the tribal residents of Prince of Wales Island, especially the coastal areas and areas associated with higher wildlife values, which include the Roadless areas around _____

WHEREAS, any rule that weakens or eliminates Roadless Rule protections in what became Inventoried Roadless Areas of the Tongass National Forest will substantially affect the existing 2016 Tongass Land Management Plan Amendment and the decades of work, millions of dollars, and extensive amount of time and effort that went into crafting this Plan and associated documents; and,

WHEREAS, the 2016 Plan Amendment excluded all roadless areas from the available timber base, and the agency's failure to consult and coordinate forest planning with the aforementioned tribes will impair the ability of the Tribes to fulfill their responsibility to "protect the general welfare and security" of their tribal citizens; and,

WHEREAS, the USDA and Forest Service entered into a MOU with the State of Alaska and issued the NOI for preparation of an EIS for an Alaska-specific Roadless Rule without prior consultation or collaboration with the aforementioned tribes that claim jurisdiction over their respective traditional territories; and,

WHEREAS, as provided in FSM 1563.03, the policy of the Forest Service that "Indian tribes will be provided the opportunity for timely and meaningful government to government consultation regarding actions which may have tribal implications" and such consultation requires the agency to "[c]ollaboratively involve Indian tribes, as early as possible, in the development of regulatory and management policies, resource and land management plans, study plans and actions, and Federal undertakings that may have tribal implications;" and,

WHEREAS, the aforementioned Tribes have specialized knowledge and should have a meaningful role in providing input and information on any significant actions that could lead to any restrictions or reductions in subsistence harvest opportunities or wildlife populations that are used for subsistence harvest; and,

WHEREAS, the Alaska Roadless Rulemaking process is a significant action that will make more acres on Prince of Wales Island and the surrounding islands available for industrial-scale timber harvest and associated roadbuilding, thus further negatively impacting winter habitat for Sitka black-tail deer populations and other important subsistence resources that the Tribal citizens depend on for their traditional way of life, practice of culture, and food security; and,

WHEREAS, the Forest Service and the USDA have failed to fulfill their obligation for timely and meaningful government-to-government consultation time and time again, which is evident through the denial of requests on the behalf of the Tribal cooperating agencies for an extended

commenting period and the lack of technical support and expertise provided to assist the Tribes with the analysis of the effects of the proposed rulemaking on the traditional lands of the aforementioned Tribes; and,

WHEREAS, the State of Alaska has resisted and continues to resist all efforts to develop and work in a government-to-government relationship with the aforementioned Tribes and did not consult with the Tribes before filing its' petition with the USDA; and,

WHEREAS, Governor Walker issued Administrative Order 299, establishing the Alaska Roadless Rule Citizen Advisory Committee (CAC) on September 5, 2018; and

WHEREAS, Alaska Governor Walker appointed 13 members to the CAC, including the Alaska State Forester, to advise the State of Alaska on the future management of inventoried roadless area in the Tongass National Forest; and

WHEREAS, the Governor appointed a single Alaska Native to represent the plurality of all tribal perspectives for the seventeen federally-recognized Tribes located in Southeast Alaska on the CAC, a gigantic and unrealistic burden for one person; and

WHEREAS, seventeen federally-recognized Tribes located in Southeast Alaska face diverse challenges and have divergent priorities for their tribal citizens and tribal communities, making the representation of all these perspectives a gigantic and unrealistic burden for one person; and,

WHEREAS, the only other Alaska Native on the CAC represents Sealaska Corporation, which is an Alaska Native for-profit Corporation established under the Alaska Native Claims Settlement Act, and is not a federally-recognized Tribe.

THEREFORE, BE IT RESOLVED, the Forest Service must initiate consultation, coordination and accommodation of tribal interest in any changes to the 2016 TLMP Amendment connected with this rulemaking process; and

THEREFORE, BE IT FURTHER RESOLVED, given the serious and long-lasting Tribal implications from any reduction in current Roadless Rule protections, the aforementioned Tribes strongly object to the Forest Service's failure to engage in meaningful consultation with these federally-recognized sovereign governments before deciding to grant the State of Alaska's petition and begin a review under the National Environmental Policy Act and the public rulemaking process; and

THEREFORE, BE IT FURTHER RESOLVED, the only changes to the Roadless Rule that CTA can support is an update to the inventory used to define inventoried roadless areas subject to the Rule on the Tongass to include approximately 350,000 acres excluded from the 1995 inventory used for developing the 2001 Roadless Rule, because the agency assumed approved logging development would occur - when it did not; and

NOW, THEREFORE BE IT NOW RESOLVED, the aforementioned Tribes strongly support lasting protection for all inventoried roadless areas within the Tongass National Forest as provided in the 2001 Roadless Rule; and,

NOW, THEREFORE, BE IT RESOLVED that the federally-recognized sovereign tribal governments representing tribal citizens in the communities of Craig hereby urge the Secretary of Agriculture to select a 'no-action alternative' as the preferred alternative in the Alaska Roadless Rulemaking process.

CERTIFICATION

The undersigned do hereby certify that the Tribal Council of the Craig Tribal Association is composed of seven (7) members, of which an established quorum was present at a duly convened meeting, and this resolution was adopted by the following votes and has not been rescinded, amended, or modified in any way this 21st day of May, 2019.

[5] Yes [Ø] No [Ø] Abstained [Ø] Absent [1] Excused

Attest:

Clinton E. Cook, Sr., Tribal President

June E. Durgan, Tribal Secretary

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Assessing the value of roadless areas in a conservation reserve strategy: biodiversity and landscape connectivity in the northern Rockies

MICHELE R. CRIST,* BO WILMER† and GREGORY H. APLET‡

**Ecology and Economics Research Department, The Wilderness Society, 350 N 9th Street, Suite 302, Boise, ID 83702, USA; †Center for Landscape Analysis, The Wilderness Society, 1424 Fourth Ave, Ste 816, Seattle, WA 98101, USA; and ‡Ecology and Economics Research Department, The Wilderness Society, 7475 Dakin Street, Ste 410, Denver, CO 80221, USA*

Summary

1. Roadless areas on United States Department of Agriculture (USDA) Forest Service lands hold significant potential for the conservation of native biodiversity and ecosystem processes, primarily because of their size and location. We examined the potential increase in land-cover types, elevation representation and landscape connectivity that inventoried roadless areas would provide in a northern Rockies (USA) conservation reserve strategy, if these roadless areas received full protection.
2. For the northern Rocky Mountain states of Montana, Wyoming and Idaho, USA, we obtained GIS data on land-cover types and a digital elevation model. We calculated the percentage of land-cover types and elevation ranges of current protected areas (wilderness, national parks and national wildlife refuges) and compared these with the percentages calculated for roadless and protected areas combined. Using five landscape metrics and corresponding statistics, we quantified how roadless areas, when assessed with current protected areas, affect three elements of landscape connectivity: area, isolation and aggregation.
3. Roadless areas, when added to existing federal-protected areas in the northern Rockies, increase the representation of virtually all land-cover types, some by more than 100%, and increase the protection of relatively undisturbed lower elevation lands, which are exceedingly rare in the northern Rockies. In fact, roadless areas protect more rare and declining land-cover types, such as aspen, whitebark pine, sagebrush and grassland communities, than existing protected areas.
4. *Synthesis and applications*. Landscape metric results for the three elements of landscape connectivity (area, isolation and aggregation) demonstrate how roadless areas adjacent to protected areas increase connectivity by creating larger and more cohesive protected area 'patches.' Roadless areas enhance overall landscape connectivity by reducing isolation among protected areas and creating a more dispersed conservation reserve network, important for maintaining wide-ranging species movements. We advocate that the USDA Forest Service should retain the Roadless Area Conservation Rule and manage roadless areas as an integral part of the conservation reserve network for the northern Rockies.

Key-words: conservation, elevation zones, land-cover types, landscape metrics, national forests, reserve design

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Correspondence: M. R. Crist, Ecology and Economics Research Department, The Wilderness Society, 350 N 9th St, Suite 302, Boise, ID 83702, USA (fax +208 343 8184; e-mail michele_crist@tws.org; mcrist@gmail.com).

Introduction

A growing body of scientific evidence indicates that the current USA system of federal protected areas (designated wilderness areas, national parks and national wildlife refuges) may be too small and disconnected to protect against the decline and loss of native species diversity or to accommodate large natural ecosystem processes (Wright, Dixon & Thompson 1933; MacArthur & Wilson 1967; White 1987; Wilcove 1989; Baker 1992; Turner *et al*. 1993; Noss & Cooperrider 1994; Reice 1994; Newmark 1995; Sinclair *et al*. 1995; Soule & Terborgh 1999). Expanding road networks, human settlements, resource extraction and other encroachments on the landscape have increased the fragmentation and loss of natural areas. Such disturbances have isolated many protected areas, causing them to function as terrestrial 'islands' surrounded by a matrix of lower quality altered lands (Harris 1984; Pickett & White 1985; Wiens, Crawford & Gosz 1985; Turner 1989; Saunders, Hobbs & Margules 1991). The long-term persistence of many species within protected areas is dependent on the degree of human activities and land-use practices on lands adjacent to and near protected areas. There is a need to identify relatively undisturbed lands located outside protected areas that may increase the potential of protected areas in maintaining native biodiversity and certain ecological processes, and to include these lands within the conservation reserve system before they are lost or altered.

Inventoried roadless areas, large tracts of relatively undisturbed land on USA Forest Service lands, are often left out of landscape assessments for identifying functional conservation reserves. Only two studies (DeVelice & Martin 2001; Strittholt & DellaSala 2001) have analysed the contribution that roadless areas make to the current protected areas reserve network. However, more than one-third of inventoried roadless areas on national forests are adjacent to protected areas (DeVelice & Martin 2001). They hold the potential to increase the size and connectivity of designated wilderness areas, national parks and national wildlife refuges, thus increasing the ability of protected areas to maintain natural landscape dynamics and native species population viability over the long term. Smaller, isolated roadless areas are also important because they may contain rare species, capture more habitat variation, including underrepresented habitat types, and may function as 'stepping stones' that connect current protected areas across a landscape (Shafer 1995; Strittholt & DellaSala 2001).

There is a precedent for the protection of national forest roadless areas. The USA Congress has designated as wilderness more than half, 6 million ha, of roadless areas that the Forest Service inventoried in national forests in the 1970s. In 1998, the Forest Service began to devise regulations aimed at protection of roadless area characteristics in national forests. In May 2000, the agency released its proposed rule, familiarly known as the Roadless Rule, and draft environmental impact statement. Eight months later, the Forest Service adopted the rule. In July 2004, the Forest Service proposed to repeal the Roadless Rule and replace it with a state petition and rule-making process, which would offer less protection by presumably opening national roadless areas to all forest service activities and requiring state governors to 'opt in' Roadless Rule protections affirmatively for any roadless area.

Included in the Roadless Rule environmental impact statement was an evaluation of the potential contribution that protection of roadless areas could make to the conservation of biodiversity at a national scale (USDA Forest Service 2000b). In that evaluation, DeVelice & Martin (2001) found that the inclusion of roadless areas in the network of federal protected areas would expand representation of ecoregions in protected areas, increase the acreage of reserved areas at lower elevations, and increase the number of areas large enough to provide refuge for wide-ranging species.

Strittholt & DellaSala (2001) focused on similar questions at a regional scale for the Klamath-Sikiyou area in southern Oregon and northern California, USA. They found that roadless areas protect a wide range of ecological attributes, especially at mid- to lower elevations, important in this region. They also concluded that roadless areas increase the connectivity among ecoregions.

The northern Rocky Mountain states of Montana, Wyoming and Idaho comprise a region particularly rich in roadless areas, roughly 2·6 million ha, providing a unique opportunity to create a relatively intact reserve design that captures important elements of conservation for the northern Rockies. Using two key concepts in conservation biology, biodiversity representation and landscape connectivity, we investigated the potential contributions of national forest roadless areas to the protected areas reserve network across the northern Rocky Mountain region.

DIVERSITY REPRESENTATION

An important goal in the design and establishment of conservation reserves is to represent a full range of native biodiversity (Shelford 1926; Margules, Nicholls & Pressey 1988; Church, Stoms & Davis 1996; Possingham, Ball & Andelman 2000). Even though this goal has been articulated for some time, most protected areas are demarcated around areas with high scenic and recreational attributes (Davis *et al*. 1996). As a result, existing protected areas in the northern Rockies are, for the most part, concentrated at higher elevations, where other important elements of biodiversity are most likely to be poorly represented (Scott *et al*. 2001).

Representation of a full range of biodiversity in reserves requires an understanding of all species and ecosystem processes operating within a given landscape. However, many researchers have used ecological communities and elevation ranges as coarse-scale

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surrogates for native biodiversity in the design of conservation reserves (Scott *et al*. 1993; Host *et al*. 1996). This concept is based on the idea that if a full range of ecological communities and elevation ranges is protected, it is more likely that many ecological communities, wide-ranging species and ecosystem processes will be maintained in the reserves. In the northern Rockies, ecological communities are often associated with elevation gradients (Hansen & Rotella 1999). Hence, roadless areas situated at middle and lower elevations may make valuable contributions in protecting many elements of biodiversity that are currently not well represented in protected areas (DeVelice & Martin 2001).

LANDSCAPE CONNECTIVITY

Connectivity refers to the degree to which the structure of a landscape helps or hinders the movement of wildlife species or natural processes such as fire (Wiens, Crawford & Gosz 1985; Turner *et al*. 1993; Noss & Cooperrider 1994; Bascompte & Solé 1996; With 1999). A 'well-connected' area can sustain important elements of ecosystem integrity, namely the ability of species to move and natural processes to function, and is more likely to maintain its overall integrity compared with a highly fragmented area.

Roads are highlighted in the scientific literature as major causes of landscape fragmentation, and function as barriers to organism movements, resulting in a reduction of overall landscape connectivity for many native species. The effects of roads are broad and include mortality from collisions, modification of animal behaviour, disruption of the physical environment, alteration of chemical environments, spread of exotic and invasive species, habitat loss, increase in edge effects, interference with wildlife life-history functions and degradation of aquatic habitats through alteration of stream banks and increased sediment loads (Franklin & Forman 1987; Andrews 1990; Noss & Cooperrider 1994; Reice 1994; Reed, Johnson-Barnard & Baker 1996; Trombulak & Frissell 2000; McGarigal *et al*. 2001). Thus, the addition of roadless areas to existing protected areas reserve is likely to maintain or increase landscape connectivity, as well as increase the integrity of protected areas.

With the advent of landscape metrics, it is now possible to quantify connectivity for landscapes, land-cover types, species' habitats, species' movements and ecosystem processes across a given region (O'Neill *et al*. 1988; McGarigal & Marks 1995; Gustafson 1998; With 1999). Many different metrics that quantify spatial characteristics of patches or entire landscape mosaics have been described (Turner & Gardner 1991; McGarigal & Marks 1995; Ritters *et al*. 1995; Hargis, Bisonette & David 1998; Dale 2000; Jaeger 2000; McGarigal & Holmes 2002). We chose metrics that measure three elements of landscape connectivity: area, isolation and aggregation.

Area

It is known that larger areas (patches) generally contain more species, more individuals, more species with large home ranges and/or sensitive to human activity, and more intact ecosystem processes than smaller areas (Robbins, Dawson & Dowell 1989; Turner *et al*. 1993; Newmark 1995; Shafer 1995). Higher numbers of patches will usually contribute to greater resilience of populations and may also increase the utility of patches that act as 'stepping stones' or connectors across a landscape (Buechner 1989; Lamberson *et al*. 1992).

Isolation

The distance between patches plays an important role in many ecological processes. Studies have shown that patch isolation is the reason that fragmented habitats often contain fewer bird and mammal species than contiguous habitats (Murphy & Noon 1992; Reed, Johnson-Barnard & Baker 1996; Beauvais 2000; Hansen & Rotella 2000). As habitat is lost or fragmented, residual habitat patches become smaller and more isolated from each other, species movement is disrupted, and individual species and local populations become isolated (Shinneman & Baker 2000).

Aggregation

The spatial arrangement of patches may help to explain how certain species are found in patches located close together and are not found in patches that are more isolated, or vice versa (Ritters *et al*. 1995; He, DeZonia, & Mladenoff 2000). This concept generally follows the ideas developed in island biogeography theory (MacArthur & Wilson 1967) and metapopulation theory (Levins 1969, 1970).

For some species or natural processes, the isolation or aggregation of patches across the landscape may be more important, for others, area may be the key element. Together, these three elements offer a comprehensive assessment of the importance of roadless areas to the maintenance of overall landscape connectivity and ecosystem integrity of current protected areas in the northern Rockies.

In this study, we aimed to assess the extent to which roadless areas increase biodiversity representation and landscape connectivity when they are included in the protected areas reserve network for the northern Rockies.

Methods

STUDY AREA

Of the 84 million ha of land that stretch across Montana, Wyoming and Idaho in the USA, roadless areas cover 2·6 million ha and existing federal protected areas (wilderness areas, national parks, special management areas and national wildlife refuges) protect almost 8·7 million ha. Within this region, three large, relatively

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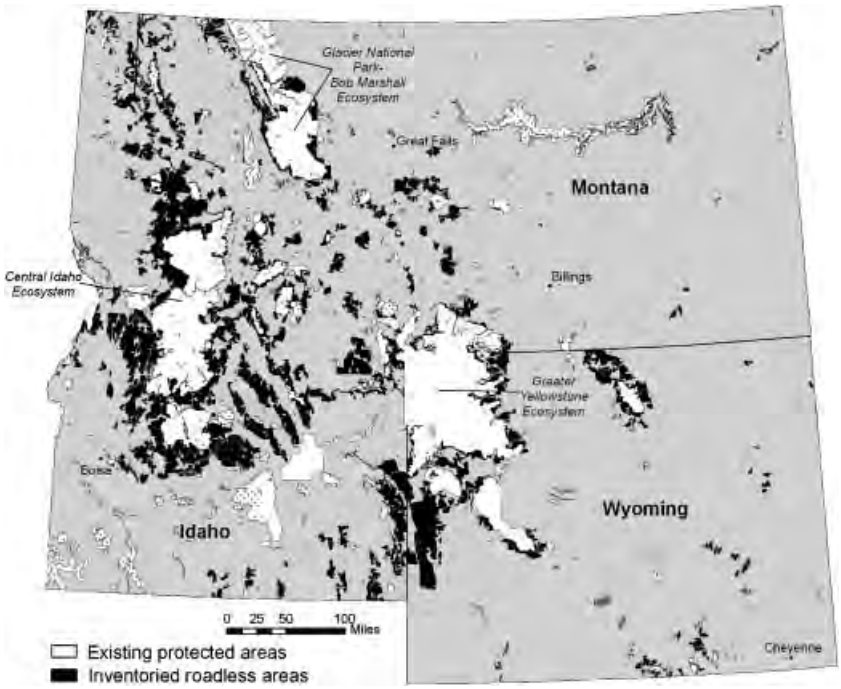

Fig. 1. Roadless areas and protected areas across the states of Idaho, Montana and Wyoming, USA.

undisturbed, mountain ecosystems are delineated around national parks and/or wilderness complexes. These are the Greater Yellowstone Ecosystem, Glacier National Park–Bob Marshall Ecosystem, and the Central Idaho Ecosystem (Fig. 1).

The topography of the northern Rocky Mountain states spans steep physical gradients in elevation, slope, aspect, temperature and precipitation that give rise to diverse vegetation types. Elevations range from 150 m to 4200 m. Average precipitation ranges from 28 cm to 51 cm (Franklin 1983). The northern Rockies comprise a variety of non-forested and coniferous forest types. Low-lying valleys are characterized by grasslands, sagebrush (*Artemisia* spp.) and desert shrublands, interspersed with juniper (*Juniperus* spp.) and riparian woodlands. Ponderosa pine *Pinus ponderosa* dominates lower elevation montane forests, while xeric coniferous forests of mainly Douglas fir *Psuedotsuga mensiezia*, ponderosa pine, grand fir *Abies grandis*, lodgepole pine *Pinus contorta* and aspen *Populus tremuloides* occur at mid-elevations. Mesic forests in the north and west largely contain western larch *Larix occidentalis*, grand fir, western red cedar *Thuja plicata* and mountain hemlock *Tsuga mertensiana*. Higher elevations are composed of Engelmann spruce *Picea engelmannii*, subalpine fir *Abies lasiocarpa*, alpine larch *Larix lyalli* and white-bark pine *Pinus albicaulis* intermixed with subalpine meadows. Herb lands, rock, alder *Alnus sinuata* shrub-fields and snowfields/ice occur at the highest elevations.

DATA COLLECTION

We used a land management status GIS coverage and classification system developed by the USA Geological Survey's Biological Resources Division in its nation-wide GAP Analysis Programme (Scott, Tear & Davis 1996) to delineate 'protected areas'. This programme devised a ranking scheme to represent various levels of protection, ranging from the least protected lands (category 4, e.g. private lands) to those with the highest level of protection (category 1, e.g. wilderness areas) for all public lands in the GIS spatial database. For this study, we assumed that categories 1 and 2 represent adequate protection as their primary management objective is conservation (Scott, Tear & Davis 1996), and selected these categories as our protected areas on all forest service lands located in the three states.

We used the federal inventoried roadless areas GIS database (USDA Forest Service 2000a). This includes areas that are greater than 2000 ha in size, where road building is prohibited under current National Forest Plan decisions and where road building is presently allowed. We recognize that our decision leaves out smaller roadless areas that were not considered during the inventory of federal roadless areas and that these areas serve important conservation goals (Strittholt & DellaSala 2001). For this study, the term 'roadless areas' refers to inventoried roadless areas.

We used three independently derived land cover maps for Montana, Wyoming and Idaho from the GAP Analysis Programme (Scott, Tear & Davis 1996). The Montana and Idaho GAP products were produced based on classification techniques by Redmond *et al.* (1998) for raw Landsat Thematic Mapper (TM) satellite imagery. Spatial resolution of the grid was 90 m for Montana and 30 m for Idaho. The Wyoming GAP Analysis Programme digitized land cover data in a vector format from Landsat TM satellite imagery at a

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scale of 1 : 100 000 (Gap Analysis Wyoming 1996). We converted Wyoming's vector map into a grid format and resampled the three data sets to 90-m resolution. Then we merged the three land cover maps into a single image and a common land cover classification scheme (Appendix 1).

Similar to most GIS databases, errors are associated with the land management status, inventoried roadless areas and land-cover grids. These grids represent a composite of data from many sources and include variations in mapping procedures and possible misclassifications that could potentially cause inconsistencies that are difficult to detect. However, we believe, based on professional judgement, that the error rate is not large enough to affect conclusions drawn from this large regional-scale analysis.

To investigate the representation of roadless areas at various elevation classes, we downloaded a digital elevation model from the 30-m National Elevation Dataset produced by the USA Geological Survey's EROS Data Center (Sioux Falls, SD). We reclassified the elevation range into 21 equal-interval classes ranging in 200-m increments from approximately 150 m to 4200 m.

DATA ANALYSIS

All data analyses were conducted in ARC/INFO and ArcView GIS software from Environmental Systems Research Institute (Redlands, CA).

Land cover representation

Using ARC/INFO, we combined the protected areas database with the land cover map. To calculate the percentage representation of each land-cover type, we divided the protected portion of each land-cover type by the total area of each land-cover type across the study area. Next, we appended the national forest inventoried roadless areas to the existing protected areas and repeated the same calculation described above to measure the additional representation of each land-cover type because of the inclusion of roadless areas. In addition, we calculated the percentage increase between each land cover percentage representation for protected areas alone and protected areas and roadless areas combined. This measure quantified the 'relative' ecological contribution from roadless areas for each land-cover type. We then ranked these land-cover types according to the level of representation within the existing protected areas.

Elevation representation

Using ARC/INFO, we combined the protected areas database with the 30-m digital elevation model. Similar to the procedure for land-cover types described above, we added the roadless areas to the existing protected areas, intersected this image with the elevation data, and calculated the change in representation for each elevation class provided by protection of roadless areas.

To examine the potential increase of landscape connectivity caused by roadless areas, we used ARC/INFO and FRAGSTATS (McGarigal & Marks 1995; McGarigal & Holmes 2002), a computer program developed to quantify heterogeneity of the landscape. We identified five landscape metrics available in FRAGSTATS to assess our three elements of landscape connectivity (McGarigal & Holmes 2002). To assess area, we used the metrics percentage land (PLAND), number of patches (NP) and patch size (AREA). We included the metrics NP and AREA to help explain the context of an increase in PLAND. For example, an increase in PLAND and AREA and a decrease in NP would indicate that the added roadless patches were located next to existing conservation patches, resulting in an increase in the size of patches and a decrease in the number of patches across the landscape. Conversely, a decrease in AREA and an increase in NP would indicate that the added patches were generally smaller and did not combine with existing patches.

To assess isolation we used nearest neighbour distance (ENN). A decrease or increase in ENN would indicate that patches are either located closer together or farther apart, respectively, across the landscape.

To assess aggregation, we used contagion (CONTAG). An increase in CONTAG would indicate that patches are, to a certain extent, aggregated together across the landscape.

Using FRAGSTATS, we selected and ran our five landscape metrics on the two grids described above (current protected areas only, and roadless areas and current protected areas combined). Each grid was a binary map where all grid cells that comprised the 'protected' and 'roadless' patches were classified as 1 and all other 'non-protected' grid cells were masked out as background (–99). For each landscape metric, we computed the mean, area-weighted mean and coefficient of variation where applicable. We then compared the differences in metrics between the two grids. In addition, differences in the mean, area-weighted mean and coefficient of variation helped to explain how the range of values for each metric were distributed when existing protected areas were compared with the conservation system including roadless areas.

Results

LAND COVER REPRESENTATION

In existing protected areas, burned forest and snowfields/ice had the highest land cover representation, 88% and 86%, respectively. Representation of other land-cover types, such as alpine meadows, whitebark pine, exposed rock/soil, subalpine meadows, wetlands, mixed subalpine forest and lodgepole pine, ranged from 31% to 71%.

The inclusion of roadless areas increased the representation of all land-cover types except for one, sand dunes (Table 1). Relative percentage increases ranged

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Table 1. Additional representation and percentage increase in representation of each land-cover type across the northern Rockies when national forest roadless areas are added to existing protected areas

Land-cover type	Existing level of representation (%)	Potential level of representation including roadless areas (%)	Percentage increase including roadless areas
Burned forest	88·12	93·09	5·65
Snowfields/ice	86·12	97·48	13·19
Alpine meadow	71·51	94·18	31·70
Mixed whitebark pine	59·62	84·94	42·46
Exposed rock/soil	44·67	59·92	34·12
Subalpine meadow	40·49	68·85	70·05
Wetlands	37·34	38·68	3·61
Mixed subalpine forest	32·20	68·63	113·11
Lodgepole pine	31·35	59·42	89·54
Mixed barren lands	21·66	22·61	4·37
Sand dunes	18·44	18·44	0·00
Mixed conifer	16·97	37·24	119·44
Mesic upland shrub	10·74	26·14	143·44
Shrub-dominated riparian	7·98	12·77	59·91
Forest-dominated riparian	7·18	12·14	69·11
Sagebrush	6·33	9·91	56·55
Juniper	5·87	6·80	15·95
Xeric upland shrub	5·85	7·97	36·33
Vegetated sand dunes	5·69	6·03	5·89
Western red cedar	5·57	22·00	295·08
Mud flats	5·33	7·39	38·79
Ponderosa pine	4·94	9·88	99·97
Aspen	4·48	25·99	479·80
Shrub–grassland associations	4·25	5·89	38·46
Western hemlock	3·36	23·62	602·54
Grasslands	2·49	3·64	46·31
Grass-dominated riparian	2·15	3·07	43·01
Salt-desert shrub flats	1·58	1·71	8·63
Bur oak woodland	0·00	2·40	NA

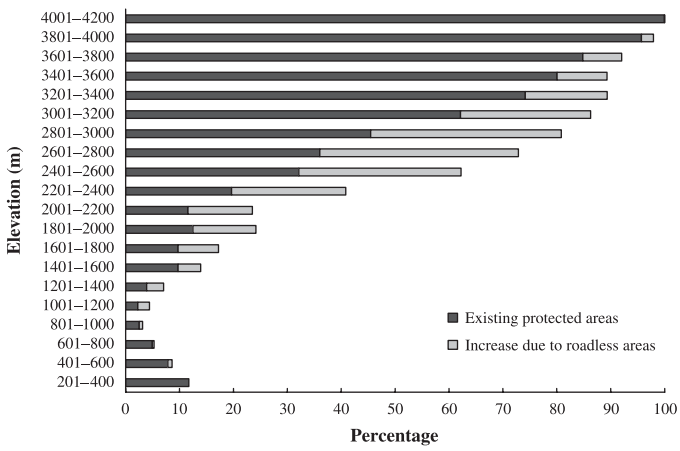

Fig. 2. Additional representation of elevation ranges resulting from the inclusion of roadless areas with protected areas for the northern Rockies. The *x*-axis represents elevation in 200-m increments and the *y*-axis shows absolute increase in percentage representation when roadless areas are added to protected areas. Black bars represent protected areas and grey bars represent roadless areas.

from 5% to 600%. Fifteen land-cover types increased by more than 40%, among them important ecological communities, western hemlock, aspen, ponderosa pine, western red cedar and sagebrush, each of which has less than 10% representation in current protected areas. Moreover, the addition of roadless areas represented one land-cover type, bur oak *Quercus macrocarpa* woodland, not present in protected areas.

ELEVATION REPRESENTATION

Our elevation analyses showed that elevations in the range of 2200–4200 m were well represented in protected areas (Fig. 2). The addition of roadless areas resulted in a large increase in representation of lands at elevations ranging from 1000 m to approximately 3400 m. For elevation ranges below 1000 m and above 3400 m, the

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Table 2. Landscape metrics comparing the spatial pattern of protected areas alone with a scenario that includes protected areas and national forest roadless areas combined for the northern Rockies. + and – indicate an increase or decrease, respectively, in the metric value caused by the addition of roadless areas

Landscape Metrics	Protected areas	Protected and roadless areas	+/ –
Area			
Class area (ha)	8 814 900	15 673 600	+
Percentage land	9	16	+
Number of patches	770	722	–
Patch size (mean, ha)	11 447·92	21 708·59	+
Patch size (area-weighted mean)	1 105 055·78	2 505 909·11	+
Patch size (coefficient of variation)	977·39	1 069·74	+
Isolation			
Nearest neighbour (m)	7 013·72	5 353·11	–
Nearest neighbour (area-weighted mean)	3 153·73	2 518·75	–
Nearest neighbour (coefficient of variation)	122·47	134·16	+
Aggregation			
Contagion index	72·56	58·64	–

contribution of roadless areas was small. However, the proportion of area represented at lower elevations increased when we included roadless areas with protected areas.

CONNECTIVITY

Results from the landscape metrics showed that the addition of roadless areas increased regional connectivity for all three connectivity elements (Table 2). Area metrics demonstrated that the addition of roadless areas almost doubled the amount of area protected, rising from 9% to 16%, and the mean patch size in protected areas changed from 11448 ha to 21709 ha. The number of patches decreased from 770 to 722. Area-weighted mean patch size increases and the patch size coefficient of variation increased from 977 to 1070. Isolation metrics showed a decrease in the mean and area-weighted mean nearest-neighbour metrics when roadless areas were added. The mean distance between nearest protected patches decreased from 7014 m to 5353 m. The decrease in the area-weighted mean was less than the overall mean when patches of all sizes were considered. The coefficient of variation also increased for this metric. The aggregation metric (contagion) decreased from 72·56 to 58·64 when roadless areas were included, signifying more dispersion of patches across the landscape.

Discussion

BIODIVERSITY REPRESENTATION

A review of the literature suggests that a given vegetation community is adequately represented when 12–25% of it is included in a conservation area (World Commission on Environment & Development 1987; Noss & Cooperrider 1994), although it is not certain that these thresholds are truly adequate to protect vegetation communities. Based on this range, we define land-cover types above 25% as adequately protected, land-cover types within the range of 12–25% as minimally protected, and those below 12% as underrepresented, similar to DeVelice & Martin (2001).

Our results show that roadless areas make a substantial contribution in maintaining regional biodiversity. One of our most important findings is that roadless areas would protect a wider range of land-cover types and elevation ranges than protected areas alone, especially those characteristic of mid- to low elevations that are underrepresented in protected areas. These lands are among the last remnants of biologically productive lands that have not been significantly altered through human settlements, resource extraction and road construction (Scott *et al.* 2001; Strittholt & DellaSala 2001). We also found that protected areas adequately represent land-cover types that are characteristic of higher elevations. This finding supports the generally accepted notion that wilderness areas and national parks mainly protect higher elevation ecological communities (Davis *et al.* 1996; Possingham, Ball & Andelman 2000). Contrary to DeVelice & Martin (2001), whose study found that roadless areas mainly occurred at mid- to lower elevations, but similar to Strittholt & DellaSala (2001), we found that roadless areas considerably increase the protection of higher elevations and corresponding cover types as well. The different results are probably because of the scale at which the studies were implemented. DeVelice & Martin's (2001) study included all roadless areas across the nation, incorporating a wide range of elevations from sea level to the highest peaks. Our study, and that of Strittholt & DellaSala (2001), focused on smaller regions at higher elevations.

Across the northern Rockies region (Montana, Wyoming and Idaho), protected areas adequately represent nine land-cover types, whereas five biologically important land-cover types, western hemlock, aspen, ponderosa pine, western red cedar and mesic upland shrub, are underrepresented in protected areas. However, the addition of roadless areas increases representation of two cover types (western hemlock and western red

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cedar) to the minimally protected threshold and two cover types (aspen and mesic upland shrub) to the adequately represented threshold (greater than 25%). Ponderosa pine, even though it increases by nearly 100%, remains underrepresented. Overall, the magnitude of the increased representation, from 100% to 600%, indicates that roadless areas can make substantial contributions to the protection of land-cover types that are not well represented in protected areas.

Increased representation of certain rare ecological communities is particularly important in a northern Rockies conservation strategy. Aspen, for example, is thought to be declining in the northern Rockies (Gallent *et al*. 1998). When roadless areas are added to protected areas, aspen moves up two full categories: from underrepresented to adequately represented, a 480% increase in representation for this forest type, on which many avian species depend upon (Hansen & Rotella 2000). Representation of whitebark pine changes from 60% to 85% when roadless areas are added. Whitebark pine is declining throughout North America due to blister rust *Cronartium ribicola*, an introduced disease, and is a 'keystone species' important for many higher elevation species (Keane, Morgan & Menakis 1994).

Elevation representation results demonstrate that protected areas are mainly located at higher elevations. We also found that roadless areas are generally concentrated at mid- to high elevations and represent a wider range of elevations, especially low- to mid elevations, than protected areas. However, our results show that protected areas encompass more lower elevation lands than roadless areas. This situation is somewhat deceiving. Representation of lower elevations in protected areas is largely a result of two well-placed low-elevation conservation areas: Hell's Canyon National Recreation Area and Missouri Breaks National Monument. In fact, low-elevation lands below 1000 m are not well represented in either protected areas or roadless areas. As a majority of lower elevation lands in the northern Rockies have been converted to other uses, it is of utmost importance to increase representation of lower elevation sites in protected areas (Strittholt & DellaSala 2001). Protection of these lower elevation roadless areas would contribute greatly to the conservation of lower elevation species and ecological communities that are poorly represented in protected areas.

LANDSCAPE CONNECTIVITY

Our analyses of three elements of connectivity show that roadless areas increase connectivity across the northern Rockies, and increase both the area and size of protected area patches. In addition, the number of protected area patches decreases with the addition of roadless areas because they combine with protected areas to form one larger patch. Larger patches will protect more species and more individuals, species with large home ranges, species sensitive to human activity, and more intact ecosystem processes than smaller areas (Askins, Philbrick & Sugeno 1987; Robbins, Dawson & Dowell 1989; Turner *et al*. 1993; Newmark 1995; Shafer 1995). Roadless areas also reduce the distance between protected areas and create a more evenly dispersed reserve system, critical for maintaining many species' movements and a large distribution of local populations (MacArthur & Wilson 1967; Murphy & Noon 1992; Reed, Johnson-Barnard & Baker 1996; Ritters *et al*. 1996; Beauvais 2000; Hansen & Rotella 2000; He, DeZonia, & Mladenoff 2000; Shinneman & Baker 2000).

Our results show an increase in the coefficient of variation for patch size and isolation metrics, which may be an important consideration in delineating conservation reserve systems capable of maintaining movements of various species and ecological processes (Wiens & Milne 1989; Wilcove & Murphy 1991; Noss 1992; Noss *et al*. 1996; O'Neill *et al*. 1996). Smaller patches may supplement larger reserves by protecting rare species that occur only in certain areas (Franklin & Forman 1987; Hansen *et al*. 1991; Shafer 1995). The dispersion of roadless areas may also contribute to greater resilience or survival of island populations by allowing a greater chance for species exchange, essentially maintaining a metapopulation or source–sink population structure (Wiens, Crawford & Gosz 1985; Pullium 1988; Gilpin & Hanski 1991; Murphy & Noon 1992). Many studies are investigating how species move through landscapes and their use of stepping-stone habitats, especially in fragmented landscapes (Freemark *et al*. 1993; With 1999; Beauvais 2000; Hansen & Rotella 2000; Holloway, Griffiths & Richardson 2003; Johnson, Seip & Boyce 2004). Being relatively undisturbed and well-distributed among protected areas, roadless areas are top candidates for the delineation of high-quality 'habitat connections' across the northern Rockies, particularly those that target rare or declining species. The loss or alteration of roadless areas may further reduce the movement of species among interdependent island populations located in protected areas and roadless areas, resulting in greater isolation.

Moreover, the addition of roadless areas increases the effective size of the three largest wilderness and national park complexes in the northern Rockies: the Greater Yellowstone Ecosystem, the Glacier National Park–Bob Marshall Ecosystem and the Central Idaho Ecosystem, where management challenges include maintaining large-scale ecological processes such as species' movements and natural fire across jurisdictional boundaries (Pickett & White 1985; Turner *et al*. 1993). Roadless areas not immediately adjacent to these complexes are dispersed in the surrounding landscape, which helps to decrease the degree of isolation between the complexes and possibly allows for species movement among these ecosystems.

MANAGEMENT IMPLICATIONS

Using research to guide reserve design and develop land protection policies is the strongest approach in

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conservation. The importance of intact, functioning natural ecosystems to the maintenance of native biodiversity and ecological processes is unquestioned (Wright, Dixon & Thompson 1933; MacArthur & Wilson 1967; Usher 1987; White 1987; Shafer 1995; Noss, O'Connell & Murphy 1997). The negative impacts of roads in natural areas are well known (Andrews 1990; Foreman & Wolke 1992; Reed, Johnson-Barnard & Baker 1996; Spellerberg 1998; Trombulak & Frissell 2000; McGarigal *et al*. 2001). Our landscape assessment demonstrates how roadless areas, the remaining relatively undisturbed forested lands in the northern Rockies, are essential for maintaining biodiversity and landscape connectivity in a conservation reserve strategy for this area. This has direct bearing on management decisions regarding the protection of roadless areas in this region. Our results, along with the findings of DeVelice & Martin (2001) and Strittholt & DellaSala (2001), highlight the important role of roadless areas in USA conservation efforts and contribute to the larger policy dialogue surrounding roadless areas.

The methods used in this study can help land managers determine appropriate guidelines to identify and assess roadless areas that are critical in maintaining regional biodiversity, ecosystem processes, landscape connectivity and overall intact ecosystem integrity. Land managers should avoid activities such as road building, logging, spread of exotic species, off-road vehicle use and exurban development in roadless areas that would result in their degradation or loss. If roadless areas are not protected from these activities as a matter of priority, it is possible that their potential contribution to conservation effort in the future will be diminished and existing protected areas surrounded by or in close proximity to roadless areas will be negatively affected as well. We recommend that roadless areas receive full protection and are managed responsibly, so that they can function as an important part of the current conservation reserve system in the USA.

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Supplementary material

The following material is available from http://www.blackwellpublishing.com/products/journals/suppmat/JPE/JPE996/JPE996sm.htm.

Appendix 1. Land-cover types across the northern Rocky Mountain region reclassified from USA Geological Survey's Biological GAP Analysis Programme (Scott, Tear & Davis 1996).

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Southeast Alaska economics
A resource-abundant region competing in a global marketplace

Lisa K. Crone*

USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 2770 Lane, Juneau, AK 99801, USA

Abstract

Questions related to economics figured prominently in the priority information needs identified in the 1997 Tongass Land Management Plan. Follow-on studies in economics were designed to improve understanding of aspects of the competitiveness of the Alaska forest sector, links between Alaska timber markets and other markets as evident in prices, and the relationship between resource allocation decisions and economic conditions in communities and the region. Analysis of the role of recreation and tourism in the regional economy was added to the topics addressed, based on early results of work to describe economic dynamics.

Comparisons are undertaken to evaluate the economic changes taking place in southeast Alaska, to analyze the sources of these changes, and to determine if and why they differ from the changes taking place at larger scales and those occurring in a similar rural and resource-abundant region. Divergent views regarding the current role of the Tongass in the regional economy are summarized and assessed by using contemporary evidence.

A variety of factors contribute to comparative and competitive disadvantages for the forest products sector in southeast Alaska. Alaska product and log markets are effectively integrated with other markets supplied by producers in British Columbia and the Pacific Northwest.

Empirical evidence suggests the need to re-examine assumptions regarding the relation between changes in "basic" sector activities and employment (such as timber harvesting and wood products manufacturing) and "nonbasic" (or support sector) employment in the rural communities of southeast Alaska. Many of the changes occurring in the economy of rural southeast Alaska are driven by changes in the international markets in which Alaskan products compete, and are largely independent of Tongass forest management.

Unearned income and tourism have replaced resource-extractive industries as the principal sources of income growth in the region. The contribution of the Tongass National Forest to the regional economy has become more complex and difficult to quantify. Forest management policies that enhance the comparative advantages the region enjoys in providing both tourism opportunities and quality of life attributes will aid communities in maintaining and expanding their economic opportunities.
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* Present address: USDA Forest Service, Recreation Solutions, P.O. Box 1165, Troy, MT 59935, USA.
Tel.: +1 406 295 7484; fax: +1 406 295 7440.
E-mail address: lcrone@fs.fed.us.

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1. Context and introduction

Natural resource management and policy issues, especially debates over management of national forests, are complex and contentious. Disputes arise over both the objectives of management and the facts on which management decisions are based. During the latest plan revision for the Tongass National Forest, public comments and political debates revealed disagreements over goals of forest management in Alaska, effective and efficient methods to achieve consensus on goals, and the definition of equitable distribution of gains or losses resulting from allocation decisions. In southeast Alaska, as elsewhere, progress in addressing enduring and contentious disputes affecting forest management depends on timely and reliable information. The factual basis for planning, and the contribution of science to planning, includes knowledge of all systems affected by management: biophysical, social, and economic. Economic science provides some of the technical information needed to inform and evaluate management decisions. Economic research and synthesis can also provide critically important context for managers to use in understanding the public choice decisions associated with all national forest planning.

The economies and communities of southeast Alaska are affected by forest management in ways that are complex and not completely understood. The region has experienced significant economic change that has fundamentally altered the importance of Tongass resource production, and social change that has altered preferences for the mix of outputs. Changes in the mix of management objectives for the Tongass clearly are among the factors contributing to these changes. However, many of the factors leading to social and economic change are outside agency influence, resulting in uncertainty and disagreement over how to use social and economic information to evaluate management decisions. The challenge is to provide a science-based description of consequences of management decisions in the broader context of social and economic dynamics. This situation is not unique to Alaska, although data and analyses that take Alaska's circumstances into account are limited. The subsistence use of resources is an important but confounding consideration when analyzing indicators of social and economic well-being at the community scale in Alaska.[1]

Southeast Alaska presents a complicated combination of scope, diversity, and scale for resource-based social science. Unlike most biophysical sciences, social science theories and methods generally apply at large spatial scales. Significant diversity among communities and mixed signals from community-scale indicators of social and economic conditions—both within and across communities—complicate efforts to describe the region-wide effects of land management decisions. Information on regional trends, although easier to obtain, often masks differences in conditions and trends at the scale of communities. In other words, information is scale-specific and is often unavailable at the desired scale.

Owing largely to the abundance of natural resources and the dominance of primary product outputs, economic activity in southeast Alaska has always been highly dependent on conditions in the national and global economy.[2] In recent years, increased globalization along with changes in key Alaskan product markets has increased this dependence. The changes occurring at larger scales have trickled down to affect communities in the region, but the effects have not been uniformly distributed across or within communities.

Questions related to economics figured prominently in the priority information needs identified in the 1997 Tongass Land Management Plan (TLMP). Two of these needs focused on economic aspects of the forest sector: (1) Determine prices and costs in Alaska timber production and product supply. (2) Determine Alaska timber prices and market arbitrage in the Pacific Northwest. Two studies were conducted in response to these information needs and to verify assumptions used by Brooks and Haynes (1997) in their market projections for Alaska timber and products by examining both the competitiveness of the Alaska forest sector and

[1] Studies focusing on subsistence use are discussed in the social synthesis portion of this journal issue (Kruger).

[2] Tsournos, P., Haynes, R.W., in preparation. An assessment of growth and development paths for resource abundant regions. Gen. Tech. Rep. PNW-GTR. USDA Forest Service, Pacific Northwest Research Station, Portland, OR, provide a review of the economic literature on development paths in resource-abundant regions, including theories and evidence suggesting that some resource-abundant regions may never develop diverse economies.

the links between Alaska timber markets and other markets. A third priority information need focused on the effects of forest management on rural communities: (3) Determine the relationship between socioeconomic conditions in rural communities and resource allocations on the Tongass National Forest. Three additional studies were designed to improve understanding of the relationship between resource allocation decisions and economic conditions in communities and to examine growth and structural changes in the regional economy.

The purpose of this synthesis is to present our findings and their management and policy implications for the Tongass National Forest. I begin with a review of the forest sector economics studies (Robertson and Brooks, 2001; Stevens and Brooks, 2003) including objectives, methods, findings, and implications. An overview of current conditions in the Alaska forest sector and the markets in which Alaskan timber competes is presented both to analyze the validity of study findings and provide an outlook for the future. In Section 2, I discuss the objectives, methods, findings, and implications of a study analyzing employment multiplier impacts at the community scale in southeast Alaska (Robertson, 2003). As discussed above, economic, social, and biophysical systems are dynamic at all scales. Thus, any attempt to explain economic changes occurring at the regional or community scale must also consider influences from changes occurring at higher scales. In Section 3, I review the objectives, methods, findings, and implications of two studies (Robertson, in press; Crone, in press) that examined changes in the southeast Alaska economy within this larger context. In recent years, there have been increasingly divergent views on the importance of Tongass timber harvests to the regional economy. In Section 5, I introduce these alternative views and use historical and contemporary evidence to assess the past, present, and future roles of the Tongass National Forest in the regional economy.

2. Forest sector studies

Forest sector models have been used to prepare perspectives on future developments in the forest sector as a framework for policy formation and decision-making in the United States, Europe, and elsewhere. One modeling approach known as scenario planning differs from traditional modeling (and particularly forecasting) approaches by incorporating uncertainty directly in the analysis. In this approach, the modeler does not attempt to predict the future, but instead postulates a set of plausible futures (scenarios), each dependent on the assumptions underlying that future. Most forestry applications of this method employ a classical sensitivity analysis approach in which a limited number of key exogenous and endogenous elements are varied and key projection results are examined for differences. The exogenous elements are also commonly referred to as assumptions.[3]

In 1997, Pacific Northwest Research Station economists used this modeling approach to produce revised and updated projections of derived demand for Alaska national forest timber. Earlier projections (also based on the scenario modeling approach) had been done in 1990 and in 1994. New projections were necessary because significant changes in the structure of the Alaska forest sector, in markets for Alaska products, and in conditions faced by Alaska's competitors (the Pacific Northwest and British Columbia), resulted in violations of implicit assumptions used in the earlier analyses.[4] The closure of the two pulp mills in southeast Alaska altered both the structure and scale of its forest products industry by eliminating the primary market for low-grade and utility logs as well as local markets for residues from lumber production. Traditionally, Japan has been the primary market for Alaska logs and lumber. The Asian recession beginning in 1989 resulted in both decreased demand and increased price sensitivity in the Japanese market. Other changes in the Japanese market included increased acceptance of engineered wood products and increased preference for kiln-dried products. Additionally, new suppliers from Europe entered the market in response to the initial high prices resulting from public harvest restrictions in the Pacific Northwest. In the 1994 projections, it was assumed that Alaska might enjoy a competitive advantage over the Pacific Northwest because of these harvest restrictions. However, increased production from Canada, the

[3] This brief description of forest sector models and the scenario approach was drawn from Haynes (1993).

[4] Some of the assumptions (such as the number of operating pulp mills) were implicit in Brooks and Haynes (1994) but were examined through scenario analysis.

Table 1
Projections of the average annual-derived demand for Alaska national forest timber

Period	Alternative scenario projections	Previous projections	
		1990	1994
1000 m^3			
1993–1997	820–910	1830	1359
1998–2002	435–565	1825	1427
2003–2007	589–824	1798	1504
2008–2010	598–1010	1816	1517

Source: Brooks and Haynes (1990, 1994, 1997).

entrance of new competitors, and Alaska's high costs eliminated this advantage.

Owing to the high degree of uncertainty surrounding important components of the demand projections, Brooks and Haynes developed high, medium, and low scenario projections by varying the levels of components. The scenario components were Alaska's share of North American shipments to Japan, North America's share of Japanese softwood lumber imports, the share of Alaska's shipments going to export markets, and the efficiency of Alaska lumber production. Table 1 shows the range of the projections based on the three scenarios as well as the previous base projections (Brooks and Haynes, 1990, 1994, 1997). Because of the changes discussed above, demand was estimated to be substantially lower under all three scenarios than in the previous projections. The authors also conducted a sensitivity analysis of the model to changes in the individual scenario components. This analysis revealed the model results to be most sensitive to changes in Alaska's share of North American shipments of softwood lumber to Japan. This finding and the desire to verify implicit assumptions used in developing the model provided the impetus for the two forest sector TLMP follow-on studies to which I now turn.

The objective of the first study by Robertson and Brooks, Assessment of the Competitive Position of the Forest Products Sector in Southeast Alaska, 1985–1994, was to provide quantitative measures of production costs and product revenues for softwood lumber produced in southeast Alaska and to compare them with those for the Pacific Northwest and coastal British Columbia. The goals of this study were to provide information relevant to the debate over Tongass timber harvest levels and to assess the economic efficiency implications of efforts to increase value-added processing in the region.

The theory of comparative advantage holds that a region will specialize in products that use more of the inputs that are abundant (and thus cheaper) in their region, and minimize the use of inputs that are relatively scarce (and thus more expensive). Competitive advantage can be defined as the ability to supply a good at a lower cost than other producers can. Two methods can be used to measure the competitiveness of southeast Alaska producers. The first is to look at unit input costs, defined as the cost of factors used to produce one unit of output. The second method is to examine stumpage prices, defined as the price buyers pay for timber. The following areas were analyzed to determine competitiveness:

(1) the forest resource in which timber stocks are combined with labor, machinery, and other inputs to produce raw logs;
(2) the processing sector in which logs are combined with other factors to produce products such as lumber and chips (including mill residues);
(3) end markets where purchasers compare the price and physical characteristics of southeast Alaska products to those from other regions (Robertson and Brooks, 2001, p. 5).

An examination of 1995 harvests in southeast Alaska revealed that 24% of the harvest was Sitka spruce (*Picea sitchensis*) and 54% of the harvest was western hemlock (*Tsuga heterophylla*). Most of the Sitka spruce harvested was saw log quality or better (70%), whereas much of the hemlock harvest was classified as low-grade saw logs or utility logs (51.6%). Southeast Alaska is the major supplier of Sitka spruce (70% of North American production in 1995), but only a minor supplier of hemlock (17% of total western hemlock production for export to Pacific Rim markets).

Over the 1987–1994 period, it took an average of 35% more labor inputs (logging hours) to produce a 1000 m^3 in southeast Alaska compared to the Pacific Northwest. The authors attribute this difference primarily to the remoteness and rugged terrain of southeast Alaska. This combined with higher wages (23% higher on average) in southeast Alaska resulted in 65% higher unit labor costs in southeast Alaska. The authors posit that other unit factor costs (capital, energy,

and materials) will also be higher in Alaska for similar reasons.

In the sawmill sector, owing to both higher per unit labor inputs and higher hourly wages, unit labor costs were 49% higher in southeast Alaska. Reasons suggested for this difference are that southeast Alaska mills are less mechanized, use older and low-quality equipment, and were operating at less than full capacity throughout the study period. Again, the authors believe that the other unit factor costs will be higher as well. Southeast Alaska was also less efficient in converting logs to lumber. This may be caused by the substitution of relatively less expensive inputs (logs, especially low-grade hemlock) for relatively more expensive inputs (labor and capital). Additionally, the demand for mill residues by pulp mills reduced the incentive to maximize conversion rates.

Using prices to examine southeast Alaska's relative market position in the Japanese log and lumber markets (Alaska's primary export end-product markets) revealed that Alaska producers enjoy a competitive advantage and market power as the major supplier of Sitka spruce. This derives from the scarcity value and uniqueness to Alaska of this species. It is a different story for hemlock, for which Alaska is a minority producer and price taker. External factors such as the substitution of radiata pine from the southern hemisphere and whitewood from northern Europe (where suppliers also have more backhaul opportunities than Alaska; Wiita, 2001), the shift from green lumber to kiln-dried lumber and engineered wood products, reductions in Japanese housing starts, and changes in Japanese building codes will play a larger role in determining Alaska's competitiveness in this market than the characteristics of the log inputs. Because of the high cost of value-added inputs (labor and capital), Alaska is at a competitive disadvantage in this market and will suffer first from price decreases.

Analyzing total lumber production costs (stumpage, harvest, and manufacturing costs) across all species in 1994, the authors found southeast Alaska to be the highest cost producer followed by British Columbia and the Pacific Northwest. Southeast Alaska had both the highest manufacturing costs and the lowest stumpage prices. They argued that the low stumpage prices are indicative of the competitive disadvantage Alaska has in harvesting and manufacturing compared to the Pacific Northwest. Southeast Alaska processors are essentially charging some of their higher production costs against the scarcity value of timber, so that the national forest bears some of the cost of regional inefficiencies.

Analysis of stumpage values for individual species in southeast Alaska revealed high values for Alaska yellow-cedar (*Chamaecyparis nootkatensis*), lower values for Sitka spruce, and the lowest values for hemlock. Because hemlock accounts for the largest proportion of volume, this implies that the profitability of southeast Alaska processors will be very sensitive to changes in the price of hemlock.

Major findings from the study are:

1. Southeast Alaska is at a comparative disadvantage in the forest products sector because of both higher factor costs and lower productivity of factor inputs.
2. Southeast Alaska is able to profitably produce certain types of wood products because of the scarcity value of some of the better species and log grades in the region.
3. Southeast Alaska's advantage lies in its ability to supply these scarce raw materials (logs), not in providing value-added inputs (labor and capital) with them.[5]

Implications of these findings are that value-added processing in the region could be successful for the higher-valued species and grades of Alaska yellow-cedar and Sitka spruce, especially niche industries using primarily Sitka spruce to produce commodities that are not in direct competition with major producers in other regions. For the lower-valued species and grades that make up the majority of southeast Alaska's timber inventory, the authors believe policies that promote cost minimization and economies of scale in processing are preferred. Alternatively, reducing or eliminating processing of this material, by allowing partial harvesting or relaxing processing requirements for certain lower grades, might improve southeast Alaska's competitive position.

Although the findings from the study overwhelmingly indicate that southeast Alaska is a high-cost pro-

[5] Irland Group and Market Decisions (1990) found that, in general, high-wage areas are not suited to value-added wood processing. Galston and Baehler (1995) wrote, "In the case of forestry and wood products, a general rule of thumb states that the more value added to the lumber, the more likely the enterprises are to concentrate near urban areas." (p. 101).

ducer, some analysts (McDowell Group, 2000b) have argued that Alaska's timber resources are unique, and therefore insulated from competition, potential substitutes, and market declines. If this argument is true, Alaska's higher costs will not matter. This view contrasts with that of Brooks and Haynes (1997)[6] who expected future Alaskan timber production to be very sensitive to market cycles and competition from other regions. These opposing views provided the catalyst for the second timber economics study.

The objective of "Alaska Softwood Market Price Arbitrage" (Stevens and Brooks, 2003) was to determine whether or not Alaskan lumber and logs are integrated with those of similar products from the Pacific Northwest and British Columbia in the Japanese market. Using co-integration statistical testing methods suggested by Johansen (1995, 1997), the authors sought to determine whether the law of one price applies for these products. This principle holds that in long-run equilibrium, and in the absence of artificial (i.e., tariffs) and natural (i.e., transportation costs) barriers to trade, identical goods should sell for the same price in international markets. Ideally, the prices of similar products from all three regions could be included in a simultaneous multivariate co-integration test. The lack of a single data source for comparable products precluded this approach. Instead three independent data sets were used in pair-wise co-integration tests of long-run movements of prices in Alaska and the Pacific Northwest, and Alaska and British Columbia.

Annual data from the Japan Lumber Journal were used to compare Alaska Sitka spruce with British Columbia Sitka spruce log prices from 1979 to 1998. Quarterly data from this source were used in a comparison of Sitka spruce lumber prices from Alaska and British Columbia from 1974 to 1998. U.S. Department of Commerce quarterly data were used to compare Alaska and Pacific Northwest hemlock log prices (1989–1997) and hemlock lumber prices (1990–1997). Finally, monthly data from Japan Lumber Reports were used to compare Alaska and British Columbia hemlock prices from 1986 to 1997.

Results from their analysis supported the conclusion that Alaskan western hemlock and Sitka spruce logs share an integrated market (Japan) with logs from British Columbia and the Pacific Northwest. The test for the Sitka spruce lumber market also supported integration. Original results for Alaskan and Pacific Northwest hemlock lumber were less robust. However, when a pre- and post-1994 dummy variable was included to account for the closure of Alaska's pulp mills and structural changes in the Japanese lumber markets, the results supported an integrated market.

These results strongly indicate that in Japan most of Alaska's forest product exports do compete with products from the other regions. This implies that Alaskan production and exports will be sensitive to international market conditions and that the cost structure of the Alaskan forest products industry compared to competing regions is very important. An overview of the current situation in the Japanese market and Alaska's forest sector provides a litmus test for this and the previous study's conclusions.

Between 1995 and 2001, Japanese softwood lumber imports decreased by 24% from 10.73 to 8.13 million m^3. The North American share of these imports decreased from 72 to 53%. Alaska's share of North American softwood lumber exports to Japan decreased from 1.4 to 0.2% over this period. As discussed above, factors contributing to these declines were the emergence of low-cost suppliers of kiln-dried material (Europeans), the decrease in Japanese housing starts associated with the 1997 recession, and both regulatory and structural changes in the Japanese housing sector. The fact that Alaska's share of North American exports to Japan decreased supports Stevens and Brooks (2003) findings of an integrated market. Between 1995 and 2000, as competition in the shrinking Japanese market increased, the estimated share of Alaska sawnwood shipments going to export markets decreased from 82 to 28%. Much of the sawnwood shipped to the lower 48 states consists of shop hemlock going to the Puget Sound region where it is remanufactured into millwork, such as door and window casings and shipped to other locations. (Southeast Alaska Regional Timber Industry Task Force, 1997; Wescott, 2002; Fay, 2003). Some southeast Alaska mill owners report that they have found excellent markets and are receiving good prices for these products.[7] Spruce sawnwood sold in the domestic market has primarily

[6] See also Irland Group (1991, 1992), which concur with the Brooks and Haynes outlook.

[7] Personal e-mail communication. Brink, S., 2002. sbrink@fs.fed.us (22 December).

Table 2
National forest timber program in Alaska, 1997–2001

Year	Offered	Sold	Harvested
1000 m^3			
1997	915	960	525
1998	847	109	548
1999	525	281	661
2000	390	775	666
2001	308	226	217
5-year average	598	471	525

Source: USDA Forest Service (2002a).

been dimension lumber. Some locally produced dry surfaced lumber, paneling, and other construction products are being used in Alaska (Southeast Alaska Regional Timber Industry Task Force, 1997).

Table 2 shows the volume of timber offered, sold, and harvested from Alaska national forests from 1997 to 2001. The 5-year average annual harvest level of 525 000 m^3 falls near the Brooks and Haynes (1997) medium scenario-derived demand projection for the 1998–2002 period of 512 000 m^3. Based on their sensitivity analysis, this result implies that the decreases in demand predicted by decreases in both the North American share of Japanese softwood lumber imports and in Alaska's share of North American shipments to Japan have been at least partially offset by increases in the share of Alaska's shipments going to domestic markets. Table 3 provides further measures of trends in important aspects of the Alaska forest products sector and clearly reveals a substantially reduced forest products industry in southeast Alaska.

Stevens and Brooks (2003) provided evidence that most of Alaska's forest products must compete with products from other regions in the Japanese market, and Robertson and Brooks (2001) provided evidence that southeast Alaska is a high-cost producer in this market. As predicted by Brooks and Haynes (1997) and borne out by the statistics presented in Tables 2 and 3, these factors have combined to make Alaska timber production very sensitive to market cycles and competition from other regions and have contributed to the decline in this sector from 1995 to 2000. In Section 4, I will provide further analysis of the underlying causes of this decline within a historical and global context. Irrespective of the causes, at the time of the TLMP revision the effects of reduced harvests on the communities and economies of southeast Alaska were the subject of much debate (MacMullan and Niemi, 1994; McDowell Group, 1995). There was little disagreement that jobs in the wood products sector would decrease with reduced harvests; rather the indirect or secondary impacts were at issue. This debate spurred the third TLMP economics follow-on study.

3. Testing the export base model in southeast Alaska

Community-level impacts have long been a concern for U.S. Forest Service policymakers. Originally, much of the concern focused on the direct impacts of timber flows on logging and sawmilling activity, but more recently the focus has shifted to the overall impact of reductions in timber activity on the local economy at large (Robertson, 1999). The most common models used to estimate economic impacts to areas from changes in forest activities are export base or input–output (I–O) models. These models assume a linear relationship between export-oriented economic activity, from basic sectors such as logging and wood products manufacturing, and local-support economic activity, from non-basic sectors such as grocery stores and banks. The idea here is that the basic sectors bring new money into

Table 3
Alaska forest product statistics, 1996–2000

Variable	1995	2000	Change (%)
Volume of softwood lumber exports (1000 m^3)	119	8.5	−93
Percentage of wood product exports going to Japan (% value basis)	78	59	−19
Average value of softwood log exports (dollars per m^3)	156	94	−40
Average value of softwood lumber exports (dollars per m^3)	329	382	16
Average value of chip exports (dollars per metric ton)	92	45	−51
Southeast Alaska wood products employment	1911	994	−48

Source: USDA Forest Service (2002b).

the local economy, and the circulation of this money creates economic activity in the nonbasic sectors. An economic multiplier is an estimate of the amount of additional (or reduced) economic activity that will be generated from an increase (or decrease) in basic sector activity. Income or employment is generally used as an indirect measure of economic activity because detailed surveys of exactly what gets exported are difficult and costly. The existence of a strong linear relationship between basic activity, such as timber harvesting and processing, and nonbasic activity is widely upheld in both public and professional debates (Robertson, 1999).

Owing to a lack of data at smaller scales, a common approach to estimating local impacts has been to adjust broader-scale models for local conditions, and rarely have these estimates been compared to actual economic performance over time. In recent years, economic data at smaller spatial scales is becoming more common, allowing for empirical tests of multiplier effects at the community scale.

In the TLMP revision, an employment multiplier of 1.72 was used to estimate the secondary impacts of changes in the timber employment in the region. This implies, for example, if 10 jobs were lost in the timber sector, an additional 7.2 jobs would be lost in nonbasic sectors. The objective of the third economics TLMP follow-on study, "A test of the economic base hypothesis in the small forest communities of Southeast Alaska" (Robertson, 2003) was to determine whether this multiplier effect existed at the smaller scale of communities.

Past studies that have attempted to measure impact multipliers by using time series data include Sasaki (1963), Weiss and Gooding (1968), Moody and Puffer (1970), Moriarty (1976), and Henry and Nyankori (1981). All these studies suffered from small sample size problems. Fortunately, Robertson was able to obtain quarterly employment data from the Alaska Department of Labor covering the period from 1981 to 1996 for 15 forest communities in southeast Alaska, thus eliminating the small sample size problems and allowing for a more robust and detailed estimation of linear impact multipliers.

Robertson followed the general approach of the earlier studies by specifying a linear regression model with nonbasic employment as the dependent variable and basic employment as the independent variable. Basic employment included the majority of manufacturing employment and state and federal government employment, and nonbasic employment included all other employment. Although tourism represents basic activity, it is not reported as a separate category in employment statistics and thus cannot be separated from retail trade and services for local consumption. This problem was handled by the inclusion of a trend variable in the regression model, which served as a proxy for tourism employment and unearned income.[8] To smooth out seasonal variations in the data, the author used quarterly differencing in which data from the same quarter of the previous year is subtracted from the current quarter.

Because impacts in the nonbasic sector caused by changes in the basic sector are not instantaneous, the independent variable (basic sector employment) was lagged. Because of the high degree of multicollinearity between lagged periods, Robertson used an Almon lag structure specification. An intercept term, a fourth-order moving average error correction term, and a first-order autoregressive correction term, were also included in the model specification.

Results from this analysis are shown in Table 4. The multipliers shown are partial multipliers and reflect the estimated change in nonbasic sector employment associated with one additional job in basic sectors. To obtain a total multiplier, add 1 to the partial multipliers listed in the table. Negative estimates imply that when jobs increase in a basic sector, jobs in the nonbasic sectors are estimated to decrease. Examining the table reveals that in 4 of the 15 communities there was a statistically significant (at 90% level or higher) positive linear relationship between basic and nonbasic employment. However, in four of the communities there was a negative statistically significant linear relationship, and in seven of the communities there was no statistically significant linear relationship. The average partial multiplier across all communities was essentially zero. Based on this evidence, the author could not reject the null hypothesis of no positive linear relationship between basic and nonbasic employment in southeast Alaska communities. In none of the communities did the total multiplier reach the 1.72 multiplier used in the TLMP revision to estimate impacts.

[8] Both unearned income, which will be defined and discussed in detail in Section 4, and tourism have shown a steady increase over much of the study period (1981–1996).

Table 4
Estimated employment multipliers

Community	Estimated multiplier	t-Statistic
Angoon	0.29	3.55**
Gutavus	−.79	−1.79**
Haines	0.04	0.42
Hollis	−0.50	−3.12**
Hoonah	−0.22	−1.61*
Hydaburg	0.07	1.57*
Juneau	−0.08	−0.07
Kake	−0.56	−2.87**
Ketchikan	0.1	0.39
Metlakatla	0.28	0.32
Petersburg	−0.35	−0.85
Sitka	0.20	0.68
Thorne Bay	−0.09	−0.37
Wrangell	0.20	1.80**
Yakutat	0.48	6.42**
Average	−0.06	1.72[a]

Source: Robertson (2003).
[a] Average for the absolute value of community multiplier t-statistics.
* 90% significance level.
** 95% significance level.

The author offers several suggestions for low, zero, and in some cases negative multipliers at the community level. First, there may be low "local endogenity" in many of these small communities. What this means is that most purchases by locals are made elsewhere, either because of lower prices or low or nonexistent availability in their community. This implies that most of the secondary effects from changes in basic sector employment will occur outside the local community. The high percentage of nonresident workers in the wood products sector, 31.5% in 2000 (Hadland and Landry, 2002), also contributes to low local endogeneity. Second, factor supplies, especially labor, may be constrained in semi-isolated regions and communities. In other words, if a mill expands, workers may leave lower paying jobs in a nonbasic sector and go to work at the mill. Conversely, if a mill shuts down, workers may have to work two jobs in the nonbasic sector to make the same amount of money. Third, unemployment benefits, income maintenance programs, and spending out of savings by unemployed basic-sector employees may mitigate effects on nonbasic economic activity. Finally, the combination of regional economic growth and tight labor supplies over the analysis period indicates that workers released from basic-sector employment could likely find employment in other sectors, again decreasing the multiplier effect.

Robertson (2003) stresses that regional growth is a critical element in this argument and notes, "In a shrinking economy, excess labor supply as well as excess capacity in the nonbasic sectors is likely, and the linear impact multipliers hypothesized by economic base and I–O models may be more applicable than they (linear impact multipliers) are in a growing economy." (p. 77).

Based on the results in Table 4, two conclusions were drawn. First, the study findings indicate that on average there was no significant relationship between basic and nonbasic activity in the forest communities of southeast Alaska. Second, there was significant variation in the nonbasic sector response to changes in basic sector employment across communities. These results strongly suggest that the use of a single positive economic multiplier to estimate community impacts from changes in forest activity levels may be unwise. Policy implications concern both economic impact analysis and the distribution of relief funds. The author states (Robertson, 2003, p. 79):[9]

If it cannot be demonstrated that a mill closure or similar economic shock will have significant economic ramifications in other sectors of the local economy, then ... it is meaningless to speak of "community impacts" as something other than simply the local portion of direct impacts of the closure, and the focus on community may serve to divert resources from those individuals who need help most. As a result, relief funding should be targeted to help directly impacted individuals in the sector receiving the exogenous shock rather than broader development projects aimed to bolster the economic prospects of the community at large.

The results from this study indicate that the assumption and application of a positive sector-specific impact multiplier is not a viable approach to impact estimation in small communities. Yet, in many of the economic impact assessments included in the environmental impact statements that must now accompany major for-

[9] The author offers the caveat to this implication that in the absence of reliable estimates of expected secondary impacts, the best policy may be to focus on the directly impacted sector but include monitoring programs in areas where secondary impacts are believed likely.

est planning decisions, a standard procedure is to use just such I–O-derived multipliers in conjunction with projections of basic employment or income to estimate total economic impacts. Often the impact multiplier exists as an unquestioned assumption buried in the analysis. Based on his results, Robertson suggests that more scrutiny should be given to these multipliers and their resulting impact assessments. The variation in the estimated multipliers across communities in his study further suggests that within the general debate regarding natural resource policy decisions, the presence of significant secondary impacts resulting from changes in resource-based economic activity cannot be taken as a matter of fact. Rather the effects of resource-based policy changes must be considered within the broader local, regional, and national contexts in which economic impacts occur. This is the subject of the last two TLMP economics follow-on studies.

4. Growth and change in southeast Alaska

The objectives of the follow-on studies by Robertson (in press) and Crone (in press) were to evaluate the economic changes taking place in rural southeast Alaska, to determine the sources of these changes, and to determine if and why they differ from the changes taking place at larger scales. In both studies, southeast Alaska is broken into two subregions: the borough of Juneau and rural southeast Alaska, defined as everywhere besides Juneau. This is done not only because of Juneau's relative size (42% of the population in southeast Alaska in 2000), but also because of the importance of state government operations in its economy. Because of its size, Juneau tends to dominate aggregate statistics for southeast Alaska, and its economic activity is often more reflective of trends within Alaska as a whole than is the economic activity in the less populated areas of southeast Alaska.

In his paper, "An Analysis of Changing Income Sources", Robertson (in press) provided a descriptive analysis of changes in the southeast Alaska economy in terms of primary income sources. The author used Bureau of Economic Analysis (BEA) data (U.S. Bureau of Economic Analysis, 1998) to compare total personal income in Juneau versus rural southeast Alaska from 1969 to 1996. This analysis revealed different patterns of growth, especially since 1990 when Juneau income grew while rural southeast Alaska income stagnated and declined. Based on this finding, the author separated his analysis into two periods, 1969–1989 and 1990–1996.

In his analysis of changes in total personal income, the author pointed out the important distinction between "earned income" (wages or profits to self-proprietors), which is tied to current employment in the area, and "unearned income" (dividends, interest, and rent; and transfer payments[10]), which is not. He notes that until recently the role of unearned income in stimulating economic growth has been largely overlooked. In rural southeast Alaska, he found that between 1969 and 1996, unearned income increased four-fold and increased its share of total personal income from 16% in 1969 to 33% in 1996. Within the unearned income component, the rate of growth in transfer payments has been somewhat higher than that of dividends, interest, and rent, especially since 1990, but both have been increasing in absolute levels and in their share of total personal income.

Both unearned and earned incomes in rural southeast Alaska had positive average annual rates of growth (6 and 2%, respectively) between 1969 and 1989. While unearned income continued to grow (2% per annum) between 1990 and 1996, earned income posted a 2% annual rate of decline over this period. An examination of individual industrial sectors revealed that the manufacturing sector accounted for the vast majority of the decline in earnings, declining at a rate of 7% per annum between 1990 and 1996. Manufacturing's share of earned income fell from 34% in 1969 to 17% in 1996, while service sector earnings have shown steady growth and increased in share from 8% in 1969 to 16% in 1996.

Robertson also examined changes in wages in rural southeast Alaska and concluded that decreases in wages within sectors have been far more important than shifts in employment between sectors in decreasing average wages in the region over the long term. However, he also found that over the short-term period from 1990

[10] Transfer payments include retirement and disability payments (social security, workers compensation, federal, state, and local government retirement), medical payments (Medicare), income maintenance (aid to families with dependent children, food stamps, and supplemental security income), unemployment insurance, and other benefits (veterans benefits, federal education and training, Bureau of Indian affairs, and Alaska permanent fund dividends).

to 1996, declines in timber harvesting and manufacturing were the primary cause of the decrease in average earnings in the region.

Robertson used Alaska Department of Labor employment data for rural southeast Alaska to identify employment sectors declining and advancing between 1981 and 1996. He found that employment gains were broadly distributed across a range of sectors, including local government, health services, retail, and construction. In contrast, employment declines in rural southeast Alaska were concentrated in the wood product industry and occurred primarily over the 1990–1996 period.

For context, Robertson compared changes in income in rural southeast Alaska to changes occurring in the United States at large and to changes in nonmetropolitan counties in the Pacific Northwest (PNW) region.[11] These areas also exhibited an increase in unearned income as a share of total personal income and a shift from manufacturing to services. However, when comparing industry structure in 1996, he found that rural southeast Alaska relied more on government activity while trade and services were under-represented in the region relative to the nonmetropolitan PNW counties and the nation at large.

Robertson also compared relative rates of employment growth in specific industries in rural southeast Alaska with rates of growth in nonmetropolitan Oregon counties between 1981 and 1995. He found that eating and drinking places, health services, and food stores were high-growth industries in both areas. On the other hand, business services and electronics manufacturing were high-growth sectors in nonmetropolitan Oregon counties, but displayed little or no growth in rural southeast Alaska.

Based on his analysis, Robertson concluded that the major sources of income growth in the rural southeast Alaska region were no longer in the traditional resource industries of timber, fishing, or mining. Rather, much of the growth came from increases in government, services, and retail activity. He also stated, "While increase in tourism is no doubt responsible for a substantial portion of this growth, the steady growth in unearned income, especially retirement and medical benefits, is likely a more important (though less noticed) factor." He also concluded that the increasing importance of unearned income and the shift from manufacturing to services in rural southeast Alaska and in nonmetro PNW were reflective of broad-scale changes in the nation at large. Growth in services, especially business services, has been slower in rural southeast Alaska than in the nation or in the nonmetro PNW.

In addressing the implications of his findings for forest policy, Robertson concluded that in the absence of significant increases in national forest timber sales (and the market to support them), the ability of forest policy to impact the regional economy via the timber sector will be small. Thus, the focus should shift to ways in which forest policy can affect the new drivers of economic activity in the region—tourism and unearned income. Forest policies that attract both visitors and new residents and keep existing residents from leaving will contribute to economic growth in the region. However, different user groups may prefer the provision of different amenities from the forest. The desires of residents and those of visitors are not always compatible, and the factors that make a place attractive to live are more complex and not always the same as those that make it attractive to visit. Because of these conflicts and those between tourism and recreational users of the forest and resource-extractive users, Robertson concluded that understanding and predicting the influence of forest management practices on local employment and income has become less direct and more complex.

In her paper, "Rural Manufacturing and the Wood Products Industry: Trends and Influences on Rural Areas" Crone (in press) expanded on the findings from Robertson. She began by using BEA data (U.S. Bureau of Economic Analysis, 2002) from 1969 to 2000 to examine trends in population, income, and earnings at the national (U.S.), state (Alaska), and regional (Juneau and rural southeast Alaska) levels. Because many of the concerns expressed about the economic effects of forest policy and management focus on impacts on small rural areas, Crone also compared trends in rural southeast Alaska with those in another rural and resource-abundant region. The other rural area chosen for comparison included four counties in Idaho and two counties in Montana. These counties were selected according to the following rule set: (1) county population less than 16 000 in 2000, (2) county not adjacent to a metro area (as defined by Cook and Miser,

[11] These counties were in Oregon and Washington and their populations ranged from 1642 to 135 833 in 1996.

1994), (3) at least 8% of county employment was in the wood products sector in 1995; (4) the county was not farming-dependent (as defined by Cook and Miser, 1994), and (5) National Forest System lands make up at least 50% of the county's land area. These criteria were used because they are representative of conditions in rural southeast Alaska.[12] This group of counties will be referred to as rural IdMt.

Crone found that, in general, the Alaska and Juneau economies have not followed the business cycles of the United States, and during most cycles appear to operate in a counter-cyclical manner. On the other hand, until recently, the rural areas did appear to be heavily influenced by national business cycles. Much of the susceptibility of the rural economies to national business cycles can be traced to their manufacturing sectors (Galston and Baehler, 1995). The manufacturing sector in rural southeast Alaska experienced negative average annual growth rates during all the U.S. recessions.

The apparent influence of the manufacturing sector on changes in earnings in rural southeast Alaska and rural IdMt led the author to take a closer look at the history of this sector. To better understand the influence of this sector on the rural areas under investigation here, she focused on rural manufacturing in general and the wood products sector in particular.

Crone found that the establishment of the pulp mills beginning in 1951 coincided with a general dispersal of manufacturing in the United States to rural areas during the 1950s and 1960s. Reasons cited for this national trend were the availability of low-cost labor,[13] cheap land, relatively relaxed regulations, weak or nonexistent unions, lower taxes, and often government incentives and subsidies. Although cheap labor did not draw the pulp companies to Alaska, the 50-year contracts they were granted effectively gave them a subsidy in the form of cheap timber.[14] In addition, the companies were granted exemptions from state and local taxes.

From 1970 to the mid-1980s, oscillations in both rural southeast Alaska and rural IdMt manufacturing sectors were clearly reflective of both trends in rural manufacturing in general and of cycles in the U.S. wood products sector in particular. The recession years of the early 1980s hit rural manufacturing and the U.S. wood products sector particularly hard. This was because U.S. policymakers increased both interest rates and the value of the dollar in an attempt to control inflation. The higher interest rates radiated to other countries such as Japan, and the combination of high interest rates and high value of the U.S. dollar decreased the demand for U.S. exports. This affected rural areas more because they had relatively more employment in the goods-producing industries and were more export-dependent than urban areas (Hamrick, 1997). Lumber and wood products employment in the United States reached its lowest level of the past 50 years in 1982.

In rural IdMt manufacturing employment decreased 27% between 1978 and 1982 while earnings in its wood products sector fell 56% over this same period. Manufacturing employment in rural southeast Alaska decreased by 33% between 1980 and 1984. Employment in Alaska's logging and pulp industries decreased by 27 and 42%, respectively, between 1980 and 1984, while employment in its lumber industry decreased by 73% between 1980 and 1986.

Manufacturing employment in rural IdMt never fully recovered from the recession of the early 1980s as many wood products plants either closed outright or invested in cost-cutting and efficiency measures for their plants. The end result in Idaho and Montana (as well in the Pacific Northwest) was a more mechanized wood products industry that employed fewer people. For example, in Montana, wood products output was higher in 1986 than in 1979, but 2400 fewer people were employed (Corporation for Enterprise Development, 1989). In 1990, manufacturing employment was only 86% of its 1978 peak, and by 2000 it was only 75% of that peak. In 1990, earnings in the wood products sector were only 54% of their 1978 peak level, and by 2000 they were only 36% of that peak.

In general, the wood products mills in rural southeast Alaska did not undertake the same efficiency-enhancing mechanization updates that were occurring elsewhere. Yet, rural southeast Alaska manufacturing employment and the wood products sector did recover (temporarily) from the early 1980s downturn.

[12] None of the nonmetropolitan counties in Oregon or Washington met all these criteria.

[13] Increases in agricultural productivity in the 1950s and 1960s created a large pool of surplus labor in rural areas.

[14] In order to attract the necessary large-scale investment to this remote and high-cost region, the Tongass Timber Act of 1947 authorized the construction of pulp mills on the forest and the use of 50-year timber sale contracts to supply their multiproduct wood-processing operations (Haycox, 1997).

The Irland Group (1991) attributed the rebound in the late 1980s to four factors: the declining dollar–yen exchange rate, the strong world pulp market, a stabilization of the dissolving pulp market, and the strong peak in log exports (primarily from Alaska Native Lands) that boosted logging jobs.

Historically, the integrated pulp mill operators in southeast Alaska were able to offset losses during low points in the pulp markets with the higher revenues they received when markets improved, but in the 1990s this ability disappeared for several reasons. First, the Tongass Timber Reform Act (TTRA) of 1990 removed many of the direct and indirect subsidies the pulp companies had been granted in their 50-year contracts and in the Alaska National Interest Lands Conservation Act (ANILCA) of 1981. Because the most accessible timber had already been harvested, the pulp companies' costs were already increasing and the cumulative effect of the TTRA was to push them higher.

The second factor affecting the profit margin of the pulp companies was the declining world demand for their primary product, dissolving pulp. Because the Alaska pulp mills were high-cost producers, they suffered first when the price of dissolving pulp decreased. The Alaska Pulp Company mill in Sitka shut down in 1993 and Ketchikan Pulp Company mill closed in 1997. Since the mid-1990s, more than 90 U.S. pulp, paper, and paperboard mills have shut down, and about 1 in every 12 industry-wide jobs have disappeared.

When the pulp mills closed, the marginal position of Alaska wood products manufacturers in the cyclical and global wood products industry became more evident and acute. With the pulp mills gone as ready markets for their mill residues and chips, Alaska firms must now compete with more efficient and lower cost suppliers from other regions in the global marketplace. The primary processing requirement for Forest Service land[15] and the long-term contracts with the pulp companies undoubtedly led to increased population and contributed to a more diversified economy in the region. However, this particular development strategy may have retarded the development of a competitive lumber and value-added industry. On the other hand, some would argue that because of Alaska's numerous cost disadvantages, it is doubtful that such an industry would have developed under any scenario (Irland Group, 1992). Today, many of the obstacles the industry faced in the 1950s remain and others have emerged (Southeast Alaska Regional Timber Industry Task Force, 1997; Morse, 2000).

The wood products firms that operate in southeast Alaska today have survived by finding niche markets. Crone discussed efforts underway to improve the competitiveness of Alaska wood products and expand into other niche markets. A lumber-grading project sponsored by the Alaska Manufacturer's Association and the Alaska Science and Technology Foundation has included testing to quantify the superior mechanical properties of Alaska species (Alaska Manufacturer's Association, 2002; Ketchikan Daily News, 2002). Additionally, the U.S. Forest Service Alaska Wood Utilization Research and Development Center in Sitka has undertaken projects to find technological and economic solutions for a durable forest products industry in Alaska. An update on their research findings states (Rapp, 2003, p.1):

They have found that wood recovery could be improved at nearly every sawmill in Alaska. Strong potential exists for sawmills to add dry kilns to dry lumber, which could sell at higher prices. Special grades based on the qualities of Alaska woods would increase lumber values further. Wood residues such as chips and sawdust should be viewed as resources instead of wastes. Value-added products could supply the domestic market in Alaska. Specialty products such as birch craft and cabinetry may bring higher prices if they are sold as made-in-Alaska products.

Because most observers believe it is unlikely that production and employment in the wood products industry will return to their previous levels, Crone examined other economic opportunities in the region. She found that the current situation in the other major natural resource industries, salmon fishing and mining, were similar to the wood products situation. The entrance of farmed salmon on world markets has dramatically decreased both Alaska salmon exports to Japan and sales in the U.S. market (McDowell et al., 2001). In the mineral industry, historic low metal prices and ris-

[15] In 1926, Congress prohibited the export of round logs from the Tongass. U.S. Congress Act of 12 April 1926, Exportation of Timber, P.L. 69-100. Ch. 117; 44 Stat. 242, as amended; 16 U.S. C. 616, 617. Harvests from Alaska Native lands are not subject to this log export ban.

ing costs, especially fuel, have decreased expenditures for exploration and development projects and revenue from mineral production (Swainbank et al., 2002).

Rural southeast Alaska is not unique with respect to changes in its resource extractive industries. Globalization of the markets for coal, timber, and agricultural products have decreased prices and reduced employment in local economies reliant on extractive industries (McLaughlin, 2002). Johnson (2000) writes that rural areas still tied to traditional rural industries face big challenges, as commodity producers face stiff competition and thin profit margins in the global economy, leaving many rural communities unsure of their best strategies. In many rural areas, the traditional sources of rural comparative advantage—abundant and cheap land (natural resources) and labor—have been replaced by a new comparative advantage—quality of life. For example, in rural IdMt despite the decreases in manufacturing employment and earnings in the wood products industry in the 1990s, population and total personal income increased throughout the decade.

Schroeder and others[16] document the dramatic rise in tourism (especially cruise tourism) in southeast Alaska. Using the same methodology previously employed by McDowell Group (1991, 1999), Crone estimated the number of tourism-related jobs in rural southeast Alaska at 1752 in 1999. In addition to attracting tourists, rural southeast Alaska enjoys a comparative advantage in attracting migrants seeking quality of life improvements. The area features outstanding recreation opportunities and what Isserman (2000) referred to as AMENities—freedom from congestion, crime, commuting, pollution, and other conflicts of urban life. Much has been written in recent years regarding the ability of these types of amenities to stimulate rural population growth and economic development by attracting both individuals and firms.[17]

Crone used data from the 2000 U.S. census (U.S. Department of Commerce Bureau of the Census, 2002) to examine indicators of socioeconomic well-being for the small (less than 2500 people) rural southeast Alaska communities. Using the amount of per capita funds each community received between 1996 and 2002 from the Southeast Economic Disaster Fund[18] as a proxy for forest products dependence, she classified the communities as timber-dependent if they received more than $1000 per capita from this fund. Using this classification system, she found there was no significant difference in mean median household income, mean per capita income, mean unemployment rate, or mean percentage of families in poverty between this group of 14 communities and the other 13 small rural southeast Alaska communities as a group. Based on this analysis, she could not conclude that the small communities that had a higher dependence on timber harvests were worse off than the other small communities in the region.

The growth of each community will depend on their individual resiliency. Resiliency, in this sense, is defined as adaptability to change and is influenced by more than just the economic structure of a community. It also depends on community leadership, activities like planning for the future, the presence and management of amenities that might attract and keep people in the area, and physical infrastructure (roads, sewers, water) (Crone and Haynes, 2001). Whether the smaller, more isolated and less economically diverse communities in rural southeast Alaska will be able to leverage their many natural amenities to overcome these development obstacles is likely to depend on each community's unique characteristics.

The implications for forest policy of these findings are similar to those from Robertson (in press). Forest management activities that maintain or enhance the comparative advantage of rural southeast Alaska communities in attracting tourists and new residents while maintaining or enhancing quality of life attributes for existing residents are likely to contribute to socioeconomic well-being in these communities. Efforts to improve the competitiveness of Alaska wood products can contribute to economic diversity in some communities,

[16] Schroeder, R., Cerveny, L., Robertson, G. Tourism growth in southeast Alaska: trends, projections and issues. Unpublished manuscript. On file with: L. Kruger, USDA Forest Service, Forestry Sciences Laboratory, 2770 Sherwood Lane, Juneau, AK 99801, USA.

[17] For examples and additional references, see Rudzitis (1999), Johnson and Rasker (1993), Beyers and Lindahl (1996), Crone and Haynes (1999), Nelson (1999), Vias (1999), McGranahan (2000), Southwick Associates (2000), and Pezzini and Wojan (2001).

[18] This fund, established by Congress in 1996, directed the Secretary of Agriculture to "allocate funds to local communities suffering economic hardship because of mill closures and economic dislocation in the timber industry to employ unemployed timber workers and for related community redevelopment projects" (Public Law 104-134, section 101, Title II (a-c4), Public Law 106-113, Title II and Public Law 106-391, Title II).

but it is unlikely that wood products production and employment will ever return to their previous levels in southeast Alaska.

5. Role of the Tongass in the regional economy: past, present, and future

Given the fact that the Tongass National Forest accounts for over 80% of the land base in southeast Alaska, since its creation, the management of this forest has affected the economy of southeast Alaska in many ways. With the establishment of the pulp companies and long-term contracts in the late 1950s, the primary influence on the economy was the provision of timber to the wood products industry. Indeed, as Crone (in press) notes, the early pulp mill proponents envisioned that these multiproduct wood-processing operations would be the principal component of a rural development strategy for the region. Haycox (1997) wrote, "For forty years and more, pulp was king of the Tongass .... Production from the mills proved marketable in a period of remarkable national growth, and the population of southeast Alaska increased significantly."

Beginning in the 1960s and throughout the 1970s, however, as the environmental movement gained momentum, many Americans began to embrace the notion of natural resource preservation. In southeast Alaska, local and national environmental groups began to question forest management practices and began filing suits to halt timber sales in the area (Sisk, 1989). In the 1980s when the timber market plummeted, these groups continued to scrutinize and attack Tongass management policies on both environmental and economic efficiency grounds. Haycox (1997) argued that the increased use of the waters in the forest by fishermen and the dramatic increase in tourism served to elevate the debate over management of the Tongass to the national spotlight. The end result of this debate was the TTRA (1990), which did not end the pulp companies' 50-year contracts but modified them to prevent below-cost sales and removed what many considered subsidies to the pulp companies.[19] This act removed the Tongass timber supply target of 20.4 million m^3 per decade (ANILCA, 1981)(established as part of ANILCA in 1981) and reduced the annual appropriation to support timber sale offerings from $40 million to $4 million. Additionally, the act directed the Forest Service to set the harvest level each year to meet "market demand" and sell timber at a profitable price.

In recent years, there have been increasingly divergent views on the importance of timber harvests from the Tongass to the regional economy. The McDowell Group (1995, 2000a) holds the view that the primary cause of economic decline in southeast Alaska is the reduction in timber supply from the Tongass. Others such as MacMullan and Niemi (1994), Whitelaw et al. (1998), Erickson and Associates (1999) and Erickson (2000) disagree, arguing that there are many other factors occurring both within and outside the forest sector that are greater influences on the economy. The discussion and analysis in previous sections of this paper provide support for the latter view for several reasons.

First, although reductions in timber harvests from the Tongass obviously imply a reduction in logging employment on the Tongass, the reduction in harvests from Native Corporation lands since 1990 have been just as large. Between 1996 and 2001, Alaska timber harvests from private lands decreased by 70% from 1 478 773 to 450 707 m^3 (see Table 19 in Warren, 2004). Koncor, once the second largest timber producer among Alaska Native corporations, ceased logging operations in 2001 (Gilbertsen, 2002). This company's president cited permanent changes in the market for Alaska logs (primarily in the Japanese market) as the main factor in their decision to exit the industry (Wheeler, 2001).

Second, from 1980 to 1987 the Forest Service prepared and offered an annual average volume of 2 100 000 m^3 of timber, but the volume sold and harvested averaged only 1 300 000 m^3. During the market peak from 1988 to 1990, the offered volumes were greater than the sold volumes but less than harvest volumes. For the 1991–2001 period, average volume of-

[19] The long-term sales were revised to make timber sales authorized under these contracts more consistent with independent timber sales in terms of planning, management requirements, and environmental assessment procedures. The revisions also included stipulations to eliminate the practice of over-harvesting old growth, to re-offer timber rejected by the pulp companies as independent sales and subtract this volume from the long-term contract volume, to adjust the price of timber offered under the long-term contracts to levels comparable to independent sale prices, to count utility logs against the contract volume, to assure purchaser road credits are treated the same as in independent sales, and to assure the timber offered meets the same economic criteria used for independent sales.

fered exceeded both the average volume sold and the average volume harvested. The fact that more timber has been offered than sold is likely attributable to the marginal position of Alaska wood products firms in the cyclical, integrated, and increasingly competitive markets for their products (Brooks and Haynes, 1997; Robertson and Brooks, 2001; Stevens and Brooks, 2003).

Third, the decreasing availability of cheap timber was not the only reason the pulp companies pulled out of Alaska. The price volatility and general decline in the demand for dissolving pulp as well as the increasing costs associated with operating older, polluting mills were likely equal, if not greater, factors in their decision to shut down (Crone, in press).

Fourth, the larger forces of globalization and adoption of more efficient production technologies (Crone, in press) and more aggressive marketing strategies in competing regions (Eastin, 2002) have further increased the comparative and competitive disadvantages (as identified in Robertson and Brooks, 2001) of the forest products sector in southeast Alaska. Owing in large part to the unique characteristics of the Alaska forest products sector, including the pulp companies and their long-term contracts, the primary processing requirement, and the increase in logging activity associated with Alaska Native harvests, this sector was able to withstand the severe market downturn in the 1980s without investing in the technological improvements that mills in other regions undertook in order to survive and compete in the global economy. The Southeast Alaska Regional Timber Industry Task Force (1997, p. 31) described the competitive environment in which the older and less efficient Alaska mills must now operate:

Computer assisted cutting, trimming, sorting, grading and shipping have increased both resource recovery and production speed and shipment in regions which already enjoyed cost advantages over Alaska. Advanced mills are not single-product operations but integrated manufacturing plants generating their own energy from wood chips and turning out an array of structurally superior engineered building products. While the cost and quality of timber supply continues to be an issue for the industry, perhaps even more critical is the strategic development of technology to more productively extract the full value from a costly resource.

Fifth, at least some of the recent economic decline must be attributed to the poor market situation for Alaska salmon (McDowell et al., 2001). Alaska's contribution to the global salmon market declined from 40 to 50% in the early 1980s to less than 20% by 2000, primarily because of competition from farmed salmon. Peak salmon prices in 2002 were 54–92% lower than they were in 1988, while total ex-vessel values in the Alaska salmon fishing-industry fell from $600 million to $150 million, and average selling prices for southeastern Alaska purse seine permits fell from $110 000 to $23 000 between 1992 and 2002 (Naylor et al., 2003).

Finally, because the Alaskan economy was growing slower with wages decreasing toward the national average (Robertson, in press), it is likely that some of the late 1990s decline in rural southeast Alaska employment was due to workers seeking better opportunities in faster growing areas in the lower 48 states (Erickson, 2000; Robinson, 2001). Crone (in press) wrote, "Rural southeast Alaska like many other rural areas did not share in the 'new economy' earnings growth of the late 1990s. One reason for this is that rural areas had less employment in and less growth of the producer services sector. This sector ... contributed most to increased urban earnings." Urban areas also tend to specialize in high-tech manufacturing industries, which provided most of the growth in manufacturing earnings, whereas rural areas tend to specialize in slower growing value-added and routine technology manufacturing (Gale and McGranahan, 2001). The wood products-manufacturing and seafood-processing industries in rural southeast Alaska fall into the latter category.

In his test of the economic base model in southeast Alaska communities, Robertson (2003) found that the presence of significant secondary impacts resulting from changes in resource-based economic activity cannot be taken as a matter of fact. In her examination of community indicators of socioeconomic well-being, Crone (in press) did not find a significant difference between small timber-dependent communities and other small communities in rural southeast Alaska. Robertson (in press) and Crone (in press) found that in the increasingly diverse and service-oriented economy that exists in southeast Alaska today, the contribution of the Tongass to the regional economy is much more complex and difficult to quantify.

Although timber from the Tongass continues to play a role and efforts to assist the wood products industry restructure should continue, timber is not likely to be the most important contributor to future socioeconomic well-being in the area. Based on regional, national, and international economic and demographic trends, the roles the Tongass plays as a provider of tourism and recreation opportunities and as the custodian of many of the unique natural amenities and ecosystem values that both attract tourists and enhance the quality of life for existing and potential residents, is likely to be of more importance to the economic vitality of the region. An important aspect of this role for Tongass National Forest managers will be a continuation of their ongoing efforts to help communities identify and implement strategies to enhance their comparative advantage and capture more of the economic benefits from recreation and tourism in ways that least detract from local quality of life.

Metric equivalents

When you know	Multiply by	To find
Cubic meters (log)	220	Board feet, log scale
Cubic meters (lumber)	420	Board feet, lumber scale
Dollars per m^3 (log scale)	4.53	Dollars per thousand board feet (log scale)
Dollars per m^3 (full sawn lumber)	2.358	Dollars per thousand board feet (full sawn lumber)
Dollars per metric ton	0.91	Dollars per standard ton

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Human impacts and the global distribution of extinction risk

Richard G. Davies[1], C. David L. Orme[2], Valerie Olson[3], Gavin H. Thomas[4], Simon G. Ross[1], Tzung-Su Ding[5], Pamela C. Rasmussen[6], Ali J. Stattersfield[7], Peter M. Bennett[3], Tim M. Blackburn[4], Ian P. F. Owens[2,8] and Kevin J. Gaston[1,*]

[1]*Biodiversity & Macroecology Group, Department of Animal & Plant Sciences, University of Sheffield, Sheffield S10 2TN, UK*
[2]*Division of Biology, Imperial College London, Silwood Park, Ascot, Berkshire SL5 7PY, UK*
[3]*Institute of Zoology, Zoological Society of London, Regent's Park, London NW1 4RY, UK*
[4]*School of Biosciences, University of Birmingham, Edgbaston, Birmingham B15 2TT, UK*
[5]*School of Forestry & Resource Conservation, National Taiwan University, 1, Sec 4, Roosevelt Road, Taipei 106, Taiwan, ROC*
[6]*Michigan State University Museum, West Circle Drive, East Lansing, MI 48824-1045, USA*
[7]*BirdLife International, Wellbrook Court, Girton, Girton Road, Cambridge CB3 0NA, UK*
[8]*NERC Centre for Population Biology, Imperial College London, Silwood Park, Ascot, Berkshire SL5 7PY, UK*

Understanding the global geographical distribution of extinction risk is a key challenge in conservation biology. It remains controversial, however, to what extent areas become threat hotspots simply because of high human impacts or due to predisposing ecological conditions. Limits to the taxonomic and geographical extent, resolution and quality of previously available data have precluded a full global assessment of the relative roles of these factors. Here, we use a new global database on the geographical distributions of birds on continents and continental islands to show that, after controlling for species richness, the best predictors of the global pattern of extinction risk are measures of human impact. Ecological gradients are of secondary importance at a global scale. The converse is true for individual biogeographic realms, within which variation in human impact is reduced and its influence on extinction risk globally is therefore underestimated. These results underline the importance of a global perspective on the mechanisms driving spatial patterns of extinction risk, and the key role of anthropogenic factors in driving the current extinction crisis.

Keywords: extinction risk; global biodiversity; human population; species richness; threatened species

1. INTRODUCTION

Understanding the geographical distribution of extinction risk and its causes are key challenges in conservation biology, and are central to determining spatial priorities for the focus of conservation responses. Major determinants of extinction risk across space include not only anthropogenic environmental impacts but also variation in predisposing ecological conditions (Forester & Machlis 1996). The former include human population density, agricultural and urban land-use, species exploitation, introduced species and disease, and anthropogenic climate change (Soulé 1991; Forester & Machlis 1996). Predisposing ecological conditions include, but are not exclusive to, the availability of ambient environmental energy which is thought to influence speciation rates and thus the occurrence of neoendemics (Rohde 1992), the availability of productive environmental energy which is thought to limit overall and individual species population numbers (Wright 1983), absolute species numbers which may influence food web structure and thus the likelihood of extinction cascades (Gaston 2002), and surface topography which influences the occurrence of narrowly distributed species (Richerson & Lum 1980).

Despite a number of valuable regional studies, restrictions to the taxonomic and geographical extent, resolution and quality of previously available data, have thus far largely precluded a full global assessment for a major taxon of the relative roles of human impact and predisposing ecological factors in determining threatened species richness (Kerr & Currie 1995; Balmford *et al.* 2001; McKinney 2001; Norris & Pain 2002; Luck *et al.* 2004; Scharlemann *et al.* 2005). Given that the sensitivity of individual species to human population density has been shown to vary within and between biogeographical regions (Woodroffe 2000), it remains an open question whether the relative importance of factors indicated by individual regional studies will generalize to other regions or globally. The frequently incomplete representation, within individual continents, of global variation in human impact and ecological gradients, as well as the distinct

* Author for correspondence (k.j.gaston@sheffield.ac.uk).

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evolutionary histories of species occurring in different regions, further contribute to this uncertainty. The relevance to conservation policy-making of the answer lies both in its indication of the wider applicability of regional findings and in the confirmation of ultimate causes operating globally. Here, we present an analysis of human and ecological determinants of spatial patterns of extinction risk at continental and global scales. We use a new database on the geographical distribution of the breeding ranges of extant bird species on continents and continental islands on an equal-area grid at a resolution comparable to 1° latitude×longitude (Orme *et al.* 2005).

We tested equal numbers of predictors of human impact and ecological condition so as to avoid *a priori* skewing analyses in favour of finding the greater importance of predictors from one category over the other. Building on previous demonstrations of their potential importance in shaping spatial patterns of threatened species richness, for indices of human impact we used human population density, economic activity (purchase power parity gross domestic product, GDP), and extent of agricultural and urban land-area (Kerr & Currie 1995; Balmford *et al.* 2001; McKinney 2001; Norris & Pain 2002; Luck *et al.* 2004; Scharlemann *et al.* 2005). For ecological gradients, we used mean annual temperature as a measure of available ambient energy, while for productive energy we used the Normalized Difference Vegetation Index (NDVI). In addition to topographical variability (elevation range), we used number of land-cover types (habitat diversity) as an alternative measure of habitat heterogeneity. To minimize the risk of including spurious variables in our analyses, we built a multivariate minimum adequate model (MAM) based on regression methods that accounted for spatial autocorrelation in the response variable. To test whether the results of our global model could have been predicted by analyses conducted at a smaller geographical extent, we used the same methodology to construct models separately for six major biogeographic realms (Olson *et al.* 2001).

2. MATERIAL AND METHODS

(a) *Species data*

The analyses presented here are based on a database of distribution maps for 9626 extant, recognized bird species constructed using a variety of published sources (for details see Orme *et al.* 2005, in press) and following a standard avian taxonomy (Sibley & Monroe 1990). The polygon breeding range vector maps were converted to an equal-area grid using a Behrmann projection at a cell resolution of 96 486.2 m. This grid cell size is equivalent to 1° longitude and 1° latitude at 30° latitude N/S (1/360th of the width of the globe under a Behrmann projection using the WGS84 datum). The global grid therefore contains 360 by 152 cells, omitting the partial cells at latitudes higher than 87.13°. Species were scored as present in a grid cell if any of the available sources indicated that the breeding range fell within the cell boundaries. Threatened species were those classified as Critical, Endangered and Vulnerable, but not those in lower risk categories (Near Threatened, Least Concern) or other categories (Data Deficient and Not Evaluated; BirdLife International 2000). Where necessary, we converted the taxonomy used in BirdLife International (2000) back to the standard avian taxonomy (Sibley & Monroe 1990), and calculated the number of threatened species in each grid cell. Biogeographic realms were delimited using the World Wildlife Fund ecoregions map (Nearctic, Palaearctic, Neotropical, Afrotropical, Indo-Malayan, and Australasian; Olson *et al.* 2001). The final dataset used for analyses omitted grid cells falling within Oceania or Antarctica, since environmental data were not available for these realms. Remaining true oceanic islands, defined as any land area located further than 200 km from the edge of continental shelf, were also omitted since these are known to differ markedly in the kinds and intensities of evolutionary and threatening processes affecting constituent avifauna, in comparison with continental locations (Manne *et al.* 1999; Blackburn *et al.* 2004; Duncan & Blackburn 2004). Finally, so as to avoid bias in terms of the contribution of coastal land-area to the regression models, grid cells with less than 50% land-cover were omitted from the final dataset.

(b) *Environmental data*

Data for the eight selected environmental and human impact predictors (see above) were each re-projected and re-sampled to the same equal-area grid as the species richness data. Human population density, GDP, NDVI, agricultural and urban land-area, and elevation range were all $\log_{10}$-transformed for the analysis.

Sources and raw resolutions of the eight selected environmental variables are as follows: (a) human population density (people km^{-2}) for 1995 at 2.5 arc-min resolution (CIESIN 2003) derived from human population census data for 127 105 administrative units, and based on national population estimates that have been adjusted to match the UN national estimated population for each country; (b) purchase power parity GDP data (US $) for 1990 at 0.25° resolution (CIESIN 2005); (c) agricultural land-area (km^2); (d) urban and built-up land-area (km^2); and (e) number of land-cover types (habitat diversity) occurring in a grid cell, all computed using remotely sensed data for the 12-month period between April 1992 and March 1993 at 30 arc-s resolution classified to the US Geological Survey (USGS) 25-category land-cover classification (USGS 2003*a*); (f) elevation range (m), maximum minus minimum elevation within each grid cell, from 30 arc-s resolution data (USGS 2003*b*); (g) mean annual temperature data (°C) for the period 1961–1990 at 10 min resolution interpolated from station means (New *et al.* 2002); (h) mean annual remotely sensed NDVI for the period 1982–1996 at 0.25° resolution (ISLSCP Initiative II 2005). Agricultural land-area (above) was computed as the sum of all agricultural land-use classes from the USGS data (USGS 2003*a*) (2, dryland cropland and pasture; 3, irrigated cropland and pasture; 4, mixed dryland/irrigated cropland and pasture; 5, cropland/grassland mosaic; 6, cropland/woodland mosaic). In order to standardize the definition of terrestrial land-area across raw environmental datasets, each was overlaid with a high-resolution terrestrial areas map (ESRI 1993) prior to re-sampling to the 1° Behrmann gird. Raw-data cells, or portions of cells, falling outside this definition of land-area were excluded from re-sampling calculations, and the latter were weighted by the land-area associated with each remaining raw-data cell.

(c) *Statistical analyses*

To deal simultaneously with spatial autocorrelation and a response variable that was not normally distributed, analyses were based on a Poisson errors generalized linear mixed

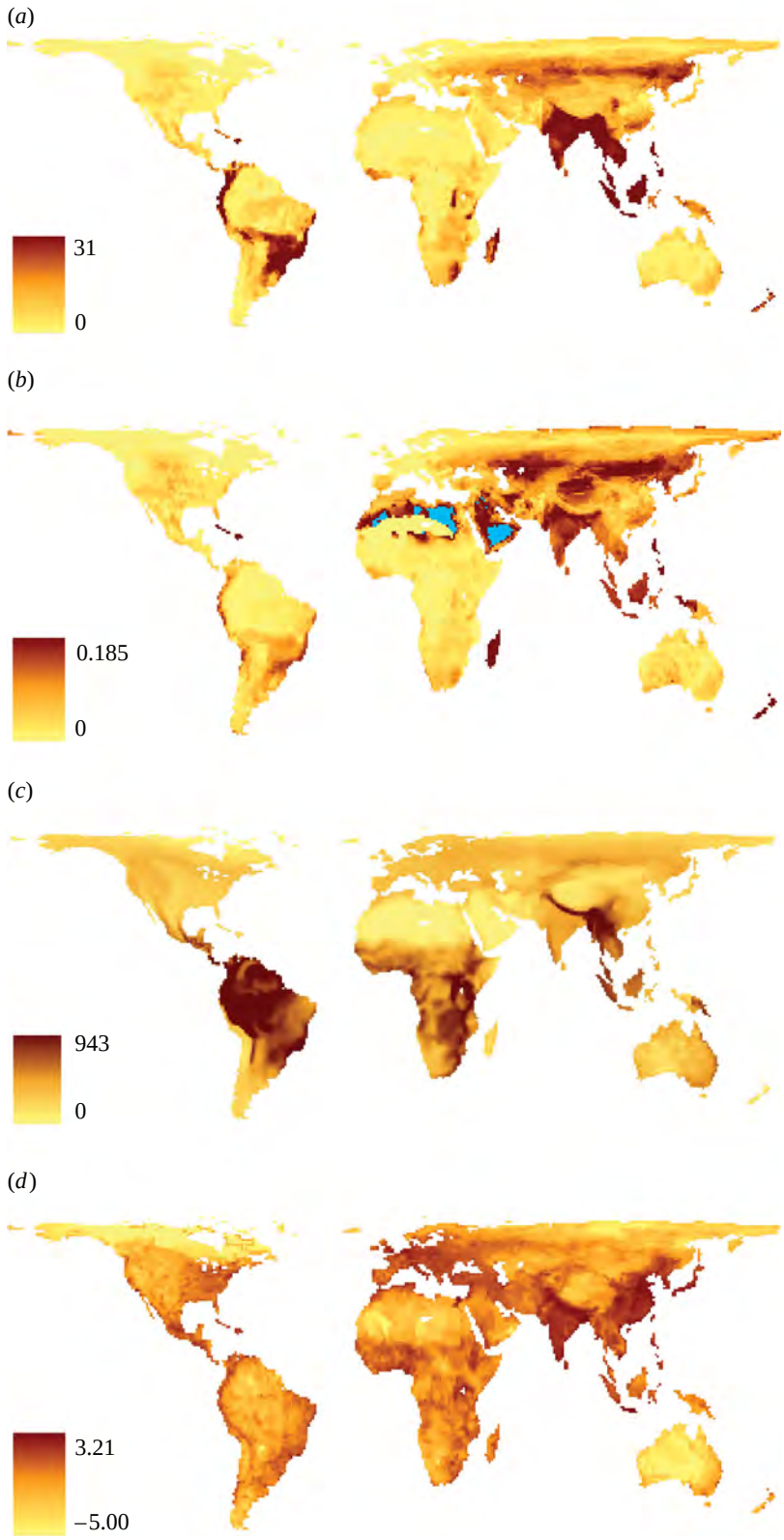

Figure 1. Geographical distribution of avian threatened species richness and of human population density. (*a*) Threatened species richness. (*b*) The proportion of species that are threatened (with a blue mask over those areas with high proportional threat (greater than or equal to 0.1) and low species richness (less than 30), to enable for illustrative purposes use of a scaling which reveals general details of the map which are otherwise obscured). (*c*) Non-threatened species richness. (*d*) Log_{10} (human population density).

modelling (GLMM) method (SAS, Littell *et al.* 1996) in which an exponential spatial covariance structure is fitted with longitudinal and latitudinal cell centroid values as spatial variables using Proc Glimmix v. 1.0 add-in in SAS v. 9.1.3. The choice of the exponential, over other spatial covariance structures, was based on inspection of semi-variograms of non-spatial Poisson error model residuals. Spatial GLMMs took account of the differences among major biogeographical realms in spatial autocorrelation by estimating the maximum geographic distance or range parameter (ρ), measured in degrees, over which spatial autocorrelation in equivalent independent errors model residuals was observed to occur.

This involved estimating ρ from the semi-variogram of residuals of non-spatial Poisson errors models that included the relevant combination of predictors, separately for each realm. All six estimates of ρ were then entered as spatial covariance parameters in the global model, with spatial autocorrelation assumed for observations within the same realm. GLMMs used the pseudo-likelihood (PL) procedure (Wolfinger & O'Connell 1993) that obtains a maximum-likelihood-like estimate of the scale parameter (φ) (Littell *et al.* 1996). PL does not compute a true log-likelihood, precluding use of model selection procedures based on Akaike's Information Criterion, and forward stepwise model-building procedures were employed to determine MAMs. The fit of quadratic as well as linear terms for predictors was tested in order to allow for nonlinear relationships. Estimates of variance explained cannot be derived from spatial models that use PL, so we used percentage of total deviance explained from equivalent non-spatial models as an indication. For all geographical areas modelled, we explored collinearity among predictor variables using tolerance levels (Quinn & Keough 2002). Tolerance levels were sufficiently high (i.e. greater than 0.1, following Quinn & Keough 2002) in all cases except for the Nearctic and Neotropical realms where some redundancy was observed between human population density, GDP and urban land-area (for tolerance values see table 1 in electronic supplementary material). Hence, this had the relatively minor consequence that significance of human population density in the Neotropical MAM could not be separated from the effect of the other two socio-economic predictors.

3. RESULTS AND DISCUSSION

The global distribution of threatened avian species richness exhibits marked large-scale spatial heterogeneity (figure 1*a*), being highest across much of the Indo-Malayan realm and parts of the Neotropics, including areas of the Andes, Amazonia and the Atlantic coastal forests. Previous work has shown that the geographical distribution of threatened species richness is, to some extent, dependent on that of overall species richness (Kerr & Currie 1995; McKee *et al.* 2003). However, the proportion of species threatened (figure 1*b*) is far from constant and does not simply mirror the patterns for absolute numbers of either threatened (figure 1*a*) or non-threatened species (figure 1*c*). Instead, the richest continental or larger-island areas with respect to proportional threat are the southern Palaearctic, Madagascar, and New Zealand.

Our global model revealed that, after controlling for the effect of spatial variation in non-threatened species richness, human population density (figure 1*d*) was the primary global driver of geographical patterns of numbers of threatened bird species, followed by extent of agricultural activity as a secondary human influence (table 1). Environmental factors played a more minor role, with elevation range and NDVI entering as subsidiary factors in the global MAM. Accounting for the effects of human factors and non-threatened avian richness, we might expect gradients such as elevation range and primary productivity to be inversely related to the remaining variation in numbers of threatened species, since we could expect fewer threatened species in pristine areas that coincide with areas of lower human impact. However, the importance of elevation range as a positive predictor of threatened species richness globally may result from the influence of topography on the occurrence of restricted-range species (Jetz *et al.* 2004) and their inherent associated vulnerability to population decline (Stattersfield *et al.* 1998; Manne *et al.* 1999). Alternatively, in some cases this occurrence may result from range contractions from human-impacted lower elevation areas leaving remnant populations in more mountainous regions. Similarly, the positive influence of NDVI may be linked to contraction of the geographic distribution of threatened species to remaining areas of high plant productivity. The global MAM also included the significant negative influence of GDP indicating that, having accounted for other factors, areas of high economic development are coincident with lower numbers of threatened species. This is more likely to result from local extinction resulting in threatened species range contractions from areas of highest economic activity (including urban areas), rather than from any positive influence of economic development on conservation investment and effectiveness.

In contrast to the global model, our analyses of threat within biogeographic realms generally suggested that human impact variables ranked relatively lower in importance compared with ecological predictors (table 2). The only exception to this was Australasia, which showed human population density to be the primary predictor of threatened species richness, ahead even of non-threatened species richness. However, the latter was the primary predictor of numbers of threatened species for all remaining realms, and was the only predictor to enter MAMs for the Nearctic and Palaearctic, the two higher latitude realms with the lowest absolute numbers of threatened species. For the major tropical realms, subsequent predictors entered MAMs in different rank orders, with NDVI, elevation range, and temperature being the second strongest predictors, respectively, for Afrotropical, Neotropical, and Indo-Malayan realms. Likely reasons for the positive slopes for NDVI and elevation range are the same as those proposed for the global MAM, whereas the importance of temperature over NDVI or elevation range in Indo-Malaya is indicative of the impact of widespread lowland deforestation on the large-scale distribution of plant productivity.

These results have important consequences for understanding the mechanisms that underlie contemporary extinction processes. For instance, the differences between the results of our global analyses and those of the realm-specific analyses indicate that the combination of predictors of threatened species richness observed for one realm cannot be assumed to apply either to other realms or globally. One reason for this appears to be that individual realms often contain relatively limited geographical variation in human impacts (figure 2*a*) and/or the numbers of threatened species (figure 2*b*). For example, the Indo-Malayan realm shows the maximum global average levels both of threatened species richness (figure 2*b*) and of human population density, these being an order of magnitude higher than in any other realm. In spite of this, human population density does not even enter the MAM for Indo-Malaya since variation in human population density within the realm is relatively limited at this spatial resolution (figure 2*a*). Hence, the weak

Table 1. Global minimum adequate model for geographic variation in the number of threatened birds. (The minimum adequate model was obtained using forward stepwise procedures. % expl. deviance is the overall percentage of total deviance explained by non-spatial models using the same combinations of predictors as spatial GLMMs. All other test statistics (slope, 95% confidence interval of slope, and *F*-value) refer to spatial GLMM results. Abbreviations are as follows: non-threat spp., number of non-threatened species; GDP, gross domestic product; NDVI, Normalized Difference Vegetation Index (superscripts refer to quadratic terms). Population density, elevation range, agricultural-area, GDP and NDVI were all $\log_{10}$-transformed. $^{***}p<0.001$; $^{**}0.001\leq p<0.01$; $^{*}0.01\leq p<0.05$.)

predictor	estimate	95% conf. interval	$F_{1,13\,858}$	% expl. deviance
non-threat spp.	0.0051	±0.0004	560.63^{***}	41.4
non-threat spp.^{2}	−0.0000003	±0.0000	201.55^{***}	
population density^2	0.0052	±0.0016	43.50^{***}	
agricultural-area^2	0.0049	±0.0021	20.77^{***}	
elevation range^2	0.0090	±0.0040	19.75^{***}	
GDP^2	−0.0158	±0.0084	13.74^{***}	
NDVI^2	1.6600	±1.0894	8.92^{**}	
land-area	0.1090	±0.1162	3.38^{*}	
intercept	−0.5110	±0.4690		

Table 2. Minimum adequate models for geographic variation in the number of threatened birds within biogeographical realms. (Abbreviations and methodology as in table 1.)

realm	predictor	estimate	95% conf. interval	*F*-value	% expl. deviance
Australasian (d.f. = 932)	population density^2	0.0128	±0.0053	21.93^{***}	66.1
	elevation range^2	0.0322	±0.0140	20.25^{***}	
	non-threat spp.	0.000003	±0.0000	18.67^{***}	
	temperature	−0.2783	±0.1634	11.14^{***}	
	temperature^2	0.0025	±0.0016	9.36^{**}	
	NDVI	1.2802	±1.0929	5.27^{*}	
	land-area	−0.0859	±0.2509	0.45	
	intercept	8.4088	±4.3179		
Afrotropical (d.f. = 2313)	non-threat spp.	0.0044	±0.0014	38.92^{***}	26.6
	NDVI	1.5221	±0.6795	19.28^{***}	
	land-area	0.4889	±0.2777	11.90^{***}	
	GDP	0.0655	±0.0419	9.38^{**}	
	non-threat spp.^{2}	−0.000002	±0.0000	4.72^{*}	
	intercept	−1.6776	±1.3679		
Indo-Malayan (d.f. = 881)	non-threat spp.	0.0014	±0.0003	105.94^{***}	51.1
	temperature	0.1484	±0.0685	18.04^{***}	
	temperature^2	−0.0013	±0.0007	14.42^{***}	
	NDVI	0.8523	±0.4784	12.19^{***}	
	agricultural-area	0.0262	±0.0196	6.89^{**}	
	land-area	−0.0442	±0.1226	0.50	
	intercept	−2.1804	±1.7534		
Nearctic (d.f. = 2061)	non-threat spp.	0.0270	±0.0121	19.18^{***}	21.4
	non-threat spp.^{2}	−0.0001	±0.0000	10.03^{**}	
	land-area	0.1586	±0.6768	0.21	
	intercept	−3.7346	±3.0745		
Neotropical (d.f. = 2039)	non-threat spp.	0.0037	±0.0008	78.14^{***}	59.4
	non-threat spp.^{2}	−0.000002	±0.0000	50.28^{***}	
	elevation range^2	0.0228	±0.0083	28.86^{***}	
	population density^2	0.0052	±0.0026	14.80^{***}	
	agricultural-area^2	0.0215	±0.0113	13.93^{***}	
	land-area	0.5051	±0.2740	13.06^{***}	
	agricultural-area	−0.0746	±0.0484	9.13^{**}	
	temperature^2	−0.0008	±0.0005	8.35^{**}	
	temperature	0.0654	±0.0523	6.01^{*}	
	intercept	−2.6566	±1.9331		
Palaearctic (d.f. = 5603)	non-threat spp.	0.0041	±0.0004	356.29^{***}	28.3
	land-area	0.0271	±0.1905	0.08	
	intercept	0.2982	±0.7752		

predictive strength of population density within Indo-Malaya belies the fact that this variable is largely driving the peaks in global avian threatened species richness. The same pattern is repeated across several of the other realms, with intra-realm variation typically being relatively minor in comparison to the global patterns, resulting in the regional models having considerably less power than the full global analysis.

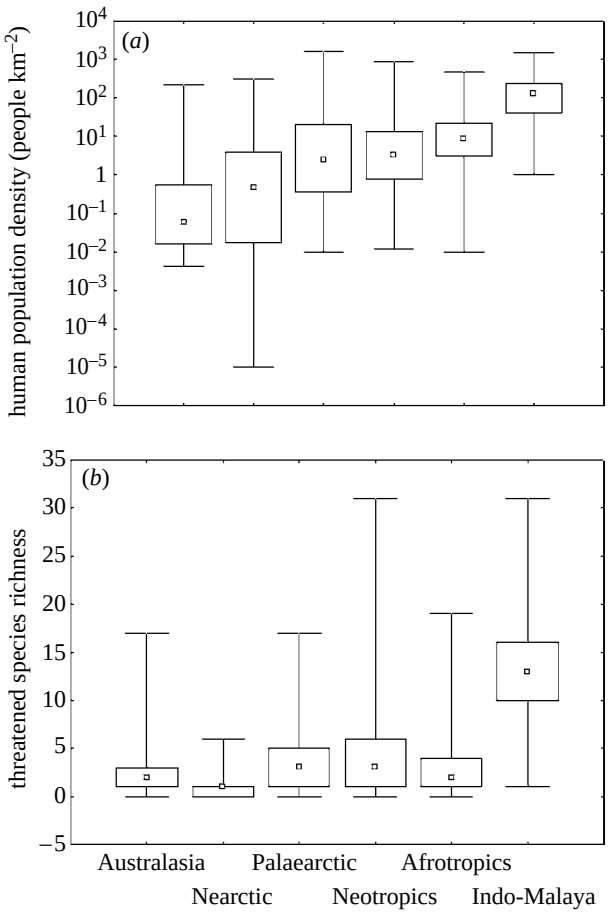

Figure 2. Variation in human population density and avian threatened species richness among biogeographical realms. (*a*) Human population density (log-scaled) per grid cell. (*b*) Numbers of threatened species per grid cell. Points represent medians, boxes indicate interquartile ranges, and whiskers are minimum and maximum values.

Our analyses also indicate likely directions for future work in this area. In particular, the estimates of the percentage of deviance explained by models range from 21 to 66%, indicating that considerable variation in the number of threatened species remains to be explained (tables 1 and 2). This is expected, since time lags in species' responses to threatening processes on the one hand, and removal by local extinction of threatened species from high-threat areas (extinction filtration) on the other, will tend to reduce variation in numbers of threatened species present along spatial gradients in the duration and/or intensity of a given driver of threat (Pimm *et al.* 1995; Balmford 1996; Forester & Machlis 1996). Even globally, therefore, the full impact of human activities will be largely underestimated in the continuing absence of comprehensive historical species distribution data (Cardillo *et al.* 2004). Our findings thus lend support to conservation prioritization initiatives focused on areas of high human population density and impact, as well as those with the best remaining natural habitat. Moreover, lack of marked congruence in the intra-regional spatial distribution of numbers of threatened species across major taxa (Kerr & Currie 1995; Dobson *et al.* 1997) may belie a stronger emergent signal of taxonomic congruence globally. Hence, global studies of other tractable taxa are urgently needed to test the wider validity of these models. Nevertheless, being the first global models of extinction risk for the best-known vertebrate taxon, our findings represent one of the highest-resolution analyses of human impacts on global biodiversity currently available to inform conservation policy.

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ORIGINAL ARTICLE

A multilocus evaluation of ermine (*Mustela erminea*) across the Holarctic, testing hypotheses of Pleistocene diversification in response to climate change

Natalie G. Dawson[1,2], Andrew G. Hope[3]*, Sandra L. Talbot[3] and Joseph A. Cook[1]

[1]*Department of Biology and Museum of Southwestern Biology, University of New Mexico, MSC03 2020, Albuquerque, NM 87131, USA,* [2]*College of Forestry and Conservation, University of Montana, Missoula, MT 59812, USA,* [3]*U.S. Geological Survey, Alaska Science Center, 4210 University Drive, Anchorage, AK 99508, USA*

*Correspondence: Andrew G. Hope, US Geological Survey, Alaska Science Center, 4210 University Drive, Anchorage, AK 99508, USA.
E-mail: ahope@usgs.gov

ABSTRACT

Aim We examined data for ermine (*Mustela erminea*) to test two sets of diversification hypotheses concerning the number and location of late Pleistocene refugia, the timing and mode of diversification, and the evolutionary influence of insularization.

Location Temperate and sub-Arctic Northern Hemisphere.

Methods We used up to two mitochondrial and four nuclear loci from 237 specimens for statistical phylogeographical and demographic analyses. Coalescent species-tree estimation used a Bayesian approach for clade divergence based on external mutation rate calibrations. Approximate Bayesian methods were used to assess population size, timing of divergence and gene flow.

Results Limited structure coupled with evidence of population growth across broad regions, including previously ice-covered areas, indicated expansion from multiple centres of differentiation, but high endemism along the North Pacific coast (NPC). A bifurcating model of diversification with recent growth spanning three glacial cycles best explained the empirical data.

Main conclusions A newly identified clade in North America indicated a fourth refugial area for ermine. The shallow coalescence of all extant ermine reflects a recent history of diversification overlying a deeper fossil record. Post-glacial colonization has led to potential contact zones for multiple lineages in north-western North America. A model of diversification of ermine accompanied by recent gene flow was marginally less well supported than a model of divergence of major clades in response to the most recent glacial cycles.

Keywords
Approximate Bayesian computation, Beringia, coastal refugium, gene flow, island endemism, Mustelidae, North Pacific coast, phylogeography, species tree estimation.

INTRODUCTION

Cyclical climate change through the Pleistocene (2.6 Ma–12 ka) has been a catalyst for rapid speciation (Brochmann & Brysting, 2008; McCulloch *et al.*, 2010). At higher latitudes, continental ice sheets periodically forced dramatic distributional shifts (Hewitt, 1996), including episodes of range expansion followed by isolation and subsequent divergence. These repeated events underlie the generalized process of diversification associated with climate cycles (Hope *et al.*, 2012), although responses varied widely. Generally, the ranges of tundra-associated species expanded during glacial advances, while those of forest-associated species contracted (Fedorov *et al.*, 2008; Hope *et al.*, 2013). Diversification has been complicated by migration, secondary contact and associated gene flow, coupled with isolation in multiple refugia (Abbott *et al.*, 2013). In Europe and North America, a model of refugial isolation followed by latitudinal shifts has been

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the foundation for understanding the evolutionary consequences of ice ages (Hewitt, 1996; Arbogast & Kenagy, 2001; Petit *et al.*, 2003). Phylogeographical investigations have also tied diversification to repeated longitudinal shifts, including intercontinental movement through Eurasia and across the Bering Isthmus (Cook *et al.*, 2005; Waltari *et al.*, 2007; Hoberg *et al.*, 2012).

We examined these biogeographical concepts by assessing the diversification of the ermine (also called the stoat; *Mustela erminea* Linnaeus, 1758), one of the most widely ranging terrestrial mammals (Reid & Helgen, 2008). Ermine occur through boreal Europe, Asia, North America and Greenland (Fig. 1), on adjacent land-bridge archipelagos (e.g. the British Isles, Japan, Alaska's Alexander Archipelago), and in southern forest remnants that constitute montane sky islands. The earliest ermine fossils in Eurasia and North America date to the late Pliocene (*c.* 3 Ma; Kurtén, 1968) and the middle Pleistocene (*c.* 0.7 Ma; Kurtén & Anderson, 1980), respectively.

Despite a broad distribution, previous molecular assessments have uncovered only shallow regional phylogeographical structure. Fleming & Cook (2002) discovered three proximal but distinct mtDNA lineages: one distributed widely across the Palaearctic and north-western North America (hereafter, the Holarctic clade); a second widespread in North America (the continental clade); and a third endemic to a few islands along the North Pacific coast (NPC; the island clade) coinciding with a hypothesized coastal refugium. Martínková *et al.* (2007) detected another coastal refugium in Ireland dated to the Last Glacial Maximum (LGM; *c.* 21 ka). Ermine may also have occupied High Arctic refugia, where prey including collared lemming (Fedorov & Stenseth, 2002) and ptarmigan (Holder *et al.*, 1999) persisted. Northern and coastal refugia indicate that ermine can persist in periglacial areas despite their typical association with boreal forest (Fleming & Cook, 2002; Martínková *et al.*, 2007).

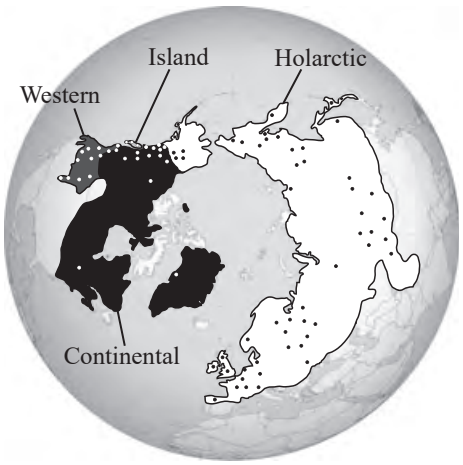

Figure 1 Map of the study area illustrating sampling localities and the range-wide distribution for four major clades of ermine (*Mustela erminea*) as detected in the current study.

Refugial locations and recolonization processes may be inferred from molecular signatures. Other species that share similar distributions with ermine in North America exhibit distinct West Coast lineages, reflecting isolation in coastal refugia (Demboski *et al.*, 1999; Conroy & Cook, 2000) or isolation west of the Rocky Mountains and/or Cascade Range (e.g. Demboski & Cook, 2001; Chavez & Kenagy, 2010). In addition, species consisting of distinct regional lineages across the northern continents indicate multiple refugial areas south of major continental ice sheets during glacial phases (Runck & Cook, 2005; Himes & Kenagy, 2010; Hope *et al.*, 2012). Repeated phylogeographical signatures of other boreal taxa (Arbogast & Kenagy, 2001) allow us to test competing hypotheses concerning the location of Pleistocene refugia for ermine in North America and associated source(s) for recolonization (H_1), and changes in population size, timing and sequence of divergence, and gene flow among contemporary lineages (H_2). We also link previous regional studies of ermine by means of expanded nuclear and range-wide geographical sampling, which are critical for understanding diversification across the Holarctic and for reassessing the conservation implications of coastal endemism along the NPC.

Hypotheses

H_{1a} – North American ermine occupied three glacial refugia, or H_{1b} – more than three refugia during the late Pleistocene. Three distinct clades were identified in north-western North America, and the island clade is likely to have persisted in a NPC refugium (Fleming & Cook, 2002). Fossil ermine dated to the Wisconsinan glacial (*c.* 130 to 12 ka) are known from Alaska and the Yukon Territory (i.e. eastern Beringian refugium) and throughout the conterminous United States (Kurtén & Anderson, 1980; Harington, 2011).

H_{2a} – Cladogenesis of extant ermine lineages began early in their evolutionary history, or H_{2b} – lineages only recently diverged. Considering early- to mid-Pleistocene fossils, ermine may have diverged over deep time-scales. However, if the current distribution is a consequence of recent expansion from a single source followed by allopatric divergence over one or two glacial cycles, clades should exhibit shallow structure. Similarly, gene flow following secondary contact may limit diversification (Abbott *et al.*, 2013). Finally, fluctuations in population size can influence coalescence times. We performed coalescent analyses of population size change and species tree estimation using multiple loci and a mutation rate based on documented vicariance events that are appropriate for investigating recent diversification within ermine. We then used empirical data to form and test multiple models of diversification from simulated data using approximate Bayesian computation (ABC). ABC tests provide a rigorous assessment of population size change, timing of diversification and the extent of gene flow in a mammal species that has been influenced by regional environments across the Holarctic.

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MATERIALS AND METHODS

Sampling and sequencing

We analysed DNA sequences from 237 specimens of *M. erminea* (see Appendix S1 in Supporting Information). Our final analyses were based on 541 bp of the cytochrome *b* gene (cyt *b*, n_{tot} = 160), 272 bp of the control region (CR, n_{tot} = 210) and four independent nuclear loci: 261 bp of the serotonin receptor gene (*HTR1B* intron; n = 39), 480 bp of the agouti signalling protein gene (*ASIP* intron + exon; n = 46), 458 bp of the feline sarcoma proto-oncogene (*FES* intron + exon; n = 54), and 520 bp of the growth hormone receptor gene (*GHR* intron + exon; n = 35). A single individual of *Mustela frenata* was the outgroup (Appendix S1).

Clade demographics

Genetic diversity and demography were analysed for all clades collectively across loci, and for separate clades with mitochondrial data only. Summary statistics calculated in DnaSP 5 (Librado & Rozas, 2009) included genetic diversity indices, and Tajima's D (Tajima, 1989) and Fu's F_S (Fu, 1997) tests for demographic expansion. Population size changes through time were calculated from the coalescent extended Bayesian skyline plot (EBSP) with tree prior for multiple loci (Heled & Drummond, 2008).

Estimation of mutation rate

Few fossils of *M. erminea* (*sensu stricto*) exist and fossil dates are broad (Paleobiology Database; http://www.paleodb.org/). Koepfli *et al.* (2008) suggested a time to the most recent common ancestor (TMRCA) for *M. erminea* and other mustelids of *c.* 3.5 Ma but with a confidence interval *c.* 2 Myr wide. We estimated a mutation rate for cyt *b* using isolation dates (bathymetric and isotopic data; Yoshikawa *et al.*, 2007) for an extant insular lineage of *Mustela nivalis* on Honshu, Japan. We assumed that the closely related ermine shares a similar mutation rate (Kumar & Subramanian, 2002).

Methods for estimating mutation rate were adapted from Hope *et al.* (2010). Briefly, we retrieved GenBank cyt *b* sequences of *M. nivalis* from Honshu, Hokkaido, Russian Primorye and South Korea. Honshu is isolated by the Tsushima and Tsugaru straits, both *c.* 130 m in depth, allowing only limited connections through the Pleistocene. At 0.33 Ma [marine isotope stage (MIS)-10], 0.14 Ma (MIS-6; Yoshikawa *et al.*, 2007) and the LGM (0.021 Ma; MIS-2), sea levels dropped, creating land-bridge connections or narrow frozen straits. A rate greater than 20% Myr^{-1} for the cyt *b* gene would be necessary to accommodate an LGM connection, so this possibility was excluded. In BEAST 1.6.1 (Drummond & Rambaut, 2007), we grouped samples from Honshu separately and estimated clock rate with a tree root fixed to either 0.33 Ma or 0.14 Ma. These two calibrations resulted in rates for the cyt *b* gene of 2.1% Myr^{-1} and 5.1% Myr^{-1} (equivalent to 4.2% and 10.2% divergence rates). Although fossils were not incorporated into mutation rate estimation, we used fossils to refine rate estimates a posteriori. Between the two point mutation rates, we considered 2.1% Myr^{-1} to be the most realistic because the coalescence time of ermine and the closely related *M. frenata* that used this rate and based on multiple loci coincided closely with the earliest ermine fossils, and because fossil dates generally underestimate the first occurrence.

Gene trees, species trees and divergence dating

Markov chain Monte Carlo (MCMC) searches of tree space were performed in MrBayes 3.1 (Ronquist & Huelsenbeck, 2003). The convergence of MCMC runs (with burn-ins) was assessed in TRACER 1.4 (Rambaut & Drummond, 2007), with burn-ins adjusted for convergence among analyses. Phylograms were visualized in FigTree 1.2.2 (Rambaut, 2009). Haplotype networks for all loci were constructed in Network 4.610 (Bandelt *et al.*, 1999). We used *BEAST (Heled & Drummond, 2010) to co-estimate all genealogies embedded within a species tree. We also repeated *BEAST analyses using only mitochondrial loci and using only nuclear loci.

ABC model simulations

We developed 12 models of diversification with accompanying parameters and command lines (see Fig. S1 in Appendix S2) to test hierarchical models of population size, timing and levels of gene flow associated with cladogenesis. For each model, one million simulated datasets were produced in DIYABC 1.0.4.46beta (Cornuet *et al.*, 2010) for two mtDNA loci and four autosomal nuclear loci. All parameters were assigned a uniform prior (interval of $\pm$ 0.5 $\times$ parameter value) except for the rate of gene flow (*ra*, for which the interval remained at default values). All models considered four clades: Holarctic, island, continental and western. The Holarctic and island clades formed a lineage that was sister to the continental and western clades together. Mutation rates for all models were based on empirical values estimated from species tree analyses assuming a cyt *b* rate of 2.1% Myr^{-1}. Transition/transversion ratios (K) and the proportion of invariant sites (P_{inv}) were set to mean values for autosomal and mitochondrial loci from empirical data.

The first four models (N_e models; Fig. S1a–d) tested scenarios of population size change through time. All shared a topology consistent with the species tree reconstruction, and divergence times remained constant. All clades underwent a 'bottleneck' following initial divergence, and then their populations grew to reflect population size changes calculated from EBSPs. The population sizes for N_e model 1 followed values obtained from individual clade EBSPs; the population sizes for N_e models 2 to 4 reflected values obtained from an EBSP for all clades combined to represent size changes for the entire species. For N_e model 2, population size was assigned relative to the population sizes of clades, in the ratio

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of sizes provided from individual EBSPs, and resulting in a constant total population size through the tree. For N_e model 3, population size was assigned equally among clades and total population was constant through the tree. For N_e model 4, population sizes were constant along each branch, meaning a fourfold increase in total population through the tree from root to tips (i.e. expansion through time). Following simulations, the N_e model most similar to the empirical dataset provided the population sizes that were used in subsequent models.

The second four models (temporal models; Fig. S1e–h) tested different timings of cladogenesis. Temporal model 1 reflected a recent (Pleistocene/Holocene; 21–7 ka) fragmentation of all clades. Temporal model 2 reflected a recent divergence of the Holarctic and island clades and continental and western clades, respectively, but deep divergence of both combined lineages. For temporal model 3, the divergences of extant clades were dated to the onset of the Wisconsinan glacial (*c.* 100 ka); for temporal model 4, the divergences of extant clades were dated to the onset of the Illinoian glacial (*c.* 200 ka). Divergence times from the temporal model most similar to the empirical dataset were assigned to all subsequent models.

The final four models (gene flow models; Fig. S1i–l) tested scenarios of gene flow. Gene flow model 1 was identical to the best temporal model (no gene flow). Model 2 assigned gene flow recently (Pleistocene/Holocene transition) between closely related clades. For model 3, gene flow was assigned both recently and coincident with the Sangamon interglacial (*c.* 130–100 ka). The final model assigned gene flow coincident with the Holocene (21–7 ka) and the two most recent interglacials (*c.* 100 ka; *c.* 200 ka).

Statistical evaluation of simulated data from each set of models included pre-evaluation of model–prior combinations to detect models unable to explain the empirical dataset and the computation of posterior probabilities of models including both direct and logistic regression methods with default values. For additional details of methods, see Appendix S2.

RESULTS

Clade demographics

Mitochondrial genealogies identified four clades generally coincident with geography (Fig. 1). Low genetic diversity characterized all loci (Table 1). Significant geographical expansion (and growth) was evident for the continental and Holarctic clades, with only minimal expansion in the western and island clades (Table 1). Among the nuclear loci, only *ASIP* indicated significant expansion. Timing of population size change from EBSPs indicated growth beginning prior to 50 ka for the Holarctic clade (Fig. 2a) and at *c.* 25 ka for the continental clade (Fig. 2b).

Gene trees, species trees and divergence dating

The relationships among the four major clades were generally consistent across mitochondrial loci, while nuclear genealogies were largely unresolved (Fig. S2 in Appendix S2), although analysis of all loci from species tree reconstruction lent strong support for the four clades, with a sister relationship between the continental and western clades and between the Holarctic and island clades (Fig. 3). Both clade pairs (continental + western; Holarctic + island) were reciprocally monophyletic. Haplotype networks generally reflected the low divergence and low resolution among clades for nuclear loci (Fig. S3 in Appendix S2). Haplotypes within clades were geographically widespread, although some regional affiliations included Europe and Central Asia within the Holarctic clade.

TMRCA estimates for the main clades varied depending on the loci used (Table 2; Fig. 3): estimates were shallower with all six loci and deeper when using only mitochondrial loci.

Table 1 Range-wide demographic statistics for 237 samples of ermine (*Mustela erminea*). Loci include partial mtDNA control region (CR) and cytochrome *b* (cyt *b*) and partial introns from nuclear 5-HT1B receptor gene: serotonin receptor (*HTR1B*), agouti signalling protein (*ASIP*), feline sarcoma proto-oncogene (*FES*) and growth hormone receptor (*GHR*). Statistics include sample size (n), sequence length (l), number of haplotypes (h), haplotype diversity (Hd), nucleotide diversity (π), Tajima's D and Fu's F_S. Significance of D and F_S is indicated as follows: $^*P \leq 0.05$; $^{**}P \leq 0.01$; $^{***}P \leq 0.001$.

Gene – group	n	l (bp)	h	Hd	π	D	F_S
CR – all	210	271	42	0.906	0.0117	−1.696**	−30.143***
CR – continental	52	263	14	0.710	0.0047	−1.945**	−9.931***
CR – Holarctic	129	264	28	0.875	0.0074	−1.978**	−21.830***
CR – island	14	264	3	0.560	0.0029	0.558	0.292
CR – western	15	262	6	0.790	0.0056	−0.198	−1.709
cyt *b* – all	160	541	46	0.914	0.0174	−0.523	−9.959*
cyt *b* – continental	31	541	8	0.499	0.0018	−2.053***	−3.964**
cyt *b* – Holarctic	114	541	33	0.873	0.0045	−1.990**	−26.784***
cyt *b* – island	4	541	2	0.500	0.0009	−0.612	0.172
cyt *b* – western	11	541	3	0.345	0.0010	−1.600*	−0.537
HTR1B	39	259	4	0.538	0.0023	−0.325	−0.571
ASIP	46	480	13	0.872	0.0036	−0.477	−6.127***
FES	54	449	4	0.535	0.0013	−0.265	−0.463
GHR	35	520	6	0.645	0.0016	−0.794	−1.956

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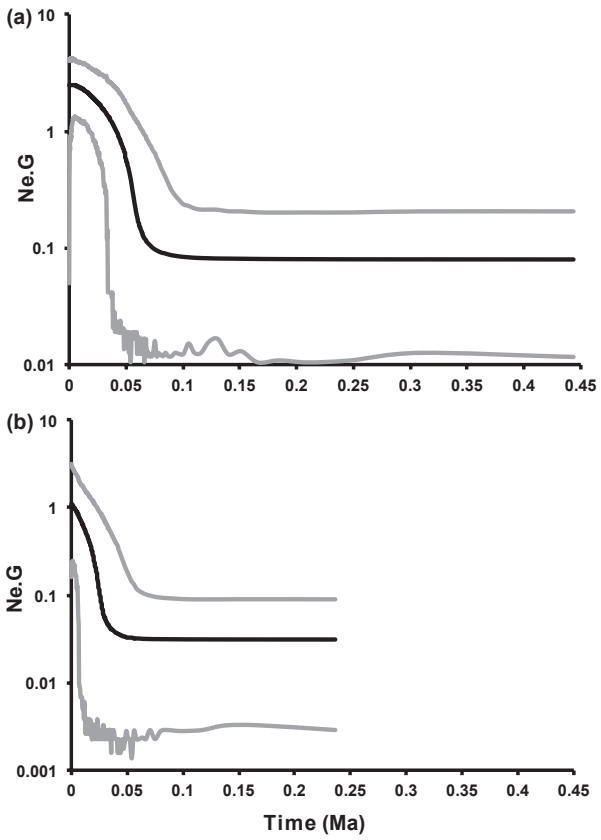

Figure 2 Extended Bayesian skyline plots using six independent loci for (a) the Holarctic clade and (b) the continental clade of ermine (*Mustela erminea*). Plots indicate population growth from past (right) to present (left) including median population size through time (black line) and 95% highest probability distribution (grey interval). Log-transformed *y*-axes represent population size as a function of effective size (N_e) and generation time (G).

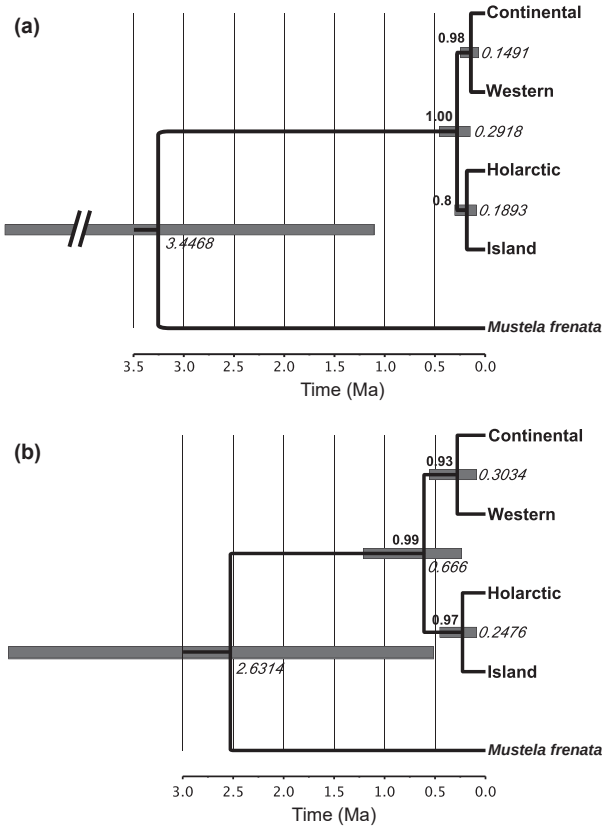

Figure 3 Species trees using (a) six loci or (b) two mitochondrial loci for ermine (*Mustela erminea*). Analyses, performed in *BEAST, provide posterior probability nodal support (bold, left of node) and coalescence times (millions of years ago, Ma; italics, right of node). Shaded bars indicate 95% confidence intervals for coalescence estimates, consistent with values in Table 2. Topologies are ultrametric and proportional to evolutionary time (bottom), which extends from present (right) to past (left).

Conversely, tree roots were older when estimated using six loci than when based only on mtDNA (Fig. 3). This observation was also true for a species tree based only on nuclear loci (Fig. S4 in Appendix S2). Using all loci, the TMRCA for each clade pair coincided with the Illinoian–Saalian glacial (*c.* 150–190 ka; Gibbard & van Kolfschoten, 2004) and the TMRCA of all clades coincided with the pre-Illinoian (*c.* 300 ka; Table 2, Fig. 3). Using only mitochondrial loci increased the divergence estimates to *c.* 300 ka or *c.* 250 ka for pairs of clades and *c.* 660 ka for the whole species (Table 2, Fig. 3). The TMRCA of *M. erminea* and *M. frenata* was estimated at 3.4 Ma or 2.6 Ma (based on six or two loci, respectively; Table 2, Fig. 3).

ABC model simulations

The N_e model most closely resembling the empirical data divided the population size from the combined ermine dataset along branches in proportions reflecting the population sizes that were calculated for individual clades (N_e model 2; Fig. S1). Both direct and logistic regression methods supported N_e

Table 2 Time to most recent common ancestor (TMRCA) estimates (mean and 95% confidence intervals in million years ago, Ma) for ermine (*Mustela erminea*) from species tree reconstructions based on 237 specimens and six loci: cytochrome *b* (cyt *b*); control region (CR); partial 5-HT1B receptor gene; agouti signalling protein; feline sarcoma proto-oncogene; and growth hormone receptor. Reconstructions based on two loci used cyt *b* and CR. TMRCA estimates correspond to those in Fig. 3.

Number of loci	Group	TMRCA (Ma)	95% CI (TMRCA; Ma)
6	1: Continental + western	0.1491	0.0639–0.2442
	2: Holarctic + island	0.1893	0.0863–0.3038
	3: 1 + 2	0.2918	0.1489–0.4552
	4: Tree root	3.4468	1.1169–5.9687
2	1: Continental + western	0.3034	0.0878–0.4527
	2: Holarctic + island	0.2476	0.0868–0.4527
	3: 1 + 2	0.6660	0.2348–1.2117
	4: Tree root	2.6314	0.5125–4.7269

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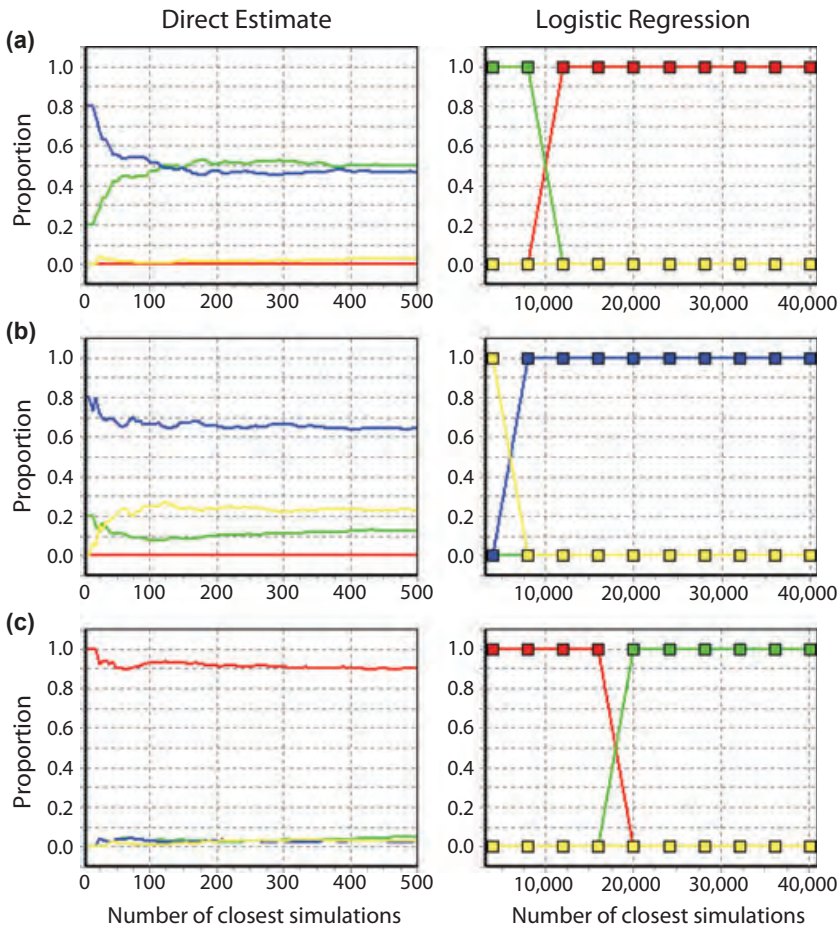

Figure 4 Approximate Bayesian computation model comparisons considering results of direct (left) and logistic regression (right) analyses for (a) N_e models, (b) temporal models and (c) gene flow models of diversification for ermine (*Mustela erminea*). Models are colour-coded within each set: red, model 1; green, model 2; blue, model 3; yellow, model 4. Direct analyses considered the proportion of times that a model was represented in the *n* closest datasets to empirical data from the 500 closest simulations. Logistic regressions considered the *n* closest datasets to empirical data reported as 10 point-estimates (regression intercept) with the proportion of the model represented as the dependent variable, and differences between the observed and simulated dataset summary statistics as the independent variable. Models correspond to Fig. S2 in Appendix S2 and data values are reported in Table S1 of Appendix S2.

model 2 (Fig. 4; Table S1 in Appendix S2), although the distributions of simulated data for N_e models 2 and 3 were closely overlapping (Fig. S5 in Appendix S2). Using the population sizes from N_e model 2, simulations indicated that temporal model 3 (divergences dated to the onset of the Wisconsinan glacial) best coincided with empirical data (see Figs 4 & S5 and Table S1). With divergence times set to temporal model 3, gene flow model 1 (no gene flow) most closely resembled the empirical data, although gene flow model 2 (recent gene flow) was a close second, and all four gene flow models produced closely overlapping distributions (Fig. S5).

DISCUSSION

H_1: Location of Pleistocene refugia and sources for post-glacial recolonization

We documented a new clade of ermine in western North America, consistent with the persistence of the species in four refugia in North America during the Wisconsinan glacial. The western and continental clades indicated two southern refugia for ermine in North America that parallel the primary east/west division of several other mammals (Arbogast & Kenagy, 2001) and the Holarctic and island clades reflect Beringian and NPC refugia. Ermine persisted through several glacial cycles in North America, with the earliest fossils, dating to *c.* 0.7 Ma, coming from Arkansas and Kansas (Kurtén & Anderson, 1980). Fossils from the last interglacial (Sangamon; 130 ka) have been found in the Yukon Territory and Utah, while fossils from the Wisconsinan glacial and the Holocene have been recovered from eastern Beringia (Yukon), the central Rockies (Idaho and Colorado), Texas and Arkansas (Kurtén & Anderson, 1980; Harington, 2011).

A lack of evidence of Holocene population expansion for the western clade, coupled with few common haplotypes, is consistent with contraction into remnant habitat in western North America since the LGM (Fig. 1). Holocene demographic contraction was also suggested for European populations (Martínková *et al.*, 2007). In contrast, the broad distribution and signature of expansion evident for the continental clade were consistent with Holocene colonization of deglaciated regions in Canada and Alaska. There was no evidence to support a High Arctic refugium for ermine during the LGM, although ermine may have occurred in the High Arctic earlier in the Wisconsinan. The lack of significant genetic structure in the continental clade indicated a single geographical source. We suggest that continental ermine spread north from a southern refugium (consistent with fossil evidence) located east of the Rocky Mountains.

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Northward movement was likely to have been rapid and coincident with the expansion of boreal vegetation following the LGM (Jaramillo-Correa *et al.*, 2004; Runck & Cook, 2005; de Lafontaine *et al.*, 2010; Hope *et al.*, 2012). Further north, significant expansion also characterized the Holarctic clade (Table 1), reflecting the rapid spread of ermine across Eurasia. The geographical distribution of haplotypes through Eurasia indicated multiple potential sources in Ireland, continental Europe and Central Asia. Fossil evidence from Beringia during the Wisconsinan (Kurtén & Anderson, 1980; Harington, 2011) may represent populations that were subsequently extirpated. Finally, the island clade was restricted to the Prince of Wales (POW) and nearby Haida Gwaii archipelagos (Fleming & Cook, 2002). Low genetic diversity is likely to indicate a small refugial population with a coalescent history that may only span the most recent glacial period.

H_2: Timing and sequence of evolution among ermine lineages

This multilocus coalescent approach revealed a recent history of diversification among extant ermine. TMRCA was no earlier than *c.* 660 ka and possibly as recent as *c.* 122 ka (Table 2, Fig. 3), a surprising finding given that there are European fossils dated to *c.* 3 Ma (Kurtén, 1968). The knowledge that divergence times calculated from mitochondrial loci are often overestimated compared with nuclear loci lends further support to recent diversification (Smith & Klicka, 2013). Even the deeper mitochondrial divergence estimates revealed that the current geographical structure resulted from a recent pulse of diversification, possibly as a consequence of stronger and longer glacial phases in the late Pleistocene (Hope *et al.*, 2013). Cladogenesis at 0.66 Ma (Table 2, Fig. 3b) may coincide with the first intercontinental colonization based on the earliest North American fossils (*c.* 0.7 Ma; Kurtén & Anderson, 1980). Genetic analysis of fossils from the Conrad Fissure in Arkansas might clarify the number and timing of intercontinental colonizations.

Our analyses supported three intercontinental colonization events for ermine. Given the fossil record, ermine are likely to have entered North America through Beringia in the mid-to-late Pleistocene and diverged around 300 ka. A second invasion from Eurasia occurred as late as the penultimate glacial (Illinoian; *c.* 130–200 ka), probably along a southern coastal route through western Beringia (Hope *et al.*, 2013), with subsequent isolation along the NPC during the Wisconsinan glacial. In a similar time-frame, southern populations are likely to have diverged along an east/west axis, possibly across the Rocky Mountains. Finally, during the most recent glacial leading up to the Holocene, ermine representing the Holarctic clade colonized eastwards through Beringia. As the Wisconsinan ice sheets receded, the continental clade expanded northwards and attained proximity to the Holarctic and island clades (Fig. 1).

Although there are potential pitfalls of using multiple discordant gene trees to infer clade relationships (Degnan & Rosenburg, 2006), species trees using all loci (Fig. 3) or only nuclear loci (Fig. S4 in Appendix S2) resulted in the same topology and demonstrated that adding loci narrows the confidence intervals around node ages and reduces potential homoplasy near the tree root (Fig. 3). Despite well-supported species trees, stochastic lineage sorting is inherently difficult to distinguish from gene flow (Reid *et al.*, 2012). Multiple mitochondrial haplotypes were widely distributed within both Holarctic and continental clades, indicating population expansion and within-clade connectivity. However, some nuclear haplotypes were shared across clades and individuals sharing alleles were often located near major contact zones (Figs S2 & S3 in Appendix S2), particularly in eastern Beringia between the Holarctic and island clades and in western North America between the continental and western clades (Fig. 1). This may reflect recent colonization, the retention of nuclear alleles, the maintenance of gene flow or some combination of these. Gene flow both within and among major regions may therefore be an important feature of ermine biogeography. The periodicity of glacial cycling, particularly during the early to middle Pleistocene, may have been too brief to initiate persistent differentiation. With additional loci and emerging genomic methods available to assess the degree and timing of hybridization, ermine constitute an excellent group for exploring the dynamics between divergence and intermittent gene flow (Miller *et al.*, 2012; Twyford & Ennos, 2012).

The timing of divergence and potential gene flow is also more rigorously evaluated by statistical simulation of realistic evolutionary scenarios (Beaumont, 2010). Phylogeographical analyses such as ABC set the stage for rigorous statistical tests of hypotheses regarding high-latitude diversification processes (Beaumont, 2010). Preliminary genealogies and species trees provided a null topology for the timing and sequence of differentiation events, while demographic statistics provided estimates of population changes through time. Hierarchical model simulations then provided controls for sets of variables. The best-supported N_e model used population sizes for clades that reflected their relative empirical genetic diversity. The most strongly supported topology from ABC analyses was consistent with cladogenesis resulting from the onset of glacial phases and corresponding regional isolation. This model (no gene flow) exhibited minimally higher support than a model incorporating recent gene flow between the most closely related populations, so the possibility of contemporary hybridization deserves further investigation.

A closer look at islands of the NPC

Refugia coincident with coastal land-bridge archipelagos may support populations that have experienced prolonged isolation due to rising sea levels that maintained separation following glacial phases. The distributions of two distinct and geographically restricted ermine phylogroups are consis-

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tent with independent coastal refugia – an Irish refugium (Martínková *et al.*, 2007) and an NPC refugium in North America (Fleming & Cook, 2002; Dawson, 2008). Both constitute land-bridge archipelagos, although the Irish lineage maintained relatively high diversity and is minimally divergent from 'mainland' ermine of the Holarctic clade (isolation dating to *c.* 23 kyr), whereas ermine of the island clade of the NPC exhibited low genetic diversity with population isolation persisting until at least the Last Glacial (prior to 100 ka; Table 2, Fig. 3). The relative timing of divergence and observed levels of genetic diversity have implications for the conservation of distinct lineages and for understanding the influence of coastal refugia on regional patterns of diversity and community assembly. Small populations that have experienced extended isolation in insular refugia – such as those shown by endemic genetic signatures in ermine along the NPC – may represent significant components of a species' genetic legacy, and may be important for considering future persistence.

During the Last Glacial (*c.* 120–11 ka), coastal communities of the NPC were shifted onto the exposed continental shelf, isolated from other mainland areas by the Cordilleran ice sheet and are likely to have been further fragmented by major tidewater glaciers (see Fig. 3 in Carrara *et al.*, 2007; Cook & MacDonald, 2013). Molecular studies have identified cold-tolerant species that exhibit distinct island lineages along the NPC (e.g. Talbot & Shields, 1996; Demboski & Cook, 2001; Topp & Winker, 2008), although few of these endemic lineages are as divergent as in ermine (Soltis *et al.*, 1997; Weckworth *et al.*, 2005; Cook *et al.*, 2006; Dawson *et al.*, 2007; Shafer *et al.*, 2011). Molecular studies on salmon (*Oncorhynchus keta*; Kondzela *et al.*, 1994), spruce grouse (*Falcipennis canadensis*; Barry & Tallmon, 2010) and numerous plants (e.g. Soltis *et al.*, 1997), and a fossil record from caves representing a continuous sequence of radiocarbon ages from 40 to 10 ka (Heaton & Grady, 2003), provide further evidence that POW was a refugium for relatively diverse communities during the Last Glacial, although the community composition changed as the climate warmed. The flora during the Last Glacial consisted of graminoid tundra with willows (*Salix* spp.; $\geq$ 30 ka) suggesting a cold, arid climate (Brubaker *et al.*, 2005; Zazula *et al.*, 2005). This community was replaced by birch shrub and sedge–moss tundra (15–13 ka) indicating moister summers and warmer winters. The earliest trees arrived *c.* 12 ka (*Picea*) and 11 ka (*Pinus*), based on dated stumps and pollen found near Haida Gwaii (Fedje & Josenhans, 2000). Species such as ermine that persisted in the NPC despite marked community turnover are thus characterized by broad ecological tolerances, although these species may have experienced significant reductions in genetic variability.

Ermine of the island clade are endemic to only a few islands of the POW (MacDonald & Cook, 2009) and Haida Gwaii (Heusser, 1989) archipelagos, but representatives of both the continental and Holarctic clades of ermine have recently colonized a few islands of the NPC. The close proximity of multiple lineages presents strong conservation implications for the maintenance of diversity and for evolutionary dynamics including competition, hybridization and the exchange of parasites (Koehler *et al.*, 2007). Further documentation of these three clades throughout the NPC is needed. The close genetic associations of island-clade ermine on Haida Gwaii and POW islands suggest a Last Glacial connection between these islands. Steller's jay (Topp & Winker, 2008) and several lichens and fungi (Geiser *et al.*, 1998) conceivably share similar signatures.

The divergences between pairs of ermine lineages was assessed with the K/θ ratio (Birky, 2013) with values of 5.5 (based on cyt *b*) for island and continental clades and 0.8 for island and Holarctic clades. Values greater than or equal to 4 may represent species-level differentiation under the evolutionary genetic species concept (Birky, 2013). Although we do not yet recommend taxonomic revision, these values, coupled with multilocus analyses, provide strong evidence for multiple evolutionarily significant units (Moritz, 2002) along the NPC. Understanding the vulnerability of archipelago populations is essential, particularly considering that high endemism coupled with environmental variability increases the risk of extinction or extirpation (Weir & Schluter, 2007).

CONCLUSIONS

Multiple species, as exemplified by ermine, attained vast distributions across the northern continents in relatively short time-frames through the late Pleistocene, with intercontinental movement through Beringia being an important consequence of expansion. Common phylogeographical structure over numerous ecological components of Arctic diversity presents the possibility of parallel differentiation among related taxa (Dawson, 2012), but the realization of both common and idiosyncratic responses also provides key insights. Because several mustelids are widely distributed across the Holarctic, assumptions of parallel evolution with niche partitioning or interdependence should be tested. For carnivores such as ermine, we may also predict codifferentiation between specialized predators and their prey, but whereas many small prey species have developed multiple distinct phylogroups across the Holarctic attributable to Pleistocene climate cycles (Hewitt, 1996), predators such as ermine seem to maintain a higher degree of continent-wide connectivity through glacial cycles, highlighting the value of comparative investigation.

Ermine were influenced by localized isolation in coastal and periglacial areas or broad regional differentiation due to barriers such as continental ice sheets. However, reference to these as 'refugia' during glacial periods is potentially misleading for a species that can seemingly thrive and maintain a broad distribution during glacial advances (Bennett & Provan, 2008). Rather, coastal refugia along the NPC or in Ireland (Martínková *et al.*, 2007) are indicative of confined isolates that may or may not have experienced a significant

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loss of genetic diversity through potential bottlenecks. True interglacial refugia may be represented by isolated patches of temperate habitat at the southern extent of the boreal biome in the form of sky islands, and yet the genetic evidence still suggests the long-term maintenance of close associations among these terrestrial islands.

The diversification of ermine through the last three glacial cycles parallels the late-Pleistocene evolutionary history of numerous temperate taxa, although the extent of interaction between contemporary lineages along zones of contact remains to be explored. North-western North America, as shown by an increasing number of genetic assessments, was a dynamic evolutionary arena with multiple transition zones between lineages, biomes and regional environments.

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SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article:

Appendix S1 Specimens examined, sample localities, GenBank accession numbers.

Appendix S2 Supplementary methods, results (including supplementary tables and figures) and discussion.

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BIOSKETCH

This research is part of **Natalie G. Dawson**'s doctoral dissertation. She studies Nearctic mustelids as part of her research and education programme at the University of Montana. She is associate director of the Wilderness Institute within the College of Forestry and Conservation at the University of Montana (UM) and currently directs and teaches the Wilderness and Civilization Program, an experiential education programme at UM.

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Historical biogeography sets the foundation for contemporary conservation of martens (genus *Martes*) in northwestern North America

Natalie G. Dawson, Jocelyn P. Colella,* Maureen P. Small, Karen D. Stone, Sandra L. Talbot, and Joseph A. Cook

Wilderness Institute, University of Montana, 32 Campus Drive, Missoula, MT 59812, USA (NGD)
Department of Biology and Museum of Southwestern Biology, University of New Mexico, MSC03-2020, Albuquerque, NM 87131, USA (JPC, JAC)
Washington Department of Fish and Wildlife, 1111 Washington St SE, Olympia, WA 98501, USA (MPS)
Department of Biology, Southern Oregon University, Ashland, OR 97520, USA (KDS)
United States Geological Survey, Alaska Science Center, 4200 University Dr., Anchorage, AK 99508, USA (SLT)

** Correspondent: jcolella.jc@gmail.com*

Effective conservation of insular populations requires careful consideration of biogeography, including colonization histories and patterns of endemism. Across the Pacific Northwest of North America, Pacific martens (*Martes caurina*) and American pine martens (*Martes americana*) are parapatric sister species with distinctive postglacial histories. Using mitochondrial DNA and 12 nuclear microsatellite loci, we examine processes of island colonization and anthropogenic introductions across 25 populations of martens. Along the North Pacific Coast (NPC), *M. caurina* is now found on only 2 islands, whereas *M. americana* occurs on mainland Alaska and British Columbia and multiple associated islands. Island populations of *M. caurina* have a longer history of isolation reflected in divergent haplotypes, private microsatellite alleles, and relatively low within-population diversity. In contrast, insular *M. americana* have lower among-population divergence and higher metrics of within-population diversity. On some NPC islands, introductions of *M. americana* may be related to decline of *M. caurina*. Long-term persistence of these species likely has been influenced by anthropogenic manipulations, including wildlife translocations and industrial-scale deforestation, yet, the distinctive histories of these martens have not been incorporated into natural resource policies.

Key words: carnivore, endemic, introductions, islands, marten

Genetic parameters provide key indicators of the vulnerability of natural populations to extinction (Frankham 2005). Island populations, because they are typically small and isolated, often exhibit reduced genetic variability and elevated levels of inbreeding when juxtaposed against conspecific mainland (continental) populations (Frankham 1997; Bidlack and Cook 2001). Cold saltwater and strong currents are effective barriers to gene flow in many terrestrial vertebrate species (Williamson 1981; Baker et al. 1990), so populations on archipelagos often are differentiated from mainland populations. Although the relationship between genetic variability and extinction is complex (Aguilar et al. 2004), reduced variability and low levels of immigration may increase the probability of extirpation for isolated populations (MacArthur and Wilson 1967; Frankham 2005). Although island populations typically contain low levels of genetic variation (Frankham 1997), they may be essential to long-term persistence of the species and to the origin of new lineages because of their tendency to accumulate novel variation (Wilson et al. 2009). Isolation, therefore, increases the potential for divergence, while also increasing the risk of extinction (MacArthur and Wilson 1967), and these 2 factors have made island populations the focus of research and conservation efforts worldwide (Losos and Ricklefs 2009). Understanding how historical biogeography influences the contemporary distribution of genetic diversity within natural populations is a critical first step towards effective management of island biodiversity.

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Early scientific surveys along the northern North Pacific Coast (NPC) of North America identified high numbers of endemic or regionally distinct organisms (Swarth 1936; Hall 1944) and subsequent investigations of the region's fauna (e.g., Reid et al. 1999), combined with advances in molecular techniques, uncovered a suite of genetically distinct populations and species of conservation concern (Cook et al. 2006). Recent investigations are providing more detailed views of these endemics (Topp and Winker 2008; Sawyer and Cook 2016) and revealing new areas supporting high genetic diversity along the NPC (Latch et al. 2008). This region includes 2 large, north-south oriented archipelagos: the Alexander Archipelago in southeastern Alaska (> 2,000 islands) and to the south, the Haida Gwaii Archipelago along coastal British Columbia (> 150 islands). These archipelagos, and the nearby narrow mainland bounded to the east by the Coast Range, support the largest remaining tract of old-growth temperate rainforest in the world (Alaback 1988). Potential Late Quaternary glacial refugia in this region have been proposed (Byun et al. 1997) and have been explored from geological (Carrara et al. 2003, 2007), paleontological (Heaton and Grady 2003; Ramsey et al. 2004), anthropological (Carlson and Baichtal 2015), and phylogeographical perspectives (Cook et al. 2006; Dawson et al. 2014).

Pacific martens (*Martes caurina* Merriam 1890) are distributed along the northern NPC of North America with disjunct populations found southward into California and the southern Rocky Mountains (Fig. 1). More widespread American pine martens (*M. americana* Turton 1806) have a parapatric northern distribution across North America. Originally described as distinct species, *M. caurina* was later subsumed within *M. americana* on the basis of a hypothesized zone of introgression in Montana (Wright 1953). We use the original 2-species taxonomic classification of Merriam (1890), which has been supported by molecular (Carr and Hicks 1997; Dawson and Cook 2012) and parasitological (Koehler et al. 2009; Hoberg et al. 2012) evidence. Reassessment of genetic diversity and conservation status of these 2 species provides a foundation for effective wildlife management prescriptions within the archipelagos and elsewhere. Further, 3 populations (Kuiu Island, northern and southern Montana) include areas of sympatry for

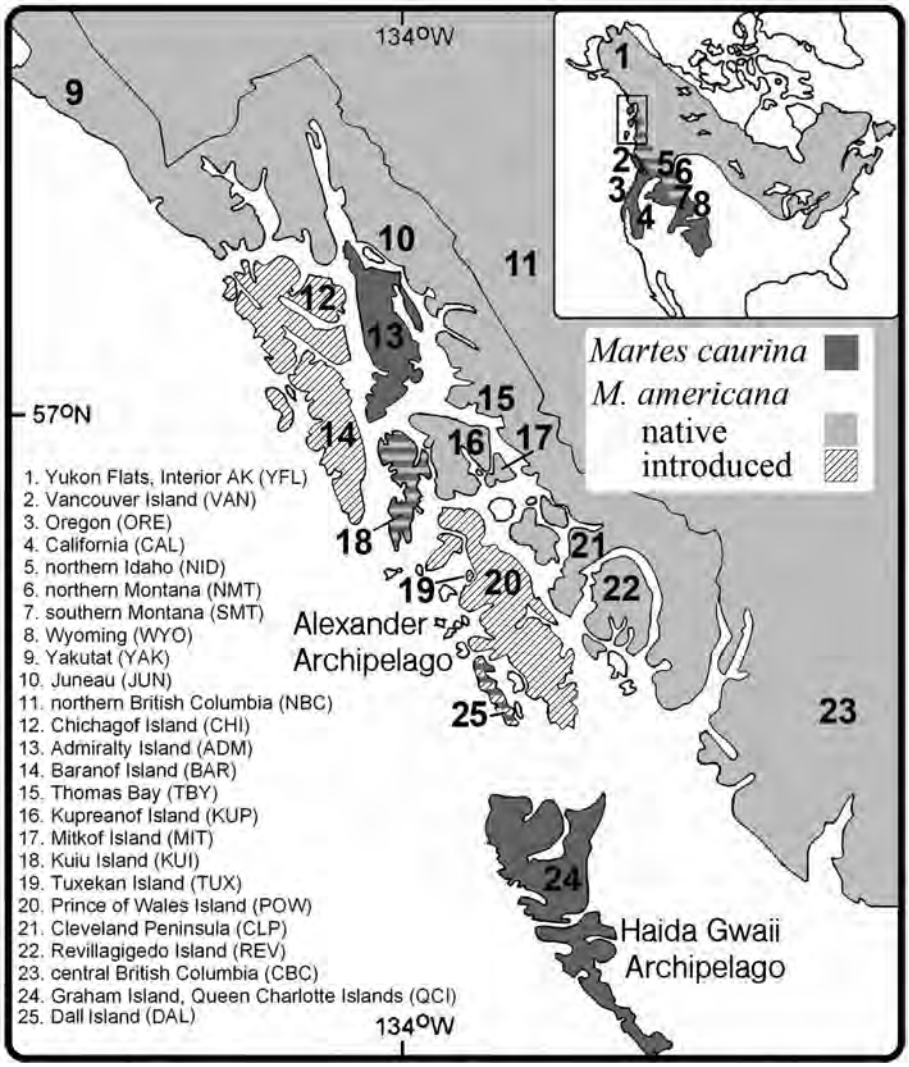

Fig. 1.—Map of localities for all *Martes americana* (light gray) and *M. caurina* (dark gray) populations. Bold hatched lines indicate hybrid zones between the 2 species, light hatching indicates islands that received *M. americana* introductions.

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the 2 species and represent documented hybrid zones (morphological intergradation—Wright 1953; intermixed microsatellite alleles—Small et al. 2003).

A series of wildlife translocations conducted from 1930 to 1950 by the Alaska Game Commission from the coastal mainland and elsewhere into multiple islands of the Alexander Archipelago (Chichagof, Baranof, and Prince of Wales islands) further complicates our understanding of the distribution of these species (Paul 2009). We deliberately include introduced populations in our analysis to facilitate exploration of the dynamics of natural versus human-mediated colonization, an aspect unaddressed by previous phylogenetic studies in this system. We expand upon previous genetic studies of North American martens (Stone and Cook 2000; Stone et al. 2002; Small et al. 2003) in an effort to refine contact zone geography and reconstruct historical biogeographic patterns relevant to conservation. We add key locations, including new insular (Dall Island, Tuxekan Island) and continental (southern Montana, northern Idaho, California) populations, and an additional mitochondrial gene (control region) to fully characterize variation in these species. By placing our new analyses into the context of past studies (Stone et al. 2002; Small et al. 2003), we evaluate historical demographics relevant to conservation of insular and continental populations. We hypothesize that *M. caurina* exhibits a characteristic signature of long-term persistence and diversification in island populations (Stone et al. 2002). Persistence would be consistent with extensive geographic structure uncovered in an associated endoparasitic nematode across island populations of the northern NPC (Koehler et al. 2009). In contrast, we expect *M. americana* will exhibit minimal geographic structure among populations in the Pacific Northwest as a result of recent expansion into the region (Stone et al. 2002). In this paper, we seek to: 1) describe diversity within and among insular populations of 2 species with distinct biogeographic histories; 2) compare genetic structure of both indigenous and introduced insular populations with mainland populations; 3) identify vulnerable island populations based on genetic variability; and 4) discuss these results within the context of future management considerations for this complex coastal island ecosystem.

Materials and Methods

Sampling.—Marten samples were obtained from furbearer trappers through cooperative efforts with state, federal, or provincial natural resource agencies for disposition in the University Alaska Museum of the North at the University of Alaska in Fairbanks, Alaska and Museum of Southwestern Biology at the University of New Mexico in Albuquerque, New Mexico. Other specimens were obtained from the Museum of Vertebrate Zoology at the University of California in Berkeley, California. Samples represent 268 individuals (Supplementary Data SD1) and 25 populations (Fig. 1; Table 1). Populations were defined based on their locations (e.g., islands). The use of vertebrate specimens conforms to American Society of Mammalogists guidelines (Sikes et al. 2016) and institutional requirements. Sampling effort focused on the western region of North America and most intensively on the northern NPC. Of the 6 "pure" *M. caurina* populations (Stone et al. 2002; Small et al. 2003), 3 are restricted to islands and 3 are on the mainland (Fig. 1). *Martes americana* is represented by 16 "pure" populations, including 8 island populations (2 naturally colonized recently, 3 documented deliberate human introductions within the last 80 years, and 1 probable introduction [Tuxekan], and 2 populations of unclear origin [Dall and Revillagigedo]; Table 1—Stone et al. 2002; Small et al. 2003; Paul 2009). The remaining 8 mainland populations are either coastal (west of the Coast Range) or continental (east of the Coast Range).

Laboratory procedures.—DNA was extracted following methods in Fleming and Cook (2002) and Slauson et al. (2008). PCR amplifications were in 50 µl volumes with the following reagents: 1.5 mM $MgCl_2$, 1× dNTPs, 1.0 µM of each primer, 0.05 µl of AmpliTaq polymerase, and 10× polymerase PCR buffer with approximately 1–100 µg of DNA. Control region PCR was completed with a PTC-0200 (MJ Research, Inc., Waltham, Massachusetts) thermocycler with the following conditions: 1 cycle of 94°C for 45 s, followed by 35 denaturation cycles at 94°C for 10 s, annealing at 45°C for 15 s, and an extension at 72°C for 45 s, followed by a final extension at 72°C for 3 min (Stone et al. 2002). Negative and positive controls were included in all experiments. PCR products were cleaned with a PEG (30%) cleanup procedure. The following primer pair was used to amplify 304 bp of the control region: TDKD (5′-CCT GAA GTA GGA ACC AGA TG—Kocher et al. 1993) and CTRL-L (5′-CAC YWT YAACWC CCA AAG CT—Bidlack and Cook 2001). Both forward and reverse strands were sequenced for each individual using either an ABI 377 or ABI 3100 (Wyoming sequences) automated sequencer. Big Dye 1.0 and 3.1 Terminators (Applied Biosystems, Foster City, California) were used in DNA sequencing. Dall Island sequences (n = 5) were generated using an Illumina MiSeq small genome analyzer.

Sequencher v4.6 (Gene Codes Corporation, Ann Arbor, Michigan) was used to navigate sequences, in both forward and reverse directions. Two programs, MacClade v4.0 (Maddison and Maddison 1992) and ClustalX v1.8 (Thompson et al. 1997), were used to align sequences and identify insertions and deletions (indels). We analyzed control region sequences separately and in a second analysis we concatenated control region sequences with cytochrome *b* sequences from Stone et al. (2002) to accommodate linked mitochondrial (mtDNA) inheritance. Genetic diversity and differentiation metrics generated from mtDNA data were compared to data generated from 12 nuclear microsatellite loci (Small et al. 2003). Microsatellite locus Mvis20 of Small et al. (2003) was not included in the current analysis, as Dawson (2008) found that the locus was X-linked. Sequences (n = 268) were deposited in GenBank (KX807723–KX807981).

Evolutionary relationships.—For 304 bp of the mtDNA control region (n = 268), we used jModelTest v2.1.7 (Guindon and Gascuel 2003; Darriba et al. 2012) to find the best model of evolution to fit the data. Akaike information criteria, Bayesian

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Table 1.—Summary statistics from nucleotide sequences of the mtDNA cytochrome *b* (Stone et al. 2002 and this study) and control region (this study) for *Martes americana* and *M. caurina* from northwestern North America, and fragment data from 12 nuclear microsatellites (Small et al. 2003). Bold F_{IS} values denote significant inbreeding (Bonferroni correction applied). BC = British Columbia; SE AK = Southeast Alaska.

Populations	Abbreviation	Cytochrome *b*			Control region			Nuclear microsatellites (12 loci)					
		n	Haplotypes	Haplotype richness	*n*	Haplotypes	Haplotype richness	*n*	Alleles	Allelic richness	H_E	H_O	F_{IS}
Martes americana													
Mainland													
Northern Idaho	NID				19	7	0.37						
Central British Columbia	CBC	6	2	0.33	10	6	0.60	17	5.46	4.13	0.70	0.61	0.12
Yukon Flats, interior AK	YFL	10	3	0.30	9	6	0.67	25	5.69	3.63	0.23	0.55	**0.13**
Thomas Bay, SE AK	TBY	5	3	0.60	10	5	0.50	20	4.23	3.34	0.62	0.50	**0.21**
Northern British Columbia	NBC	5	3	0.60	5	4	0.80	5	4.00	4.00	0.67	0.66	0.02
Juneau, SE AK	JUN	5	2	0.40	12	4	0.33	25	5.08	3.46	0.61	0.61	**0.17**
Cleveland Peninsula, SE AK	CLP	5	3	0.60	10	3	0.30	25	5.31	3.66	0.63	0.56	0.12
Yakutat, SE AK	YAK	5	1	0.20	10	3	0.30	22	4.00	2.91	0.54	0.47	0.13
Island, non-introduced													
Dall Island, SE AK[a]	DAL	6	2	0.33	5	2	0.40						
Mitkof Island, SE AK	MIT	5	2	0.40	10	4	0.40	25	4.69	3.39	0.59	0.48	**0.18**
Kupreanof Island, SE AK	KUP	2	1	0.50	11	3	0.27	25	4.45	3.10	0.56	0.49	**0.13**
Revillagigedo Island, SE AK[a]	REV	5	1	0.20	10	2	0.20	25	3.54	2.74	0.48	0.40	0.26
Island, introduced													
Chichagof Island, SE AK	CHI	15	3	0.20	14	6	0.43	25	4.23	3.24	0.57	0.50	0.11
Prince of Wales Island, SE AK	POW	10	2	0.20	11	3	0.30	25	4.38	3.29	0.62	0.47	**0.24**
Baranof Island, SE AK	BAR	10	1	0.10	11	1	0.09	26	3.54	2.46	0.45	0.36	**0.20**
Tuxekan Island, SE AK	TUX				4	1	0.25						
Hybrid *M. americana*/*caurina*													
Northern Montana	NMT	2	2	1.00	9/1	7	0.70	11	4.54	3.72	0.63	0.55	0.14
Southern Montana	SMT	2	1/1	1.00	9/1	7	0.70	11	5.31	4.20	0.68	0.48	**0.31**
Kuiu Island, SE AK	KUI	23	3/1	0.17	1/9	6	0.50	25	4.85	3.20	0.54	0.45	**0.18**
Martes caurina													
Mainland													
California	CAL				20	5	0.25						
Wyoming	WYO	5	1	0.20	5	3	0.60						
Oregon	ORE	6	1	0.17	15	1	0.07	16	3.31	2.73	0.50	0.53	-0.04
Island													
Vancouver Island, BC	VAN	2	1	0.50	14	3	0.21	23	2.92	2.08	0.31	0.29	0.07
Admiralty Island, SE AK	ADM	21	1	0.05	12	1	0.08	25	1.31	1.23	0.09	0.08	**0.09**
Queen Charlotte Islands, BC	QCI	5	1	0.20	9	1	0.10	11	2.00	1.87	0.30	0.28	0.05
Totals													
Martes americana		96			178			301					
Martes caurina		64			90			112					

[a]Populations of uncertain origin.

information criteria, and decision theory all selected the HKY (Hasegawa, Kishino, and Yano 1985) model of evolution (Yang et al. 1994). We included nucleotide sequence data from portions of mtDNA control region and cytochrome *b* from 1 European pine marten (*M. martes*; GenBank AF336969 and AF154975, respectively) and 1 sable (*M. zibellina*; GenBank AF336970 and JQ343004, respectively) as outgroups for phylogenetic reconstruction. Phylogenetic trees were constructed using MrBayes (Huelsenbeck and Ronquist 2001; Ronquist and Huelsenbeck 2003) and RAxML (Stamatakis 2014). In this case, we constructed trees using control region and cytochrome *b* independently and then by concatenating the 2 loci. We ran HKY and GTR (general time reversible) reverse-jump models of evolution as a comparative framework. We completed 6 runs over 5,000,000 generations with a burn-in period of 1,250,000 (25%) at which point the log-likelihood values became stationary. Consensus phylogenies were visualized in FigTree v1.4.2 (http://tree.bio.ed.ac.uk/software/figtree/). For network construction, we employed the Median-Joining Network (MJN—Bandelt et al. 1999), a method that yields the best genealogies under strict parsimony conditions (Cassens et al. 2003). Networks are often employed for large datasets based on closely related species or for population-level studies (Posada and Crandall 2001) as they show the number of base-pair differences between each haplotype. We used the R{Pegas} package (Paradis 2010) to construct the shortest network separating *M. americana* from *M. caurina* based on control region variation. Arlequin v3.01 (Excoffier et al. 2005) was used to calculate haplotype (h) and nucleotide diversity (π) for each population.

Population structure.—All population-level analyses were carried out using the 304 bp segment of the mitochondrial control region and then compared across the cytochrome *b* and microsatellite data (Stone et al. 2002; Small et al. 2003) for

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corresponding specimens (n = 209). We conducted a hierarchical analysis of molecular variance (AMOVA) to examine population structure within and between species using Arlequin. Significance was assessed using nonparametric permutations, followed by recalculation of all statistics to form a null distribution ($\alpha < 0.05$—Excoffier et al. 1992). An AMOVA was conducted independently for both species to determine the relative partitioning of variance among individuals within populations and among island and mainland populations. Introduced populations (Table 1) were excluded from the AMOVAs to ensure that the tests summarized genetic variation within and among groups with no a priori assumptions (Araya-Anchetta et al. 2013). We generated pairwise F_{ST} statistics (Weir and Cockerham 1984) in Arlequin to assess interpopulation differentiation and compared the results to independent values generated from 12 microsatellite loci for a subset of the populations, as reported elsewhere (Small et al. 2003).

Population expansion.—We tested for signals of expansion in each population using DNAsp v5 (Librado and Rozas 2009) for mismatch analysis, Lamarc v2.1.10 (Kuhner 2006) for g-statistics, and Arlequin for Tajima's D and Fu's F_S ($\alpha < 0.05$—Fu 1997). All analyses were run with 10,000 iterations unless otherwise noted. The mismatch distribution test measures population expansion based on pairwise differences between haplotypes (Rogers and Harpending 1992) and assumes that the taxa are related, the differences are mutations, and the mutations occur at a constant rate through time. Fu's F statistics (F_S—Fu 1997) and Tajima's D (Tajima 1989) allow for additional, independent tests of expansion by examining deviations from neutrality. Fu's F_S uses pairwise differences and D compares theta ($\theta = 2N_f\mu$) values based on nucleotide site differences (Excoffier et al. 2005). Significantly negative values of F_S or D indicate an excess of low-frequency mutations over that expected under a standard model of neutrality for populations. Fu's F_S and Tajima's D are used widely in tests for signals of expansion and are more conservative and arguably have greater statistical power than other tests, such as mismatch distributions (Ramos-Onsins and Rozas 2002). Although most demographic analyses are prone to overestimation, especially in closely related populations, they can still provide a relative measure of growth or contraction for species at similar levels of divergence (Grant 2015).

We used g-statistics as an additional test for expansion in each population, excluding all invariable populations (Admiralty Island, Baranof Island, Tuxekan Island, Haida Gwaii, and Oregon). The g-statistic measures exponential population growth using estimates of θ ($2N_f\mu$). The g-statistic measures growth in populations and θ measures effective population size relative to mutation rate, $\theta = 2N_f\mu$, where μ is the mutation rate of the control region and N_f the female effective population size (Domingues et al. 2006) when based on mtDNA. Negative values of g indicate that a population is in decline, while positive values indicate population growth. In this analysis, θ indicates time to coalescence for genealogies of a specific population. When compared to unbiased estimates such as number of pairwise differences or nucleotide diversity, a larger θ reflects a longer time to coalescence across population genealogies given the observed population growth (Delport et al. 2007). The g-statistic has an upward bias (Kuhner et al. 1998), so we used 3 *SD*s as our test of significance (Lessa et al. 2003).

Lastly, we generated control region Extended Bayesian Skyline Plots (EBSPs; Supplementary Data SD2) for each putative species with admixed localities excluded. Analyses were run in Beast v2.4.3 (Bouckaert et al. 2014) with a chain length of 500 million generations, a strict clock, and sampling every 5,000 generations with a burn-in of 25% resulting in 10,000 final trees. General set up followed the online Beast2 EBSP tutorial (Heled and Vaughan: beast2.org/tutorials). Convergence (effective sample size > 300) was evaluated in Tracer v1.6 (Rambaut et al. 2014) and plots were visualized in R.

Results

Control region.—Control region sequences were variable in both *M. americana* and *M. caurina* populations with a total of 49 haplotypes identified with 38 variable sites, 33 substitutions (28 transitions, 6 transversions), and 6 indels. There were 7 sites with fixed differences between *M. americana* and *M. caurina* (Fig. 2). Across 178 *M. americana*, 29 distinct haplotypes (Table 2a) were identified, whereas 92 *M. caurina* had 20 haplotypes (Table 2b). Admiralty, Baranof, Tuxekan, and Haida Gwaii islands, and Oregon were invariable; these populations were excluded from population-level analyses but included in phylogenetic analyses. Sixteen of 29 (55%) *M. americana* haplotypes were unique to single populations,

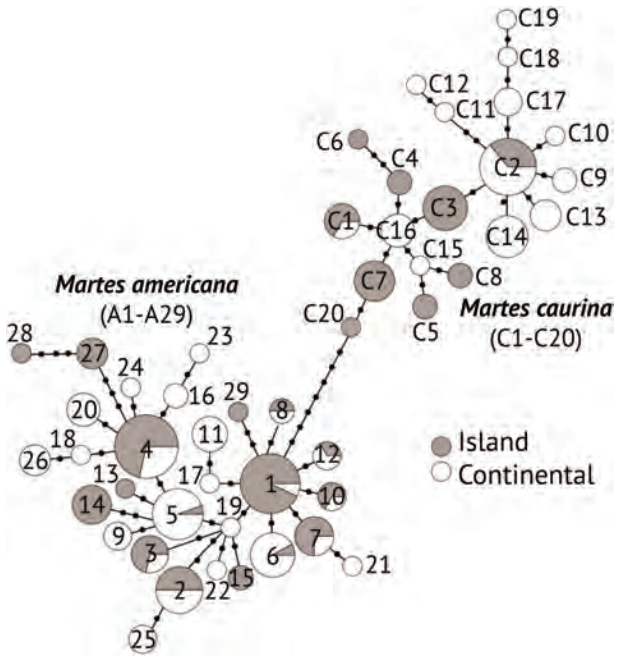

Fig. 2.—Haplotype network for *Martes americana* and *M. caurina* mitochondrial control region sequences. Circle size is proportional to the number of individuals with each haplotype; the number of mutations separating haplotypes is indicated by dots along the links. Pie colors indicate mainland haplotypes (white) and island haplotypes (gray).

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Table 2.—Control region haplotypes by locality for populations of a) *M. americana* and b) *M. caurina*. Bold populations are recognized hybrid populations.

a) *M. americana*

	Haplotype																												
Population	A1	A2	A3	A4	A5	A6	A7	A8	A9	A10	A11	A12	A13	A14	A15	A16	A17	A18	A19	A20	A21	A22	A23	A24	A25	A26	A27	A28	A29
BAR	11																												
CHI	4	2	5	1	1	1																							
DAL																											4	1	
KUI	3			3									1																
KUP	5			4			2																						
MIT	1	3		2			4																						
POW		1		8				1																					1
REV														8	2														
TUX				4																									
CBC	1				4											2	1	1	1										
NBC		1		1					2	1																			
NID		3		5			2					2								5	1	1							
NMT		1	2	2		2			1		1																		
SMT											1																		
YFL					2	2		1			2												1	1					
CLP		2		2	6																								
JUN	1				5	3																			3				
TBY	3	1		3						2		1																	
YAK						4					2															4			
Total	29	14	7	35	18	12	8	2	3	3	6	3	1	8	2	2	1	1	1	5	1	1	1	1	3	4	4	1	1

b) *M. caurina*

Population	C1	C2	C3	C4	C5	C6	C7	C8	C9	C10	C11	C12	C13	C14	C15	C16	C17	C18	C19	C20
ADM			12																	
KUI				2	2	1														
QCI							9													
VAN	4	8						2												
NMT																				1
SMT	2	2							2	1	1	1								
ORE		15																		
CAL													4	10	1	5				
WYO																	3	1	1	
Total	6	25	12	2	2	1	9	2	2	1	1	1	4	10	1	5	3	1	1	1

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and 17 of 20 (85%) *M. caurina* haplotypes were unique to single populations. Sequences in 6 populations exhibited transversions (1 *M. caurina*: Wyoming; *2 M. americana*: Dall Island and Prince of Wales Island; 3 hybrid populations: northern and southern Montana and Kuiu Island).

We calculated haplotype richness (Table 1) for each population by dividing the number of control region haplotypes within a population by the total number of haplotypes and correcting for sample size. Haplotype richness is an analogue to allelic richness (Kawamoto et al. 2008) and was higher for the control region compared to cytochrome *b* in 7 of the 14 *M. americana* populations that had both sequences. Three island populations of *M. americana* had equivalent richness across both loci, while 4 localities (1 island, 3 mainland) had lower haplotype richness for control region over cytochrome *b*. In contrast, only the Wyoming population of *M. caurina* showed greater haplotype richness for control region compared to cytochrome *b*. The remaining 4 *M. caurina* populations had elevated cytochrome *b* richness. Populations demonstrating the highest haplotype and nucleotide diversity occurred in recognized hybrid populations. Haplotype diversity was highest in southern and northern Montana, and nucleotide diversity was highest in Kuiu Island and northern Montana where haplotypes characteristic of both species were present (Table 3).

Phylogenetic relationships.—For the Bayesian consensus tree (Fig. 3), control region sequences were concatenated with cytochrome *b* sequences only for the 23 individuals with complete cytochrome *b* sequences (1,140 bp) described in Stone et al. (2002). Some localities are not represented because corresponding cytochrome *b* sequences were unavailable. The 50% majority rule consensus tree consistently separated *M. caurina* from *M. americana* populations.

Comparatively, cytochrome *b* analyses identified 9 mutations separating these 2 species. Network analyses also highlight differences in haplotype diversity within each group. Whereas common *M. americana* haplotypes are widely dispersed across both continental and island locations for this species (i.e., Haps A1, A3, A4, A5, A6; Fig. 2), *M. caurina* haplotypes tend to be more limited in geographic distribution (i.e., haplotypes C4–C8; Fig. 2). Approximately 32% of *M. caurina* haplotypes are endemic to a single island, whereas only 10% of *M. americana* haplotypes are endemic to an island.

Population structure.—Overall, AMOVA values across all *M. americana* and all *M. caurina* populations (Table 4) indicate greater percent haplotypic variation within populations compared to among populations; however, there are differences between species when island and mainland populations are compared. Across insular *M. caurina* populations, there is significantly greater genetic variation among populations and relatively little variation within populations (α = 0.05 level; Table 4). In contrast, island *M. americana* populations have haplotypic variation more evenly partitioned among and within populations. For mainland populations of *M. caurina*, there is nearly twice as much variation partitioned among individuals within populations as there is among populations and this pattern is amplified in mainland *M. americana* which harbor substantially more haplotypic variation within populations than among populations.

Across all *M. americana* populations except Dall and Revillagigedo islands, pairwise F_{ST} values were rarely higher than 0.2 (Table 5) and lower than all pairwise F_{ST} values among *M. caurina* populations. Pairwise F_{ST} values among *M. americana* populations were within the range of pairwise

Table 3.—Results of demographic tests for 2 species of *Martes* from northwestern North America. *SD*s are provided for nucleotide diversity (π) and haplotype diversity (h). Tests of expansion (Fu's F_S, Tajima's *D*, *g*-statistics) are reported for all variable populations of *M. americana* and *M. caurina*. All *g*-statistics are nonsignificant (within 3 *SD*s of zero) and their associated θ ($2N_f\mu$) values are reported.

Species	Population	π	h	*D*	F_S	*g*	θ
M. americana							
	NID	0.007 ± 0.005	0.854 ± 0.047	0.95	−0.99	161.56	0.0106
	CBC	0.007 ± 0.005	0.844 ± 0.103	0.53	−1.58	675.05	0.0293
	YFL	0.008 ± 0.005	0.911 ± 0.062	1.29	−1.72	383.86	0.0138
	TBY	0.007 ± 0.005	0.844 ± 0.080	1.47	−0.65	253.08	0.0093
	NBC	0.009 ± 0.006	0.833 ± 0.222	1.09	0.01	380.53	0.0105
	JUN	0.008 ± 0.005	0.758 ± 0.081	0.60	1.31	5.26	0.0051
	CLP	0.003 ± 0.003	0.622 ± 0.138	0.83	0.46	236.30	0.0029
	YAK	0.004 ± 0.003	0.711 ± 0.086	1.64	0.6	100.23	0.0026
	DAL	0.004 ± 0.004	0.400 ± 0.237	−1.05	1.69	−37.25	0.0038
	MIT	0.007 ± 0.005	0.778 ± 0.091	1.59	0.59	58.00	0.0049
	REV	0.005 ± 0.004	0.356 ± 0.159	0.02	3.03	−173.92	0.0026
	KUP	0.006 ± 0.004	0.691 ± 0.086	1.31	1.96	−79.15	0.0061
	CHI	0.006 ± 0.004	0.833 ± 0.071	1.03	−1.51	512.17	0.0081
	POW	0.006 ± 0.004	0.491 ± 0.175	−0.89	0.36	43.32	0.0062
Hybrid populations							
	KUI	0.023 ± 0.013	0.879 ± 0.060	1.70	1.98	−0.26	0.0151
	NMT	0.012 ± 0.007	0.933 ± 0.062	−1.04	−1.61	11.49	0.0136
	SMT	0.006 ± 0.004	0.933 ± 0.062	−1.14	−1.53	42.94	0.0201
M. caurina							
	CAL	0.006 ± 0.004	0.679 ± 0.074	0.95	1.55	−21.29	0.0043
	VAN	0.003 ± 0.002	0.615 ± 0.102	0.70	0.38	161.32	0.0024
	WYO	0.003 ± 0.003	0.700 ± 0.218	0.24	−0.48	911.12	0.0092

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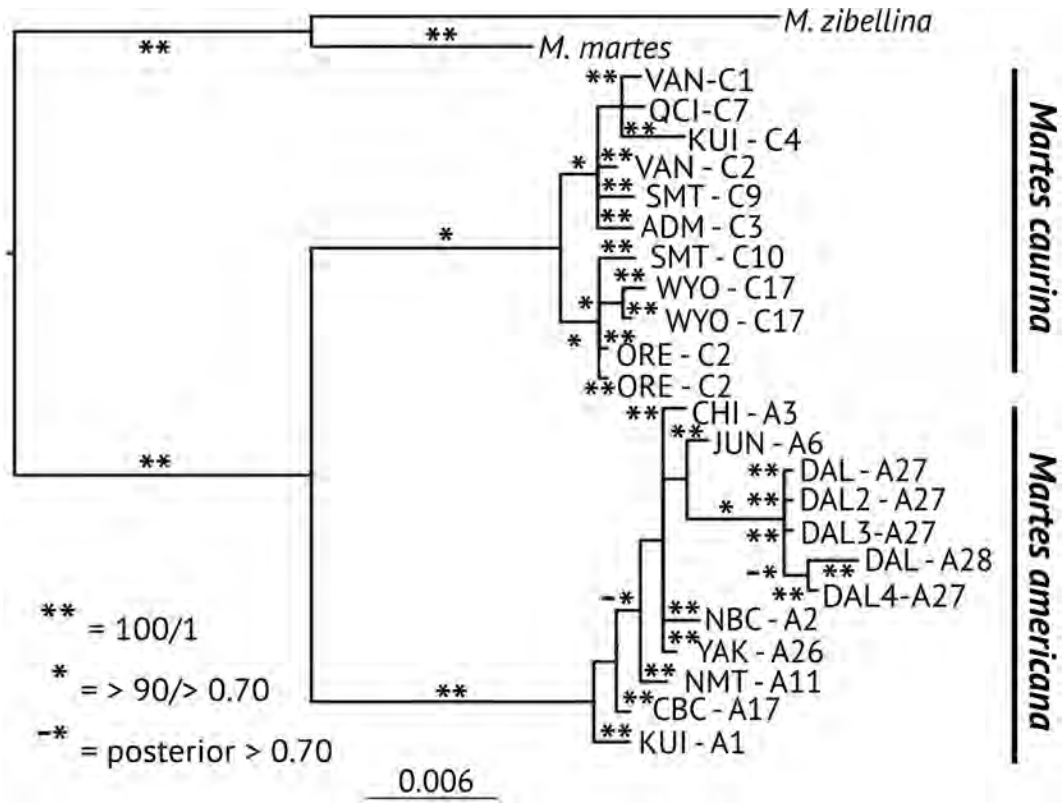

Fig. 3.—Phylogeny of *Martes americana* and *M. caurina* based on maximum likelihood (ML) and Bayesian inference (rooted). Analyses are based on mtDNA from the control region (this study: 304 bp) and cytochrome *b* (Stone et al. 2002) using an HKY + I evolutionary model. ML bootstrap and Bayesian posterior probabilities are displayed at each node (bootstrap support value/posterior probability).

F_{ST} values measured in other mammal species with moderate gene flow between populations (Hellborg et al. 2002). The highest pairwise F_{ST} values between *M. americana* populations are between Dall and Revillagigedo Islands when compared to other populations. These 2 islands were also the only *M. americana* populations that were significantly different from all other *M. americana* populations. Dall Island, in particular, harbors 2 endemic haplotypes (Fig. 2: A27, A28), separated from the nearest *M. americana* haplotype by multiple mutations.

Although average pairwise F_{ST} values for the mtDNA control region for *M. caurina* was more than double that of *M. americana* populations (0.69 and 0.33, respectively), these values were not significantly different (2 tailed students *t*-test $\alpha > 0.01$; Table 5). Microsatellite F_{ST} values (0.54 and 0.13, $\alpha < 0.01$, for *M. caurina* and *M. americana*, respectively) were significantly different and support greater population structure and lower gene flow among populations of *M. caurina* compared to *M. americana*. The highest pairwise F_{ST} value, other than between the invariable populations (Admiralty, Baranof, and Haida Gwaii islands and Oregon), was between Wyoming and other *M. caurina* populations.

Population expansion.—Mismatch distributions (Supplementary Data SD3) show weak support for recent expansion in *M. americana* and *M. caurina* populations. However, mismatch distributions have low statistical power (Ramos-Onsins and Rozas 2002) and are sensitive to recently bottlenecked populations (Roques and Negro 2005). A recent bottleneck is a distinct possibility for several insular populations including Admiralty and Haida Gwaii islands (populations with extremely low variation), and Baranof, Prince of Wales, and Chichagof islands, which received recent introductions of few individuals. The mismatch distributions for Yukon Flats, Yakutat, Idaho, central British Columbia, Prince of Wales Island, and northern and southern Montana show strict, unimodal distributions, corroborating the hypothesis of recent (Holocene) expansion (Rogers and Harpending 1992). We can reject a model of demographic expansion (sum of squared deviation [SSD]; $\alpha < 0.05$) for central British Columbia, Wyoming, Yakutat, Northern Idaho, and Revillagigedo Island populations, suggesting demographic stability for these localities. Significantly bimodal or "ragged" distributions ($\alpha < 0.05$) further support demographic equilibrium of central British Columbia and Yakutat populations.

For each population, we used Fu's *F* statistic (F_S), Tajima's *D* (*D*), the *g* parameter (*g*), and theta (θ; Fluctuate—Kuhner et al. 1998) as tests of demographic expansion for the mtDNA (Table 3). Fu's F_S showed significant signals of expansion averaged over all populations for both *M. caurina* (−7.98; $\alpha < 0.005$) and *M. americana* (−15.55; $\alpha < 0.001$), but no single population was significant. Tajima's *D* was not significant for any population of either species. All populations were nonsignificant for recent demographic expansion (e.g., zero was within the 95% confidence intervals or within 3 *SD*s [values not shown] for *g*-statistics for all populations). EBSPs were

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Table 4.—Analysis of molecular variance (AMOVA) results reported for populations of *Martes americana* and *M. caurina* in this study. We compared across each species, as well as each species group of island or mainland populations. Overall F_{ST} values for each group are reported in the last column.

AMOVA	Source of variation	Sum of squares	Variance components	% Haplotype variation	Overall F_{ST}
All *M. americana*	Among	18.10	0.13	27.88	0.12
	Within	81.09	0.95	72.12	
All *M. caurina*	Among	8.24	0.31	33.58	0.34
	Within	21.71	0.60	66.42	
All *M. caurina* islands	Among	28.86	1.20	88.51	0.89
	Within	5.14	0.16	11.49	
All *M. caurina* mainland	Among	8.45	0.32	37.27	0.37
	Within	19.35	0.54	62.73	
All *M. americana* islands	Among	15.33	0.66	42.72	0.43
	Within	24.67	0.88	57.28	
All *M. americana* mainland	Among	18.10	0.13	11.76	0.12
	Within	81.09	0.95	88.24	

Table 5.—Weir and Cockerham's (1984) pairwise F_{ST} values for a) *M. americana* and b) *M. caurina* for 13 and 12 microsatellite loci[a], respectively (above diagonal; microsatellite data from Small et al. 2003) and mtDNA control region (below diagonal; this study). Bold values are significantly different from zero ($\alpha = 0.05$, Bonferroni correction applied to microsatellite data). n/a = not available.

a) *M. americana*

	BAR	CBC	CHI	CLP	DAL	JUN	KUP	MIT	NBC	NID	POW	REV	TBY	TUX	YAK	YFL
BAR		0.16	0.25	0.21	n/a	0.23	0.22	0.21	0.21	n/a	n/a	0.31	0.15	n/a	0.27	0.21
CBC	**0.54**		0.08	**0.05**	n/a	**0.03**	**0.11**	**0.06**	0.00	n/a	n/a	**0.15**	**0.07**	n/a	**0.11**	**0.03**
CHI	**0.46**	**0.14**		0.09	n/a	0.13	0.13	0.16	0.09	n/a	n/a	0.19	0.12	n/a	0.24	0.08
CLP	**0.81**	0.00	**0.27**		n/a	**0.05**	**0.12**	**0.08**	0.04	n/a	n/a	**0.12**	**0.09**	n/a	**0.17**	**0.07**
DAL	**0.96**	**0.73**	**0.78**	**0.83**		n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
JUN	**0.49**	0.05	0.08	0.11	**0.73**		**0.15**	**0.12**	0.03	n/a	n/a	**0.19**	**0.12**	n/a	**0.14**	0.02
KUP	**0.27**	0.11	0.08	**0.26**	**0.77**	**0.15**		**0.06**	**0.06**	n/a	n/a	**0.24**	**0.14**	n/a	**0.19**	**0.10**
MIT	**0.39**	**0.17**	0.07	**0.31**	**0.77**	**0.16**	−0.04		0.01	n/a	n/a	**0.27**	**0.10**	n/a	**0.21**	**0.07**
NBC	**0.82**	−0.07	**0.22**	0.06	**0.77**	0.10	0.23	0.24		n/a	n/a	**0.21**	0.05	n/a	**0.08**	−0.01
NID	**0.50**	0.08	**0.14**	0.08	**0.71**	**0.16**	0.10	0.09	0.07		n/a	n/a	n/a	n/a	n/a	n/a
POW	**0.70**	**0.14**	**0.21**	**0.14**	**0.74**	**0.22**	**0.24**	**0.24**	0.10	0.03		n/a	n/a	n/a	n/a	n/a
REV	**0.81**	**0.43**	**0.56**	**0.54**	**0.84**	**0.45**	**0.54**	**0.56**	**0.50**	**0.42**	**0.57**		0.20	n/a	0.27	**0.21**
TBY	**0.32**	**0.08**	**0.07**	**0.23**	**0.76**	0.13	−0.06	0.03	0.19	0.10	**0.20**	**0.53**		n/a	0.21	**0.12**
TUX	**1.00**	0.20	**0.39**	**0.30**	**0.87**	**0.30**	0.41	**0.41**	0.27	0.07	−0.06	**0.67**	0.37		n/a	n/a
YAK	**0.74**	**0.26**	**0.24**	**0.48**	**0.85**	**0.25**	**0.35**	**0.30**	0.22	**0.33**	**0.44**	**0.65**	**0.34**	**0.66**		**0.11**
YFL	**0.44**	−0.03	0.01	0.09	**0.75**	0.03	0.01	0.04	−0.02	0.06	**0.13**	**0.48**	−0.01	**0.27**	0.14	

b) *M. caurina*

	CAL	ADM	ORE	QCI	VAN	WYO
CAL		n/a	n/a	n/a	n/a	n/a
ADM	**0.70**		**0.58**	**0.75**	**0.65**	n/a
ORE	**0.27**	**1.00**		**0.39**	**0.35**	n/a
QCI	**0.58**	**1.00**	**1.00**		**0.51**	n/a
VAN	**0.18**	**0.88**	**0.31**	**0.73**		n/a
WYO	**0.46**	**0.94**	**0.83**	**0.90**	**0.61**	

[a]Microsatellite locus Mvis20 originally included in Small et al. (2003) was found to be X-linked (Dawson 2008) and not included in the current analysis for both species. MA15 was monomorphic in *M. caurina* and removed from F_{ST} analyses of *M. caurina*.

uninformative (Supplementary Data SD2), suggesting either an extreme historic bottleneck event in both populations that effectively prevented interpretation of N_f or that additional loci are required.

Discussion

Within the archipelagos of North America's northern NPC, contemporary genetic diversity and patterns of endemism result from complex geologic processes, natural fragmentation of the temperate rainforests, long-term population isolation, and more recently, human-mediated disturbances (Murie 1959; Bailey 1993; Paul 2009; Cook and MacDonald 2013). Thus, parsing signatures of demographic change in martens potentially illustrates the impact of both evolutionary processes and anthropogenic manipulation on island diversity. As a federally designated Management Indicator Species, martens function as a proxy for forest health (Simon 1980; Hargis and McCullough 1984;

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Flynn and Schumacher 1997). Because negative responses to insularity are evident in specialized, higher trophic taxa (Holt et al. 1999; Krauss et al. 2003), these mesocarnivores have an increased risk of extirpation. Taxonomy, biogeographic history, introductions, pathogens, hybridization, and the consequences of isolation on islands are all critical factors to consider in management, particularly within the context of accelerating environmental change (Ruesink et al. 1995; Pyšek and Richardson 2010; Malaney and Cook 2013; Robertson et al. 2014).

History highlights diversity.—Understanding Pleistocene refugial distributions and subsequent colonization histories enables rigorous interpretation of contemporary genetic diversity (Waltari et al. 2007). Along the NPC, significant variability between island and continental sister species or lineages reflects the consequences of historical processes on contemporary species assemblages (e.g., Dawson et al. 2014; Sawyer et al. 2017). Following the Last Glacial Maximum (LGM), *M. americana* tracked boreal forest expansion northwestward from a refugium in eastern North America (Graham and Graham 1994). This species arrived recently (late Holocene) along the NPC (Fig. 1) via river corridors through the Coast Range (Stone et al. 2002) and subsequently colonized or was translocated to several islands. In contrast, *M. caurina* was thought to be isolated in 1 or more southern LGM refugia along the west coast, spreading northward along the Pacific Coast (early Holocene—Stone et al. 2002) or eastward to the southern Rocky Mountains (Graham and Graham 1994). Haplotype networks (Fig. 2) and phylogenetic reconstructions (Fig. 3) illustrate the historical divergence within each species and each population's contribution to geographic structure and overall genetic diversity. Unique haplotypes and elevated differentiation (F_{ST}) on Revillagigedo and Dall islands suggest that geographic proximity does not necessarily translate to higher genetic similarity (Fig. 2; Tables 2a and 4). This curious result, also seen in red squirrels (*Tamiasciurus hudsonicus*) on Revillagigedo Island (Hope et al. 2016), requires additional investigation in other forest-associated species. The history of the Dall Island population, which plausibly received migrants from the neighboring introduced population on Prince of Wales Island, also needs more detailed sampling and assessment. Whether the 2 endemic haplotypes, separated from other *M. americana* haplotypes by multiple mutations, reflect earlier colonization event of martens on this remote western island is unclear.

Overall low F_{ST} values within *M. americana* suggests less population structure compared to *M. caurina* populations. The degree of divergence among private mtDNA haplotypes in Southeast Alaskan *M. americana* populations and other *M. americana* haplotypes is less than that between the private insular haplotypes among *M. caurina* populations (e.g., Admiralty, Haida Gwaii, Kuiu islands). Greater divergence among insular *M. caurina* populations reflects longer persistence and isolation in the region and possibly smaller population sizes, despite apparently viable densities for Kuiu and Admiralty islands (R. Flynn, Alaska Department of Fish and Game, pers. comm.). In addition, several mitochondrial and microsatellite alleles are shared widely across *M. americana* populations. One widespread *M. americana* haplotype (A4) is found from northern Montana to Kuiu Island. These sites, largely covered by ice during the LGM, were recently colonized by *M. americana*. *Martes caurina* on average has more private alleles (both control region and microsatellite) per population (2.33 and 0.78, respectively) compared to *M. americana* (1.67 and 0.69, respectively). High within-population diversity (and heterozygosity) and fewer private alleles in *M. americana* suggest higher historic or contemporary connectivity among populations, consistent with recent (late Holocene) expansion across North America. This general pattern is mirrored in other forest-associated organisms that colonized deglaciated areas at the end of the Pleistocene (Lessa et al. 2003; Hope et al. 2012; Chavez et al. 2014; Kerhoulas et al. 2015).

Many of the patterns we document are also found in genetic signatures of some marten parasites. Genetic structure in *Soboliphyme baturini*, a parasitic nematode (Koehler et al. 2009) of northern mustelids, mirrors host patterns, including long-term separation of *M. caurina* nematodes endemic to Admiralty Island and Haida Gwaii, recent westward expansion of *M. americana* into Southeast Alaska with subsequent colonization by the nematode, and also multiple, independent anthropogenic transplants of *M. americana* to Chichagof Island (Koehler et al. 2009; Hoberg et al. 2012).

Introduced populations.—Species introductions can facilitate mixing of previously distinct populations of a single species (Peacock et al. 2009) or may introduce novel parasites or invasive species. The Alexander Archipelago has experienced a series of introductions ranging from amphibians (Pauly et al. 2008) to ungulates (Cook et al. 2006; MacDonald and Cook 2007), yet the consequences are poorly understood. Because martens are a commercially important component of the furbearer industry (MacDonald and Cook 2007; Alaska Department of Fish and Game 2010), the Alaska Game Commission established "new" populations or supplemented existing populations during the 1930–1950s on Baranof, Chichagof, and Prince of Wales islands (Elkins and Nelson 1954; Burris and McKnight 1973; Alaska Department of Fish and Game 2010) from several mainland *M. americana* sources. Molecular signatures largely corresponded with the written records of introductions (Stone et al. 2002; Small et al. 2003), although the possibility of prior occupation by *M. caurina* on islands such as Prince of Wales (Pauli et al. 2015) should be more rigorously explored using genome-level analyses aimed at detecting admixture.

Some introduced *M. americana* populations exhibit relatively high genetic diversity (Prince of Wales, Chichagof islands) while others have lower diversity (Baranof Island) when compared to naturally colonized islands (Table 1). Introductions from multiple source populations, as recorded for martens on Prince of Wales and Chichagof islands, may lessen the impact of founder effects (i.e., loss of variation) that typify many introduced populations (Dlugosch and Parker 2007). Chichagof Island martens are derived from populations from Baranof Island, Revillagigedo Island, the Stikine River area, Wrangell Island, Mitkof Island, and a site near Anchorage, Alaska (Elkins and Nelson 1954; Burris and McKnight 1973). Baranof

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and Prince of Wales islands martens were from coastal mainland populations taken near Behm Canal and Thomas Bay. The different levels of variation in these translocated populations may reflect severe founder events, subsequent bottlenecks, or simply differing histories of colonization and introduction (Nei et al. 1975). Other island populations of *M. americana* in Southeast Alaska (MacDonald and Cook 2007) may have originated through colonization from nearby island source populations (e.g., Dall from Prince of Wales). Future investigations could monitor the effects of colonization on the genetic structure of these introduced populations if specimen archives were augmented annually. Because these islands function as independent tests of the impact of translocations, they have the potential to provide key insight into changing conditions.

Additional layers of genetic complexity in North American martens include 2 natural hybrid zones previously hypothesized in Montana and on Kuiu Island (Small et al. 2003). Both hybrid zones are characterized by admixed microsatellite alleles and mtDNA haplotypes characteristic of both species (Table 1, 2), producing higher haplotype or nucleotide diversity in hybrid populations relative to parental populations (Hewitt 2004; Swenson and Howard 2005). Of the 11 control region sequences from Kuiu Island, 2 haplotypes are shared throughout the range of *M. americana* including island and continental populations. Four new haplotypes were identified on Kuiu, among which 1 is *americana*-like (A13) and 3 are *caurina*-like (C4, C5, C6). Elevated and novel haplotypic diversity in this contact zone is consistent with patterns in other hybrid zones (e.g., Bradley et al. 1993). Hybridization can lead to decreased hybrid fitness (Muhlfeld et al. 2009), extinction of parental species (Rhymer and Simberloff 1996), prevention of adaptation (Eroukhmanoff et al. 2013), an increase in invasive success (Blumler 2003; Rieseberg et al. 2007), and outbreeding depression due to the disruption of co-adapted gene complexes (Shields 1987). Hybridization also may enhance the fitness of colonizing lineages via the capture of local adaptation (Racimo et al. 2015) and may promulgate speciation (Rheindt and Edwards 2011).

Conservation implications for insular populations.— Populations on islands, especially small islands, often contain less genetic variation than mainland populations (Bidlack and Cook 2001; Hayaishi and Kawamoto 2006). Decreased variation can lower fitness and evolutionary potential, ultimately leading to extinction in extreme cases (e.g., an extinction vortex—Gilpin and Soulé 1986; Newman and Pilson 1997; Frankham 1998, 2005; Alsos et al. 2012).

Martens provide a case study in conservation biology at multiple temporal and spatial scales (Cook et al. 2006; Dawson et al. 2007). Although they are tightly associated with high-volume old-growth forests (Potvin et al. 1999; Flynn and Schumacher 2016), federal timber harvest standards in the Tongass National Forest were adopted (United States Department of Agriculture, Forest Service, Alaska Region 1997; United States Department of Agriculture, Forest Service 2016) without a clear understanding of the complexity of insular marten populations and without the knowledge of 2 species in the region. This lack of clarity resulted in uneven application of timber harvest rules (Cook et al. 2006; United States Department of Agriculture, Forest Service 2016). For example, Kuiu Island is currently scheduled for significant habitat modification in old-growth reserves originally identified as central for martens' persistence (Stewart 2016). Kuiu Island supported an endemic population of *M. caurina*, but this population is now apparently being introgressed by *M. americana* (Small et al. 2003). Furthermore, the low genetic variability of the other *M. caurina* population potentially portends an increased risk of extinction (Frankham 2005; Whittaker and Fernández-Palacios 2007). Given their federal status related to old-growth forests and our growing understanding of how fundamental principles of island biology (insularity, distribution of genetic diversity, species assemblages—MacArthur and Wilson 1967; Whittaker and Fernandez-Palacios 2007) are impacting martens, we encourage the incorporation of new scientific information into management prescriptions for this vast and dynamic archipelago.

We note that declining populations of the Alexander Archipelago wolf (*Canis lupus ligoni*) did not figure into recent policy decisions (United States Fish and Wildlife Service 2016), despite documented genetic discreteness of populations at island and regional scales (Weckworth et al. 2005, 2015; Muñoz-Fuentes et al. 2009, 2010; Stronen et al. 2014; Cronin et al. 2015; Fredrickson et al. 2015). The Alexander Archipelago continues to be identified as a center of endemism in the northern latitudes due to its distinct biological diversity (Smith 2016). Unfortunately, the maintenance of endemism is only tangentially incorporated into contemporary forest conservation practices.

Wildlife management in the region may benefit from an island-by-island approach (MacDonald and Cook 2007). Understanding the historical framework that contributed to the evolutionary distinction of insular populations, in particular, is critical to the maintenance of contemporary biodiversity. Only within the last several decades have modern molecular tools demonstrated the hidden diversity of this coastal biome, long suspected by early 20th century naturalists (e.g. Swarth 1911, 1936). Still, significant human-induced change has already impacted islands (Cook and MacDonald 2013). For *M. caurina*, control region haplotypes on 3 NPC islands represent over one-third of documented mtDNA diversity (Table 1). Other old-growth obligates sharing similar patterns of endemism and evolutionary histories (flying squirrels, *Glaucomys sabrinus*—Bidlack and Cook 2002; ermines, *Mustela erminea*— Fleming and Cook 2002; Alexander Archipelago wolves—Weckworth et al. 2015) may be impacted by decreased volume of old-growth forests. In a period of accelerating change, understanding the evolutionary history of the Alexander Archipelago's biota could provide resource managers with key perspectives to build effective management prescriptions (Smith 2016). This pair of species, in particular, represents the results of 1 of several intriguing evolutionary experiments that played out along the NPC producing unique variability and complex interactions. This natural experiment, however, has been impacted by a series of anthropogenic manipulations that deserve further scrutiny and debate.

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Supplementary Data

Supplementary data are available at *Journal of Mammalogy* online.

Supplementary Data SD1. Appendix of all samples used, listed by museum identification number (UAM numbers, unless otherwise noted), with haplotype assignment and frequencies (N). "A" haplotypes indicate those belonging to *M. americana* and "C" haplotypes, to *M. caurina*.

Supplementary Data SD2. Extended Bayesian Skyline Plots and associated histograms for *M. americana* and *M. caurina* control region samples, with hybrid localities removed. Plots show a relatively constant N_f through time for both species, however histogram results indicate there is not enough information in a single mitochondrial locus to accurately infer N_f through time and these distributions likely reflect the priors.

Supplementary Data SD3. Mismatch distribution α values ($\alpha < 0.05$*, $\alpha < 0.01$**, $\alpha < 0.001$***) for mitochondrial control region sequences from each sampled locality.

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USDA United States Department of Agriculture | Forest Service | Tongass National Forest Alaska Region | 648 Mission Street Ketchikan, AK 99901 907-225-3101

File Code: 1950
Date: December 8, 2016

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EARTHJUSTICE

Dear Reader,

I am pleased to announce the availability of the Tongass Land and Resource Management Plan Amendment (Forest Plan Amendment). The effective date of the Forest Plan Amendment is 30 days after publication of the notice of Forest Plan Amendment approval in the newspaper of record, the *Ketchikan Daily News*. We have worked hard to develop these final documents and have incorporated valuable input from you during opportunities for public participation. Thank you for your interest and involvement in this effort.

The Forest Plan Amendment amends the 2008 Tongass Land and Resource Management Plan (2008 Forest Plan) and describes resource management practices, levels of resource production and management, and the availability and suitability of lands for different kinds of resource management. The Forest Plan Amendment guides all natural resource management projects and activities and establishes management direction for the Tongass National Forest. The Forest Plan Amendment was developed using the current Planning Rule, issued in 2012, and embodies the provisions of the National Forest Management Act, the implementing regulations, and other guiding documents.

The Forest Plan Amendment was shaped by best available science, current laws, and public participation including Ketchikan High School youth; participation of a cooperating agency (U.S. Fish and Wildlife Service); consultation with Alaska Native tribes and Alaska Native Corporations; advice and recommendations from the Tongass Advisory Committee, a Federal Advisory Committee established by the U. S. Department of Agriculture; and significant public contributions from 12 open house meetings, nine subsistence hearings, and the receipt of over 165,000 public comments. Guidance in the amended Forest Plan provides for:

- Transitioning to a more economically, socially and ecologically sustainable timber program on the Tongass; and
- Promoting more sustainable and diverse local economies by encouraging renewable energy production.

Highlights of my decision include:

1. Accelerating the transition (from 32 years under the 2008 Forest Plan to 16 years) from old-growth to young-growth forest management, and maintaining a viable timber industry.
2. Promoting more sustainable and diverse local economies by encouraging renewable energy production.
3. Maintaining the integrity of the Tongass Conservation Strategy to provide ecological conditions that maintain the diversity of plant and animal communities and support the persistence of most native species in the plan area.

4. Providing greater clarity regarding the intended long-term management of the Tongass 77 Watersheds and The Nature Conservancy/Audubon Conservation Priority Areas.

5. Committing to producing a report that documents the results of the ongoing inventory work under the Challenge Cost Share Agreement with State and Private Forestry and the State of Alaska, and to make this report available to industry stakeholders and the public for review.

6. Committing to posting the current draft five-year timber sale schedule on the public Tongass web page, by December 31, 2016, and updating and posting annually by December 31st of each year.

7. Committing to a consideration of the "no net loss" concept recommended by the Tongass Advisory Committee (Forest Plan, Appendix B) as the Forest monitors the effectiveness of the Forest Plan's plan components.

8. Making corrections (known as errata) to the June 2016 Tongass Land and Resource Management Plan final environmental impact statement (FEIS).

Highlights of the amended Forest Plan include:

1. Removing the Transportation and Utility System overlay Land Use Designation (LUD) as well as all associated direction (i.e., "window" and "avoidance area") in the management prescriptions in Chapter 3.

2. Including a new Chapter (Chapter 5) that includes plan content that was developed under the 2012 Planning Rule (36 Code of Federal Regulations [CFR] 219) that provides applicable plan direction related to Young Growth, Renewable Energy, and Transportation Systems Corridors and other plan direction that applies to the plan area (forest-wide) or to specific LUDs.

3. Adding management approaches to Young Growth, Renewable Energy, and Transportation Systems Corridors direction, and forest-wide direction in Chapter 5.

Forest Plan Amendment documents are available on the Tongass National Forest Plan Amendment Web site at: http://www.fs.usda.gov/goto/R10/Tongass/PlanAmend. These documents are also available for viewing at the Forest Supervisor's Office in Ketchikan, and at Tongass National Forest Ranger District Offices.

For more information, contact Sue Jennings, Forest Planner at 907-772-5864 or sjennings@fs.fed.us.

Again, I sincerely thank you for your continued involvement in the Forest Plan Amendment. I look forward to working with you as we implement the amended Forest Plan in the coming years.

Sincerely,

M. EARL STEWART
Forest Supervisor, Tongass NF

THE TONGASS RAINFOREST AS ALASKA'S FIRST LINE OF CLIMATE CHANGE DEFENSE AND IMPORTANCE TO THE PARIS CLIMATE CHANGE AGREEMENTS

Dominick A. DellaSala, Ph.D.
Chief Scientist, Geos Institute (Dominick@geosinstitute.org)

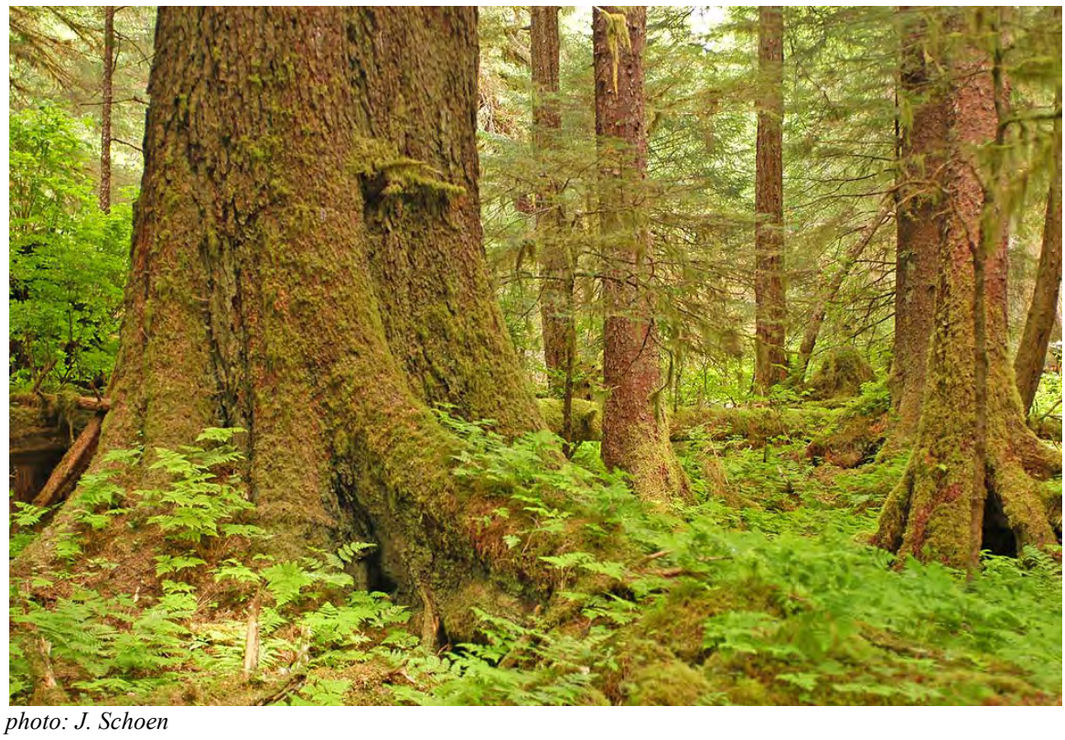


photo: J. Schoen

Executive Summary: the Tongass is a global champion in sequestering (absorbing) atmospheric carbon and storing it long-term in its ancient trees, productive soils, and dense rainforest foliage. Because it is one of the world's last relatively intact temperate rainforests, and it has a maritime climate, the Tongass is Alaska's first line of climate change defense and a climate refuge for its world-class salmon and wildlife populations. Logging of the Tongass rainforest produces greenhouse gas emissions that damages the region's contribution to a safe climate. Recognizing the critical need to reduce greenhouse gas emissions to keep global warming temperatures below a dangerous 2º C (~4º F) anticipated increase, a climate change agreement was reached in Paris by 195 members of the Conference of Parties (COP 21 also known as the 2015 Paris Climate Conference), including the USA. Articles of the agreement called for forests to be managed as a global "sink" for carbon. Therefore, protecting carbon sinks and reducing forestry emissions are pivotal steps to ensure a safe climate for Alaskans and for future generations.

Given the global importance of the Tongass as a carbon sink, we wanted to: (1) determine if the Tongass Draft Forest Plan Amendment (preferred alternative) was generally consistent with the Paris articles regarding managing forests as a carbon sink; (2) consistent with the Obama Administration's policies on climate change; and (3) whether the timeline for the proposed transition out of old-growth logging was consistent with efforts to end global deforestation under global forest and climate change agreements (e.g., COP 2, NY Forest Declaration). Thus, we estimated CO_2 emissions anticipated from logging old growth and young-growth forests as proposed by the Forest Service on the Tongass over the next 25 and 100 years and compared them to emissions under a conservation alternative designed to speed up the transition by relying mostly on soon-to-be-ready-for logging young growth as a replacement for old-growth logging.

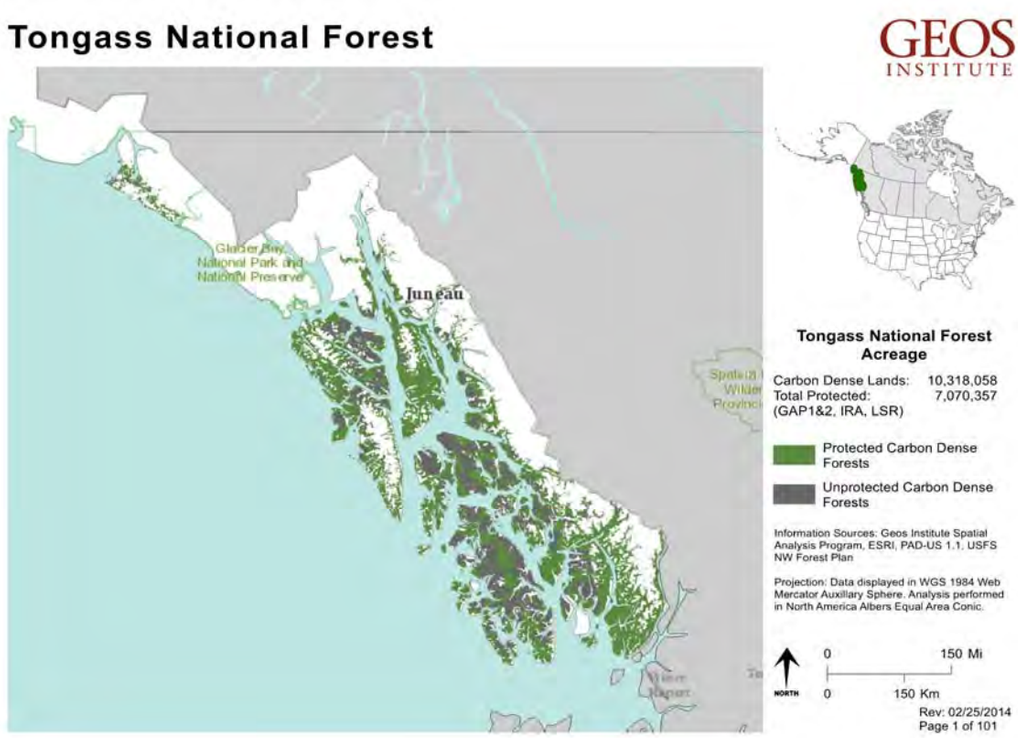
Key Findings (for 100 years):

- The agencies' preferred alternative would log 43,167 acres of old growth (OG) and 261,850 acres of young growth (YG) resulting in the equivalent emissions of ~4 million vehicles annually on Alaska roads for the next 100 years. These

estimates account for carbon stored in wood products and capture of carbon by forest regrowth.

- Logging emissions are ~175 times greater than the "reference point" for project emissions recommended by the White House's Council of Environmental Quality (CEQ). Emissions would result in a "social cost of carbon"conservatively estimated at >$100 million annually in global warming damages by the end of the century. Losses are ~10 times the projected timber revenues on the Tongass.
- A conservation alternative proposed by conservation groups (but dismissed by the Forest Service) would rely predominately on 76,000 acres of low controversy YG to support the transition with much less OG (9,125 acres over 100 years) to support specialty products. This alternative yields the equivalent emissions of over ~400,000 vehicles annually for 100 years, 16 times above CEQ emissions reference, but a tenth of the emissions from Forest Service proposed logging.
- The Tongass preferred alternative is out-of-step with efforts by the global community to reduce emissions. The conservation alternative better complies with CEQ guidelines, the Paris climate agreement, and efforts to reduce climate damages from CO_2 pollution.
- President Obama showed great interest in Alaska's already extensive climate impacts during his September 2015 Alaska visit to showcase his climate change initiatives prior to the Paris conference. Continued OG logging on the Tongass would further jeopardize Alaska's climate and is out of step with the President's climate change agenda.

NO OTHER NATIONAL FOREST STORES MORE CARBON THAN THE TONGASS (map shows concentration of Tongass forest-carbon stores)



THE TONGASS IS A NATIONAL CARBON SINK

Photo: D. DellaSala

"This is as good of a signpost as any when it comes to the impacts of climate change." President Obama during his September 2015 tour to Alaska glaciers.

Alaska's First Line of Climate Defense–Alaska is at the front lines of climate change, experiencing higher temperature increases than any other region in the nation along with increasing floods, coastal erosion and displacement of native villages, interior wildfires, die off of certain conifers, thawing of permafrost, and glacial melting (among other changes anticipated over the coming century)[1]. If Alaska is on the front lines, then the Tongass is Alaska's first line of climate defense.

At 16.8 million acres, the Tongass National Forest in southeast Alaska is the crown jewel of the national forest system. It is the nation's largest national forest and one of the world's last relatively intact temperate rainforests and thus it has global significance[2]. Its world-class salmon runs are the backbone of a thriving subsistence, commercial fishery, and recreation-based economy[3]. The Tongass is by far the nation's champion in storing carbon long-term[4] and, in doing so, represents a unique opportunity for the Obama Administration to lead by example regarding its global commitments to the Paris climate change agreements designed to keep global warming below the dangerous 2º C (~4º F) presumed tipping point. During COP 21, the parties recognized the importance of forests as global "sinks" for storing greenhouse gases and called for steps by the global

[1]Alaska Department of Environmental Conservation. 2010. Alaska's climate change strategy: addressing impacts in Alaska. http://www.climatechange.alaska.gov
[2]DellaSala, D.A. 2011. Temperate and boreal rainforests of the world: ecology and conservation. Island Press: Washington, D.C.
[3]Crane, L.K., and J.R. Mehrkens. 2013. Indigenous and commercial uses of the natural resources of the North Pacific Rainforest with a focus on Southeast Alaska and Haida Gwaii. Pp. 89-126. In G.H. Orians & J.W. Schoen (eds.). North Pacific Temperate Rainforests. University of Washington Press, Seattle.
[4]Leighty, W.W. et al. 2006. Effects of management on carbon sequestration in forest biomass in southeast Alaska. Ecosystems 9:1051-1065

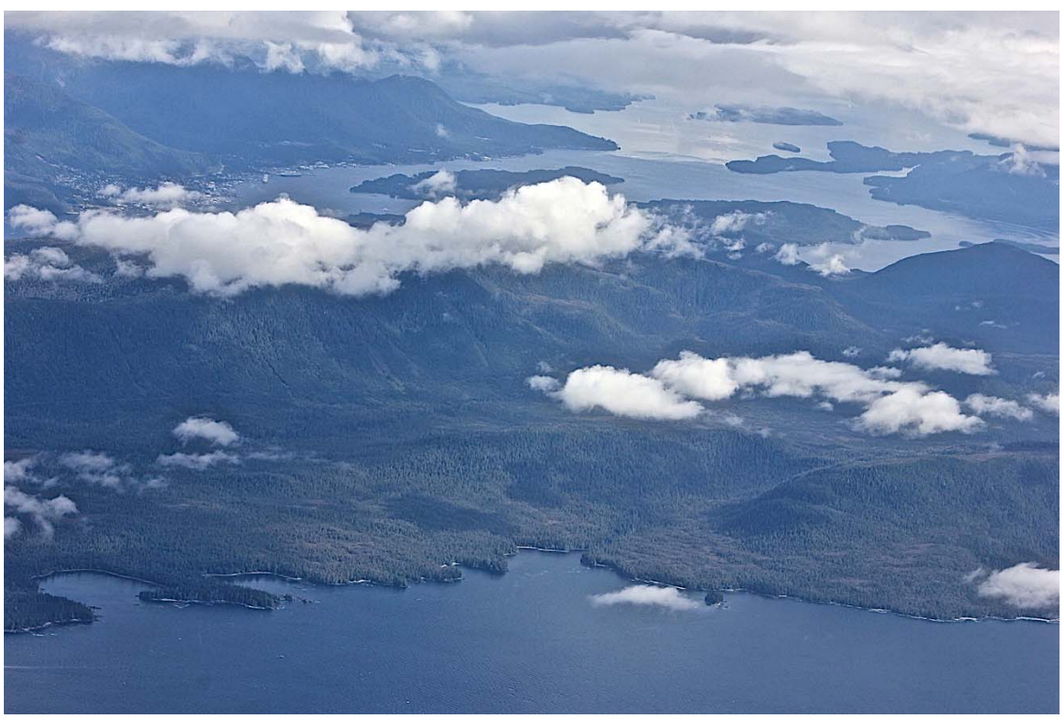
community to *conserve and enhance* forest sinks to help stabilize what may soon become run-away climate chaos.

Conference of the Parties (COP 21) Twenty-First session, Paris, December 12, 2015

"Recognizes the importance of adequate and predictable financial resources, including for results-based payments, as appropriate, for the implementation of policy approaches and positive incentives for reducing emissions from deforestation and forest degradation, and the role of conservation, sustainable management of forests and enhancement of forest carbon stocks; as well as alternative policy approaches, such as joint mitigation and adaptation approaches for the integral and sustainable management of forests…..

Parties should take action to conserve and enhance, as appropriate, sinks and reservoirs of greenhouse gases as referred to in Article 4, paragraph 1(d), of the Convention, including forests."

Photo: D. DellaSala

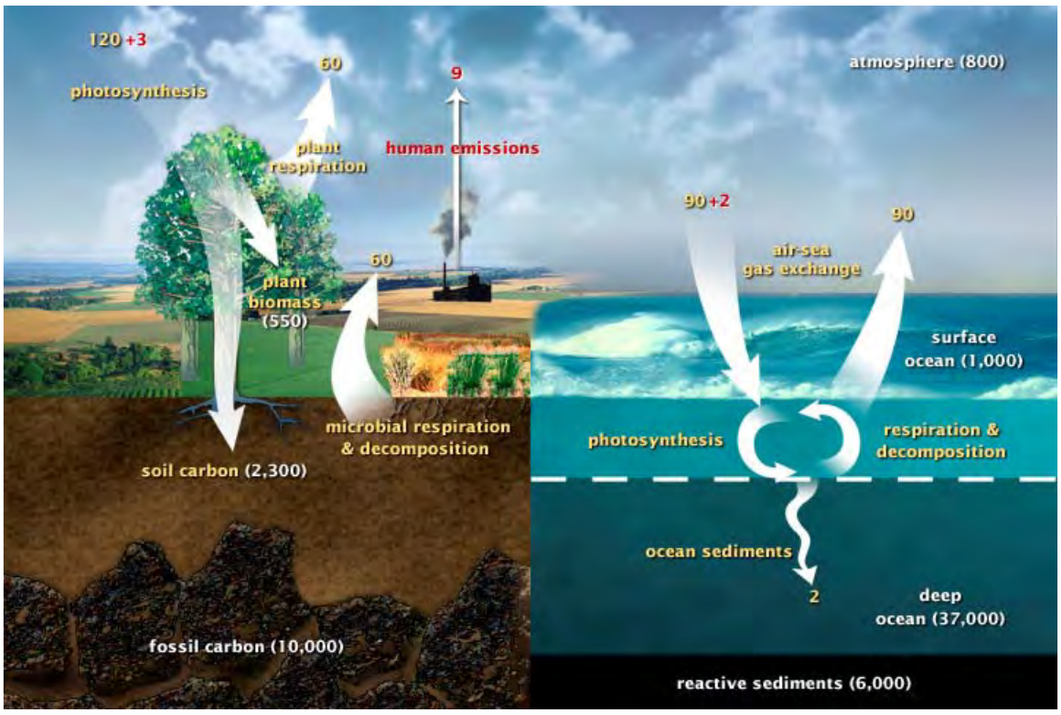


The Tongass is pivotal to the Obama Administration's climate change commitments. The region's forests not only store more carbon than any national forest, but also may function as a climate refuge (i.e., first line of defense) given maritime influences may moderate more extreme climate events anticipated for interior Alaska and temperate rainforests further south[5]. Relatively intact watersheds provide a refuge for old-growth dependent species (including many that are important to subsistence needs), and buffer salmon populations from cumulative effects of climate change and more extensive logging in the surroundings (non-federal lands)[6].

Notably, prior estimates of net carbon flux from logging scenarios on the Tongass indicate that *only a no-logging scenario* maintains carbon stores through time[4]. Carbon

[5]DellaSala, D.A. et al. 2015. Climate change may trigger broad shifts in North America's Pacific coastal rainforests. Online module – Earth Systems and Environmental Sciences – published by Science Direct
[6]For examples, see Watson, et al. 2013. Mapping vulnerability and conservation adaptation strategies under climate change. Nature Climate Change 3:989-994.

also has future economic value in terms of avoided costs from global warming pollution and development of carbon-offset markets. For instance, if carbon were stored long-term in old-growth forests instead of being released to the atmosphere by logging, the estimated annual economic value of carbon would be comparable to revenue generated from Tongass timber sales should carbon markets mature[4]. Moreover, the Interagency Working Group on Social Cost of Carbon estimated the cost of carbon in economic impacts from global warming would be $27-221 per ton by 2050[7]. Recent evidence suggests the anticipated costs maybe much higher, including large demographic displacements of human populations along coastlines[8].



Planetary carbon cycle with exchange of carbon among land, atmosphere, and oceans (billions of tons of carbon per year)[9]. Yellow numbers represent natural carbon fluxes, red are carbon dioxide emissions in billions of tons of carbon per year. White numbers show stored carbon. Note the fossil fuel related carbon stores in the diagram. Forests are integral to the earth's carbon filtration system. http://en.wikipedia.org/wiki/Carbon_cycle

Photo: D. DellaSala

Forests as a Carbon Sink – forests are a vital part of the global atmospheric carbon cycle that contribute to climate stabilization by absorbing (sequestering) and storing vast amounts of carbon dioxide (CO_2) in trees (live and dead), soils, and understory foliage. As a forest ages, it continues to sequester and store carbon, functioning as a net "sink" for centuries if undisturbed. Ongoing carbon sequestration and storage has been measured in forests >800 years old[10].

When a forest is cut down, roughly 66% to 80% of the stored carbon in the forest[11] is released overtime as CO_2 (some carbon is stored in wood products) thereby converting forests from a sink to a "source" or "emitter." The minimal storage in wood products is an accounting misstep typical of federal agency carbon

[7] Interagency Working Group on Social Cost of Carbon, United States Government. 2013. *Technical Update of the Social Cost of Carbon for Regulatory Impact Analysis – Under Executive Order 12866*. May.
[8] Pizer et al. 2014. Using and improving the social cost of carbon. *Science* 346:1189-1190. DOI:10.1126/science.125974
[9]Reprinted from DellaSala, D.A. In 2013. The carbon cycle and global change: too much of a good thing. Reference Module in Earth Systems and Environmental Sciences, Elsevier. 3 pp. http://dx.doi.org/10.1016/B978-0-12-409548-9.05874-7
[10]Luyssaert, S. et al. 2008. Old-growth forests as global carbon sinks. Nature 455:213-215
[11]Wayburn, L.A. 2000 (several citations included). Forest carbon in the United States: opportunities and options for private lands. Pacific Forest Trust, San Francisco.

pronouncements that over value carbon in wood products[12].

Soon after logging, carbon is emitted to the atmosphere via rapid decomposition of logging slash, fossil-fuel emissions from transport and wood processing, and decay or combustion (within 40-50 years) of forest products in landfills[13]. Planting or growing young trees or storing carbon in wood products does not make up for emissions released from a logged forest. Indeed, after an old forest is clearcut, the young forest remains a net CO_2 emitter for 5 to 50 years, depending on site productivity[14].

Logging on the Tongass is global warming pollution(photo: D. DellaSala)

Globally, deforestation (8-15%) and forest degradation (6-13%) **contribute more greenhouse gas pollution than the world's entire transportation network**[15], which is why countries, including the U.S., have committed to reducing emissions and protecting forest sinks (COP 21 climate agreements). Recognizing the importance of unlogged forests as carbon sinks, scientists also have repeatedly called on countries to protect their vast forest carbon stores as integral to stabilizing global climate change[16].

[12]The White House. 2015. Climate change and the land sector: improving measurement, mitigation and resilience of our natural resources.
[13]Harmon, M.E. W.K Ferrel, J. F. Franklin. 1990. Effects on carbon storage of conversion of old –growth forests to young forests. Science 247:699-702.
[14]Law, B. E., and M.E. Harmon. 2011. Forest sector carbon management, measurement and verification, and discussion of policy related to climate change. Carbon Management 2:73-84.
[15]Estimates are conservative as they were mainly derived from the tropics where the majority of forest losses occur – boreal and temperate losses are not available at this time. Intergovernmental Panel on Climate Change. 2007. Synthesis report. An assessment of the IPCC on climate change. Houghton, R.A., B.Byers, and A.A. Nassikas. 2012. A role for tropical forests in stabilizing atmospheric CO_2. Nature Climate Change 5:1022-1023.
[16]MackeyB., et al. 2014. Policy options for the world's primary forests in multilateral environmental agreements. Conservation Letters 8:139-147 DOI: 10.1111/conl.12120. Also letters sent to the Forest Service and USDA in 2015 signed by 7 scientific societies and hundreds of the nation's leading natural resource scientists calling on the Administration to protect the Tongass old-growth rainforest sink.

Photo: The Big Thorne logging operation on Prince of Wales Island converted Tongass old-growth rainforest from a carbon sink to a source of emissions (S. Ballhorn)

"The Tongass National Forest is a national treasure. Today, I am outlining a series of actions by USDA and the Forest Service that will protect the old-growth forests of the Tongass while preserving forest jobs in southeast Alaska. I am asking the Forest Service to immediately begin planning for the transition to harvesting second growth timber while reducing old-growth harvesting over time." July 3, 2013 Press Release, USDA Secretary Tom Vilsack.

Tongass Is Transitioning But Not Soon Enough – Agriculture Secretary Tom Vilsack announced in July 2013 that a transition away from old-growth logging would need to occur rapidly on the Tongass National Forest while maintaining a viable timber industry. In November 2015, the Forest Service released a Draft Environmental Impact Statement (DEIS) Plan Amendment to transition the Tongass from predominately old growth to predominately young-growth logging with the preferred alternative adopting recommendations of a multi-stakeholder Tongass Advisory Committee that incorporated years of additional old growth volume as "bridge timber" to accommodate the transition. Here, we compare the Forest Service preferred alternative to a conservation alternative prematurely dismissed by the Forest Service as not producing enough volume. The agencies' decision to dismiss this alternative occurred before completion of independent field inventories that now show sufficient volume from young growth can accommodate a more rapid transition with minimal old growth (Appendix I, report in preparation).

In conducting theTongass logging emissions analysis, we compared the following:

- *Forest Service Preferred Alternative* – proposes logging 43,167 acres of old growth and 261,850 acres of young growth over 100 years with extensive road building (road building was not calculated in emissions scenarios although it certainly contributes to emissions).

- *Conservation Alternative* – proposed by conservation groups to accelerate the transition while meeting timber demand targets of the Forest Service using much less old growth (OG) to transition. Young growth (YG) estimates were provided by Mater Engineering (Appendix I) from field-verified 55-year old pre-commercially thinned (PCT) YG sampled from a land base of 76,000 acres of relatively low controversy areas (i.e., areas not considered environmentally sensitive based on a suite of attributes, manuscript in preparation). An additional 9,125 acres of old growth was estimated for specialty wood products over 100 years (Appendix I).

We estimated carbon stored in young and old forests by interpolating data from prior estimates on the Tongass[4] for above ground biomass, which was higher than estimates used by the Forest Service for live tree carbon only. We projected logging emissions of the two alternatives over 25- and 100-year increments. We then converted logging emissions to equivalent emissions from vehicles using EPAs equivalencies calculator and compared these projected emissions to CEQ's draft "reference point" for minimizing emissions of federal actions. CEQ directs agencies to adopt projects with low emission using a reference of 25,000 metric tons of $CO_2(e)$[17] on an annual basis[18]. We used the CEQ reference for two reasons: (1) to determine if the preferred alternative is generally consistent with the Obama Administration's global warming commitments (COP 21, Paris agreements); and (2) to provide an appropriate regional comparison of logging emissions that is based on easy to understand emissions comparable. Notably, the Forest Service based logging emissions projections on comparisons to the entire U.S. annual greenhouse gas emissions (the wrong scale of comparison), masking the severity of regionally specific climateimpacts.

ESTIMATING LOGGING EMISSIONS USING VEHICLE EQUIVALENTS

Photo: Juneauempire.com

Forest Service Preferred Alternative – In general, the agencies' preferred alternative to log substantially more OG and YG than proposed by the conservation alternative is estimated to generate **annual** emissions that are:

- equivalent to 4 million vehicles annually for 100-years (Appendix II); and
- 175 times > the CEQ emissions reference.

Conservation Alternative – the transition proposed by the conservation alternative uses much less OG and is estimated to generate **annual** emissions that are:

- equivalent to 419,535 vehicles annually (Appendix II); and
- 16 times > the CEQ emissions reference.

The conservation alternative, while also exceeding CEQ's reference, yields 10 times less emissions in the long-term compared to the agencies' preferred alternative and therefore should have been kept in the DEIS as a reasonable alternative under NEPA. The agencies' preferred alternative is generally inconsistent with the COP 21 climate agreements

[17]Carbon dioxide equivalents (CO_2e) are an internationally accepted term for comparing different greenhouse gas emissions using a common (standardized) unit of analysis.
[18]CEQ 2014. Draft published for public review and comment Dec. 2014. White House.https://www.whitehouse.gov/sites/default/files/docs/nepa_revised_draft_ghg_guidance_searchable.pdf

(Article 4 on greenhouse sinks) to conserve forests as a sink for atmospheric carbon and is well above the CEQ emissions reference.

SOCIAL COSTS OF CARBON
Photo: S. Ballhorn

Executive Order 12866 requires federal agencies to "*assess both the costs and benefits of the intended regulation and, recognizing that some costs and benefits are difficult to quantify, propose or adopt a regulation only upon a reasoned determination that benefits of the intended regulation justify its costs*."

We provide an estimate of the social cost of carbon (SCC) derived from relevant published sources as a means for costing emissions in a regional context and to illustrate how the Forest Service could achieve compliance with the Executive Order by documenting climate costs of logging and the benefits of maintaining the Tongass carbon sink.

In any cost-benefit analysis, it is imperative to incorporate the *benefits* (or cost savings) of avoiding damages to the environment, or, in this case, the climate, so as to level the economic playing field (although many ecosystem services critical to properly functioning forests are difficult to quantify). In this case, SCC is expressed as monetized damages associated with incremental increases in emissions, including, but not limited to changes in net agricultural productivity, human health, property damages from increased flood risk, and the value of ecosystem services. An Interagency Working Group on SCC estimated the **annual** cost of releasing emissions to be $27-221 per ton of carbon using 2050 projections. For this analysis, we used the lower bound of $27 per metric ton of CO_2(e) to estimate potential costs of logging emissions recognizing costs will escalate overtime as a result of the accumulation of regional and global emissions under status quo emissions scenarios.

Forest Service Preferred Alternative - CO_2 (e) released from logging would contribute to:

- ~$108 million annually in global warming costs over 100 years. Estimated costs are 10 times greater than the $8-10 million in annual wood products value anticipated by the Forest Service (DEIS Table 3.22-16).

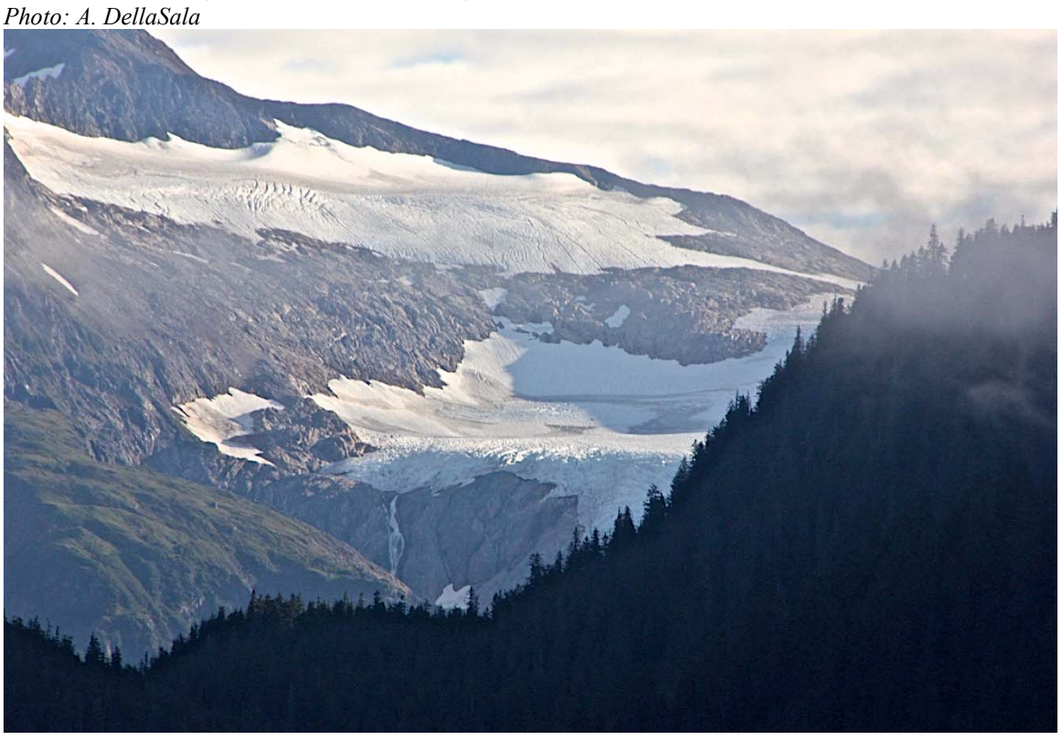
Conservation Alternative - CO_2(e) released from logging would contribute to:

- ~$11 million annually in global warming costs, a tenth as costly as the Forest Service alternative.

Thus, the conservation alternative represents a cost savings to the foreseeable future climate compared to the Forest Service's preferred alternative that would result in much higher costs due to greater logging emissions and this should have been included in the agencies' NEPA analysis. It should be noted that *only a no-logging alternative* results in maximizing carbon sinks and generating apositive SCC. This is because removing carbon from a forest always results in some costs to the climate (costs are based on the combination of regional logging intensity and global emissions contributions).

LIMITATIONS, UNCERTAINTIES, AND THE FUTURE CLIMATE

Photo: A. DellaSala



Follow Up Research and Monitoring – accurately estimating carbon in regional forest assessments requires the use of new carbon assessment tools and improved inventories (including soils) along with inclusion of sequestration rates (e.g., Net Ecosystem Productivity). Carbon assessments are costly but necessary to develop proper carbon flux estimates from logging and to evaluate SCC as a multiple-use objective. In this case, we approximated emissions from published sources, published estimates of carbon stored in wood products (using conversion factors), and published estimates of carbon capture via forest regrowth (using nationally recognized online carbon tools).

Without the benefit of a comparable analysis, however, the Forest Service claims that logging old-growth forests could result in *either a net loss or gain* of carbon depending on logging practices even though clearcut logging (a substantial emissions source) is the method of choice on the Tongass (some young tree retentions and small (<10 ac) clearcuts are proposed in young forests within Old Growth Reserves and Beach buffers by the agency). Our findings are meant to provide a better estimate of emissions than the DEIS. Moreover, we used an appropriate scale of analysis that tiers to CEQ emissions guidelines and used comparable emission sources (e.g., vehicle equivalents that are locally applicable) to evaluate the magnitude of regional impacts. Follow up work, ideally conducted by the Forest Service in collaboration with scientists, is needed to improve upon these estimates and address uncertainties.

Climate Shift Happens – Notably, the effects of climate change on forest productivity represents additional uncertainties. As the climate warms in Alaska, other vegetation types may replace conifer forests that evolved under a cooler climate[3]. For instance, during the Miocene millions of years ago Alaska was a much warmer place dominated by hardwood forests. As climate change now accelerates, it could lower carbon storage in conifer forests as the climate conducive to hardwoods gradually replaces conifers and some conifers die off from climate change effects (thereby releasing CO_2 as is currently

happening with an extensive die-off of Alaska yellow cedar[19]). However, the maritime climate of the Tongass might ameliorate some of these shifts compared to more extreme changes anticipated for interior Alaska and temperate rainforests to the south[3].
Photo: A. DellaSala

ALASKA'S FIRST LINE OF CLIMATE CHANGE DEFENSE AT RISK: CONCLUSIONS

Although the Obama Administration took a leadership position during the climate negotiations in Paris, its global commitments to lower emissions and end deforestation ostensibly do not extend to Alaska's globally significant Tongass rainforest carbon sink.

The Administration has a unique opportunity to demonstrate to the world that it takes its climate change commitments seriously by quickening the pace of transition without relying on controversial timber sales that will cost more in future economic losses from climate change than the revenues generated by logging. The Forest Service has not conducted a logging emissions analysis as directed by CEQ. It has not conducted a cost-benefit analysis of the SCC implications of more OG logging and is out of compliance with Executive Order 12866. The feasibility of an accelerated transition was demonstrated in the conservation alternative summarily dismissed by the agency but which uses much less OG and generates far less emissions over time.

A robust analysis using carbon life cycle accounting is needed to more fully assess the social cost of carbon using advancements in forest carbon accounting as declared in recent climate change policies of the White House[11]. The Tongass is a known carbon sink, yet land-use emissions[11]references the importance of climate resilience best achieved through ecosystem and landscape conservation. Ecosystem resilience, and therefore the Tongass carbon sink, will decline on the Tongass with another 100 years of OG logging and road building. Proposed logging will be occurring at a time when the climate is changing the likelihood that the Tongass can function as a climate refuge[3].

"I loved Alaska and met so many inspiring people. Have to keep up the fight on climate change for their sake—and ours." President Obama on his September visit

The international community clearly spoke up in Paris about the strategic value of forest sinks in keeping global warming below the dangerous 2º C threshold. Choosing a climate responsible alternative for the Tongass would allow the Obama Administration to live up to its commitments to safeguard Alaska's climate, comply with the COP 21 climate agreements and its pledge to end global deforestation.

[19]Hennon P.E.et al. 2012. Shifting climate, altered niche, and a dynamic conservation strategy for yellow-cedar in the North PacificCoastal Rainforest. Bioscience 62: 147–158.

"We share the vision of slowing, halting, and reversing global forest loss while simultaneously enhancing food security for all. Reducing emissions from deforestation and increasing forest restoration will be extremely important in limiting global warming to 2°C." United Nations Climate Summit New York Declaration on Forests (agreed to by 157 governments, including the U.S, indigenous groups, corporations, NGOs, and others)

APPENDIX I. YOUNG GROWTH LOGGING LEVELS NEEDED TO HIT TIMBER DEMAND THRESHOLDS OF THE FOREST SERVICE CALCULATED FROM MATER 2015 PHASE II CRUISE RESULTS (IN PUBLICATION PREPARATION).

	Estimated Avg Length of Tree Residual for PCT Stands When Harvest at 55 yrs Based on 2015 Cruise Results *(lineal ft from top of merchantable log to tree top)*		
	Trees producing *two* 34' logs (ft)	Trees producing *one and a half* 34' logs (ft)	Trees producing *only one* 34' log (ft)
2020	36	27.4	40.86
2025	38.7	41.1	45.23
2030	33.4	31.9	39.39

		Conservation Alternative • With 80% merchantable volume growth every 5 yrs per 2015 cruise results • Focus on *one and a half* log-producing trees
		Annual mbf harvested 2020-2024
		33,074
		Annual mbf harvested starting 2025
Industry Demand (DEIS)	46,000	46,000
	Public Law 113-291	
Total acres harvested; first 10 years	15,000 ac max	13,272 ac
Annual acres harvested; first 5 years	3,000 ac max	1,696 ac
Annual acres harvested; next 5 years	3,000 ac max	958 ac
Total acres harvested overall	50,000 ac max	44,636 ac
Acres harvested per year after 10 years	5,000 ac max	697 ac
Total acres harvested: 25 years		23,727
Total acres harvested: 100 years		76,002 (44,636 before re-harvest)

APPENDIX II: ESTIMATED LOGGING EMISSIONS UNDER THE FOREST SERVICE PREFERRED ALTERNATIVE (DEIS) VS. A CONSERVATION ALTERNATIVE IN 25- AND 100-YEAR INCREMENTS AND IN RELATION TO CEQ'S EMISSIONS GUIDELINES (25,000 METRIC TONS CO2 (E) ANNUALLY)

	DEIS • Comply with PL 113-291 (harvest below CMAI) on 14% of acres • Harvest above CMAI on 86% of acres • Harvest above stated FS demand volume (46 mmbf/yr)	Conservation Alternative • Comply with PL 113-291 (harvest below CMAI) on 100% of acres • Harvest capped at FS demand volume (46 mmbf/yr)
Acres to be harvested first 25 years	• 37,390 ac SG below CMAI (@65 yrs) • 23,223 ac OG (@ 120 yrs)	• 23,727 ac SG below CMAI (@55 yrs) • 3,500 ac OG (@ 120 yrs)
Acres to be harvested next 75 years	• 224,460 ac SG above CMAI (@120 yrs) • 19,944 ac OG (@ 120 yrs)	• 52,273 ac SG below CMAI (@55 yrs) • 5,625 ac OG (@ 120 yrs)
	After initial harvest, re-growth, and (where applicable) re-harvest:	
Total CO2(e) emissions	In 25 years	In 25 years
	105,347,668 tons (4,213,907 tons/yr)	36,576,407 tons (1,463,056 tons/yr)
	In 100 years	In 100 years
	441,068,733 tons (4,410,733 tons/yr)	41,953,532 tons (419,535 tons/yr)
Multiplier above CEQ annual CO2(e) emissions limit of 25,000/yr	In 25 years	In 25 years
	x 168	x 58
	In 100 years	In 100 years
	x 175	x 16

Calculation Notes (all other calculations will be posted online):

- Carbon values interpolated from Leighty et al. 2006 Fig. 2 for age classes as follows: 55 years (494 tons per ac), 65 years (585 tons per acre), 120 years (776 tons per acre).
- Emissions adjusted to account for wood products stores using published estimates in footnote 10 and then multiplied by 3.67 to convert to metric tons CO_2 (e).
- Logging emissions are equivalent to passenger vehicle emissions http://www.epa.gov/energy/greenhouse-gas-equivalencies-calculator.
- CEQ reference = 25,000 metric tons CO_2 (e): https://www.federalregister.gov/articles/2014/12/24/2014-30035/revised-draft-guidance-for-federal-departments-and-agencies-on-consideration-of-greenhouse-gas
- PL 113-291 requires: no more than 50,000 acres of initial YG (not including re-harvest acres) logging; total YG logging in first ten years cannot exceed 15,000 ac; 3,000 ac annual acres in first five years; 3,000 acres annual in 6-10 yrs; and 5,000 YG acres annual after 10 years. If the timber volume goal is 46 mmbf/yr and compliance with PL113-291, the conservation alternative would log: 8,480 acres YG in 2020-2024 (1,696 ac/yr @ 13mbf/ac with a 1.5 multiplier for long log to short log recovery factor) producing 33 mmbf/yr.; not enough pre-commercially thinned 55-yr old stands are available at this time to meet the timber target exclusively from YG); 4,790 acres in 2025-2029(958 ac/yr @ 32mbf/ac with a 1.5 multiplier for long log to short log recovery factor meets that target); 697 acres YG annual logging beginning in 2030 (1.5 multiplier for long log to short log recovery factor producing 46 mmbf/yr @ 44 mbf/ac). See Appendix Ifor Mater 2015 YG numbers plus specialty OG products (e.g., 3 mmbf/yr = 75 ac OG logged per year using a mid point of 40,000 board feet per acre Class 6 old growth (Tongass DEIS: 3-295) to back calculate to acres logged).

Analysis of Carbon Storage in Roadless Areas of the Tongass National Forest

Prepared by Dominick A. DellaSala, Ph.D, in consultation with Brian Buma, Ph.D

We have reviewed the Draft Environmental Impact Statement (DEIS) for the proposed Alaska Roadless Rulemaking and, in particular, its analysis of carbon storage in the inventoried roadless areas of the Tongass National Forest. The DEIS substantially undervalues the global and national importance of old-growth trees on the Tongass for carbon storage. Research shows, for example, that primary (unlogged) forests on the Tongass store much more carbon than logged forests because of the relatively high percentage of old growth and long stable residence times of carbon stored in these forests. The DEIS incorrectly assumes the carbon emitted from logging represents a zero-sum game with carbon recapture in wood product pools and reforestation – this argument is completely false (see below).

- The Tongass is part of a global network of temperate rainforests that make up ~2.5% of the world's total forest coverage but these rainforests have exceptional carbon stores for their relatively small spatial extent and are critically important in climate regulation collectively and individually.[1]

- The Tongass is one of only 4 other temperate rainforests world-wide that is still largely intact, which is a value of global importance grossly undervalued in the DEIS.[2]

- The Tongass occurs within the Pacific Coastal Temperate Rainforest bioregion (extends from Coast Redwoods to Alaska) that includes temperate rainforest ecoregions and climatically distinguishable subregions (subpolar, perhumid, seasonal, warm temperate) considered globally outstanding for their biodiversity and that collectively comprise over one-third of the world's entire temperate rainforest biome based on latest rainforest mapping that should be cited and elevated in importance in the DEIS.[3]

- Tongass carbon stores are substantially greater than any other national forest in the US and are irreplaceable as carbon sinks.[4]

- Primary (unlogged) forests on the Tongass store much more carbon than logged forests because of the relatively high percentage of old growth and long stable residence times of carbon stored in these forests, and in fact old growth forests are accruing biomass at a rate of approximately a Teragram a year.[5] The DEIS incorrectly assumes the carbon

[1] DellaSala et al. 2011.

[2] DellaSala et al. 2011.

[3] DellaSala et al. 2011.

[4] Leighty et al. 2006; Keith et al. 2009; Buma and Thompson 2019. Also, using the dataset in Krankina et al. 2014, the Tongass is a national carbon champion.

[5] *See* Leighty et al. 2006; Keith et al. 2009; Buma and Barrett 2015

emitted from logging represents a zero sum game with carbon recapture in wood product pools and reforestation – this is completely false (see below).

- The Tongass may function as a climate refuge for species facing more extreme climatic conditions in the interior of Alaska and coastal rainforests further south if managed to protect old-growth forests and roadless areas, based on climate envelope modeling and downscaled climate projections for the region.[6]

- Globally, wilderness and intact areas have been declining at an accelerated rate, contain irreplaceable biodiversity and carbon stores, and these losses can be attributed to the "degazetting" (removal of protection status) globally – while roadless areas are not designated wilderness per se – the DEIS continues the alarming global trend of degazetting wild, irreplaceable places.[7] Instead, maintaining and restoring the integrity of intact forests and wild places is an urgent global priority for conservation and sustainability efforts designed to halt the biodiversity and climate crises.[8] Intact areas are also much more likely to retain their native biodiversity than fragmented areas in a rapidly changing climate.[9]

- Large, old growth trees are critically important globally and scientists are calling for protecting places like the Tongass where large trees are especially concentrated to help avoid a biodiversity crisis.[10]

- Because of the global importance of primary (unlogged) forests and high concentration of old-growth forests on the Tongass, scientists are calling on governments to manage these forests to reach their maximum carbon potential via "proforestation" (nature-based climate solutions that allow forests to mature) in order to mitigate climate change.[11]

- The best option for storing carbon long term on public lands is the "no harvest option" for the Tongass and all US public timberlands.[12] Forgoing timber harvest in these areas is projected to result in a net increase of 43% in carbon stores nation-wide, for instance, and an increase in sequestration potential on the national forests such as the Tongass[12]. The DEIS needs to reflect these published estimates and provide a science-based assessment of carbon stored by old forests and estimated emissions from proposed logging given the national and global significance of the Tongass.

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[6] DellaSala et al. 2015.

[7] Watson et al. 2016a.

[8] Watson et al. 2017; Ripple et al. 2019.

[9] Watson et al. 2016b.

[10] Keith et al. 2009; Lindenmayer et al. 2012, 2013; Krankina et al. 2014.

[11] Mackey et al. 2014; Moomaw 2019.

[12] Leighty et al. 2006; Depro et al. 2008

In sum, the Forest Service is responsible for stewarding arguably the most important national forest in the nation and has an ethical-moral and legal obligation to maintain remaining untrammeled areas on the Tongass as irreplaceable assets within the national forest system (as noted by 234 scientists in an October 2019 letter calling on land managers to leave the Roadless Rule in place in Alaska). These irreplaceable values need to be fully acknowledged and protected for their national and global significance.

Additionally, the Forest Service is taking unacceptable climate and biodiversity risks at a time when thousands of scientists have been calling for stricter protections as climate mitigation/adaptation strategies due to the global biodiversity and climate crises we now face.[13] The best alternative for storing carbon long term on public lands is a "no harvest option" for the Tongass and all US public timberlands.[14] Forgoing timber harvest in these areas is projected to result in a net increase of 43% in carbon stores nation-wide, for instance, and an increase in sequestration potential on national forests such as the Tongass. The DEIS needs to reflect these published estimates and provide a science-based assessment of carbon stored by old forests and emitted from proposed logging given the national and global significance of the Tongass and in relation to these cited studies.

A. THE DEIS UNDERVALUES FOREST CARBON AND GROSSLY UNDERESTIMATES EMISSIONS ATTRIBUTABLE TO LOGGING.

NEPA regulations state that:

> NEPA procedures must insure that environmental information is available to public officials and citizens before decisions are made and before actions are taken. The information must be of high quality. Accurate scientific analysis, expert agency comments, and public scrutiny are essential to implementing NEPA.[15]

To ensure that the agency has taken the required "hard look," courts hold that the agency must utilize "public comment and the best available scientific information."[16]

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[13] Watson et al. 2016a,b; Ripple et al. 2017; Ripple et al. 2019.

[14] Leighty et al. 2006; Depro et al. 2008.

[15] 40 C.F.R. § 1500.1(b).

[16] *Biodiversity Cons. Alliance v. Jiron*, 762 F.3d 1036, 1086 (10th Cir. 2014) (internal citation omitted). Regulations implementing the planning provisions of National Forest Management Act (NFMA) also require the use of the best available scientific information (BASI). 36 C.F.R. § 219.3. As noted above, the proposed action includes adding 185,000 acres to the suitable timber base, which requires that the Forest Service amend the Tongass Forest Plan. The Forest Service's planning regulations apply to Forest Plan amendments. 36 C.F.R. § 219.1. Even if they do not apply, they establish sound agency practice and comport with NEPA's mandates regarding best available scientific information and high quality data.

Further, NEPA requires agencies to explain opposing viewpoints and their rationale for choosing one viewpoint over the other.[17] Federal courts have set aside NEPA analysis where the agency failed to respond to scientific analysis that calls into question the agency's assumptions or conclusions.[18]

The DEIS does not present or consider the best available scientific information about the impact of the proposed action on forest carbon. The DEIS presents a contradictory, scientifically flawed, inappropriately scaled and biased accounting of forest carbon losses associated with suspending the national roadless conservation rule on the Tongass. Not a single forest carbon life cycle analysis is presented, yet, the Forest Service draws sweeping conclusions that undervalue the global importance of carbon stored in old growth and roadless areas (IRAs) on the Tongass, while inappropriately minimizing the emissions footprint from roadless entry at a time when overwhelming scientific consensus urges governments to avoid additional emissions from forest degradation and to store more carbon in forest ecosystems.[19] Because agencies and academics have quantified and compared the carbon emissions of alternative logging proposals, the Forest Service cannot fail to undertake a similar analysis on the basis that it is too complex or complicated. Dr. DellaSala's 2016 report addressed carbon stores from wood products and concluded that logging Tongass old-growth forest under the 2016 Forest Plan would result in net annual CO2 emissions totaling between 4.2 million tons and 4.4 million tons, depending on the time horizon chosen.[20] The Bureau of Land Management a decade ago completed an EIS for its Western Oregon Resource Management Plan in which that agency also predicted and quantified the net carbon emissions from its forest and other resource management programs.[21]

Opening up roadless areas and logging in old-growth forests, as the proposed rule would do, conflicts with published research showing the most effective/efficient means to maintain the

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[17] 40 C.F.R. § 1502.9(b) (requiring agencies to disclose, discuss, and respond to "any responsible opposing view").

[18] *See Ctr. for Biological Diversity v. U.S. Forest Serv.*, 349 F.3d 1157, 1168 (9th Cir. 2003) (finding Forest Service's failure to disclose and respond to evidence and opinions challenging EIS's scientific assumptions violated NEPA); *Seattle Audubon Soc'y v. Moseley,* 798 F. Supp. 1473, 1482 (W.D. Wash. 1992) ("The agency's explanation is insufficient under NEPA – not because experts disagree, but because the FEIS lacks reasoned discussion of major scientific objections."), *aff'd sub nom. Seattle Audubon Soc'y v. Espy,* 998 F.2d 699, 704 (9th Cir. 1993) ("[i]t would not further NEPA's aims for environmental protection to allow the Forest Service to ignore reputable scientific criticisms that have surfaced"); *High Country Conservation Advocates v. Forest Service*, 52 F. Supp. 3d 1174, 1198 (D. Colo. 2014) (finding Forest Service violated NEPA by failing to mention or respond to expert report on climate impacts).

[19] Mackey et al. 2013, Mackey 2014, Mackey et al. 2016a,b, Griscom et al. 2017, Law et al. 2018, Ripple et al. 2019, Moomaw 2019.

[20] D. DellaSala, *The Tongass Rainforest as Alaska's First Line of Climate Change Defense and Importance to the Paris Climate Change Agreements* (2016) at 14, and available at https://forestlegacies.org/wp-content/uploads/2016/01/tongass-report-emissions-2016-01.pdf (last viewed Dec. 13, 2019).

[21] *See* Bureau of Land Management, Western Oregon Proposed RMP Final EIS (2009) at 165-181, excerpts attached.

enormous Tongass carbon sink is to protect all remaining old-growth forests from logging.[22] The DEIS carbon assessment does not present the best scientific information, particularly in reference to the global climate emergency[23] or the importance of keeping carbon tied up in Tongass forests as recommended by scientists.[24] In fact, the DEIS goes as far as to boldly proclaim, without a single published scientific reference, that "the management mechanisms applied in all alternatives are consistent with internationally recognized climate change adaptation and mitigation practices identified by the IPCC (IPCC 2000, 2007)."[25] To the contrary, the IPCC (2018)[26] does not endorse roadless development as an appropriate climate mitigation/adaptation strategy. Rather, the IPCC has repeatedly recommended storing more carbon in ecosystems by avoiding additional emissions in the land sector.[27] The same is true for the published sources cited in these comments. We are unaware of any other research that supports the DEIS assertion that clearcutting old-growth rainforests and building roads into intact watersheds is consistent with adaptation and mitigation strategies.

Based on the recent IPCC assessment (2019), an estimated 23% of total anthropogenic greenhouse gas emissions (2007-2016) derive from agriculture, forestry, and other land use. Thus, IPCC recommends avoiding additional emissions from these sectors.

Notably from the IPCC (2019)

> "Achieving land degradation neutrality will involve a balance of measures that avoid and reduce land degradation, through adoption of sustainable land management, and measures to reverse degradation through rehabilitation and restoration of degraded land. Many interventions to achieve land degradation neutrality commonly also deliver climate change adaptation and mitigation benefits. The pursuit of land degradation neutrality provides impetus to address land degradation and climate change simultaneously (high confidence)."

There are at least two fundamental flaws (inherent *biases*) in the DEIS carbon assessment: (1) undervaluing long-term carbon stored in intact watersheds and old-growth forests compared to logged areas; and (2) understating cumulative emissions from logging and road building by using an inappropriate analysis scale and by overstating wood product stores that do not comport with recent published estimates (discussed below).

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[22] Leighty et al. 2006.

[23] Ripple et al. 2019.

[24] Leighty et al. 2006, DellaSala et al. 2011, Moomaw 2019.

[25] DEIS at 3-128.

[26] Given the large size of this report and the fact that the IPCC report is readily available online, we have provided only the only link and not the full pdf - https://report.ipcc.ch/sr15/pdf/sr15_spm_final.pdf.

[27] *See also* Griscom et al. 2017, Moomaw 2019.

The DEIS also does not sufficiently meet the Forest Service's substantive obligation to protect Tongass resources because it: (1) proposes to enter intact watersheds that are acting as irreplaceable strongholds for fish and wildlife populations in a changing climate;[28] and (2) degrades intact areas containing nationally recognized carbon sinks at a time when scientists recommend avoiding entry into intact areas as critical to preventing the escalating climate and biodiversity crises underway globally.[29] Specifically, the DEIS should continue *to protect, preserve, manage, and restore* natural systems (roadless, old growth) on the Tongass, rather than degrade them by development, and then expecting them to somehow be miraculously restored and recovered with all emissions offset by regrowth and wood product stores – an assumption directly contradicted by the best available science (see below).

To assess properly the impacts of the proposed exemption on carbon emissions and sequestration, the agency must address the following key elements and information not now considered in the DEIS.

Trees accumulate carbon over their entire lifespan. While growth efficiency declines as the tree matures, corresponding increases in a tree's total leaf area overcome this slow down as the **whole-tree carbon accumulation rate increases with age and tree size** (Figure 1 – the figure below and some of the text in this section was modified from materials sent to DellaSala by M.G. Anderson, pers. comm). A study of 673,046 trees across six countries and 403 species found that at the extreme, a large old tree may sequester as much carbon in one year as growing an entire medium size tree.[30] At one site, large trees comprised 6 percent of the trees but 33 percent of the annual forest growth. More recent studies show the largest 1% of trees in old-growth forests worldwide store ~50% of the total stand level carbon.[31] In the Tongass, old growth forests continue to accrue biomass and carbon at an amazing rate[32]. In sum, young trees grow fast, but old trees store a disproportionate amount of carbon over time given the larger leaf surface area for absorption and massive tree trunks and root wads that represent centuries of accumulated carbon.

Quoting directly from the abstract in Lutz et al. (2018):

> Main conclusions: Because large-diameter trees constitute roughly half of the mature forest biomass worldwide, their dynamics and sensitivities to environmental change represent potentially large controls on global forest carbon cycling. We recommend managing forests for conservation of existing large-diameter trees or those that can soon

[28] *See* DellaSala et al. 2011, DellaSala et al. 2015, Watson et al. 2016a,b; 2017.

[29] Watson et al. 2016a,b; 2017; Ripple et al. 2019.

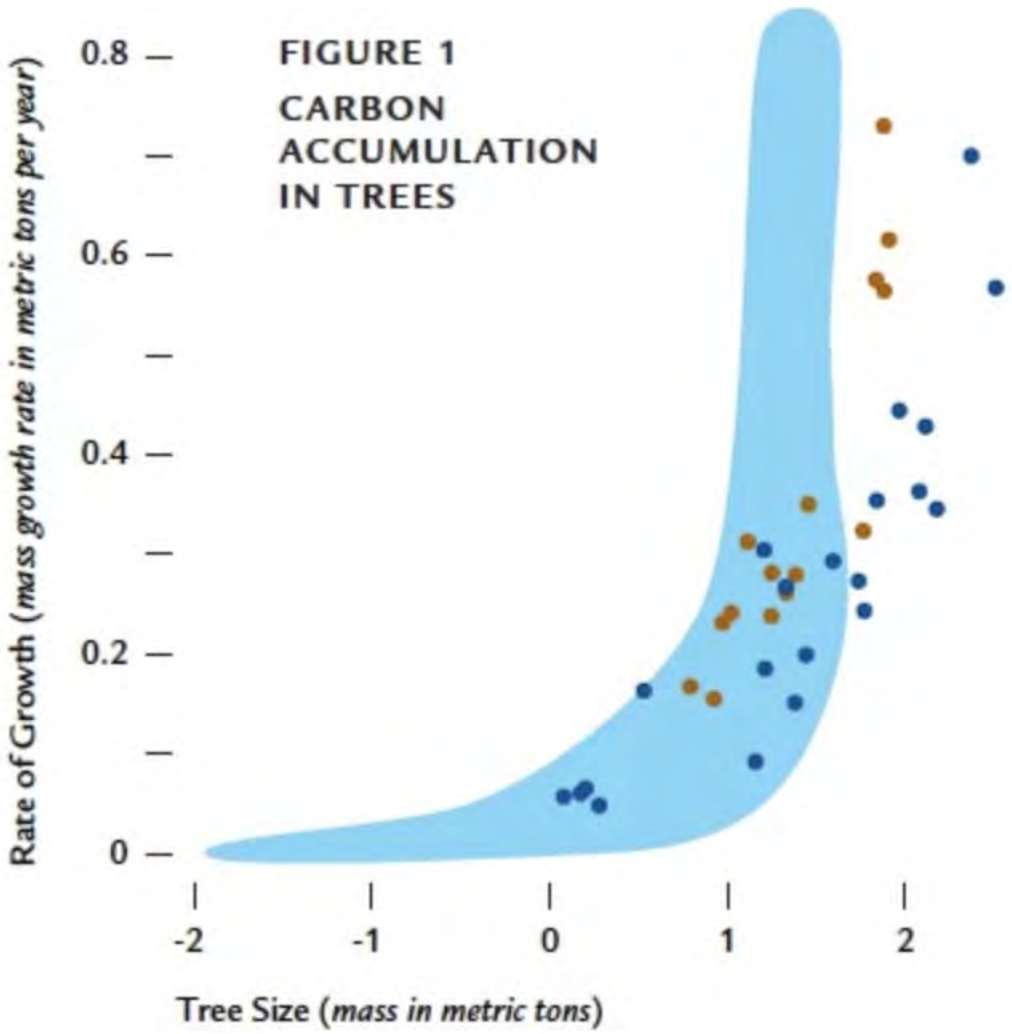
[30] Stephenson et al. 2014.

[31] Lutz et al. 2018.

[32] Buma and Barrett 2015

[33] Lutz et al. 2018

reach large diameters as a simple way to conserve and potentially enhance ecosystem services.[33]



Aboveground mass growth rates for 58 species (shaded area) juxtaposed with two of the most massive tree species on earth: Swamp Gum (*Eucalyptus regnans*—brown dots) and Coast Redwood (*Sequoia sempervirens*—blue dots). Mass growth rate equals the total mass accumulated each year after accounting for respiration. The mass of a tree is primarily carbon, so the figure shows that annual carbon accumulation increases with the size of the tree. (Adapted from Stephenson et al. 2014.)

Old forests accumulate carbon and contain vast quantities of it. Although individual trees experience an increasing rate of carbon sequestration, forest stands experience an "S-curve" of

net sequestration rates (e.g. slow, rapid, slow).[32] The expected decline in older stands is due to tree growth balanced by mortality and decomposition. For instance, an international team of scientists reviewed 519 published forest carbon-flux estimates from stands 15 to 800 years old and found that, in fact, net carbon storage was positive for 75 percent of the stands over 180 years old and the chance of finding an old-growth forest that was carbon neutral was less than 1 in 10.[33] They concluded that old-growth forests are substantial carbon sinks, steadily accumulating carbon over centuries and containing vast quantities of it in relatively stable form.

Old forests accumulate carbon in soils. Soil organic carbon levels in old forests are generally thought to be in a steady state. However, as Alaska's climate increasingly overheats (twice the rate of the rest of the US), soils will be exposed to increased drying and reduced snowpack, and this will lead to methane release. Notably, Tongass soils store >50% of the carbon in the already incredibly dense ecosystem[33]. Moreover, protecting remaining unlogged forests provides for more stable microclimates (with less desiccation and lower temperatures). In fact, recent research shows that old-growth forests may act as a climate buffer as studies comparing logged vs. old growth in the Oregon Cascades found that old growth reduced maximum spring and summer air temperatures as much as 2.5 degrees C.[34] Thus, scientists have repeatedly acknowledged the superior climate benefits inherent to old-growth forests that are irreplaceable in human lifetimes.

Forests share carbon among tree species. Trees compete for sunlight and soil resources, and competition for resources is commonly considered the predominant tree species interactions in forests. However, recent research on carbon isotope labeling has shown that trees interact in more complex ways, including substantial exchange and sharing of carbon below ground. Aided by mycorrhiza networks, interspecific transfer among trees accounts for 40% of the fine root carbon: totally ~280 kg ha-1 per year tree-to-tree transfer.[35] Morrien et al. (2017), found that mycorrhiza soil networks become more connected and take up more carbon as forest succession progresses even without major changes in dominant species composition. Notably, old-growth forests compared to young growth contain more complex below-ground processes that connect trees at the subsurface level.[36] Thus, the Forest Service needs to provide information on the impacts of logging on soil microbial and mycorrhizae carbon exchange before concluding it is insignificant. Failure to include such information would violate NEPA's hard look and BASI mandates.

Primary forest carbon can help slow climate change. Griscom et al. (2017) systematically evaluated 20 conservation, restoration, and improved land management actions that increase carbon storage and avoid greenhouse gas emissions. They found that the maximum potential of

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[33] McNicol et al. 2019

[33] Luyssaert et al. 2014.

[34] Frey et al. 2016.

[35] Klein et al. 2016.

[36] Morrien et al. 2017.

natural climate solutions was ~2.4 Pg of carbon per-year while safeguarding food security and biodiversity.[37] To put the Tongass in this perspective – total Tongass stores = 2.8 Pg carbon with 16%-23% of that in IRAs – additionally, by maximizing carbon in IRAs and old growth (the scientifically recommended climate strategy) – the entire national forest benefits through the maintenance of linked ecosystem services and biodiversity (i.e., multifunctionality of forests maintained via carbon management).[38] New research (see below) suggests this strategy is the most cost-feasible option by a large margin[39] (also see below) and it should receive highest priority as a policy consideration[40] especially on the Tongass.[41] In addition to carbon, old forests also build soil, cycle nutrients, mitigate pollution, purify water, release oxygen, and provide habitat for wildlife at levels far superior than logged forests.[42]

Primary (unlogged) forests are far superior to logged forests in climate mitigation and biodiversity benefits. Globally, primary forests store 30-50% more carbon than logged forests (which is similar to the estimates provided in the DEIS on mature vs. logged Tongass forest stores[43]) and up to half of the carbon stored in a forest is represented by the largest/oldest 1% of trees at the stand level as noted.[44] As stated, logging primary forests results in a net carbon debt and other irreplaceable losses that are not made up for via reforestation or wood product stores as the carbon present in primary forests and soils takes centuries to accumulate compared to much shorter-lived wood products that represent only a fraction of the original forest store.

In part because the DEIS analysis fails adequately to account for this basic scientific information relevant to an assessment of the impact of the proposed exemption on carbon and climate impacts, the DEIS is flawed in at least the specific ways described herein.

Tongass carbon stores need to be prioritized as globally and nationally significant climate mitigation/adaptation strategies to be protected, preserved, and managed as unique ecological communities. Old-growth forests, in general, store massive amounts of carbon in trees, foliage, and soils. Pacific coastal rainforests, in particular, are global champions in this regard.[45] Of relevance, temperate rainforests in Alaska store >2.8 Petagrams (Pg) C (1 Pg = 1 billion tonnes) in biomass and soils, the equivalent of >8% of the carbon in all contiguous US forests, most of which is on the Tongass.[46] Based on FIA datasets, Tongass roadless areas

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[37] Griscom et al. 2017.

[38] *See* Brandt et al. 2014.

[39] Moomaw et al. 2019.

[40] McKinley et al. 2011.

[41] Leighty et al. 2006; Buma and Thompson 2019

[42] Mackey et al. 2014, Brandt et al. 2014.

[43] DEIS at 3-124.

[44] Lutz et al. 2018.

[45] Leighty et al. 2006, Keith et al. 2009, Krankina et al. 2014.

[46] Leighty et al. 2006; Buma and Thompson 2019; McNicol et al. 2019

represent ~16% to 23% of total carbon on the Tongass forest depending on categories used (Table 1, 2). Thus, roadless areas – especially those with old-growth forests – are uniquely valuable as a long-term stable carbon sink compared to logged areas that emit most of their carbon (see below).

The Tongass stores a massive amount of carbon--the total carbon stored in Tongass roadless areas are equivalent to annual emissions of ~128, 550-watt coal-fired power plants.[47] Keeping carbon in forests is a fundamental climate mitigation strategy directly responsive to the climate emergency[48] and essential to offsetting some of the emissions from the energy sector. The Tongass stores a massive amount of carbon in its old growth forests, at levels that if emitted into the atmosphere would approach the emission equivalents of coal-fired power plants. At a time when the world is looking for leadership on cutting emissions at all scales, removing protection for this carbon storage is unsupportable. Table 1 provides a breakdown of Tongass old-growth roadless carbon values (including congressionally withdrawn areas), Table 2 just the IRA carbon values, and Figure 1 shows the spatial distribution of carbon stores on Tongass IRAs. Table 3 shows that Alternative 6 will place at risk 71.5% of the carbon stored in old-growth forests and soils, with most of that carbon emitted to the atmosphere (see Leighty et al. 2006). Table 4 provides an economic estimate of the carbon value at risk to logging on the Tongass under Alternative 6 (>$234 million), which may far exceed timber values. Additionally, if the Forest Service enters all roadless areas in this century >$2.2 billion in carbon assets will be squandered away, should an offset market develop. All these data were available to the Forest Service (Forest Inventory Assessment - FIA) and they need to be fully analyzed in the DEIS to provide reliable estimates of carbon assets and their relative (to timber), tradeoffs involved, and the economic importance on the Tongass of carbon, along with reliable estimates of emissions from logging. Disclosing these tradeoffs is especially relevant at a time when the IPCC (2018, 2019) and other reports (Ripple et al. 2017, 2019) have warned that we have about 10 years before severe climate impacts are locked in with irreversible consequences to biodiversity and the planet's life-giving systems.

[47] https://www.oregonlive.com/business/2010/12/pges_coal-fired_boardman_plant.html.

[48] *See* Moomaw 2019, Ripple et al. 2019.

Table 1. Estimated carbon biomass in old-growth forest among categories of roadless areas on the Tongass NF						
OwnerType_2019	**USDA FOREST SERVICE**					
POG	**OG Only**					
	low estimate	**high estimate**	**Total area**		**Estimate C in OG forest biomass**	
Row Labels	**Average of Carbon_ratio 58**	**Average of Carbon_ratio 46**	**Sum of GIS_Hectares**		**(low estimage in Mg)**	**(high est. in Mg)**
Roadless in 2001 Rule	264.7	333.7	999,179		264,443,913	333,444,106
Lg. Roadless Areas not in	257.1	324.1	76,166		19,580,156	24,689,023
Small Rdls Areas	242.4	305.6	58,329		14,139,031	17,828,070
Roaded-Roadless	249.3	314.4	14,357		3,579,634	4,513,860
Roaded Areas	263.3	332	156,023		41,076,507	51,794,893
Wilderness or NM	247.7	312.3	662,496		164,101,837	206,915,638
Non-USFS Lands	261.5	329.8	4,171		1,090,861	1,375,453
Unknown	187.5	236.5	982		184,209	232,273
Grand Total	**255.6**	**322.3**	**1,971,704**		**503,969,209**	**635,467,036**

Table 2. Carbon stored in roadless area categories on the Tongass.

Large Roadless Areas Average Mg per ha	Total Mg Carbon	"Roaded Roadless" Average Mg per ha	Total Mg Carbon	Small Rdls Areas Average Mg per ha	Total Mg Carbon	Roaded Areas Average Mg per ha	Total Mg Carbon	Protected by Congress Average Mg per ha	Total Mg Carbon	Non-USFS Lands Average Mg per ha	Total Mg Carbon	Total Average Mg per ha	Total Total Mg Carbon
242.58	**47,208,021**	**274.78**	**98,058**	**313.89**	**1,867,886**	**361.48**	**10,026,227**	**207.71**	**66,457,805**	**263.80**	**5,436,143**	**260.64**	**131,094,140**
245.92	2,081,610					317.13	83,671	224.49	36,976,406	205.86	248,378	224.82	39,390,064
224.39	7,939,491	248.04		361.21	317,926	374.39	1,469,914	179.03	2,579,177	277.13	198,836	277.09	12,513,023
255.70	18,610,022	295.11	7,679	305.53	1,322,520	356.44	6,423,528	210.15	11,972,020	286.92	3,550,567	278.05	41,964,449
231.40	17,172,289	226.63	85,792	304.03	227,439	371.92	2,025,237	144.80	7,123,632	252.95	1,407,175	261.84	27,960,360
190.38	1,404,610		4,587			234.51	23,876	185.47	7,806,570	228.97	31,187	192.89	9,266,244
232.20	**58,182,471**	**262.79**	**1,429,232**	**261.26**	**4,248,583**	**303.66**	**15,895,643**	**196.32**	**22,381,595**	**253.61**	**11,984,173**	**249.11**	**114,121,697**
236.71	9,363,695	232.25	279,800	213.54	756,243	291.33	3,979,300	167.72	2,705,767	252.31	1,048,227	244.24	18,133,031
248.42	4,621,788			282.27	61,790	345.31	264,235	231.39	6,290,381	200.47	149,533	246.54	11,387,728

201.63	14,044,010	240.78	675,871	236.58	1,737,670	276.70	4,000,336	199.16	2,694,688	216.69	3,418,960	225.90	26,571,535
289.39	4,013,721	317.04	8,473	337.44	1,008,771	340.32	3,176,035	284.31	1,032,343	291.65	521,090	311.71	9,760,434
237.96	26,139,257	301.06	465,088	286.21	684,109	320.59	4,475,736	177.93	9,658,416	268.36	6,846,362	254.25	48,268,969
229.96	**31,318,111**	**247.66**	**892,278**	**238.48**	**2,941,262**	**290.26**	**16,874,911**	**239.86**	**16,293,468**	**268.92**	**21,060,394**	**255.37**	**89,380,425**
264.73	5,634,563			226.80	397	356.33	2,846			265.96	3,679,426	266.56	9,317,233
219.88	15,720,049	244.81	775,492	234.97	2,586,253	287.67	15,078,975	258.07	7,425,974	261.59	14,491,136	252.13	56,077,879
281.10	2,088,047	230.67	21,443	251.59	236,829	294.96	1,465,398	196.99	4,999,443	315.98	468,652	278.44	9,279,811
231.49	7,875,451	293.14	95,343	284.60	117,783	322.67	327,692	192.56	3,868,051	296.39	2,421,180	256.94	14,705,501
186.13	**53,884,796**	**336.78**	**287,412**	**333.72**	**595,565**	**312.46**	**2,942,711**	**149.44**	**66,534,550**	**259.89**	**5,757,139**	**213.28**	**130,002,173**
192.56	11,619,409	369.34	120,627	327.18	275,972	313.76	861,295	118.57	3,208,927	268.13	1,822,554	235.07	17,908,784
62.94	1,228,345			368.10	96,490	136.68	78,974	159.84	18,154,192	284.05	46,875	168.06	19,604,876
						123.88	45	181.02	31,003,210	148.94	170	179.52	31,003,425
187.62	21,671,798	288.49	136,481	309.91	31,943	325.86	1,478,419	118.41	7,974,912	260.51	721,358	212.07	32,014,912
192.02	19,365,243	314.33	30,304	440.91	191,161	322.72	523,978	118.67	6,193,309	251.30	3,166,182	215.24	29,470,176
225.32	**190,593,399**	**264.38**	**2,706,980**	**265.82**	**9,653,295**	**312.40**	**45,739,492**	**196.09**	**171,667,419**	**263.11**	**44,237,849**	**247.37**	**464,598,434**

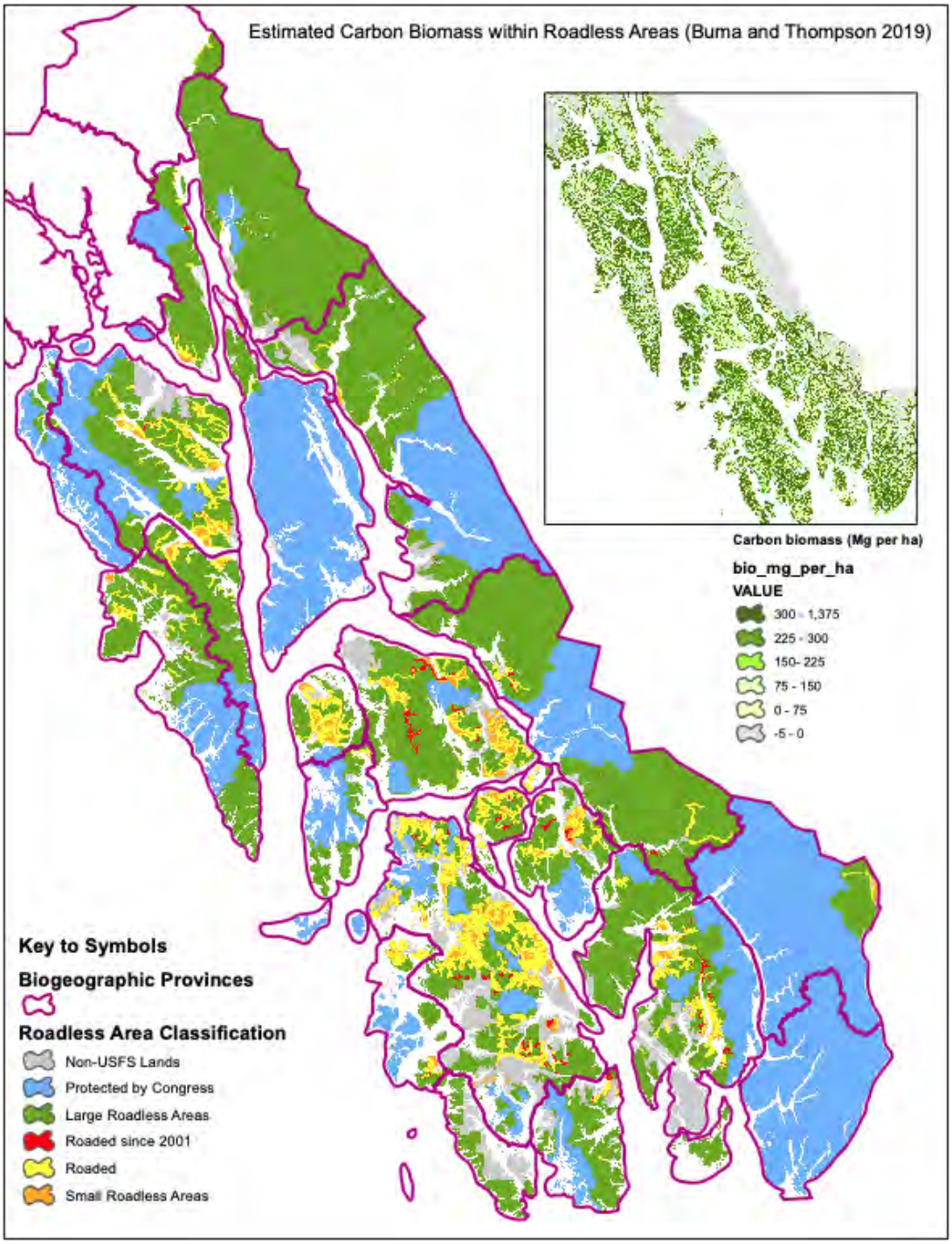
Estimated Carbon Biomass within Roadless Areas (Buma and Thompson 2019)
Carbon biomass (Mg per ha)
bio_mg_per_ha
VALUE
300 - 1,375
225 - 300
150- 225
75 - 150
0 - 75
-5 - 0
Key to Symbols
Biogeographic Provinces
Roadless Area Classification
Non-USFS Lands
Protected by Congress
Large Roadless Areas
Roaded since 2001
Roaded
Small Roadless Areas

Table 3. Estimated Mg of forest and soil carbon on lands suitable for old-growth logging under DEIS Alternatives

	Alternatives					
Forest & Soil Carbon Estimates	**1**	**2**	**3**	**4**	**5**	**6**
Suitable Acres (with Data)	229,564	249,888	307,778	387,941	394,997	394,997
Net Change from Alt 1 (acres)	0	20,325	78,214	158,377	165,433	165,433
Suitable Hectares (w/data)	92,901	101,126	124,554	156,994	159,850	159,850
Net Change from Alt 1 (hectares)	0	8,225	31,652	64,093	66,948	66,948
Total Forest C (low est)	23,625,799	25,643,535	31,591,558	39,655,731	40,508,557	40,508,557
% Increase from Alt 1 (low est)	0.0%	8.5%	33.7%	67.8%	71.5%	71.5%
Total Forest C (high est)	29,790,661	32,334,884	39,834,901	50,003,261	51,078,589	51,078,589
% Increase from Alt 1 (high est)	0.0%	8.5%	33.7%	67.8%	71.5%	71.5%
Total Soil C	34,284,875	37,153,086	45,699,226	56,497,163	57,468,262	57,468,262
% increase from Alt 1 (soil)	0.0%	8.4%	33.3%	64.8%	67.6%	67.6%
Forest + Soil C (low)	57,910,675	62,796,621	77,290,783	96,152,894	97,976,819	97,976,819
Forest + Soil C (high)	64,075,536	69,487,970	85,534,126	106,500,424	108,546,851	108,546,851
% Increase from Alt 1 (high)	0.0%	8.4%	33.5%	66.2%	69.4%	69.4%

Table 4. Economic value of at-risk carbon (Alternative 6 plus all suitable)

	Alt 6 suitable timber at-risk	All suitable timber at-risk
Acres	42,500	394,997
Low est total carbon	40,508,577	40,508,577
CO2 (carbon x 3.67)	148,666,478	148,666,478
Value of CO2 at-risk in suitable timber base* at $15/ton CO2	$240,839,694 40% logged in first decade = $96.3 million	$2.2 billion

*Suitable timber base = 10.8% of at-risk carbon under Alt 6, 100% at risk under all suitable acres

Carbon emissions assessment by the Forest Service provides a misleading comparison to other emissions and fails to include a social cost analysis. The DEIS is woefully inadequate as it compares emissions (prior and current logging) on the Tongass to gross emissions from the **entire US electric power sector in 2012** and **all US emissions in 2017**.[49] Federal courts have rejected this kind of skewed comparisons.[50] This arbitrary baseline ignores the incremental nature of carbon emissions and impacts and is inconsistent with recommendations of the IPCC (2018) to avoid additional emissions, and with the broader scientific consensus of fully protecting carbon sinks like the Tongass.[51] To comply with NEPA, the Forest Service must, at a minimum, explain why it is choosing to ignore these expert conclusions. The global community also has signaled its intent to protect carbon sinks under Article 5 of the Paris Climate Agreement. While the US is irresponsibly withdrawing from the agreement, it is also irresponsible for the Forest Service to downplay the substantial regional emissions from Tongass roadless and old growth logging when the rest of the world is looking for ways to reduce and avoid emissions **at all scales**. Instead, the agency should choose the alternative with the least

[49] DEIS at 3-124.

[50] *See High Country*, 52 F. Supp. 3d at 1190 ("Beyond quantifying the amount of emissions relative to state and national emissions and giving general discussion to the impacts of global climate change, [the agencies] did not discuss the impacts caused by these emissions."); *Mont. Envtl. Info. Ctr. v. U.S. Office of Surface Mining*, 274 F. Supp. 3d 1074, 1096–99 (D. Mont. 2017) (rejecting the argument that the agency "reasonably considered the impact of greenhouse gas emissions by quantifying the emissions which would be released if the [coal] mine expansion is approved, and comparing that amount to the net emissions of the United States"); *WildEarth Guardians v. Zinke*, 368 F. Supp. 3d 41, 76-78 (D.D.C. 2019) (holding BLM's conclusion that the emissions from oil and gas leases "represent an incremental contribution to the total regional and global GHG emissions level" was arbitrary and capricious because it was not supported by any data).

[51] Keith et al. 2009, DellaSala et al. 2011, Stephenson et al. 2014, Mackey et al. 2014, Law et al. 2018, Moomaw et al. 2019.

emissions – no action – and compare alternatives against no action with respect to reliable and accurate direct, indirect, and cumulative emissions. This also needs to be expressed in carbon dioxide equivalents to estimate the socio-economic cost of carbon. Additionally, the emissions need to be expressed at an appropriate **regionally specific scale**, as for instance, coal-fired power plant equivalents as mentioned above so that the public understands the true regional climate consequences of opening roadless areas to logging and development.

Further, the DEIS falsely asserts that "it is difficult and highly uncertain to ascertain the indirect effects of emissions resulting from these alternatives on global climate."[52] The Forest Service could easily express the indirect impacts of climate emissions by quantifying or estimating climate pollution volumes by alternative (as noted above in our analysis) and then using the social cost of carbon (SCC) to assess and compare the significance of the effects on global climate. The very purpose of the SCC is to assist decisionmakers in (conservatively) estimating the marginal damages from each additional ton of greenhouse gas emissions. To avoid this analysis irresponsibly kicks the emissions can down the road.

The DEIS incorrectly states that most of the carbon in trees after logging will be recovered via reforestation and stored in wood products for buildings instead of stored in forest ecosystems and this is completely false. As noted, a substantial portion of the total forest carbon is contained in foliage, branches and bark, root wads and soils.[53] Because much of the carbon in logs hauled to mills becomes waste, only a relatively minor portion of the total tree carbon ultimately ends up in wood products.[54] Up to 40% of the harvested material does not become forest products and is burned or decomposes quickly on site, and a majority of manufacturing waste is burned for heat. One study found that 65% of the carbon from West Coast forests logged over the past 100 years is still in the atmosphere with just 19% stored in long-lived products; the remainder is in landfills.[55] Additionally, Leighty et al. (2006) reported that a century of Tongass logging has emitted 6.4-17.2 Tg C that is still in the atmosphere (again – it matters most what the atmosphere "sees" more than what is stored in wood products). Further, Hudiburg et al. (2019) note that state and federal reporting of emissions has erroneously excluded some product-related emissions, resulting in 25-55% underestimation of total CO2 emissions from logging. Thus, the Forest Service needs to fully disclose and provide reliable estimates on how much carbon is emitted by clearcutting given the substantial fall down and problems with underestimating emissions as noted. Large amounts of logs, stumps, root wads and slash are left on the ground after clearcutting and soils are noticeably disturbed by heavy equipment. This cannot be simply dismissed as an insignificant impact in the DEIS.

It is also wrong for the Forest Service to assert that carbon stored in Tongass saw logs (wood product pools) compensates for carbon emitted by logging long-lived (hundreds of years) trees in

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[52] DEIS at 3-127

[53] Campbell et al. 2007.

[54] *See, e.g.*, Harmon et al. 1990, Harmon et al. 1996, Ingerson 2008, Law et al. 2018, Harmon 2019.

[55] Hudiburg et al. 2019.

the Tongass old-growth carbon sink.[56] The carbon debt created by expansive clearcut logging (past, present, future) must be calculated using reliable and accurate estimates via a carbon life cycle analysis that accounts for how long carbon remains in the atmosphere (after all, it's what the atmosphere "sees" that matters most in the long run). Thus, at a minimum, NEPA requires that the Forest Service conduct a carbon life cycle analysis using published sources and the Forest Service should use FIA/timber stand data on estimated carbon uptake and stores in old growth vs. young growth to calculate age-related differences in carbon stores and associated emissions from logging (e.g., using the carbon values for Tongass old growth and IRAs in our comments) at the regional scale. The following analysis components should be included in the DEIS:

- In-boundary emissions – at the stand and landscape level, this includes carbon entering the atmosphere from the substantial "fall down" and defect of uneconomical logs, slash, and stumps – based on Tongass timber stand inventory data (2016-18) fall down alone (uneconomical material) may be as high as 70% of felled trees (carbon emitted directly to the atmosphere) with old-growth defect at least 30%.

- Out-of-boundary emissions – this includes: (1) carbon emitted via wood processing waste at the mill (see Law et al. 2018 for example); (2) fossil fuels used in transport and manufacture of wood products, including emissions from log exports sent to China and then exported for distribution as products, the lower 48 states and elsewhere (note - transport emissions are easily obtained from the Alaska Department of Environmental Conservation Division of Air Quality (greenhouse gas emission inventory[57]; and (3) estimated emissions from road building.[58]

- Use more recent studies on wood product substitution estimates – Harmon (2019), for instance, re-examined substitution assumptions questioning their *reliability* in life cycle analysis and concluding that any benefits depend on duration of fossil carbon displacement, longevity of buildings being assumed, and nature of the forest supplying building materials (also see below):

  "Substitution of wood for more fossil carbon intensive building materials has been projected to result in major climate mitigation benefits often exceeding those of the forests themselves. A reexamination of the fundamental assumptions underlying these projections indicates long-term mitigation benefits related to product substitution may have been overestimated *2- to 100-fold (emphasis added)*. This suggests that while product substitution has limited climate mitigation benefits, to be effective the value and duration of the fossil carbon displacement, the longevity of buildings, and the

[56] *See, e.g.*, DEIS at 3-127.

[57] *See* https://dec.alaska.gov/air/anpms/projects-reports/greenhouse-gas-inventory)

[58] *See* Loeffler et al. 2008 for how to estimate this -note – this is a Forest Service publication easily accessible to the agency.

nature of the forest supplying building materials must be considered."[59] Failure to address this scientific study would violate NEPA's and NFMA's mandate that the Forest Service use the best available science and that the agency explain why its approach differs from that of experts.

- The need for reliable published references to estimate wood product stores– Researchers report most carbon is emitted to the atmosphere when old trees are logged, accounting for wood product stores is only a fraction of the carbon pool (e.g., ~35% of the live carbon is rapidly emitted when an old-growth forest is logged with another 30% emitted at the mill and even more in transportation).[60]

- The reference to an albedo effect in the DEIS (at 3-123) is unreliable, cannot be verified, is inconsistent with the BASI requirement, and should be dropped. The DEIS provides no citation or support for its unsubstantiated albedo assumption, which likely was extrapolated from the boreal regions where albedo has been reported as having a potential cooling affect because of the reflectance properties of snow. The Forest Service cannot make this same claim for the Tongass given that low-elevation temperate rainforests experience relatively little snow (and therefore have low albedo/reflectance properties), especially in a changing climate (as noted in the DEIS). Without a life cycle analysis that first estimates logging emissions and then compares emissions to whatever insignificant albedo effect is anticipated in temperate regions with little snow, the albedo cooling assumption is falsified and cannot be used for disclosing climate impacts of Tongass logging. In sum, large regional and ecosystem type variations have been observed in albedo and one cannot compare albedo from one region to another or one forest type to another.

In this regard, the DEIS echoes unsupportable claims and assumptions by the wood products industry that substituting wood for concrete and steel reduces the overall carbon footprint of buildings and thus is unreliable and inaccurate. The agencies' wood production substitution claim has been refuted by recent analyses that reveal forest industries have been using unrealistic and erroneous assumptions in their models, overestimating the long-term mitigation benefits of substitution by 2- to 100-fold.[61] An additional recent analysis concluded that the carbon footprint of wood is 6% higher than concrete (Stiebert et al. 2019), and that assessment did not include the reduced forest carbon sequestration and storage caused by forest losses as discussed. Importantly, a very recent breakthrough in solar energy production will soon make it possible to dramatically reduce the carbon footprint of concrete and steel even further.[62] Additionally, regarding the noted problems with exaggerated wood substitution benefits,[63] there is no

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[59] Harmon 2019. –

[60] *See* Harmon et al. 1990, Harmon et al. 1996, Law et al. 2018, Harmon 2019.

[61] As discussed, Law et al. 2018, Harmon 2019.

[62] https://www.cnn.com/2019/11/19/business/heliogen-solar-energy-bill-gates/index.html.

[63] *See* DEIS at 3-123.

assurance that concrete and steel replaced by wood will not be used in application somewhere else (i.e., leakage from using steel/concrete used elsewhere). For the substitution benefit to accrue, an equivalent amount of concrete or steel would need to not be produced and used in construction; otherwise, substitution is purely speculative (not best science) and unreliable. Further, the DEIS did not account for the high recycled content in most steel or recent/future anticipated advances in reducing the carbon footprint of concrete. For instance, changing manufacturing methods impact embodied energy, as for example, if fly ash is added to concreate it could yield 22-38% reductions in embodied energy required in manufacturing processes, thereby reducing the displacement value of wood.[64] Using clean, renewable energy instead of coal in concrete and steel manufacturing also can lower the substitution value and is part of the mix of energy sources being expand upon by the global community (i.e., over the next few decades new energy sources and processing efficiencies will emerge to reduce concrete/steel emissions and this needs to be factored into a "best case scenario" for energy efficiency upgrades in the DEIS). This change is already underway.[65]

To construct a proper life cycle analysis that provides a science-based assessment of carbon stocks and flows on the Tongass, the DEIS should adopt a method similar to the approach used by Hudiburg et al. in their 2019 life cycle analysis of emissions from logging. The following abstract summarizes their methodologies:

> Abstract
> Atmospheric greenhouse gases (GHGs) must be reduced to avoid an unsustainable climate. Because carbon dioxide is removed from the atmosphere and sequestered in forests and wood products, mitigation strategies to sustain and increase forest carbon sequestration are being developed. These strategies require full accounting of forest sector GHG budgets. Here, we describe a rigorous approach using over one million observations from forest inventory data and a regionally calibrated life-cycle assessment for calculating cradle-to-grave forest sector emissions and sequestration. We find that Western US forests are net sinks because there is a positive net balance of forest carbon uptake exceeding losses due to harvesting, wood product use, and combustion by wildfire. However, over 100 years of wood product usage is reducing the potential annual sink by an average of 21%, suggesting forest carbon storage can become more effective in climate mitigation through reduction in harvest, longer rotations, or more efficient wood product usage (emphasis added). Of the∼10,700 million metric tonnes of carbon dioxide equivalents removed from west coast forests since 1900, 81% of it has been returned to the atmosphere or deposited in landfills (emphasis added). Moreover, state and federal reporting have erroneously excluded some product-related emissions, resulting in 25%–55% underestimation of state total CO2 emissions. For states seeking to reach GHG reduction mandates by 2030, it is important that state CO2 budgets are

[64] Harmon 2019.

[65] *See* J. Gillis, The Steel Mill That Helped Build the American West Goes Green, The New York Times (Oct. 16, 2019) (describing Colorado steel mill's decision to manufacture steel using only renewable energy), available at https://www.nytimes.com/2019/10/16/opinion/solar-colorado-steel-mill.html (last viewed Dec. 13, 2019).

effectively determined or claimed reductions will be insufficient to mitigate climate change.[66]

Logging involves transportation of trucks and machinery across long distances between the forest, the mill, and point of distribution and the DEIS needs to properly disclose these emission sources. For every ton of carbon emitted from logging, an additional ~17% is estimated from fossil fuel consumption to support transportation, extraction, and processing of wood[67], not including the significant emissions from building roads.[68] There is no indication that this was even accounted for in the DEIS.[69] As noted, the Forest Service should consult with state emissions data to obtain reliable estimates of emissions from transport and manufacturing of wood products, particularly the incredibly long hauling distances involved with exporting logs to China and the burning of fossil fuels to get them there (plus when manufactured products are shipped again to retail and distribution areas). In the Tongass this is an especially valid concern given the remote location, no road access (necessitating saltwater barges), and weather which requires extensive and long transportation chains.

The DEIS does not account for the reduction in carbon sequestration and storage potential in forests due to logging-caused soil compaction and nutrient loss. This is despite the fact that these combined impacts can reduce forest carbon storage potential contributing to an overall carbon debt not explained or assessed in the DEIS. We note that this debt is not trivial because ~60% of the carbon lost through logging since 1700s has not yet been recovered by the land sector[70] and 81% of carbon previously stored in West Coast forests has been returned to the atmosphere via logging since 1900.[71] These are centuries-long atmospheric carbon emissions coming at a time when we are in a climate emergency.[72] This is why scientists are calling for policies that avoid emissions and store more carbon in forests compared to wood product pools.[73] Additionally, there are other greenhouse gas effects such as methane and nitrous oxide emissions from soil impacts that will impact the climate from logging.[74]

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[66] Hudiburg et al. 2019.

[67] Ingerson 2008.

[68] *See* Loeffler et al. 2008.

[69] The DEIS at 3-127 includes "transporting wood products" in a laundry list of potential cumulative impacts to consider in its climate analysis, but provides no analysis at all of the scale or nature of that impact, violating NEPA's hard look mandate.

[70] McKinley et al. 2011.

[71] Hudiburg et al. 2019.

[72] Ripple et al. 2019.

[73] Hamon et al. 1990, 1996, Leighty et al. 2006, McKinley et al. 2011, Mackey et al. 2016a,b, Law et al. 2018, Moomaw 2019.

[74] McKinley et al. 2011.

In sum, the DEIS fails to include peer-reviewed science on forest carbon and emissions that shows: (1) primary (unlogged) forests are far superior to logged forests at carbon uptake and storage long term; (2) trees accumulate carbon over their entire lifespan; older trees capture and store far more carbon than young trees; (3) old, primary forests accumulate far more carbon than they lose through decomposition and respiration, thus acting as net carbon sinks; (4) logged forests are an emission source for at least the first decade and never fully recapture the emitted carbon stored in the pre-logged old-growth forest due to short rotation harvests and carbon losses throughout the wood product distribution chain; and (5) the superior carbon benefits of old forests are especially evident when taking into account the role of undisturbed soils (which may contain ~50% of carbon stores[75],) and below ground carbon exchange losses from logging and climate change impacts.

B. DEIS CLAIMS ABOUT TEMPERATE RAINFORESTS AND FOREST MANAGEMENT ARE NOT BASED ON BEST AVAILABLE SCIENCE

In addition to failing to analyze important information about the Tongass and its value for climate and carbon storage, the DEIS fails to analyze important information about the value of temperate rainforests.

- **Temperate rainforest amount reported in the DEIS is incorrect** – The DEIS grossly underestimates the global importance of coastal temperate rainforests, including the Tongass, for carbon regulation (0.5% global cover; no citation given).[76] DellaSala et al. (2011) provided the first computer generated map of all the world's temperate rainforests reporting that the total area for this rainforest biome is actually 2.5% of all forests globally (5 times that reported in the DEIS). The Pacific Coastal rainforests (California Coast Redwoods to Alaska) are globally significant as they represent over one-third of all temperate rainforests world-wide and because the Tongass is one of only 4 other relatively intact temperate rainforests (Great Bear – BC; Valdivia – Chile; Russian Far East; Southern Siberia). Thus, even though the overall global footprint of this rainforest biome is relatively small, the climate regulation properties of these forests – because of their enormous carbon stores – along with their myriad biodiversity and ecosystem benefits[77] – are globally significant and irreplaceable.[78] The Forest Service therefore has a national and global responsibility to maintain the intactness of this region and opening up roadless areas will have global ramifications contributing to the pace and scale of forest degradation globally. This is why 234 scientists signed a letter urging the Forest Service to protect the region's roadless areas (attached). The decision to open up roadless areas therefore is not based on best available science. At an absolute minimum, the Forest Service must correct its evaluation of the global importance of the Tongass's temperate rain forests and respond to these expert reports.

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[75] Campbell et al. 2007; McNicol et al. 2019

[76] DEIS at 3-122.

[77] Brandt et al. 2014.

[78] DellaSala et al. 2011.

- **Unsubstantiated claims are made about management activities approximating and promoting natural processes** – The Forest Service states, without a single citation, that logging and prescribed fire tend to approximate and promote natural processes and that such actions can result in long-term carbon uptake and storage that somehow increases resilience.[79] We note that prescribed fire is not even relevant on the Tongass rainforest and has no purpose in this DEIS. The statement overall also has no basis in the ecological literature, and certainly none for the Tongass's temperate rainforest, and seems to imply that forest degradation is a net gain in carbon and ecosystem processes even though the IPCC (2018, 2019) and numerous scientific studies indicate otherwise.[80] As discussed, the Forest Service needs to provide a reliable life cycle analysis and evidence-based review of the literature to back assertions that clearcut logging and road building somehow resemble natural disturbance processes – including effects on biodiversity (e.g., deer, wolves, murrelets and other old growth species). The statement, in fact, is reflective of old-school forestry ideologies long dismissed in the ecological literature and even by many foresters. Notably, given the lack of fire on the Tongass, primary disturbance agents are blow down from wind storms (canopy gap, stand, landscape level), landslides (watershed-landscape level), and tree mortality (stand level – canopy gaps – and watershed-landscape yellow cedar death from climate impacts). In no way do clearcuts, roads, mines, dams, etc. resemble any of these natural disturbances as natural disturbances leave prodigious amounts of biological legacies[81] that "life-boat" a forest through successional stages while these developments in old growth and IRAs will remove nearly all biological legacies. The long return interval of natural disturbances allows for old growth to develop over centuries, whereas, logged areas can be logged again in <100 years; this is insufficient time for forests to recoup carbon emitted from logging and to reach their maximum carbon potential.[82] We note that Public Law 113-291 (2014) allows up to 15,000 acres of young growth to be logged from 2016-2025 in stands < 95% CMAI and there is flexibility in NFMA to allow a continuation of harvesting at young ages beyond 2025 – thus, the carbon debt from re-logging these forests on a sustained yield basis is never recaptured and remains in the atmosphere for over a century at a time when we are in a climate emergency. The Forest Service needs to properly account for this carbon debt in the DEIS.

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[79] DEIS at 3-123.

[80] *See e.g.*, Harmon et al. 1990, Harmon et al. 1996, Mackey et al. 2014, Law et al. 2018, Moomaw 2019.

[81] DellaSala 2019; Buma et al. 2019.

[82] Moomaw 2019.

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Temperate and Boreal Rainforests of the World: Ecology and Conservation

Edited by

Dominick A. DellaSala

Geos Institute

Foreword by David Suzuki

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CHAPTER 1

Just What Are Temperate and Boreal Rainforests?

Dominick A. DellaSala, Paul Alaback, Toby Spribille, Henrik von Wehrden, and Richard S. Nauman

When most people think of rainforests, they think of lush, tropical "jungles" teeming with poison arrow frogs (*Dendrobates* spp.), toucans (e.g., *Ramphastos sulfuratus*), mountain gorillas (*Gorilla gorilla beringei*), and jaguars (*Panthera* spp.). Tropical rainforests are indeed special places, as they account for over half the terrestrial species on Earth (Meyers et al. 2000) while representing just 12 percent of the world's forest cover (Ritter 2008). Their temperate and boreal counterparts are another story, though, one yet to receive the kind of global recognition rightfully merited by tropical rainforests. Their story is told here, beginning with historical and recent accounts to define and map the temperate and boreal rainforests of the world.

Any discussion of rainforests must begin with what we mean by this term and how we map rainforests. Definitions and mapping standards are the mortar with which scientists visually construct biome delineations such as temperate and boreal rainforests. Consequently, the modeling techniques used in this chapter frame the entire book, as each of the regional chapters is built from the approaches set herein. In cases where it is necessary to deviate from globally based models and maps, explanations are given by regional authors of the book. Nevertheless, we now build on earlier approaches and definitions of temperate and boreal rainforests by providing a standardized modeling approach and a consistent methodology for mapping these rainforests. While it was our original intent that readers of this book would use our approach as the up-to-date standard for defining and delineating temperate and boreal rainforests, we note that

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this is a work-in-progress requiring further refinement and real-world verification as new data sets become available. Similarly, in Chapter 10, we present standardized mapping techniques aimed at determining just how much of this rainforest biome is in strict protection, a necessary step for developing a unifying vision for rainforests globally and for calling on decision makers to protect these rainforests as we do in Chapter 11. Because the process used in this opening chapter is central to the entire book, we put more emphasis here compared to the regional chapters that follow.

SCIENTIFIC HISTORY OF TEMPERATE AND BOREAL RAINFORESTS

Throughout this book we refer to either temperate or boreal rainforests that differ mainly with respect to latitude, climate, and plant associations. For descriptive purposes we separate these rainforest types in this chapter but refer to them jointly throughout much of the book.

Temperate Rainforests

Temperate rainforests have been recognized in some fashion by ecologists for nearly a century (Köppen 1918; Holderidge et al. 1971; Whittaker 1975; Jarmon and Brown 1983; Veblen 1985; Read and Hill 1985; Omernick 1987; Moore 1990; Hickey 1990; Alaback 1991; Kirk and Franklin 1992; Kellogg 1992, 1995; Gallant 1996; Lawford et al. 1996; Schoonmaker et al. 1997; Moen 1999). Most researchers classify them as distinct biomes based on broad differences in dominant vegetation and/or climate, or as inclusions within larger ecoregions (large areas distinguished by their dominant vegetation, climate, and land form). Yet a simple internet search for "temperate rainforest" yields inconsistencies in mapping locations due to gross differences in definitions and mapping techniques.

An earlier term, "high-latitude rainforest," was proposed by researchers to describe the pan-American portion of the biome (Lawford et al. 1996), since this is the most simple and unambiguous way to define *temperate* as contrasted with *tropical* (low-latitude) rainforests, but "high-latitude rainforests" has increasingly been replaced by "temperate rainforests," which generally have milder climates than boreal rainforests, due primarily to comparatively low latitudes. A number of temperate rainforest subtypes are described later in this chapter in order to distinguish rainforests from one another, and this terminology is used throughout this book.

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Boreal Rainforests

The border between boreal and temperate has traditionally been defined as the zone where conifer forests give way to deciduous forests, or, in drier regions, grasslands, roughly equated by Köppen (1918) with the –3°C January isotherm in the south (Tuhkanen 1984). The delineation of boreal versus temperate is blurred in montane regions, where temperate coniferous forest transitions seamlessly to boreal conifer forest. The important thing to note here is that *boreal* is a latitudinal zone and should not be conflated with terms such as *continental*; biogeographers are unanimous in recognizing some high-precipitation oceanic regions as part of the boreal zone. Tuhkanen (1984) compared a wide variety of different approaches to delineating the northern and southern limits of the boreal zone, and in the integrated classification he proposed that several of the rainforest regions treated here as "temperate" would be considered part of the boreal zone. Nonetheless, throughout this book, we use the term *boreal* to describe the cold northern rainforests of what in other studies have been more generally termed *subpolar*. As we will see later, these include the Pacific Coast of North America north of ~55°N latitude (chapter 2), the northern half of the inland rainforest of Northwestern North America (chapter 3), much of the wet forests of Eastern Canada (chapter 4), portions of Norway (chapter 6), and Inland Southern Siberia (chapter 9). Because there is no boreal zone in the Southern Hemisphere, relatively colder areas in this hemisphere are considered subpolar.

In reality, many temperate rainforests straddle the abiotic (nonliving chemical and physical factors) boundaries between temperate and boreal, both latitudinally and altitudinally, and more so for oceanic boreal systems. Thus, these rainforests serve as a phytogeographical bridge, facilitating the exchange of mesic (moist) floral elements among neighboring systems and as corridors of latitude- and slope- related south-to-north, north-to-south and slope-up, slope-down migrations of wildlife during periods of climate change. How much of the forests included in this book is boreal versus temperate depends on which classification system chosen. The fact that highly similar forest-species assemblages can be found on both sides of artificially drawn lines is a topic best reconciled to biogeography debates.

RAINFOREST DEFINITIONS

Where and how to draw the line between temperate and boreal rainforests has changed over time as more and better data have become available regarding these unique rainforests and the conditions that have created them. Several

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geographers who developed classifications for the world's climate included a category for temperate rainforest based, for instance, on some combination of cool temperatures and high rainfall, or cool temperatures and a small annual range of temperatures (see below). Whittaker (1975) in his classic ecology text *Communities and Ecosystems* also identified a temperate rainforest type. Most of these early efforts separated the Southern Hemisphere forests into a broadleaf evergreen forest type, further complicating a comprehensive global definition. These classifications vary widely in how they portray the distribution of temperate rainforests, and especially what types of temperate rainforests occur on Earth.

The prevailing definition of temperate rainforest began with work in the 1980s, when the environmental group Ecotrust and its collaborators proposed a more precise definition so that more accurate global maps and conservation strategies could be developed (Alaback 1991, 1996; Kellogg 1992, 1995). The first iteration of this work included a definition for these rainforests consisting of: (1) annual precipitation exceeding 1,200 millimeters with 10 percent or more occurring during summer months; (2) mean July temperature of 16°C or less; (3) cool dormant seasons; and (4) infrequent fire that is an unimportant evolutionary factor (Alaback 1991). Soon it became apparent that this definition was too restrictive, and more important, it did not accurately characterize availability of moisture, since there was no direct link between evaporation and the required minimum amount of rainfall. The most biophysically precise method of doing this would be to calculate potential evapotranspiration, which corrects for latitude—with increasing latitude, less precipitation is required to maintain the same humidity levels (Stephenson 1990). Potential evapotranspiration was also later shown to precisely predict the distribution of at least one common rainforest tree in northwestern North America, western hemlock (*Tsuga heterophylla*), even including its distribution in interior rainforests of northwestern North America (Gavin and Hu 2006). In the absence of detailed models and global spatial coverages, a more inclusive definition was proffered by Alaback (1996). In this case, temperate rainforests meeting the original criteria for annual rainfall were divided into four subtypes (or zones, including boreal), analogous to subtypes of tropical forests, based on seasonality of precipitation and annual temperatures:

- *Subpolar*—summer rainfall is above 20 percent of the annual total, summers are cool, and snow is persistent in winter, with mean annual temperature below 4°C.
- *Perhumid*—summer rainfall is above 10 percent of the annual total, summers are cool, and typically transient snow is present in winter, with

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mean annual temperature of 7°C. "Cool-temperate" also has been used in this context.

- *Seasonal*—summer droughts and fires can periodically occur, summer rainfall is less than 10 percent of the annual total, with mean annual temperature of 10°C.
- *Warm-temperate*—summer precipitation is less than 5 percent of the annual total, winter snow is rare, drought can occur during any season, and mean annual temperature is 12°C or above (Alaback 1996; Veblen and Alaback 1996; Alaback and Pojar 1997).

The threshold values of temperature and precipitation for each of the forest subtypes was determined by examining climatic conditions in areas along the west coast of North and South America that possessed key ecological characteristics associated with rainforests. This has been the prevailing set of definitional parameters for describing rainforest regions used throughout the chapters of this book.

A NEW GLOBAL RAINFOREST MODEL

Building on concepts from Alaback (1991), we developed a strongly organism/ecosystem–driven model for temperate and boreal rainforests that has identified a very small amount of land surface of the earth within the same biome and sharing climatic characteristics and associated ecological processes that rightfully and generally can be called temperate and boreal rainforest. The processes described herein build on earlier work of rainforest ecologists by providing a broad suite of climatic criteria and a standardized approach to mapping rainforests globally.

In this chapter, we use computer modeling to develop defensible criteria for identifying temperate and boreal rainforests and to locate forests not widely recognized as rainforest but meeting our criteria. Further, we create a computer model with high-resolution climate data and compare it to maps created by regional experts.

Rainforest Distribution Model

This book's chapter authors, from a wide range of rainforest regions, provided locations of sites they considered typical of temperate or boreal rainforest in their area. Based on this input, we used climate data for 117 localities from six regions for the initial modeling step: the Pacific Coast of North America

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(n = 55, mostly coastal); Chile and Argentina (n = 9); New Zealand (n = 10); Tasmania (n = 6); Norway (n = 15); and Japan (n = 22). These regions were selected because we had localities from collaborators, and because there was little dispute that the locations represent rainforests (especially the Pacific Coast of North America, Chile, and New Zealand). Baseline predictors were extrapolated from a global climate data set (Hijmans et al. 2005); redundancy in the model variables was reduced based on a principal-components analysis of the complete data set. The final model was constructed using a MaxEnt modeling approach (Phillips et al. 2006), consisting only of predictors that improved the model. This yielded 11 discrete climate-related parameters. We used the MaxEnt model since it is known to be more conservative compared to other presence-only models, which tend to overestimate occurrence of a particular variable of interest (in this case, temperate and boreal rainforest).

The model was evaluated with a bootstrapping method (Burnham and Anderson 2002), resulting in strong support of the predictive ability of the model (AUC = 0.90; values less than 0.5 indicate no predictive capabilities; see Phillips et al. 2006). Based on 100 repeated runs, we quantified the heterogeneity of the ground-truth climate data set, thus ensuring a demarcation of core zones with a high probability of rainforest occurrence in comparison to areas with a lower probability (for mapping simplicity, only high-probability areas were depicted).

The rainforest distribution model generated four additional regions with climate suitable for temperate and boreal rainforests: the Inland Northwest of North America (figure 1-1, middle-right portion of panel a—inland British Columbia), Eastern Canada (figure 1-1, panel b), Great Britain and Ireland (figure 1-1, western corner of panel d), and portions of the Alps (figure 1-1, lower middle of panel d). Notably, two of these regions have not been widely recognized as rainforest by scientists, including the wettest parts of Eastern Canada, which appeared in some form in all map iterations, and some valleys of the eastern Alps, in particular the Salzburg Alps and mountain ranges of western Slovenia. Interestingly, these regions support rainforest lichen assemblages remarkably similar to those of the Pacific Northwest of North America or coastal Norway.

Two lower-latitude regions often considered rainforest by some (e.g., Kellogg 1992), such as the Colchic (Georgia) and Hyrcanic (Iran) forests of the Western Eurasian Caucasus, and the forests of the southern cape of South Africa, were shown to be in a class of their own compared to the more definitive rainforests of the Pacific Coast of North America and Valdivia. Including these warmer and drier outliers in the model calibration invariably resulted in overestimating the global extent of these rainforests by also including South

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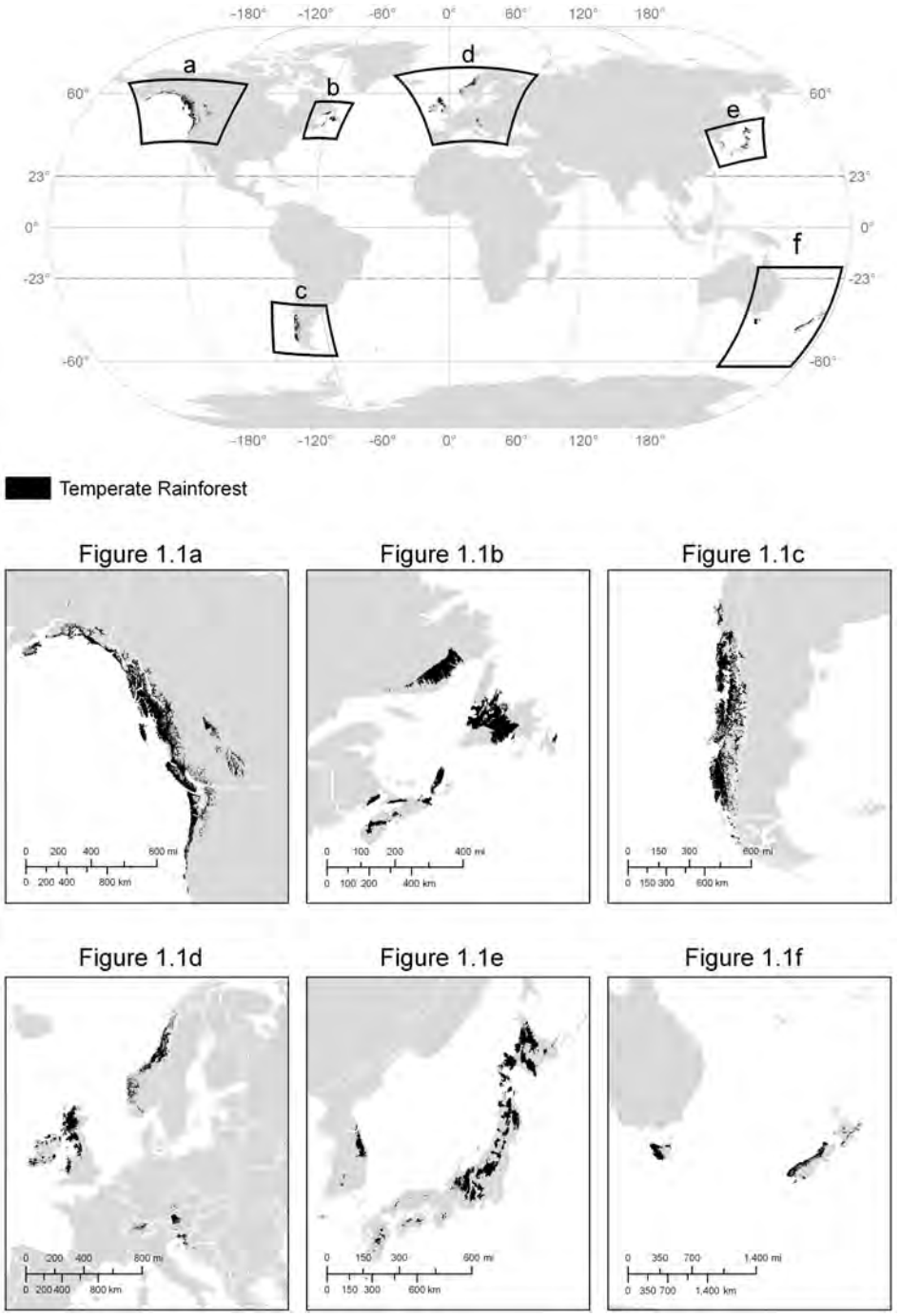

Figure 1-1. Temperate and boreal rainforests of the world based on the global rainforest distribution model, including: (a) Pacific Coast and Inland Northwestern North America; (b) Eastern Canada; (c) Chile and Argentina; (d) Europe; (e) Japan and Korea; and (f) Australasia.

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American páramo, high-elevation African equatorial fog forests, and nearly half of the Alps. Retention of the eastern Black Sea region (Colchic), in particular, resulted in model inclusion of large areas of eastern North America, parts of which are indeed climatically similar, but did not agree with our initial criteria on several counts. We settled on a conservative definition of temperate and boreal rainforest based generally on the climate data (see table 1-1; figures 1-2, 1-3) presented for nine regions (some were combined from the set above) as follows:

- Annual (minimum, maximum) temperatures from ~4 to 12°C.
- Annual (minimum, maximum) precipitation from 846 to 5,600 millimeters.
- Snowy winters in high latitudes.
- Significant precipitation (that is, up to 25 percent of annual precipitation) during the driest quarter.
- Low annual temperature fluctuation (based on low annual temperature variability).
- Temperature of warmest quarter (summer) from 7 to 23°C.

This is the first time a spatially explicit global data set was made available for the world's temperate and boreal rainforests that was based on a suite of climate variables obtained from a global data set (available in raster—or grid—GIS format), improvements in computer processing capacity, and statistical models. The model therefore represents an initial cut at producing a global rainforest map, requiring further refinements through the use of regional climate data sets, regional rainforest classifications, and regional maps. Notably, while the minimum precipitation and maximum temperature values reported seem extreme in comparison to earlier rainforest definitions, rainforest communities persist in these regions due to compensatory factors as discussed below and in the regional chapters of this book. This is why regional ground-truth of the model and further study of rainforest classifications are essential.

CLIMATIC PATTERNS OF TEMPERATE AND BOREAL RAINFORESTS

Based on the rainforest distribution model, rainforests were clustered along precipitation and temperature gradients that distinguished them from one another and other forest types.

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Table 1-1. Abiotic conditions (climate, altitude) of temperate and boreal rainforests used in the rainforest distribution model, based on a global climate data set.[a]

Model Variable[b]		Norway	Great Britain	Inland Northwest of North America	Eastern Canada	Alps	Pacific Coast of North America	Japan and Korea	Chile and Argentina	Australasia
Mean Diurnal Range (mean of monthly; max temp–min temp; °C)	Min	3.9	5.4	9.2	7.0	6.3	4.3	5.8	4.2	6.1
	Max	7.3	7.6	12.1	11.9	10.4	11.9	11.1	12.1	11.4
	Mean	6.1	6.6	10.8	8.9	8.9	7.6	8.6	7.2	8.9
Isothermality (diurnal temperature/annual range of temperature, °C)	Min	24.1	30.4	27.6	22.3	27.8	24.9	19.2	42.8	44.1
	Max	31.2	42.2	33.2	32.3	34.6	63.6	31.9	60.1	52.2
	Mean	27.9	36.4	30.2	25.9	31.8	33.4	25.1	48.3	47.6
Mean Temperature of Driest Quarter (°C)	Min	–0.9	5.5	–0.1	–11.1	–4.8	–7.0	–14.8	1.2	–1.0
	Max	9.3	13.6	15.2	15.0	15.1	17.9	10.5	16.2	15.2
	Mean	5.4	9.3	3.2	1.7	0.4	9.9	–2.6	9.9	10.0
Mean Temperature of Warmest Quarter (°C)	Min	10.3	10.3	10.8	10.4	10.9	9.9	10.6	6.8	6.8
	Max	14.8	15.2	17.9	18.7	19.5	17.9	22.8	16.3	17.8
	Mean	11.5	12.9	13.5	13.6	15.4	12.5	17.9	11.9	13.2
Precipitation of Driest Quarter (mm)	Min	124	169	106	162	188	6	97	74	143
	Max	447	352	198	399	317	619	487	1378	1169
	Mean	262	248	152	248	245	252	233	499	458

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Table 1-1. Continued

Model Variable[b]		Norway	Great Britain	Inland Northwest of North America	Eastern Canada	Alps	Pacific Coast of North America	Japan and Korea	Chile and Argentina	Australasia
Precipitation of Coldest Quarter (mm)	Min	210	244	187	209	209	125	97	230	241
	Max	859	697	458	439	388	1473	506	1403	1803
	Mean	428	431	267	309	269	642	263	801	613
Altitude (m)	Min	25	74	356	86	284	0	69	0	4
	Max	754	842	1487	1319	1548	1263	2612	1066	2334
	Mean	327	314	973	267	826	364	811	278	580

[a]See Hijmans et al. 2005.

[b]Annual mean temperature, temperature annual range, annual precipitation, and precipitation seasonality (coefficient of variation) were included in the model but not in this table as the data are summarized in figures 1-2 and 1-3.

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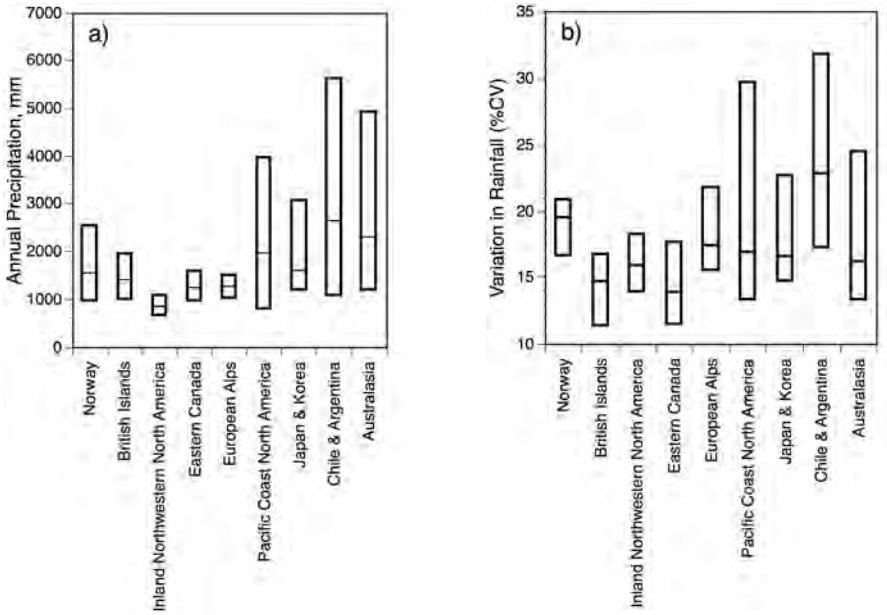

Figure 1-2. Annual precipitation (a) and variation in rainfall (b) of definitive temperate and boreal rainforests, based on a global climate data set (Hijmans et al. 2005) and the rainforest distribution model.

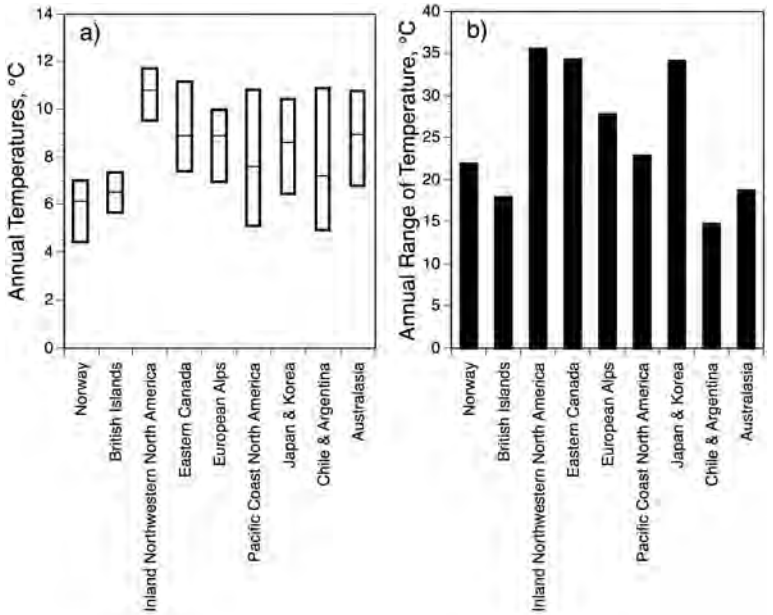

Figure 1-3. Annual temperature (a) and annual range of temperature (b) of definitive temperate and boreal rainforests, based on a global climate data set (Hijmans et al. 2005) and the rainforest distribution model.

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Precipitation Gradient

A broad range of annual rainfall amounts occurs in the "classic" temperate rainforests of the Pacific Coast of North America, Chile and Argentina, and Australasia (see figure 1-2a). As in tropical rainforests, seasonality of precipitation is a key element of rainforest climate that can influence rates of decomposition, the roles of fire, drought, epiphytes, and species composition (Alaback 1996; Losos and Leigh 2004). Just looking at the coefficient of variation of monthly precipitation shows the greatest range in the rainforest regions with the greatest latitudinal ranges (e.g., the Pacific Coast of North America, Chile and Argentina) and also the greatest seasonality, but a less clear pattern in the seasonality of precipitation in smaller regions (see figure 1-2b). More work is needed to clarify how seasonality of precipitation helps effect such differences among rainforest regions.

Temperature Gradient

Based on the global climate data set, Norway had the coolest annual temperature and Inland Northwestern North America (based on southern locales) the warmest (see figure 1-3a). Notably, climate data sets derived from a global reference (Hijmans et al. 2005) may differ from data sets presented in the regional chapters due, for instance, to topographical influences on local climate and the location and density of weather stations.

The comparatively wide range of annual temperatures on the Pacific Coast of North America and in the Valdivian temperate rainforest reflects both its broad latitudinal distribution and a large range in climates from boreal and subpolar to nearly subtropical. Similarly, the Japanese archipelago spans many climate types (alpine to subtropical, and continental to oceanic), with rainforests distributed zonally.

The annual range of temperature provides a good measure of seasonality of a given region (see figure 1-3b). The regions with the greatest influence from interior climates, such as Inland Northwestern North America, Eastern Canada, Japan, and Korea, all clearly show this influence. The more oceanic climates, such as Norway, the British Islands, and the Southern Hemisphere rainforests, by contrast show a much smaller range of monthly temperatures. This also helps explain why some of the forests in these regions can develop rainforest characteristics with less rainfall than in comparable continental regions.

In sum, rainforests can be grouped both by differences in annual temperature and annual precipitation, with the Inland Northwest of North America the warmest, driest rainforest globally, Norway the coolest (with moderate precipi-

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tation), and Chile, Argentina, and Australasia the wettest, with relatively cool-to-moderate temperatures.

OUTLIERS AND OTHER CONSIDERATIONS

The rainforest distribution model did not predict some areas as rainforest which, upon further inspection, showed signs of rainforest conditions or communities. We chose to include some of these as "rainforests at the margins" (or outliers), based on input from regional scientists specializing in the specific regions (see chapter 9). For instance, in some places rainforest communities can persist at precipitation levels lower than the range used in the model as long as there is enough moisture at critical times of the year (e.g., warm summer months) to support moisture-loving species such as lichens and mosses, either directly through some rainfall or indirectly through compensatory mechanisms (e.g., low evapotranspiration rates, high humidity, cool summer nighttime temperatures, and fog). Evidence for this exists for the Knysna-Tsitsikamma forests of South Africa and the Colchic and Hyrcanic forests of the Western Eurasian Caucasus, where persistent fog and high humidity compensate for low summer precipitation and/or hot summers (chapter 9). Such conditions prove suitable for oceanic lichens and humidity-dependent vegetation. The Ussuri taiga of the Russian Far East and the Sayani Mountains of Inland Southern Siberia were too dry for inclusion in the model but have relatively low temperatures and high humidity (chapter 9). Low evaporative losses apparently compensate for drier conditions, allowing humidity-dependent forests to flourish.

The rainforest distribution model also did not identify rainforest in some areas previously suspected to be rainforest. For instance, while Taiwanese montane forests receive sufficient rainfall and cool-enough temperatures zonally (at high elevations) to be considered "temperate rainforest" by some (see Wikipedia[1]; also see Farjon 2005), the lack of a well-defined cool dormant season makes them more ecologically equivalent to cloud or subtropical forests (Alaback 1991), and thus we did not give further consideration to these forests. Iceland's scant boreal forests, though recognized as rainforest by Kellogg (1992), were not included in our rainforest model because the mean annual temperature is below even the minimum used to define rainforests. Icelandic forests also lack the structural complexity associated with temperate and boreal rainforests,

[1] www.en.wikipedia.org/wiki/Temperate_rain_forest

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such as well-defined canopy layers and gap-phase disturbance dynamics, as trees usually are not long-lived or productive due to severe weather. There are no naturally occurring boreonemoral tree species (see chapter 6) such as elms (*Ulmus* sp.) and oaks (*Quercus* sp.), and there are few of the rainforest lichens common to Norway's rainforests (e.g., *Biatora toensbergii*, *Fuscopannaria ahlneri*, and *Lobaria hallii*).

Although the Appalachian mixed-mesophytic forest of the southeastern United States has been recognized as temperate rainforests by some (see Netencyclo.com[2]; Shanks 1954; see chapter 4), it was not predicted by the rainforest distribution model, presumably because the region has relatively high year-round temperatures and dry summers. However, because there was evidence of rainforest conditions at high elevations (moist pockets of spruce-fir within the larger ecoregion), we briefly mentioned them as a southerly extension of Appalachian boreal rainforests from Eastern Canada that require further study (see chapter 4). In sum, we hope the techniques used here will inspire additional research into these areas in order to further refine our approach.

INTRODUCING TEMPERATE AND BOREAL RAINFORESTS

In the following sections of this book, we discuss seven definitive regions (some regions from above were combined) identified by the model and three outlier regions that collectively make up the global network of temperate and boreal rainforests.[3] We generally organized regions north to south (Western Hemisphere) and west to east (Eastern Hemisphere), as presented sequentially as the book's regional chapters.

Definitive Regions

- Pacific Coast of North America (chapter 2)
- Inland Northwestern North America (chapter 3)
- Eastern Canada (chapter 4)
- Chile and Argentina (chapter 5)
- Europe: Norway, Ireland, Great Britain, portions of the Alps, the Bohemian region, and the Balkans (chapter 6)
- Japan (chapter 7—note that Korea was included in the Russian Far East and Inland Southern Siberia profile, based on author expertise)
- Australasia: Australia, Tasmania, and New Zealand (chapter 8)

[2] www.netencyclo.com/en/Temperate_rain_forest

[3] Maps available at www.databasin.org

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Outliers (chapter 9)

- Western Eurasia Caucasus (Colchic and Hyrcanic forests)
- Russian Far East and Inland Southern Siberia
- South Africa (Knysna-Tsitsikamma forests)

REGIONAL VS. RAINFOREST DISTRIBUTION MAPS

While the global model was useful in predicting general locations of temperate and boreal rainforests, we often found differences in global projections versus regional delineations made by local experts (see table 1-2). Thus, comparing predicted distributions with regional maps was necessary to ensure that an agreed-upon set of maps was used in the regional chapters. Digital maps for this step were obtained for the Pacific Coast of North America (Kellogg 1995; see figure 4), Inland Northwestern North America (Craighead and Cross 2007), Eastern Canada (described below), Chile and Argentina (provided by Patricio Pliscoff—see below), Australasia (Kirkpatrick and Dickerson 1984), Japan (Miyawaki et al. 1980–89), and Norway (described below). Here, we describe the differences in mapping delineations and reasons for including regional maps, where we had them, in the chapters of the book that follow.

Pacific Coast of North America

Differences in mapping estimates between the global model and regional mapping (Kellogg 1995) were fairly minor (see table 1-2; figure 1-4). The rainforest distribution model yielded a rainforest estimate that was ~9 percent higher than regional mapping (see table 1-2). We present the map by Kellogg (1995) in Chapter 2 because it allowed us to base conservation priorities on regionally specific zones (finer scale) that were not apparent from the coarser rainforest distribution model.

Inland Northwestern North America

The model predicted rainforest to occur on nearly 2.2 million hectares, but only for eastern British Columbia (see figure 1-5; table 1-2). In comparison, using the distribution of western red cedar (*Thuja plicata*) and western hemlock (*Tsuga heterophylla*) (i.e., Interior Cedar Hemlock forests) yielded over 3 times the amount of rainforest at 7.3 million hectares (Craighead and Cross 2007; see chapter 3), with nearly equal amounts in British Columbia and the United States. While the rainforest distribution model and the vegetation-based map showed strong agreement in British Columbia, the Interior Cedar Hemlock

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Table 1-2. Global (rainforest distribution model, Kellogg 1992) and regional (based on digital maps from published sources) estimates for temperate and boreal rainforests.

Region	Rainforest distribution model (ha)	(%)	Regionally based estimates[a] (ha)	Kellogg (1992) (ha)	Kellogg (1992) (%)
Pacific Coast of North America[b]	27,274,225	35.0	25,097,930	20,726,700	50.3
Inland Northwestern North America					
British Columbia	2,179,733	2.8	3,879,730		
United States	0	0.0	3,366,874		
Total Inland Northwestern North America	2,179,733	2.8	7,246,604		
Eastern Canada	5,969,641	7.7	6,085,063		
Valdivia					
Chile	12,211,573	15.7	9,752,451	11,675,100	28.4
Argentina	348,371	0.4	2,211,888	323,300	0.79
Total Valdivia	12,559,944	16.1	11,964,339	11,998,400	29.1
European Relicts					
Iceland				195,200	0.47
Norway	4,887,739	6.3	3,747,090	1,459,000	3.5
Great Britain	5,064,759	6.5		1,149,300	2.8
Ireland/Republic of Ireland	1,578,545	2.0		157,300	0.38
Northeast Alps and Swiss Prealps	745,915	1.0			
Bohemia	220,199	0.3			
Southeastern Alps and Northwest Balkans	577,425	0.7			
Total European relicts	13,074,582	16.8		2,960,800	7.2
Japan and Korea	8,295,241	10.6	2,404,404		
Australasia					
Australia	55,989	0.07	1,652,933		
New Zealand	5,458,170	7.0	4,969,590	4,040,400	9.8
Tasmania	3,132,684	4.0	692,300	551,700	1.3
Total Australasia	8,646,843	11.1	7,314,823	4,592,100	11.2
Total Rainforest	78,000,209	1.95[c]		41,177,500	1.1
Outliers[d]					
South Africa (Knysna-Tsitsikamma)	235,483	1.2			

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Table 1-2. Continued

Region	Rainforest distribution model (ha)	(%)	Regionally based estimates[a] (ha)	Kellogg (1992) (ha)	Kellogg (1992) (%)
Western Eurasia					
Hyrcanic	1,960,000	10.3			
Colchic[e]	3,000,000	15.8		899,500	2.2
Total Western Eurasia	4,960,000	26.1			
Russia/Siberia					
Russian Far East	6,800,000	35.8			
Inland Southern Siberia	7,000,000	36.9			
Total Russia/Siberia	13,800,000	72.6			
Total Outliers	18,995,483	0.47[c]			
Combined temperate and boreal rainforest total	96,995,692	2.42[c]			

[a]Regional estimates were provided for comparisons to the rainforest distribution model but, due to differences in mapping methodologies, did not include percentages except in the case of Kellogg (1992), which was based on more consistent mapping methodologies.

[b]Differences in rainforest estimates between the two Kellogg references (1992, 1995) are presumed due to refinements in mapping techniques, mainly the addition of the western Cascades in Washington and Oregon, which were not included in the original maps.

[c]Percentages were derived from global forest cover (all forest types) estimated at 4 billion hectares based on FAO (2005) estimates that define forests as >10% tree cover. Plantations are included in estimates.

[d]Outlier estimates, provided by regional authors, were derived from different mapping methodologies not directly comparable to rainforest distribution estimates or other regional estimates.

[e]Kellogg (1992) lists this region as Eastern Black Sea (Turkey, Georgia).

map extends this rainforest type southward for roughly 430 kilometers into northeastern Washington, northern Idaho, and northwestern Montana (see figure 1-5a). Based on local knowledge, we choose the map of Interior Cedar Hemlock forests for Chapter 3.

Eastern Canada

For this region, we overlaid the Thornthwaite (1948) index for perhumid regions (100+ moisture index) onto digital layers of vegetation obtained from coniferous and mixed forest types (source: Canadian Vegetation and Land Cover data set, www.nrcan.gc.ca). This shapefile is based on satellite data obtained in 1995 by the Advance Very High Resolution Radiometer

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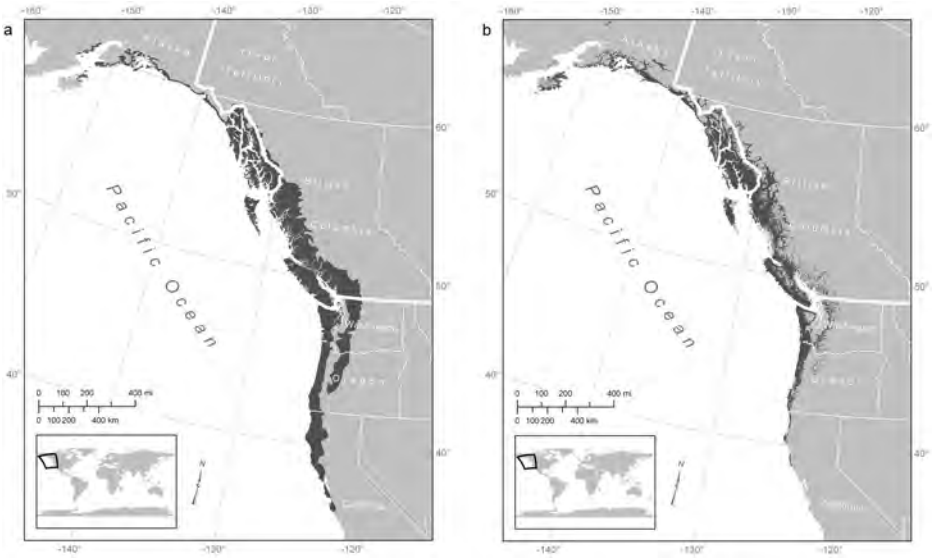

Figure 1-4. Temperate and boreal rainforests of the Pacific Coast of North America based on (a) regional mapping (Ecotrust 1995) and (b) the rainforest distribution model.

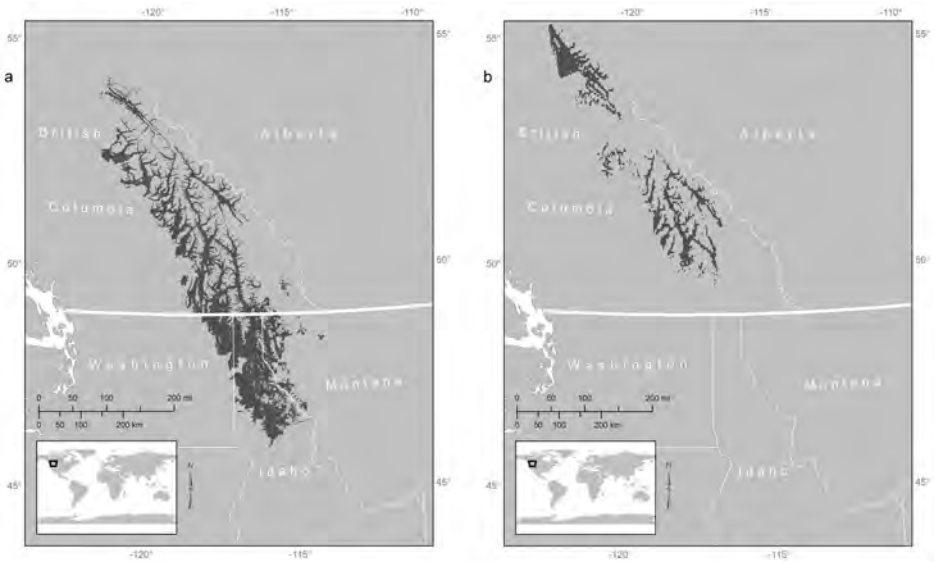

Figure 1-5. Temperate and boreal rainforests of Inland Northwestern North America based on (a) regional mapping (Craighead and Cross 2007) and (b) the rainforest distribution model.

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(AVHRR) on board the NOAA-14 (National Oceanic and Atmospheric Administration) satellite. We assumed these forest types were most likely to include important lichen assemblages and rainforest structure that matched perhumid climatic conditions in the region.

Both the rainforest distribution model and regional map (Thornthwaite 1948) yielded nearly identical area estimates (see table 1-2). However, predicted locations of rainforests from the rainforest distribution model vs. regional mapping differed appreciably (see figure 1-6). Thus, we used the regional map in

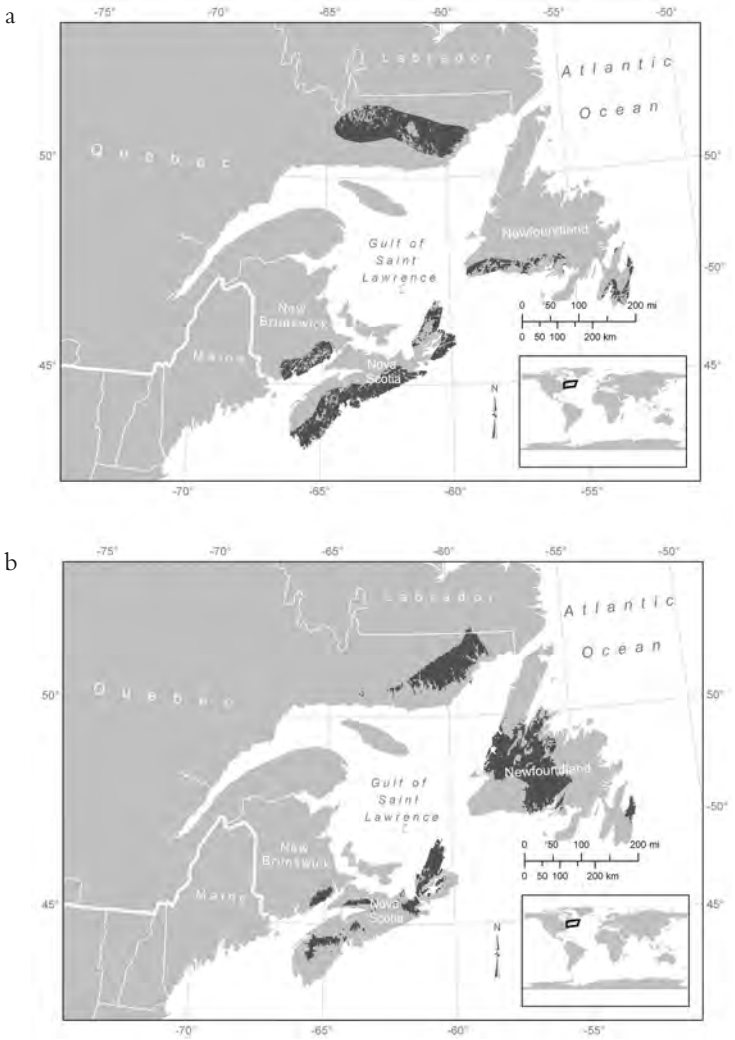

Figure 1-6. Perhumid boreal and hemiboreal rainforests of Eastern Canada based (a) on regional mapping (modified from Thornthwaite 1948) and (b) the rainforest distribution model.

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Chapter 4 because it was thought to have higher predictability and greater concordance with forests supporting rainforest lichen assemblages based on local knowledge.

Chile and Argentina

The primary map source for Chile was the national vegetation survey. This was originally produced using aerial photography at a scale of 1:50,000 and with varied level of verification on the ground. Later updates to this information were produced using Landsat imagery and essentially serve to track loss of forest cover. As a representation of forest cover, the national vegetation survey is widely used in Chile, is embraced the official source by Chile's Native Forest Law of 2008, and is fairly reliable. For Argentina no such forest survey exists; thus we used the same criteria and methods from Chile's national survey and a series of aerial photos to produce a forest-cover map at 1:500,000 scale without ground verification.

The rainforest distribution model and regional map yielded similar area estimates for Valdivia (see table 1-2; figure 1-7). However, there were significant differences in rainforest locations, with the rainforest distribution model extending farther south into the Magellanic (subpolar) rainforests, considered a separate ecoregion by Chilean scientists (see chapter 5), but missing important

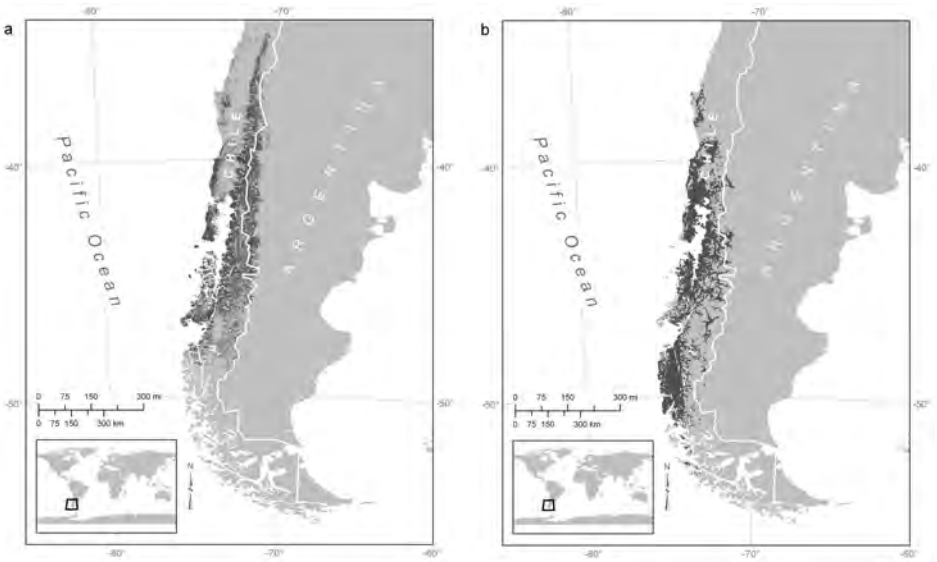

Figure 1-7. Valdivian temperate rainforests of Chile and Argentina based on (a) regional mapping (digitized from national vegetation surveys) and (b) the rainforest distribution model.

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rainforest locations in the north and in Argentina. Notably, because the Magellanic forests can be considered rainforest by the standards set forth herein, regional authors included some mention of them in Chapter 5. We used the regional map in Chapter 5 because it is widely accepted in regional conservation planning.

Europe

Norway was the only regional map available for comparisons to the rainforest distribution model in Europe. The regional map in this case was based solely on floristic data, namely distribution of epiphytic lichens housed at the Norwegian Lichen Database.[4] Notably, the core area of boreal rainforest in Norway (and Europe) is rather well outlined by the distribution of just two lichens—*Rinodina disjuncta* and *Pyrrhospora (Lecidea) subcinnabarina*—also known from the Pacific Coast of North America (see Tønsberg 1992, 1993; Sheard 1995). The distribution of three other lichens demark the northern and southern limits, with *Lobaria hallii* delimiting boreal forests with occurrences in ravines and by waterfalls, and *Leptogium burgessii* and *Pyrenula occidentalis* the southern boreonemoral (temperate) rainforests.

The rainforest distribution map of Norway estimated about 1.1 million hectares (~30 percent) more rainforest than the estimate generated by regional authors (see table 1-2; figure 1-8). In this case, the rainforest distribution model may have correctly predicted conditions suitable for rainforests but local differences in soils, wind exposure, or human disturbance may preclude rainforest development. Therefore, the Norway regional map was used because it was prepared with regional forest inventories based on known rainforest lichen assemblages (see chapter 6).

Japan

About 5.9 million hectares (over 3 times) more rainforest was estimated by the rainforest distribution model compared to a digitized map of Japan's rainforest zones (see table 1-2); figure 1-9), which were based on finer-scale mapping and therefore used in Chapter 7.

Australasia

About 1.3 million hectares (18 percent, table 1-2) more rainforest was predicted by the rainforest distribution model compared to regional mapping (see figure 1-10). Differences were greatest for Tasmania, where the rainforest

[4] www.nhm.uio.no/botanisk/lav/index.html

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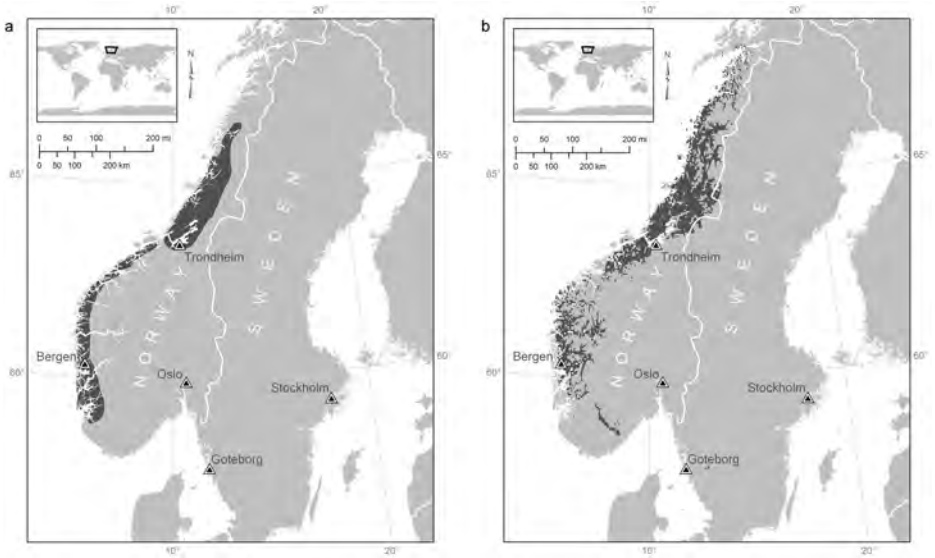

Figure 1-8. Boreal and boreonemoral rainforests of Norway based on (a) regional mapping (derived from lichen distribution maps) and (b) the rainforest distribution model.

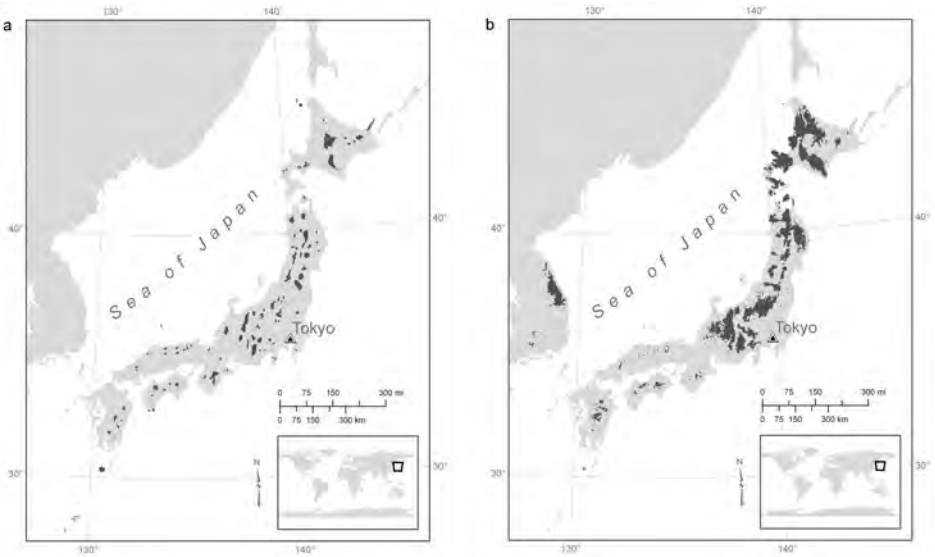

Figure 1-9. Temperate rainforests of Japan based on (a) regional mapping (Miyawaki et al. 1980–1989) and (b) the rainforest distribution model.

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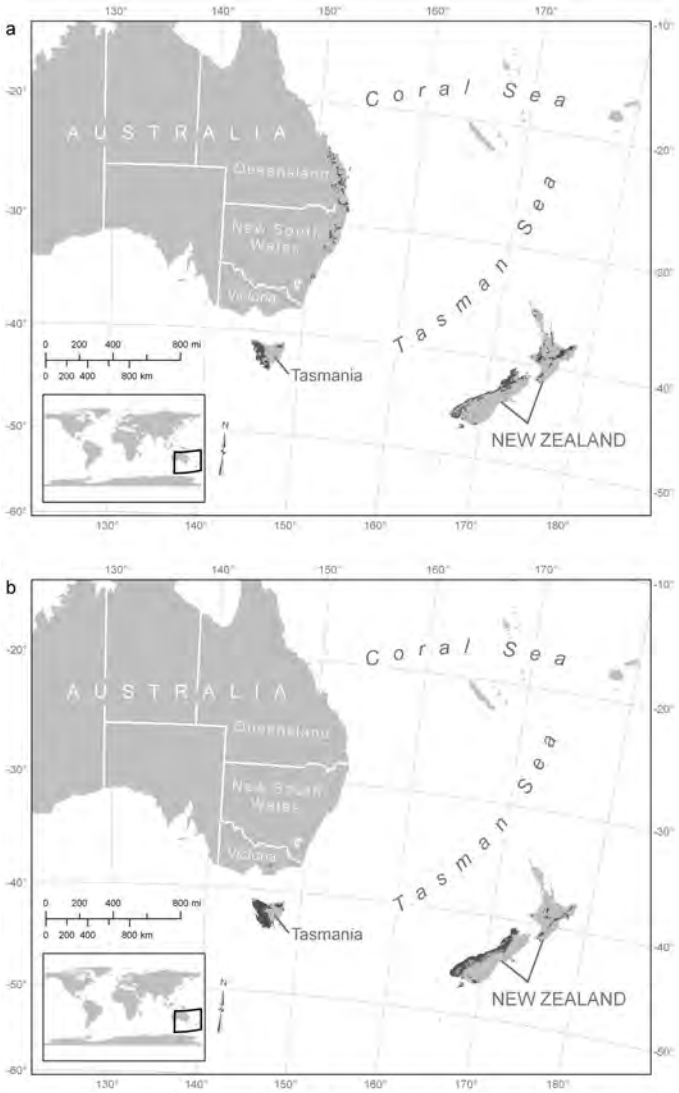

Figure 1-10. Temperate rainforests of Australasia based on (a) regional mapping (Kirkpatrick and Dickerson 1984) and (b) the rainforest distribution model.

distribution model estimated about 2.4 million hectares (over 4 times) more rainforest than the regional map. Conversely, the rainforest distribution map estimated about 1.6 million hectares less rainforest along the Australian coastline (New South Wales). Notably, about 151,173 hectares and 830,769 hectares of the regionally based totals (Kirkpatrick and Dickerson 1984) were classified as clear felled or forests patchily distributed, respectively, at the time. So the overestimate of rainforest by the model may have been partially compensated by the

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mapping of cleared forests by regional experts. Because the regional maps included more of the Australian coastline where rainforests are known to occur, they were used in Chapter 8.

In sum, the rainforest distribution model was useful in establishing an objective upper range of potential rainforest, was the only standardized data set available for comparisons among regions, and provided a reliable global rainforest total. However, the model had a tendency to overestimate rainforest extent in most, but not all, regions when compared to site-specific mapping and regional expertise. The rainforest distribution model was potentially confounded by human disturbance and local site conditions. Rainforest estimates derived from regional maps, however, also have limitations, as they cannot be compared among regions due to differences in mapping techniques, data sources, and mapping scales. Thus, in making relative comparisons among regions and predicting new localities, the global rainforest distribution model performs quite well; however, for regional specificity we relied on regional maps, as they had a higher degree of reliability at that scale. Follow-up mapping assessments and modeling is recommended in both cases—regionally and globally—to improve rainforest estimates and mapping techniques.

TEMPERATE AND BOREAL RAINFOREST TOTALS

Based on the rainforest distribution model, the Pacific Coast of North America (British Columbia and the United States combined) by far contains the most expansive temperate and boreal rainforests globally, representing over one-third of the world's totals (see table 1-2). Our estimate for this region is notably less than prior estimates (50 percent). Differences are due largely to rainforest areas added in the rainforest distribution model and different mapping techniques, which obviously affected regional totals. Nonetheless, in decreasing order, rainforest extent was then highest for European rainforest relicts (disjunctly distributed); Chile and Argentina; Australasia; Japan; Eastern Canada; and Inland Northwestern North America. However, these percentages do not indicate intactness of rainforests within a given region. For instance, some of the last remaining large blocks of temperate rainforests in the world occur in Valdivia, Tasmania, and New Zealand (see chapters 5 and 8), in comparison to highly fragmented European relicts (see chapter 6); and some of the most intact old-growth rainforests occur in the British Columbia and Alaska (see chapter 2). However, regional totals are not affected by conservation status.

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In addition to definitive regions, outliers added nearly 19 million hectares to the global temperate and boreal rainforest total (roughly 0.5 percent), with the Russian Far East and Inland Southern Siberia by far containing the largest (73 percent) expanse and South Africa the smallest (~1 percent, table 1-2).

In sum, our estimate for global temperate and boreal rainforest extent (2.42 percent) was more than twice that of previous estimates (1.1 percent; Kellogg 1992), due largely to additional regions estimated by the rainforest distribution model and differences in mapping techniques. However, some regions (Iceland) previously considered rainforest (Kellogg 1992) were not included here as they do not appear to support rainforest communities. Nonetheless, despite these differences there was considerable overlap in regional estimates, with the net result that temperate and boreal rainforests still represent just a fraction of the global forest cover.

RAINFOREST DIFFERENCES IN THE NORTHERN AND SOUTHERN HEMISPHERES

In this section, we examine major differences in gross rainforest characteristics that can be readily grouped by differences in biogeography between hemispheres where these rainforests are found.

Northern Hemisphere

Temperate and boreal rainforests in the Northern Hemisphere are remarkably similar in species composition, at least at the genus level. The largest of these rainforests in terms of areal extent are dominated by conifers (e.g., Pacific Coastal and Inland Northwest North America, parts of Japan, Norway), usually broadly distributed but closely related species of the pine family, including hemlock, true firs (*Abies* spp.), Douglas-fir (*Pseudotsuga menzeisii*), spruce (*Picea* spp.) or pine (*Pinus spp.*), and species of Cupressaceae, especially red cedars (*Thuja* spp.). Other, smaller regions are dominated especially by beeches (*Fagus* spp.; found in Japan and central European fragments) or beech-spruce mixtures (found, for example, in Norway). In general, temperate and boreal rainforests of the Northern Hemisphere have a dense understory of largely deciduous woody shrubs, a variety of widely distributed (often circumboreal) herbaceous plants and a thick mat of bryophytes (mosses and liverworts), lichens, and many fern species. The broad commonalities among these rainforests make sense from a biogeographical standpoint, since the floras of the Northern

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Hemisphere are believed to have been derived in large part from common Tertiary ancestors 60–80 million years ago (see Axelrod 1976).

Southern Hemisphere

In Southern Hemisphere rainforests (southern Chile, Argentina, New Zealand, Tasmania and nearby areas), most trees are broad-leaved evergreens, which form a patchy canopy with many layers beneath the dominant overstory, including a broad diversity of both evergreen and deciduous trees and shrubs. The trees are tall and dense, with small tough leaves (Veblen et al. 1996).

Southern vs. Northern Hemisphere

Southern Hemisphere trees are unlike most of the familiar broad-leaved trees in the North. The "southern beech" or *Nothofagus* trees, for example, are not closely related to beeches of the Northern Hemisphere. They are in their own family (Nothofagaceae) and originated in ancient Gondwana before it split into what have become the small areas of temperate rainforest scattered across the Southern Hemisphere (Veblen et al. 1996). This explains why there are many species of trees that are shared at least at the genus level among rainforests in New Zealand, South America, and Australia (Ezcurra et al. 2008). Another big surprise is in the pine family. While pines, spruces, firs, and related species dominate high-latitude forests of the Northern Hemisphere, this entire family is absent in the Southern Hemisphere (Lusk 2008). The principal tree families shared are the most ancient ones, such as the cedars and cypress species (family Cupressaceae), that were well developed before the continents split apart.

While the Northern Hemisphere is dominated by conifers in the pine family (Pinaceae), trees in temperate rainforests of the Southern Hemisphere belong to a wide assortment of mostly small, specialized families. Among these, the myrtle family (Myrtaceae) is often the most diverse. Some other, more-modern families are also shared between the Northern and Southern Hemispheres, such as the heath and heather family (Ericaceae). In this case, these plants are particularly well adapted to cool, moist conditions, either alpine or subalpine, and have apparently been able to disperse along the Rockies and Sierra Madre in North America down the Andes all the way to Tierra del Fuego. The crowberry (*Empetrum nigra*), for example, has black berries in rainforests of the Northern Hemisphere, but red berries in the Southern Hemisphere (*E. rubrum*), and otherwise looks very similar between hemispheres. The occurrence of these two families may be, in part, attributable to dispersal of the seeds by migratory birds moving between hemispheres, a prospect that also has

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been proposed for some lichens. A striking exception to the pattern of divergence is the case of an increasing number of possibly relictual lichen lineages being discovered to be shared between the Pacific Coast of North America and Tasmanian and/or Valdivian rainforests (Spribille et al. 2010). However, the overwhelming pattern is one of disparity, with contrasting assemblages recurring with bryophytes, most nonmigratory birds, mammals, fishes, and insects. Why are these forests so taxonomically different between hemispheres? Let's explore some of the leading hypotheses.

Continental Drift and Isolation

This is generally considered the key factor explaining hemispheric differences. While the continents in the Northern Hemisphere were well connected many times in the past, including as recently as a few tens of thousands of years ago during glacial cycles, in the Southern Hemisphere many of the land masses that now have temperate rainforests have been isolated from each other since the late Tertiary period (over 60 million years ago—see Lawford et al. 1996; Veblen et al. 1996; Arroyo et al. 2000). This has lead to adaptive radiation events in species with ancient lineages, resulting in many unique forms (endemics).

Geography

Most of the Southern Hemisphere is dominated by ocean, and at the high latitudes land masses are highly fragmented and have been since the upper Tertiary some 2 million years ago, when the rainforest zone became progressively isolated by xeric climates to the east and north triggered by the uplift of the Andes (Arroyo et al. 1996). Thus, most temperate rainforests have milder winter climates with rainfall evenly distributed over the growing season. This unique climate leads to a more subdued role for wildfire and to a more limited adaptation to extreme cold. Even subalpine species from the Southern Hemisphere are generally not hardy enough to survive in continental rainforests of the Northern Hemisphere (Lawford et al. 1996; Veblen and Kitzberger 2002).

Endemism

The vast majority of species in temperate and subpolar rainforests of the Southern Hemisphere are unique to each continent (South America, Africa, and Australasia), and sometimes to a specific area due to their relictual taxonomic status and long periods of isolation (Lawford et al. 1996; Smith-Ramirez 2004; Hinojosa et al. 2006; also see chapters 5 and 8). By contrast, in the Northern Hemisphere fewer species are limited to specific habitats or

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areas, although island biogeographical effects in northern coastal latitudes have triggered speciation events at the subspecies level (see chapter 2).

Species Mutualisms

Many species in the Southern Hemisphere evolved from tropical affinities (e.g., Valdivia—see chapter 5), including complex interactions between plants, herbivores, pollinators, and seed-dispersing species. Further, most trees in Southern rainforests produce edible fruits and have co-evolved with seed-dispersing animal species (Armesto et al. 1996). In contrast, most rainforest trees of the Northern Hemisphere are conifers with less direct and specific co-evolution with pollinators and seed dispersers (e.g., Willson et al. 1990).

TEMPERATE AND BOREAL RAINFORESTS VS. TROPICAL MOIST RAINFORESTS

Tropical rainforests, as their name implies, are bracketed by the tropics of Cancer and Capricorn (see figure 1-11; table 1-3). They cover about 6 times more area than temperate and boreal rainforests (~2 percent versus 12 percent of the world's forests). Tropical rainforests are generally drenched in warm, moist climates with little seasonal temperature variation within 1 kilometer of sea level. On the other hand, temperate and boreal rainforests are generally but not

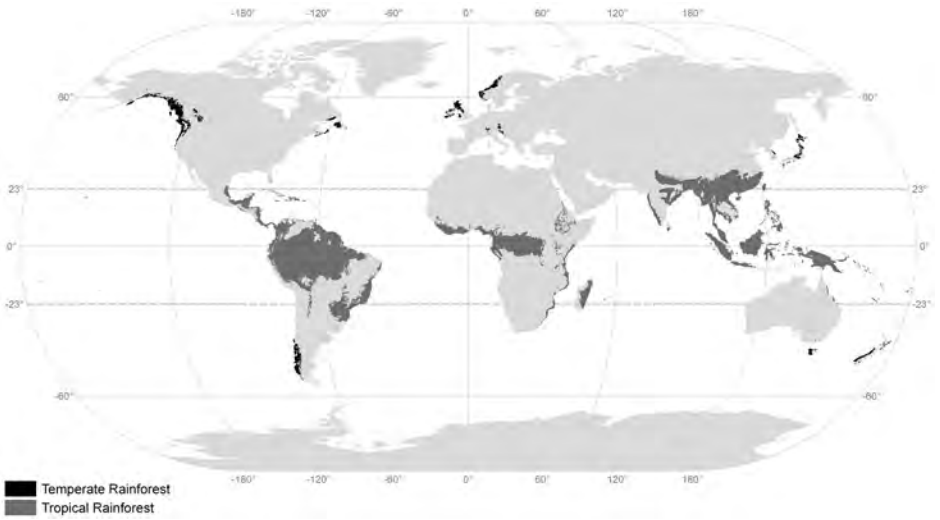

Figure 1-11. Tropical moist (Olson and Dinerstein 1998) and temperate and boreal rainforests of the world.

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Table 1-3. General features distinguishing tropical moist rainforests from temperate and boreal rainforests.

Feature	Tropical Moist Rainforest[a]	Temperate and Boreal Rainforest
Distribution	up to 23° latitude from the equator: large belts across South America, Central America, Southeast Asia, and Africa	~30–69° latitude, disjunct, mainly coastal: Pacific Northwest, Alaska, British Columbia, Chile, Argentina, Tasmania, New Zealand, Australia, Japan, Europe
Extent	~12% of present global forest cover, reduced by over half of estimated historic levels	~2% of present global forest cover, reduced by ~half of estimated historic levels
Deforestation (2000–2005)	1-2% annual[b], especially high in South America and Africa, mostly converted to agriculture	forest cover generally increasing, but old growth replaced by tree plantations
Annual Mean Temperature	23–27° Celsius	~4–12° Celsius
Seasonality	uniform temperature with wide variation in rainfall patterns (up to a 3-month dry season)	varied temperatures, snow in winter, greater precipitation in fall and winter with summer rains over 14% of annual precipitation
Moisture	over 1,700 mm, high humidity, high evapotranspiration	846–2658 mm, high humidity, low evapotranspiration
Canopy diversity	multilayered, rich epiphytes (orchids, bromeliads), and abundant lianas	generally multilayered, rich epiphytes (lichens, mosses), lianas less developed
Forest height	20–50 m	10–70 m
Soils	thin litter layer, infertile and severely leached except in volcanic and riparian areas; large nutrient pools in trees	rich humus, highly productive and rich in invertebrates, large amount of coarse, woody debris
Biomass	moderate (100–250 metric tons/ha), highest in dipterocarps (Southeast Asia)	low (Europe) to exceptional (redwoods, Pacific Northwest, Tasmania, Valdivia) (100–1867 metric tons/ha)
Productivity	high-exceptional	exceptional (marine, freshwater, terrestrial)
Nutrient cycling	rapid decomposition rates	slow decomposition rates
Pollination	exceptional	low in conifers
Plant and animal richness	exceptional, over half of terrestrial species on Earth, generally 5–10 times that of temperate forests	low (Europe) to moderate (Japan, Valdivia), but high for mosses and lichens
Endemism	exceptional, many species unique	low (Europe), moderate (California), high (Chile and Argentina)
Tree richness	exceptional (50–200 species/ha)	low to moderate (1–20 species/ha)

[a]Synthesized from Terborg (1992); Richards (1996); Kricher (1997); Myers et al. (2000); and Losos and Leigh (2004).

[b]Deforestation rates based on total forest cover lost on a continental scale (FAO 2005). Individual countries with rainforest, however, may have higher or lower rates of deforestation or show afforestation due to tree planting.

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exclusively found along coastlines at middle to upper latitudes, and can extend to nearly timberline (exceptions include Inland Northwest of North America, the Alps, and Inland Southern Siberia).

Climatically, temperate and boreal rainforests have a more distinctive seasonality (especially wider temperature swings), and greater range of precipitation types including snow and sleet, than tropical counterparts (see table 1-3). High temperatures in the tropics lead to high evaporation rates and the development of daily clouds above the forest, so that they can recycle 70 percent or more of their annual rainfall. Temperate rainforests, on the other hand, are cool and wet, with slower rates of decomposition and low evaporation rates. To better understand the differences between these rainforest types, we turn to some key concepts in forest ecology.

Ecologists today generally recognize that forest ecosystems are comprised of three main "ingredients": *composition*—the mix of species in a forest; *structure*—the vertical and horizontal dimensions and spatial patterns of a forest; and *function*—the workings of a forest expressed through nutrient cycling, food-web and disturbance dynamics, forest succession, pollination, and many other processes (Perry et al. 2008). The regions identified as temperate and boreal rainforest in this book have a suite of underlying characteristics along these lines that can be used to further distinguish them from each other as well as from their tropical counterparts.

Structure

Both temperate and tropical rainforests (boreal less so) have complex forest canopies composed of many canopy layers, creating dense and continuous vegetation cover that provides for rich fauna from the ground up. In both forest types, canopy gaps and emergent crowns of dominant trees create complex spatial patterns in the lower strata. A key difference in rainforest canopies is that temperate rainforests are dominated by conifers (except in the Southern Hemisphere, where they are dominated by broadleaf evergreens, and in Japan and Europe, where they can be deciduous), while tropical rainforests are dominated by broad-leaved trees enveloped by numerous lianas (Valdivia, New Zealand, Hyrcanic, and South African temperate rainforests also have lianas). Both rainforest types often have a high degree of standing dead trees (snags) and fallen logs that provide structure and habitat for scores of plant and animal species (Baker et al. 2007; Perry et al. 2008).

Function

Biomass in temperate rainforests is exceptional on a global scale, exceeding that of tropical rainforests (Smithwick et al. 2002; Losos and Leigh 2004; Keith et al.

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2009; see table 1-3). For instance, one study of a young temperate rainforest in Oregon showed that it could fix as much carbon per year as some mature tropical rainforests (e.g., 36 metric tons of organic matter per hectare annually—Fujimori 1971). Another study found primary forests in Australia capable of storing up to 1,867 metric tons per hectare, the world's highest known total biomass carbon density (Keith et al. 2009). However, while tropical forests are not exceptionally carbon-dense systems, they still play the dominant role for forest contributions to global carbon cycles due to their high rates of productivity, decomposition, long growing seasons, and the large land area they still occupy.

Evergreen needles (or leaves) are a common characteristic of the vast majority of tree species that grow in temperate and boreal rainforest climates. They allow rainforest plants to photosynthesize throughout the year in most coastal temperate areas, helping to explain the high productivity of these rainforests (Waring and Franklin 1979). The mild climate of these rainforest regions may explain why most of the tallest trees in the world grow there. Examples from around the world include towering *Eucalyptus* forests in southeastern Australia, massive coastal redwoods and alerce in California and Chile, respectively, and ancient coastal Douglas-fir (*Pseudotsuga menziesii* var. *menziesii*) of northern California and the Pacific Northwest. Finally, a continuously mild, wet climate, combined with minimal genetic losses during Pleistocene glaciations, may have played a role in maintaining the rich genetic diversity of conifer species in the Pacific Northwest but led to losses in other regions (Waring and Franklin 1979; Premoli et al. 2000).

Coastal rainforests also are productive places for marine life, with strong linkages between marine and terrestrial ecosystems (Simenstad et al. 1997). Well-known examples include the marbled murrelet (*Brachyramphus marmoratus*) of the Pacific Coast of North America, a coastal seabird that summers at sea but breeds and nests in the tops of old-growth trees; and historical links between Pacific sea-run salmon (*Oncorhynchus* sp.) and terrestrial predators such as bears (*Ursus* spp.) and wolves (*Canis lupus*), which, in the Great Bear Rainforest of British Columbia, prey upon salmon and help fertilize coastal riparian forests through their droppings (see chapter 2).

Composition

Compared to the tropics, in Northern Hemisphere rainforests plant and animal species richness is generally low, and endemism low to moderate, with some noted exceptions (see table 1-3), including island systems (e.g., Cook et al. 2001). However, lichens appear to be much more diversified at high latitudes than in the tropics (witness ~750 species for a single southeast Alaskan rainforest

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fjord compared to ~550 species in all of Thailand; Spribille et al. 2010). Even if many more lichens are discovered in the tropics and the relative richness gap closes, it appears that the tropics are by no means richer on the orders of magnitude that apply to some other groups of organisms. Outstandingly high levels of species richness also have been documented in basidiomycete fungi ("mushrooms") with hyper-diverse floras documented in coastal rainforests of British Columbia (Roberts et al. 2004) and over 750 macro-fungal species from a single stand of old-growth forest on a hill in rural Victoria on Vancouver Island (Češka 2009). Here, too, numbers may be far higher than in the tropics, especially of ectomycorrhizal fungal species (a type of mycorrhizae composed of a fungus sheath around the outside of root tips—Allen et al. 1995). How these numbers stack up in the long term against species numbers in the more poorly known Tropics remains to be seen, but the fact that key physiological processes for many fungal and lichen species are optimal at cool temperatures through community adaptation (Friedman and Sun 2005) suggests that, for lichens at least, the pattern may hold.

The generally low diversity of trees species in temperate rainforests, with some noted exceptions such as Valdivia and Japan (see table 1-3), should not seem too surprising, since these rainforests tend to have dense overstory canopies and occur in cloudy climates at high latitudes, leaving little light available for understory canopy layers. Many endemic plant species are associated with warm-temperate or seasonal rainforests, such as the forests in south-central Chile and northern California, as well as all rainforests that occur on islands, and other areas in the Southern Hemisphere. In addition, many moisture-adapted taxa that provide a unique physiognomy and structure closely tied to these rainforests, including epiphytic mosses, liverworts, and lichens, are associated with moist rainforest climates (Goward and Spribille 2005; see table 1-3). In these groups, endemism is locally high in Tasmania and New Zealand, Japan, Valdivia, and parts of northwest North America while it is low to nonexistent in the isolated patches of rainforest in Europe and Eastern Canada. This is likely correlated with the extent of glaciation and/or availability of extensive glacial refugia, combined with a long history of good dispersal across and between continents in these regions. Other species-rich taxa in these rainforests include insects (mostly soil and canopy species) and gastropods (mainly in the Pacific Northwest), with high levels of endemism in certain taxa. Apart from that, tropical rainforests are exceptional across taxa (see table 1-3).

Disturbance Dynamics

Stand-replacing disturbances are relatively rare in temperate and boreal rainforests, as they are in tropical moist forests. As a result, both rainforest types are

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dominated by ancient trees that have a complex structure and pattern, due to a long history of small patch or gap disturbances (see box 1-1). This history, along with the evolution of tree defenses against diseases, has allowed certain tree species to reach very old ages (Waring and Franklin 1979) in not only temperate rainforests (see above examples for tree species) but tropical rainforest trees as well (e.g., *Hymenolobium mesoamericanum* of Costa Rica can live for hundreds of

BOX 1-1

Gap Phase Dynamics of Temperate and Boreal Rainforests.

While most temperate and boreal rainforests are subject to various stand-replacing disturbances such as canopy fires, hurricanes, and landslides, forests in moist climates often have small-scale disturbances that serve to maintain the species composition and structure of the forest over time. Some authors have called these disturbances "maintenance dynamics" (see Veblen and Alaback 1996; Perry et al. 2008). A key ecological consequence of frequent gap disturbances is that a wide range of light environments and ecological conditions can be maintained in a forest that enriches its structural and compositional diversity. This also promotes a rich assortment of plant and animal species requiring vastly different light levels (e.g., both shade-tolerant and -intolerant species), and implies forest structure and composition can be theoretically maintained indefinitely. The extent to which a given rainforest is dominated by gap dynamics depends on many factors, including susceptibility to intense windstorms or geomorphic disturbances (landslides and flooding), as well as the susceptibility of individual trees to mortality, insects, and disease.

Key disturbance features of temperate and boreal rainforests are summarized as:

- Usually small-scale events affecting 1–4 percent of the forest area annually, although these gaps are eventually filled by light-seeking plants, creating a continuous push-pull dynamic between gap-dependent and gap-avoiding (anti-gap) species (Nowacki and Kramer 1998; Franklin et al. 2002).
- A small number of trees are killed in each disturbance event, usually fewer than 10 trees (Lertzman et al. 1996; Ott and Juday 2002).
- Gaps vary widely in size and shape, creating a rich mosaic of conditions in the forest (Ott and Juday 2002).

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BOX 1-1
Continued

- When gaps are created by wind events, root-throw can create a rich diversity of soils and microhabitat conditions in the forest, including "pit and mound" micro-topography (Bormann et al. 1995) and nesting sites for birds (e.g., winter wrens *Troglodytes troglodytes* often nest in root-wads).
- Tree architecture, including rooting depth, height and exposure of canopy, and resistance to decay fungi play key roles in determining susceptibility to windthrow.
- Openings in canopy created by gaps promote regeneration of tree and understory species, leading to greater diversity in the forest (Spies et al. 1990; Franklin et al. 2002).
- While in theory gap disturbances can maintain the structure and composition of the forest indefinitely, in practice gap dynamics can lead to changes in forests due to changes in the environment at the time of gap creation, including seed availability and dispersal, microclimate, and specific characteristics of a given gap event (see Lertzman et al. 1996).

years—Fichtler et al. 2003). The infrequency of natural fires in both rainforests adds to tree longevity (e.g., see Gavin et al. 2003).

While both tropical and temperate rainforests are affected by and in turn affect regional climates, tropical rainforests, along with the world's oceans, play a major role in the planet's climate regulation. When either rainforest type is cut down, much of their stored carbon is released as carbon dioxide, thus contributing to global warming as well as regional changes in moisture (evaporative losses) and temperature (as discussed in Chapter 11). Understanding this basic fact is key to climate change negotiations for protecting the world's mature forests in both the tropics and temperate zones for their pivotal role in long-term carbon storage (see chapters 10, 11).

RAINFORESTS: GOING, GOING, GONE?

Unfortunately, both temperate-boreal and tropical rainforests have been reduced by at least half their estimated original extent (i.e., before widespread

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human-related destruction of rainforests—see Bryant et al. 1997; Myers et al. 2000; Ritter 2008; see table 1-3). Logging in the tropics is typically accompanied by the burning of vegetation and conversion of biologically rich forest to agriculture fields also used by livestock. A recent development is the clearing of rainforest to grow crops for biofuels (e.g., Borneo, Malaysia, and forest thinning in the temperate zone). In the tropics, this comes with severe depletion of already nutrient-deficient laterite (acidic) soils due to the leaching of nutrients otherwise held in place by rainforest trees, thus hampering afforestation efforts. Temperate and boreal forests, on the other hand, mainly have been degraded by conversion of biologically rich, older rainforest to simplistic tree plantations, or have been high-graded, where old high-value trees (or forest patches) are removed without providing for adequate rates of regeneration of older age classes or ecological types (as discussed throughout this book).

Notably, some researchers (Kauppi et al. 2006) contend that the world's forests have been increasing over a 15-year period (1990–2005) measured by accruing wood volume, biomass, and captured carbon (growing stock). While this is certainly a positive development, it misses the point about ongoing losses to intact and high-quality forests such as old-growth or primary forests. Globally, very few large, intact primary forests (e.g., "frontier forests") remain (Bryant et al. 1997). In addition, according to estimates provided by the World Wildlife Fund, approximately 13 million hectares of forests are destroyed globally each year mainly in the tropics.[5] But these losses are not just restricted to the tropics. For instance, the United States was recently ranked seventh in the world in deforestation, an annual rate of 215,000 hectares (FAO 2005). These alarming losses come at a time when deforestation (including forest conversion as used here) was second only to fossil-fuel emissions in global contributions to greenhouse-gas pollutants, although growth in emissions from forestry slowed from 1970 to 2004 (IPCC 2007). These forests are not equated by tree farms achieved through planting, as the difference in terms of quality of forest composition, genetics, function, structure, and long-term storage of carbon (and its release by forestry operations) is hard to measure at a global scale, but such comparison is certainly feasible at regional scales through measures of forest quality, remote sensing, and landscape change-detection analysis.

Ongoing consumption of wood products, particularly in the United States, Canada, Japan, and Europe (where per capita consumption levels are highest), will continue this alarming trend of forest conversion in the temperate zone and complete deforestation in the tropics. Recycling, the use of alternative

[5] www.worldwildlife.org/climate/northsouthpartner.html

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fibers, and improvements in manufacturing technologies are offsetting this trend somewhat. Greater interest in the conversion of cellulosic fiber from forests to liquid fuel (biofuels), however, will put more pressure on the world's forests, both tropical (UNEP 2009) and temperate/boreal (Searchinger et al. 2009). Afforestation cannot keep pace with ongoing demand without further degradation of rainforest biota from the loss of primary forests and the suite of ecosystem services they uniquely provide.

ABOUT THIS BOOK

This book, while focused primarily on the ecology of temperate and boreal rainforests, is intended as a rallying call for global action to conserve these rainforests, which, like so many of the world's rainforests, are at a critical juncture. Each of the regional chapters is a closer examination of the history and ecological characteristics of the largest remaining examples of temperate and boreal rainforest, and provides essential information that can be used to make clearer global priorities for the conservation of these important rainforests.

The regional chapters (chapters 2–9) largely maintain a consistent structure throughout that includes basic information on rainforest location and types, climatic conditions, significant ecological attributes of regional and global importance, ecological processes such as natural disturbances and forest succession, keystone or exemplary rainforest species, regional rainforest classifications (zones or subtypes), threats, and conservation priorities. In Chapter 10, we summarize key findings from each of the rainforest regions in order to stitch together a unifying vision, based on fundamental concepts of conservation biology, for conserving the world's temperate and boreal rainforests. We end the book in Chapter 11 with a call for an international accord to prepare these rainforests for the inevitable consequences of climate change. Most important, we hope that the principles and concepts outlined in this book provide a scientific foundation for expanding rainforest protections around the globe, so that these remarkable rainforests will continue to meet the growing demands of human communities for the life-giving services that these forests have provided to us for millennia.

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Edited by

Dominick A. DellaSala

Geos Institute

Foreword by David Suzuki

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CHAPTER 2

Temperate and Boreal Rainforests of the Pacific Coast of North America

Dominick A. DellaSala, Faisal Moola, Paul Alaback, Paul C. Paquet, John W. Schoen, and Reed F. Noss

The world's most expansive stretch of coastal temperate and boreal rainforest is sandwiched between the Pacific Ocean and a chain of coastal mountains spanning 23 degrees of latitude and some 3,600 kilometers. Here, rainforests extend from northern Kodiak Island and Prince William Sound, Alaska (61°N latitude) to just south of San Francisco Bay, California (38°N latitude; see figure 2-1). Along the coastline, rainforest is limited to moist climates extending to as much as 160 kilometers inland in southeast Alaska and adjacent British Columbia to 60 kilometers or less in California and the Pacific Northwest. A secondary belt of rainforest up to 100 kilometers wide occurs along the western slopes of the Cascades from southern British Columbia to central Oregon.

RAINFOREST VITALS AND GLOBAL ACCOLADES

With over a third of the world's temperate and boreal rainforests and some of its most intact watersheds, the rainforests of British Columbia and southeast Alaska are a temperate "Amazonia." Southeast Alaska also boasts nearly a third of the world's old-growth temperate rainforest (Carstensen et al. 2007; see table 2-1). And coastal redwood (*Sequoia sempervirens*) is exceptional in having whole forests that have persisted for over 2,000 years.

Temperate rainforests of the Pacific Coast of North America are among the world's champions of storing carbon, primarily in their massive tree trunks, logs

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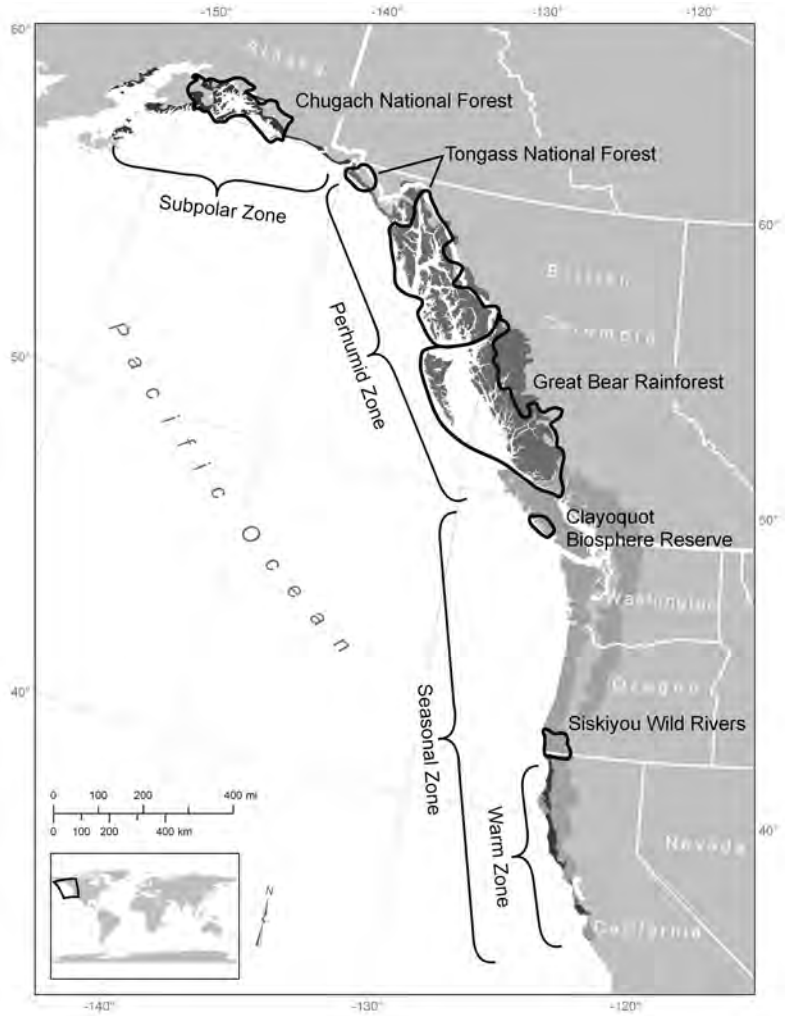

Figure 2-1. Temperate and boreal rainforests of the Pacific Coast of North America, showing rainforest zones (modified from Kellogg 1995) and conservation priority areas. Despite overlap, seasonal and warm temperate (redwood) zones differ in climate and vegetation.

on the forest floor, and thick rich humus layers in the soils (see table 2-1). Coastal redwoods, for instance, store nearly 3,000 metric tons of organic matter per hectare compared with 100–500 metric tons in most tropical and temperate forests, up to approximately 1,000 metric tons in the Pacific Northwest (Fujimori 1971; Beebe 1991; Sawyer et al. 2000; Smithwick et al. 2002), and 1,867 metric tons per hectare in mountain ash (*Eucalpytus regnans*) forests of southeastern Australia and the Central Highlands of Victoria (Keith et al. 2009).

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Table 2-1. Globally significant attributes of temperate and boreal rainforests of the Pacific Coast of North America.

Attribute	Importance
More than a third of the world's temperate and boreal rainforests	High conservation value due to global rarity
Some of the largest, oldest trees and most dense carbon pools	Wildlife habitat, long-term carbon storage
High levels of intactness and old growth (northern latitudes)	Habitat for fragmentation sensitive and old-forest dependent species
Expansive mountain fjords, glaciers, and limestone caves	Landscape complexity associated with high levels of beta diversity (turnover in species richness across edaphic gradients)
Exceptional levels of marine, freshwater, and terrestrial productivity, including prolific salmon runs	Rich and abundant wildlife also important for subsistence and cultural values
Exceptional richness of invertebrates, bryophytes, and epiphytes	Food for insectivorous species and nesting platforms for murrelets and other animals

Pacific Coastal rainforests are extraordinarily rich in epiphytes (canopy-dwelling plants), and soil and canopy invertebrates (e.g., see Cooperrider et al. 2000) due, in part, to a cool, wet climate, complex forest canopies, and rich soils (Sillett 1999; Sawyer et al. 2000).

Even the high-latitude rainforests are rich in lichens and bryophytes (e.g., mosses and liverworts—Worley 1972; Schofield 1988; Alaback 1995; Goward et al. 2001; see table 2-1). They also contain 380 vertebrate species, including unique ones (Cook and MacDonald 2001), and support the most prodigious salmon runs on Earth (e.g., *Oncorhynchus* spp. in the Fraser and Skeena Rivers of British Columbia; the Chilkat and Taku Rivers of Tongass National Forest; the Kenai and Copper Rivers of Chugach National Forest; North and Central Coastal British Columbia). Historically, rivers of the Pacific Northwest, especially the Columbia River of Washington and Oregon, and the Klamath River of California and Oregon, had massive salmon runs. Salmon returning to these rivers today represent but a fraction of historic runs, with losses attributed mainly to habitat degradation from logging, hydroelectric dams, agricultural practices, and overfishing in places.

These rainforests also lie at the intersection of land, sea, freshwater, and glacier (ecotones), one of the most productive living tapestries in the world (Ricketts et al. 1999). Rivers transport sediments from glacial melt, known as "glacial

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flour" for its murky appearance, which aid in the formation of wetlands and deliver nutrients to aquatic species (Montgomery 1997). Downed rainforest trees embedded in streams offer hiding places and resting pools for migrating salmon in search of cool, shaded waters to spawn. When dislodged, logs float downriver where they are washed ashore, anchoring the beachfront for colonizing vegetation. Depending on stream volume, coastal salinity is diluted at the mouth of rivers, affecting marine productivity (e.g., the massive plume of the Columbia River—Simenstad et al. 1997). Tidal waters flush saline water back through narrow fjords, providing cues for salmon to migrate (Simenstad et al. 1997).

THE MAKING OF RAINFOREST

Pacific Coastal rainforests of North America are dynamic places, shaped by the interplay of geological and glacial forces and ongoing disturbances of various sizes and intensities. Although the retreat of glaciers in the early Holocene some 12,000 years ago opened up vast coastal areas and thousands of nearshore islands for both rainforest species and indigenous people, contemporary rainforests did not take their present form until some 7,000 to 4,000 years ago (Hebda and Whitlock 1997). Amazingly, these rainforests have only been around for just a few generations of their oldest trees (e.g., coastal redwoods), and in Alaska some rainforest communities (Prince William Sound, Yakutat and Kodiak Islands) have been present for less than 1,000 years.

Pacific Coastal rainforests are surrounded by expansive mountain fjords and glaciers from British Columbia northward. Tidal glaciers are especially pronounced in high-latitude areas (LeConte is the southernmost tidewater glacier at 56°N latitude) where for thousands of years they carved U-shaped valleys (mountain fjords) and fingerlike bays, projecting deep into rainforest interior and once extending 80 kilometers or more off the contemporary outer coastline. Despite this massive accumulation of ice, several regions escaped glaciation, including Haida Gwaii (Queen Charlotte Islands), Brooks Peninsula on northern Vancouver Island, the outer coast of Glacier Bay, Dall Island (southeast Alaska), Kodiak Island, and forests south of the Olympic Peninsula. These areas provided refugia for rainforest species during the Pleistocene glaciations, including endemic subspecies found nowhere else on Earth (Alaback and Pojar 1997; Cook and MacDonald 2001; Coast Information Team Report 2004). Pacific salmon also include many genetically unique stocks (Allendorf and Waples 1996).

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RAINFOREST GRADIENTS AND CLIMATE

Because this coastal rainforest spans such great north-south distances, differences in rainforest communities are especially pronounced. Ecologists like to group these differences along environmental gradients determined by abiotic (chemical and physical) factors underlining the distribution of living communities (biotic factors).

Distributional Gradients

In the most northerly rainforests, tree line dips down below 200 meters elevation and conifers include just two species: Sitka spruce (*Picea sitchensis*) on Kodiak Island and mountain hemlock (*Tsuga mertensiana*) on the Kenai Peninsula. Conversely, conifer richness and endemism in southern locales are much higher, particularly where coastal and inland areas overlap (Alaback 1995; DellaSala et al. 1999). In these southern areas, tree line is sometimes present at the tallest peaks (up ~2,744 meters elevation in places) where alpine tundra takes over. Terrestrial productivity also increases southward, owing to long growing seasons and high nutrient availability. Despite the high productivity of southern forests, forest landscapes become extremely patchy and water-limited, with the southernmost forests restricted to coves and riparian draws until Mediterranean scrub vegetation eventually dominates.

While species richness is generally low in northern latitudes, some species such as bald eagle (*Haliaeetus leucocephalus*) and brown bear (*Ursus arctos*) attain exceptional densities due, in part, to a superabundant food source—salmon—and relatively high levels of intact forest in places. Species that are well adapted to cool, moist conditions, such as lichens, liverworts and mosses, as well as many marine invertebrates, typically have greater diversity in these northern ecosystems than they do in southern areas (Schofield 1988; Alaback 1995; Simenstad et al. 1997).

Rainforest Climate

The collision of saturated marine air against coastal mountains soaks these rainforests in life-giving precipitation, with some locales receiving up to 5,300 millimeters annually (Alaback 1995; Western Regional Climate Center 2006). The regional climate lacks extended warm or cold periods, has a high frequency of cloud cover (60–80 percent monthly), and fog (especially in the redwood region; Redmond and Taylor 1997). In the most northerly locales, rainfall is persistent year round (over 25 percent annual precipitation in summer). South-

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ward, summers become progressively drier, especially in the redwood region, where under 2 percent of annual rainfall occurs in summer months, rendering these forests as more seasonal rainforests (Alaback 1995). Climate is increasingly continental inland, as the moderating influence of the Pacific Ocean diminishes, and snow, uncommon throughout, occurs at higher elevations and latitudes. Precipitation is highest on the windward side of mountains and oceanic islands, with rain shadows on the leeward side.

This persistent moisture has several direct implications to rainforest ecology. The most significant factor is the reduced role of fire, in contrast with drier forest types. South of the Canadian border in the seasonal rainforests of the Pacific Northwest, infrequent fire, every 300–600 years, plays a key role in establishing long-lived species such as Douglas-fir (*Pseudotsuga menziesii*) (Veblen and Alaback 1995; Franklin et al. 2002; Gavin et al. 2003). Farther north in the perhumid and subpolar rainforests (see classifications below), fire has only a localized influence. The other key implication of a persistently humid climate is the development of thick organic layers on the forest floor and slow rates of decomposition. This ultimately leads to the development of peat bogs in northern forests.

ISLAND BIOGEOGRAPHY

The northern rainforest region contains thousands of small islands and some of the largest (e.g., Prince of Wales Island, southeast Alaska, is the fourth largest in the United States) and most mountainous ones (elevations above 1,800 meters) in North America. This is a place where island biogeography, the size of individual islands and their distance from the mainland and one another, combine to influence the composition of rainforest communities. In southeast Alaska, brown bears are distributed along the mainland coast and northern islands of the Alexander Archipelago (the so-called ABC islands—Admiralty, Baranof, Chichagof, and smaller adjacent islands) but are absent on the southern islands (south of Fredrick Sound) that are inhabited instead by black bears (*U. americanus*) and wolves (*Canis lupus*), which also occur on the mainland coast. The Alexander Archipelago of southeast Alaska and the Queen Charlotte Islands contain endemic subspecies of invertebrates, birds, and small mammals, some of which maybe especially vulnerable to extinction due to insularity effects (Cook and MacDonald 2001; Smith 2004).

Other unique rainforest inhabitants include: the Alexander Archipelago wolf (*C. l. ligoni*) from Yakutat Bay to Dixon Entrance of southeast Alaska

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except for the ABC Islands; ermine (*Mustela erminea haidarum*), a threatened forest dwelling, weasel-like mammal restricted to the Queen Charlottes of British Columbia; and spruce grouse (*Falcipennis canadensis isleibi*) on Prince of Wales Island, Alaska. Notably, coastal wolves of British Columbia are highly distinct and representative of a unique ecosystem, whereas gray wolves of interior British Columbia are more similar to adjacent populations of wolves located in Alaska, Alberta, and Northwest Territories (Muñoz-Fuentes et al. 2009). Given their unique ecological, morphological, behavioral, and genetic characteristics, gray wolves of coastal British Columbia should be considered an Evolutionary Significant Unit (ESU) and, consequently, warrant special conservation status (see Muñoz-Fuentes et al. 2009). Many plant and tree species also are isolated on islands in British Columbia and Alaska, including alpine and whole forests of subalpine fir (*Abies bifolia*) and silver fir (*Abies amabilis*) (Alaback 1995; Fedje and Mathewes 2005).

Another key element that leads to heterogeneity of plant communities and consequently high levels of biological diversity is the patchy and complex nature of the geological formations, particularly across the archipelago portion of the region. Most notable is the existence of belts of karst, or limestone deposits, which occur on several southern islands in Alaska (especially Prince of Wales Island) and Princess Royal Island in British Columbia, where they are associated with extensive cave systems, many of which contain fossilized ancient carnivores and the remains of other extinct animals. Karst deposits provide sites for many unique mosses and vascular plants as well as for what were the most historically significant patches of large Sitka spruce in southeastern Alaska (Alaback 1995; Carstensen et al. 2007).

Because much of the northern portion of this rainforest region is insular, island biogeography, combined with human-related disturbances, may increase extirpation risks of rainforest subspecies (see Hanley et al. 2004; Cook et al. 2006). Other vulnerable species include the Alexander Archipelago wolf, the Prince of Wales flying squirrel (*Glaucomys sabrinus griseifrons*), ermine (*Mustela ermine*—several endemic subspecies), and American marten (*Martes americana caurina* and *M. a. americana*) (Hanley et al. 2004; Flynn et al. 2004, Cook et al. 2006). Additional rainforest biota may be sensitive to human impacts due to their limited dispersal (e.g., flightless arthropods, endemic mollusks, mosses, and lichens) and dependence on older forest structures (e.g., downed decayed logs, tree-fall gaps). In fact, recent research has shown that even remnant patches of rainforest are critical to the survival of species sensitive to forest fragmentation (Moola et al. 2004; Halpern et al. 2005).

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FOREST SUCCESSION AND DISTURBANCE DYNAMICS

Coastal rainforests of North America are shaped by natural disturbances ranging from small gaps in the overstory canopy created by the death of individual trees to stand- and landscape-level insect outbreaks, high-wind events (especially in archipelagos—Nowacki and Kramer 1998), and landslides, although infrequent but severe fire is a disturbance agent mainly south. Landslides deliver massive amounts of sediments to streams, while extensive floods inundate rainforest areas, creating nutrient and energetic exchanges that influence species composition, food-web dynamics, and ecosystem productivity (Veblen and Alaback 1995; Alaback and Pojar 1997).

In the moist and cool rainforests of the north, fire is nearly absent and rainforests persist relatively unaltered for centuries. The transitional and seasonal rainforests to the south; however, have less old growth, owing to infrequent but large fires and higher logging levels. Old-growth rainforests in the Oregon Coast Range, for instance, historically accounted for ~25 to 75 percent of the forest age classes, depending on disturbance dynamics (Wimberly et al. 2000). Farther north, nearly all (roughly 90 percent) of the productive rainforests in southeast Alaska and the north coast of British Columbia were old growth, affected little, if at all, by fire (Gavin et al. 2003; Albert and Schoen 2007a). Small-scale windthrow is the dominant natural disturbance agent in these forests (Lertzman et al 1996). Large-scale blowdown and landslides on steep slopes and wind-exposed terrains also occur (Veblen and Alaback 1995; Alaback and Pojar 1997).

Following stand-replacing disturbance events, plant communities proceed along a successional continuum propagated initially by fast-growing, colonizing forbs, grasses, shrubs, and conifer seedlings. This early seral stage is then followed by young, densely packed conifers ("stem-exclusion phase") that overtop and shade out most understory plants, usually within 15–20 years, and by several intermediate stages leading to old-growth forest (Alaback 1982, 1984; Alaback and Juday 1989; Franklin et al. 2002; Spies 2004). Barring large disturbances, the progression to old-growth forest takes some 150–400 years, depending on site conditions (Alaback 1982; Franklin et al. 2002; Spies 2004; VanPelt 2008). Due to the recurrence of small disturbances, old-growth rainforests are patchy places, with pioneering stages persisting at finer spatial scales (i.e., fine-grain heterogeneity) wherever disturbance resets nature's successional clock. For instance, blowdown of a giant conifer creates a gap in the tallest tree crowns for life-giving sunlight to penetrate deep within the forest understory, allowing

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fast-growing plants to fill the gap and creating a structurally diverse forest from the ground up (Franklin et al. 2002). Therefore, an individual old-growth forest site (or stand) represents a perpetual tug-of-war between gaps created by the death of overstory trees and light-seeking plants racing to fill them (also see discussion of maintenance disturbances in chapter 1). Such fine-scale processes make older forests much more structurally complex and biologically rich than the vast areas of biologically impoverished tree plantations replacing them (Franklin et al. 2002; Lindenmayer and Franklin 2002; Spies 2004).

In general, older forests and large-tree stands tend to have very high levels of species richness and exceptional structural complexity (Lindenmayer and Franklin 2002; Franklin et al. 2002; Spies 2004). They also store vast amounts of carbon important in climate regulation (Smithwick et al. 2002; Luyssaert et al. 2008; Keith et al. 2009). While most conservation efforts rightfully have focused on protecting older forests (Schoen et al. 1988; Strittholt et al. 2006), unlogged, early-successional rainforest communities created by natural disturbance events also are exceptionally rich (Franklin and Agee 2003; Swanson et al 2010). The tail ends of the rainforest successional continuum (natural young and old) include high levels of species richness, complex ecosystem functions, and important forest structures (especially biological legacies) that are either absent or present in much lower levels in tree plantations (see table 2-2). Because they are managed intensely for fiber production, tree farms often include chemical inputs (herbicides, fertilizers, pesticides), and their ongoing maintenance represents chronic ecosystem stressors (e.g., runoff of sediments facilitated by roads and logging on steep slopes that pollute streams).

Other examples of rainforest communities that are of conservation interest yet are not old-growth are periglacial forests, which have developed in the wake of melting glaciers since the Little Ice Age, and shoreline forests that have developed on uplifted beaches in northern southeast Alaska and Prince William Sound (e.g., Carstensen et al. 2007). These are the first forests that have developed since glaciers or the ocean scoured soils from bedrock or sediments. Their soils are poorly developed, and they are quite sensitive to logging and other disturbances that could reduce nutrient availability.

KEYSTONE SPECIES

Keystone species drive the abundance, distribution, and ecological requirements of other species within the rainforest community. Perhaps no species typifies this role better than Pacific salmon (Willson and Halupka 1995; Gende

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Table 2-2. Comparisons of old-growth rainforest, naturally regenerating rainforest, and tree plantations along the Pacific Coast of North America.

Type	Feature	Importance
Old-growth rainforest	continuous, multilayered canopy	habitat for wildlife such as neotropical migratory birds, spotted owls, shading for fish
	abundant, large, old trees	sites for owls, murrelets, and lichens; carbon storage
	abundant dead trees (snags)	nesting and foraging sites, anchor soils
	abundant logs	fish habitat, soil stabilization, "nurse" logs for conifer seedlings, salamander and invertebrate sites, soil richness, mycorrhizae fungi
	diverse shrubs/forbs	habitat for ground-nesting birds and mammals, especially deer
	small canopy gaps	structural complexity, habitat for early seral species
Young, naturally regenerating rainforest[a]	biological legacies (logs/snags/large trees) carried over from old forest following distrubance	habitat for old-growth associates, "anchor" soils, shade conifer seedlings from intense sunlight, sites for mycorrhizae fungi
	exceptional shrub and forb richness	habitat for deer and other early seral species, nutrient cycling
	complex vertical and horizontal structure	habitat for rich array of early-seral and some late-seral species
	complex nutrient cycling and energy (food-web) pathways	productive soils and high species diversity
Tree plantations[b]	young, densely packed trees	simplistic vertical and horizontal structure due to dense spacing of small trees
	low understory light levels	sparse forb and shrub layer
	simplistic genomes with seedlings grown to match local site conditions	reduced resilience to climate change
	low levels of nutrient cycling and mycorrhizae fungi	diminished below-ground processes
	lack of biological legacies	impoverished wildlife habitat and ecological processes
	chronic erosion and depletion of soils, particularly from roads and use of heavy machinery	long-term diminished site productivity and/or use of fertilizers and other chemicals to supplement nutrient deficiencies
	high risk of severe fires in drier areas due to high fuel loads and roads (ignition factors)	altered successional pathways

[a]Generated by natural disturbance events such as blowdown, volcanic eruptions, wildfires, and landslides.

[b]Degree of differences vary depending on scale and intensity of forest management.

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et al. 2002). More than 190 species of plants and animals eat salmon, including marine mammals, birds, bears, and, surprisingly, even wolves (Cederholm et al. 2000). By way of their dual oceanic and riverine life cycles and their importance to predators, salmon uniquely span marine, freshwater, and terrestrial systems. During salmon runs, for instance, coastal bears derive up to 90 percent of their total annual dietary requirements as well as essential fat stores for hibernation (Temple 2005). Wolves are seasonally dependent on salmon as a primary source of food in this region (Darimont et al. 2008). Amazingly, up to 80 percent of yearly nitrogen uptake in old trees is from nutrients derived from rotting, spawned-out salmon carcasses (Reimchen 2000).

From a conservation standpoint, the potential extirpation of keystone species like salmon could have cascading ecological effects that reverberate through rainforest communities (Lichatowich 1999). For instance, salmon declines are known to reduce nutrient levels affecting food-web dynamics in streams (Gresh et al. 2000) and have the potential to adversely affect regional economies that depend on productive fish runs (Sisk 2007a). Notably, nearly one of four salmon populations is at risk of extinction in this region (Augerot 2004), and several have vanished already (Nehlsen and Lichatowich 1997; Price et al. 2008).

REGIONAL RAINFOREST CLASSIFICATIONS

To account for variability in species assemblages and broad climatic differences across such an expansive region, ecologists have subdivided coastal rainforests using various classifications. For instance, Gallant (1996) developed an ecoregional classification system based on land use, land surface form, potential natural vegetation, and soils, delineating two Alaskan ecoregions that support coastal rainforests: Pacific Coastal Mountains and Coastal Western Hemlock–Sitka Spruce Forests. Nowacki et al. (2001) refined this approach (finer scale) by mapping five areas with coastal rainforest: Alexander Archipelago, Boundary Ranges, Chugach-St. Elias Mountains, Gulf of Alaska Coast, and Kodiak Island. Ricketts et al. (1999) developed ecoregional classifications based on broad differences in vegetation, climate, and landform, identifying five ecoregions in North America with coastal temperate rainforests: North Pacific Coastal Forests (southeast Alaska, Prince William Sound, and eastern Kodiak Island), British Columbia Mainland Coastal Forests, Queen Charlotte Islands, Central Pacific Coastal Forests (Vancouver Island south to southern Oregon), and Northern California Coastal Forests.

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In this chapter, we used the classification system of Alaback (1995) and Alaback and Pojar (1997) described in further detail in Chapter 1 primarily because these classifications were developed specifically for this region, were based largely on differences in summer rainfall and temperature, and were useful in grouping conservation priorities. The four zones (see figure 2-1) include:

- *Subpolar*—highest-latitude coastal forests
- *Perhumid*—northern Vancouver Island, British Columbia to southeast Alaska
- *Seasonal*—southern Oregon to central Vancouver Island
- *Warm*—San Francisco Bay to southern Oregon coast.

Perhaps the most conspicuous occupants spanning all rainforest zones, however, are the conifers. Species composition varies but the predominant conifers are Sitka spruce, western hemlock (*T. heterophylla*), western red cedar (*Thuja plicata*), Douglas-fir, silver or amabilis fir (*Abies amabilis*), shore pine (*Pinus contorta*), yellow cypress or yellow cedar (*Chamaecyparis nootkatensis*), and coastal redwood (south).

CONSERVATION PRIORITIES

We nested conservation priorities and rainforest threats within each of the four rainforest zones. Threats varied by zones but, in general, logging was greatest southward (but moving north) and industrial tourism, mining, and salmon farming more prevalent northward (see box 2-1).

SUBPOLAR (BOREAL) RAINFOREST CONSERVATION PRIORITIES

The subpolar zone includes high-latitude rainforest with distinct subpolar attributes (Alaback and Juday 1989; Alaback 1995). Forests are bracketed by glaciers interspersed among alpine meadows, muskegs, and other wetlands and are dominated by mountain hemlock and Sitka spruce. Due to the short growing season and poor soil conditions, conifers rarely exceed 30 meters tall and are found on protected slopes where soil drainage is good (Alaback and Pojar 1997). Annual precipitation is often greater than 3,810 millimeters and snow is more common than in southerly latitudes.

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BOX 2-1

Threats to Temperate and Boreal Rainforests of the Pacific Coast of North America.

More than 40 percent of rainforests throughout this region have been fragmented by land-use activities, with highest fragmentation levels southward (see Cascadia Scorecard 2007; sightline.org[1]); logging has been advancing northward more recently. The amount of protected rainforest varies from approximately 5 percent that is strictly protected in the Pacific Northwest to nearly 85 percent of the Chugach National Forest in Alaska, but generally below representation targets for most areas. Types of protection range from Wilderness (strictly protected) and National Parks (although commercial recreation and even some development are allowed in Canadian parks) to conservation reserves (some or no commercial logging allowed but other development activities, such as mining or road building, are permissible). Mining and energy development (e.g., Prince William Sound) are ongoing risks to fish and wildlife. Industrial tourism, especially along the inside passage of Alaska and Prince William Sound is increasing. (Tourism is also a benefit to local economies when properly managed.) Fish hatcheries and salmon farming compete with and adversely affect wild fish runs, and overfishing in nearshore waters is reducing fish runs in portions of British Columbia. Extirpation of grizzly bears and wolves (e.g., south of Canada) has altered food-web dynamics through removal of apex predators. Although trophy hunting of wolves, bears, and other large carnivores is legally allowed throughout most areas north of the Canadian border (including most parks and protected areas), such mortality can be managed, whereas the poaching and killing of large carnivores by people for defense of life or property is relatively unpredictable.

Rainforest conservation is hampered by weak wildlife regulations and laws (e.g., British Columbia Forest Range and Practices Act and limitations on the Canadian Species At Risk Act), as well as inconsistent policies (e.g., uncertain federal roadless-area policies in the United States). Other threats include exotic-species invasions (particularly south), rapid growth and expanded access for off-highway vehicles in previously inaccessible areas, and, perhaps greatest of all, climate change. Cumulative impacts have led to loss of salmon runs from southern British Columbia southward, and the imminent extirpation of the northern spotted owl in southwest

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BOX 2-1
Continued

British Columbia and the Olympic Peninsula of Washington. Global warming is melting tidal glaciers and shrinking mountain snowpack (Mote et al. 2005), and is expected to drop river flows and produce water temperatures too warm for spring and summer salmon runs. Other anticipated climate-change impacts include increased erosion from floods and landslides.

[1]www.sightline.org/maps/maps/forests_over_cs04m

The subpolar zone reaches its western terminus on Afognak Island, in the northern portion of the Kodiak Archipelago, and its northern terminus in the Prince William Sound, Alaska. Owing to island biogeography and extreme northern latitude, forests arrived recently (less than 1,000 years ago) and conifers are limited to just Sitka spruce. The rest of the archipelago is treeless (although spruce is naturally migrating southward on Kodiak Island) and home to some of the most abundant salmon and brown bear populations in the world. Because of its remoteness and relatively high level of intactness on public lands, the Chugach National Forest is a conservation priority.

Chugach National Forest

Along the shoreline of Prince William Sound (see figure 2-1) is a naturally discontinuous band of rainforest where three conifer species meet—Sitka spruce, western hemlock, and mountain hemlock. The Chugach includes an extensive coastline of more than 4,800 kilometers rimmed by tidewater glaciers. A mosaic of rainforest, peatland bog, and beach fringe (an ecotonal zone where rainforest and beachfront meet) mixes with meadow, shrubland, and tundra. The retreat of glaciers allowed plants to colonize from Asia and regions north and west (Cook and MacDonald 2001; Cook et al. 2006).

The Chugach National Forest is the second largest national forest in the U.S. (2.24 million hectares), encompassing the Kenai Peninsula, Prince William Sound, and the Copper River Delta, one of the most productive migratory-shorebird stopovers in the world (Ricketts et al. 1999). About a third of the National Forest consists of active glaciers and nonforest habitat, including the largest coastal wetlands on the Pacific Coast. Most of the Chugach (85 percent

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of the forest base) is protected (see chapter 10) and the National Forest is managed primarily for fish, wildlife, and recreation so that threats are relatively minimized (although federal roadless-area policies are currently in flux; Turner 2009).

Industrial-scale logging and road building (primarily on Native Corporation lands and largely completed), mining, coastal transport of crude oil, energy development, and expanding recreation (mainly commercial tourism from large cruise ships and growing off-highway vehicle access) are principal threats. Nearly all (98 percent—2.16 million hectares) of the National Forest is roadless and, therefore, of international importance.

Currently, roadless-area policies in the United States are in flux: a roadless conservation rule excluding nearly all forms of logging and road building from all federally inventoried roadless areas (larger than 2,000 hectares) on national forests was enacted by President Clinton in 2001, repealed by President Bush in 2005, and partially reinstated by the courts in 2006 and 2008 with some notable exceptions (Tongass National Forest was exempted in 2003 by President Bush but not yet reinstated by court or administrative decisions), where it remains today in judicial and administrative uncertainty (although congressional legislation was recently introduced and the Obama administration has upheld most of the original rule thus far). In addition, ongoing effects to coastal species from the running aground of the Exxon Valdez in 1989 (at the time, the largest oil spill in U.S. history) are still being felt in some places (Short et al. 2007).

Conservation priorities for the Chugach include:

- Comprehensive planning for the growth of recreation-based tourism that is bringing both positive (economic diversification) and negative impacts (overuse) to the region.
- Tighter restrictions and enforcement of off-highway vehicle access that contributes to wildlife harassment, pollution, and soil erosion and compaction.
- Permanent protection of roadless areas, maintaining much of the pristine character of the rainforest.
- Full implementation of the Exxon Valdez oil-spill restoration plan, especially land acquisitions that compensate Native Corporations willing to sell their lands to the government so they can be protected from development (e.g., Afognak Island Native corporation lands, see www.cooperativeconservatonamerica.org).

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PERHUMID RAINFOREST CONSERVATION PRIORITIES

These are wet places year round, with cool summers delivering 10–20 percent of the annual precipitation (Alaback and Pojar 1997). Snow is common but relatively transient near the coast. Conifer richness is higher than more northerly latitudes and includes Sitka spruce, western hemlock, mountain hemlock, amabilis fir, shore pine, western red cedar, and Alaska yellow cedar. High-priority conservation areas in this zone include the Tongass National Forest in southeast Alaska, the British Columbia Raincoast (also known as the Great Bear Rainforest and Haida Gwaii), and Clayoquot Sound (British Columbia). These areas were chosen because they support relatively high proportions of intact, old-growth rainforest, and abundant salmon and other wildlife, including apex carnivores.

Tongass National Forest

A vast chain of islands (including more than 5,500 larger than 0.4 hectares) and a narrow mainland coast stretch from the Yakutat Bay (north of Glacier Bay) to Dixon Entrance at the southern end of Prince of Wales Island, a distance of over 835 kilometers (Albert and Schoen 2007a; see figure 2-1). The "crown jewel" of the national forest system and a land that time nearly forgot, the Tongass is the largest (6.8 million hectares) national forest in the U.S. Verdant rainforest is spread across an extensive shoreline of nearly 30,000 kilometers, interspersed with tidewater glaciers and rugged islands, a profusion of salmon-spawning streams, and bisected by deeply dissected fjords (Albert and Schoen 2007a). About half (3.3 million hectares) of the total land base is forested but only about a third of the total land base (2.3 million hectares) is considered productive (i.e., commercially valuable) forest land; of that, about 90 percent (2 million hectares) is old growth (Albert and Schoen 2007a).

But not all old growth is the same in these rainforests. Stands of the largest old-growth trees (measured by total timber volume per hectare) have always been rare and today represent just 3 percent (~216,000 hectares) of the entire land base on the Tongass, because that is where much of the past logging has been concentrated (Albert and Schoen 2007a; USDA Forest Service 2008). These stands include some of the most valuable fish and wildlife habitats (DellaSala et al. 1996; Albert and Schoen 2007a) in the area. Past high-grade logging (focused on the largest trees) eliminated the best old growth concentrated on the most productive regions (especially Prince of Wales, Mitkof, and Chichagof islands) and the most ecologically valuable and intact watersheds (Albert and Schoen 2007a, b).

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All six species of Pacific salmon—chum (*O. keta*), coho or silver (*O. kisutch*), king or chinook (*O. tshawytscha*), pink (*O. gorbuscha*), sockeye (*O. nerka*), and steelhead (*O. mykiss*)—spawn in Tongass waters, supporting some of the largest concentrations of brown bears and bald eagles in the world (Albert and Schoen 2007a). Fishing (commercial, subsistence, recreation) provides the greatest number of natural-resource jobs (Everest 2005). Many of the resources that people value are associated with intact watersheds having abundant old growth and healthy populations of fish and wildlife.

On the Tongass, logging and road building began in earnest after World War II, during which three 50-year government contracts (cancelled in the mid-1990s) were signed by the USDA Forest Service with private companies in Ketchikan, Wrangell, and Sitka to process Tongass lumber, mainly for export to Asia. These contracts were the first and only in the nation and focused logging efforts on high-volume rainforests with the largest trees (Sisk 2007b). A postwar logging boom began rainforest depletion that was slowed by litigation, poor export markets, and ultimately the loss of large blocks of easily accessible and economically valuable timber. Meanwhile, the Alaska National Interest Lands Conservation Act (ANILCA 1980) left an indelible mark on the Tongass landscape. One of ANILCA's legislative achievements was to protect some key places with productive fish and wildlife habitats such as Admiralty Island. However, much of the wilderness protection occurred in areas dominated by rock and ice that were not the most productive fish and wildlife habitats available (Albert and Schoen 2007b). Worst of all, ANILCA mandated large logging subsidies and unsustainable rates of logging that led to significant losses of old growth during the 1980s. The Tongass Timber Reform Act (1990) intended to eliminate these logging subsidies by reducing logging pressure on the high-volume (large-tree) rainforests and by protecting a number of invaluable old-growth watersheds, although abuses have been documented (Katz 1992).

Decades of rainforest logging on the Tongass have left many scars easily viewed in a flyover in one of Alaska's many float planes (a primary mode of Alaska transport, see plates 1a and 1b). This includes more than 7,900 kilometers of logging roads (USDA Forest Service 2008), especially on Prince of Wales Island (figure 2-2 shows the progression) and Long Island (over 90 percent of the forests were logged after they were transferred to Native corporations), the most biologically productive islands on the Tongass. In addition, whereas over half of the Tongass is without roads (3.74 million hectares), current national roadless-areas policies do not provide inviolate protection for rainforest (the region was exempted by a recent court ruling), with logging projects proposed for roadless areas by the Forest Service. Moreover, under the

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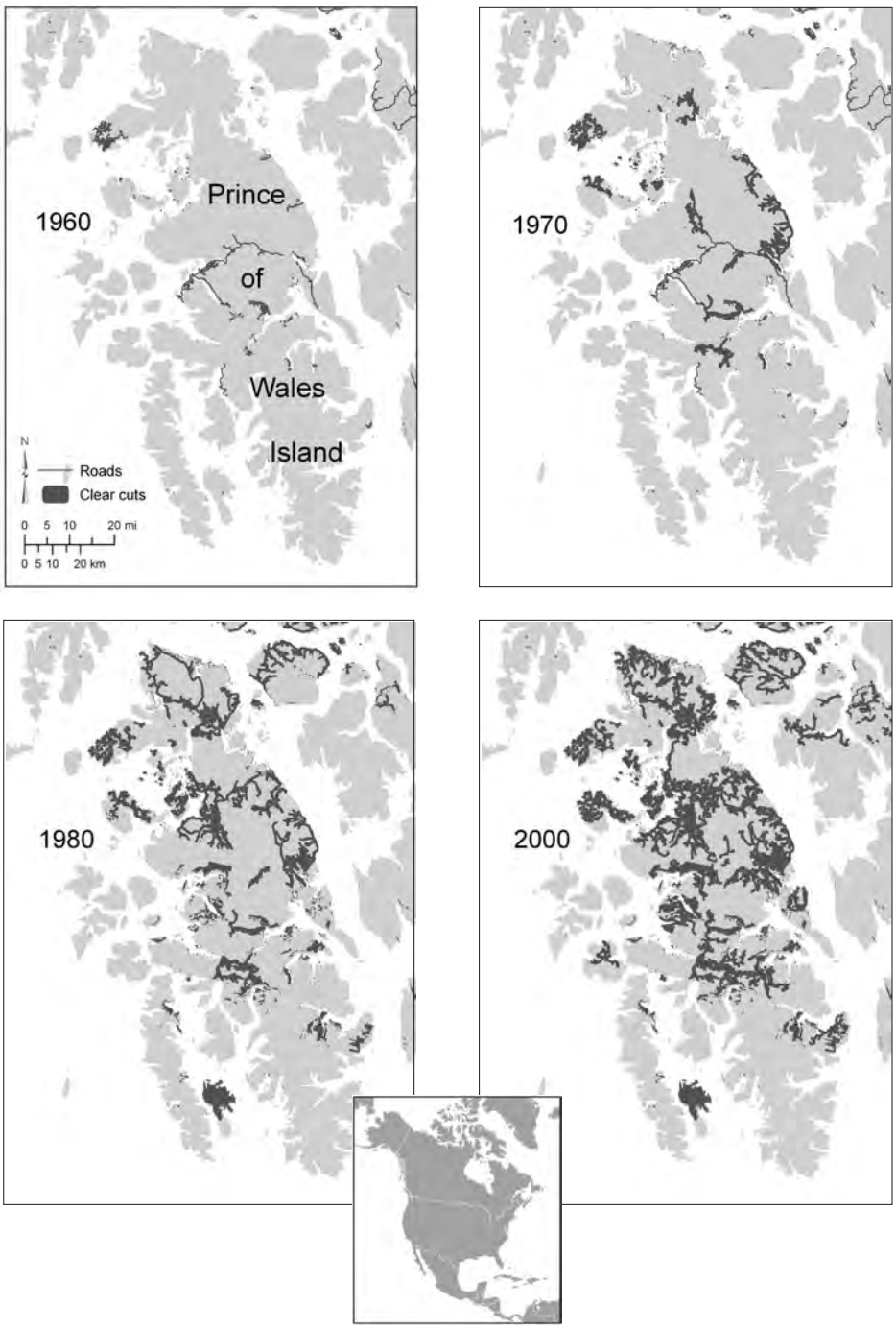

Figure 2-2. Chronosequence (1960–2000) of logging and road building on the Tongass National Forest, Prince of Wales Island, southeast Alaska. Source: Conservation Biology Institute and World Resources Institute.

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current Tongass Land Management Plan, nearly 6,000 kilometers of new logging roads could be constructed and an additional 180,000 hectares of old forest logged by the end of this century (USDA Forest Service 2008).

In response to these concerns, The Nature Conservancy and Audubon Alaska employed a technique known as conservation area design (CAD) to identify and propose for protection intact watersheds of highest conservation value and to prioritize developed watersheds for restoration. The CAD also identifies potential timber harvest areas where logging can be concentrated in the smallest land base within watersheds with preexisting road infrastructure (Schoen and Albert 2007). This approach builds on the existing land-management strategy of the Tongass by adding intact watersheds with conservation priorities to the reserve network in order to more effectively ensure representation of diverse forest types in protected areas widely distributed across the Tongass. For instance, under the current Tongass land allocation, about a third of the forest's habitat values for focal species (salmon, deer, bear, murrelets), estuaries, and large-tree old growth are strictly protected. Implementing the CAD would place an additional 34 percent of the forest's habitat values in protected watersheds and another 15 percent in integrated management watersheds (a combination of protection for core old-growth areas, restoration of riparian areas, and some sustainable harvest of upland second growth) (Schoen and Albert 2007).

Great Bear Rainforest and Haida Gwaii (aka Raincoast)

As one of the few remaining large blocks of comparatively unmodified landscapes on Earth, the Great Bear Rainforest and the adjacent offshore archipelago of Haida Gwaii are biologically rich, aesthetically unique, and rare (see figure 2-1; plate 2a). This 7.4 million–hectare region encompasses the North Coast, Central Coast, and Haida Gwaii forest districts of British Columbia (Coast Information Team Report 2004). It includes over a quarter of the Pacific Coastal rainforests of North America (based on the rainforest distribution model, chapter 1) and some of the largest, relatively intact coastal rainforests in the world. The Great Bear and Haida Gwaii also could have been easily dubbed Rainforest Fjords, for the extensive network of mountainous fjords, or the Large Carnivore–Salmon Rainforest for the keystone role that salmon provide to apex predators like wolves and bears.

Like the Tongass, the Great Bear Rainforest and Haida Gwaii harbor one of the last opportunities for studying the outcome of long-term evolution on a geographic scale, and observing highly specialized and coevolved interactions that are being replaced elsewhere with invasive species or intensively managed

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landscapes (Paquet et al. 2006). Unfortunately, the rate of human-induced environmental change has been and may continue to be so rapid (Moola et al. 2004) that many species may not be able to keep pace with accelerating habitat losses.

Although the Raincoast has remained remarkably inaccessible when compared with other British Columbia regions, nevertheless extensive habitat loss and fragmentation have occurred in the southern portion of the Great Bear Rainforest and throughout much of central and northern Haida Gwaii. Even though most of the remaining forested land base is considered unsuitable for forestry (Pojar et al. 1999), many of the unprotected and biologically rich valley-bottom, old-growth forests have been logged or have been leased for logging to several major forestry companies and First Nations. Furthermore, new technologies such as helicopter logging provide access to old-growth timber once thought to be out of the reach of commercial interests (e.g., steep-forested slopes).

These magnificent forests are home to prodigious salmon runs (~2,500—Temple 2005), grizzly bears, wolves, Queen Charlotte goshawks (*Accipiter gentilis laingi*), and marbled murrelets (*Brachyramphus marmoratus*). Notably, a unique white phase of black bear, the Kermode bear (*U. americanus kermodei* or "Spirit Bear" (see plate 2b) because of its spiritual significance to First Nations, ranges from Roderick-Pooley and Princess Royal Islands in the south to the Nass Valley in the northern portion of the Great Bear Rainforest of British Columbia. Roughly 1 in 10 bears are born with a white coat (Ritland and Marshall 2001). Gribbell Island (20,600 hectares) on the British Columbia Central Coast has been identified in recent genetic studies as one of four important nearshore islands on the coast for this subspecies. Over 30 percent of the small bear population on the island are estimated to be white-phase bears, and this island is "richest in white bears" and has "substantial genetic isolation" (Ritland et al. 2001). Apparently, "Kermodism" has been established and maintained in populations by a combination of genetic isolation and somewhat reduced population sizes in insular habitat. Further, McCrory and Paquet (2008) consider the small insular population of between 90 and 130 bears to be vulnerable to ongoing logging and other human disturbances.

Fortunately, a recent land-use agreement between officials from the British Columbia government and local First Nations increased protection of forests from all industrial development in the Great Bear Rainforest portion of the Raincoast from ~9 percent to 28 percent (2.1 million hectares). An additional 5 percent of the land base (370,000 hectares) is off limits to logging but is not strictly protected from mining, road-building and other development. The remainder of the region's forests (so-called "matrix landscapes") are available to

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special forest management, called ecosystem-based management (EBM—see box 2-2) under which the amount of old-growth forest that can be logged across the landscape, in each watershed, and in each ecosystem type will be capped. Across the region as a whole a minimum of 50 percent of the natural level of old-growth forest of each ecosystem type will have to be maintained (or recruited over time in areas that have been heavily impacted already) under new EBM regulations. This translates to an additional 700,000 hectares of old-growth forest that will be off limits to logging (but not other development) outside of formally designated conservation areas in the region. Over the next

BOX 2-2

Ecosystem-Based Management of the Great Bear Rainforest: Solution or Incomplete Strategy?

The Great Bear Rainforest Agreement includes ecosystem-based management of industrial forests situated outside of the formally designated protected areas (i.e., the matrix landscape). These "EBM Zones" cover most (~70 percent) of the Great Bear Rainforest, including most of the timber harvesting landbase (e.g., valley bottoms and adjacent mid-slopes targeted by the logging industry—Martin et al. 2004), and the most biologically valuable areas in the region—areas of relative intactness and high aggregate conservation value (Rumsey et al. 2004). Though EBM was narrowly defined by an independent science panel for the region as a comprehensive suite of "best-practices" for forestry operations at multiple spatial scales (e.g., protection of rare and endangered plant communities, establishment of unlogged buffers adjacent to high-value fish habitat—see www.citbc.org), not all of the prescriptive elements recommended by the science panel have been legislated. Rather, EBM remains subject to further negotiations among First Nations, government, and other stakeholders, and is supposed to be fully implemented by 2014.

Nevertheless, there remains great uncertainty as to the ecological efficacy of newly established EBM planning and on-the-ground logging practices to maintain biodiversity on the landscape outside of formally designated protected areas. The lack of a representative reserve network necessitates establishment of a rigorous EBM process that eventually incorporates the 40–70 percent protection targets recommended by scientists through additional set-asides.

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five years, government, environmental groups, industry, and First Nations have committed to revise the EBM regulations to increase the amount of old growth that will have to be maintained on the land base outside of protected areas (i.e., maintain 70 percent of natural levels of old growth over time). However, much uncertainty remains with the EBM process, which took 9 years to develop and another 5 years to implement, while logging continues. This means that considerable more old growth and biological diversity will have been lost by the time the agreements are finally implemented in 2014.

An accompanying recent land-use agreement with local First Nations in Haida Gwaii increased overall protection (strictest designation) on the islands to 0.5 million hectares. At least half of this archipelago is now protected and, as in the Great Bear Rainforest, new logging practices under EBM agreements will apply to forestry operations outside of established parks and protected areas.

Overall, the Great Bear Rainforest and Haida Gwaii agreements provide a significant increase in the overall area now strictly protected across the Raincoast (2.6 million hectares; 35 percent) from logging and all other industrial practices (e.g., mining) in parks, conservancies, and other legally designated areas. However, concerns have been raised over the inadequate size and spatial configuration of the protected areas network (Gilbert et al. 2004; Martin et al. 2004), incomplete representation of ecosystems (Wells et al. 2003), including coastal islands and productive old-growth stands (Paquet et al. 2004), inadequate levels of protection and omissions of key areas of critical habitat for focal species such as bears and wolves that were recommended by scientists (Gonzales et al. 2003; Martin et al. 2004; Paquet et al. 2004), and lack of full protection from all potentially deleterious human impacts (Gilbert et al. 2004; Moola et al. 2004). Perhaps more fundamentally, because much of the Great Bear Rainforest and Haida Gwaii are still relatively intact compared to other regions, it presents one of the last opportunities on the planet to conserve large, expansive wildlands (including old-growth rainforests) in their natural state.

For these and other reasons, both the Great Bear Rainforest and the Tongass are globally significant (Ricketts et al. 1999) in that the core elements necessary for conservation are mostly present and not in need of restoration at this time (e.g., predator-prey relationships are still intact—see Noss 2000). In particular, this region is especially deserving of core and comprehensive protection at the regional scale to ensure the persistence of these ecological values in perpetuity. For the current agreement to provide lasting conservation benefits, it needs to expand the size of protected areas that are presently still too small to maintain viable populations of wide-ranging species such as grizzly bears,

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coastal wolves, and salmon; to increase connectivity among existing and newly created protected areas; and to improve representation of rare, sensitive, and at-risk ecosystems such as islands and intact watersheds, along with rare, large old-growth trees.

Finally, ecosystem management and forestry operations throughout British Columbia should include both more responsible practices such as those of the Forest Stewardship Council (FSC),[1] and also protection of critical habitat of declining species (see box 2-3). Notably, about 1 million hectares of the EBM zones were recently certified under FSC standards, although it is unclear whether certification will make marked improvement to on-the-ground EBM measures at this time. Improved logging practices outside protected areas, in general, would reduce extirpation risks of vulnerable species (e.g., wolves, bears). But ultimately the reserve network needs to be expanded to include protection of large blocks (greater than 5,000 hectares) of intact and underrepresented watersheds, as these areas are crucial to salmon and other wildlife (e.g., large carnivores) and few intact watersheds this large remain outside Alaska and northern British Columbia (Beebe 1991). Given that the most rapid human-caused extinctions have occurred on islands worldwide (Pimm 1991), existing land-use plans for the Great Bear Rainforest and Haida Gwaii, like the Tongass, need to address fully the archipelago environment and associated extirpation risks.

Clayoquot Sound

The west coast of Vancouver Island (see figure 2-1) includes 265,000 hectares of mostly old-growth rainforest within the larger Clayoquot Sound region (3 million hectares). Vancouver Island has lost three-quarters of its productive old-growth forest (e.g., valley bottoms where the biggest trees typically grow) to logging, mining, agriculture, and urban development to date (www.viforest.org). Currently, ~35 percent (91,400 hectares) of the Sound's forest is under strict legal protection in parks and other protected areas (however, see table 10-1 for minor differences in protected-areas estimates based on GIS analysis). An additional 67,800 hectares within the Scientific Panel Watershed Reserves (EBM zones) is off limits to logging, but are still vulnerable to mining, roads, and other land-use activities. These areas were not included in the protected-areas database used in Chapter 10, as they were not strictly protected, but they do count toward assessments of regional protections.

[1] www.fsc-bc.org/britishcolumbia.htm

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BOX 2-3

Endangered Birds of Southwest British Columbia: Going, Going, Gone?

The southwest Coast Mountains of British Columbia are home to the northern spotted owl, a forest raptor clinging to existence in the only place where it is found in Canada. An apex avian predator of older forests, fewer than a dozen owls remained in the wild in 2008, until the Canadian government removed most of them to establish a captive breeding program. Unfortunately, Canada's Minister of the Environment at the time denied habitat-protection measures mandated under Canada's endangered species law (i.e., the Species At Risk Act), deciding instead that "the northern spotted owl does not currently face imminent threats to its survival or recovery." This surprising decision was made as owl habitat continues to be logged, primarily by the government itself, and where over 80 percent of the old growth has already been logged since the 1940s.[a] Therefore, if conservation measures are not taken soon, the owl will be extirpated from its rainforest habitat within a wing beat of its evolutionary time line.

But the spotted owl is not the only bird in trouble. An unusual coastal seabird, the marbled murrelet, has declined by 70 percent over a 25-year period throughout Alaska and British Columbia (Piatt et al. 2006). This robin-sized seabird nests reclusively in clumps of moss in the tallest and oldest rainforest trees, returning infrequently to feed its young the catch-of-the day caught at sea. Cumulative impacts from climate-related changes in the marine ecosystem and human activities (e.g., logging of nest sites, forest fragmentation, gillnet bycatch, oil pollution) are suspected in the bird's decline (USFWS 1997). Immediate conservation is needed to stem further losses.

[a]See www.davidsuzuki.org/Forests/Canada/BC/Spotted_Owl.asp

The Sound's surroundings consist of highly dissected mountains blanketed by coastal rainforests and peppered by fjords and salmon-bearing rivers, although portions (e.g., the Upper Kennedy River) of this region are actually within the transitional rainforest type (Clayoquot Sound Scientific Panel 1995). Recognizing these distinctions, researchers subdivided the Sound into three biogeoclimatic units based on elevation, vegetation, and maritime influences. Within the units, old-growth rainforests are characterized by trees ranging from

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saplings to 1,000-year-old giants (Clayoquot Sound Scientific Panel 1995). Notably, most forest-dwelling species in this region nest in riparian (streamside) areas (72 percent), followed by older (greater than 140 years) forests (46 percent), and lastly by young, even-aged plantations (9 percent—Clayoquot Sound Scientific Panel 1995).

In 2000, Clayoquot Sound was designated a UNESCO Biosphere Reserve, though this was purely symbolic and did not result in any additional protected areas. Logging (under new EBM rules), fish-farming, exploratory drilling for a potential open-pit copper mine, and other ecologically damaging practices continue. Numerous open net-cage salmon farms now populate the Sound, as well as elsewhere in British Columbia. Open net-cage salmon farming has been linked to parasitic sea lice in juvenile salmon; toxic-waste effluent, including methylmercury contaminants, fish dyes, and growth hormones; and the release of antibiotics (Krkosek et al. 2005). Farmed Atlantic salmon kept in pens also are known to escape, where they genetically dilute wild Pacific stocks through hybridization.

To keep Clayoquot Sound's global status as a UNESCO Biosphere Reserve, we recommend that the British Columbia government legally protect remaining intact valleys, co-managed with First Nations, and provide funds in support of a conservation-based economy and restoration of degraded watersheds. The use of closed-containment facilities for farming salmon, combined with the elimination of exotic Atlantic salmon and improved waste water treatment, would dramatically elevate conditions for wild salmon in the Sound and throughout British Columbia in general.

SEASONAL RAINFOREST CONSERVATION PRIORITIES

Seasonal rainforests stretch from central Vancouver Island along the United States Pacific coastline to southern Oregon, with a secondary band farther inland along the western Cascades, a distance of some 604 kilometers (see figure 2-1). These rainforests are cloaked in fog and ocean spray, with annual precipitation ranging from ~2,000 to 4,080 millimeters. Contrary to the near year-round rain in the perhumid zone, most rainfall is distributed in the fall and winter months, with less than 10 percent in the dry summer (Alaback and Pojar 1997). Because of intermittent seasonal rainfall and the irregular but more frequent occurrence of fire (stands replaced every 90 to 250 years; Spies 2004), some researchers have elected not to classify them as temperate rainforest. However, we included them as well as the warm-temperate rainforests to the

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south because of their coastal association and similarities in plant and animal communities with their northern counterparts (also see Alaback and Pojar 1997). Conservation priorities in this zone include the Pacific Northwest, Siskiyou Wild Rivers (a subset of the Pacific Northwest Coast), and the Western Cascades (old forests).

Pacific Northwest

Pacific Northwest rainforests (4.8 million hectares) extend from the Olympic Peninsula to southern Oregon (see figure 2-1). They are ecologically more diverse than northern rainforests and are dominated by Douglas-fir, western hemlock, western red cedar, Sitka spruce, and shore pine (especially where salt spray is a factor), mixing with broadleaf hardwood understories in places. Some of world's largest hemlock and spruce are found on the Olympic Peninsula, where trees can exceed 2 meters in diameter and tower to over 60 meters (Kirk and Franklin 1992). Olympic rainforests are rich in plants from the ground up, with numerous ferns and shrubs blanketing the understory and unique epiphytes on the tops of the tallest trees (Kirk and Franklin 1992, plate 3).

Siskiyou Wild Rivers

At the terminus of the seasonal coastal rainforest in southwest Oregon is a distinct area known as Siskiyou Wild Rivers. This ~405,000-hectare area stands out as the "Pacific Coastal Outback" because it contains the largest complex of coastal roadless areas between the Mexican and Canadian borders. It supports one of the most prolific wild salmon runs south of Canada, exceptional concentrations of endemic plant species, large stretches of undeveloped and scenic rivers, and high levels of amphibian and avian richness (DellaSala et al. 1999). The western, coastal portion of this area is rainforest: plant communities farther inland are characteristic of drier, more fire-adapted forest types, although high-elevation areas retain rainforest characteristics due to increased moisture levels.

Western Cascades

This secondary rainforest band farther inland (see figure 2-1) includes coniferous forests dominated mainly by Douglas-fir, western hemlock, and western red cedar, and is distributed within three major physiographic provinces: the Northern Cascades, the Southern Washington Cascades, and the Western Cascades (Franklin and Dyrness 1973). Rainforest transitions to drier types at the Cascade Crest heading eastward toward drier continental conditions.

Old-growth rainforests have declined throughout each of these regions, from nearly two-thirds of the forest age classes historically to roughly 20

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BOX 2-4

Impacts of Logging and Road Building in Temperate and Boreal Rainforests of the Pacific Coast of North America.

Logging-related disturbances can have significant impacts on rainforest biota, primarily from the mechanical disruption of understory plants and forest soils, and from the oversimplification of the composition and structure of tree species, including the loss of coarse, woody debris and legacy trees (old trees and structures carried over from the pre-disturbed forest). Repeated logging on the same site every 30–100 years creates a long-term "successional debt" whereby old-growth forest is depleted and thus wildlife habitat and other ecosystem services degraded. Following logging, single species of trees (monocultures) are often planted in tight rows (in the north, however, natural regeneration precludes the need for replanting), creating impoverished forests compared to the complex, multispecies, multilayered rainforests they replaced.

The long-term adverse consequences of clearcut logging and road building are many and include greater vulnerability of deer to loss of optimal winter habitat (Schoen et al. 1988) and human-caused wildlife mortality along roads, particularly for brown bears, wolves, and marten (Schoen et al. 1994; Person et al. 1996; Trombulak and Frissell 2000; Person 2001; Flynn et al. 2004). Other terrestrial consequences include decline of wolves dependent on deer as a primary prey species, and increased likelihood of conflicts between hunters and wolves for deer (Darimont and Paquet 2002). Federally threatened species like the northern spotted owl and marbled murrelet have declined precipitously due, in large part, to loss of old-growth habitat caused by high rates of logging. Aquatic consequences include chronic degradation of salmon habitat by erosion and siltation, increased water temperature due to loss of forest-related shading along streams (see Reeves et al. 2006), and greater risk of mass-wasting events (landslides) associated with road building and logging (Guthrie 2002).

Many plants and animals, particularly salmon, are now listed under the U.S. Endangered Species Act and/or the Canadian Species at Risk Act, due, in part, to logging, although legal listing has not necessarily led to effective protection of critical habitat (Yezerinac and Moola 2006). Consequently, resource managers need to consider long-term consequences of

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BOX 2-4
Continued

forest management (including roads and logging techniques) on predator-prey dynamics and threatened species, recognizing that mitigation of those consequences may not always be possible. The archipelago landscape of coastal Alaska and British Columbia may amplify such impacts, given the greater vulnerability of island populations to extirpations (Hanley et al. 2004, Cook et al. 2006).

percent today (this estimate includes all temperate forests and not just rainforest—Strittholt et al. 2006). Notably, few old-growth rainforests remain along the coastline from southern Oregon to the Olympics, where staggered logging units have created a "checkerboard" pattern of alternating rainforest and clearcuts, with much of the remaining unlogged rainforest restricted to federal lands. Logging, particularly of older forests, has impacted the area in many ways (see box 2-4).

The three regions are no strangers to logging conflicts, as made widely known by the 1990 listing of the northern spotted owl (*Strix occidentalis caurina*) as a federally threatened species (other species listings soon followed). A court injunction in 1993 shut down logging on federal lands, resulting in a compromise, known as the Northwest Forest Plan, which lowered allowable logging levels on federal lands by 80 percent while placing about a third of the region's public forests in late-successional reserves (LSRs). Since 1994, nearly 9.8 million hectares of federal lands have been managed under this plan, a global model in biodiversity protection and ecosystem management (DellaSala and Williams 2006). While the region is managed under various land-use categories, only about 5 percent is strictly protected by parks and wilderness areas (see chapter 10), although lower levels of protection are also provided on federal lands managed under the Northwest Forest Plan as late-successional reserves. There have been administrative attempts to weaken this reserve network, and logging has occurred in some reserves following wildland fire (DellaSala and Williams 2006).

Notably, the conservation foundation for the Northwest Forest Plan on federal lands is a state-of-the art reserve network developed by scientists (FEMAT 1993) to maintain the viability of old-growth associated species, primarily the northern spotted owl and marbled murrelet. Despite its significance

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as a global conservation model, only about 60 percent of the ~3 million hectares of reserves actually include old-growth forests (older than 150 years), with the rest made up of plantations that over time (50–100 years) may eventually acquire, through restorative silvicultural actions, older forest attributes (Strittholt et al. 2006). In addition, reserves are not inviolate, as forest thinning is permitted under certain conditions and post-disturbance logging has degraded reserves in places (e.g., Siskiyou Wild Rivers). Significant amounts of older forests (e.g., 0.4 million hectares of late-successional forest) remain in the "matrix" where logging is concentrated. In particular, logging on state and private lands is 3 to 4 times greater than on federal lands (Staus et al. 2002), making federal lands stick out as the last stronghold for old-growth rainforests and intact watersheds in this region. Invasive species, spread by logging operations, road building, livestock, and other dispersal agents, threaten to replace native species. Sudden Oak Death syndrome, caused by an introduced fungal species (*Phytophthora ramorum*), has crept into southern Oregon, where it threatens a host of trees, including coastal redwoods just to the south. Climate change may facilitate the spread of at least some of these deleterious invaders.

Conservation priorities for all three areas include:

- Protect remaining mature (100 years old) and old-growth forests on public lands; mature forests provide replacement trees for older ones that die from natural disturbances such as fire and wind throw.
- Prioritize intact watersheds and roadless areas (e.g., Siskiyou Wild Rivers) for protection as climate refugia for salmon, and other areas for restoration, particularly where road densities are high (e.g., the Oregon Coast Range).
- Implement invasive-species containment measures, particularly by curtailment of vectors that facilitate their spread (e.g., roads, livestock grazing, logging, and vehicles).
- Strengthen federal protection for threatened species, including northern spotted owl, marbled murrelet, and coho salmon, by increasing habitat protections for older forests and roadless areas.
- Thin forests by removing small trees in order to restore forest structure and composition in overly simplified tree plantations, and extend timber harvest rotations to grow older forests.
- Promote responsible forest management (such as FSC certification) on nonfederal lands to complement conservation strategies on federal lands.
- Designate new protected areas, particularly relatively intact areas spanning elevation gradients and important migration and travel corridors in order to allow species to find new habitat in response to climate change.

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WARM RAINFOREST CONSERVATION PRIORITIES

Warm-temperate rainforests stretch in a long, narrow, and discontinuous band (skipping San Francisco Bay) no wider than 50 kilometers all the way from just above the Oregon-California border to just below the Bay area (see figure 2-1). This zone, which includes coastal redwood, is characterized by mild, wet winters and cool, dry summers on the coast with warmer dry summers inland. Less than 5 percent of the annual rainfall occurs during dry summer months, and snow is rare. Annual precipitation in the northern redwoods (Del Norte and Humboldt counties) exceeds 3,200 millimeters (Sawyer et al. 2000); it is generally drier southward.

Few trees on Earth are as impressive as coastal redwoods in tree size or age (see Sawyer et al. 2000 for trees over 100 meters tall). Only the *alerce* of Chile (see chapter 5), Huon pine (*Lagorastrobus franklinii*) of Tasmania (see chapter 8), and mountain ash (*Eucalyptus regnans*) of southeastern Australian (see chapter 8) approximate redwood trees in age or stature. Overall, the redwood region is globally significant (Ricketts et al. 1999) because it is the only place on Earth where coastal redwoods thrive, and because the area has higher species richness and more endemics than any of the Northern Hemisphere's rainforests (Alaback 1995; Noss et al. 2000).

A redwood's immense stature is enabled, in part, by its immersion in year-round fog that helps to meet moisture requirements, as well as by the biogeochemical properties and hydrological cycles of this rainforest (Sawyer et al. 2000). Interestingly, when redwood forests are cut down, fog production declines due to diminished evapotranspiration (Sawyer et al. 2000). Fire occurs infrequently (once every 250 to 500 years) in northern redwoods, more frequently inland and to the south (33 to 50 years), and more so in upland areas (less than 17 to 175 years—Sawyer et al. 2000). However, the more frequent fires are mostly ground based and this has allowed redwoods to reach enormous size and impressive longevity.

Redwood forests cover an estimated 647,000 (Sawyer et al. 2000) to 877,396 hectares, from extreme southwestern Oregon to southern Monterey County, California (35°N latitude).[2] Noss et al. (2000) subdivided the region into northern, central, and southern zones (with 25 additional subsections), owing to differences in plant assemblages, precipitation, and the influence of fire. In general, northern redwood forests more closely resemble rainforest counterparts to the north (in terms of conifer species composition), while central and southern redwood forests are a mixture of conifer and hardwood trees

[2]www.frap.cdf.ca.gov/data/frapgisdata/download.asp?rec=redwood/

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(Sawyer et al. 2000). In addition to coastal redwood, these rainforests include white fir (*A. concolor*), sugar pine (*P. lambertiana*), incense cedar (*Calocedrus decurrens*), ponderosa pine (*P. ponderosa*), Sitka spruce, western red cedar, western hemlock, and a rich assortment of understory hardwoods, shrubs, ferns, and forbs (Sawyer et al. 2000).

The wildlife composition of redwood rainforest is similar to that of the transitional temperate rainforest to the north, with considerable overlap in species distributions (Cooperrider et al. 2000). Much of the richness of these forests is invisible to the untrained eye. At the top of the tallest trees, a diverse canopy ecosystem consists of unique invertebrates, epiphytes, mosses, lichens, small mammals, and even some salamanders that never venture from the rainforest canopy and some that never leave the same tree (see Noss et al. 2000). Likewise, the soil fauna are exceptionally rich, even more so than in tropical rainforests (Cooperrider et al. 2000).

Less than 4 percent of the redwood forests remain intact (Ricketts et al. 1999) and much of this is vulnerable to logging, as the region is mostly (83 percent) in private ownerships. Noss et al. (2000) indicate about 13 percent of the redwood region is strictly protected (but see table 10-1 for differences), and most of this is limited to just three areas in northern California—Humboldt Redwoods State Park, King Range National Conservation Area, and Redwoods National Park. Consequently, in the absence of additional conservation measures, old-growth redwood forests will remain restricted to a few isolated parks, important on many levels, but not in the form of intact ancient redwoods. (Much of the parks' forests are now recovering from logging prior to their protective designations.) Principal threats include the logging of remaining redwood groves (clusters of huge trees) mainly on nonfederal lands, exotic-species invasions (including Sudden Oak Death, which is starting to show up in redwood), and decline of salmon runs primarily from logging and road building (see Noss et al. 2000). Climate change brings uncertain prospects for these coastal giants, particularly as coastal fog has declined markedly in recent decades, as attributed to climate change (Johnstone and Dawson 2010).

Perhaps nowhere else in the coastal rainforest region is conservation more critical and urgent than in the redwoods. Without stepped-up conservation, the few remaining old-growth redwood forests will become outdoor museum pieces, reminders of a bygone era when the redwoods prospered. In response, Noss et al. (2000) introduced a three-part conservation strategy based on a CAD approach that recommends: (1) increased representation of various ecosystem types in strictly protected reserves for all three subregions; (2) conserva-

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tion of focal species such as the Pacific fisher (*Martes pennanti pacifica*)—an old-growth-redwood meso-carnivore; and (3) protection of special elements such as "hot spots" of rare species, critical watersheds, and redwood groves. Building on the work of these scientists, the following conservation priorities for the redwood rainforest region are recommended:

- Establish new protected areas by purchasing redwood groves on private lands (for land acquisition priorities—www.savetheredwoods.org).
- Restore (regrow) degraded forests and watersheds as in several of the redwood parks.
- Greatly expand invasive-species and vector-containment measures, particularly through additional research on Sudden Oak Death.
- Maintain viable populations of focal species such as the Pacific fisher.
- Research the effects of climate change, including development of appropriate strategies that better enable redwood forests to adapt.

WHAT WILL IT TAKE TO SAVE PACIFIC COASTAL RAINFORESTS?

North America's coastal rainforests have stood the test of time against a backdrop of volcanic eruptions, retreating glaciers, wind storms, and occasional fires. As glaciers retreated, new areas opened up for colonizing species with ancient Beringia, continental, and coastal affinities. Today, tidal glaciers in the north are melting and logging has eliminated many of the region's giant trees. The naturally fragmented nature of the northern rainforest archipelago has served as a cradle of evolution, an outdoor laboratory for island biogeography, and a wake-up call alerting us to the vulnerabilities of rainforest island systems to human disturbances that often exceed the capacity of rainforest species to adapt.

It is the dawn of a new era for these remarkable rainforests, and their fate will be determined by whether prudent conservation measures are adopted by local, regional, and national governments in a time of accelerating global climate change. Both the U.S. and Canadian governments have demonstrated that enlightened conservation leadership can secure an enduring legacy for rainforest species and human communities. Although rainforest conservation will continue to be informed by the best available science of the times such as that provided by the scientific panels in British Columbia, Clayoquot Sound, and the Pacific Northwest, its mercurial nature depends on the will of the people, the politicians they elect, and the ongoing search for innovative and responsible

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solutions to humanity's ever-growing environmental footprint. Ultimately, these remarkable rainforests and the human societies around them are joined at the hip, "like grizzly bear to salmon and forest to rain." If we allow the remaining intact areas and the vulnerable species in them to drift toward oblivion, humanity will suffer irreplaceable losses, including the inability to experience rainforest giants that sprang from tiny seedlings when indigenous people and the earliest explorers paid homage to their splendor.

North America's coastal rainforests are indeed deserving of stepped-up attention if they are to persist in these challenging times. This is urgently needed in order to ensure that the rainforest giants in coastal redwood forests and older forests of the region, rich early successional rainforests, roadless watersheds, and the apex carnivore–salmon food web continues to flourish. Humanity will increasingly depend on the myriad ecosystem services these rainforests have provided through the ages. But these rainforests will only continue to nurture us if they are protected and properly managed.

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Climate Change May Trigger Broad Shifts in North America's Pacific Coastal Rainforests

DA DellaSala, Geos Institute, Ashland, OR, USA; Oregon State University, Corvallis, OR, USA
P Brandt, Karlsruhe Institute of Technology, Karlsruhe, Germany; International Livestock Research Institute (ILRI), Nairobi, Kenya
M Koopman, Geos Institute, Ashland, OR, USA
J Leonard, Geos Institute, Ashland, OR, USA
C Meisch, Leuphana University Lüneburg, Lüneburg, Germany
P Herzog, Martin Luther University Halle-Wittenberg, Halle, Germany
P Alaback, University of Montana, Missoula, MT, USA
MI Goldstein, University of Alaska Southeast, Juneau, AK, USA
S Jovan, Portland Forestry Sciences Lab, Portland, OR, USA
A MacKinnon, BC Forest Service, Victoria, BC, Canada
H von Wehrden, Research Institute of Wildlife Ecology, Vienna, Austria

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Introduction

Climate change threatens biodiversity and ecosystem integrity all over the globe (IPCC, 2014) and is already triggering pronounced shifts of species and ecosystems (Chen et al., 2011; Parmesan et al., 2000). Climate change is also expected to exacerbate effects of forest fragmentation (Bossuyt and Hermy, 2002; Opdam and Wascher, 2004), especially where only small fractions of formerly intact ecosystems remain (Heilman et al., 2002), presumably by magnifying local edge effects (Chen et al., 1995; Harper et al., 2005) and by reducing opportunities for dispersal and range expansion (Thompson et al., 2009; Watson et al., 2013). Thus, mitigating such effects in areas of global conservation importance is critical as biodiversity losses are especially significant.

The conservation importance of the coastal temperate rainforest region of North America is exemplified by the inclusion of six World Wildlife Fund Global 200 ecoregions (Ricketts et al., 1999), some of the most carbon dense ecosystems on earth (Leighty et al., 2006; Smithwick et al., 2002), extraordinarily productive salmon (*Oncorhynchus* spp.) runs and relatively intact forests northward (DellaSala et al., 2011). The highest epiphytic lichen biomass of any forest system also occurs here (McCune and Geiser, 2009). Thus, maintaining extant biodiversity in a changing climate has biodiversity significance on a global scale given the region's importance.

Already confirmed climate change effects in this region include elevated temperatures (Karl et al., 2009), declining mountain snowpack (Mote et al., 2005), shifts in species distributions (Wang et al., 2012), and reduced fog levels (Johnstone and Dawson, 2010). Diminished snowpack combined with late winter freezes has triggered dieback of Alaska yellow-cedar (*Cupressus nootkatensis*) in southeast Alaska (Hennon et al., 2012) and northern British Columbia (Wooten and Klinkenberg, 2011).

Vegetation along the northern Pacific coast has been sensitive to climatic changes since the last glaciation, resulting in large shifts in species distributions, and providing strong evidence that future climate change will result in substantial ecological changes (Brubaker, 1988; Heusser et al., 1985). Even small changes in temperature often result in large species displacements, which

explains contemporary pattern of species distributions along the coastal region (Alaback, 1996). A 125 000-year record of vegetation change from the eastern slope of the Cascades, for example, shows that while species movements are individualistic, depending on species characteristics and geography, at the millennial scale global climatic variation is the dominant factor controlling vegetation distribution (Whitlock and Bartlein, 1997). Conifer species' distributions have changed since the glacial maximum reflecting differences in dispersal ability, effects of refugia, and changes in glacial dynamics from central Alaska southward. The physiography of the region, with north-south tending cordillera, has facilitated species movements, helps explain the rarity of species extinctions in the past, and importance of dispersal in the future if species are to adapt to even more abrupt climatic changes. Additionally, dramatic changes in vegetation in the past 20 000 years (Whitlock, 1992) corresponded to warming of 2.5–7.8 °C (median values, including uncertainty) that is similar to what most general circulation models (GCMs) predict for the Western USA by the end of the twenty-first century (IPCC, 2014).

There is no broad adaptation plan that addresses potential range-wide shifts of ecologically and commercially valuable species in this region, although there is a growing body of relevant adaptation work as reflected by the North Pacific Landscape Conservation Cooperative (NPLCC) of the U.S. Fish & Wildlife Service (http://northpacificlcc.org, accessed October 14, 2014). Our primary objectives were therefore to: (1) model current potential distributions of focal conifers considered of commercial importance to land managers and to project future potential distributions of focal species and broad rainforest vegetation types in response to anticipated climate change; (2) identify areas that may exhibit higher vegetation stability, including those in currently protected areas where biodiversity conservation is emphasized; and (3) illustrate how uncertainty can be addressed in designing effective adaptation strategies in a changing climate.

Notably, attempts to predict future shifts in species' ranges have employed a variety of approaches. One widespread approach, climate envelope modeling, considers the climate conditions where a species is currently or historically distributed and estimates where those same suitable climate conditions are expected to be found in the future based on GCM outputs. This approach has both benefits and shortcomings, which have been thoroughly reviewed (Wiens et al., 2009). A criticism of climate envelope modeling is the strict focus on climate variables with little to no consideration of non-climate drivers such as competition, predation, soils, elevation, and dispersal. Thus, in our assessment of potential climate change effects, we employed both climate envelope models and a dynamic vegetation model, despite differences in input data and analysis scales, to qualitatively compare gross differences regarding the spatial patterns produced. Using correlative and mechanistic modeling approaches independently might increase the reliability of predictions (see Coops and Waring, 2011; Kearney et al., 2010), reducing uncertainties inherent in relying on any individual modeling effort.

Also, in this paper, our findings are used to illustrate some key concepts in climate adaptation planning for managers wishing to maintain extant biodiversity in a changing climate for a rainforest region that straddles two countries (USA and Canada) and large swaths of public and private lands. Additional analyses not presented, including detailed appendices and datasets, are available online (http://databasin.org/articles/172d089c062b4fb686cf18565df7dc57; accessed October 28, 2014).

North America Pacific Coast Temperate Rainforest Region

The Pacific Coast of North America contains the largest proportion of temperate rainforests in the world, representing 35% of the global total (DellaSala et al., 2011). Stretching from the coast redwoods (38° N), California to northern Kodiak Island and Prince William Sound (61° N), Alaska, these rainforests span a wide climatic gradient (Alaback, 1996). Coastal rainforests are associated with cool, moist oceanic air masses, a narrow range of temperature extremes, high frequency of clouds and fog, and high annual precipitation, with most precipitation in the winter (Redmond and Taylor, 1997) and up to 20% in the summer in northern latitudes (DellaSala et al., 2011). The region consists of four distinct rainforest zones that differ climatically and floristically: (1) subpolar – north of southeast Alaska to Prince William Sound and Kodiak Island; (2) perhumid – southeast Alaska to northern Vancouver Island; (3) seasonal – central Vancouver Island to southern Oregon; and (4) warm – southern Oregon coast to San Francisco Bay area (Alaback, 1996; **Figure 1**).

Climate Data

In order to predict potential shifts in species and rainforest distributions, we used the downscaled WorldClim dataset at 30 arc-s (1-km) resolution (Hijmans et al., 2005). We obtained 19 climatic variables for baseline conditions (1950–2000) and for two future time periods (2050s, 2080s) under the A2A ensemble-high-emissions scenario. This scenario assumes continued global population growth and focus on regional economic growth rather than global collaboration. It is one of the scenarios that most closely tracked the emissions trajectory at the time of our 2012 study. Thus, we used three GCMs: CCCMA-CGCM2 (third assessment, Flato and Boer, 2001), CSIRO-MK2 (third assessment, Gordon et al., 2002), and HADCM3 (third assessment – Johns et al., 2003) that covered a broad range of temperature and precipitation projections spanning dry and wet projections.

For climate envelope modeling, we employed a 1000-km buffer on the coastal rainforest study area to capture the entire current ranges of focal species and potential future shifts. Due to the small distribution of coast redwood, the buffer for the baseline model was set to 100 km around the most outer available localities.

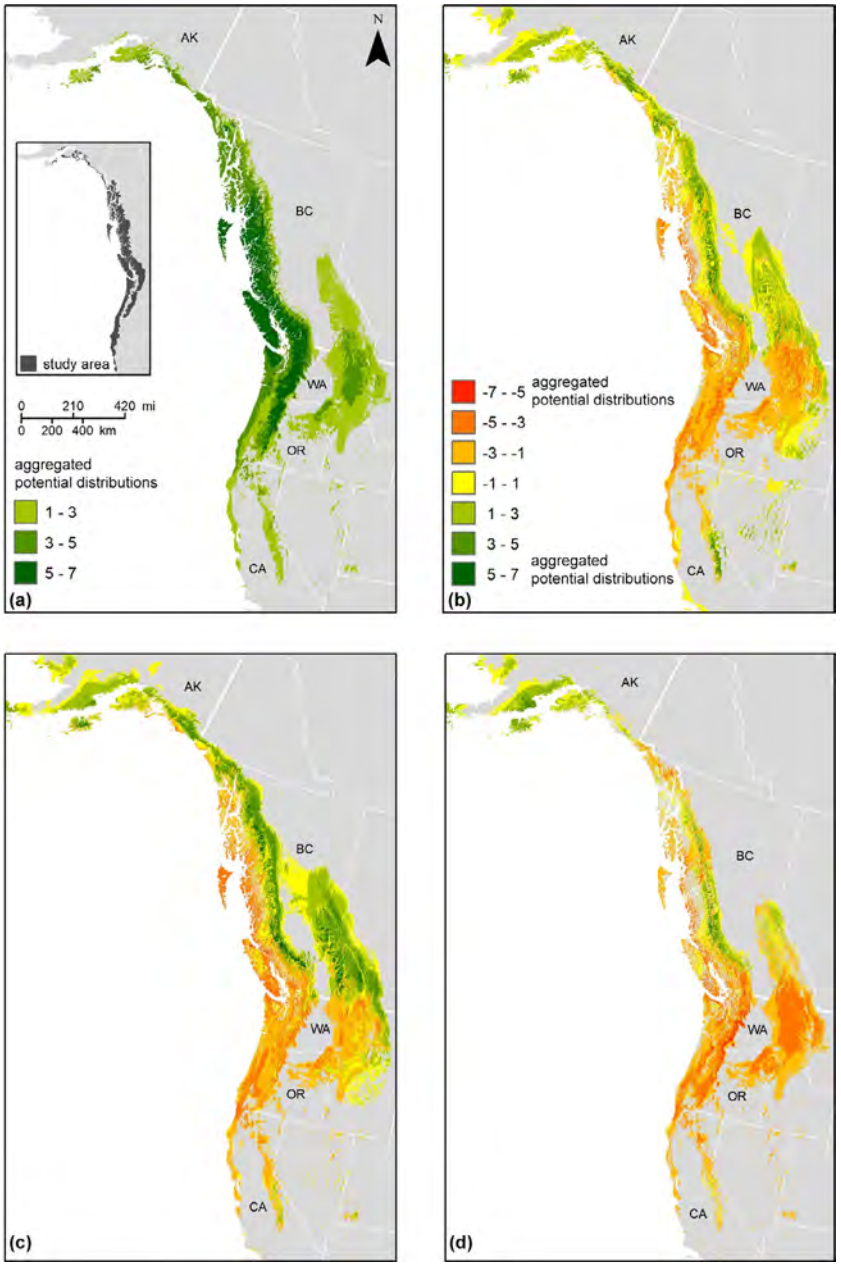

Figure 1 Aggregated potential distribution of eight focal conifer species (Pacific silver and grand fir, Alaska yellow-cedar, Sitka spruce, western red cedar, western and mountain hemlock, coast redwood) for the baseline period (a) and the richness changes for 2080s under scenario A2A ensemble-emissions based on three General Circulation Models (CSIRO (b), CCCMA (c), and HADCM3 (d)).

Selection of Focal Species of Commercial Importance

Based on prior discussions with land managers, we selected eight dominant conifer species of commercial, conservation, and cultural importance to model potential range shifts related to climate change. These species also were chosen because there was readily available location data and their geographic range overlapped primarily with our study area. They included Sitka spruce (*Picea sitchensis*), western and mountain hemlock (*Tsuga heterophylla*, *T. mertensiana*), western red cedar (*Thuja plicata*), Alaska yellow-cedar, Pacific silver and grand fir (*Abies amabilis*, *A. grandis*), and coast redwood (*Sequoia sempervirens*). We did not include other conifers with wide distributions that extended well outside our study area buffer such as Douglas-fir (*Pseudotsuga menziesii*, see Coops and Waring, 2011) or hardwoods (see Hamann and Wang, 2006) given their lower importance to land managers in this region.

Presence-only Modeling of Focal Species

To build the baseline species distribution models, we obtained presence-only data (point and polygon locations) for focal species from numerous databases (USDA Forest Inventory Assessment DataMart v5.1 – apps.fs.fed.us/fiadb-downloads/datamart.html; Biogeoclimatic Ecosystem Classification Program – www.for.gov.bc.ca/hre/becweb/resources/codes-standards/standards-becdb.html, active October 14, 2014; herbaria collections; museum records; published atlases) and from regional specialists that provided more than 30 000 species localities ranging from 710 occurrence points for coast redwood to 7999 points for western hemlock.

Presence-only models outline areas that are predicted as suitable space for a given species according to the predictor dataset (Soberón and Peterson, 2005); these models are known to overestimate realized distributions due to missing information of unvisited locations (Kent and Carmel, 2011). To examine the impact of climate change on species distributions, we only took climatic predictors into account, therefore, focusing on a species' climate envelope (Pearson and Dawson, 2003). Potential distribution was thus determined by projecting this climate envelope across the geographic study area (Soberón and Peterson, 2005).

We applied Maxent 3.3.3k (Elith et al., 2011; Phillips et al., 2006) to model current and future potential distribution for each focal species. Maxent frequently outperforms other presence-only modeling algorithms (Wisz et al., 2008). Instead of real absences, Maxent uses random background points to approximate the best fitting probability distribution for estimating habitat suitability (Elith et al., 2011). We used area under curve (AUC) statistics to assess model discrimination performance (Phillips et al., 2006). All models were replicated 25 times using the bootstrap replicate run type. The final average outputs were used for further analyses. The species datasets were split into 70% training and 30% test data sets randomly chosen for each model run.

We used jackknife procedures from initial model runs to exclude predictors that showed low importance in predicting included presence points when modeled in isolation, expressed by low values of model gain. We activated the 'fade by clamping' option in Maxent to mitigate clamping issues arising from projection values extending beyond the range of training data (Phillips et al., 2006) and chose the logistic output format. The automatic feature selection was applied since it has been validated with respect to a broad range of species, environmental conditions, numbers of occurrences, and degrees of sample selection bias (Phillips and Dudík, 2008). Using ARCGIS 10, the continuous grid outputs of the Maxent models were transformed into binary data showing either potential presence or modeled absence of a given species based on species-specific thresholds that minimized falsely excluded presences while retaining the similarity to published ranges (Little, 1978). Thus, for every species we created one baseline (1950–2000) potential distribution layer and six future potential distribution layers based on the two time periods (2050s, 2080s) and three GCMs.

Identifying Areas of Persistence, Gain, and Loss

For each focal species, we analyzed and mapped differences and commonalities between current and all variants of future potential distributions that were categorized as: (1) 'persistent distribution' where baseline and future potential distributions overlap, (2) 'distribution gain' where baseline potential distributions are absent but future potential distributions are present, and (3) - 'distribution loss' where baseline potential distributions are present but future potential distributions are absent. This is important for managers wishing to assess broad patterns in species distributions related to projected climate changes.

GCM outputs may differ widely, leading to variation in output among different climate envelope projections (Beaumont et al., 2008). Using three GCMs that spanned much of the range of possible futures, from wetter to drier and from faster warming to slower warming, allowed us to assess the level of disagreement among model output as an indirect measure of model uncertainty for managers wishing to plan for future distribution shifts. Importantly, we were able to assess climate envelope model outputs regarding model uncertainty inherent in climatic projections: uncertainty being lowest in areas where future potential distributions of all model projections showed a full consensus (spatial agreement) and highest in cases where they completely differed (Araújo and New, 2007). Obviously, model uncertainty is still inherent based on the complexity of climate and ecological systems, the potential for unexpected events related to climate change, and human behavior concerning greenhouse gas emissions abatement. Nonetheless, we propose that projections with relatively high agreement among models are useful in predicting broad trends important in robust reserve design and forest management decisions.

We calculated Cohen's kappa coefficients (K) (R Development Core Team, 2013 v. 2.1.12), indicating the degree of agreement (Fielding and Bell, 1997) between baseline and future potential distribution for all modeled species in order to quantify possible divergences in potential distributions over time as a proxy for expected shifts in species distribution (online appendix).

Outputs of climate envelope models can also be used to compile richness maps based on aggregated potential species distributions (McKenney et al., 2007). We used binary, aggregated potential distributions of focal tree species as an index of broad potential changes in species richness patterns across the entire study area.

Future Vegetation Stability, Intact Late-Seral Forests, and Current Protection Schemes

In addition to potential species shifts, we used the MC1 dynamic vegetation model outputs, biogeography module (Bachelet et al., 2001) to assess potential stability of dominant types of vegetation under a changing climate. The MC1 model was derived from physiologically based biogeographic rules derived from the MAPSS model (Neilson, 1995) adapted to dynamic environmental gradients using site production information (Bachelet et al., 2001). While the Maxent climate envelope analysis (above) focused on individual rainforest species and species richness, the MC1 output provided information on overall functional types of potential vegetation (temperate coastal needleleaf forest, for example) but not individual species. We compiled MC1 outputs produced under current and future climatic conditions using three GCMs (third assessment models): Hadley (HadCM3; Johns et al., 2003), MIROC (Hasumi and Emori, 2004), and CSIRO (Gordon et al., 2002) under the A2 emissions scenario. MC1 explicitly simulates vegetation dynamics, nutrient cycles and dynamic impacts of disturbance due to fire and has been used in analyses of vegetation responses to climate change (Lenihan et al., 2008). However, MC1 does not incorporate anthropogenic disturbances such as timber harvest, agriculture, urbanization, invasive species introductions, and human-wildfire ignition sources.

All applied MC1 model outputs have a 1/12°, unprojected, grid-cell resolution that is nominally 8-km (Daly et al., 2008). We assessed vegetation stability by comparing the dominant type of vegetation predicted to be supported under modeled baseline conditions (1961–1990) to that predicted to be supported for two future time periods (2035–45 and 2075–85). We identified areas as 'stable' or 'unstable' based on whether the future climate is expected to continue to support the same dominant vegetation type through late-century based on agreement across the three GCMs.

Notably, Pacific coastal temperate rainforests are highly fragmented in southern locales, which may be more vulnerable to large-scale changes in precipitation and temperature if magnified by local edge effects. Therefore, we accessed the most current intact late-seral rainforest datasets to identify areas that overlap with stable vegetation areas as potential refugia (Keppel et al., 2012; Olson et al., 2012; Watson et al., 2013). For intactness, we downloaded the only seamless forest fragmentation dataset available for the entire Pacific coastal temperate rainforest region and published in 1995 (http://databasin.org/datasets/7f72a68ac6c343bda3ffff4bef3926de; accessed October 28, 2014).

We also intersected protected area feature classes with the MC1 stability areas to determine areas that are currently protected and projected to support climatically stable vegetation types overtime. In the USA, we used GAP status codes 1 ('strict') and 2 ('relaxed') obtained from the Protected Area Database (PAD-US CBI edition v1.1). In most cases, this database does not include administrative protections such as late-successional reserves of the Northwest Forest Plan (USFS and BLM, 1994) unless they overlapped with more stringent protections such as Wilderness and Congressionally Withdrawn Areas. The protected area data in British Columbia were obtained from Global Forest Watch Canada. Thus, we were able to show how areas of future stable vegetation, current late-seral forests and protected areas coincide in order to assess if the current conservation scheme across the entire region is well adapted to climate change or not.

Climate Envelope Model Evaluation and Most Important Climate Parameters

For the focal species, the AUC values based on the test data averaged across Maxent model runs ranged from 0.82 (western hemlock) to 0.93 (coast redwood), indicating that the models satisfactorily discriminated between presence and background information (online appendix).

The two most influential variables from the Worldclim dataset that most frequently show highest prediction power among the predictive Maxent models for focal species were 'Precipitation of Coldest Quarter' and 'Precipitation of Driest Quarter' (online appendix).

Key Findings for Focal Species and Rainforest Assemblages

Shifts of Potential Species Distributions

Aggregated potential distributions of focal conifer tree species predicted a shift for all applied GCMs by 2080s (**Figure 1**). More detailed species by species analysis are available in the online appendix. Although the intensity of shifts differed slightly among GCMs, the overall pattern showed a substantial reduction of aggregated potential species distributions for large parts of the seasonal and warm rainforest zones (south) and a broadly stable richness pattern of aggregated potential species distributions along the perhumid zone (north) – except for some northerly, island parts, and rain shadow areas (e.g., Olympic Peninsula). Quantitative comparisons of potential species distributions through time periods indicated that future distributions, in part, differ substantially compared to their baseline counterparts (**Table 1**). Averaged Cohen's kappa coefficients across all species and applied GCMs per time period revealed that differences are more pronounced by 2050s ($K=0.71$) compared to 2080s ($K=0.57$) in relation to baseline distributions.

By 2080s, potential distributions of western red cedar, Sitka spruce, and western hemlock show marked persistence (55–82%) mainly in northern portions of their range with minor contractions (2–7%) in the south (**Table 1**, **Figure 1**). Pacific silver fir, grand fir, Alaska yellow-cedar, and mountain hemlock had more substantial reductions (15–39%) in potential distributions throughout their range by 2080s. Coast redwood is expected to experience reduction of nearly one-fourth of its modeled climate envelope by 2080 (**Figure 2**, inset). Small (3%) climate related potential distribution gains were possible to the north; however, these are gone by 2080.

Table 1 Percent of baseline (1950–2000) potential distribution loss, persistence, and gain for focal species in the Pacific Coastal temperate rainforest by two time periods (2050s, 2080s), the A2A ensemble-emissions scenario, and full agreement among three General Circulation Models (CCCMA-CGCM2; CSIRO-MK2; and HADCM3)

Species	*Period*	*Loss (%)*	*Persistence (%)*	*Gain (%)*
Western red cedar	2050s	4	65	18
	2080s	6	59	28
Sitka spruce	2050s	0	83	9
	2080s	2	82	15
Western hemlock	2050s	4	74	8
	2080s	7	55	12
Pacific silver fir	2050s	24	35	3
	2080s	39	21	5
Grand fir	2050s	20	35	6
	2080s	36	17	10
Alaska yellow-cedar	2050s	8	66	4
	2080s	21	34	4
Moutain hemlock	2050s	7	59	7
	2080s	15	33	4
Coast redwood	2050s	21	16	3
	2080s	23	1	0

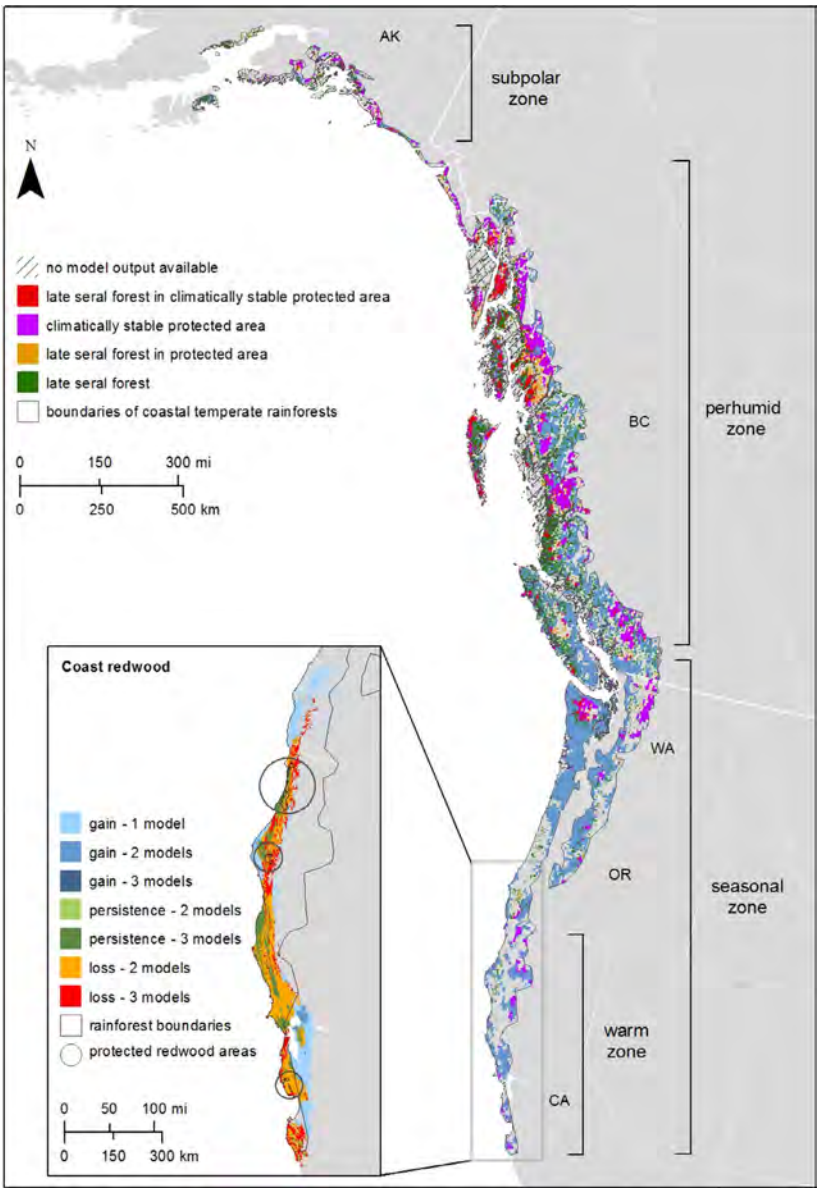

Figure 2 Predicted areas of vegetation stability (scenario A2, 2080s), protected areas, and late-seral forests in the Pacific coastal rainforests. Inset map shows potential distribution gain, persistence, and loss of coast redwood based on three GCMs (CSIRO, CCCMA, and HADCM3). The three circled areas in the redwood insert indicate protected areas where redwoods are currently found. Only the upper circled area has parks that coincide with projected redwood persistence in green.

Future State of the Ecosystem and Conservation Areas

Results from the MC1 dynamic vegetation model largely resembled the pattern obtained from climate envelope models on a broader scale (Figure 3 vs. Figure 1). Areas with potentially stable dominant vegetation communities were most densely spread across the perhumid zone and the coastal regions of the northern seasonal zone while southern areas changed more dramatically as also depicted in the species distribution models. In general, northern regions are expected to retain climate suitable for the baseline dominant vegetation types through 2080s, mostly the maritime evergreen needleleaf (e.g., western hemlock, Sitka spruce) type. Unstable areas also occur in the North, including portions of the Queen Charlotte and Haida Gwaii island and much of the mid and southern British Columbia coastline where temperate deciduous broadleaf woodland (e.g., red alder, *Alnus rubra*) is expected to expand, and the Kenai Peninsula of Alaska where the climate is expected to be more suited to temperate cool mixed forest rather than the baseline needleleaf forest. The climate currently supporting baseline subalpine forest in many areas is expected to shift toward conditions more suitable for patches of maritime evergreen needleleaf forest, temperate evergreen needleleaf forest, and temperate deciduous broadleaf forest.

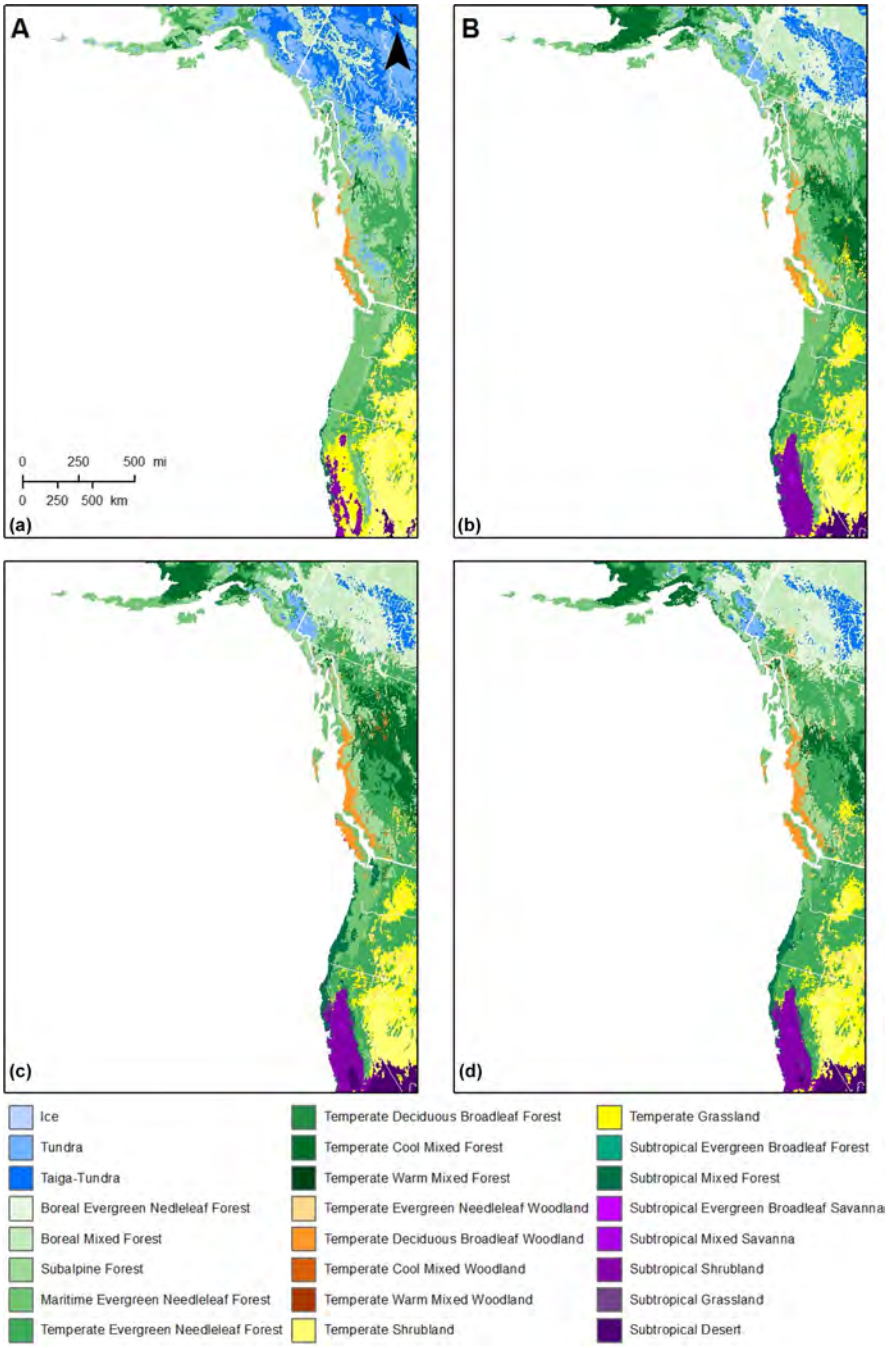

Figure 3 Outputs from MC1 functional vegetation model show baseline (a) and future dominant types of vegetation for 2080s (2075–85) based on three GCMs: CSIRO (b), MIROC (c), and HADCM3 (d).

In southern areas, shifts in dominant vegetation types were well dispersed throughout the warm zone and within the seasonal zone, especially the Cascades and southern coastal areas. For instance, starting just north of the Oregon/California border, the climate niche supporting maritime evergreen needleleaf (redwood, Douglas-fir zone) is expected to contract.

There was often a mismatch between current protected areas of coastal temperate rainforests with areas of future potential stability in dominant vegetation types, or with larger extents of late-seral forests, in particular, within the perhumid zone where older forests are especially concentrated and relatively intact (see Figure 2). This pattern was also shown when the proportion of vegetation stability for all protected areas that are completely located within the study area is plotted per state or province that intersects the coastal temperate rainforests (Figure 4). For instance, Washington and Oregon show the lowest vegetation stability, British Columbia the highest.

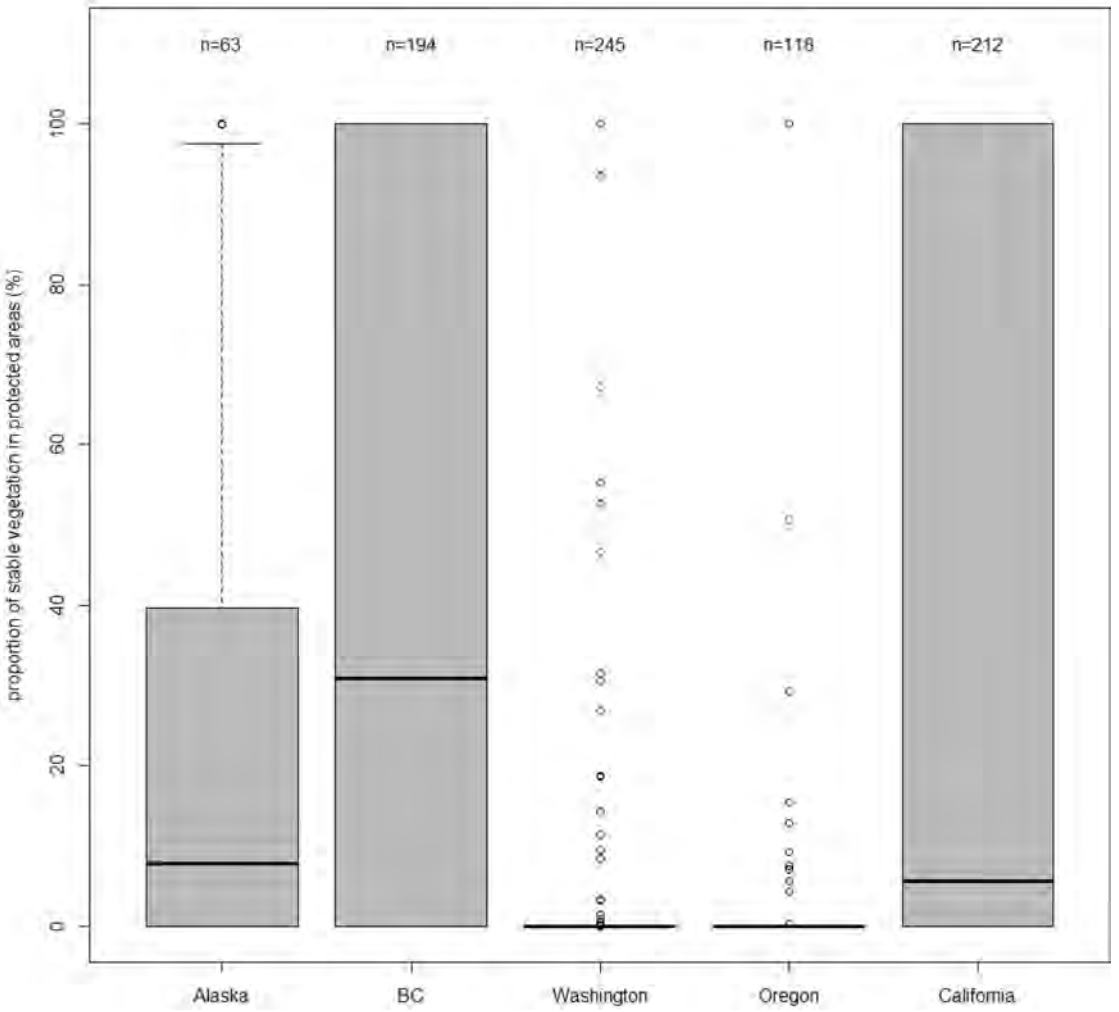

Figure 4 Predicted vegetation stability in protected areas per state or province derived from outputs of the MC1 model based on the agreement of three GCMs under the A2 scenario for 2080s (2075–85) (BC = British Columbia).

Relevance to Climate Adaptation Strategies and Land Management

Shifting Potential Distributions as a Surrogate for Ecosystem Change

Our focal species results correspond well with recent literature on range shifts of tree species caused by climate change (Chen et al., 2011; Hickling et al., 2006; Parmesan and Yohe, 2003; Shafer et al., 2001; Wang et al., 2012) and, while the magnitude of shifts differed, the trends were similar. For instance, using different GCMs than ours, Hamann and Wang (2006) found the distribution of western hemlock may increase by 50% over baseline area in British Columbia, shifting up in elevation and northward under the A2 emissions scenario by 2085. Coops and Waring (2011) also found a 50% gain for western hemlock and for other coastal

conifers that are likely to remain 'highly adapted' through the 2080s under the A2 emissions scenario. Others also have predicted northward shifts and shrinking baseline ranges of tree species in North America (McKenney et al., 2007; Murphy et al., 2010).

We found a core zone featuring the highest richness of potential focal species distributions in British Columbia between Vancouver Island and southeast Alaska, and areas of higher potential vegetation stability in these same areas. These regions could potentially act as refugia for temperate rainforest conifer species and assemblages and, because they have the lowest levels of forest fragmentation, may also be relatively insulated from edge-related local climate effects (Chen et al, 1995; Harper et al., 2005). Similarly, both approaches indicated greater loss and instability in the southern portion of the study area, particularly within the seasonal zone, supporting the generalized patterns of declining focal species richness southward.

What Is Driving the Projected Shifts?

A downside of our modeling approaches is that they do not provide us with definitive information on what is driving the projected shifts in communities or species. However, increases in frequencies and duration of extreme events have been documented in many regions and are expected to increase (Field et al., 2012). Extreme events are expected to be the primary drivers for many species and ecosystem impacts (Jentsch and Beierkuhnlein, 2008). Droughts have been correlated with elevated rates of forest dieback in North America due to water deficiency (Birdsey and Pan, 2011; Michaelian et al., 2011; van Mantgem et al., 2009), and might thus be crucial drivers of future distribution of temperate rainforest (DellaSala et al., 2011). For instance, water deficit may contribute to reductions of species distributions (both aggregated and species-specific) in the drier, southern parts of coastal temperate rainforests in our study area. However, declining low elevation snow and summer fog (southern rainforest distribution), not modeled in our study, might have a bigger effect on the distribution of yellow-cedar (Hennon et al., 2012) and coast redwood (Johnstone and Dawson, 2010), respectively, than the climate variables that we modeled. Further, projected increases in fires in southern rainforest areas may exacerbate climate-related changes to rainforest assemblages (Littell et al., 2009).

Model Limitations and Uncertainties

Climate envelope models are often criticized for relying on over-simplistic assumptions such as equilibria among species and their environment, omitting other predictors such as biotic interactions that might determine the fundamental niche (Araújo and Pearson, 2005), and lacking predictor quality (Soria-Auza et al., 2010). Biotic interactions and dispersal limitations are known to contribute to mismatches between model outputs and reality (Soberón and Peterson, 2005; Zimmermann et al., 2009). However, climate envelopes are known to perform best at a regional scale because they show general ecological trends and patterns (Boucher-Lalonde et al., 2012; Warren, 2012), as was the case in our study area. Moreover, the Worldclim predictor set is currently the most abundantly used set of climatic parameters, and to date the only one allowing for high resolution predictive modeling on a global scale. The applied model scale is appropriate, especially for species featuring smaller ranges or for modeling of complex terrain (Seo et al., 2009).

The MC1 dynamic vegetation model has been frequently used to investigate potential ecosystem vulnerability to climate change (Gonzalez et al., 2010). Comparing static climate envelope predictions with the dynamic MC1 vegetation model outputs revealed a more robust pattern (Kearney et al., 2010) of the bigger picture of shifting vegetation types across the Pacific coastal temperate rainforest region and also allowed us to apply our results on different data and spatial scales.

None of the models integrate human disturbances. There is no quantitative connection between Maxent and MC1 model outputs because focal tree species do not fully coincide with broad vegetation types. However, information derived from both model types complement each other on a coarse level and thus can more reliably inform management decisions by reducing uncertainty arising from any one model alone (also see Coops and Waring, 2011 for similar cross-model applications). Moreover, we propose that human impact is most likely to increase throughout the region, thus our models most likely under-estimate climate change effects exacerbated by human disturbance.

Rainforest Management Implications

At broad spatial scales, northern coastal regions and their protected areas (BC, Alaska) may be more resilient to climate change than southern areas that are highly fragmented and more vulnerable to edge effects (also see Thompson et al., 2009). That pattern holds true for coastal regions compared to interior drier regions (Wang et al., 2012) perhaps because of climatic buffering of maritime climates. Our results therefore are important for maintaining ecological integrity and climate resilience in high priority conservation areas from north to south such as the Tongass Rainforest of Alaska, Great Bear rainforest of BC, Olympic National Park of Washington, portions of the Western Cascades, and coast redwoods (DellaSala et al., 2011). Notably, ecological integrity and climate resilience are emphasized in the 2012 National Forest Planning Rule and climate resilience is emphasized in President Obama's Climate Action Plan (Executive Office of the President, 2013). Thus, the largely intact nature of the Tongass National Forest should provide important opportunities for meeting both policy objectives and for the northward expansion of rainforest communities in the face of climate change. Managers may also increase resilience potential by maintaining or restoring climatically stable vegetation along elevation and north-south gradients to accommodate shifting distributions. However, the slightly reduced richness of potential distributions and climatic instability in southern parts of the region show that some of the currently protected old forest stands are also vulnerable to climate change (online appendix) and may require additional actions. In particular, declines

in yellow-cedar may warrant consideration of assisted migration if this species is not able to colonize new climate spaces (Loss et al., 2011).

The Great Bear rainforest located in the perhumid zone is among the world's last remaining large extents of old-growth rainforest (DellaSala et al., 2011). Large portions of this rainforest show vegetation stability under a changing climate, including large extents of remaining old forest and high richness of focal tree species' potential distributions. Thus, we suggest that this region might also serve as broad-scale refugia if sufficiently protected from anthropogenic stressors that might exacerbate climate change impacts (Thompson et al., 2009; Watson et al., 2013).

Olympic National Park is situated in the seasonal rainforest zone and features exceptional plant richness, including many unique epiphytes (McCune and Geiser, 2009). Climate envelope richness of focal tree species is high within the core area of the park suggesting upslope shifts assuming melting glaciers. Importantly, the boundary regions of the park, including old-forest stands, show potential stability (online appendix) but are surrounded by highly fragmented private lands where conservation incentives are needed to retain stable dominant vegetation.

The Western Cascades are a secondary rainforest belt located in the northern portion of the seasonal zone that has been subjected to intensive logging. Lower resilience to climate change is indicated by unstable vegetation and decreasing climate envelope richness of focal tree species. Large proportions of remaining old forest remnants will likely be affected. While the larger protected areas, such as North Cascades National Park, Glacier National Park, and Alpine Lakes Wilderness show potential vegetation stability, some smaller areas (generally <1000 km^2) may experience climate-related stress to the dominant vegetation (online appendix).

Coast redwoods are situated in the warm zone within the most southern region of coastal temperate rainforests; the last, heavily diminished, redwoods are a conservation priority (Noss, 2000) and the apparent vulnerability of redwood to climate change in a significant portion of its range adds to conservation significance. Restorative actions within higher stability but previously logged areas may help to alleviate climate stressors for redwood. In addition, it is possible that redwood is resilient, at least initially, to shifts in its climate niche as increased growth rates measured in old-growth redwood forests are thought to be related to a lengthening of the growing season (Sillett et al., 2010). Our projections indicate that this apparent positive climate response of redwood might be short lived due to its projected shrinking climate niche.

Conclusions

Future temperate rainforest communities of the Pacific Coast of North America may persist mainly in northern latitudes and upper elevations where land-use disturbances are less likely to exacerbate changes to the focal species' climate envelope. They also may persist in pockets of relatively stable microrefugia (e.g., north-facing older forests) in the south if buffered from human disturbances (Olson et al., 2012). Projected changes in dominant vegetation types and focal species distributions, and identification of relatively stable intact patches, can aid managers in developing strategies for persistence of extant rainforest communities. Our work also provides valuable management insights into where important tree species may require assisted migration (e.g., yellow-cedar and redwood).

Finally, we note that in the time to peer review and publish this manuscript (>2 years) climate change models have been updated (IPCC, 2014). Thus, our projections need to be continuously updated (every five years or when new models come out) based on ongoing refinements to downscaled GCMs. Nevertheless, our broad-scale findings should prove useful in helping managers with comprehensive adaptation planning now for climate shifts in rainforest species and assemblages over a large region in order to avoid ecologically costly lags in conservation and management options given climate shifts are already underway.

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Climate Change May Trigger Broad Shifts in North America's Pacific Coastal Rainforests

DA DellaSala, Geos Institute, Ashland, OR, USA; Oregon State University, Corvallis, OR, USA
P Brandt, Karlsruhe Institute of Technology, Karlsruhe, Germany; International Livestock Research Institute (ILRI), Nairobi, Kenya
M Koopman, Geos Institute, Ashland, OR, USA
J Leonard, Geos Institute, Ashland, OR, USA
C Meisch, Leuphana University Lüneburg, Lüneburg, Germany
P Herzog, Martin Luther University Halle-Wittenberg, Halle, Germany
P Alaback, University of Montana, Missoula, MT, USA
MI Goldstein, University of Alaska Southeast, Juneau, AK, USA
S Jovan, Portland Forestry Sciences Lab, Portland, OR, USA
A MacKinnon, BC Forest Service, Victoria, BC, Canada
H von Wehrden, Research Institute of Wildlife Ecology, Vienna, Austria

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Introduction

Climate change threatens biodiversity and ecosystem integrity all over the globe (IPCC, 2014) and is already triggering pronounced shifts of species and ecosystems (Chen et al., 2011; Parmesan et al., 2000). Climate change is also expected to exacerbate effects of forest fragmentation (Bossuyt and Hermy, 2002; Opdam and Wascher, 2004), especially where only small fractions of formerly intact ecosystems remain (Heilman et al., 2002), presumably by magnifying local edge effects (Chen et al., 1995; Harper et al., 2005) and by reducing opportunities for dispersal and range expansion (Thompson et al., 2009; Watson et al., 2013). Thus, mitigating such effects in areas of global conservation importance is critical as biodiversity losses are especially significant.

The conservation importance of the coastal temperate rainforest region of North America is exemplified by the inclusion of six World Wildlife Fund Global 200 ecoregions (Ricketts et al., 1999), some of the most carbon dense ecosystems on earth (Leighty et al., 2006; Smithwick et al., 2002), extraordinarily productive salmon (*Oncorhynchus* spp.) runs and relatively intact forests northward (DellaSala et al., 2011). The highest epiphytic lichen biomass of any forest system also occurs here (McCune and Geiser, 2009). Thus, maintaining extant biodiversity in a changing climate has biodiversity significance on a global scale given the region's importance.

Already confirmed climate change effects in this region include elevated temperatures (Karl et al., 2009), declining mountain snowpack (Mote et al., 2005), shifts in species distributions (Wang et al., 2012), and reduced fog levels (Johnstone and Dawson, 2010). Diminished snowpack combined with late winter freezes has triggered dieback of Alaska yellow-cedar (*Cupressus nootkatensis*) in southeast Alaska (Hennon et al., 2012) and northern British Columbia (Wooten and Klinkenberg, 2011).

Vegetation along the northern Pacific coast has been sensitive to climatic changes since the last glaciation, resulting in large shifts in species distributions, and providing strong evidence that future climate change will result in substantial ecological changes (Brubaker, 1988; Heusser et al., 1985). Even small changes in temperature often result in large species displacements, which

explains contemporary pattern of species distributions along the coastal region (Alaback, 1996). A 125 000-year record of vegetation change from the eastern slope of the Cascades, for example, shows that while species movements are individualistic, depending on species characteristics and geography, at the millennial scale global climatic variation is the dominant factor controlling vegetation distribution (Whitlock and Bartlein, 1997). Conifer species' distributions have changed since the glacial maximum reflecting differences in dispersal ability, effects of refugia, and changes in glacial dynamics from central Alaska southward. The physiography of the region, with north-south tending cordillera, has facilitated species movements, helps explain the rarity of species extinctions in the past, and importance of dispersal in the future if species are to adapt to even more abrupt climatic changes. Additionally, dramatic changes in vegetation in the past 20 000 years (Whitlock, 1992) corresponded to warming of 2.5–7.8 °C (median values, including uncertainty) that is similar to what most general circulation models (GCMs) predict for the Western USA by the end of the twenty-first century (IPCC, 2014).

There is no broad adaptation plan that addresses potential range-wide shifts of ecologically and commercially valuable species in this region, although there is a growing body of relevant adaptation work as reflected by the North Pacific Landscape Conservation Cooperative (NPLCC) of the U.S. Fish & Wildlife Service (http://northpacificlcc.org, accessed October 14, 2014). Our primary objectives were therefore to: (1) model current potential distributions of focal conifers considered of commercial importance to land managers and to project future potential distributions of focal species and broad rainforest vegetation types in response to anticipated climate change; (2) identify areas that may exhibit higher vegetation stability, including those in currently protected areas where biodiversity conservation is emphasized; and (3) illustrate how uncertainty can be addressed in designing effective adaptation strategies in a changing climate.

Notably, attempts to predict future shifts in species' ranges have employed a variety of approaches. One widespread approach, climate envelope modeling, considers the climate conditions where a species is currently or historically distributed and estimates where those same suitable climate conditions are expected to be found in the future based on GCM outputs. This approach has both benefits and shortcomings, which have been thoroughly reviewed (Wiens et al., 2009). A criticism of climate envelope modeling is the strict focus on climate variables with little to no consideration of non-climate drivers such as competition, predation, soils, elevation, and dispersal. Thus, in our assessment of potential climate change effects, we employed both climate envelope models and a dynamic vegetation model, despite differences in input data and analysis scales, to qualitatively compare gross differences regarding the spatial patterns produced. Using correlative and mechanistic modeling approaches independently might increase the reliability of predictions (see Coops and Waring, 2011; Kearney et al., 2010), reducing uncertainties inherent in relying on any individual modeling effort.

Also, in this paper, our findings are used to illustrate some key concepts in climate adaptation planning for managers wishing to maintain extant biodiversity in a changing climate for a rainforest region that straddles two countries (USA and Canada) and large swaths of public and private lands. Additional analyses not presented, including detailed appendices and datasets, are available online (http://databasin.org/articles/172d089c062b4fb686cf18565df7dc57; accessed October 28, 2014).

North America Pacific Coast Temperate Rainforest Region

The Pacific Coast of North America contains the largest proportion of temperate rainforests in the world, representing 35% of the global total (DellaSala et al., 2011). Stretching from the coast redwoods (38° N), California to northern Kodiak Island and Prince William Sound (61° N), Alaska, these rainforests span a wide climatic gradient (Alaback, 1996). Coastal rainforests are associated with cool, moist oceanic air masses, a narrow range of temperature extremes, high frequency of clouds and fog, and high annual precipitation, with most precipitation in the winter (Redmond and Taylor, 1997) and up to 20% in the summer in northern latitudes (DellaSala et al., 2011). The region consists of four distinct rainforest zones that differ climatically and floristically: (1) subpolar – north of southeast Alaska to Prince William Sound and Kodiak Island; (2) perhumid – southeast Alaska to northern Vancouver Island; (3) seasonal – central Vancouver Island to southern Oregon; and (4) warm – southern Oregon coast to San Francisco Bay area (Alaback, 1996; **Figure 1**).

Climate Data

In order to predict potential shifts in species and rainforest distributions, we used the downscaled WorldClim dataset at 30 arc-s (1-km) resolution (Hijmans et al., 2005). We obtained 19 climatic variables for baseline conditions (1950–2000) and for two future time periods (2050s, 2080s) under the A2A ensemble-high-emissions scenario. This scenario assumes continued global population growth and focus on regional economic growth rather than global collaboration. It is one of the scenarios that most closely tracked the emissions trajectory at the time of our 2012 study. Thus, we used three GCMs: CCCMA-CGCM2 (third assessment, Flato and Boer, 2001), CSIRO-MK2 (third assessment, Gordon et al., 2002), and HADCM3 (third assessment – Johns et al., 2003) that covered a broad range of temperature and precipitation projections spanning dry and wet projections.

For climate envelope modeling, we employed a 1000-km buffer on the coastal rainforest study area to capture the entire current ranges of focal species and potential future shifts. Due to the small distribution of coast redwood, the buffer for the baseline model was set to 100 km around the most outer available localities.

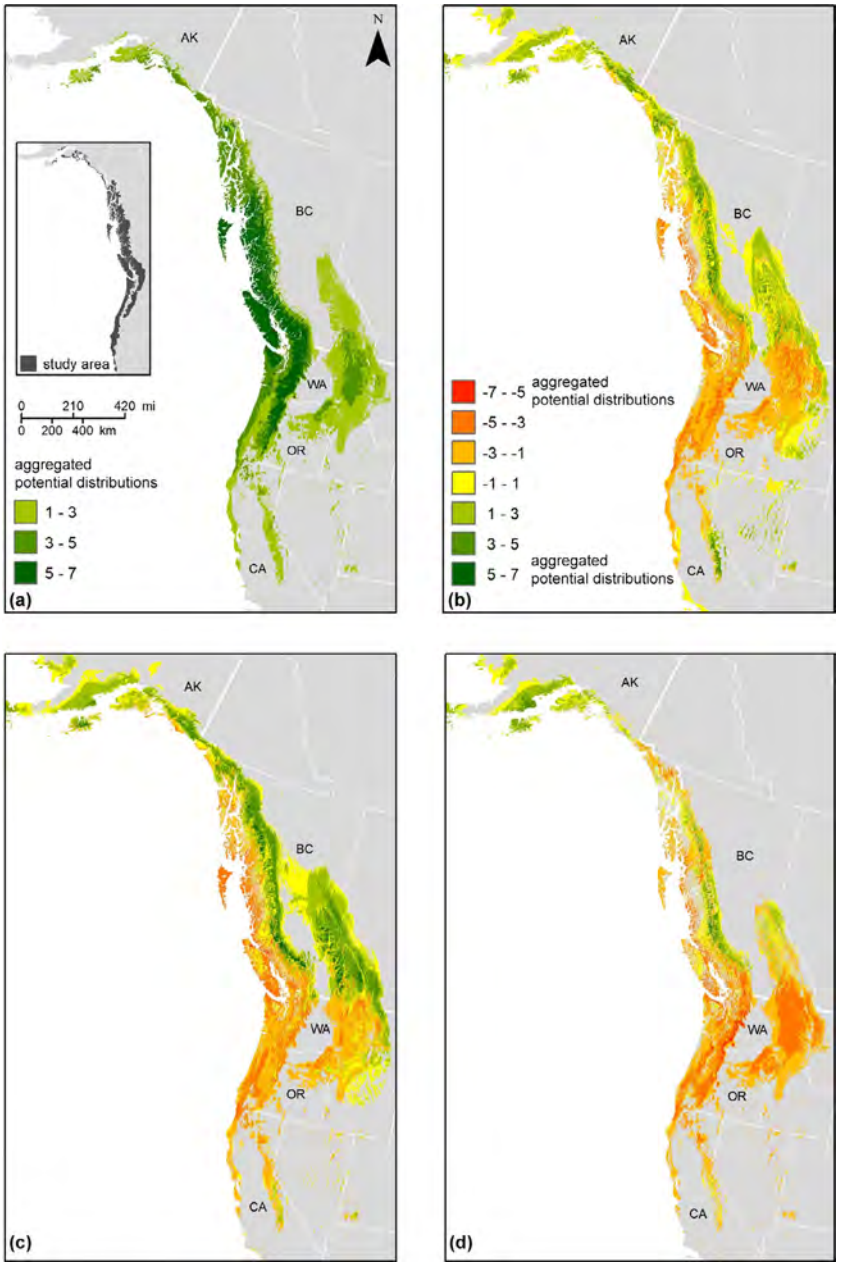

Figure 1 Aggregated potential distribution of eight focal conifer species (Pacific silver and grand fir, Alaska yellow-cedar, Sitka spruce, western red cedar, western and mountain hemlock, coast redwood) for the baseline period (a) and the richness changes for 2080s under scenario A2A ensemble-emissions based on three General Circulation Models (CSIRO (b), CCCMA (c), and HADCM3 (d)).

Selection of Focal Species of Commercial Importance

Based on prior discussions with land managers, we selected eight dominant conifer species of commercial, conservation, and cultural importance to model potential range shifts related to climate change. These species also were chosen because there was readily available location data and their geographic range overlapped primarily with our study area. They included Sitka spruce (*Picea sitchensis*), western and mountain hemlock (*Tsuga heterophylla*, *T. mertensiana*), western red cedar (*Thuja plicata*), Alaska yellow-cedar, Pacific silver and grand fir (*Abies amabilis*, *A. grandis*), and coast redwood (*Sequoia sempervirens*). We did not include other conifers with wide distributions that extended well outside our study area buffer such as Douglas-fir (*Pseudotsuga menziesii*, see Coops and Waring, 2011) or hardwoods (see Hamann and Wang, 2006) given their lower importance to land managers in this region.

Presence-only Modeling of Focal Species

To build the baseline species distribution models, we obtained presence-only data (point and polygon locations) for focal species from numerous databases (USDA Forest Inventory Assessment DataMart v5.1 – apps.fs.fed.us/fiadb-downloads/datamart.html; Biogeoclimatic Ecosystem Classification Program – www.for.gov.bc.ca/hre/becweb/resources/codes-standards/standards-becdb.html, active October 14, 2014; herbaria collections; museum records; published atlases) and from regional specialists that provided more than 30 000 species localities ranging from 710 occurrence points for coast redwood to 7999 points for western hemlock.

Presence-only models outline areas that are predicted as suitable space for a given species according to the predictor dataset (Soberón and Peterson, 2005); these models are known to overestimate realized distributions due to missing information of unvisited locations (Kent and Carmel, 2011). To examine the impact of climate change on species distributions, we only took climatic predictors into account, therefore, focusing on a species' climate envelope (Pearson and Dawson, 2003). Potential distribution was thus determined by projecting this climate envelope across the geographic study area (Soberón and Peterson, 2005).

We applied Maxent 3.3.3k (Elith et al., 2011; Phillips et al., 2006) to model current and future potential distribution for each focal species. Maxent frequently outperforms other presence-only modeling algorithms (Wisz et al., 2008). Instead of real absences, Maxent uses random background points to approximate the best fitting probability distribution for estimating habitat suitability (Elith et al., 2011). We used area under curve (AUC) statistics to assess model discrimination performance (Phillips et al., 2006). All models were replicated 25 times using the bootstrap replicate run type. The final average outputs were used for further analyses. The species datasets were split into 70% training and 30% test data sets randomly chosen for each model run.

We used jackknife procedures from initial model runs to exclude predictors that showed low importance in predicting included presence points when modeled in isolation, expressed by low values of model gain. We activated the 'fade by clamping' option in Maxent to mitigate clamping issues arising from projection values extending beyond the range of training data (Phillips et al., 2006) and chose the logistic output format. The automatic feature selection was applied since it has been validated with respect to a broad range of species, environmental conditions, numbers of occurrences, and degrees of sample selection bias (Phillips and Dudík, 2008). Using ARCGIS 10, the continuous grid outputs of the Maxent models were transformed into binary data showing either potential presence or modeled absence of a given species based on species-specific thresholds that minimized falsely excluded presences while retaining the similarity to published ranges (Little, 1978). Thus, for every species we created one baseline (1950–2000) potential distribution layer and six future potential distribution layers based on the two time periods (2050s, 2080s) and three GCMs.

Identifying Areas of Persistence, Gain, and Loss

For each focal species, we analyzed and mapped differences and commonalities between current and all variants of future potential distributions that were categorized as: (1) 'persistent distribution' where baseline and future potential distributions overlap, (2) 'distribution gain' where baseline potential distributions are absent but future potential distributions are present, and (3) - 'distribution loss' where baseline potential distributions are present but future potential distributions are absent. This is important for managers wishing to assess broad patterns in species distributions related to projected climate changes.

GCM outputs may differ widely, leading to variation in output among different climate envelope projections (Beaumont et al., 2008). Using three GCMs that spanned much of the range of possible futures, from wetter to drier and from faster warming to slower warming, allowed us to assess the level of disagreement among model output as an indirect measure of model uncertainty for managers wishing to plan for future distribution shifts. Importantly, we were able to assess climate envelope model outputs regarding model uncertainty inherent in climatic projections: uncertainty being lowest in areas where future potential distributions of all model projections showed a full consensus (spatial agreement) and highest in cases where they completely differed (Araújo and New, 2007). Obviously, model uncertainty is still inherent based on the complexity of climate and ecological systems, the potential for unexpected events related to climate change, and human behavior concerning greenhouse gas emissions abatement. Nonetheless, we propose that projections with relatively high agreement among models are useful in predicting broad trends important in robust reserve design and forest management decisions.

We calculated Cohen's kappa coefficients (K) (R Development Core Team, 2013 v. 2.1.12), indicating the degree of agreement (Fielding and Bell, 1997) between baseline and future potential distribution for all modeled species in order to quantify possible divergences in potential distributions over time as a proxy for expected shifts in species distribution (online appendix).

Outputs of climate envelope models can also be used to compile richness maps based on aggregated potential species distributions (McKenney et al., 2007). We used binary, aggregated potential distributions of focal tree species as an index of broad potential changes in species richness patterns across the entire study area.

Future Vegetation Stability, Intact Late-Seral Forests, and Current Protection Schemes

In addition to potential species shifts, we used the MC1 dynamic vegetation model outputs, biogeography module (Bachelet et al., 2001) to assess potential stability of dominant types of vegetation under a changing climate. The MC1 model was derived from physiologically based biogeographic rules derived from the MAPSS model (Neilson, 1995) adapted to dynamic environmental gradients using site production information (Bachelet et al., 2001). While the Maxent climate envelope analysis (above) focused on individual rainforest species and species richness, the MC1 output provided information on overall functional types of potential vegetation (temperate coastal needleleaf forest, for example) but not individual species. We compiled MC1 outputs produced under current and future climatic conditions using three GCMs (third assessment models): Hadley (HadCM3; Johns et al., 2003), MIROC (Hasumi and Emori, 2004), and CSIRO (Gordon et al., 2002) under the A2 emissions scenario. MC1 explicitly simulates vegetation dynamics, nutrient cycles and dynamic impacts of disturbance due to fire and has been used in analyses of vegetation responses to climate change (Lenihan et al., 2008). However, MC1 does not incorporate anthropogenic disturbances such as timber harvest, agriculture, urbanization, invasive species introductions, and human-wildfire ignition sources.

All applied MC1 model outputs have a 1/12°, unprojected, grid-cell resolution that is nominally 8-km (Daly et al., 2008). We assessed vegetation stability by comparing the dominant type of vegetation predicted to be supported under modeled baseline conditions (1961–1990) to that predicted to be supported for two future time periods (2035–45 and 2075–85). We identified areas as 'stable' or 'unstable' based on whether the future climate is expected to continue to support the same dominant vegetation type through late-century based on agreement across the three GCMs.

Notably, Pacific coastal temperate rainforests are highly fragmented in southern locales, which may be more vulnerable to large-scale changes in precipitation and temperature if magnified by local edge effects. Therefore, we accessed the most current intact late-seral rainforest datasets to identify areas that overlap with stable vegetation areas as potential refugia (Keppel et al., 2012; Olson et al., 2012; Watson et al., 2013). For intactness, we downloaded the only seamless forest fragmentation dataset available for the entire Pacific coastal temperate rainforest region and published in 1995 (http://databasin.org/datasets/7f72a68ac6c343bda3ffff4bef3926de; accessed October 28, 2014).

We also intersected protected area feature classes with the MC1 stability areas to determine areas that are currently protected and projected to support climatically stable vegetation types overtime. In the USA, we used GAP status codes 1 ('strict') and 2 ('relaxed') obtained from the Protected Area Database (PAD-US CBI edition v1.1). In most cases, this database does not include administrative protections such as late-successional reserves of the Northwest Forest Plan (USFS and BLM, 1994) unless they overlapped with more stringent protections such as Wilderness and Congressionally Withdrawn Areas. The protected area data in British Columbia were obtained from Global Forest Watch Canada. Thus, we were able to show how areas of future stable vegetation, current late-seral forests and protected areas coincide in order to assess if the current conservation scheme across the entire region is well adapted to climate change or not.

Climate Envelope Model Evaluation and Most Important Climate Parameters

For the focal species, the AUC values based on the test data averaged across Maxent model runs ranged from 0.82 (western hemlock) to 0.93 (coast redwood), indicating that the models satisfactorily discriminated between presence and background information (online appendix).

The two most influential variables from the Worldclim dataset that most frequently show highest prediction power among the predictive Maxent models for focal species were 'Precipitation of Coldest Quarter' and 'Precipitation of Driest Quarter' (online appendix).

Key Findings for Focal Species and Rainforest Assemblages

Shifts of Potential Species Distributions

Aggregated potential distributions of focal conifer tree species predicted a shift for all applied GCMs by 2080s (**Figure 1**). More detailed species by species analysis are available in the online appendix. Although the intensity of shifts differed slightly among GCMs, the overall pattern showed a substantial reduction of aggregated potential species distributions for large parts of the seasonal and warm rainforest zones (south) and a broadly stable richness pattern of aggregated potential species distributions along the perhumid zone (north) – except for some northerly, island parts, and rain shadow areas (e.g., Olympic Peninsula). Quantitative comparisons of potential species distributions through time periods indicated that future distributions, in part, differ substantially compared to their baseline counterparts (**Table 1**). Averaged Cohen's kappa coefficients across all species and applied GCMs per time period revealed that differences are more pronounced by 2050s ($K=0.71$) compared to 2080s ($K=0.57$) in relation to baseline distributions.

By 2080s, potential distributions of western red cedar, Sitka spruce, and western hemlock show marked persistence (55–82%) mainly in northern portions of their range with minor contractions (2–7%) in the south (**Table 1**, **Figure 1**). Pacific silver fir, grand fir, Alaska yellow-cedar, and mountain hemlock had more substantial reductions (15–39%) in potential distributions throughout their range by 2080s. Coast redwood is expected to experience reduction of nearly one-fourth of its modeled climate envelope by 2080 (**Figure 2**, inset). Small (3%) climate related potential distribution gains were possible to the north; however, these are gone by 2080.

Table 1 Percent of baseline (1950–2000) potential distribution loss, persistence, and gain for focal species in the Pacific Coastal temperate rainforest by two time periods (2050s, 2080s), the A2A ensemble-emissions scenario, and full agreement among three General Circulation Models (CCCMA-CGCM2; CSIRO-MK2; and HADCM3)

Species	*Period*	*Loss (%)*	*Persistence (%)*	*Gain (%)*
Western red cedar	2050s	4	65	18
	2080s	6	59	28
Sitka spruce	2050s	0	83	9
	2080s	2	82	15
Western hemlock	2050s	4	74	8
	2080s	7	55	12
Pacific silver fir	2050s	24	35	3
	2080s	39	21	5
Grand fir	2050s	20	35	6
	2080s	36	17	10
Alaska yellow-cedar	2050s	8	66	4
	2080s	21	34	4
Moutain hemlock	2050s	7	59	7
	2080s	15	33	4
Coast redwood	2050s	21	16	3
	2080s	23	1	0

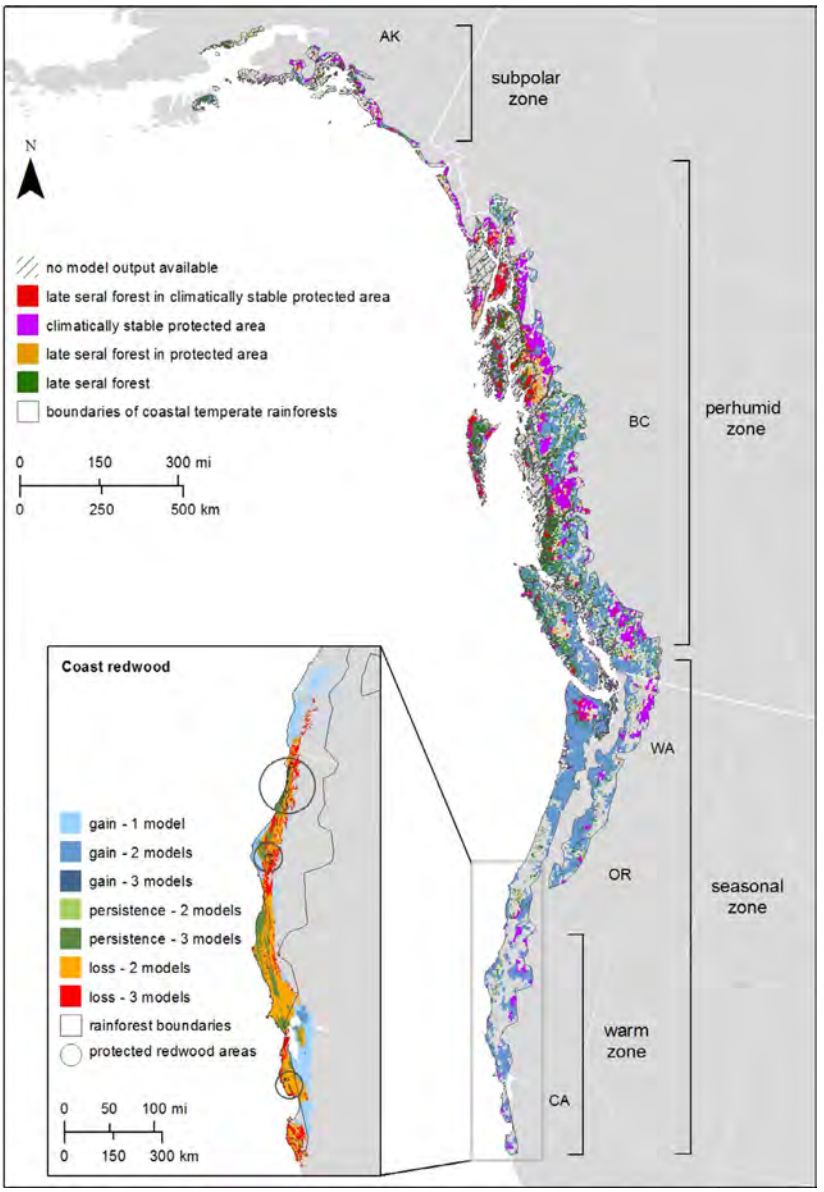

Figure 2 Predicted areas of vegetation stability (scenario A2, 2080s), protected areas, and late-seral forests in the Pacific coastal rainforests. Inset map shows potential distribution gain, persistence, and loss of coast redwood based on three GCMs (CSIRO, CCCMA, and HADCM3). The three circled areas in the redwood insert indicate protected areas where redwoods are currently found. Only the upper circled area has parks that coincide with projected redwood persistence in green.

Future State of the Ecosystem and Conservation Areas

Results from the MC1 dynamic vegetation model largely resembled the pattern obtained from climate envelope models on a broader scale (Figure 3 vs. Figure 1). Areas with potentially stable dominant vegetation communities were most densely spread across the perhumid zone and the coastal regions of the northern seasonal zone while southern areas changed more dramatically as also depicted in the species distribution models. In general, northern regions are expected to retain climate suitable for the baseline dominant vegetation types through 2080s, mostly the maritime evergreen needleleaf (e.g., western hemlock, Sitka spruce) type. Unstable areas also occur in the North, including portions of the Queen Charlotte and Haida Gwaii island and much of the mid and southern British Columbia coastline where temperate deciduous broadleaf woodland (e.g., red alder, *Alnus rubra*) is expected to expand, and the Kenai Peninsula of Alaska where the climate is expected to be more suited to temperate cool mixed forest rather than the baseline needleleaf forest. The climate currently supporting baseline subalpine forest in many areas is expected to shift toward conditions more suitable for patches of maritime evergreen needleleaf forest, temperate evergreen needleleaf forest, and temperate deciduous broadleaf forest.

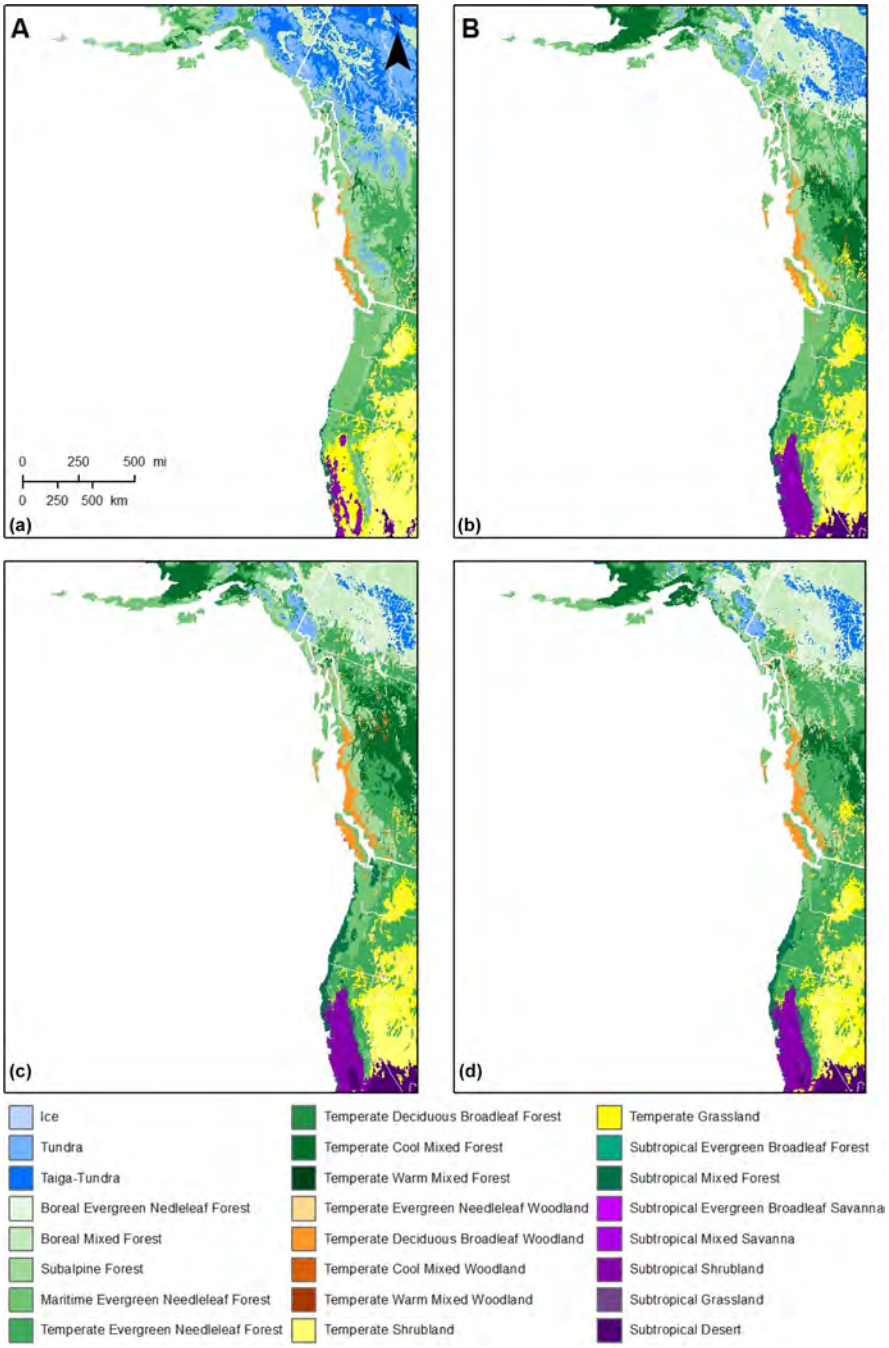

Figure 3 Outputs from MC1 functional vegetation model show baseline (a) and future dominant types of vegetation for 2080s (2075–85) based on three GCMs: CSIRO (b), MIROC (c), and HADCM3 (d).

In southern areas, shifts in dominant vegetation types were well dispersed throughout the warm zone and within the seasonal zone, especially the Cascades and southern coastal areas. For instance, starting just north of the Oregon/California border, the climate niche supporting maritime evergreen needleleaf (redwood, Douglas-fir zone) is expected to contract.

There was often a mismatch between current protected areas of coastal temperate rainforests with areas of future potential stability in dominant vegetation types, or with larger extents of late-seral forests, in particular, within the perhumid zone where older forests are especially concentrated and relatively intact (see Figure 2). This pattern was also shown when the proportion of vegetation stability for all protected areas that are completely located within the study area is plotted per state or province that intersects the coastal temperate rainforests (Figure 4). For instance, Washington and Oregon show the lowest vegetation stability, British Columbia the highest.

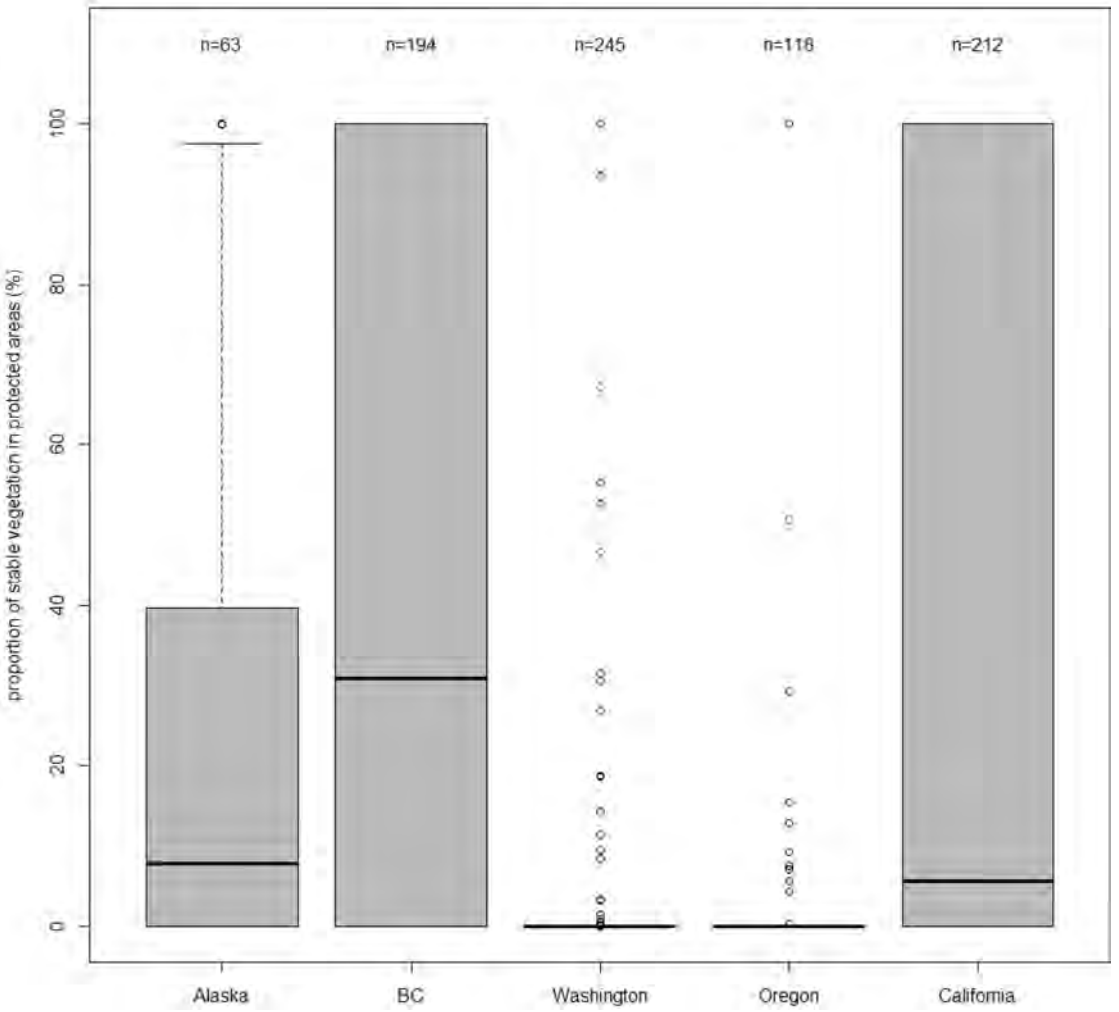

Figure 4 Predicted vegetation stability in protected areas per state or province derived from outputs of the MC1 model based on the agreement of three GCMs under the A2 scenario for 2080s (2075–85) (BC = British Columbia).

Relevance to Climate Adaptation Strategies and Land Management

Shifting Potential Distributions as a Surrogate for Ecosystem Change

Our focal species results correspond well with recent literature on range shifts of tree species caused by climate change (Chen et al., 2011; Hickling et al., 2006; Parmesan and Yohe, 2003; Shafer et al., 2001; Wang et al., 2012) and, while the magnitude of shifts differed, the trends were similar. For instance, using different GCMs than ours, Hamann and Wang (2006) found the distribution of western hemlock may increase by 50% over baseline area in British Columbia, shifting up in elevation and northward under the A2 emissions scenario by 2085. Coops and Waring (2011) also found a 50% gain for western hemlock and for other coastal

conifers that are likely to remain 'highly adapted' through the 2080s under the A2 emissions scenario. Others also have predicted northward shifts and shrinking baseline ranges of tree species in North America (McKenney et al., 2007; Murphy et al., 2010).

We found a core zone featuring the highest richness of potential focal species distributions in British Columbia between Vancouver Island and southeast Alaska, and areas of higher potential vegetation stability in these same areas. These regions could potentially act as refugia for temperate rainforest conifer species and assemblages and, because they have the lowest levels of forest fragmentation, may also be relatively insulated from edge-related local climate effects (Chen et al, 1995; Harper et al., 2005). Similarly, both approaches indicated greater loss and instability in the southern portion of the study area, particularly within the seasonal zone, supporting the generalized patterns of declining focal species richness southward.

What Is Driving the Projected Shifts?

A downside of our modeling approaches is that they do not provide us with definitive information on what is driving the projected shifts in communities or species. However, increases in frequencies and duration of extreme events have been documented in many regions and are expected to increase (Field et al., 2012). Extreme events are expected to be the primary drivers for many species and ecosystem impacts (Jentsch and Beierkuhnlein, 2008). Droughts have been correlated with elevated rates of forest dieback in North America due to water deficiency (Birdsey and Pan, 2011; Michaelian et al., 2011; van Mantgem et al., 2009), and might thus be crucial drivers of future distribution of temperate rainforest (DellaSala et al., 2011). For instance, water deficit may contribute to reductions of species distributions (both aggregated and species-specific) in the drier, southern parts of coastal temperate rainforests in our study area. However, declining low elevation snow and summer fog (southern rainforest distribution), not modeled in our study, might have a bigger effect on the distribution of yellow-cedar (Hennon et al., 2012) and coast redwood (Johnstone and Dawson, 2010), respectively, than the climate variables that we modeled. Further, projected increases in fires in southern rainforest areas may exacerbate climate-related changes to rainforest assemblages (Littell et al., 2009).

Model Limitations and Uncertainties

Climate envelope models are often criticized for relying on over-simplistic assumptions such as equilibria among species and their environment, omitting other predictors such as biotic interactions that might determine the fundamental niche (Araújo and Pearson, 2005), and lacking predictor quality (Soria-Auza et al., 2010). Biotic interactions and dispersal limitations are known to contribute to mismatches between model outputs and reality (Soberón and Peterson, 2005; Zimmermann et al., 2009). However, climate envelopes are known to perform best at a regional scale because they show general ecological trends and patterns (Boucher-Lalonde et al., 2012; Warren, 2012), as was the case in our study area. Moreover, the Worldclim predictor set is currently the most abundantly used set of climatic parameters, and to date the only one allowing for high resolution predictive modeling on a global scale. The applied model scale is appropriate, especially for species featuring smaller ranges or for modeling of complex terrain (Seo et al., 2009).

The MC1 dynamic vegetation model has been frequently used to investigate potential ecosystem vulnerability to climate change (Gonzalez et al., 2010). Comparing static climate envelope predictions with the dynamic MC1 vegetation model outputs revealed a more robust pattern (Kearney et al., 2010) of the bigger picture of shifting vegetation types across the Pacific coastal temperate rainforest region and also allowed us to apply our results on different data and spatial scales.

None of the models integrate human disturbances. There is no quantitative connection between Maxent and MC1 model outputs because focal tree species do not fully coincide with broad vegetation types. However, information derived from both model types complement each other on a coarse level and thus can more reliably inform management decisions by reducing uncertainty arising from any one model alone (also see Coops and Waring, 2011 for similar cross-model applications). Moreover, we propose that human impact is most likely to increase throughout the region, thus our models most likely under-estimate climate change effects exacerbated by human disturbance.

Rainforest Management Implications

At broad spatial scales, northern coastal regions and their protected areas (BC, Alaska) may be more resilient to climate change than southern areas that are highly fragmented and more vulnerable to edge effects (also see Thompson et al., 2009). That pattern holds true for coastal regions compared to interior drier regions (Wang et al., 2012) perhaps because of climatic buffering of maritime climates. Our results therefore are important for maintaining ecological integrity and climate resilience in high priority conservation areas from north to south such as the Tongass Rainforest of Alaska, Great Bear rainforest of BC, Olympic National Park of Washington, portions of the Western Cascades, and coast redwoods (DellaSala et al., 2011). Notably, ecological integrity and climate resilience are emphasized in the 2012 National Forest Planning Rule and climate resilience is emphasized in President Obama's Climate Action Plan (Executive Office of the President, 2013). Thus, the largely intact nature of the Tongass National Forest should provide important opportunities for meeting both policy objectives and for the northward expansion of rainforest communities in the face of climate change. Managers may also increase resilience potential by maintaining or restoring climatically stable vegetation along elevation and north-south gradients to accommodate shifting distributions. However, the slightly reduced richness of potential distributions and climatic instability in southern parts of the region show that some of the currently protected old forest stands are also vulnerable to climate change (online appendix) and may require additional actions. In particular, declines

in yellow-cedar may warrant consideration of assisted migration if this species is not able to colonize new climate spaces (Loss et al., 2011).

The Great Bear rainforest located in the perhumid zone is among the world's last remaining large extents of old-growth rainforest (DellaSala et al., 2011). Large portions of this rainforest show vegetation stability under a changing climate, including large extents of remaining old forest and high richness of focal tree species' potential distributions. Thus, we suggest that this region might also serve as broad-scale refugia if sufficiently protected from anthropogenic stressors that might exacerbate climate change impacts (Thompson et al., 2009; Watson et al., 2013).

Olympic National Park is situated in the seasonal rainforest zone and features exceptional plant richness, including many unique epiphytes (McCune and Geiser, 2009). Climate envelope richness of focal tree species is high within the core area of the park suggesting upslope shifts assuming melting glaciers. Importantly, the boundary regions of the park, including old-forest stands, show potential stability (online appendix) but are surrounded by highly fragmented private lands where conservation incentives are needed to retain stable dominant vegetation.

The Western Cascades are a secondary rainforest belt located in the northern portion of the seasonal zone that has been subjected to intensive logging. Lower resilience to climate change is indicated by unstable vegetation and decreasing climate envelope richness of focal tree species. Large proportions of remaining old forest remnants will likely be affected. While the larger protected areas, such as North Cascades National Park, Glacier National Park, and Alpine Lakes Wilderness show potential vegetation stability, some smaller areas (generally <1000 km^2) may experience climate-related stress to the dominant vegetation (online appendix).

Coast redwoods are situated in the warm zone within the most southern region of coastal temperate rainforests; the last, heavily diminished, redwoods are a conservation priority (Noss, 2000) and the apparent vulnerability of redwood to climate change in a significant portion of its range adds to conservation significance. Restorative actions within higher stability but previously logged areas may help to alleviate climate stressors for redwood. In addition, it is possible that redwood is resilient, at least initially, to shifts in its climate niche as increased growth rates measured in old-growth redwood forests are thought to be related to a lengthening of the growing season (Sillett et al., 2010). Our projections indicate that this apparent positive climate response of redwood might be short lived due to its projected shrinking climate niche.

Conclusions

Future temperate rainforest communities of the Pacific Coast of North America may persist mainly in northern latitudes and upper elevations where land-use disturbances are less likely to exacerbate changes to the focal species' climate envelope. They also may persist in pockets of relatively stable microrefugia (e.g., north-facing older forests) in the south if buffered from human disturbances (Olson et al., 2012). Projected changes in dominant vegetation types and focal species distributions, and identification of relatively stable intact patches, can aid managers in developing strategies for persistence of extant rainforest communities. Our work also provides valuable management insights into where important tree species may require assisted migration (e.g., yellow-cedar and redwood).

Finally, we note that in the time to peer review and publish this manuscript (>2 years) climate change models have been updated (IPCC, 2014). Thus, our projections need to be continuously updated (every five years or when new models come out) based on ongoing refinements to downscaled GCMs. Nevertheless, our broad-scale findings should prove useful in helping managers with comprehensive adaptation planning now for climate shifts in rainforest species and assemblages over a large region in order to avoid ecologically costly lags in conservation and management options given climate shifts are already underway.

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FEATURE

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Roadless areas and clean water

Dominick A. DellaSala, James R. Karr, and David M. Olson

Clean water, like biodiversity, is most closely linked to undisturbed natural ecosystems. When undisturbed watersheds in roadless and protected areas (e.g., national parks, state parks, wilderness areas, national monuments) are fragmented by roads, logging, and intensive recreation development, both water quality and biodiversity decline as hydrological integrity is lost (USFS 1972, 1979, 2001; Alexander and Gorte 2008; Anderson 2008). In the United States, inventoried roadless areas (IRAs) are lands without roads exceeding 2,000 ha (5,000 ac) that have been inventoried by the USDA Forest Service. IRAs collectively amount to approximately one third of the 77 million ha (193 million ac) of the 155 national forests but are disproportionately concentrated in western states (figure 1) (Trout Unlimited 2004; Anderson 2008). The roaded, intensively managed landscapes of the other national forest lands have been closely correlated with heavily sediment-laden streams and dramatic changes in flow regimes (Espinosa et al. 1997; Trombulak and Frissell 2000; CBD et al. 2001; Coffin 2007; Frissell and Carnefix 2007). While the biodiversity benefits of IRAs are well documented (DeVelice and Martin 2001; Strittholt and DellaSala 2001; Loucks et al. 2003; Strittholt et al. 2004; Gelbardi and Harrison 2005), little has been made of the importance of IRA water for downstream users and wildlife.

In this paper, we assess the importance of IRAs from a water quality perspective, including the likely water quality effects of developing IRAs. We provide conservative estimates of the economic impact of intact unroaded watersheds on national forests for clean water and associated water resource benefits. In particular, rising demand and shrinking water supply associated with changing climate will likely make intact areas in drought-prone regions of the West even more valuable and crucial to protect. Thus, our findings are especially relevant to drought-prone states considering development of IRAs. The state of Colorado, for example, with approximately 1.7 million ha (4.2 million ac) of IRAs, has been seeking federal permission to develop its IRAs for logging, expanding ski areas, coal-bed mining, and producing oil and gas (figure 2) (Anderson 2008; Colorado Division of Wildlife 2010; Colorado, State of 2010; Straub 2010, USFS 2011). Although we focus on IRAs throughout the western United States, we also emphasize the importance of uninventoried roadless areas (unroaded) <2,000 ha (Henjum et al. 1994; Greenwald 1998; Beschta et al. 2004) that collectively cover an area roughly 1.5 times that of the total IRA network (USFS 2000; Strittholt et al. 2004). Those smaller unroaded areas also play a strategic role in maintaining reliable supplies of high-quality water and protecting aquatic ecosystems.

Dominick A. DellaSala is Chief Scientist and President of the Geos Institute, Ashland, Oregon. **James R. Karr** is professor emeritus of ecology and environmental policy at the University of Washington, Seattle, Washington. **David M. Olson** is a conservation biologist at the Conservation Earth Consulting, Burbank, California.

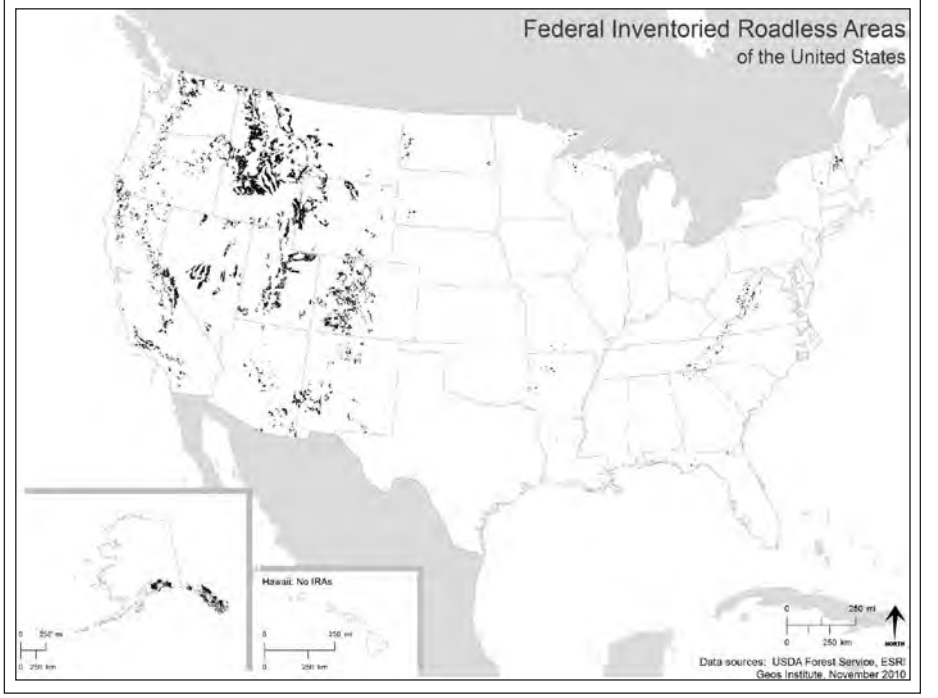
ROADLESS AREAS PROVIDE SUBSTANTIAL WATER RESOURCE BENEFITS

IRAs benefit society in many ways, including providing a valuable and increasingly rare natural supply of abundant, clean, and naturally reliable water (Sedell et al. 2000); affordable drinking water for municipal and rural communities; water for agricultural and industrial uses; flood control; in-stream aquatic recreation; aquifer recharge; flood protection; reliable water supply; diverse and productive fisheries; healthy aquatic ecosystems; resident and migratory waterfowl habitat; recovery of endangered species; and, increasingly, the vitality and sustainability of local economies (table 1). These benefits accrue nationally and at the local and regional levels.

National Benefits of Clean Roadless-Area Water. At least 124 million Americans directly benefit from water

Figure 1

Federal inventoried roadless areas (IRAs) of the United States (Source: USDA Forest Service).



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originating from national forests (Sedell et al. 2000). In fact, national forests provide about 15% of the nation's runoff with an estimated net value of $3.7 (Sedell et al. 2000) to $27 billion (Krieger 2001). The water treatment value alone of National Forests ranges from $490 million (Loomis 2005) to $18 billion (Krieger 2001).

Because IRAs represent roughly a third of national forestland, by inference they contribute significantly to the overall runoff volume and value (Anderson 1997, 2008) estimated in billions of dollars annually (Loomis and Richardson 2001; Sechhi et al. 2005). For instance, using Forest Service data (USFS 2000), IRAs make up 661 of the 914 national forest watersheds, with 55% of the 914 watersheds acting as source areas for facilities that treat and distribute drinking water to the public. The cost-savings to water treatment plants and highway departments from avoiding sedimentation caused by logging in IRA watersheds is estimated at up to $18 billion annually (Loomis 1988). IRAs provide $490 million annually in waste treat-

Figure 2

Colorado's 2001 inventoried roadless areas (IRAs) are shown in light gray, the 2011 proposed Colorado roadless areas (CRAs) are shown in gray, and overlap between CRAs and IRAs is shown in black. Water quality will be most impacted by changes of allowable activities within existing IRAs relative to changes in designated areas (USFS 2011).

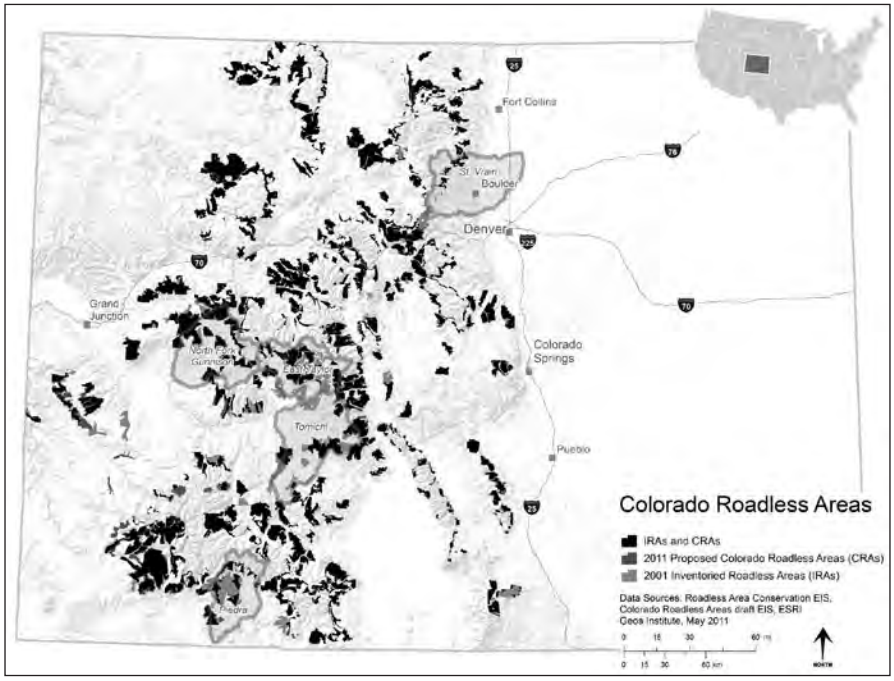

Table 1

General ecosystem services and benefits related to water that are provided by undisturbed IRAs and watersheds (derived from Greenway 1996; Costanza et al. 1997; Talberth and Moskowitz 1999; GAO 2000;Heal 2000, Loomis and Richardson 2001; Sedell et al. 2000; Krieger 2001; Dombeck 2003; Berrens et al. 2006).

	Benefits
Off-stream benefits	Low treatment costs for water for all beneficiaries
	Low price per unit volume costs for water for all beneficiaries
	High-quality and abundant drinking water for rural communities and municipal water supplies
	High-quality water for agricultural and industrial purposes
	High-quality water for downstream livestock production
	High-quality water for reduced health care and epidemic control
	Reduced costs of flood damage and flood control; enhanced local economies and property values
	Community benefits, including jobs, income, favorable trends for key economic indicators, and economic sustainability and stability
	Recharging of groundwater aquifers
	Healthy terrestrial and riparian ecosystems and their component species, sustained ecological and evolutionary processes, and resilient ecosystems
In-stream benefits	Healthy aquatic ecosystems
	Recovery of endangered species and protection of refugia
	Diverse and productive fisheries
	High-quality habitat for wildlife, including migratory waterfowl and game and nongame species
	Aquatic recreation such as swimming, rafting, and boating; enhancement of hiking and camping
	The inherent value of wild rivers and wilderness (including passive use benefits such as option, bequest, and existence values)
	Moderation of runoff and streamflows (e.g., lower peak flows, higher low flows, year-round water)
	Soil stabilization and erosion control
	Scientific value (intact watersheds are very rare today)
	Maintaining sediment production to streams at normal background rates
	Reducing potential for damage to downstream properties and water users during periods of high flow
	Breakdown and containment of waste and toxins (e.g., atmospheric, prior use)

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ment services through recovering mobile nutrients and cleansing the environment, both processes that involve water flow through intact watersheds (Loomis and Richardson 2001).

Regional Benefits of Clean Roadless-Area Water. In the US Rocky Mountains, roughly one third of utilized streamflow is derived directly from IRAs (which cover a quarter of Colorado's headwaters), with cities like Denver receiving about 30% of their water supply from IRA watersheds. Annually, IRAs in Colorado are estimated to provide an equivalent of nearly 2.5 times Denver's annual water use (Doyle and Gardner 2010; Denver Water 2010). Similarly, IRAs in New Mexico provide an estimated water quality benefit up to $42 million annually (Berrens et al. 2006).

Flood Control Protection and Inventoried Roadless Areas. The intact watersheds of IRAs are especially important for ameliorating the frequency and intensity of flooding, saving millions of dollars annually from averted floods and associated sedimentation, a service that will only increase in value as climate change drives more floods (Seeds 2010). Dredging reservoirs to increase capacity and channels to enable navigation costs cities, states, and ultimately taxpayers millions annually. Salem, Oregon, spent approximately $100 million on new treatment facilities after logging in upper watersheds created conditions leading to mass sedimentation in its watershed following storms in 1996 (Schwickert and Mauldin 1997; Talberth and Moskowitz 1999). In addition, Seattle, Washington, deferred a $150 million filtration plant expenditure through an intensive watershed rehabilitation program that will decommission 480 km (300 mi) of roads over a 10-year period, fix road erosion problems, and limit access and high-risk activities for fire and sedimentation within their watersheds (Seeds 2010).

Recreation Benefits and Strong Local Economies. IRA water benefits outdoor recreation and the people that either engage in or earn their living from outdoor recreation. The nation's IRAs generate $600 million annually from recreation (Loomis and Richardson 2001). Passive-use values (i.e., the intrinsic value of wilderness, wildlands, and benefits for the future) are estimated at an additional $280 million annually. At the regional scale, New Mexico IRA water provides an estimated $27 million active outdoor recreation benefit and a $14 million passive-use benefit annually (Berrens et al. 2006). For many visitors, much of the attraction to wildlands is associated with the presence of clean and abundant water—a dwindling resource as logging, grazing, and road-building continues across mountain landscapes and droughts from a changing climate intensify in much of the West (Saunders et al. 2008).

Freshwater Biodiversity and Healthy Fisheries. Clean water from IRAs also maintains healthy fisheries, such as salmon and trout fisheries, sustains viable aquatic ecosystems, and helps protect threatened species and ecosystems (Abell et al. 2000; Trout Unlimited 2004). Indeed, IRAs may act as important refugia for many salmon and trout populations, as well as for a diversity of endangered freshwater species (Henjum et al. 1994; Huntington 1998; NRC 1996; Trombulak and Frissell 2000; CBD et al. 2001; Strittholt and DellaSala 2001; Oechsli and Frissell 2002; Strittholt et al. 2004; Petersen 2005). Restoration of salmon and trout fisheries in places with high road densities will likely fail without the pivotal role provided by IRAs as fishery strongholds.

ROADLESS AREAS ARE IMPORTANT SOURCES FOR DRINKING WATER

The distribution of IRAs across prime hydrologic real estate—headwaters and upper watersheds—makes them particularly valuable for providing reliable supplies of clean water. In Colorado, IRAs occur in the headwaters of all major drainages, covering roughly a third of upper watersheds in the state. Indeed, most IRAs are located in mountainous terrain in western states, including Oregon, Idaho, New Mexico, Utah, Montana, California, and Washington. This extensive coverage of IRAs in headwaters, and because they are often the last minimally disturbed watersheds within larger landscapes of degraded lands, makes them hydrologic hotspots—areas with relatively small spatial extent that have a disproportionately important role in producing abundant and reliable clean water (Frissell and Carnefix 2007).

For many major drainages (entire watersheds of major rivers, such as the Columbia River Basin), IRAs and other wilderness areas represent the last few percentages (typically 1% to 5%) of the landscape with a minimally disturbed, or near natural, hydrology. As in many other ecological contexts, losing the last relatively natural systems typically results in major losses in water resource benefits, losses that can only be compensated by very expensive actions. The known relationship between watershed degradation and water quality decline deserves to be more rigorously incorporated as a central foundation for decisions on watershed management and protection.

Developing Roadless Areas Degrades Water Quality. In addition to their keystone location within watersheds, roadless areas typically encompass the most fragile of natural landscapes—montane forests and meadows. Road building and other intensive management in these otherwise intact areas damage their ability to provide clean water for downstream communities and biodiversity over both short and long terms (Beschta 1978; Forman and Alexander 1998; Lugo and Gucinski 2000; Trombulak and Frissell 2000; Gucinski et al. 2001; Coffin 2007). Logging, including post-disturbance, fire-risk reduction, forest health, and insect control; livestock grazing; mining; and road building are responsible for chronic and acute sedimentation of aquatic ecosystems, alter overland flow and stream structure, and change a range of physical and biological features by causing more frequent and intense floods, decreasing available water throughout the year, increasing stream and ambient temperatures, and elevating turbidity and nutrient levels (Beschta 1978; Fleischner 1994; Trombulak and Frissell 2000; DellaSala et al. 2006; Coffin 2007). Logging roads have been linked to great increases in erosion rates and sediment delivery to streams—up to 850% over rates in undisturbed habitat—with long-term and often catastrophic impacts on stream biota, aquatic ecosystems, and water quality (Fredricksen 1970; Megahan and Kidd 1972; Amaranthus et al. 1985; Bilby

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et al. 1989; King 1989, 1993; Haynes and Horne 1997; Jones et al. 2000; Wemple and Jones 2003).

Depending on severity and duration of impacts, disturbance can elevate average turbidity levels well above background levels (Seeds [2010] provides examples from Oregon), along with triggering more frequent and intense turbidity spikes that are a major source of excess costs to municipal water supply departments. Relative to roadless watersheds with intact natural vegetation, intensively managed watersheds also produce less available water (i.e., average monthly usable raw water) due to intensified high flows with very high turbidity and exacerbated low flow conditions (Seeds 2010). The monthly reliability of water is also diminished.

Even small disturbances in upper watersheds can result in significant, cumulative, and long-term impacts to downstream water and aquatic ecosystems (Platts and Nelson 1985; Boise National Forest 1993; McIntosh et al. 1994, 1995). In unstable terrain, for instance, small areas (e.g., less than 10% of a watershed's area) of low-intensity disturbance, including roads, may greatly increase the frequency and size of mass erosion events, with subsequent acute and chronic reduction in downstream water quality. Management activities that damage natural vegetation typically result in loads of suspended solids that exceed background levels and more frequent and intense spikes in suspended solids stemming from an increase in mass erosion events like landslides, debris flows, and bank failures. These impacts are strongly correlated with roads, as well as with logging and grazing (Amaranthus et al. 1985; Fleischner 1994; Trombulak and Frissell 2000; Coffin 2007).

Rising Demand and Climate Change Diminish Water Supply. Population in the West is projected to increase by 300% within just 30 years, with similar increases in demand for water (Sedell et al. 2000). Urban and exurban areas are growing exponentially, including communities adjacent to wilderness areas and IRAs (Theobald 2005). The demand for water in Colorado is expected to triple by 2050. Similarly, the number of people relying on national forest water has doubled in Oregon in the last 30 years, and 86% of the population of Washington rely on national forest water to some degree (Sedell et al. 2000).

The dramatic population growth in the West is concurrent with a warming and drying climate in many places. Temperatures are increasing, snow pack is declining and melting sooner, and drought and summer water deficits are more frequent and longer (Barnett et al. 2008; Mohammed and Tarboton 2008; Saunders et al. 2008; Miller et al. 2010). Streamflow reductions ranging from 10% to 35% are likely for the western states over the next half century as a consequence of climate change (Barnett and Pierce 2009). A 10% drop in streamflow is considered calamitous by municipal water districts. More frequent and intense flood events are also likely in places (Raff et al. 2009), despite drying conditions. Costs for flood control, repair and reconstruction, and insurance rates will also increase (GAO 2007). These events will worsen the severe and unprecedented droughts already afflicting much of the West (Drechsler et al. 2006; Saunders et al. 2008).

SOLUTION: A LIGHT HYDROLOGICAL FOOTPRINT IN ROADLESS AREAS

IRAs should be managed in the same way many municipalities manage their watersheds—sustaining a light ecological and hydrological footprint and hydrologic restoration through decommissioning or, even better, obliteration of roads (Barten et al. 1998; NRC 2000; Payne et al. 2004; Gallo et al. 2005; Postel and Thompson 2005; Seeds 2010). The most cost-effective and prudent approach to maintain water supplies and high-quality fresh water in the face of population growth and climate change is to manage upper watersheds in a roadless condition with undisturbed natural vegetation. The high, long-term economic cost of degrading clean water for millions of people, by itself, is argument strong enough to continue protection of the current roadless areas network either at national or state levels. Development of IRAs, as proposed in Colorado, would primarily provide opportunities for short-term gains, but the substantial and long-term impacts on water quality and availability will come at a time of increasing demand and shrinking supply. Managers should, therefore, treat IRAs as natural reservoirs of high quality water for downstream users before approving development projects. Cost-benefit analyses should include regionally and locally specific estimates of water quality to better inform project management decisions that may reduce the value of high-quality water in the short and long run.

CONCLUSIONS

Roadless areas and the relatively intact ecosystems they maintain provide many important biodiversity benefits, including acting as strongholds for threatened freshwater species. Beyond these important values, their role in producing clean and reliable water for people and economies is more likely to compel decision-makers to leave roadless areas undeveloped. We reviewed the importance of inventoried roadless areas on national forests in the United States to determine their importance in providing clean water for downstream users. We concluded that (1) many intact watersheds are in headwaters, (2) they supply downstream users with high-quality drinking water, and (3) developing these watersheds comes at significant costs associated with declining water quality and availability. Several case studies from the western United States, particularly Colorado, demonstrated the importance of assessing the diverse consequences of developing roadless areas. Managers should perform comprehensive cost-benefit analyses when weighing development options. A light-touch hydrological footprint is recommended to sustain the many values that derive from roadless areas, especially clean and abundant water.

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chapter three

Fire-mediated Biological Legacies in Dry Forested Ecosystems of the Pacific Northwest, USA

Dominick A. DellaSala

3.1 What are Biological Legacies?

Biological legacies are the type, quantities, and patterns of biotic structures and organisms present before a disturbance that transfer their functions over to the post-disturbance environment (Franklin et al. 2002; Dale et al. 2005). Large (landscape-scale) fires of mixed-severity effects within dry-fire adapted Ponderosa pine (*Pinus ponderosa*) and mixed conifer forests of the Pacific Northwest USA, the subject of this chapter, generate pulses of biological legacies that can sustain ecosystems for decades to centuries (Franklin et al. 2000; Lindenmayer et al. 2008; Swanson et al. 2011; Donato et al. 2012; DellaSala and Hanson 2015). Whereas forest managers have focused mainly on legacies at the stand level, legacies also exist at landscape scales and as plant and wildlife populations persisting before and after a disturbance. Therefore, scale and context matter in managing legacies for biodiversity benefits.

In this chapter, my main objectives are to summarize the importance of forest legacies at multiple spatiotemporal scales, describe the ecosystem benefits of large fires that accrue legacies in space and time in these forests, and distinguish natural disturbance events (fire) from chronic ones caused by intensive land management (e.g., roads and short-rotation logging). I also provide ways to maintain legacies in areas intensively managed for timber production and where large landscape conservation is emphasized. I focus on fires of mixed-severity effects on dry forests, as a principal disturbance agent for legacy replenishment; other natural disturbances are not covered here. Mixed-severity fires generate a characteristic burn mosaic pattern in low- to mid-elevation dry forests in this region. The fire mosaic includes variably sized patches of lightly burned, moderately burned, and severely burned patches (DellaSala and Hanson 2015). This type of fire-mediated mosaic has been described as "pyrodiversity" and is associated with high

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levels of biodiversity in these forests (i.e., pyrodiversity begets biodiversity in this case, Parr and Brockett 1999; DellaSala and Hanson 2015). I focus on dry forests within the Northwest Forest Plan (NWFP) area of the Pacific Northwest, where extensive research on biological legacies has occurred (Franklin et al. 2000, 2002; Dale et al. 2005; Fontaine et al. 2009; Swanson et al. 2011; Donato et al. 2012; Odion et al. 2014), and where there is conservation interest in maintaining ecosystem benefits of large fires (DellaSala and Hanson 2015; also see DellaSala et al. 2017 for the Sierra region). Notably, the NWFP shifted federal land management from timber dominance to ecosystem management and biodiversity conservation across 10 million ha from northern California to Washington. The plan is considered a global model for large-landscape conservation, including maintenance of natural disturbance processes (DellaSala et al. 2015a).

3.2 Vegetation Types and Fire Regimes of the Pacific Northwest

Fire regime groups for this region ranged from frequently (0–35 years) occurring fires with low-severity effects on vegetation (i.e., most trees survive) to infrequent (>200 years) fires with high-severity (stand replacement) effects (i.e., most trees fire killed; Figure 3.1). Multiple lines of evidence indicate the preponderance of mixed-severity fire effects in dry forests of this region and the biodiversity importance of high-severity burn patches (>70 percent overstory mortality) within fire complexes (Hessburg et al. 2006; Perry et al. 2011; Baker 2012, 2015; Donato et al. 2012; Williams and Baker 2012; Odion et al. 2014). Importantly, the proportion and extent of high-severity fire has remained relatively constant from 1984–2011 in this region, replenishing legacies at landscape scales (Odion et al. 2014; Law and Waring 2015).

Mixed-severity fires generate pulses of biological legacies that result in structurally complex-early seral forests (Swanson et al. 2011). Although fires of generally low intensity may generate patches of high-tree mortality (fine-scale heterogeneity) in small-scale flare ups, they do not produce the landscape heterogeneity characteristic of mixed-severity fires. It is this precise variation in patch patterns that sets mixed-severity fire effects apart from fires of low-severity effects in terms of legacy production.

3.3 Legacies

Landscape

When viewed at the landscape scale, a collection of intact forest blocks in different post-disturbance stages can act as a legacy landscape. The

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heterogeneity present within intact areas is most notably distinguishable from the oversimplified human-dominated landscape (Figure 3.2a vs. 3.2b). Most of the Pacific Northwest is in a highly fragmented and altered condition consisting of few natural legacy blocks interspersed within a highly industrially managed matrix (Heilman et al. 2002; Strittholt and DellaSala 2001). Highly fragmented landscapes have very little legacy functions and the fragmented patches are mostly edge environments (Chen et al. 1992) that are degrading to the habitat of many plants and wildlife species, although individual patches, such as roadless areas embedded within the human-dominated matrix, maintain important legacy functions that may serve as "blueprints" for restoring larger blocks (Strittholt and DellaSala 2001). Legacy landscapes also include a mix of undisturbed and naturally disturbed sites that provide habitat for a suite of species distributed across the disturbance gradient (Figure 3.2c; Mackey et al. 2002; Thompson et al. 2007; Lindenmayer et al. 2008). Interestingly, what may function as an undisturbed patch for disturbance-avoiding species, may not for disturbance-adapted ones until the patch is perturbed, rendering it temporarily unsuitable for disturbance avoiders. Therefore, heterogeneity is important for maintaining biodiversity at the landscape scale.

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Legacies produced by small-scale disturbances such as tree fall gaps or a flare up within a predominately low-intensity burn provide fine-scale heterogeneity and production of localized legacies. Examples include large snags, downed logs, below-ground seed banks, and flowering plants that were present pre-disturbance and that persist in the newly created disturbed patch (Figure 3.3). Stand legacies perform myriad ecosystem functions as keystone structures (Franklin et al. 2000, 2002; Dale et al. 2005, Box 3.1).

BOX 3.1 STAND-LEVEL AND POPULATION LEGACIES

Many species of insects, birds, mammals, and herpetofauna rely on deadwood (snags and logs) for nesting, roosting, foraging, and hiding. In fact, some 67 species of vertebrates in this region are strongly dependent (use them >50 percent of the time) on snags (Bunnell 2013). The root wads of snags anchor soils, preventing erosion. Large snags shade conifer seedlings from intense sunlight aiding in reforestation. Downed logs absorb and slowly release runoff, which is especially important to moisture-dependent bryophytes, salamanders, terrestrial mollusks and many invertebrates in dry

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summer months. Logs decompose slowly and recycle soil nutrients while acting as "nurseries" for new plant growth (Figure 3.4). Wildlife populations present before the disturbance also may act as source populations post-disturbance (Lindenmayer et al. 2005). Animal carcasses connect the deceased with the living (scavengers) as a food legacy. Interestingly, spawned-out salmon (*Oncorhynchus* spp.) carcasses replenish soil nutrients for plants in riparian areas (Reimchen 2000).

From a social perspective, forest legacies connect human generations to the natural world in a way that is not easily quantifiable but important in providing solitude, wilderness, and a sense of the great outdoors (Figure 3.5). A hike in old-growth forest is a surreal connection to an earlier time when primeval forests prospered but today are remnants of what may be a bygone era because of climate change and ongoing land-use pressures.

3.4 Legacies Link Successional Pathways

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> When we try to pick out anything by itself, we find it hitched to everything else in the universe.
>
> **(John Muir)**

The quantity and quality of legacies at the stand and landscape levels depend on what was present before the disturbance. In this fashion, the post-disturbance environment is "hitched" to the quality of its predecessor stage and vice versa (Donato et al. 2012). For instance, an intense fire in a mature forest triggers the onset of the complex pioneering stage of forests characterized by abundant legacies that then set the stage for development of older forest conditions over time (Swanson et al. 2011; Figure 3.3 and 3.6a). This early stage can persist for up to two decades (depending on site productivity) until forest canopy closure; however, it rarely persists in managed areas because of post-fire logging (Figure 3.6b). The early stage is often disrupted by intense forest management as discussed below.

3.5 Pulse vs. Chronic Disturbances

Disturbance can be characterized as having pulse (rapid) or chronic (long-term) effects on ecosystems, depending on intensity and frequency (Frelich 2008). For instance, legacies are continuously replenished by reoccurring pulse disturbances, the timing of which is governed by localized

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natural disturbance dynamics. If the fire return interval is over a century, the later stages of forest succession (old growth) may ensue over time with legacies continuously replenished at the stand level by small-scale events (e.g., gap-phase dynamics). Additionally, large fires are the main event for perpetuating abundant quantities of legacies at landscape scales. Most ecosystems and species evolved with these types of pulse disturbances (stand and landscape) and have unique adaptations to cope and even thrive in the post-disturbance environment (Box 3.2).

BOX 3.2 DISTURBANCE ADAPTATIONS

Many native species have unique adaptations that allow them to prosper, resist, and/or be resilient to natural disturbances. The thick bark of large older fire-resistant trees, such as Ponderosa pine, insulate the tree from low- to moderate-fire intensities. The giant sequoia (*Sequoiadendron giganteum*) is considered the most fire-resistant conifer on Earth, capable of withstanding even crown fires. Other species not only are resilient to disturbed areas but prosper in them as well. Examples include seritonous cones of certain lodgepole pine (*P. contorta*) populations that release their seeds after intense fire heat. Below-ground seed propagules of some shrub species (e.g., *Ceanothus* spp.) also require intense heat to crack open the seed coat. Stump sprouting in madrone (*Arbutus menziesii*) and coast redwood are natural adaptations to disturbances. Many animal species are well adapted to colonize disturbed areas. The Black-backed Woodpecker (*Picoides arcticus*), aptly named for its cryptically colored back against charred tree boles, reaches its highest population levels in severely burned forests with abundant legacies that support beetle larvae, the woodpecker's predominant food-source (Hutto 1995). Some wood-boring beetles have specialized infrared receptors (pit organs, e.g., *Melanophila* spp.) for detecting heat or smoke as far away as 60 km (Schmitz and Bousack 2012). These "fire-chasers" quickly hone in on fire-killed trees and are essential to nutrient cycling post-disturbance.

In contrast, chronic disturbances from industrial-scale logging and road building differ from pulse disturbances in cumulative effects to ecosystems. Such disturbances often overlap and accumulate in space and time (e.g., post-fire logging, roads, grazing, and mining, of which two or more may occur on the same site at the same time), which can flip ecosystem dynamics and species composition to highly altered (novel) states (Paine et al. 1998; Lindenmayer et al. 2011, 2017; Box 3.3; Figure 3.7). The result is a homogenized landscape that lacks legacies and biodiversity.

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BOX 3.3 CHRONIC DISTURBANCES

Extensive logging and road building following fire is common in the Pacific Northwest (Figure 3.7; DellaSala et al. 2015b). Acting synergistically, several linked anthropogenic disturbances often exceed the evolutionary capacity of species and ecosystems to respond to cumulative disturbances (Paine et al. 1998). An example comes from the highly fragmented landscapes of the Klamath-Siskiyou ecoregion of southwest Oregon/northern California where industrial-scale logging has contributed to uncharacteristically severe fires because of the accumulation of logging slash and small densely stocked and flammable trees that have replaced native fire-resistant forests (Odion et al. 2004; Zald and Dunn 2018) (Figures 3.2b, 3.7). Most of the dead and live legacy trees are removed and this is followed by planting trees that often burn intensely in the next fire (Thompson et al. 2007). The logged area sometimes includes livestock grazing, colonization by invasive plants, and off-highway vehicle use (soil compaction) that can exceed disturbance thresholds of many native species. Such landscape-scale conversions set in motion a management feedback loop, whereby a natural disturbance regime is transformed to frequently occurring and cumulative disturbances with concomitant losses in biodiversity and forest legacies (Lindenmayer et al. 2011, 2017; DellaSala et al. 2015b).

3.6 *Large Fires and Biological Legacies*

Landscape-scale fires occur infrequently (decades, centuries) in forest ecosystems of this region but have disproportionate effects on ecosystems compared with small or low-intensity fires. Large fires of varied patch sizes and severities include severely burned patches with super-abundant legacies (Figure 3.2c, Figures 3.3 and 3.6). Such fires result in high levels of alpha (within stand) and beta diversity (species turn over across gradients) because of complex structures present at the stand-level and varied patch severities at the landscape scale (Hanson et al. 2015). In general, prescribed fires and low-intensity burns lack this heterogeneity (Swanson et al. 2011; DellaSala et al. 2017).

Notably, under "extreme" weather conditions (drought, high winds, high temperatures), fires are likely to reach landscape proportions and therefore result in a pulse of legacies. Drought monitoring, in particular, is useful in predicting the likelihood of large fires and therefore where legacy pulses may occur. Managers of national parks, for instance, may want to know when and where there is potential for a major-fire event

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headed their way so that they can manage the fire for ecosystem benefits. US drought-monitor maps (https://droughtmonitor.unl.edu) have commonly been used to alert fire suppression forces, but not to predict where ecosystem benefits may accrue from large fires.

3.7 Landscape Conservation and Disturbance Dynamics

Landscape-scale conservation requires knowledge of disturbance dynamics in conservation biology approaches (Noss and Cooperrider 1994; DellaSala et al. 2017). In mixed-severity fire regimes, this means developing a conservation strategy that:

1. Includes a protected-areas network representative of patch types (e.g., fire severities) well-distributed in redundant patterns that collectively are resilient to natural disturbances (i.e., the disturbance does not overwhelm the functionality of the reserve network given redundant types, DellaSala et al. 2015a, 2017);
2. Maintains viable populations of disturbance-adapted and disturbance-avoiders well-distributed across a planning area; and
3. Provides landscape connectivity of pre- and post-disturbed areas to allow dispersal of plants and wildlife, particularly in a changing climate.

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The reserve network of the NWFP incorporated resilience concepts into the design of the network. For instance, late-successional reserves (LSRs) are well-distributed and in redundant patterns so the loss of any particular LSR(s) should not overwhelm the integrity of the overall reserve network (DellaSala et al. 2015a). However, very few reserves are large enough to fully encompass large fires and therefore managing fire for ecosystem benefits across large landscapes needs to include both protected and other land-use designations. For example, although 71,351 ha (39 percent) of the 184,354 ha 2002 Biscuit fire perimeter in southwest Oregon occurred within the Kalmiopsis Wilderness, extensive fire suppression was employed throughout the burn area, bulldozed fire lines created travel corridors for weedy species, post-fire logging damaged LSRs and encouraged on wilderness boundaries, and intense logging was conducted in the surroundings, rendering the landscape prone to future intense burns and biodiversity loss (DellaSala 2006). Over-reliance on fire suppression and post-fire logging is a shortcoming of fire management strategies in general and the NWFP in particular (DellaSala et al. 2015a). Specifically, the NWFP does not include strict provisions for maintaining LSRs post-intense burn, nor does it focus on viable populations of fire-dependent species and is therefore prone to post-fire biodiversity losses from logging.

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3.8 Legacy Management

> Ecosystems are not only more complex than we know, they are more complex than we can know.
>
> **(Former U.S. Forest Service Chief, Jack Ward Thomas (but the quote has more historical origins))**

Given the importance of legacies, can management mimic natural landscape patterns and disturbance processes responsible for legacies?

This question has been central to much research, trial and error management (often without trials), and considerable controversy. In the Pacific Northwest, most of the older forests have been logged (Strittholt et al. 2006) and therefore there is not much room for trial and error management, particularly practices that further degrade these forests and the post-fire conditions that replenish them over time. For instance, complex-early seral forests are now considered a rare type because they are almost always logged following a fire (DellaSala et al. 2015c). However, the closer management can come to mimicking natural processes in form and function, the more likely it can sustain at least some legacy elements that can "lifeboat" important ecosystem functions from the pre- to post-disturbed environment (Hunter 1999). Studies of legacy structures in this region provide a foundation for determining baseline or reference conditions for managers wishing to provide specific structural features, such as snags and down wood, particularly in relation to fire severity (see Sillett and Goslin 1999; Bull 2002; Weisberg 2004; Keeton and Franklin 2005; Tepley et al. 2013; Brown et al. 2013; O'Halloran et al. 2014; Merschel et al. 2014; Dunn and Bailey 2016). Here I build on these recommendations with specific actions for improving upon the NWFP so managers can aptly recognize the ecosystem benefits of fire.

Develop clear, measurable objectives that place ecosystem benefits of fire on par with suppression. An ecosystem benefits analysis (Ingalsbee and Roja 2015; DellaSala et al. 2017; also see www.forestandrangelands.gov) for fires is needed to place the importance of fire on par with that of suppression, allow managers to evaluate the conditions under which fires can be managed for such benefits, and determine how best to weigh tradeoffs and collateral damages to ecosystems from pre- and post-fire logging. Assessments should include long-term benefits of fire in creating pyrodiverse landscapes, complex-early seral structures, and fire refugia as part of multiple resource objectives. Ecosystem benefits could then be quantified and integrated with suppression, community safety, and other values (Ingalsbee and Roja 2015; DellaSala et al. 2015c, 2017).

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Manage large fires for pyrodiversity in protected areas and legacy landscapes on public lands. The more fire is managed for pyrodiversity effects, the more the landscape will generate legacies. To minimize human-safety conflicts, large fires can be compartmentalized so firefighting resources target the fire front closest to homes, whereas other portions of the fire in remote areas are managed for ecosystem benefits (Ingalsbee and Roja 2015).

Manage individual forests and landscapes for complexity before disturbance. Management objectives that include retention of pre-disturbance stand and landscape structures provide the foundation for legacies to regenerate when a natural disturbance eventually happens. Managers may provide ecosystem benefits by altering the quantity, quality, and spatial arrangement of forest structures at stand and landscape scales (Figure 3.8). In particular, restoration of degraded sites should focus on promoting large, irregularly shaped and clumped spacing of large-legacy structures well distributed at the stand and landscape scale, which is preferred over small-homogenized and widely spaced legacies.

Managers can also create structure in otherwise homogeneous plantations by girdling trees in clumps (Figure 3.8C). Irregular spacing of canopy gaps (not shown in Figure 3.8) allows shrubs and forbs to persist by extending the time a site remains in the complex-early stage (Swanson et al. 2011). Also see Zenner (2000), Aubry et al. (2009), and Keeton and Franklin (2005) for effects of varied retention standards on forest ecosystems.

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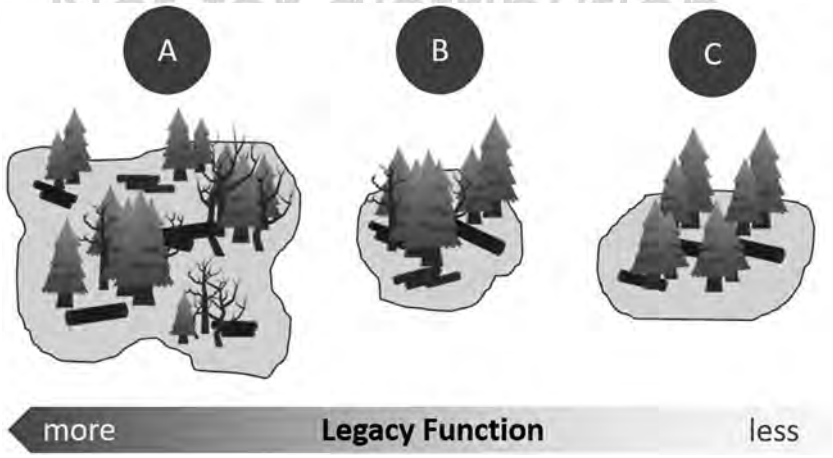

Figure 3.8 Decreasing legacy functions from left to right. Large irregular patches (A) function more as intact legacies, whereas small ones (B) have diminished functions. Clumping of legacies, particularly large snags and logs, is preferred over smaller tree sizes and regular spacing of legacies (C). Restoration involves moving degraded sites (C) to restored (A) sites with intact legacy functions based on reference conditions. Prepared by C. Mills, Geos Institute.

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Felling and tipping some large trees into streams where in-stream habitat structure is lacking creates legacies for aquatic organisms. Large logs left on the ground are "nurse logs" for seedlings and habitat for scores of moisture-dependent invertebrates, fungi, mosses, and herpetofauna. "Feathering" edges reduces contrast between managed and unmanaged areas while minimizing edge effects.

Manage individual forest stands for legacies after disturbance. Post-fire logging is a chronic disturbance that compacts soils, removes large legacies (live and dead), retards natural conifer regeneration, and is often coupled with herbicide treatments and road expansion (summarized in Lindenmayer et al. 2008; DellaSala et al. 2015b). In the Pacific Northwest, large-scale post-fire logging is most often proposed even in LSRs, roadless areas, and other "protected areas" based, in part, on the assumption that management is needed to "recover" these areas (DellaSala et al. 2015b). Instead, managers wishing to provide legacies should operate under the premise that if the area was high-quality wildlife habitat before a disturbance (e.g., LSR, roadless area) it remains so after the disturbance because what happens in the post-disturbance environment determines the trajectory of ecosystems through time (i.e., if degraded early in the succession, the subsequent stages may be irreparably harmed; Lindenmayer et al. 2014).

Exercise caution in use of back-burns to mediate large fires. Back-burns are fires intentionally set by managers to slow or reduce the intensity of an advancing fire front by reducing flammable vegetation under more "controlled" conditions before the main fire arrives. Although important as a fire safety precaution, back-burns often escape containment (e.g., under windy, dry, and hot conditions) and therefore can compound disturbance effects particularly in wilderness and other high-conservation value areas (Backer et al. 2004; Ingalsbee and Roja 2015). Back-burning is often associated with cumulative disturbances, such as bulldozed fire lines that become de facto roads and corridors for flammable weedy plants. Documenting where back-burns have occurred, avoiding additional disturbance (post-fire logging), and restoring fire lines by recontouring slopes and seeding with native plants are critical steps in reducing cumulative impacts. Wherever possible, managers should employ minimal fire suppression tactics (MIST) in roadless areas, wilderness, parks, and other lands of high-conservation value (Ingalsbee and Roja 2015).

Restore ecosystems by addressing linked human-caused stressors, particularly roads, livestock, off-highway vehicles, and other chronic disturbances. The chronic effect of roads on ecosystems is well-documented and generally includes unnatural fire ignitions, spread of invasives, chemical and air pollution, wildlife–vehicle collisions, water-quality degradation, and habitat fragmentation (see Ibisch et al. 2017). Managers may wish to reduce these chronic effects by seasonal road closures (to limit human-caused ignitions), repairing and obliterating failing roads, improving culvert

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design, and containing flammable weedy plants. The same is true for livestock grazing, off-highway vehicles, and other chronic disturbances. These should be restricted in burn areas. Restoration of areas degraded by intensive management should include both active and passive (remove the stressor) measures that reduce chronic effects.

3.9 Conclusions (So What?)

Legacies come in all different shapes, sizes, and functions. They are produced by varied disturbances acting at multiple spatiotemporal scales. Large fires of mixed-severity effects on ecosystems are critical to forest disturbance dynamics because they provide unique pulses of biological legacies not found in managed forests, except by way of fires that escape containment, nor are they mimicked by low-intensity or prescribed burning (DellaSala and Hanson 2015; DellaSala et al. 2017). The likelihood that legacy landscapes will be managed for ecosystem benefits is increasingly constrained by the economics of forest management and a preoccupation with fire suppression activities.

Recent research has shown complex-early seral forests with prolific legacies are as rich in biodiversity as the more heralded old-growth forests, yet they are greatly underappreciated (Swanson et al. 2011; DellaSala and Hanson 2015; DellaSala et al. 2017). Thus, greater awareness of the importance of these newly emerging forests, the quantity and quality of legacies before and after disturbance (i.e., antecedent conditions and past disturbance legacies), and the large fires that generate them are urgently needed (DellaSala et al. 2015b). In particular, extensive post-fire logging and fire suppression are the main drivers of ecosystem degradation in fire-adapted systems and why complex-early-seral forests are among the rarest in the Pacific Northwest (Swanson et al. 2011; DellaSala et al. 2015b). Homogenization of landscapes through fire suppression and a desire to manage fires for low-severity vegetation effects may be altering important evolutionary processes that have structured fire-resilient communities for millennia. The emergence of novel climates in conjunction with unprecedented anthropogenic disturbances may flip disturbance-adapted ecosystems to highly altered states that exceed disturbance thresholds (Paine et al. 1998; Lindenmayer et al. 2011). Thus, managers wishing to maintain pyrodiverse landscapes and their associated biota need to manage for complexity before fires, accommodate large fires for ecosystem benefits wherever possible, and prohibit post-fire logging and related activities to allow for the full expression and development of linked seral stages.

Most reserves are too small to fully incorporate large-scale disturbance dynamics unimpeded by fire suppression and thus declines in disturbance-associated biota are anticipated. Reserve-based approaches aptly have focused on late-successional forests in this region because of

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their conservation status (Strittholt et al. 2006). However, protected areas need to be as large as possible, selected to represent varied patch types in redundant distributions, and the surroundings integrated through compatible management activities that maintain viable populations of well-distributed disturbance-adapted and disturbance-avoiding species.

We encourage researchers to focus attention on: (1) the unique biota (especially rare and declining species) of these forests and their special adaptations; (2) habitat and population requirements of fire-adapted species; (3) intensity and frequency of fires necessary to maintain fire-adapted species, particularly in a changing climate; (4) spatial distribution, patch types, and sizes needed for these species; and (5) effects of different management intensities on fire-adapted species and ecosystems. Notably, comparative studies are needed that contrast homogenized-early seral clearcuts, early seral forests with partial legacy retentions, and unaltered complex-early seral forests (reference conditions). This is necessary to generate relevant legacy stand and landscape prescriptions, aid in restoration of degraded areas, and assuage concerns that fires in these forests are ecologically "catastrophic." Instead, such disturbances are ecologically necessary to maintain pyrodiverse landscapes and the pulse of complex early seral structures that follow (DellaSala and Hanson 2015; Lindenmayer et al. 2017).

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Public land, timber harvests, and climate mitigation: Quantifying carbon sequestration potential on U.S. public timberlands

Brooks M. Depro [a,*], Brian C. Murray [b], Ralph J. Alig [c], Alyssa Shanks [d]

[a] RTI International, 3040 Cornwallis Road, P.O. Box 12194, Research Triangle Park, NC 27709, United States
[b] Duke University, Durham, NC, United States
[c] USDA Forest Service, Pacific Northwest Research Station, 3200 SW Jefferson Way, Corvallis, OR 97331, United States
[d] Alaska Department of Labor and Workforce Development, Research and Analysis Section, Juneau, AK 99801, United States

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Abstract

Scientists and policy makers have long recognized the role that forests can play in countering the atmospheric buildup of carbon dioxide (CO_2), a greenhouse gas (GHG). In the United States, terrestrial carbon sequestration in private and public forests offsets approximately 11% of all GHG emissions from all sectors of the economy on an annual basis. Although much of the attention on forest carbon sequestration strategy in the United States has been on the role of private lands, public forests in the United States represent approximately 20% of the U.S. timberland area and also hold a significantly large share (30%) of the U.S. timber volume. With such a large standing timber inventory, these forested lands have considerable impact on the U.S. forest carbon balance. To help decision makers understand the carbon implications of potential changes in public timberland management, we compared a baseline timber harvest scenario with two alternative harvest scenarios and estimated annual carbon stock changes associated with each. Our analysis found that a ''no timber harvest'' scenario eliminating harvests on public lands would result in an annual increase of 17–29 million metric tonnes of carbon (MMTC) per year between 2010 and 2050—as much as a 43% increase over current sequestration levels on public timberlands and would offset up to 1.5% of total U.S. GHG emissions. In contrast, moving to a more intense harvesting policy similar to that which prevailed in the 1980s may result in annual carbon losses of 27–35 MMTC per year between 2010 and 2050. These losses would represent a significant decline (50–80%) in anticipated carbon sequestration associated with the existing timber harvest policies. If carbon sequestration were valued in the marketplace as part of a GHG offset program, the economic value of sequestered carbon on public lands could be substantial relative to timber harvest revenues.
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Keywords: Public timberland; Forestry; Climate change; Carbon sequestration

1. Introduction

Forest ecosystems play an important role in the global carbon cycle, absorbing large amounts of atmospheric carbon dioxide (CO_2) through photosynthesis and emission of CO_2 to the atmosphere through respiration, decomposition, and disturbances such as timber harvesting, fire, pest infestations, and land use change. Globally, terrestrial ecosystems are a net carbon sink[1] because removals and storage of CO_2 from the atmosphere (about 2300 million metric tonnes of carbon [MMTC] per year) exceed emissions (1600 MMTC per year) (IPCC, 2000). Most of the terrestrial sink is in forests. The global carbon balance masks some regional disparities; for instance, tropical forests are a source of emissions as deforestation outpaces regrowth, while the reverse is true currently in temperate forests, which are a net sink. The latest data for the United States indicate that land use, land use change, and forestry (predominately forest) comprises a net carbon sink of over 210 MMTC per year, offsetting about 11% of the country's GHG emissions (U.S. EPA, 2006).[2]

* Corresponding author. Tel.: +1 919 541 6729; fax: +1 919 541 6683.
E-mail address: bmd@rti.org (B.M. Depro).

[1] A carbon pool is a net sink if, over a certain time interval, more carbon is flowing into the pool than is flowing out of the pool. Conversely, a carbon pool can be a net source of CO_2 emissions if less carbon is flowing into the pool than is flowing out of the pool.

[2] Note that EPA data are reported in teragrams (million metric tonnes) of CO_2 equivalent (Tg CO_2). One ton of carbon equals 3.667 tons of CO_2.

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Expanding the area of land in forest cover, avoiding deforestation, and managing existing forests to store carbon in ecosystem stocks for longer periods by increasing the length of time between harvests can increase the net size of the carbon sink or, in some cases, turn a source into a sink. This has been recognized in the global and domestic policy arenas as a mix of mandatory and voluntary initiatives have sprung forth in the last decade that incentivize expansion of carbon sinks as a climate mitigation strategy. In the United States, much of the emphasis has been on incentives to expand carbon sinks on private lands (U.S. EPA, 2005; Lewandrowski et al., 2004; Richards and Stokes, 2004; McCarl and Schneider, 2001; Adams et al., 1999; Stavins, 1999; Plantinga et al., 1999). The more limited work regarding estimates of public lands' contribution to the U.S. carbon sink pertains to the projection of the status quo or business-as-usual case or BAU (Turner et al., 1995; Smith and Heath, 2004) or to regional contributions (e.g., Alig et al., 2006). Yet public timberlands constitute a sizable share of the U.S. forest resource in terms of both land area and timber volume (see Section 2) and thereby provide a potentially important resource to manage for climate change mitigation.

This paper departs from the literature by examining public timberlands' forest carbon sequestration potential at a national scale, not only under BAU conditions, but also under changes in forest management. The change in public forest management addressed in this paper is the level of allowable timber harvests, with two alternative scenarios to BAU defining the range of options from no timber harvest (elimination of all timber harvests on public timberlands) to a return to the historically high harvest period of the 1980s. Public land managers could consider other forms of forest management, such as modified rotations and intensive management of inputs, but those remain outside the scope of this paper.

The next section of the paper provides a brief overview of the public forestland resources in the United States, followed by a description of the data and methods used in the analysis and presentation of results for public timberlands. The paper ends with policy conclusions that can be drawn from the study and suggestions for future work.

2. Public timberland in the United States

The contiguous 48 (C48) states have approximately 228 million acres of public forests. Approximately 80% of this land, or 182 million acres, is in federal ownership (W.B. Smith et al., 2004; J. Smith et al., 2004). States, counties, and municipalities own the remaining 46 million acres; approximately 61% (138 million acres) of the public forestland is classified as *timberland* because it meets site productivity criteria and is not withdrawn from timber utilization by statute or administrative regulation.[3] Public timberland in the C48 states is concentrated in the West (west of the 100th meridian), which holds about 80% of U.S. public forestland. The top six states in order of public timberland area are Oregon, Idaho, Montana, California, and Colorado/Washington (tie).

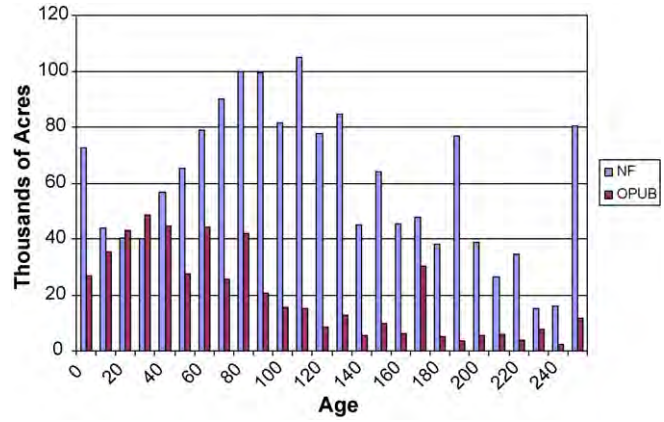

Fig. 1. Distribution of national forests and other public lands acres by age class: 2000.

Although the public owns a significant share of U.S. timber resources, they contribute a much smaller fraction of total U.S. timber removals. Public timberlands held 41% of growing stock inventory in 2001. The largest concentration of public timberlands is on National Forest (NF) lands, which alone held 30% of U.S. timber growing stock in 2001 (W.B. Smith et al., 2004; J. Smith et al., 2004). However, public timberlands produced only 8% of the U.S. timber removals in 2001, with NF lands providing just 2% of U.S. timber removals in 2001. Public policy makers have reduced timber harvests in favor of other nontimber outputs (e.g., wildlife, recreation, watershed protection, scenic amenities) since the late 1980s (Wear and Murray, 2004). Note that annual mortality is larger in volume than growing stock removals on both NF and other public (OPUB) timberlands, while net growth volume is at least two times the amount of mortality volume for those ownerships, leading to a net accumulation of growing stock and carbon. For example, in the case of NF timberlands, many acres are in young age classes with relatively rapid growth. However, public timberlands hold a relatively large share of the nation's older timber on timberland, especially on NFs, as shown in Fig. 1.

3. Analysis scenarios

Current management of U.S. public forestlands centers on a mix of environmental and socioeconomic objectives. For example, the Northwest Forest Plan (NWFP) covers almost 25 million acres and addresses northern spotted owl population and habitat, marbled murrelet population and habitat, late successional old-growth habitat, watershed conditions, and socioeconomic characteristics. Monitoring efforts are also underway to evaluate the success of the NWFP in achieving its objectives based on new scientific knowledge on key topics that include old-growth forest habitat, watersheds, and rural economies. Currently, carbon sequestration is more a by-product

[3] Timberland is defined as forestland that can produce 20 ft^3 of industrial wood per acre per year in naturally regenerated stands and that is not withdrawn from timber utilization by statute or administrative regulation (W.B. Smith et al., 2004; J. Smith et al., 2004).

than a primary management objective of the plan, but that could change with the renewed interest in climate change mitigation at the federal level in the United States (Paltsev et al., 2007).

For this analysis, we characterize a baseline (referred to as the BAU timber harvest scenario) and compare and contrast annual carbon stock changes associated with two alternative timber harvest scenarios. The baseline scenario for public timberlands identified by Mills and Zhou (2003) was derived from the USDA Forest Service's (USFS's) Washington office and represents expectations at that time based on guidelines of USFS policy. Timber harvests are drawn from a characterization that we call a ''removals scenario'' after Mills and Zhou (2003) and were allocated according to the number of acres in each age class (see below). Regeneration volumes were based on ATLAS model (Mills and Kincaid, 1992) projections of forest inventory (see below for details).

The first alternative scenario, ''no harvest,'' eliminates timber harvest completely and thereby reflects nontimber forest management objectives in the extreme. NF timber stands are assumed to grow without any timber harvest-related disturbances for the next 100 years. Mills and Zhou (2003) assumed that other naturally occurring disturbances such as fire, insects and diseases, and other natural mortality would remove timber volume and require the natural regeneration of an additional 140,000 acres annually. This acreage number came from the average rate of acres disturbed in the 10 years preceding the publication of ''Projecting National Forest Inventories for the 2000 Resources Planning Act (RPA) Timber Assessment'' by the USDA Forest Service (Mills and Zhou, 2003). The disturbed acres were taken from the two dominant forest types, those occupying the largest acreage. Within the two dominant forest types, disturbed acreage was removed from every age class above the minimum harvest age for the ATLAS model.

The second alternative, ''high-harvest/pre-1989'' scenario, follows timber harvest levels as depicted in the 1989 USFS's Timber Assessment (USDA Forest Service, 1990), the most recent period of timber harvesting on public timberlands that is above historical averages. These timber harvest levels, as reported in the 1989 RPA Assessment, for NFs came from the forest plans in effect or drafted in 1987 in response to the National Forest Management Act of 1976. NFs at that time provided about two-thirds of timber harvests from public timberlands, and NF timber harvest was assumed to increase by about 400 million ft^3, from 2.3 billion in 1986 to 2.7 billion by 2040. The 1986–2040 projected harvest levels took into consideration the anticipated impacts at that time of the Threatened and Endangered Species Act of 1973. The scenarios are intended to convey differences in forest carbon and carbon that is disposed of off-site – in products, landfills, and energy use – under different timber harvest assumptions.

As shown in Fig. 2, the BAU timber harvests per decade from public timberlands in 2010 range from 15 to 20 billion ft^3 during the period of the analysis. Approximately two-thirds of the harvests come from other public timberlands (see Fig. 2a and b), a reverse of the relative contributions of the two major sources of public timber harvest in 1986. In contrast, the pre-1989 scenario harvests per decade are significantly higher and

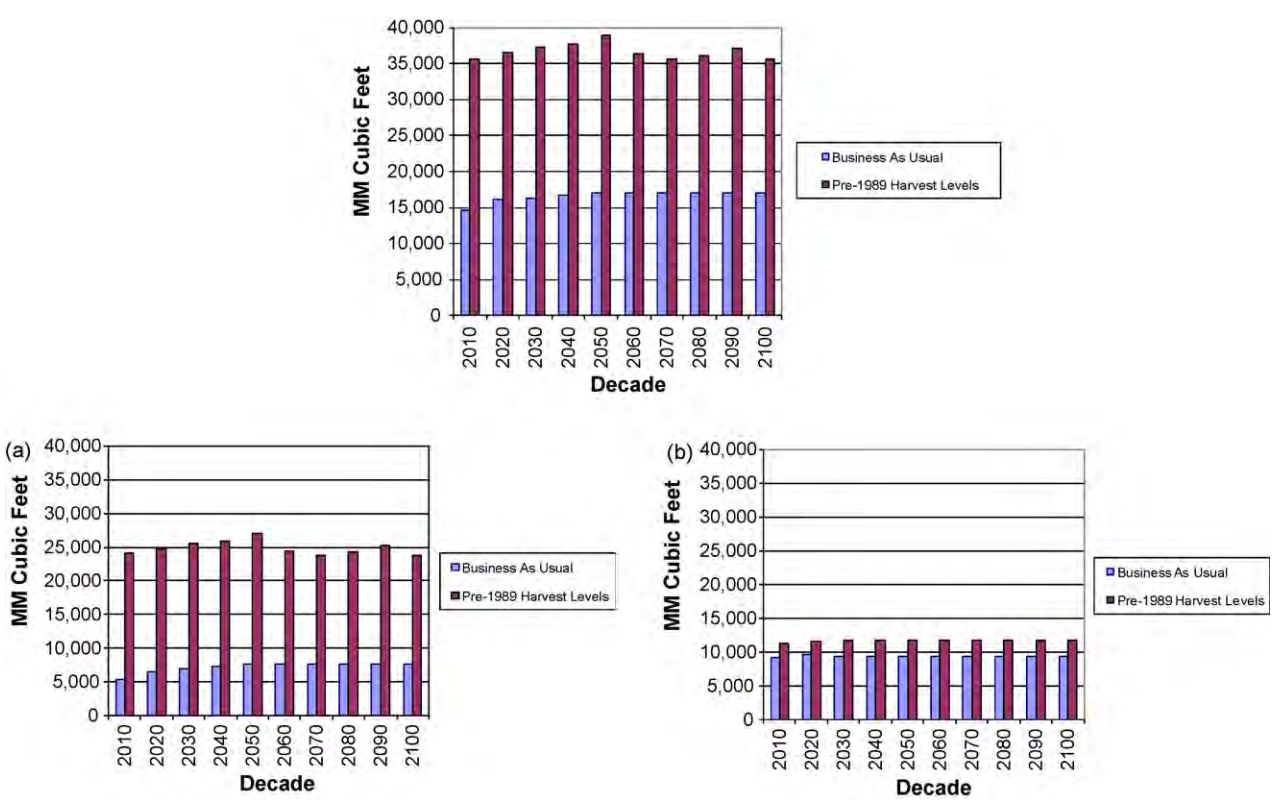

Fig. 2. Total public timberland harvests by decade and scenario 2010–2100. This includes harvests from (a) National forests and (b) other public lands.

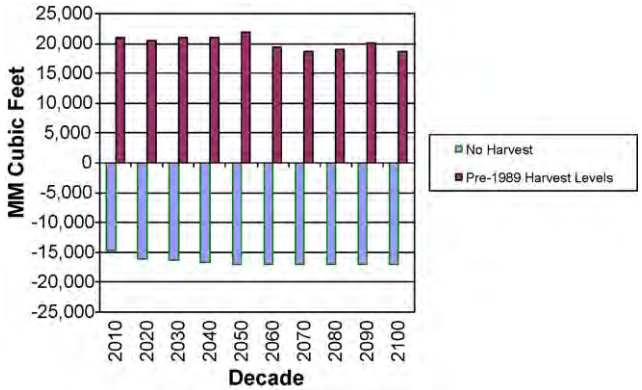

Fig. 3. National forests and other public lands: changes from BAU harvest volume by scenario.

range from 35 to 40 billion ft^3. Timber harvests in these scenarios increasingly rely on NF lands, with approximately two-thirds of the decades' total harvests coming from NFs.

As shown in Fig. 3, the no-harvest scenario reduces public timber harvests by approximately 15 billion ft^3 per decade. Presumably, this scenario will increase carbon stocks by avoiding carbon losses associated with converting standing forests into wood products. In contrast, the pre-1989 scenario increases baseline public timber harvest levels by approximately 20 billion ft^3 per decade. As a result, carbon losses will increase as more timber is removed. Our analysis is designed to estimate, compare, and contrast annual carbon stock changes associated with the two radically different timber harvest scenarios.

4. Data and methods

Simulating public forest management requires data specific to public timberlands on a range of variables, including land class, timberland area, forest type, timber yields for specified land management trajectories, growing stock or biomass volume by age class, site productivity, and regeneration yields. These data also need to be linked to data or models that quantify the relationship between these variables and carbon storage.

4.1. Timberland inventory

Public timberland data were obtained from ATLAS modeling used in the 2000 RPA Timber Assessment. We assembled the inventory data, along with existing and regenerated timberland yield projections for NF aggregates, using strata identical to those used in the private timberland tables in the Forest and Agricultural Sector Optimization Model-Greenhouse Gases, or FASOMGHG (McCarl et al., 2005; Adams et al., 1996). Data for projections came from USFS Forest Inventory Analysis (FIA) permanent sample plots. Collected data for NF and OPUB timberlands were stratified by region, ownership, forest type, and age class (Mills and Zhou, 2003). Assembling inventory data included identifying public timberland area and growing stock volumes by age, land class, region, forest type, site class, and broad management intensity class. Timber growth and yield relations were developed from a broad cross section of field plots. In ATLAS, timber management intensity classes correspond to a specific regime of silvicultural treatments to represent a regional average response for a particular forest type. The management intensity classes are initially populated with a timberland inventory derived from forest survey plots. Empirically derived parameters dictate forest stand development in terms of net growing stock volume as the ATLAS model simulates growth, timber harvesting, and regeneration. The ATLAS modeling approach has been applied in regional and national timber resource assessments, for modeling of changes on both private and public timberland.

Mills and Zhou (2003) provided public timberland data, based on USFS Forest Inventory and Analysis (FIA) plots. We used 5-year age classes to represent public timberlands, up to ages of 250+ in all regions except the South, where the oldest age class was 90+ for the generally younger forests held there. Some Northeast and South Central plots did not have age class data assigned by the FIA units; for these plots age was assigned using a method that considers volume and stocking.

Age class is one of the parameters used to calibrate the yield functions that determine volume; another parameter is region. Nine timber supply regions were designated to categorize the United States described in Mills and Zhou (2003). These regional designations help organize forest area into areas of similar growth characteristics, making the model more accurate than if only one yield function were used for the entire United States.

Across all regions, forestland was aggregated into softwood and hardwood forest-type groups. In the Pacific, Rocky Mountain, Lake States, and Corn Belt regions, all land with trees over 250 years old was aggregated into the age cohort of >250. In the Southern regions, land with trees 90 years or older was aggregated into the uppermost age cohort of >90. In the Southeast and South Central regions, ATLAS was unable to project yields of older stands for the entire 100-year time horizon. In the older stands, the total volume within the strata was used to extrapolate yield curves throughout the projection period. Based on data limitations, each stand in the inventory was assigned a medium site class. Public timberland only occurred on the FORONLY ("forest only") land class, areas not suitable or not available for conversion to crop or pasture. Because of this limitation, no conversion is allowed to agriculture on public land, which, regardless of whether it is biophysically feasible to do so, is not likely to occur for legal and political reasons.

Timber management intensity on NF timberland consists of three categories: a low intensity of even-age management, uneven-age management, and reserved (Mills and Zhou, 2003). Other public timberlands only had the low intensity of timber management. With a low intensity of timber management, no significant intermediate stand treatments are assumed to occur between stand establishment and final harvest.

Timber stands are final harvested over a range of stand ages. The uneven-age regime allows partial cutting (Mills and Zhou, 2003), where a treatment removes a portion of timber volume to reflect a stand subject to multiple entries. Timberland in a

reserved class is not available for timber harvest, but growth of the reserved stands is projected forward in time. The number of acres assigned to these regimes was derived from a survey of NF regional silviculturists (Mills and Zhou, 2003). The majority of the NF acres are assigned to either the partial cutting or reserved classes.

Timber yield estimation for regenerated stands was based on the ATLAS model approach. ATLAS calculates regeneration failures by region and used lagged yields to reflect failed cases. ATLAS has acres remain in the youngest timber age class for an extra 5 years for the South or 10 years elsewhere. Lacking data on pre- and postdisturbance forest types, all regenerated stands returned to the same forest type from which they originated in the same proportions of hardwood and softwood as they had before disturbance.

Assumptions concerning future harvest patterns and land base changes included that the public timberland area does not change over the planning horizon. All clear-cut harvested acres are regenerated as a single stratum with the other harvested acres in that same period and region. Harvests are distributed according to area in each age class; no age class or management intensity is excluded from harvest except for reserved acres.

4.2. Carbon projection methods

Our analysis calculates the stocks and flows (fluxes) of carbon on public timberlands in the United States, including NF and OPUB lands. These estimates are based on USFS projections of future timberland inventories and timber harvest levels, forest carbon accounting equations of the USFS FORCARB2 model (see below), and wood product accounting methods based on the previous work of Smith et al. (2006). As shown in Fig. 4, the carbon accounting framework separates forest carbon calculations into two parts: the accumulation of forest ecosystem carbon as forested stands mature before harvest and the disposition of carbon into various destination pools after the point of harvest. We discuss each component below.

4.2.1. Forest ecosystem carbon accumulation before harvest

On-site carbon accounting closely mirrors the FORCARB2 system used by the USFS in their aggregate assessments of forest carbon sequestration. Using this framework, carbon accumulates in four pools and we describe each below:

- trees
- understory
- forest floor and coarse woody debris
- soil

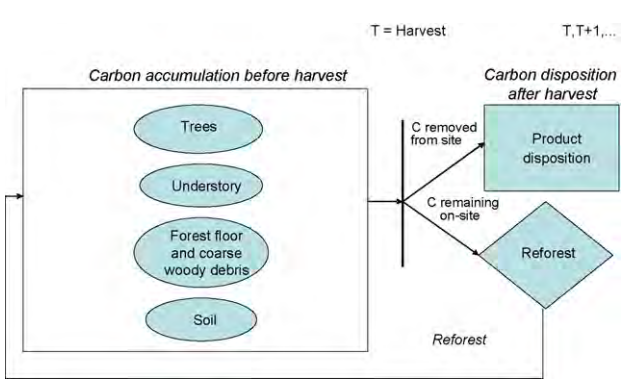

Fig. 4. Carbon accounting framework.

4.2.1.1. Trees. In FORCARB2, tree carbon is a function of two factors: merchantable timber volume and parameters of a forest volume-to-biomass model developed by USFS researchers (Smith et al., 2003). Merchantable volume, by age, on each representative stand is obtained from the timber growth and yields tables in the ATLAS model described above. Tree carbon includes live and standing dead tree carbon and is calculated using the parameters of the forest volume-to-biomass model equations for live and dead tree mass densities (above and below ground) in Smith et al. (2003).[4] Birdsey's (1992) assumption that mass of wood is approximately 50% carbon is used to derive the associated quantity of carbon:

$$C^{\mathrm{R}} = \left(\frac{D^{\mathrm{L}} + D^{\mathrm{D}}}{U^{\mathrm{B}}}\right) \times 0.5, \tag{1}$$

where live and dead tree biomass are computed as

$$D^{\mathrm{L}} = F^{\mathrm{w}} \times (G^{\mathrm{vbw}} + (1 - \exp^{(-V^{\mathrm{T}}/H^{\mathrm{vbw}})})) \tag{2}$$

$$D^{\mathrm{D}} = D^{\mathrm{L}} \times A^{\mathrm{vbw}} \times \exp^{(-(V^{\mathrm{T}}/B^{\mathrm{w}})^{C^{\mathrm{vbw}}})}. \tag{3}$$

The variables in these equations are reported in Table 1.

4.2.1.2. Understory. Understory vegetation is the smallest component of total carbon stock and includes all live vegetation except trees larger than seedlings. In this analysis, understory carbon is a fixed fraction of live tree carbon based on published ratios reported by the U.S. EPA (2003). Weighted ratios for regions/forest types are created using forestland area data reported by the USDA Forest Service (Miles, 2003).

$$C^{\mathrm{U}} = \frac{D^{\mathrm{L}}}{U^{\mathrm{B}}} \times 0.5 \times R^{\mathrm{Uw}} \tag{3}$$

The variables in this equation are defined in Table 2. The weighted parameters used are reported in Table 3.

4.2.1.3. Forest floor and coarse woody debris. Forest floor carbon constitutes the third largest carbon storage pool, but this pool is much smaller than tree or soil carbon pools. Smith and Heath (2002) developed a model for estimating forest floor carbon mass, which forms the basis for the forest floor carbon estimates used here. Their model's definition of forest floor excludes coarse woody debris (CWD) materials (i.e., pieces of dead wood that are not attached to trees). CWD includes large woody material fallen or cut and left from live and standing

[4] The parameters used are weighted for the economic model's (McCarl et al., 2005) region/forest-type designations. Forestland area data reported in the RPA Assessment (Miles, 2003) are used to calculate the appropriate weights.

Table 1
Tree carbon variables and parameters

Symbol	Description	Source
D^{L} (Mg C/ha)	Live tree mass density (above and below ground)	See Eq. (2)
D^{D} (Mg C/ha)	Dead tree mass density (above and below ground)	See Eq. (3)
C^{R} (Mg C/acre)	Total tree carbon	See Eq. (1)
V^{T} (m^3/ha)	Total timber volume	–
F^{vbw}, G^{vbw}, H^{vbw}	Weighted live tree density parameters from volume-to-biomass equations	Smith et al. (2003) Table 3 weighted by forestland area data from RPA (Miles, 2003) Tables 5 and 6
A^{vbw}, B^{vbw}, C^{vbw}	Weighted dead tree mass density parameters from volume-to-biomass equations	Smith et al. (2003) Table 4 weighted by forestland area data from RPA (Miles, 2003) Tables 5 and 6
U^{B} (1 hectare (ha) = 2.471 acres)	Units conversion factor	–

Mg C = megagram ("metric" tonne) of carbon equivalent m^3 = cubic meters of timber volume.

dead trees with a diameter of at least 7.5 cm (W.B. Smith et al., 2004; J. Smith et al., 2004). CWD accumulates over the life of a forested stand. At the time of harvest, a relatively large component of CWD may be left on site, which decays over time as the next rotation of trees grows. To account for effects of growth, mortality, disturbance, and decay of carbon in this material, we assumed CWD is a fixed fraction of tree carbon. Published ratios of CWD carbon to live tree carbon reported by the U.S. EPA (2003) were weighted for regions/forest types using forestland area data reported by the USDA Forest Service (Miles, 2003). This formulation of the CWD model clearly has limitations because CWD dynamics depend on the time since harvest and the amount of dead wood left after the disturbance. Although we view the results of the simulations using the current CWD model as fairly robust, given the relatively small factor that CWD plays in stand dynamics over time, the CWD model likely underestimates CWD stocks. Future CWD modeling work could adopt methods similar to recently published work (Smith et al., 2006).

The model for net accumulation of forest floor carbon is a continuous and increasing function of age. The rate of accumulation eventually approaches zero (i.e., a steady-state level of forest carbon):

$$C^{FFA} = \left(\frac{A^{ffw} \times \text{age}}{B^{ffw} + \text{age}}\right) / U^{B} \tag{4}$$

The variables in this equation are defined in Table 4.

Forest floor carbon mass following clear-cutting is assumed to begin at the level of carbon for a mature forest, and decay is described using an exponential function of time and average mature forest floor carbon mass:

$$C^{FFR} = (C^{ffw} \times \exp^{-(\text{age}/D^{ffw})}) / U^{B} \tag{5}$$

The variables in this equation are defined in Table 5.

For CWD, we report the weighted parameters used in Table 6.

4.2.1.4. Soil. Although the soil carbon pool is the second largest carbon storage pool in aggregate in the United States (Birdsey and Heath, 1995), Heath et al. (2002) note that little change in soil carbon occurs if forests are regenerated after harvest. This analysis assumed that all public timberland harvested returns to forest after harvest (i.e., no land is deforested), as is consistent with a mandate to manage and protect public forests. As a result, we assumed soil carbon on public timberland remains at a steady-state value (i.e., there is no change in soil carbon stock in the analysis) for the entire period of analysis.

4.2.2. Carbon disposition after harvest

At the time of harvest, some timber is removed from the site and used to make pulpwood-based products such as paper and sawlog-based products such as lumber, veneer, and

Table 2
Understory carbon variables and parameters

Symbol	Description	Source
C^{U} (Mg/acre)	Total understory carbon	See Eq. (3)
D^{L} (Mg/ha)	Live tree mass density (above and below ground)	See Eq. (2)
U^{B} (1 hectare (ha) = 2.471 acres)	Units conversion factor	–
R^{Uw} (%)	Weighted ratio of understory carbon to live tree carbon	U.S. EPA (2003) Table O-2 weighted by forestland area data from RPA (Miles, 2003) Tables 5 and 6

Table 3
Weighted ratio of understory to live tree carbon (%)

Region	Softwood	Hardwood	Planted pine	Natural pine	Oak pine	Douglas fir	Bottomland hardwood	Upland hardwood	Other softwoods
Northeast	2.6	2.2	NA	NA	NA	NA	NA	NA	NA
Lake states	2.1	2.4	NA	NA	NA	NA	NA	NA	NA
Corn Belt	2.1	2.4	NA	NA	NA	NA	NA	NA	NA
Southeast	NA	NA	6.8	6.8	4.4	NA	2.2	4.4	NA
South central	NA	NA	5.9	5.9	4.4	NA	2.2	3.7	NA
Rocky mountain	5.7	9.2	NA	NA	NA	NA	NA	NA	NA
Pacific northwest west	2.0	4.5	NA	NA	NA	2.0	NA	NA	3.2
Pacific northwest east	3.0	4.5	NA	NA	NA	NA	NA	NA	NA
Pacific southwest	5.0	2.9	NA	NA	NA	NA	NA	NA	NA

Source: Author calculations using U.S. EPA (2003) and forestland area data from RPA (Miles, 2003).

Table 4
Forest floor carbon variables and parameters: net accumulation

Symbol	Description	Source
C^{FFA} (Mg/acre)	Total forest floor carbon net accumulation	See Eq. (4)
Age (years)	Age of stand	–
A^{ffw}, B^{ffw}	Weighted forest floor carbon model coefficients	Smith and Heath (2002) Table 4 weighted by forestland area data from RPA (Miles, 2003) Tables 5 and 6
U^{B} (1 hectare (ha) = 2.471 acres)	Units conversion factor	–

Table 5
Forest floor carbon variables and parameters: decay of forest floor carbon mass existing prior to clear-cut

Symbol	Description	Source
C^{FFR} (Mg/acre)	Total forest floor carbon, residual	See Eq. (5)
Age (years)	Age of stand	–
C^{ffw}, D^{ffw}	Weighted forest floor carbon mass coefficients	Smith and Heath (2002) Table 4 weighted by forestland area data from RPA (Miles, 2003) Tables 5 and 6
U^{B} (1 hectare (ha) = 2.471 acres)	Units conversion factor	

panels. These products are then used to produce goods and services such as furniture, housing, and printed materials that are put into use for some period of time. The ultimate disposition over time of harvested carbon removed from the site depends on the products produced, their end uses, and the period of time elapsed since they were harvested and turned into product. Carbon in logging residue left on site is tracked separately in the forest floor carbon pool described above.

The wood product carbon accounting method used here is based on early versions of recent product accounting work (Smith et al., 2006). The modified approach uses calculation methods that are distinguished by the starting point of the harvest input (e.g., roundwood harvests or primary products produced). Because future NF and OPUB timberland inventories and timber harvest levels are expressed in terms of roundwood harvested rather than primary products produced, we used the roundwood harvests approach to track the fate of product carbon in the following pools:[5]

- products in use (sink),
- landfills (sink),
- energy (source or sink), and
- emissions (source).

Note, our primary analysis treats wood products allocated to the energy pool as a source of GHG emissions. However, we have also included calculations that treat energy uses as a sink

[5] In contrast, the FASOMGHG economic model (McCarl et al., 2005), which incorporates the carbon accounting methods described herein and applies them to estimate forest carbon sequestration at the national and regional levels in the United States, includes production technologies that convert roundwood harvests into primary products. Therefore, FASOMGHG's product accounting system uses the alternative starting point for product carbon calculations (i.e., quantities of primary products produced).

Table 6
Weighted ratio of coarse woody debris (CWD) to live tree carbon (%)

Region	Softwood	Hardwood	Planted pine	Natural pine	Oak pine	Douglas fir	Bottomland hardwood	Upland hardwood	Other softwoods
Northeast	12.3	11.2	NA	NA	NA	NA	NA	NA	NA
Lake states	14.1	10.8	NA	NA	NA	NA	NA	NA	NA
Corn belt	14.1	10.8	NA	NA	NA	NA	NA	NA	NA
Southeast	NA	NA	23.9	23.9	17.3	NA	21.8	24.3	NA
South central	NA	NA	18.6	18.6	17.3	NA	15.7	15	NA
Rocky mountain	12.6	26.7	NA	NA	NA	NA	NA	NA	NA
Pacific northwest west	11.9	3.9	NA	NA	NA	11.9	NA	NA	15.4
Pacific northwest east	14.8	3.9	NA	NA	NA	NA	NA	NA	NA
Pacific southwest	13.0	11.4	NA	NA	NA	NA	NA	NA	NA

Source: Author calculations using U.S. EPA (2003) and forestland area data from RPA (Miles, 2003).

for GHG emissions, assuming that biomass energy sources from the forest sector substitute for fossil fuel energy sources and serve as an offset for those emissions.

To calculate product carbon, we used cubic feet of roundwood harvested, divided into pulpwood or sawtimber products using yield tables, and converted volumes harvested into metric tonnes of carbon using factors reported in earlier versions of Smith et al. (2006). These factors include the average specific gravity, an upward adjustment to account for bark (1.18), and the carbon content of wood (0.5). Next, we allocated the carbon into the wood product pools (see Fig. 5) according to years since harvest and the disposition patterns. Examples of these patterns for the Southeast region are reported in Table 7.[6]

5. Results

Carbon sequestrations for U.S. public timberlands under the three scenarios (BAU, no harvest, and high harvest/pre-1989) are presented in Tables 8–10 respectively. Results are reported separately for all public timberlands and subcomponents (NF and OPUB) and for forest ecosystem carbon and wood product carbon. The projection time period is 10 decades, starting in 2010 and running through 2110. Tables 8–10 report detail for the first 5 decades, but summary totals are provided below for all 10 decades in the projection (Figs. 6–8). All carbon quantities are reported in average annual change in carbon stocks for that period, also known as annual flux.

Under the BAU scenario, public timberlands sequester, on average, 50 MMTC annually during the first 5 decades. This estimate ranges from 65 to 40 MMTC between 2010 and 2050, and decline after that (Fig. 6). The annual carbon flux occurs primarily in the ecosystem carbon pools of public forests prior to harvest (NF and OPUB), and the remainder is associated with postharvest wood and paper product sequestration. The ecosystem fluxes range between 82 and 92% of the total flux depending on decade and whether energy is treated as a credit. NFs account for over 60% of the annual carbon flux for all public timberlands. In 2030, for example, we estimated a total annual forest carbon flux of 33 million metric tonnes for NF timberlands compared with 15 million metric tonnes for OPUB timberlands (see Table 8). As shown in Fig. 7, over 85% of the NF forest carbon flux occurs in the West. The Rocky Mountain region accounts for 41%, followed by the Pacific northwest west (23%) and Pacific southwest (21%).

Table 8 and Fig. 6 display a positive but declining sequestration rate for public timberlands under BAU, with sequestration levels highest in the first decade and falling after that. The magnitudes of stock changes are consistent with the estimates for public forests in Smith and Heath (2004), although they do exhibit slightly different trends. These patterns reflect recent dynamics in the way public lands have been managed. Many of the current forest stands on public timberland today were regenerated after the heavier timber harvest periods of the 1960s–1980s. The net growth in such forest stands eventually slows down considerably as the stands age. Together with the recent slowdown in timber harvest levels, the age distribution of the public timberland stands will

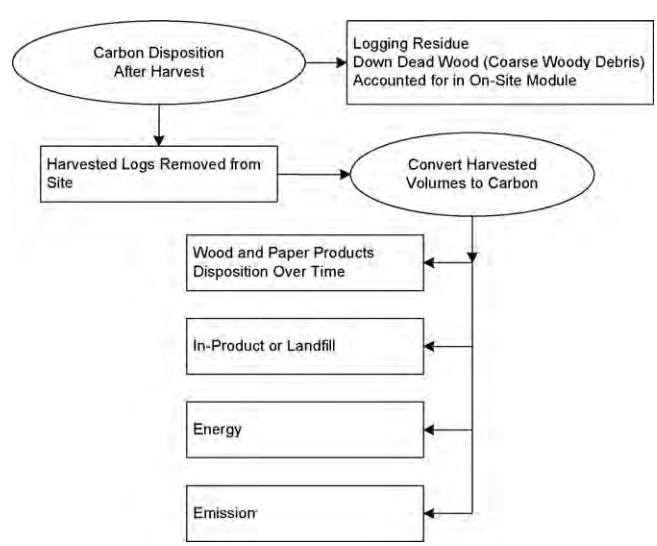

Fig. 5. Wood and paper product carbon disposition.

[6] Data for other regions are available upon request.

Table 7
Example of disposition patterns of harvested wood by region and harvest type, 100-Year period: southeast[a]

Region	Type	Product	Disposition	Years after harvest										
				0	10	20	30	40	50	60	70	80	90	100
Southeast	Softwood	Pulpwood	Products	0.30	0.07	0.05	0.04	0.03	0.03	0.03	0.03	0.03	0.03	0.02
Southeast	Softwood	Pulpwood	Landfills	0.00	0.16	0.16	0.16	0.10	0.14	0.14	0.13	0.12	0.11	0.11
Southeast	Softwood	Pulpwood	Energy	0.44	0.45	0.45	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46
Southeast	Softwood	Pulpwood	Emissions	0.26	0.32	0.34	0.35	0.41	0.37	0.38	0.39	0.40	0.41	0.41
Southeast	Softwood	Sawtimber	Products	0.47	0.28	0.24	0.21	0.18	0.17	0.15	0.14	0.13	0.13	0.12
Southeast	Softwood	Sawtimber	Landfills	0.00	0.13	0.16	0.17	0.18	0.19	0.19	0.19	0.18	0.18	0.18
Southeast	Softwood	Sawtimber	Energy	0.38	0.40	0.40	0.40	0.40	0.40	0.41	0.41	0.41	0.41	0.41
Southeast	Softwood	Sawtimber	Emissions	0.15	0.19	0.20	0.22	0.24	0.24	0.25	0.26	0.28	0.28	0.29
Southeast	Hardwood	Pulpwood	Products	0.30	0.07	0.05	0.04	0.04	0.03	0.03	0.03	0.03	0.03	0.03
Southeast	Hardwood	Pulpwood	Landfills	0.00	0.16	0.16	0.15	0.15	0.14	0.13	0.12	0.12	0.11	0.10
Southeast	Hardwood	Pulpwood	Energy	0.39	0.40	0.41	0.41	0.41	0.41	0.41	0.41	0.41	0.41	0.41
Southeast	Hardwood	Pulpwood	Emissions	0.31	0.37	0.38	0.40	0.40	0.42	0.43	0.44	0.44	0.45	0.46
Southeast	Hardwood	Sawtimber	Products	0.27	0.12	0.08	0.07	0.06	0.05	0.05	0.04	0.04	0.04	0.04
Southeast	Hardwood	Sawtimber	Landfills	0.00	0.11	0.13	0.14	0.14	0.14	0.13	0.13	0.13	0.13	0.12
Southeast	Hardwood	Sawtimber	Energy	0.42	0.43	0.43	0.44	0.44	0.44	0.44	0.44	0.44	0.44	0.44
Southeast	Hardwood	Sawtimber	Emissions	0.31	0.34	0.36	0.35	0.36	0.37	0.38	0.39	0.39	0.39	0.40

[a] These are proportions of the harvested stock allocated to each pool in the years following harvest. Column totals may not sum to one due to independent rounding.

Table 8
Annual stock changes: business-as-usual scenario (MM metric tonnes of carbon, MMTC, unless otherwise specified)

Decade	Forest carbon			Disposition of wood product carbon					Total in wood products		Total carbon stock change	
	Existing	Regenerated	Total in forest	Cumulative harvest volume since 2000 (MM cf)[a]	Decade harvest (MM cf)	Products	Land-fills	Energy	Without energy credit	With energy credit	Without energy credit	With energy credit
All public lands												
2010	55.1	4.5	59.7	28,009	14,695	2.5	2.5	2.5	5.0	7.5	64.6	67.1
2020	45.2	10.1	55.3	44,159	16,150	2.5	2.5	2.5	5.0	7.5	60.3	62.8
2030	34.0	13.5	47.5	60,477	16,318	2.2	2.2	2.2	4.3	6.5	51.9	54.0
2040	20.1	17.5	37.5	77,236	16,759	2.1	2.1	2.1	4.1	6.2	41.7	43.7
2050	15.7	20.4	36.2	94,239	17,003	1.9	1.9	1.9	3.8	5.6	39.9	41.8
National forests												
2010	50.0	1.2	51.2	9,424	5,394	1.2	1.2	1.2	2.5	3.7	53.7	54.9
2020	35.3	2.9	38.2	15,912	6,488	1.4	1.4	1.4	2.7	4.1	40.9	42.3
2030	28.6	4.1	32.8	22,862	6,950	1.3	1.3	1.3	2.6	3.9	35.4	36.6
2040	22.1	5.6	27.6	30,253	7,391	1.3	1.3	1.3	2.5	3.8	30.2	31.4
2050	17.6	7.1	24.7	37,888	7,635	1.2	1.2	1.2	2.4	3.5	27.0	28.2
Other public lands												
2010	5.2	3.3	8.4	18,585	9,301	1.2	1.2	1.2	2.5	3.7	10.9	12.2
2020	9.9	7.2	17.2	28,247	9,662	1.1	1.1	1.1	2.3	3.4	19.4	20.6
2030	5.4	9.4	14.8	37,615	9,368	0.9	0.9	0.9	1.7	2.6	16.5	17.4
2040	−2.0	11.9	9.9	46,983	9,368	0.8	0.8	0.8	1.6	2.4	11.5	12.3
2050	−1.9	13.4	11.5	56,351	9,368	0.7	0.7	0.7	1.4	2.1	12.9	13.6

[a] The cumulative harvest for periods includes all harvests for the previous decades plus the current decade.

shift to older stands in the coming decades and the growth rate will slow.[7]

A comparison of timber harvest scenarios illustrates the carbon storage trade-offs that policy makers face when considering alternative timber harvest levels from public forests. As shown in Fig. 8, moving from the baseline to a no-harvest regime leads to a significant increase in the carbon sequestered on public timberlands. Our estimates suggest an annual increase (above baseline) of 17–29 MMTC per year between 2010 and 2050, approximately a 40–50% increase in carbon storage depending on the decade. Interestingly, this is just below the 55–57% additional carbon sequestration reported by Harmon et al. (1990) when looking at the carbon sequestration potential of maintaining old-growth stands versus converting to

[7] One possible change to this growth projection is the effect of a changing climate. As shown in various studies at different spatial scales (Sohngen and Mendelsohn, 1998; Alig et al., 2002; Abt and Murray, 2001), future changes in climate can affect the growth and species distribution of forests in ways that are either favorable or unfavorable, depending on location.

Table 9
Annual stock changes: no-harvest scenario (MM metric tonnes of carbon, MMTC, unless otherwise specified)

Decade	Forest carbon			Disposition of wood product carbon					Total in wood products		Total carbon stock change	
	Existing	Regenerated	Total in forest	Cumulative harvest volume since 2000 (MM cf)	Decade harvest (MM cf)	Products	Land-fills	Energy	Without energy credit	With energy credit	Without energy credit	With energy credit
All public lands												
2010	93.3	0.0	93.3	0	0	0.0	0.0	0.0	0.0	0.0	93.3	93.3
2020	85.5	0.0	85.5	0	0	0.0	0.0	0.0	0.0	0.0	85.5	85.5
2030	76.1	0.0	76.1	0	0	0.0	0.0	0.0	0.0	0.0	76.1	76.1
2040	61.0	0.0	61.0	0	0	0.0	0.0	0.0	0.0	0.0	61.0	61.0
2050	57.3	0.0	57.3	0	0	0.0	0.0	0.0	0.0	0.0	57.3	57.3
National forests												
2010	64.3	0.0	64.3	0	0	0.0	0.0	0.0	0.0	0.0	64.3	64.3
2020	52.2	0.0	52.2	0	0	0.0	0.0	0.0	0.0	0.0	52.2	52.2
2030	46.8	0.0	46.8	0	0	0.0	0.0	0.0	0.0	0.0	46.8	46.8
2040	41.1	0.0	41.1	0	0	0.0	0.0	0.0	0.0	0.0	41.1	41.1
2050	36.9	0.0	36.9	0	0	0.0	0.0	0.0	0.0	0.0	36.9	36.9
Other public lands												
2010	29.0	0.0	29.0	0	0	0.0	0.0	0.0	0.0	0.0	29.0	29.0
2020	33.3	0.0	33.3	0	0	0.0	0.0	0.0	0.0	0.0	33.3	33.3
2030	29.4	0.0	29.4	0	0	0.0	0.0	0.0	0.0	0.0	29.4	29.4
2040	19.9	0.0	19.9	0	0	0.0	0.0	0.0	0.0	0.0	19.9	19.9
2050	20.4	0.0	20.4	0	0	0.0	0.0	0.0	0.0	0.0	20.4	20.4

sustained harvesting of stands under rotational forestry. Sequestration under the no-harvest scenario in the first 5 decades would offset between 1 and 2% of total CO_2 emissions in the United States at current levels and is equivalent to removing the emissions of about 13–24 million cars per year. Most of the additional sequestration occurs within NFs (see Table 9). This rate of additional carbon sequestration declines over time (Fig. 8).

In contrast with the no-harvest scenario, increasing the baseline harvest levels to pre-1989 levels leads to a significant decrease in the carbon sequestered in public forests. Our estimates suggest losses ranging from 27 to 35 MMTC per year

Table 10
Annual stock changes: pre-1989 harvest levels (MM metric tonnes of carbon, MMTC, unless otherwise specified)

Decade	Forest carbon			Disposition of wood product carbon					Total in wood products		Total carbon stock change	
	Existing	Regenerated	Total in forest	Cumulative harvest volume since 2000 (MM cf)[a]	Decade harvest (MM cf)	Products	Land-fills	Energy	Without energy credit	With energy credit	Without energy credit	With energy credit
All public lands												
2010	5.6	9.1	14.8	69,470	35,630	7.5	7.5	7.5	14.9	22.4	29.7	37.1
2020	−3.1	17.7	14.6	105,975	36,504	6.6	6.6	6.6	13.2	19.8	27.8	34.3
2030	−17.7	25.5	7.8	143,301	37,327	6.0	6.0	6.0	12.0	18.0	19.9	25.9
2040	−29.9	32.2	2.3	181,085	37,784	5.3	5.3	5.3	10.5	15.8	12.9	18.1
2050	−33.1	36.1	3.0	220,015	38,929	5.0	5.0	5.0	10.0	15.0	13.1	18.1
National forests												
2010	2.7	5.8	8.6	47,200	24,220	5.6	5.6	5.6	11.1	16.7	19.7	25.3
2020	−11.0	10.1	−0.8	72,040	24,840	5.0	5.0	5.0	10.1	15.1	9.2	14.3
2030	−19.2	15.4	−3.8	97,562	25,522	4.6	4.6	4.6	9.2	13.8	5.4	10.0
2040	−24.3	19.0	−5.3	123,472	25,910	4.1	4.1	4.1	8.1	12.2	2.9	6.9
2050	−27.3	21.3	−6.0	150,527	27,055	4.0	4.0	4.0	8.0	12.0	1.9	5.9
Other public lands												
2010	2.9	3.3	6.2	22,270	11,410	1.9	1.9	1.9	3.8	5.7	10.0	11.9
2020	7.8	7.6	15.4	33,935	11,664	1.6	1.6	1.6	3.1	4.7	18.5	20.1
2030	1.6	10.1	11.7	45,739	11,804	1.4	1.4	1.4	2.8	4.2	14.5	15.9
2040	−5.6	13.2	7.6	57,613	11,874	1.2	1.2	1.2	2.4	3.6	10.0	11.2
2050	−5.7	14.8	9.1	69,487	11,874	1.0	1.0	1.0	2.0	3.0	11.1	12.1

[a] The cumulative harvest for periods includes all harvest for the previous decades plus the current decade.

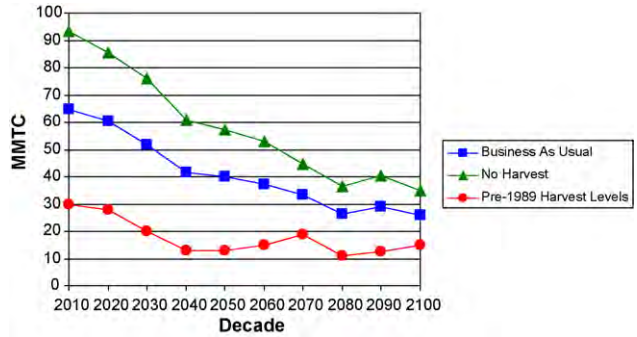

Fig. 6. Annual carbon sequestration in all public lands by scenario.

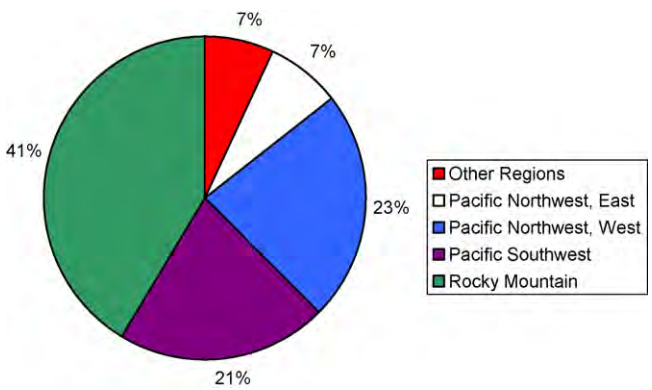

Fig. 7. Distribution of annual NF carbon stock changes by region: BAU scenario (2030).

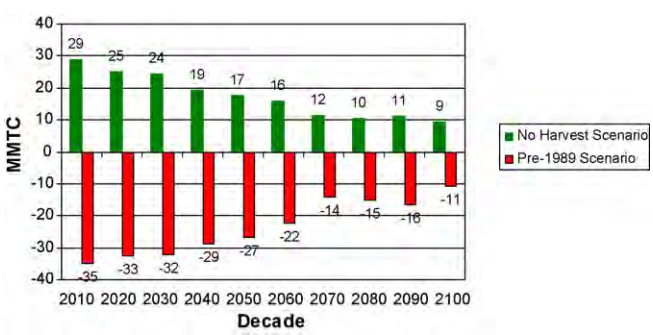

Fig. 8. Comparison of annual carbon stock changes with business-as-usual scenario.

between 2010 and 2050, approximately a 50–80% decline in carbon storage from BAU depending on the decade. This tempers some in the last 5 decades as the regrowth from harvested stands contribute more strongly to the sequestration rate. The vast majority of timber harvests come from NF timberlands: an average of 26 billion ft^3 are harvested per decade in NFs compared with 12 billion ft^3 in OPUB timberlands. As a result, returning to these high timber harvest levels would make NFs a net source of emissions between 2020 and 2050. Although OPUB timberlands continue to be carbon sinks, the annual carbon stock changes in forests are substantially lower than in the BAU case. Carbon losses associated with the more intense harvesting scenario are reduced to some degree through carbon storage in wood and paper products. Our estimates suggest that wood and paper products sequester between 10 and 15 MMTC per year. If we treat energy uses as a sink for GHG emissions, assuming energy use substitutes for other energy sources and serves as an offset for those emissions, our wood and paper product sequestration estimates increase another 5–7 MMTC per year, rising between 15 and 22 MMTC per year total, depending on the decade (see Table 10).

It is instructive to view these results in terms of the potential monetary value of sequestered carbon in the different scenarios. Payments for carbon sequestration can be viewed as part of a potential broader system to offset emissions of CO_2 and other GHGs. CO_2 emission credits are currently being traded for between US$ 15 and 30 per metric tonne (Mg) of CO_2 equivalent on the EU's Emissions Trading System (EU ETS). Translating to units of carbon, this is about US$ 55–110 per Mg C. Although forest carbon sequestration is not currently traded in the EU ETS, this range provides some sense, perhaps an upper range, of its monetary potential if sequestered carbon on public timberlands were included in a trading mechanism.[8] At this price range, the annual value of carbon sequestered on public timberlands under BAU ranges from US$ 2.2 to 7.1 billion, depending on the decade. However, GHG compensation schemes that include forest carbon offsets might not consider BAU sequestration to be creditable, focusing instead on carbon that is additional to BAU (Murray et al., 2007). We can estimate that the additional amount of carbon sequestered under the no-harvest scenario would be between US$ 0.9 and 3.2 billion per year, and foregone carbon revenue would be between US$ 1.5 and 3.9 billion per year under the pre-1989 harvest scenario. By contrast, timber harvest revenues on public lands in 2005 were approximately US$ 800–900 million (Adams, 2006). One should note that these revenue comparisons do not capture all relevant aspects of welfare. A more complete comparison would capture effects on consumer and producer surplus and thereby the net benefits to society of each harvesting plan. That is beyond the scope of this study. The revenue comparisons here, however, do indicate relative trade-offs between timber and carbon revenue that might be expected under different management regimes.

6. Conclusions

For decades, public timberlands have been managed for multiple uses and ecosystem services including timber, range, wildlife habitat, watershed protection, recreation, and visual amenities. More attention in recent years has been placed on establishing and maintaining forest carbon sinks to help regulate atmospheric GHGs and climate, but little empirical work at a national scale has estimated the biophysical potential

[8] Rather than evaluating its revenue potential in a greenhouse gas trading market, another perspective is the social cost of carbon remaining in the atmosphere. This measures the value of climate change damages caused by carbon accumulation in the atmosphere and thus the marginal benefit of carbon removed from the atmosphere. The most recent IPCC assessment report provides a range of values for social cost of carbon at about US$ 43 per tonne C or about US$ 12 per tonne CO_2 (IPCC, 2007).

of modifications in public timberland management to sequester more carbon. This paper addresses that gap by combining data on public timber inventories, timber harvest scenarios, and carbon accounting to quantify the accumulation of carbon on public timberlands and in wood product stocks from harvested timber under three scenarios: BAU, no harvest, and high harvest (equivalent to the 1980s). Findings suggest that under BAU, public timberlands will continue to sequester carbon through the next century, though at a diminishing rate. The BAU accumulation of carbon occurs because of the age class and growth dynamics of the current inventory of public timberland, which has experienced timber harvest levels in the recent past that are substantially lower than the preceding decades. These changes in timber harvest were done for a wide variety of ecological and economic reasons, but a by-product of these efforts was an increase in public timberlands' positive contribution to global climate regulation.

Variations in BAU in either direction – elimination of harvests altogether or a substantial ramp-up in public harvests to levels of 20 years ago – could substantially alter the annual carbon balance of public timberlands, at least 50% in either direction. Each action would have opportunity costs in terms of the economic and ecological value of the corresponding changes in market and nonmarket ecosystem services, but a market for sequestered carbon could alter the balance considerably with public sequestration worth potentially billions of dollars in value per year. Although markets for carbon are in their nascent stages and the level of future carbon prices are highly uncertain, public decision makers should nonetheless consider the economic value of carbon when developing national, regional, and forest-level targets for timber harvests and other public timberland outputs.

This study provides a rough estimate of the potential from a relatively few, though wide-ranging, timber harvest policy alternatives. Forest and carbon management, however, is much more subtle than simply determining how much to harvest. Many forest management decisions from the time of stand establishment through mid-rotation treatments to the timber harvest decision could be affected with carbon sequestration as a more accentuated objective. Of particular interest is the link between carbon management, fire management, and biofuel production, each of which can have a profound impact on the carbon balance, ecological integrity, and economic value of the forest. One research need is a better understanding of how such linkages are affected by the stochastic nature of certain disturbances such as fires. Future research should carefully evaluate these trade-offs and opportunities at regional, landscape, and individual forest scales.

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Testimony of Mike Dombeck
House Natural Resources Committee Oversight
Hearing on the Alaska Roadless Rule
November 13, 2019

Thank you for inviting me to testify on the Alaska Roadless Rule. I'm very familiar with this Committee's work, having testified many times as acting-Director of the Bureau of Land Management and as Chief of the Forest Service, including several times on the very topic of this hearing—the future of Roadless Areas on the National Forests. I greatly appreciate this Committee's important role in the oversight and management of our nation's precious natural resources and public lands. I am very grateful to have spent a career in public service and have witnessed a very wide variety of public land management debates.

I appreciate this Committee's interest and oversight in the need for proper multiple-use management on our public forests. Rolling back the protection provided by the Forest Service 2001 Roadless Rule in Alaska or anywhere else is bad public policy. Opening up backcountry areas to road construction will cost taxpayers far more than the economic benefits will yield. It will compromise long-term watershed health, and it will take us back to the unproductive controversies of the past.

I believe it important to review a bit of history to put the current proposal into a broader historical context. I have included the preamble to the 2001 Roadless Rule as an appendix. The preamble provides more detail regarding the rationale for the Rule and associated resource values.

No other nation in the World has this bounty of public land belonging to all citizens. Some of our public land was acquired through conquest or treaty; other land including Alaska was purchased, paid for by taxpayers from every state.

Translating multiple-use land management on the ground is no easy task. Every constituency—forest products, grazing, mining, recreation, wilderness, and so on—pushes to maximize its interest. Couple this with constantly changing economic needs and social values, and the realities and the challenge get even messier. In the last half of the last century few issues elicited as much interest and controversary as the disposition and management of this nation's last remaining wild and unfragmented land, including 58 million acres of National Forest roadless lands.

As this nation grew, by the early 1900's the great forests of the eastern U.S. had been clear cut, but millions of acres of large old growth forest remained in the West. Prior to World War II the timber industry did not want a lot of National Forest timber on the market because it would depress prices of private land timber. However, World War II changed all that with the tremendous demand for wood for the war effort followed by the post-war housing boom. With most of the

privately-owned old growth timber harvested, this left the western national forests with a bounty of big trees---old growth forest. The U.S. Forest Service at the request of Congress did its part by ramping up harvest to meet the demand.

Roads are needed for timber harvest. In the 46 years between 1950 and 1996, the National Forest road system more than doubled, from 165,000 miles to 373,000 miles. The construction and maintenance of this extensive road system was financed in large part by timber sales.

The easily accessible timber was harvested first and as years went by the Agency had to put up timber sales in increasingly remote forests. With the easily accessible high value timber already cut, the agency was forced to move into more mountainous, rugged and remote lands, locations with steep slopes and fragile soils. To maintain its high level of timber harvest, the agency had to construct roads into these increasingly remote areas. Many of these timber sales lost money and the Agency was roundly criticized for “below cost timber sales,” i.e., timber sales that cost taxpayers more money to design, access and sell than they generate in revenue. The impact on Agency budgets was profound. By 1979, only 38% of existing roads were maintained to the safety and environmental standards to which they were designed. Even with the record high level of timber production in 1989, only 47% of forest roads were maintained to standard. The timber harvest levels were unsustainable as was the ability to maintain the extensive road system.

Yet some forest roads are important to local communities. They can provide access for development, tourism and recreation and many other needs. However, when forest roads are not maintained, particularly in rugged mountainous terrain, they erode and slide into streams, muddying drinking water and ruining fish habitat. Roads fragment wildlife habitat, negatively impact rare species, and serve as vectors for invasive species. Additionally, public support for protecting remote wild lands was increasing.

When I became Forest Service Chief in 1997, everyone was upset about roads and roadless areas. County commissioners complained about the poor condition of roads on the National Forests. Wilderness advocates and many hunters and anglers strongly opposed punching roads into remote places, wanting the remaining large tracts of wild places kept wild. The controversary was at a fever pitch. There was constant controversy and litigation over timber sales and road construction into wild remote places on the National Forests. Many forest management decisions were made in the courtroom.

By 1998, the Forest Service had a 380,000-mile road system with an $8.5-billion maintenance backlog. Many inside and outside the Forest Service and Congress rightly began to question why the agency would build more roads into relatively pristine areas when it could not take care of the roads already in place.

Members of Congress and this Committee were very concerned as well. As a surrogate to protect roadless areas, in 1997 the House of Representatives came within a single vote of cutting the Forest Service's road budget by 80 percent. This would have crippled the agency and jeopardized public access and use of the National Forests.

When you find yourself in a hole, put the shovel down and stop digging. Why in the world would the Forest Service continue to build roads in remote forests such as the Tongass National Forest when they couldn't take care of the roads they already had *for timber sales that lost the taxpayers money!*

As a result of broad consultation and deliberation, the Forest Service developed a protective strategy for roadless areas over a three-year period that involved more than 600 local public meetings and a record-breaking 1.6 million public comments. More than 90% of the public comments favored protection of the entire 58 million acres of inventoried roadless areas on National Forest lands. The result was stronger protection of roadless areas.

The Roadless Rule, finalized in 2001, ended the most damaging and expensive threats to those roadless areas. But carefully designed exceptions allow for some new road construction on a case-by-case basis, where that is in the public interest. Roads are permitted for firefighting, forest health, energy development and access to private inholdings, for instance, as well as public thoroughfares between communities. For example, on the Tongass National Forest alone, all 58 requests for entry into roadless areas since the Rule has been in place have been granted. These include roads for mining projects, hydropower and intertie projects, a geothermal lease, a road realignment, road reconstruction, and U.S. Coast Guard and Alaska Army National Guard projects, among others. Most projects are approved within a month, and that time is expected to become even shorter since authority to approve requests was delegated to the regional foresters in October 2018.

The Roadless Rule was controversial on many fronts and some controversy remains so today, as is evidenced by the need for this hearing. However, the Rule has gone a long way to reducing conflict over national forests, focusing Forest Service management on what the land needs and the American people want, and better prioritizing and utilizing budget dollars. It has:

- Saved taxpayers millions of dollars from below cost timber sales and by not adding to the road maintenance financial burden.
- Saved millions of dollars in downstream water filtration costs.
- Reduced litigation over forest management.
- Calmed the waters over timber harvest, which has restored at least some level of trust and enabled collaboration among opposing interests.
- Allowed for more effort and restoration of roaded and previously harvested National Forest lands.

- Provided more certainty to both the timber industry and other forest interests.
- Allowed for more resources and work to be directed toward reducing risk and severity of wildfires, focusing on areas naturally prone to wildfire, especially near human communities.

During my tenure as 14th Chief of the Forest Service, the management of the Tongass was among the most politically charged and controversial issues I dealt with. A key question that should be considered is what does this nation want the Alaska National Forests to look like in the coming decades? What resource values should be considered?

The Tongass is not only the largest of our National Forests, but it is also one of the most valuable and important. It has unique and irreplaceable cultural value to vibrant local Alaska Native communities. It produces more salmon than all other National Forests combined, with 50 million salmon commercially harvested annually at a value of $60 million. The region's beautiful scenery, abundant fish and wildlife, and expansive tracts of undeveloped lands attract visitors and recreationists in increasingly large numbers, with current estimates projecting 2 million out-of-state visitors in the coming years. Seafood and tourism now support 26% of local jobs and $2 billion in economic contribution.

The Tongass is also one of the last remaining intact temperate rainforests in the world and a globally significant storehouse of carbon. According to Forest Service estimates, Tongass trees contain 650 million tons of carbon, which is equivalent to 2.4 billion tons of CO_2. This is nearly half (45%) of the total carbon emissions for the entire U.S. in 2017.

Recent analysis by Conservation Science Partners, a nonprofit science think-tank using advanced satellite data and geospatial analyses, shows that the continental U.S. is losing natural lands at the rate of two football fields per minute, due to factors that include road building. This along with increasing urbanization of the U.S. makes the remaining wild places more valuable each year.

The key recommendation I have for this Committee, the Congress and the Executive Branch is to focus on how to maintain the long-term health, diversity, and productivity of the land. The challenge as defined by Gifford Pinchot is to manage for the "greatest good for the greatest number for the longest time." Watershed protection and restoration were the basic concerns that led to the establishment of the National Forests. The critical role forests play in the carbon cycle and moderating climate change is perhaps the most recent value we must take seriously. The intense forest fires in parts of the West and other extreme weather patterns are reminders that maintaining and protecting forests and their sound management is of the utmost importance.

The Forest Service should be protecting the best remaining undeveloped lands and restoring the rest. It should invest in new scientific research to improve our understanding of how to reduce risk from wildfire and climate change. The Forest Service should focus its efforts where it is most needed: on addressing climate change, promoting wildfire resiliency, restoring damaged habitat for important and rare fish and wildlife species, and creating new opportunity for recreation, hunting and fishing.

For the Tongass, where two-thirds of the high-volume timber stands have already been logged at an immense social and economic cost, the Forest Service should focus on addressing the $100 million backlog of watershed restoration needs and the $68 million road maintenance backlog. And above all else, the Forest Service should stop the bleeding by keeping the Roadless Rule in place.

The 2001 Roadless Rule in my view is conservative public policy. It maintains the status quo, keeps options open for the future, and saves taxpayer money. The time and money spent attempting to roll back roadless area protections on the Tongass or any of the National Forests is a big step backwards to the era of gridlock and costly litigation.

Thank you for the opportunity to share my views with the Committee.

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Appendix: Preamble to 2001 Roadless Area Conservation Rule

Roadless Area Values and Characteristics

Inventoried roadless areas considered in this rule constitute roughly one-third of all National Forest System lands, or approximately 58.5 million acres. Although the inventoried roadless areas comprise only 2% of the land base in the continental United States, they are found within 661 of the over 2,000 major watersheds in the nation (FEIS Vol. 1, 3-50) and provide many social and ecological benefits.

As urban areas grow, undeveloped private lands continue to be converted to urban and developed areas, and rural infrastructure (such as roads, airports, and railways). An average of 3.2 million acres per year of forest, wetland, farmland, and open space were converted to more urban uses between 1992 and 1997. In comparison, 1.4 million acres per year were developed between 1982 and 1992. The rate of land

development and urbanization between 1992 and 1997 was more than twice that of the previous decade, while the population growth rate remained fairly constant (FEIS Vol. 1, 3-12). In an increasingly developed landscape, large unfragmented tracts of land become more important. For example, from 1978 to 1994, the proportion of private forest ownerships of less than 50 acres nearly doubled (Birch, T.W. 1996. Private forest-land owners of the United States, 1994. Resource Bulletin NE-134. Radnor, PA: USDA Forest Service, Northeastern Experiment Station. 183 p). Subdivision and other diminishment of tract size of these lands can discourage long-term stewardship and conservation.

Inventoried roadless areas provide clean drinking water and function as biological strongholds for populations of threatened and endangered species. They provide large, relatively undisturbed landscapes that are important to biological diversity and the long-term survival of many at risk species. Inventoried roadless areas provide opportunities for dispersed outdoor recreation, opportunities that diminish as open space and natural settings are developed elsewhere. They also serve as bulwarks against the spread of non-native invasive plant species and provide reference areas for study and research (FEIS Vol. 1, 1-1 to 1-4).

The following values or features often characterize inventoried roadless areas (FEIS Vol. 1, 3-3 to 3-7):

High quality or undisturbed soil, water, and air. These three key resources are the foundation upon which other resource values and outputs depend. Healthy watersheds catch, store, and safely release water over time, protecting downstream communities from flooding; providing clean water for domestic, agricultural, and industrial uses; helping maintain abundant and healthy fish and wildlife populations; and are the basis for many forms of outdoor recreation.

Sources of public drinking water. National Forest System lands contain watersheds that are important sources of public drinking water. Roadless areas within the National Forest System contain all or portions of 354 municipal watersheds contributing drinking water to millions of citizens. Maintaining these areas in a relatively undisturbed condition saves downstream communities millions of dollars in water filtration costs. Careful management of these watersheds is crucial in maintaining the flow and affordability of clean water to a growing population.

Diversity of plant and animal communities. Roadless areas are more likely than roaded areas to support greater ecosystem health, including the diversity of native and desired nonnative plant and animal communities due to the absence of disturbances caused by roads and accompanying activities. Inventoried roadless areas also conserve native biodiversity by serving as a bulwark against the spread of nonnative invasive species.

Habitat for threatened, endangered, proposed, candidate, and sensitive species and for those species dependent on large, undisturbed areas of land. Roadless areas function as biological strongholds and refuges for many species. Of the nation's species currently listed as threatened, endangered, or proposed for listing under the Endangered Species Act, approximately 25% of animal species and 13% of plant species are likely to have habitat within inventoried roadless areas on National Forest System lands. Roadless areas support a diversity of aquatic habitats and communities, providing or affecting habitat for more than 280 threatened, endangered, proposed, and sensitive species. More than 65% of all Forest Service sensitive species are directly or indirectly affected by inventoried roadless areas. This percentage is

composed of birds (82%), amphibians (84%), mammals (81%), plants (72%), fish (56%), reptiles (49%), and invertebrates (36%).

Primitive, Semi-Primitive Non-Motorized, and Semi-Primitive Motorized classes of dispersed recreation. Roadless areas often provide outstanding dispersed recreation opportunities such as hiking, camping, picnicking, wildlife viewing, hunting, fishing, cross-country skiing, and canoeing. While they may have many Wilderness-like attributes, unlike Wilderness the use of mountain bikes, and other mechanized means of travel is often allowed. These areas can also take pressure off heavily used wilderness areas by providing solitude and quiet, and dispersed recreation opportunities.

Reference landscapes. The body of knowledge about the effects of management activities over long periods of time and on large landscapes is very limited. Reference landscapes of relatively undisturbed areas serve as a barometer to measure the effects of development on other parts of the landscape.

Natural appearing landscapes with high scenic quality. High quality scenery, especially scenery with natural-appearing landscapes, is a primary reason that people choose to recreate. In addition, quality scenery contributes directly to real estate values in nearby communities and residential areas.

Traditional cultural properties and sacred sites. Traditional cultural properties are places, sites, structures, art, or objects that have played an important role in the cultural history of a group. Sacred sites are places that have special religious significance to a group. Traditional cultural properties and sacred sites may be eligible for protection under the National Historic Preservation Act. However, many of them have not yet been inventoried, especially those that occur in inventoried roadless areas.

Other locally identified unique characteristics. Inventoried roadless areas may offer other locally identified unique characteristics and values. Examples include uncommon geological formations, which are valued for their scientific and scenic qualities, or unique wetland complexes. Unique social, cultural, or historical characteristics may also depend on the roadless character of the landscape. Examples include ceremonial sites, places for local events, areas prized for collection of non-timber forest products, or exceptional hunting and fishing opportunities.

Fiscal Considerations

The Department is also concerned about building new roads in inventoried roadless areas, when there presently exists a backlog of about $8.4 billion in deferred maintenance and reconstruction on the more than 386,000 miles of roads in the Forest Transportation System. The agency

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estimates that at least 60,000 miles of additional unauthorized roads exist across National Forest System lands.

The agency receives less than 20% of the funds needed annually to maintain the existing road infrastructure. As funding needs remain unmet, the cost of fixing deteriorating roads increases exponentially every year. Failure to maintain existing roads can also lead to erosion and water quality degradation and other environmental problems and potential threats to human safety. It makes little fiscal or

environmental sense to build additional roads in inventoried roadless areas that have irretrievable values at risk when the agency is struggling to maintain its existing extensive road system (FEIS Vol. 1, 1-5 and 3-22). The National Forest System was founded more than 100 years ago to protect drinking water supplies and furnish a sustainable supply of timber. Neither objective is fully achievable given the present condition of the existing road system. The risks inherent in building new roads in presently roadless areas threaten environmental, social, and economic values.

Development activities in inventoried roadless areas often cost more to plan and implement than on other National Forest System lands. Some planned timber sales in inventoried roadless areas are likely to cost more to prepare and sell than they realize in revenues received. Because of the level of public controversy and analytical complexity, projects in roadless areas often require development of costly environmental impact statements for most resource development activities, including timber harvesting, in inventoried roadless areas. In some cases, road construction costs are higher due to rugged terrain or sensitive ecological factors. Many development projects in inventoried roadless areas are appealed or litigated. These factors contribute to generally higher costs for the agency to plan and implement development activities in inventoried roadless areas.

National Direction vs. Local Decisionmaking

At the national level, Forest Service officials have the responsibility to consider the ``whole picture'' regarding the management of the National Forest System, including inventoried roadless areas. Local land management planning efforts may not always recognize the national significance of inventoried roadless areas and the values they represent in an increasingly developed landscape. If management decisions for these areas were made on a case-by-case basis at a forest or regional level, inventoried roadless areas and their ecological characteristics and social values could be incrementally reduced through road construction and certain forms of timber harvest. Added together, the nation-wide results of these reductions could be a substantial loss of quality and quantity of roadless area values and characteristics over time.

In 1972, the Forest Service initiated a review of National Forest System roadless areas generally larger than 5,000 acres to determine their suitability for inclusion in the National Wilderness Preservation System. A second review process completed in 1979, known as Roadless Area Review and Evaluation II (RARE II), resulted in another nationwide inventory of roadless areas. In the more than 20 years since the completion of RARE II, Congress has designated some of these areas as Wilderness. Additional reviews have been conducted through the land management planning process and other large-scale assessments. The 58.5 million acres of inventoried roadless areas used as the basis for this analysis were identified from the most recent analysis for each national forest or grassland, including RARE II, land and resource management planning, or other large-scale assessments such as the Southern Appalachian Assessment.

Of the 58.5 million acres of inventoried roadless areas considered in the FEIS, approximately 34.3 million acres have prescriptions that allow road construction and reconstruction. The remaining 24.2 million acres are currently allocated to management prescriptions that prohibit road construction; however, protections in these existing plans may

change after future forest plan amendments or revisions.

Over the past 20 years, roads have been constructed in an estimated 2.8 million of those 34.3 million acres of inventoried roadless areas. The agency anticipates that the trend of building roads in inventoried roadless areas will gradually decrease in the future even without this rule due to economic and ecological factors already discussed, changes in agency policy, increasing controversy and litigation, and potential listings under the Endangered Species Act. While these anticipated changes may reduce some of the impact to inventoried roadless areas, they would not eliminate the future threat to roadless area values (FEIS Vol. 1, 1-14 to 1-15).

On many national forests and grasslands, roadless area management has been a major point of conflict in land management planning. The controversy continues today, particularly on most proposals to harvest timber, build roads, or otherwise develop inventoried roadless areas. The large number of appeals and lawsuits, and the extensive amount of congressional debate over the last 20 years, illustrates the need for national direction and resolution and the importance many Americans attach to the remaining inventoried roadless areas on National Forest System lands (FEIS Vol. 1, 1-16). These disputes are costly in terms of both fiscal resources and agency relationships with communities of place and communities of interest. Based on these factors, the agency decided that the best means to reduce this conflict is through a national level rule.

Importance of Watershed Protection

Watershed protection is one of the primary reasons Congress reserved or authorized the purchase of National Forest System lands. Watershed health and restoration is also one of four emphasis areas in the agency's Natural Resource Agenda. Protecting the remaining healthy components of a watershed provides multiple benefits and a strong base to anchor future restoration in unprotected portions of these watersheds. Rivers, streams, lakes, and wetlands within a watershed are the circulatory system of ecosystems, and water is the vital fluid for inhabitants of these ecosystems, including people (FEIS Vol. 1, 1-1).

Inventoried roadless areas comprise a small fraction of the national landscape, representing less than 2% of the land base of the continental United States. They are, however, disproportionately important to the small percentage of the land base they occupy. Overall, National Forest System watersheds provide about 14% of the total water flow of the nation, about 33% of water in the West (FEIS Vol. 1, 3-46). Of the watersheds on National Forest System land, 661 contain inventoried roadless areas and 354 of those watersheds serve as source areas of drinking water used by millions of people across the nation. Therefore, the health of these watersheds is important to people's health throughout the United States.

Roads have long been recognized as one of the primary human-caused sources of soil and water disturbances in forested environments (FEIS Vol. 1, 3-44). For example, while landslides are a natural process, extensive research and other investigations in the West have closely associated land management activities, particularly roading and timber harvest, with accelerated incidence of landslides by several orders of magnitude (FEIS Vol.

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1, 3-58). A joint study by the Forest Service and Bureau of Land Management in Oregon and Washington found that of 1,290 landslides reviewed in 41 sub-watersheds, 52% were related to roads, 31% to timber harvest, and 17% occurred in undisturbed forest (FEIS Vol. 1, 3-59). Another evaluation of landslides initiated by the Siuslaw National Forest found that roads were the source of 41% of landslides, harvest units less than 20 years old were the source of 36%, while natural forest processes accounted for the remaining 23%. Without the disturbance caused by roads and associated activities, stream channels are more likely to function naturally (FEIS Vol. 1, 3-54). Current road construction and timber harvest practices reduce the potential for damage associated with the use of earlier and less sophisticated techniques. However, even with today's improved design standards for road construction and timber harvest, these activities can still result in adverse effects to watersheds. These effects include pollution, changes to water temperatures and nutrient cycles, and increased sediment from storm or runoff events that exceed road design standards (FEIS Vol. 1, 3-45 to 3-50).

Improving Ecosystem Health

Inventoried roadless areas provide large, relatively undisturbed blocks of important habitat for a variety of terrestrial and aquatic wildlife and plants, including hundreds of threatened, endangered, and sensitive species. In addition to their ecological contributions to healthy watersheds, many inventoried roadless areas function as biological strongholds and refuges for a number of species and play a key role in maintaining native plant and animal communities and biological diversity (FEIS Vol. 1, 3-123 to 3-124). For example, about 60% of unroaded or very low road density sub watersheds within the Interior Columbia Basin Ecosystem Management Project (ICBEMP) assessment area are aquatic strongholds for salmonid populations (FEIS Vol. 1, 3-161). Inventoried roadless areas are key to recovery of salmon and steelhead stocks in decline, providing habitat to protect species until longer-term solutions can be developed for migration, passage, hatchery, and harvest problems associated with the decline of anadromous fish.

Species richness and native biodiversity are more likely to be effectively conserved in larger undisturbed landscapes, such as inventoried roadless areas (FEIS Vol. 1, 3-142). For example, inventoried roadless areas cover approximately 21% of the centers of biodiversity for animals and 10% for plants identified in ICBEMP (FEIS Vol. 1, 3-144 and 3-173). Inventoried roadless areas also provide reference landscapes that managers can use to gauge the health and condition of other land areas.

Road construction, reconstruction, and timber harvesting activities can result in fragmentation of ecosystems, the introduction of non-native invasive species, and other adverse consequences to the health and integrity of inventoried roadless areas (FEIS Vol. 1, 3-128 to 3-136). As human-caused fragmentation increases, the amount of core wildlife habitat decreases. This fragmentation results in decreased connectivity of wildlife habitat and wildlife movement, isolating some species and increasing the risk of local extirpations or extinctions (FEIS Vol. 1, 3-133). The value of inventoried roadless areas as habitat for threatened, endangered, and sensitive species and as biological strongholds can also be diminished due to these activities. For example, 220 species that are listed as threatened, endangered, or

proposed for listing under the Endangered Species Act and 1,930 agency-identified sensitive species rely on habitat within inventoried roadless areas (FEIS Vol. 1, 3-180). The Department of Agriculture believes that the risks associated with certain development activities in inventoried roadless areas should be minimized and that these areas should be conserved for present and future generations.

Need for Action

Promulgating this rule is necessary to protect the social and ecological values and characteristics of inventoried roadless areas from road construction and reconstruction and certain timber harvesting activities. Without immediate action, these development activities may adversely affect watershed values and ecosystem health in the short and long term, expand the road maintenance backlog which would increase the financial burden associated with road maintenance, and perpetuate public controversy and debate over the management of these areas. The new planning rules provide for review of other activities and allow for additional protection of roadless areas, if warranted. Adoption of this final rule ensures that inventoried roadless areas will be managed in a manner that sustains their values now and for future generations.

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GOSHAWKS IN CANADA: POPULATION RESPONSES TO HARVESTING AND THE APPROPRIATENESS OF USING STANDARD BIRD MONITORING TECHNIQUES TO ASSESS THEIR STATUS

Frank I. Doyle

Abstract. In this paper, I use the results from current research and from established bird monitoring techniques to highlight the inability of current techniques to establish the status of Northern Goshawk (*Accipiter gentilis atricapillus* and *A. g. laingi*) across Canada. At a national-scale monitoring of goshawks relies upon opportunistic goshawk sightings made during Breeding Bird Surveys (BBS), Christmas Bird Counts (CBC), or during migration counts. These sources indicate that the population trend is either stable (BBS and CBC), or possibly declining (migration counts over last 20–30 yr). However, recent goshawk population studies in western Canada have shown that individual subpopulations respond differently to harvesting of mature forest, with some showing a negative impact, while others appear to be thriving at the same rate of harvest. Work in the undisturbed boreal forests of the Yukon has linked goshawk density and productivity to prey abundance. Differences in the response of goshawk populations to timber harvest may therefore be primarily dependent on the prey available and the habitat used by the prey. Goshawks that are more reliant on prey associated with mature forests showed the greatest impact from harvesting. Across Canada, therefore, population responses to harvesting at the ecosystem level may vary, with the possibility that at the regional or local scale goshawk populations could be lost without this loss being detected by the present non-target monitoring techniques (CBC, BBS, and migration counts). Broad assessment of prey and prey habitat use will help managers to assess the risk to population persistence at regional and local scale.

Key Words: *Accipiter gentilis, laingi*, Canada, harvest, habitat, prey, status, threshold.

GAVILANES EN CANADÁ: RESPUESTAS POBLACIONALES AL APROVECHAMIENTO Y LO APROPIADO DEL USO DE TÉCNICAS ESTANDARIZADAS DE MONITOREO DE AVES PARA EVALUAR SU ESTADO.

Resumen. En este artículo utilizo los resultados de investigación actual, así como técnicas establecidas de monitoreo de aves, para resaltar la inhabilidad de las actuales técnicas para establecer el estado del Gavilán Azor (*Accipiter gentilis atricapillus* y *A. g. laingi*) en Canadá. A escala nacional, el monitoreo de los gavilanes reside en los avistamientos oportunos del gavilán, realizados durante Estudios de Aves Reproductoras (BBS), Conteos de Aves en Navidad (CBC) o durante los conteos de migración. Estos recursos indican que la tendencia de la población, es ya sea estable (BBS y CBC), o posiblemente decadente (conteos de migración durante los últimos 20–30 años). Sin embargo, estudios poblacionales recientes del gavilán en el oeste de Canadá, han mostrado que subpoblaciones individuales responden de forma distinta al cultivo del bosque maduro, algunas mostrando impacto negativo, mientras que otras parecen prosperar durante el cultivo. Trabajo realizado en el bosque boreal no perturbado del Yokon, ha vinculado la densidad y productividad del gavilán a la abundancia de la presa. Diferencias en la respuesta de las poblaciones del gavilán al aprovechamiento de la madera quizás se deban principalmente a la disponibilidad de la presa y al hábitat utilizado por la presa. Gavilanes que dependen más en presas asociadas con el bosque maduro, mostraron el gran impacto que causa el aprovechamiento. Es por esto que en Canadá, las respuestas al aprovechamiento a nivel de ecosistema quizás varíen, con la posibilidad de que las poblaciones de gavilán a escala regional o local se pierdan sin poder detectar dicha pérdida a través de las técnicas de monitoreo actuales de no-blanco (CBC, BBS y conteos de migración). Mayor valoración de la presa, así como de la utilización del hábitat por la presa, ayudarán a los administradores a evaluar el riesgo de la permanencia de la población a escala regional y local.

Goshawks in Canada are distributed throughout the entire forested portion of the landscape, from the US border to tree line in the Arctic, and thus potentially a large portion of the North American goshawk population is resident in Canada. My objective is to determine the status of goshawks across Canada based on all the available information on goshawk populations. In Canada, as in the US, large-scale harvesting of mature forests has taken place throughout the past century and up-to-date information on the present status of the goshawk is required to determine if this identified threat is influencing the status of goshawk population. However, recent long-term goshawk research in the west of Canada

has highlighted how inappropriate the standard bird monitoring tools may be in establishing the population status of goshawks across Canada. In this paper I set out the problems associated with the present methodology, and focus on the differing responses of goshawk subpopulations to harvesting. Harvesting and its impact on prey, versus impacts at a nest stand or other factors such as depredation or climate, being identified by the Canadian research studies as likely to be the most critical factor influencing the status of goshawks in Canada. This impact on prey and the differences seen in the scale and type of this impact between forest types indicates that specific monitoring of goshawk populations may be the only accurate method for determining the overall status of this species.

In Canada raptor populations have been monitored by the Canadian Wildlife Service using trend information from breeding bird surveys (BBS), Christmas Bird Counts (CBC) and from migration counts (Hyslop 1995, Kirk and Hyslop 1998, Kennedy 2003). These counts indicate a range in Northern Goshawk (*Accipiter gentilis atricapillus* and *A. g. laingi*) population trends depending on the source with BBS and CBC indicating a relatively stable population, whereas the migration counts have shown a decreasing trend in the numbers of goshawks, which is significant at three of the eight migration stations. No discernible geographic trend was observed.

In contrast to this opportunistic count of goshawks, the last decade has seen several intensive long-term goshawk population studies (Table 1; Fig. 1) taking place in the Yukon, and in British Columbia (BC) (Doyle and Mahon 2001, Doyle 2003, McClaren 2003, Mahon and Doyle 2003a). These studies were designed to quantify the possible impacts to this raptor of harvesting mature forests and have show that goshawk populations react differently to that impact on an ecosystem specific basis. This variation in impacts could potentially result in the loss of goshawks at a local or regional level, an impact that may be unnoticed at a national level if relying on the opportunistic BBS, CBC, and migration counts to monitor population trends. This can occur because the negative responses by goshawks to harvesting in one forest type may be balanced by a positive response in another, such that the coarse opportunistic monitoring fails to detect any significant change. Consequently the habitat thresholds that may negatively impact goshawks may be exceeded in some landscapes, such that the goshawk population is lost without being noticed.

The work showing the potential weakness in relying of non-target species monitoring techniques to monitor goshawk populations has all taken place in western Canada over the past decade. No comparable studies have been conducted elsewhere in Canada. In BC this has included work on two island populations of the threatened goshawk sub-species, the Queen Charlotte goshawk (*A. g. laingi*), while those on mainland BC and in the Yukon are working with the larger *A. g. atricapillus*. The Yukon study took place within an undisturbed northern boreal forest

TABLE 1. LONG-TERM GOSHAWK STUDY AREA IN BRITISH COLUMBIA AND YUKON, CANADA.

Location	Number of goshawk nest areas located	Length of study	Forest type	Nest area spacing (km)
Haida Gwaii-Queen Charlotte islands [a]	9	1995–present	Rain-forest coastal western hemlock	9–15
Vancouver Island [b]	66	1995–present	Rain-forest coastal western hemlock	6–8
Interior BC (Lakes and Morice Forest Districts) [c]	40	1997–present	Sub-boreal spruce and pine	4–5
Interior BC (Kispiox Forest District) [d]	33	1995–present	Interior cedar hemlock	4–5
Yukon [e]	13	1986–1996	White spruce	3 (P)[f] 12 (L)[g]

[a] Doyle 2003.
[b] McClaren 2003.
[d] Mahon and Doyle 2003a, Doyle and Mahon 2001.
[c] Doyle and Smith 1994, Doyle 2000.
[f] P = Years with a peak in prey abundance,
[g] L = Years with low prey abundance.

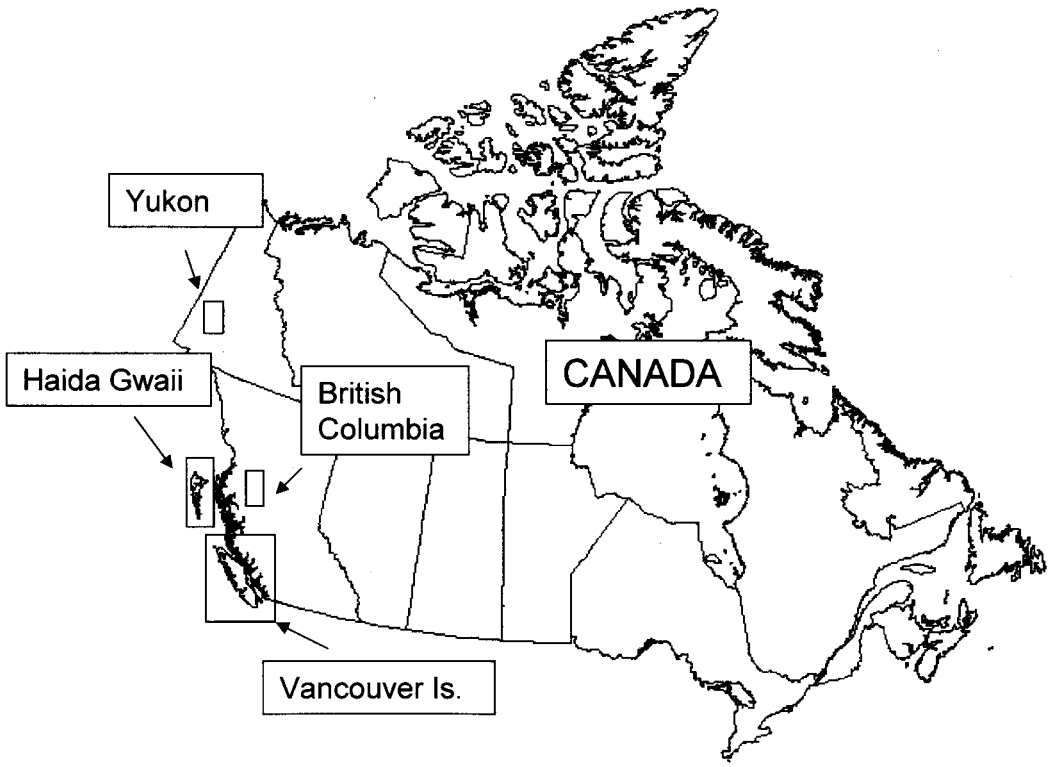


FIGURE 1. Location of the long-term goshawk study areas in Canada.

ecosystem, while those in BC are all taking place within forest types associated with the southern boreal and coastal rain forest ecosystems that have all seen 30–40% of the mature forest harvested in the last 20–30 yr (Doyle 2003, McClaren and Pendergast 2003, Mahon et al. 2003). These BC studies show very different population responses to harvesting; the rain forest *laingi* may be possibly under threat with declining populations and reduced productivity while *atricapillus,* found in the drier mainland forests, may be benefiting from harvesting, at least in the short-term.

In the Yukon, the Northern Goshawk population was studied intensively at Kluane Lake as part of a long-term boreal forest ecosystem study (Krebs et al. 2001) in which all raptors, their prey, and the environment in which they lived were monitored to establish if and how these ecosystem components were linked together. This study established that goshawks were largely resident and that the number of nests and production of young (Fig. 2) was significantly linked to the abundance of their main prey the snowshoe hare (*Lepus americanus*) and possibly also to grouse, their secondary winter and early breeding season prey. Rainfall and other factors such as nest depredation by wolverines (*Gulo gulo*; Doyle 1995) and Great Horned Owls (*Bubo virginianus*; Doyle 2000) influenced the breeding success of individual pairs; however, these impacts were insignificant compared to the density of snowshoe hares in winter. In addition, human impacts have also been cited in the past as possibly reducing breeding success at the nest stand level (Squires and Reynolds 1997), and certainly disturbance of any breeding birds can cause breeding attempts to fail. However, an ongoing long-term, adaptive-management experiment at the mainland study sites in BC (Mahon et al. 2003), has to date (3–5 yr post harvest, 73 nest areas) shown no significant impacts, with goshawks continuing to breed successfully even in highly fragmented nest stands. This does not mean a threshold fragmentation or disturbance threshold for some individuals at the nest-stand level does not exist, but it does indicate that this impact is not driving changes in population trends at this stage.

The critical role of prey in the breeding success and the subsequent status of individual goshawk populations may in part be explained by the fact

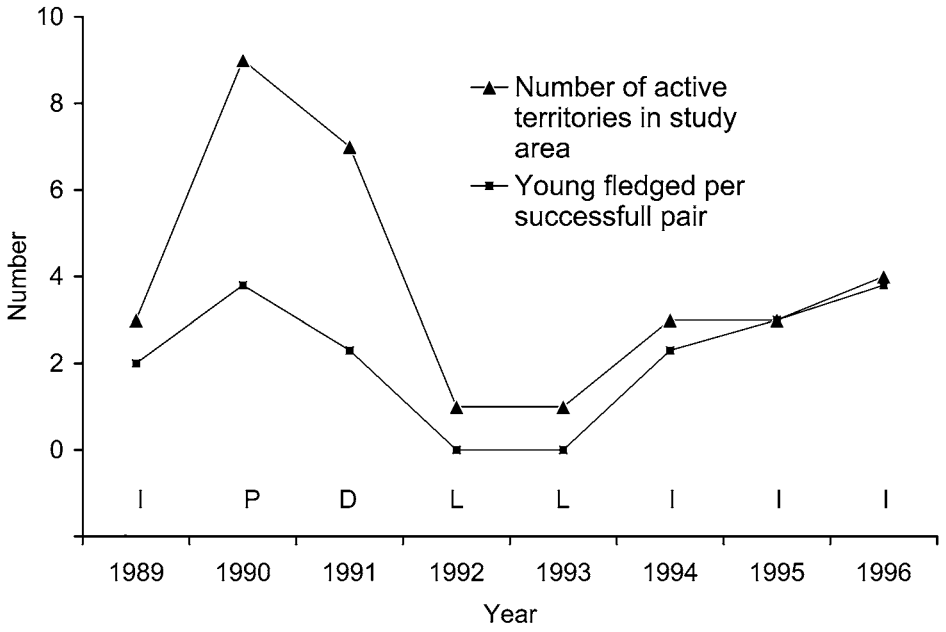


FIGURE 2. Number and productivity of successful goshawk nests in relation to the cyclic phases of the hare and grouse population at Kluane Lake, Yukon. (Doyle 2000). Phases in the abundance of hares and grouse over the period of the 10-yr snowshoe hare cycle: I = Increase, P = Peak, D = Decrease, L = Low.

that in BC as in the Yukon, populations appear to be largely resident. Birds' radio-tagged as breeding adults, and independent goshawk sightings indicate that some, and possibly all, adults and juveniles are resident in the breeding habitat during the winter months. Independent of these observations, birds in all the study locations began to breed in late winter (February–March; Doyle 2000, Mahon et al. 2003), 1–2 mo before spring or summer prey (migrant passerines, young prey or hibernating prey) were available. All birds, independent of their winter movements therefore appear to be largely dependent on the abundance and availability of winter prey to ensure their survival, body condition and subsequent ability to breed, if territories become vacant.

This likely dependence on winter prey has been identified as a possible driving factor in the observed differences in the resilience of goshawk subpopulations to habitat change (Doyle 2003). Within these same study areas in the winter months we see pronounced ecosystem differences in the species, abundance and habitat associations of the available prey (Fig. 3). In BC, two of the four long-term goshawk studies are being conducted on large islands (Haida Gwaii and Vancouver islands) off the west coast. On these islands the winter diet appears as though it may be dominated by red squirrels (*Tamiasciurus hudsonicus*) and forest grouse, while the dominant winter prey on the mainland, the snowshoe hare, is absent, and other common mainland prey, ptarmigan and grouse, are at relatively low densities. When we then look at where these prey types are found within a landscape we can see clear differences in broad habitat types. Snowshoe hares, grouse, and ptarmigan are typically associated with openings of shrub-young forest (pole sapling), while red squirrels (Mahon and Doyle 2003b), and the island's forest grouse (Zwickle 1992) are most abundant in mature coniferous forest that once dominated the entire landscape (Canning and Canning 1996). Harvesting of the mature forests that dominate these landscapes will therefore likely have very different impacts on the resident goshawk population, depending on the habitat association of their prey in that ecosystem. On the mainland the harvested openings result in habitat (after a shrub layer has formed) in which snowshoe hare and grouse densities are higher than compared to the surrounding mature forest (Mahon and Doyle 2003b). In contrast, these same harvested habitats on Vancouver Island (Ethier 1999, McClaren 2003) and on Haida Gwaii (Doyle 2003) have few prey associated with them in winter and early in the breeding season.

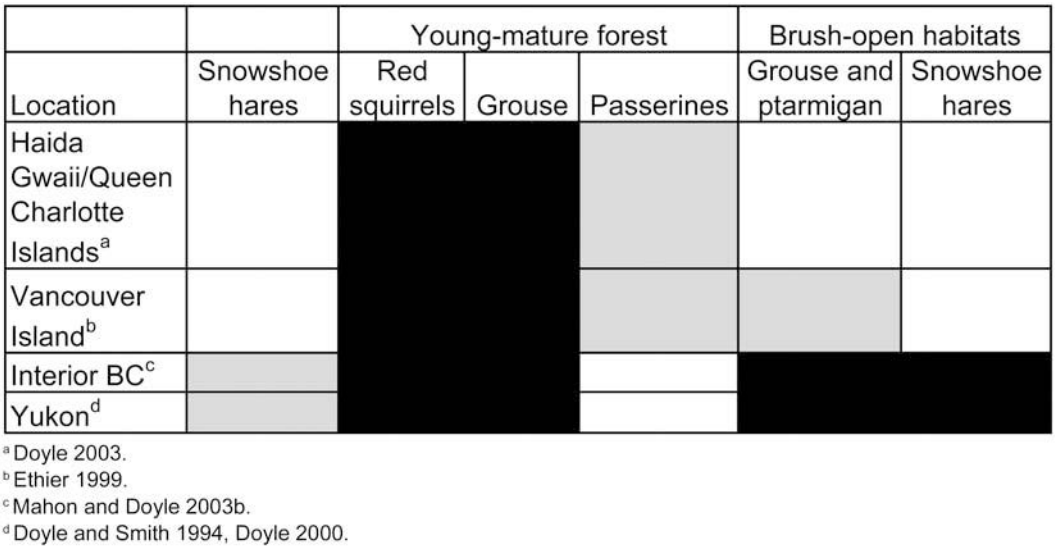


FIGURE 3. Relationship between the location (ecosystem type), and the association of habitat and the main goshawk prey in late winter. Black = species abundant; gray = species occurs but less abundant; white = species absent.

In mainland versus island we also see different patterns in goshawk productivity and territory spacing which supports this observed link between habitat and prey (Table 1). The mainland territories are at around twice the density of their island counterparts and the annual productivity (both reoccupancy rates and young fledged per breeding attempt) is higher (Doyle 2003). In addition, at the territory scale, harvesting on the mainland has not been seen to reduce breeding success (Mahon et al. 2003), but on Haida Gwaii, in areas with >30% harvesting, no active nests have been located, and on Vancouver Island (McClaren and Pendergast 2003) landscapes with the highest rates of harvesting have significantly lower reoccupancy rates.

How these observed differences in available prey, habitat, and goshawk productivity vary across Canada have not been explored, but they provide the possibility that harvesting combined with individual ecosystem differences could lead to the loss of certain goshawk populations while other populations may remain stable or indeed increase in density. However, this does not mean that we can be complacent even for those populations that appear to be relatively robust. As was seen in the Yukon hare population, which predictably peaks and then crashes cyclically, we have to ensure that enough diversity of habitat types and their associated prey remains to support goshawk population through the low in hare numbers. In particular, sufficient areas of mature forest supporting populations of red squirrels or forest grouse may be critical to the long-term persistence of goshawks and other predators. Both the Yukon study on goshawks and on another winter resident, the lynx (*Lynx canadensis*), showed a switch to prey typically associated with mature forests (red squirrels and forest grouse) during the low in hare numbers (Krebs et al. 2001). At present, the critical habitat and the threshold in habitat area required to maintain goshawk populations in landscapes in which they largely depend on cyclic prey is unknown. As we have seen the rate of timber harvesting in the goshawk study areas in BC has resulted in 30–40% of the mature forest being cut in the last 20–30 yr. In addition, this harvest has recently been spread across the landscape, such that out of the 73 or more goshawk territories located to date, none have no harvesting at a territory scale, if we assume that nest-area spacing in a landscape (Table 1) indicates the foraging area required by the birds in that landscape. If this rate and spatial arrangement of harvesting is taking place throughout the rest of the province or across large areas of Canada and a threshold in critical habitats does exist, will it allow for the retention of enough critical habitats to ensure the long-term persistence of goshawk populations across much of their range?

Finally, the possibility exists that this could all take place while information from migration stations fails to detect a notable population change at a time when action should be taking place to protect threatened populations at a regional or on a listed sub-species basis. This could occur because declines

in goshawk populations in any one area may be masked by increasing populations elsewhere. As an example, large numbers of goshawks sightings at some migration stations are thought to be influenced by the abundance and subsequent population crash of snowshoe hares in the northern boreal forests, a prey species that we know is cyclic (9–11 yr cycle) (Doyle 2000), and which also exhibits a variation in the amplitude of the density between cycles. This large number of goshawks from the northern boreal forest may therefore effectively mask declines brought about by human influences on goshawk carrying capacity of landscapes.

Other broad-scale land-bird surveys methods such as BBS and CBC are likely to be inappropriate goshawk survey methods. BBSs are focused on detection of calling by songbirds, while CBCs are not stratified across regions or habitat types, because they are centered on communities. Most communities, and therefore count sites, are located in southern Canada. In addition they do not specifically target habitat types (forested landscapes in Canada) in which we may expect to locate goshawks in the winter months. As a consequence, the present survey methods may fail to detect any significant population change.

Therefore, we do not know the long-term resilience of individual goshawk populations to habitat change and changes in prey availability outside of few local studies conducted to date in western Canada. Furthermore, a possibility exists that the broad-scale monitoring methods that are being used to monitor goshawk populations across Canada may fail to detect local or regional population declines. Additionally, it may be too late by the time these broad-scale survey methods do detect a decline. If, for example, goshawk a population increases with timber harvesting until a critical threshold is reached and then that population (genetically or regionally) declines sharply, becoming extinct in the worst case scenario. In Canada, as in other areas of North America, standardized broadcast surveys using localized detection probability functions and area occupied methods (McLeod and Andersen 1998, McClaren et al. 2003) could be used to detect changes in breeding populations. However, the challenges associated with setting up such a comprehensive monitoring system in Canada, i.e., training, money, low-detection rates in coastal forests, and cyclic goshawk populations, will likely prevent such a strategic plan being put in place until it is too late. If this is the likely outcome, research that identifies broad landscape thresholds for goshawks within individual ecosystem types based on the habitat and available prey may be necessary. Although necessarily coarse in its assessment this will at least allow landscape managers to assess the risk of their actions to the goshawks population, both at a local and regional scale.

ACKNOWLEDGMENTS

The work on goshawks across Canada is indebted to the help of many organizations and volunteers at both at the provincial and federal level, whose dedication and commitment ensures the collection of the information that comprises the invaluable long-term databases that are provided by the BBS, CBC, and migration counts. In western Canada, the Yukon study on goshawks was funded by the Natural Sciences and Research Council of Canada, and was also indebted to the hard work of many researchers and students, both from Canada and abroad. In BC, the four long-term goshawk studies were supported by BC provincial funding (Forest Renewal, Forest Innovative Account, and the Habitat Conservation Trust Fund.) and through the support of licensees including Babine Forest Products, CANFOR, Huston Forest Products, Skeena Cellulose, Timber West, Western Forest Products, and Weyerhaeuser Ltd. I also thank Erica McClaren for reviewing this manuscript, and finally a special thanks both to her, and to Todd Mahon for sharing with me a passion to understand how we may successfully manage goshawks in harvested landscapes.

STATE CAPITOL
P.O. Box 110001
Juneau, AK 99811-0001
907-465-3500

550 West Seventh Avenue, Suite 1700
Anchorage, AK 99501
907-269-7450

Governor Michael J. Dunleavy
STATE OF ALASKA

March 1, 2019

The Honorable Donald J. Trump
President of the United States
The White House
1600 Pennsylvania Avenue NW
Washington, DC 20500

Dear Mr. President:

Thank you for the opportunity to meet with you on February 28, 2019. I enjoyed our time together during your visit to Alaska. The generosity and attention you extended to our servicemen and servicewomen stationed in the Last Frontier is greatly appreciated – clearly, we share the same respect towards those who stand on the frontlines to keep our nation safe. I want to also reiterate that Alaska supports your vision for our great country and we look forward to working with you to make it a reality.

As such, I wanted to take the opportunity to share with you some matters that are of great importance to Alaskans. The following is a synopsis of issues facing Alaska that I believe by working together can advance our shared goals of economic growth and energy dominance:

U.S. Department of the Interior

- **Statehood Land Selections** – Alaska is ready to complete our statehood land selections. It's important for the Department of the Interior to lift the current secretarial orders that are preventing priority selections from attaching. One key example is Public Land Order 5150 - put in place for the Trans Alaska Pipeline System (TAPS). It is time for the state to own that right-of-way, then this land can finally generate economic development. Lifting that single Land Order will satisfy close to half of the land we were promised. The Land Orders were put in place without a land plan or environmental analysis, but now the Bureau of Land Management (BLM) is demanding both an updated plan and environmental review to lift each order. Alaska views BLM's demand as an example of a government agency creating unnecessary work that only stalls economic growth.
- **Arctic National Wildlife Refuge 1002 area lease sale** – U.S. Fish and Wildlife career employees undermined seismic work this winter, so the sale will occur without valuable data, likely lowering the bids. These same U.S. Fish and Wildlife employees are working to undermine the lease sale. ANWR has the potential to become the next Prudhoe Bay, and is a key element for your goal of America to achieve energy dominance.

- **National Petroleum Reserve – Alaska (NPR-A)** – Former President Obama took half of the NPR-A off the table for oil and gas development. This land needs to be put back into the leasing plan. A new land plan and environmental review needs to be done before that can be put up for lease.

U.S. Army Corps of Engineers (U.S. Corps) / Environmental Protection Agency (EPA)

- **Waters of the U.S. Rule (WOTUS)** – The U.S. Corps and EPA have a long history of misinterpreting the Clean Water Act to give themselves more and more power. The Clean Water Act applies to "***waters*** of the United States" – not wet***lands***. Lands are not waters. In 2017, you issued a Presidential Executive Order directing the Corps and EPA to revisit WOTUS regulation, which defines federal jurisdiction under the Clean Water Act. Instead, what the Corps and EPA have proposed isn't much different than before and doesn't give much control back to Alaska – if any at all. When you're flying over Alaska, you can see wetlands everywhere. In fact, Alaska has 174 million acres of wetlands – more than all states in the Lower 48 combined. I understand that people are concerned about the disappearing wetlands in the Lower 48; but, we don't have that problem in Alaska. We are a young state, with tremendous undeveloped resource potential that could be of great benefit to our country's energy security. However, this proposed federal regulation is something that has prevented Alaska's economy from growing and prohibited natural resource development, which ultimately is a roadblock to America becoming energy dominant – something that is a priority of yours. Please urge your departments to not adopt a regulation that is counter to your vision for our great nation.

- **Clean Water Act 404 veto** – Natural resource development in Alaska is handicapped due to the EPA's failure to officially announce that a preemptive Clean Water Act 404 veto is not going to be used anywhere in the state. The Pebble Mine Project is the poster child, but every potential investor must evaluate whether that preemptive veto could be used against their project; which results in a serious brake that chill potential resource development opportunities.

- **Clean Water Act 401 permits** – My team is available to assist the EPA in rewriting the handbook on Clean Water Act 401 Water Quality permits. By working together, we could create a federal policy that is of benefit to both the nation and Alaska.

- **Clean Air Act – Fairbanks air quality** – Fairbanks has some of the nation's coldest weather and highest costs of energy, and people have been burning wood because it's all they can afford. This is nothing new, as Fairbanks has been doing this for decades. But now, the EPA says we have an air quality problem due to woodsmoke. Keep in mind, Fairbanks has temperatures as cold as -50 below zero. Being able to heat your home isn't a luxury – it's a matter of survival. Fairbanks has been making great strides to reduce woodsmoke emissions, but EPA is telling us it's not enough and that we need to implement costly and intrusive regulations. In concert with the measures already taken, a potential long-term plan is to get affordable natural gas to Fairbanks. We need the EPA to be patient and ask they also explore potential regulatory changes that consider Fairbanks' Arctic climate and high energy costs.

Medicaid

- **Medicaid Block Grants –** Your Medicaid Administrator, Seema Verma, has urged us to be the first state to receive Medicaid dollars as a block grant. We are eager to do this, but your support of her on this "first" will keep the proper focus and speed on the application.

U.S. Forest Service (USFS)

- **Tongass National Forest / Roadless Rule –** The USFS is working hard to reverse a Clinton-era rule that virtually shut down logging in the Tongass Forest. New York State logs more timber commercially than Alaska. That rule-making must go perfectly to be completed timely and save what's left of our timber industry. In January 2018, former Governor Walker petitioned the USDA for new rulemaking to completely exempt the Tongass from the 2001 Roadless Rule and to rescind the interlocking 2016 Tongass Transition Plan. USDA began the process in August 2018 and is now considering what the preferred alternative in the DEIS should be. Secretary Perdue should maintain the policy determination made by the USDA in 2003 by selecting total exemption as the preferred alternative in the DEIS. We look forward to you providing clarification to the Secretary of Agriculture on the roadless rule exemption for the Tongass EIS. This would be a significant victory for the timber industry in Southeast Alaska and their efforts to increase the annual timber harvest, which creates jobs and fosters economic opportunity.

U.S. Customs and Border Patrol (CBP)

- **Tourism Opportunities –** Iceland continues to allow tourists to travel throughout their country for a few days without a visa, and business is booming. Anchorage Airport built a special terminal so that Asian tourists without visas could de-plane during fuel stops, shop at Duty Free, then re-board, or even get on a different flight headed to another foreign destination. These tourists could not leave the terminal. Business was booming. Then, after 9/11, CBP ended this program. It is time for CBP to allow the restart of this program. Anchorage lost all that business to the airport in Vancouver, Canada.

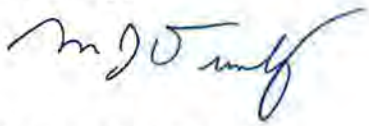
Fisheries Appointments

- **U.S. Pacific Salmon Commission –** I have submitted the appointment request for Mr. Doug Vincent Lang, Commissioner of the Alaska Department of Fish and Game, as our selection for the Alaska seat on the U.S. Pacific Salmon Commission.

- **North Pacific Fishery Management Council -** Our nominations for the North Pacific Fishery Management Council have been forwarded to Secretary of Commerce Ross, and we look forward to his appointments soon.

Thank you again for taking time to visit our nation's troops and for your voiced support of Alaska's needs. I know many of the items listed will require mutual attention from the State of Alaska in concert with the federal government. Alaska stands ready to do our part to materialize the dream of a great state supportive of a great nation, and I look forward to our next visit.

Sincerely,



Michael J. Dunleavy
Governor

Office of Governor
MIKE DUNLEAVY

Governor Applauds USDA Support to Lift Tongass Roadless Rule Exemption

October 15, 2019

October 15, 2019 (Anchorage, AK) – Today Alaska Governor Michael J. Dunleavy praised the U.S. Department of Agriculture's draft Environmental Impact Statement (EIS) on the proposed Alaska Roadless Rule exemption for the Tongass National Forest. The draft EIS lists six potential options and selects Alternative 6, a full exemption of the Tongass National Forest from the Clinton-era 2001 Roadless Rule as the preferred option.

"Today's announcement on the Roadless Rule is further proof that Alaska's economic outlook is looking brighter every day," **said Governor Dunleavy**. "The ill-advised 2001 Roadless Rule shut down the timber industry in Southeast Alaska, wiping out jobs and economic opportunity for thousands of Alaskans. I thank the USDA Forest Service and for listening to Alaskans wishes by taking the first step to rebuilding an entire industry, putting Alaskans back to work, and diversifying Alaska's economy."

Alternative 6 is fully responsive to the State of Alaska's petition to completely remove the Tongass National Forest from the Roadless Rule. This USDA preferred alternative removes all 9.2 million roadless acres and reclassifies 165,000 old-growth acres and 20,000 young-growth acres to suitable timber lands. The draft EIS only applies to the Tongass National Forest.

Governor Dunleavy worked with President Donald Trump, USDA Secretary Sonny Perdue, Congressman Don Young, Senator Dan Sullivan, Senator Lisa Murkowski and others to reopen the Tongass and wants to thank them for their support and hard work.

Governor Dunleavy urges Alaskans to submit comments in favor of Alternative 6 and a growing Alaska economy between Friday, October 18 and December 17, 2019. Please go to the Alaska Roadless Rule project website for more information.

ALASKA WILDERNESS LEAGUE – CENTER FOR BIOLOGICAL DIVERSITY
DEFENDERS OF WILDLIFE – EARTHJUSTICE – GEOS INSTITUTE
NATURAL RESOURCES DEFENSE COUNCIL – SIERRA CLUB
SOUTHEAST ALASKA CONSERVATION COUNCIL

February 22, 2016

VIA EMAIL & EXPRESS MAIL

Earl Stewart
Forest Supervisor
Tongass National Forest
Attn: Forest Plan Amendment
648 Mission Street
Ketchikan, AK 99901
Email: comments-alaska-tongass@fs.fed.us

Dear Forest Supervisor Stewart,

The U.S. Forest Service is in the process of amending the Tongass Land Management Plan and has prepared a draft plan (the Draft Forest Plan) and has prepared an accompanying Draft Environmental Impact Statement (the DEIS). These comments are submitted on behalf of the undersigned organizations. These groups have long-standing interest in the social and ecological values of the Tongass National Forest and any developments that may affect those values. For the reasons described below, the Forest Service should not adopt the Draft Forest Plan as proposed and should instead should consider and adopt a transition that rapidly—within less than five years—ends industrial-scale old-growth logging.

The undersigned groups share Secretary of Agriculture Thomas Vilsack's vision of taking immediate steps to conserve the Tongass National Forest, promote sustainable development, strengthen the wildlife conservation strategy by maintaining future options, enlist its globally significant carbon-rich old-growth forests to advance America's climate preparedness, and help reduce the effects of climate change worldwide. The Tongass, an internationally significant and nationally valued natural treasure, must be managed to conserve biological diversity, support local communities, and protect the ecological integrity of the coastal temperate rainforest in Southeast Alaska. Additionally, the Tongass is the nation's carbon forest and as such should reflect the Administration's commitment to addressing global climate change.

Since the Forest Service revised the Tongass Plan in 1997, Southeast Alaska has witnessed an explosion in demand for Southeast Alaska's fish, wildlife, and outdoor recreation resources and grappled with effects from enduring changes in global timber markets. Today, the Tongass National Forest is both a world-class salmon forest and superlative tourist destination. While the customary and traditional use of deer, salmon, and other renewable Tongass forest resources

continue to support a way of life vital to the cultural and economic resiliency of local communities.

We agree with the Forest Service's decision to bring an end to industrial-scale old-growth logging. The Draft Forest Plan, however, compromises the Tongass's ability to support essential economic, environmental, and social interests because it does not provide a fixed and rapid end to industrial-scale old-growth logging on the Tongass. Indeed, the plan as proposed has no mechanism whatsoever to bring about an end to industrial-scale old-growth logging. Instead, the Draft Forest Plan proposes 16 years and likely many more of business as usual, concentrating the Tongass's entire logging program on a handful of islands to the detriment of social, cultural, and environmental considerations.

Rather than implement the goal of a rapid-transition out of industrial old-growth logging through a targeted, narrow amendment to the Tongass Forest Plan, the Forest Service has proposed wide-spread changes that are both controversial and extraneous, such as extensive modifications to the wildlife conservation strategy. Even more troubling, however, the agency proposes these changes without the benefit of a scientifically defensible and informed process.

The undersigned groups appreciate that, consistent with current agency policy and practice on the Tongass, the Draft Forest Plan appears to provide additional conservation measures for the Tongass's inventoried roadless areas. This will improve America's climate preparedness and assure the ecological, social, and economic sustainability of the Tongass by removing roadless areas from logging to protect remaining intact old-growth habitat and to maintain existing carbon stocks while also increasing stored carbon, considered nationally and globally significant. We also appreciate that the agency has proposed additional protections for areas of high value for salmon, wildlife, and other non-logging uses, where past logging, particularly on Prince of Wales Island, has diminished old-growth habitat to perilously low levels and stressed rare species. We support these protections. Focusing on the forest's world-class fisheries, tourism, and recreational values is a positive direction for forest management.

Because most damage to the Tongass's fish, wildlife, and other natural values has been associated with old-growth logging and related road-building, a rapid transition out of industrial-scale old-growth logging will greatly reduce future harm, particularly in a rapidly changing global and regional climate. The enormous amount of old-growth volume proposed for logging under the Draft Forest Plan during and after the transition has been completed will cause significant problems for ecosystems and vulnerable species. This added harm is unnecessary and inconsistent with transition goals. To avoid the continuing controversy surrounding industrial-scale old-growth logging, the Forest Service should adopt a forest plan that brings about real change on the Tongass and fulfills the Secretary's directive to "transition to a more ecologically, socially, and economically sustainable forest management."[1] Adopting a forest plan amendment that ends industrial-scale old-growth logging on the Tongass in less than five years accomplishes those important objectives.

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[1] PR 769_01_000046 at PDF 1 (U.S. Department of Agriculture, Office of the Secretary, Secretary's Memorandum 1044-009 Addressing Sustainable Forestry in Southeast Alaska (July 2, 2013)) (Secretary Vilsack's Memo).

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PURPOSE AND NEED

I. THE DEIS MISCHARACTERIZES A KEY COMPONENT OF SECRETARY VILSACK'S MEMORANDUM AND IGNORES OTHER IMPORTANT CONSIDERATIONS.

The DEIS's statement of purpose and need section ignores critical aspects of the rationale underlying the proposed plan amendment. In July 2013, Secretary Vilsack directed the Tongass National Forest to "transition to a more ecologically, socially, and economically sustainable forest management . . . ."[2] Secretary Vilsack made it clear that "[t]he intent of the actions announced today is that second-growth timber sales will continue to increase, and within 10 to 15 years that the vast majority of timber sales on the forest will be from second-growth forests."[3] Thus, Secretary Vilsack sought a transition out of old-growth logging that was to be completed no later than 10 to 15 years after his 2013 announcement (i.e., July 2028).

In stark contrast, the Forest Service has proposed a Draft Forest Plan that has no enforceable end to old-growth logging. Indeed, the DEIS makes clear that the agency selected an alternative that increases rates of old-growth logging higher than the recent 10-year average for at least another 16 years, but even that "end" is nothing more than aspirational.

The DEIS purports to analyze the Secretary's intent "to transition [the Tongass] forest management program to be more ecologically, socially, and economically sustainable,"[4] but the DEIS focuses almost exclusively on the economic viability of the timber industry. The "Purpose" section of the DEIS, for example, never even mentions social or ecological factors.[5] Similarly, the "Need" section describes the importance of maintaining a "viable timber industry," but never explains how the agency proposes to address the "social" or "ecological" aspects of the transition.[6] The agency cannot ignore two of the three critical parts of the Secretary's direction;

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[2] Secretary Vilsack's Memo at PDF 1.

[3] U.S. Department of Agriculture, *Secretary Vilsack Announces Steps to Conserve 17-Million Acre Tongass National Forest by Transitioning to Sustainable, Second-growth Forest Management*, Press Release No. 0140.13 (July 3, 2013). All of the documents cited in these comments that are not included in the agency's planning records (including the records for the 1997 Tongass Land Management Plan (1997 Forest Plan) and the 2008 Amended Tongass Land Management Plan (2008 Amended Forest Plan) have been hand-delivered to the Forest Service and also sent via Express Mail with a copy of these comments. All of those sources as well as these comments should be included in the administrative record for the decision to amend the Tongass Land Management Plan.

[4] DEIS at 1-4.

[5] *Id*. at 1-4 to 1-5.

[6] *Id*. at 1-5.

the FEIS must expand the scope of the purpose and need statement to examine and account for these concerns.[7]

II. CHANGES TO THE FOREST PLAN'S DIRECTIVES FOR TRANSPORTATION DO NOT FULFILL THE PURPOSE AND NEED FOR THE AMENDMENT.

The Draft Forest Plan contains new direction for transportation whose purpose is "to facilitate the availability of National Forest System land for the development of existing and future transportation systems."[8] In the context of this amendment, however, it does not appear that the Forest Service has ever identified a need to facilitate transportation systems.[9] Neither the Purpose and Need section of the DEIS, nor the Purpose section of the Draft Forest Plan, nor the Notice of Intent to Prepare an EIS, nor Secretary Vilsack's Memo, describes such a need.[10]

The Purpose and Need section of the DEIS simply states that "[c]hanges to the Forest Plan are needed to make the development of renewable energy resources more permissible, including considering access and utility corridors to stimulate economic development in Southeast Alaska communities."[11] But the new Transportation Systems Corridor Direction in the Draft Forest Plan is not related to or limited to renewable energy projects.[12] Besides, the Renewable Energy Direction in the Draft Forest Plan accomplishes this more directly by opening all LUDs to renewable energy sites, which include associated access roads and transmission lines.[13]

Under the 2012 planning rule, forest plan amendments should be based on an identified need to change the plan.[14] Here, the Forest Service never identified, and still has not identified a need to change the forest plan's transportation direction.[15] As a result, neither the public nor the agency was adequately prepared to address this issue, much less the potentially sweeping transportation changes proposed in the Draft Forest Plan.[16] Their inclusion in the amendment is inappropriate.

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[7] *See* 36 C.F.R. § 219.8 (2012 Planning Rule requires plans to "provide for social, economic, and ecological sustainability").

[8] Draft Forest Plan at 5-13; *see also generally infra* pp. 111-18.

[9] *See* 36 C.F.R. § 219.13(b)(1) ("The responsible official shall . . . [b]ase an amendment on a preliminary identification of the need to change the plan.").

[10] *See generally* DEIS at 1-4 to 1-5; Draft Forest Plan at 1-1; 79 Fed. Reg. 30,074, 30,075 (May 27, 2014); Secretary Vilsack's Memo.

[11] DEIS at 1-5.

[12] *See* Draft Forest Plan at 5-13 to 5-15.

[13] *Id.* at 5-12 (making all Tongass land potentially suitable for renewable energy sites); *id.* at 7-55 (defining "renewable energy site" to include access roads and transmission lines).

[14] 36 C.F.R. § 219.13(b)(1).

[15] Draft Forest Plan at 5-13.

[16] *See infra* pp. 111-18.

RANGE OF ALTERNATIVES

Consistent with Secretary Vilsack's directive to transition out of industrial-scale old-growth logging no later than 10 to 15 years, several groups provided the agency an alternative proposing a five-year transition out of industrial-scale old-growth logging on the Tongass (the Conservation Alternative).[17] The DEIS explains that the agency refused even to consider the Conservation Alternative in its detailed consideration.[18] The agency's rationale for dismissing this alternative is arbitrary, inconsistent with the Secretary's directive, and not based on the best available information.

As an initial matter, the Forest Service must transition out of industrial-scale old-growth logging regardless of the availability of second-growth. For the ecological, social, and economical reasons outlined in this letter, the agency must bring an end to the controversy and destruction of old-growth logging. Clear-cutting Tongass old-growth is a sad reflection of bygone era. Today's economic drivers depend upon the forest's old-growth stands to support Southeast Alaska's fish, wildlife, and outdoor recreation industries. Clear-cutting these ancient trees also compromises America's climate preparedness, and reduces the country's ability to address the effects of climate change worldwide.

The DEIS explains the agency dismissed the Conservation Alternative because it failed to provide sufficient second-growth volume to facilitate a transition in five years. The comments submitted by Natural Resources Defense Council (NRDC) and Geos Institute (Geos) explain the agency's explanation is based on flawed reasoning given available information demonstrates a five-year transition meets and soon exceeds the agency's timber targets. Rather than repeat those arguments, the undersigned groups incorporate the comment letter and the supporting exhibits and materials.[19] The undersigned groups request that the agency fully analyze the Conservation Alternative, including the new information.

Additionally, all of the alternatives in the DEIS are essentially the same as to the transition out of old-growth—all action alternatives seek to provide similar volumes under similar 10-15 years timeframes. The agency artificially constrained its development of the alternatives in such a way that compromises the agency's overall analysis.[20]

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[17] *See* PR 769_02_000013 (Southeast Alaska Conservation Council *et al.*, Letter to F. Cole Re: Request for Addition of a Conservation Alternative into Tongass Transition Framework (Feb. 5, 2015)); PR 769_04_000014 (F. Cole email to R. Abt and S. Howle Re: FW: follow up on conservation alternative for transition DEIS (Mar. 6, 2015) (Attachment "FC Ltr re Additional Info on Transition Alt final.pdf"))).

[18] DEIS at 2-7 to 2-8 (Alternative 7 in the DEIS).

[19] *See* Natural Resources Defense Council and Geos Institute Letter to Earl Stewart, Tongass Forest Supervisor at Sec. I (Feb. 22, 2016).

[20] *See* Council for Environmental Quality, Effective Use of Programmatic NEPA Reviews at PDF 21-22 (Dec. 2014); *Ctr. for Biological Diversity v. Nat'l Highway Transportation Safety Administration*, 538 F.3d 1172, 1218 (9th Cir. 2008).

ROADLESS AREAS

It appears that the Draft Forest Plan is intended to keep Inventoried Roadless Areas off limits to logging in its own right, regardless of the fate of the Roadless Area Conservation Rule in two pending court actions or in any future amendments to or litigation over that rule.[21] Commenters appreciate this important amendment and strongly urge the Forest Service to adopt a final plan amendment that protects roadless areas from logging.

The Forest Service should clarify this intent in Appendix A of the Proposed Forest Plan, the suitable lands analysis. That analysis identifies Inventoried Roadless Areas as not suitable for timber production on legal grounds in "Step 1," since logging on those lands is prohibited by the Roadless Rule.[22] This is correct, but it should be made clear that even if the Roadless Rule were struck down by a court, or amended by USDA, the intent of this forest plan amendment is to preclude logging in roadless areas, as stated in the DEIS. Therefore, the "Step 2" analysis should specify that Inventoried Roadless Areas would be deemed not suited for timber production even if they lost their protection under Step 1.

SUITABLE LANDS

As discussed above, the suitable lands analysis in Appendix A of the Proposed Forest Plan should clarify that Inventoried Roadless Areas are not suited for timber production at either Step 1 or Step 2, without regard to the fate of the Roadless Area Conservation Rule.

The structure of the Proposed Forest Plan also creates confusion regarding the status of lands in Development LUDs that are not suited for timber production, and the final plan should clarify the intent. Appendix A states that Phase 2 and 3 lands from the 2008 Amended Forest Plan, as well as lands identified by Trout Unlimited and The Nature Conservancy, are deemed not suited for timber production, even though they are in Development LUDs.[23] Confusingly, however, the plan leaves in place prescriptions for Development LUDs in Chapter 3 that "[f]orest lands are suitable for timber production…."[24] The Proposed Forest Plan does make clear that Chapter 5 Plan Components (which include the changes to the suitable land base)[25] take priority over the

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[21] *See*, *e.g.*, DEIS at 3-238 ("Alternative 3 would permit old-growth and young-growth harvest in 2001 Roadless Areas, but only if the Roadless Rule changed or the Tongass Roadless Rule Exemption were reinstated. No harvest in Roadless Areas would occur under Alternatives 1, 4, and 5"); *id.* at 3-409 ("Only Alternatives 2 and 3 would allow harvest within IRAs and a change to existing regulations would be required for this to happen."); *id.* at 2-34, Table 2-14, Key Components of Alternative 5 ("No harvest is allowed in Inventoried Roadless Areas").

[22] Draft Forest Plan, App. A at A-2.

[23] Draft Forest Plan, App. A at A-4.

[24] Draft Forest Plan at 3-114 (TIM4), 3-121 (TIM4).

[25] Draft Forest Plan at 5-4 (noting that Appendix A has been updated to comply with the 2012 Planning Rule).

Chapter 3 prescriptions.[26] Therefore, it appears to be the intent of the plan that no logging would be allowed on unsuitable lands, despite the inconsistent prescriptions in Chapter 3. The plan should make this clear, either by taking the unsuitable lands out of Development LUDs, or by changing the Chapter 3 TIM4 prescriptions to make clear that not all forest lands within those LUDs are suitable for timber production.

MARKET DEMAND

The entire Draft Forest Plan and all of the alternatives (including no action) in the DEIS are based on a flawed estimate that market demand for timber will average 46 mmbf per year for the next fifteen years, regardless of the allocation between old-growth and second-growth. The Forest Service should revise this estimate downward to more accurately reflect realistic market conditions and trends, and recognize in the alternatives the likely variation from these projections. As it stands, the inflated market demand levels distort the entire DEIS, because the Forest Service has used it in restricting the range of alternatives to those that have a single unvarying timber output and in misrepresenting the ostensible jobs and economic benefits from logging. This violates NEPA, misapplies the market demand provision of the Tongass Timber Reform Act, and skews the multiple use balancing choices under the National Forest Management Act and the Multiple-Use Sustained-Yield Act.

The market demand projections are based on a draft report from the Pacific Northwest Research Station (PNW), *Tongass National Forest Timber Demand, Projections for 2015 to 2030* (Daniels *et al*.).[27] As explained in the attached comments by Natural Resource Economics, Inc.,[28] Daniels *et al*. overestimates the likely actual market demand for timber. The undersigned parties hereby attach the Natural Resource Economics analysis in its entirety as their own comments.

Daniels *et al*. perpetuates the use of a methodology that has consistently and repeatedly overestimated market demand projections in the past.[29] It does not account for the powerful economic forces that have caused logging levels to decline continuously from a peak of nearly 600 mmbf in 1973 to less than one-tenth that level today.[30] Although the most defensible assumption would be for those forces to continue, Daniels *et al*. assumes instead that they will cease and reverse.[31] Like past failed market demand projections, all of the scenarios in Daniels *et al*. show rising demand over time, contrary to the four-decade trend.[32] The Daniels *et al*.

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[26] Draft Forest Plan at 1-4.

[27] J.M. Daniels *et al.*, *Tongass National Forest Timber Demand, Projections for 2015 to 2030* (manuscript in preparation, Dec. 2, 2015). (NOTE: This document is listed on the project record index without a record number, but instead with the web link.)

[28] E. Niemi, *Socioeconomic Comments: Timber Demand* (Feb. 2015) (Niemi, *Timber Demand*).

[29] *Id.* at PDF 7-8.

[30] *Id.* at PDF 8-9.

[31] *Id.* at PDF 9.

[32] *See* DEIS App. G at G-6, Table G-1.

analysis is based on a set of implausible and unsupported assumptions that the Tongass will retain the same share it currently has of rising global demand.[33] The Forest Service should recognize that this approach has failed to accurately predict demand in the past, and should at least consider of the more realistic scenario that the long-standing historic trend will continue, with demand continuing to decline.

The DEIS should also consider the plausible scenario that future logging levels will fall below the Daniels projections and display the economic effects of the alternatives accordingly.[34]

Because of the high degree of uncertainty in projecting market demand, the forest plan should not include an objective tied to any specific annual offer level. The 2008 Forest Plan includes an objective to "[s]eek to provide an economic timber supply sufficient to meet the annual market demand," without fixing it to any projected level.[35] This allows flexibility to adjust the annual market demand projections according to actual market conditions. The Draft Forest Plan would add an objective to "offer an average of 46 MMBF annually," regardless of actual demand.[36] This is too inflexible and will likely lead the Forest Service to waste resources offering excessive, unwanted sales. This objective is unnecessary and should be deleted. Objective O-TIM-02 continues the objective to seek to meet market demand and is more flexible.

The project record also discloses that the DEIS notably inflates the volume of old-growth—5 mmbf per year—needed to sustain small timber operators.[37] In email correspondence, the Regional Economist and Dr. Daniels agreed that the volume used by small operators "is probably somewhere between 1.4 MMBF and 3.0 MMBF. 5.0 MMBF at the VERY HIGH end with VERY AGGRESSIVE marketing of products elsewhere."[38] Despite this "very aggressive" estimate, the Forest Supervisor directed, "Use 5 for the draft. . . ."[39] The result is a level of old-growth logging, in perpetuity, higher than needed to meet the established goal of serving small operators.

The exaggerated market demand forecasts—adopted across the board for old-growth, second-growth, and small operators—also violate the economic sustainability requirement of the 2012 forest planning regulations. Chapter 5 of the Draft Forest Plan explicitly adopts, for the first

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[33] Niemi, *Timber Demand* at PDF 12-16.

[34] *See id.* at PDF 16-18.

[35] 2008 Forest Plan at 2-7.

[36] Draft Forest Plan at 5-16 (O-TIM-01).

[37] *See* DEIS at 2-9 ("Old-growth volume would continue to decrease until it reaches about 5 MMBF per year and it would remain at that level, to support limited small timber operators.").

[38] PR 769_05_000794 at 1 (N. Grewe email to M. Lisowski Re: Question from Forrest (Apr. 2, 2015)) (emphasis in original).

[39] PR 769_05_000794 at 1 (F. Cole email to R. Abt Re: Fw: Fwd: Question from Forrest (Apr. 3, 2015)).

time, a Projected Timber Sale Quantity (PTSQ) of 46 mmbf.[40] Therefore, this new provision must comply with the 2012 planning rules, which require that the plan must provide for economic sustainability.[41] There are multiple reasons to believe that the PTSQ is not economically sustainable: it is based on unreasonable assumptions of a cessation and reversal of historic trends depressing the demand for timber from the Tongass; it disregards the fact that past and present logging has consistently targeted the most valuable and accessible stands of timber in the Tongass, leaving a remaining pool of old-growth timber that is less valuable and more expensive to cut; and it unrealistically assumes that Congress will continue to fund a timber sale program that results in enormous losses to taxpayers, far in excess of any reasonable measure of the benefits of the program. The DEIS contains no analysis of whether the new PTSQ, and the associated plan objective to meet it, complies with the economic sustainability requirement of the 2012 planning rules. This is an important omission that must be corrected.

Over-estimating market demand forecasts also has a negative influence on the timber sale program in its yearly implementation. Each year, the Forest Service uses the Morse methodology to calculate the volume of timber needed to offer for sale to maintain adequate inventory for purchasers.[42] This methodology uses the PNW forecasts—which, in the future, will be Daniels (2015)—as the "projected harvest."[43] If the PNW forecasts overstate market demand, the Morse methodology will, each year, tell the Forest Service to prepare more timber than actually needed to satisfy the market. The result is wasteful preparation of excessive timber that frequently is not of interest to industry and fails to appraise positively. This flaw has existed for years without correction. The Forest Service must take steps either to monitor demand on an annual basis and bring its timber offerings into line with actual market conditions or to find a different, more realistic input for the "projected harvest" in the Morse methodology.

Similarly, the Draft Forest Plan sets a goal of maintaining a three-year supply of timber under contract based on "annual timber consumption (i.e., the amount that is expected to be logged in a given year)."[44] If the expected logging is based on the PNW forecasts, it will likely be too high. This over-estimate then gets tripled in the attempt to maintain a three-year supply, again causing the Forest Service to waste public resources on excessive, uneconomic timber sale preparation.

Under the 2008 Amended Forest Plan, the Forest Service has calculated the multi-year timber supply goal in different, inconsistent ways, which should be corrected in the plan amendment. The DEIS clearly states that the goal should be three years of "the amount that is expected to be harvested in a given year."[45] Similarly, the Big Thorne Final Environmental Impact Statement

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[40] Draft Forest Plan at 5-16.

[41] 36 C.F.R. § 219.8, 219.8(b); *see also id.* § 219.19 (defining "sustainability").

[42] *See* DEIS App. G at G-1 to G-2, G-6; 2008 TLMP AR 603_0393 (K.S. Morse, *Evaluating the Demand for Tongass Timber* (1998)) (Morse).

[43] *See* DEIS App. G at G-1 to G-2, G-6; *see also* Morse at PDF 28-29.

[44] DEIS App. G at G-7.

[45] *Id.*

(Big Thorne FEIS) stated that the goal was "three times the amount of annual projected harvest."[46] When calculating the goal, though, the Big Thorne DEIS tripled not the expected harvest, as stated in the plan and the text of the EIS, but the result of the Morse methodology, a timber *offering* goal that greatly exceeds the expected harvest.[47] The result is a greatly inflated number for the goal of volume under contract, creating even further pressure to prepare wasteful, unwanted sales. In the amended forest plan, please provide direction ensuring that agency staff use the expected harvest level when calculating the three-year volume-under-contract goal.

Finally, the plan amendment should correct a change that was made without explanation between the 2008 Amended Forest Plan and the Draft Forest Plan, raising the volume-under-contract goal. In the current forest plan, the goal is "Provide 2 to 3 years supply of volume under contract…."[48] This goal was supported by an analysis in Morse showing this was a typical range.[49] The Draft Forest Plan, however, raises that goal to "Provide about 3 years of supply of volume under contract…."[50] Raising the goal from 2-1/2 years to 3 years is a 20% increase, and it is made with no explanation and no analysis. When so much unwanted timber is already being prepared, an increase in timber sale goals is unwarranted. The amended forest plan should revert to the objective in the current plan or, at a minimum, provide a thorough explanation for a significant increase in the goal.

EXPORT POLICY

Since 2007, the Regional Forester has annually re-adopted a Limited Export Policy, allowing 50% of spruce and hemlock sawlog volume to be shipped out of Southeast Alaska as round logs with no local processing.[51] The DEIS effectively ratifies this yearly decision as the de facto permanent policy of the Region by making explicit assumptions that the policy will continue indefinitely.[52] The policy has the effect of increasing the volume of timber sold on the Tongass, thereby increasing environmental impacts, while decreasing the number of jobs created per unit of timber cut. As discussed further below, this is a policy with significant environmental impacts that reflects trade-offs between environmental and economic costs and benefits. It has never been subject to NEPA review. Because the proposed plan effectively makes this a de facto permanent policy, the DEIS must consider the environmental effects of the policy and alternatives to it. The failure to do so violates NEPA. We urge the Forest Service to discontinue the Limited Export Policy.

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[46] Big Thorne FEIS Vol. II, App. A at A-13.

[47] *Id.* at A-14, Table A-2.

[48] 2008 Amended Forest Plan at 2-7.

[49] Morse at PDF 24-26.

[50] Draft Forest Plan at 2-5.

[51] DEIS at 3-453 to 3-454.

[52] DEIS at 3-481; Daniels at 10 (noting that the Limited Export Policy was a major development and incorporating it into future demand projections).

I. BACKGROUND.

Due to the combined effects of legal restrictions and unfavorable market conditions, in the mid-2000s, the Forest Service was increasingly concerned that timber sales in the Tongass National Forest were proving less viable than desired.

Applicable law restricted the export of unprocessed timber cut on the Tongass, and simultaneously limited the advertisement of timber sales to those that would appraise positively. Under Federal statute, timber cut from National Forest land "may be exported from the State or Territory where grown if, in the judgment of the Secretary of the department administering the national forests, or the public lands in Alaska, the supply of timber for local use will not be endangered thereby."[53] The statute authorizes agencies to promulgate regulations to this effect.[54] Under Forest Service regulations, "[u]nprocessed timber from National Forest System lands in Alaska may not be exported from the United States or shipped to other States without prior approval of the Regional Forester."[55] The rationale for this regulation is that a restriction on extra-Alaskan export is necessary to provide a volume of timber to sustain "adequate wood processing capacity in Alaska" such that timber from the National Forests in Alaska can be sustained.[56] In 2003, Congress passed a Consolidated Appropriations Resolution providing in relevant part that "[n]o timber sale in Region 10 shall be advertised if the indicated rate is deficit when appraised using a residual value approach."[57] This provision has remained in Congress's subsequent appropriations legislation.[58]

Meanwhile, timber processed in Alaska was proving less competitive in the market. Due to higher transportation costs, and as well as higher processing costs where the export prohibition was applicable, Alaskan processed wood products suffered on the market. As a result, with downstream difficulty in marketing timber, few timber sales in the Tongass could receive a positive appraisal. Given this market situation and the applicable laws, few Tongass timber sales were advertised.

In March 2007, in response to the falling number of Tongass timber sales, Regional Forester Dennis Bschor partially authorized the export of unprocessed Tongass logs by means of what was termed the Limited Export Policy. Specifically, the Regional Forester authorized export of 50 per cent of Sitka spruce and western hemlock sawlogs under 15-inches in diameter at the

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[53] 16 U.S.C. § 616.

[54] *Id.*

[55] 36 C.F.R. § 223.201.

[56] *Id.*

[57] Pub. L. No. 108-7, § 318, 117 Stat. 11, 273 (2003).

[58] *See* DEIS at 3-481; Big Thorne FEIS at 3-32.

small end of a 40-foot log and grade 3 or 4 logs of any diameter.[59] In so authorizing, the Regional Forester incorporated the findings of Randy Coleman, Group Leader of Policy Analysis and Economics, and Dan Castillo, Director of Forest Management, that "Forest Service appraisals of proposed Tongass timber sales would rise if they assumed that a portion of the unprocessed spruce and hemlock logs were shipped to lower 48 markets."[60] Coleman and Castillo's findings concluded that export of 50 per cent of Sitka spruce and western hemlock logs should be approved because it optimally served four objectives: (1) ensuring the continuation of economic timber sales in the Tongass; (2) ensuring the continuation of wood-processing capacity in Alaska; (3) minimizing the shipment of unprocessed logs from Alaska; and (4) maximizing the Tongass timber industry's effects on U.S. employment.[61]

The Limited Export Policy is reviewed annually by the Regional Forester and has evolved over time. In each year since 2007, citing conditions in the domestic timber market, as well as the transitional status of Alaskan timber processors, the Regional Forester has reviewed and approved a continuation of the Limited Export Policy.[62] The 2007 decision to adopt the Limited Export Policy only authorized domestic interstate shipments.[63] In 2008, Regional Forester Bschor approved an expansion of the Limited Export Policy allowing for international export of unprocessed logs.[64] In 2012, the Regional Forester began reviewing case-by-case requests for exports even higher than routinely allowed by the policy.[65] The Regional Forester has reviewed and approved the Limited Export Policy with the international export component in each year

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[59] 2008 TLMP AR 603_1064 (D.E. Bschor, Memorandum to Forest Supervisor Re: Limited Interstate Shipments of Unprocessed Sitka Spruce and Western Hemlock Timber (Mar. 14, 2007) (Bschor 2007).

[60] 2008 TLMP AR 603_1777 (R. Coleman & D. Castillo, *Tongass Timber Appraisal Issues* (Feb. 1, 2007)) (Coleman & Castillo).

[61] *Id.* at 3.

[62] *See* D. Bschor, Memorandum to Forest Supervisor Re: Time Limited Shipment of Unprocessed Hemlock and Sitka Spruce (Aug. 8, 2008) (Bschor 2008); D. Bschor, Memorandum to Forest Supervisor Re: Extension of Limited Shipment of Unprocessed Hemlock and Sitka Spruce (Nov. 10, 2009) (Bschor 2009); B. Pendleton, Memorandum to Forest Supervisor Re: Update to R10 Limited Export Policy for hemlock and Spruce (Jan. 7, 2011); B. Pendleton, Memorandum to Forest Supervisor Re: Annual Review and Update to R10 Limited Export Policy (Jan. 11, 2012); B. Pendleton, Memorandum to Forest Supervisor Re: Annual Review and Update to R10 Limited Export Policy (Feb. 20, 2013) (Pendleton 2013); B. Pendleton, Memorandum to Forest Supervisor Re: Annual Review and Update to R10 Limited Export Policy (Feb. 28, 2014); PR 769_05_000351 (B. Pendleton, Memorandum to Forest Supervisor Re: Annual Review and Update to R10 Limited Export Policy (Apr. 21, 2015)) (Pendleton 2015).

[63] Bschor 2007.

[64] *See* Bschor 2008 (authorizing export to foreign markets with a premium of $25/MMBF for hemlock and $65 per MMBF for Sitka spruce).

[65] *See* Pendleton 2015.

since 2008, stating that "challenges continue,"[66] and citing "market conditions" including declining "softwood lumber indices,"[67] declining or "fluctuat[ing]" wood products prices,[68] and then, as "improvements . . . occurred nationally," that "recovery is slow and costly for Alaska markets."[69]

II. THE DEIS FAILS TO ADDRESS ALTERNATIVES IN WHICH THE LIMITED EXPORT POLICY IS NOT CONTINUED INDEFINITELY.

The DEIS fails to consider alternatives in which the Limited Export Policy is not continued unchanged and indefinitely.[70] Variations on the Limited Export Policy are not even included among the "alternatives eliminated from detailed review."[71] No explanation is given as to why these variations were not considered. The omission is improper given that participants in the scoping process specifically requested the Forest Service to discontinue the Limited Export Policy.[72]

Though the Limited Export Policy requires annual approval, the DEIS assumes that it will continue indefinitely, premising its analysis on the "assum[ption]" that "Western hemlock and Sitka spruce volumes . . . [will] be divided equally between domestic production and export in accordance with the current limited export shipment policy."[73]

By excluding variations on the Limited Export Policy, the DEIS excludes reasonable alternatives that fall within the range dictated by the plan amendment's "purpose and need," in violation of

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[66] Pendleton 2013.

[67] Bschor 2009.

[68] U.S. Forest Service, Issue Paper, *Tongass Limited Shipping Policy* (Feb. 2010); U.S. Forest Service, Issue Paper, *Tongass Limited Shipping Policy* (Apr. 2011); U.S. Forest Service, Issue Paper, *Tongass Limited Shipping Policy* (Apr. 2012); U.S. Forest Service, Issue Paper, *Tongass Limited Shipping Policy* (Apr. 2013); U.S. Forest Service, Issue Paper, *Tongass Limited Shipping Policy* (Apr. 2014).

[69] Pendleton 2015; U.S. Forest Service, Issue Paper, *Tongass Limited Export Policy* (Apr. 2015).

[70] DEIS at 2-8 to 2-36.

[71] DEIS at 2-5 to 2-8.

[72] Southeast Alaska Conservation Council *et al.*, Letter to Forrest Cole, Forest Supervisor, Scoping Comments in Response to Notice of Intent to Prepare an Environmental Impact Statement to Support an Amendment to the 2008 Tongass National Forest Plan at 3 (June 26, 2014) ("The outcome of any 'transition' alternative should reward local, value added manufacturing and end existing export and transshipment policies on the Tongass. . . . These policies need to be scrapped because they provide a disincentive for local mills to retool for typically smaller, second-growth trees.").

[73] DEIS at 3-481; *see also id.* at 3-484; Daniels at 10 (incorporating export policy into demand projections)).

NEPA.[74] The DEIS describes the plan amendment having five purposes: to determine land suitability for timber production, to identify the projected timber sale quantity and sustained yield limits for these lands, to establish plan components for second-growth forest management and renewable energy development, to disclose and assess direct and cumulative environment impacts from the amended Plan, and to consolidate modifications made to the Plan since its approval.[75] The Limited Export Policy has a strong effect on the projected timber sale quantity, which is based on the assumption of a continued export policy, and therefore on the environmental impacts of the plan. The annual renewals of the policy have also made it, in practical effect, a modification of the Plan since 2008 being consolidated into the proposed amended plan. For these reasons, the DEIS cannot lawfully ignore alternatives that would eliminate or modify the Limited Export Policy.

Inclusion of alternatives in which the Limited Export Policy does not continue in perpetuity is necessary for the comparative analysis of environmental consequences among reasonable alternatives, required by NEPA.[76] A comparison of alternatives that do and do not continue the Limited Export Policy would highlight important tradeoffs that are currently obscured by the agency's analysis. Allowing more export increases the volume of timber sold, which has corresponding adverse environmental effects and high financial costs to the public of a timber sale program that results in a loss to the treasury. The additional volume adds relatively few jobs, since the exported volume requires no local processing. Reasonable alternatives, therefore, would address a range of possibilities. At one end, with no export, there would be less logging but more jobs per unit of timber logged and greater protection of wildlife, biological diversity, carbon stores and carbon sequestration, the public fisc, subsistence uses, and optimizes the recreation, tourism, and fishing sectors of the economy. At the other end is the current policy, which emphasizes timber production with relatively few jobs and relatively high adverse impacts and costs on all other values. Intermediate options could also be considered. The failure to consider these options in an EIS violates NEPA.

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[74] *City of Carmel-By-The-Sea v. U.S. Dep't of Transp.*, 123 F.3d 1142, 1155 (9th Cir. 1997) ("Project alternatives derive from an Environmental Impact Statement's 'Purpose and Need' section, which briefly defines the underlying purpose and need to which the agency is responding in proposing the alternatives . . . [and which] necessarily dictates the range of reasonable alternatives." (quotation marks omitted)); *Idaho Conservation League v. Mumma*, 956 F.2d 1508, 1519 (9th Cir. 1992) (holding that failure to examine a reasonable alternative renders an EIS inadequate).

[75] DEIS at 1-4 to 1-5.

[76] *Ctr. for Biological Diversity v. U.S. Dep't of Interior*, 623 F.3d 633, 645, 648 (9th Cir. 2010) ("It is black-letter law that NEPA requires a comparative analysis of the environmental consequences of the alternatives before the agency."); *Methow Valley Citizens Council. v. Reg'l Forester*, 833 F.2d 810, 815 (9th Cir. 1987) (holding that an agency must consider a range of alternatives that is "sufficient to permit a reasoned choice"), *rev'd on other grounds, Robertson v. Methow Valley Citizens Council*, 490 U.S. 332 (1989).

III. CONTINUATION OF THE LIMITED EXPORT POLICY UNDER THE PLAN REQUIRES NEPA ANALYSIS.

To date, the agency has not conducted an analysis of the environmental impacts of the Limited Export Policy. However, such an analysis is required under NEPA.

The Regional Forester's implementation of the Limited Export Policy is a major federal action, because it entailed the "[a]doption of formal plans . . . upon which future agency actions will be based."[77] By the agency's own description, the adoption of the Limited Export Policy and its continued application via annual reviews are a "major development."[78] The Forest Service expects the Limited Export Policy "to boost appraised timber values,"[79] and, by its own account, but for this change in policy, far fewer timber sales "would appraise as positive."[80] A major increase in the number of sales appraising as positive—leading ultimately to more logging—was the Policy's raison d'etre.[81] As the adoption of a formal policy on which agency action would occur, the Limited Export Policy was a major federal action.

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As a major federal action, the Regional Forester's implementation of the Limited Export Policy requires NEPA analysis.[82] With respect to those alternatives in the DEIS in which the Limited Export Policy continues, the agency must provide a "full and fair discussion of significant environmental impacts" of the Limited Export Policy, including but not limited to resultant increases in logging, the proportion of the resultant logging that will occur in old-growth stands, and the impacts on ecosystems that will follow from the resultant logging.[83] The DEIS should also include alternatives that do not involve continuation of the Limited Export Policy in order to

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[77] 40 C.F.R. §§ 1508.18(b)(1)-(2); *see also* Forty Most Asked Questions Concerning CEQ's National Environmental Policy Act Regulations, 46 Fed. Reg. 18026, 18033 (Mar. 23, 1981) ("[T]he adoption of official policy in the form of rules, regulations and interpretations . . . establishing governmental or agency policy which will substantially alter agency programs [entail major federal actions].").

[78] Daniels at 12.

[79] DEIS at 3-453.

[80] Big Thorne FEIS at 3-32.

[81] *See* Coleman & Castillo at 2-3 ("[L]imited interstate shipments would significantly increase the likelihood that timber sales in parts of the Tongass would have a positive appraisal under current market conditions. . . . [A]llowing some [international] exports of spruce and hemlock logs would have an even more powerful positive effect on appraisals").

[82] 42 U.S.C. § 4332(2)(C) (stating that when a federal agency undertakes "major Federal actions significantly affecting the quality of the human environment," it is required to provide a "detailed statement" analyzing the "environmental impact of the proposed action" and "alternatives to the proposed action").

[83] *See Conservation Cong. v. Finley*, 774 F.3d 611, 616 (9th Cir. 2014).

properly conduct the requisite NEPA analysis. Under NEPA, the agency is required to develop alternatives that would "inform decisionmakers and the public of the reasonable alternatives which would avoid or minimize adverse impacts or enhance the quality of the human environment."[84]

SOCIOECONOMIC COSTS OF LOGGING

While the DEIS acknowledges that the Tongass timber sale program causes significant adverse effects to the environment, it makes no attempt to evaluate the economic costs of those adverse effects. By presenting only the ostensible benefits of logging in the form of jobs, economic activity, and revenue, while ignoring the economic costs, the DEIS presents a seriously skewed and misleading picture in violation of NEPA. These issues are explained in greater detail in the attached comments by Natural Resource Economics, Inc.[85] The undersigned parties hereby attach the Natural Resource Economics analysis in its entirety as our own comments.

The DEIS acknowledges that the Tongass timber sale program envisioned in the forest plan amendment will have significant adverse effects on subsistence uses, sport hunting, sport fishing, and wildlife viewing. There is extensive literature available, but disregarded by the DEIS, that discloses the economic value of these uses and sectors of the economy. For example, detailed information is readily available on the dollars spent for hunting and wildlife viewing trips.[86] The number of jobs from, and the economic value associated with, these activities dwarf those from the timber sale program and thereby reveals a substantial misallocation of resources by the Forest Service.

Perhaps even more significantly, contrary to federal policy, the DEIS makes no attempt even to address the social costs of the carbon dioxide released into the atmosphere as a result of logging. As discussed in Niemi, *Logging Costs* (2015), federal agencies including USDA and the Forest Service have, in other settings, calculated the social costs of carbon from federal land

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[84] 40 C.F.R. § 1502.1 (binding NEPA regulations provide that an EIS must "inform decisionmakers and the public of the reasonable alternatives which would avoid or minimize adverse impacts or enhance the quality of the human environment"); *Native Ecosystems Council v. U.S. Forest Serv.*, 418 F.3d 953, 965 (9th Cir. 2005).

[85] E. Niemi, *Socioeconomic Comments: Logging Costs* (Feb. 2015) (Niemi, *Logging Costs*).

[86] *See id.* at PDF 5-8; ECONorthwest, *The economic importance of Alaska's wildlife in 2011: Final report to the Alaska Department of Fish and Game, Division of Wildlife Conservation* (2014).

management actions.[87] A relatively simple estimate based on information in the DEIS and the record suggests that the climate-related costs from the proposed plan would be at least 2-8 times the value of the timber produced, and probably much greater.[88]

In short, the economic costs of the timber sale program vastly exceed its benefits. The failure of the DEIS to address this comparison in a candid way is a serious shortcoming, creating a highly misleading picture and violating NEPA.

PUBLIC COSTS

For Alternative 5 and most other alternatives, the DEIS displays high net revenues to the U.S. Treasury over most time horizons, including the short term.[89] This claim strongly contradicts recent experience of large taxpayer losses from the timber sale program. It is based on a substantial error, now admitted by the Forest Service, and requires significant revision and added explanation. The DEIS misleads readers in violation of NEPA.

In the five fiscal years from 2009 to 2013, the Forest Service spent an average of $22.3 million per year on the Tongass timber sale program and received an average of $1.7 million in revenues. The resulting annual loss to taxpayers was about $20.5 million.[90] Unfortunately, the DEIS contains no information on actual revenues or expenditures on the timber sale program. This is a significant omission that must be corrected.

The DEIS shows that over the same period of time, "Total Tongass-Related Employment" in the timber industry averaged 102 jobs.[91] Thus, each timber industry job created by the Tongass timber sale program cost taxpayers a little over $200,000. That is extremely poor value for the money. It is an issue that should be disclosed thoroughly and candidly in the DEIS, but it is nowhere to be found.

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[87] Niemi, *Logging Costs* at PDF 10-14; *see* Interagency Working Group on the Social Cost of Carbon, *Technical Update of the Social Cost of Carbon for Regulatory Impact Analysis – Under Executive Order 12866* (July 2015 (Revised)) https://www.whitehouse.gov/sites/default/files/omb/inforeg/scc-tsd-final-july-2015.pdf; U.S. Environmental Protection Agency, *EPA Fact Sheet: Social Cost of Carbon* (2015), http://www.epa.gov/climatechange/Downloads/EPAactivities/social-cost-carbon.pdf; U.S. Department of the Interior, Bureau of Land Management, *Draft resource management plan/environmental impact statement: Western Oregon* (2015); U.S. Forest Service, *Rulemaking for Colorado Roadless Areas: Supplemental Draft Environmental Impact Statement* (2015), http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd485194.pdf.

[88] Niemi, *Logging Costs* at PDF 11-12.

[89] DEIS at 3-481 to 3-483.

[90] PR 769_05_000671 at PDF 24 (Headwaters Economics, *The Tongass National Forest and the Transition Framework: A New Path Forward?* (Nov. 2014)) (Headwaters Report); *see also* Taxpayers for Common Sense, *Money-Losing Timber Sales: Tongass National Forest* (March 2015).

[91] *See* DEIS at 3-449, Table 3.22-4.

The DEIS, instead, reports projected "net revenues," both discounted and undiscounted, generated by the Woodstock model.[92] To pick the most immediate and relevant example, these charts show that, in the first five years of plan implementation, the Forest Service projects undiscounted net revenues of about $65 million on old-growth timber—or about $13 million per year—for Alternative 5, the preferred alternative. It shows slightly higher revenues for the no action alternative, which would be a continuation of the status quo.[93] These projections, the DEIS says, are "net of Forest Service costs that would be incurred for National Environmental Policy Act (NEPA) preparation, sale preparation and administration, and engineering support."[94]

Agency staff have now informed the undersigned parties that this statement in the DEIS is false. In reality, "the modeling did not include agency costs…."[95] Rather, the numbers presented as "net revenues" in the tables and figures on pages 3-481 to 3-483 actually represent the value to the *purchaser* (selling value minus costs for logging, transportation, manufacturing, profit, and risk).[96] They disclose nothing about value to the U.S. Treasury. In short, they are not "net," they are not "revenues," and they do not account for public costs at all.

The result of this error is a severely misleading DEIS. As a result of the error in the presentation, the DEIS claims that that the public can expect revenues, net of agency costs, in the range of hundreds of millions of dollars—a benefit to taxpayers if true. In reality, nothing could be farther from the truth. The Forest Service loses about $20 million per year on the timber sale program, and the DEIS offers no reason to believe that would improve under the Draft Forest Plan.

In the final EIS, it will not be sufficient merely to excise the references to agency costs. To correct the severely misleading information in the DEIS, it will be necessary for the final EIS to disclose, fully and frankly, actual past and expected future costs, revenues, and losses associated with the timber sale program.

Moreover, when the so-called "net revenues" are properly understood as value to purchasers, they must either be in error or reveal that Tongass timber sale purchasers are obtaining enormous, unjustified windfalls. The DEIS and the Howle email state that those numbers take into account all of the purchasers' costs plus "normal profit and risk."[97] If this is correct, those numbers reflect the prices the purchasers should be paying for Tongass timber. If they pay less,

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[92] DEIS at 3-481 to 3-483.

[93] DEIS at 3-482, Figure 3.22-17.

[94] DEIS at 3-481; *see also* DEIS App. B at B-19 (explaining that agency costs were used in calculating the net value).

[95] Howle, S., U.S. Forest Service, Email exchange with T. Waldo, Earthjustice, Re: TLMP record (Feb. 17-18, 2016).

[96] *See id.* (quoting DEIS at 3-481).

[97] *Id.*

they are obtaining a windfall above "normal profit and risk;" if they pay more, they are incurring a loss or an inadequate return on investment.

Forest Service records show the former: timber sale purchasers are paying far less than the value of the timber reported in the DEIS. As discussed above, the values described inappropriately as "net revenues," which actually represent value to purchasers, are projected to be, on average, $13 million per year for old-growth in the first five years under Alternative 5 (undiscounted). That alternative projects an annual cut of 37 mmbf old-growth during this five-year period.[98] Values are based on recent "Forest Service Region 10 appraisal rates."[99] Thus, the Forest Service should expect a price of about $351 per mbf old-growth.[100] This is vastly in excess of recent actual revenues, which have run, on average, about $57 per mbf:[101]

Fiscal Year	Value of Sold Timber ($)	Sold Timber (MBF)	$/MBF
2011	1,626,578.87	44,189.73	36.81
2012	1,506,779.60	52,482.61	28.71
2013	266,002.02	15,866.01	16.77
2014	8,151,761.96	105,523.27	77.25
2015	2,072,083.16	22,624.98	91.58
Total	**13,623,205.61**	**240,686.60**	
5-Year Average			**56.60**

This implies either that the numbers are wrong or that Tongass timber sale purchasers have been reaping a windfall worth nearly $300 per mbf. If the numbers are wrong, the Forest Service must correct them. If they are right, and purchasers are obtaining great windfalls, the Forest Service should be charging a great deal more for the timber it sells. In that case, the Limited Export Policy should be discontinued immediately, because there is ample value to require more local processing. In either case, though, the Forest Service needs to better explain its projections. The final EIS must fully and transparently disclose past and projected future revenues, logger costs, agency costs, prices, profit, and risk.

The public is entitled to this information. The fact that the Forest Service no longer preforms the Economic Efficiency analysis required by the 1982 planning rules is irrelevant. The cost to the

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[98] DEIS at 2-32.

[99] DEIS at 3-481.

[100] $13 million / 37,000 mbf = $351 per mbf.

[101] *See* U.S. Forest Service, Cut and Sold Reports, FY 2011-2015. The volume numbers reported by the Forest Service do not distinguish between old-growth and second-growth. Since the timber sale program until now has consisted almost exclusively of old-growth, the undersigned parties assume for purpose of these calculations that all of the reported logging volume has been old-growth, recognizing that some small fraction may have been second-growth.

public of the timber sale program is a critical piece of information, and programmatic decisions about the Tongass timber sale program cannot fairly be evaluated without it.

Neither the DEIS nor the record reveals what assumptions the Forest Service used for these values. The DEIS does not disclose or discuss them, displaying only the final model outcomes, and even those are misrepresented as discussed above. The record contains the spreadsheets used in the model runs, but they contain only rows of raw data that are not placed in sufficient context for members of the public to evaluate them. NEPA requires much more.

SUSTAINED YIELD LIMIT

The only hard limit the Draft Forest Plan imposes on logging levels is the "Sustained Yield Limit," which is fixed at 248 mmbf per year in all five alternatives.[102] However, this limit includes logging on lands not suited for timber production. This violates the National Forest Management Act, the Multiple-Use Sustained-Yield Act, and implementing regulations.

The NFMA provides, "The Secretary of Agriculture shall limit the sale of timber from each national forest to a quantity equal to or less than a quantity which *can be removed* from such forest annually in perpetuity on a sustained-yield basis…."[103] In turn, the 2012 planning regulations provide that forest plans must include components to ensure this requirement is met.[104] Those regulations further state, in no uncertain terms, "No timber harvest for the purposes of timber production may occur on lands not suited for timber production."[105] The Chief is required to adopt directives "for estimating the quantity of timber that *can be removed* annually in perpetuity on a sustained-yield basis, and exceptions, consistent with 16 U.S.C. 1611."[106] Thus, the statute and regulations plainly require that the plan must set a sustained yield limit on lands where timber "can be removed" on a sustained-yield basis, which is only those lands that are actually suitable for timber production.

The Draft Forest Plan, however, sets the Sustained Yield Limit based "on lands that *may be suitable* for timber production."[107] This follows the Directive adopted by the Chief, which directs the agency to calculate sustained yield based on "the amount of timber that could be produced on all lands that *may be suitable* for timber production, assuming all of these lands were managed to produce timber without considering other multiple uses or fiscal or

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[102] DEIS at 3-311 to 3-312. The Sustained Yield Limit is not actually mentioned in the Draft Forest Plan, but only in the DEIS. It was apparently intended to be part of the plan, though, and should be added to it as required by 36 C.F.R. § 219.11(d)(6).

[103] 16 U.S.C. § 1611(a) (emphasis added).

[104] 36 C.F.R. § 219.11(d)(6).

[105] *Id.* § 219.11(d)(1).

[106] *Id.* § 219.11(d)(6)(iii) (emphasis added).

[107] DEIS at 3-311 (emphasis in original).

organizational capability."[108] The Directive thus requires calculation of sustained yield assuming logging on lands not suitable for timber production and without regard to multiple use. That violates NFMA's requirement to limit harvest to the sustained yield of timber that "can be removed,"[109] as well as the Multiple-Use Sustained-Yield Act.

The error makes a big difference in the Draft Forest Plan. The proposed plan identifies 942,592 acres that "may be suited for timber production," but only 485,574 acres that are actually suitable.[110] To achieve the Sustained Yield Limit over time would require logging on over 450,000 acres deemed not suited for timber production in order to ensure multiple use as well as ecological and economic sustainability under the 2012 forest planning regulations. The amended forest plan should recalculate the Sustained Yield Limit based only on lands suitable for timber production.

CLIMATE CHANGE AND CARBON STORAGE

The Tongass is America's most important carbon forest. It is the largest single forest-carbon sink in the United States, storing hundreds of millions, if not over a billion, tons of carbon. Within the forest, old-growth stands account for most of the Tongass's net carbon contribution. As such, it should take a specially protected place within the Federal Government's efforts to address climate change. The United States recently joined governments from around the world in Paris for negotiations on a new agreement under the United Nations Framework Convention on Climate Change. The resulting agreement specifically highlighted the role of forest protection in the effort to mitigate climate change. The Paris Agreement is only one instance in which the Federal Government has committed to protect forests to this end. Given these commitments, it is unacceptable for the Service to countenance the continued large-scale destruction of its most carbon-valuable old-growth stands in America's most important carbon forest.

To meet the United States' carbon reduction commitments, the Service should adopt an alternative that minimizes old-growth destruction by transitioning out of old-growth logging rapidly and no later than five years. The destruction of old-growth stands releases the huge stores of carbon currently captured in centuries-old root systems and undisturbed forest soils—a loss that cannot be offset by forest regeneration in the timeframe relevant to climate change. Only by transitioning out of old-growth logging more rapidly than any of the alternatives in the DEIS can the Federal Government act as a leader and exemplar in the global effort to mitigate climate change.

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[108] Forest Service Handbook 1909.12.64.31 (emphasis in original).

[109] 16 U.S.C. § 1611(a).

[110] Draft Forest Plan App. A at A-2, Table A-1.

I. THE DEIS FAILS TO CONTEXTUALIZE TONGASS-SPECIFIC CARBON SEQUESTRATION AND STORAGE WITHIN THE CONSENSUS—RECOGNIZED BY THE FEDERAL GOVERNMENT—THAT FORESTS PLAY A CENTRAL ROLE IN CLIMATE-CHANGE MITIGATION.

The DEIS fails to situate the Draft Forest Plan within a global context in which the United States has taken a leading role in identifying forest protection as a policy goal to mitigate climate change. Forest preservation plays a central role in climate-change mitigation. Global forests' cumulative annual carbon uptake is around 2.3 billion metric tons, and their net effect on the carbon cycle is to function as a sink of around 1.2 billion metric tons of carbon per year (measured between 2000 and 2007).[111] Temperate forests alone, totaling around 767 million hectares, sink around 800 million metric tons of carbon per year.[112] Relative to global CO_2 emissions, in the past few decades forests have absorbed around 30 per cent of anthropogenic CO_2 emissions.[113] Forests have climate benefits beyond the indirect contribution via carbon sequestration and storage. For example, forests may directly mitigate warming by means of their transpiration, which contributes to the formation of clouds that reflect sunlight and thus have a cooling effect.[114]

When a forest is logged, along with opportunity costs from foregone sequestration and storage (as described above), forest carbon stores are released, making the disturbed forest a carbon source.[115] Emissions from deforestation are estimated to total 5.8 gigatonnes of CO_2 annually.[116]

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[111] Y. Pan *et al.*, *A Large and Persistent Carbon Sink in the World's Forests*, 333 SCIENCE 988, 989 (2011) (Pan).

[112] *Id.*

[113] V. Bellassen & S. Luyssaert, *Managing Forests in Uncertain Times*, 506 NATURE 153, 153 (2014).

[114] M. G. Ryan *et al.*, *A Synthesis of the Science on Forests and Carbon for U.S. Forests*, in ISSUES IN ECOLOGY REPORT NUMBER 13, at 5 (Spring 2010) (Ryan); *see also* H. Salwasser, *Introduction: Forests, Carbon and Climate—Continual Change and Many Possibilities*, in FOREST, CARBON AND CLIMATE CHANGE: A SYNTHESIS OF SCIENCE FINDINGS at 4 (C. R. Millar *et al.* eds., 2006) (Salwasser) ("Forests play important roles in climate through other mechanisms in addition to carbon exchange. These mechanisms may be as or more important than that of carbon exchange. The massive amounts of water transpired by forests ultimately change the global distribution of energy in the atmosphere, affecting rainfall patterns, cloudiness, and storms.").

[115] O.N. Krankina & M. E. Harmon, *Forest Management Strategies for Carbon Storage*, in FOREST, CARBON AND CLIMATE CHANGE: A SYNTHESES OF SCIENCE FINDINGS at 83 (2006) ("Disturbance events such as [inter alia] timber harvest have a profound impact on the carbon balance of forest ecosystems. . . . As the carbon uptake by living trees is interrupted and the emissions from decomposition increase, a disturbed forest stand shifts from sink to source of carbon relative to the atmosphere.") (Krankina & Harmon).

By way of comparison, global road transport accounts for about 6.3 gigatonnes of CO_2 emissions per year.[117] "Globally, deforestation (8-15%) and forest degradation (6-13%) contribute significant amounts to the world's annual greenhouse gas pollutants."[118] The carbon release associated with deforestation results from the loss of carbon stored in wood, as well as from heightened release of CO_2 via decomposition of materials that had been under the forest floor.[119] As a document in the Service's record concludes, "the U.S. continues to lose forests to development . . . with projected net losses of up to 23 million acres by 2050. . . . This trend needs to be reversed if forests are to play positive roles in carbon storage and climate."[120]

The Federal executive has acknowledged the central role of forest preservation in the mitigation of climate change. A study led by the Service in 2011, estimated the stock of carbon stored in forests globally to be 861 billion metric tons, with 44% of the carbon stored in soil and 42% in live trees.[121] The EPA estimated that, in 2008, U.S. forests' net carbon flux (excess of uptake over release) was about 792 million metric tons of carbon dioxide equivalent.[122]

The importance of protecting forest thus has been highly prioritized in the executive's efforts to mitigate climate change. As stated in the President's 2013 Climate Action Plan, "America's forests play a critical role in addressing carbon pollution, removing nearly 12 per cent of total U.S. greenhouse gas emissions each year. . . . Conservation and sustainable management can help to ensure that our forests continue to remove carbon from the atmosphere . . ."[123] The crucial role of forests in urgent efforts to mitigate the harmful effects of anthropogenic climate change was also emphasized in the Paris Agreement under the United Nations Framework Convention on Climate Change, which the United States signed. As stated in Article 5(1) of the

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[116] Intergovernmental Panel on Climate Change, CLIMATE CHANGE 2007: MITIGATION. CONTRIBUTION OF WORKING GROUP III TO THE FOURTH ASSESSMENT REPORT OF THE INTERGOVERNMENTAL PANEL ON CLIMATE CHANGE at 543 (B. Metz *et al.*, eds, 2007) ("Emissions from deforestation in the 1990s are estimated at 5.8 GtCO2/yr.").

[117] *Id.* at 325.

[118] D. A. DellaSala, *Enlisting the Tongass National Forest in the Historic Global Climate Change Agreements* at 6 (2016).

[119] PR 769_05_000073 (M. E. Harmon *et al.*, *Effects on Carbon Storage of Conversion of Old-Growth Forests to Young Forests*, 247 SCIENCE 699, 700 (1990)) (Harmon).

[120] Salwasser at 13.

[121] Pan at 989.

[122] U.S. Environmental Protection Agency, *Forest Carbon Storage* at 1 (2010), *available at* http://www3.epa.gov/climatechange/wycd/waste/downloads/forest-carbon-storage10-28-10PDF.

[123] Exec. Office of the President, The President's Climate Action Plan at 11 (June 2013), *available at* https://www.whitehouse.gov/sites/default/files/image/president27sclimate actionplanPDF (President's Climate Action Plan).

Agreement, as part of the efforts to mitigate climate change, "Parties should take action to conserve and enhance . . . sinks and reservoirs of greenhouse gases . . . including forests."[124]

Notwithstanding the prevailing scientific consensus, as well as recognition and prioritization by the Federal executive, the DEIS fails to recognize forests' crucial role in global carbon sequestration and storage. In Chapter 3, the DEIS discusses each alternative's impact on the Tongass's ability to sequester carbon dioxide, and thus contribute to the mitigation of harmful effects of anthropogenic climate change. The DEIS's discussion is inadequate because it fails to afford due importance to the climate-change mitigation contributions of forests in general, and of the Tongass in particular. In the FEIS, the Service should situate its evaluation of Plan alternatives within the scientific literature and discussion of the important role of undisturbed forest preservation by other Federal actors, including by the Service itself.

II. THE DEIS FAILS TO SITUATE DISCUSSION OF ALTERNATIVES' CARBON EFFECTS WITHIN THE CONSENSUS THAT THE TONGASS PLAYS AN OUTSIZED ROLE IN FOREST CLIMATE-CHANGE MITIGATION EFFORTS.

The Service must recognize that the Plan stands as the nation's policy choice with respect to its most important carbon forest, and thus is a crucial expression of our country's policy towards climate change generally. Forests of the Western United States, and specifically the Tongass, figure largely in global forests' contribution to climate-change mitigation.[125] In general, rainforests are crucially important to the carbon cycle, but among global rainforests, northern temperate rainforests play a predominant role in rainforest carbon sequestration: "Because the tropical gross deforestation emission is mostly compensated by the [carbon] uptakes in both tropical intact and regrowth forests, the net global forest [carbon] sink [of roughly 1.1 billion metric tons of carbon per year] resides mainly in the temperate and boreal forests."[126] Northern coastal temperate rainforest biomes are "disproportionately important in regional carbon cycling."[127] Forests of the Pacific Northwest and Southeast Alaska store "exceptional levels" of carbon, and "are among the most [carbon] dense ecosystems in the world.[128] Moreover, "[t]he national forests of the Pacific Northwest Region . . . have a higher proportion of forests in old

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[124] Paris Agreement under the United Nations Framework Convention on Climate Change art. 5(1) at PDF 22 (opened for signature Apr. 22, 2016).

[125] Ryan at 5-7.

[126] Pan at 992.

[127] PR 769_05_000065 (D. D'Amore & R. Edwards, *Climate and Carbon in Southeast Alaska: Beyond the Threshold of Change in a Dynamic Landscape* (2014)) (D'Amore & Edwards).

[128] O. N. Krankina *et al.*, *High-Biomass Forests of the Pacific Northwest: Who Manages Them and How Much is Protected?*, 54 ENVIRONMENTAL MANAGEMENT 112, 113 (2014); A. N. Gray & T. R. Whittier, *Carbon stocks and Changes on Pacific Northwest National Forests and the Role of Disturbance, Management, and Growth*, 328 FOREST ECOLOGY & MANAGEMENT 167, 168 (2014) ("The national forests of the Pacific Northwest Region attain some of the highest [carbon] densities in the U.S.") (Gray & Whittier).

age classes (>100 years) than other ownerships,"[129] and "Southeast Alaska . . . boasts nearly a third of the world's old-growth temperate rainforest."[130]

The Tongass is the largest single forest carbon sink in the United States, storing hundreds of millions, if not over a billion tons of carbon. As the Service has recognized elsewhere, the Tongass is the largest intact area of coastal temperate rainforest biome remaining globally.[131] According to one study, "[t]he Tongass National Forest stores substantially more forest carbon than any other national forest in the United States."[132] The Tongass annually removes about 2,787 pounds of atmospheric CO_2 per acre per year through growth and recruitment.[133] Looking only at aboveground carbon, one study found that the Tongass stores "about 650 million tons in aboveground tree carbon, equivalent to 2.4 billion tons of CO_2."[134] The study estimated that below-ground carbon pools "could be as large as the aboveground stores."[135] A 2005 study found that aboveground carbon constitutes around 40 percent of total carbon stored in the Tongass, with a conservative estimate that 66 per cent of the total was found in soils and 4 per cent in roots,[136] a distribution consistent with carbon inventories in other ecosystems.[137] Extrapolating from aboveground carbon figures in the Tongass and from its relative supra- versus sub-terrestrial carbon distribution, the total carbon store of the Tongass would around 1.625 billion tons of carbon.

The DEIS's discussion of each alternative's impact on the Tongass's ability to sequester carbon dioxide is inadequate because it ignores the Tongass's outsized role in forest climate-change mitigation. To the extent that the DEIS discusses the Tongass's place in global forests' carbon mitigation, it ignores the Tongass's outsized role. The DEIS recognizes that the Tongass's carbon stock accounts for 8 percent of the total carbon stored in U.S. forests.[138] However, by relying merely on this figure, the Service fails to capture the full importance of the Tongass's

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[129] Gray & Whittier at 168.

[130] D. A. DellaSala *et al*., *Temperate and Boreal Rainforests of the Pacific Coast of North America, in* TEMPERATE AND BOREAL RAINFORESTS OF THE WORLD: ECOLOGY AND CONSERVATION at 42 (D. A. DellaSala ed., 2011) (DellaSala).

[131] U.S. Forest Service, *Carbon Cycle Science: Quantifying the Carbon Cycle and Potential for Carbon Sequestration on the Tongass National Forest* at 1 (2010) (Carbon Cycle Science).

[132] PR 769_05_000062 at 39 (T. M. Barrett, *Storage and Flux of Carbon in Live Trees, Snags, and Logs in the Chugach and Tongass National Forests* (2014)).

[133] *Id.* at 39.

[134] *Id*.

[135] *Id.* at 36.

[136] W. W. Leighty *et al.*, *Effects of Management on Carbon Sequestration in Forest Biomass in Southeast Alaska*, ECOSYSTEMS 1051, 1059 (2006).

[137] *Id.* at 1062.

[138] DEIS at 3-14.

role as a carbon forest within the nation's leadership in global climate change policy, as discussed below. Moreover, even on the numbers, the DEIS is mistaken. The DEIS states that Southeast Alaska as a whole accounts for 0.5 percent of the world's forests,[139] a figure contradicted by the only published global study—supported by GIS and computer modeling—which found that the Tongass alone had an eight-fold larger share of the global total.[140]

In the FEIS, the Service should evaluate Plan alternatives in light of prevailing scientific literature that quantifies the stores of carbon within the Tongass, as well as assesses the Tongass's relative global importance as a forest carbon sink and store.

III. THE DEIS IGNORES THE DIFFERENTIAL ABILITY OF OLD-GROWTH FOREST TO SEQUESTER CARBON DIOXIDE AND STORE CARBON WITHIN THE TIMEFRAME RELEVANT TO CLIMATE CHANGE MITIGATION.

Scientific literature reflects that protection of America's carbon forests necessarily entails protection of old-growth stands. For this reason, the Service must recognize that the unnecessary continuation of old-growth logging would repudiate of the Federal Government's commitment to pursuing forest protection in addressing climate change.

Contrary to previous beliefs, forests continue to sequester carbon for centuries after they have become old-growth.[141] Studies have found that primary forest in the boreal and temperate regions of the northern hemisphere "alone sequester about 1.3 +/- 0.5 gigatonnes of carbon per year."[142] Pacific Northwestern forests were found to increase in biomass even at 300 and 600 years of age.[143] These results demonstrate that, although a tree's rate of carbon absorption might decline beyond 80 years of age, "old-growth forests can continue to accumulate carbon."[144]

Logging an old-growth forest results in a net carbon release; this is true notwithstanding the effects of limited carbon storage in wood products and growth of young forest in its place. As a

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[139] DEIS at 3-13.

[140] D. A. DellaSala *et al*., *Crosscutting Issues and Conservation Strategies, in* TEMPERATE AND BOREAL RAINFORESTS OF THE WORLD: ECOLOGY AND CONSERVATION at 247-49, Table 10-1 (D. A. DellaSala ed., 2011).

[141] *See* K. Lorenz & R. Lal, CARBON SEQUESTRATION IN FOREST ECOSYSTEMS at 120 (2009) ("The [carbon] balance of undisturbed, pristine forests is not in equilibrium as previously hypothesized but these forests continue to sequester carbon. Thus pristine old-growth forests are important components of the global terrestrial [carbon] budget.") (Lorenz & Lal).

[142] S. Luyssaert *et al.*, *Old-growth Forests as Global Carbon Sinks*, 455 NATURE 213 at 213 (2008) (Luyssaert).

[143] Lorenz & Lal at 120.

[144] Luyssaert at 213.

result of disturbing old-growth forest, forest carbon storage is reduced "for at least 250 years."[145] Although young forests grow relatively quickly, "the creation of new forests (whether naturally or by humans) frequently follows disturbance to soil and the previous vegetation, resulting in a decomposition rate of coarse woody debris, litter and soil organic matter (measured as heterotrophic respiration) that exceeds the [net primary productivity] of the regrowth."[146] In other words, when old-growth forest is logged and replaced by young forest, the young trees' capture of CO_2 in aboveground carbon stores is offset by the more rapid and voluminous release of carbon hitherto stored underground.

While recognizing detrimental carbon effects of forest disturbance, specifically from old-growth logging, the DEIS inaccurately suggests that resulting carbon losses are uncertain or could be offset. The DEIS recognizes that "timber harvesting activities . . . can result in the release of greenhouse gases" and "can . . . affect the rate and amount of carbon sequestration that occurs on the National Forest."[147] Specifically, carbon losses are likely to result from old-growth logging, which "creates an initial net release of CO_2 into the atmosphere relative to leaving stands unmanaged, [which] can continue for years as long as logs and snags left after harvest decompose."[148] But, while logging would in some circumstances reduce the Tongass's carbon sequestration, the DEIS hypothesizes that this reduction might be offset by uses of wood products or would vary depending on the timeframe considered.[149]

The DEIS's analysis is inaccurate because it ignores the differential ability of old-growth forest to sequester carbon dioxide and store carbon within the timeframe relevant to climate change mitigation. The DEIS is correct that not only old-growth forests sequester and store carbon: all else equal, forests between the ages of 15 and 800 years usually function as carbon sinks.[150] But old-growth forests store far more carbon than young forests.[151] Old-growth stores entail not only

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[145] Harmon at 700; *see also* PR 769_05_000091 (J. E. Janisch & M. E. Harmon, *Successional changes in live and dead wood carbon stores: implications for net ecosystem productivity*, 22 TREE PHYSIOLOGY 77 (2002)) (Janisch & Harmon). Studies more particularly focused on temperate and boreal rainforests of the Pacific Northwest put the progression to old-growth at 150-400 years. DellaSala at 49.

[146] Luyssaert at 213; Krankina & Harmon at 85 ("Following timber harvest, carbon emissions from decomposing slash usually exceed carbon accumulation in young trees (in spite of their vigorous growth) for about a decade.").

[147] DEIS at 3-19.

[148] *Id*.

[149] *Id*. at 3-20 ("Alternatives that harvest more old-growth forests compared to young-growth and/or have a larger total projected harvest would have a greater effect to the potential carbon sequestration of the Tongass National Forest compares to alternatives that harvest less timber.").

[150] Luyssaert at 213.

carbon stored in the larger mass of trees, but forest soil carbon, which "will move back to the atmosphere if these forests are disturbed."[152] As one study put it, "[a]lternative management schemes never match old-growth when evaluated based on the time integral of total carbon in live and dead organic matter."[153]

The DEIS's positing of a hypothetical offset via storage in wood products contradicts studies on the subject. Even when storage of carbon in wood products is included in models, logging results in a net flux of CO_2 to the atmosphere,[154] because storage of aboveground carbon in wood products suffers inefficiencies in processing. "At least 15% of the wood fiber in a typical harvest is left behind as broken or defective."[155] Subsequent to the logging, "[a]s much as 50% of the harvested material is released to the atmosphere within a few years."[156] Synthesizing data from multiple studies, Ann Ingerson found that "as little as 1% of the carbon present in the standing tree may remain in solid wood products in use after 100 years."[157] Use of wood products might have beneficial effects only when substituting for sufficiently more carbon-intensive concrete or metal products.[158] However, the DEIS has not made—let alone substantiated—any claim that such substitution in fact occurs. Moreover, it has not addressed whether these substitutions would necessarily occur at levels sufficient to ensure that the net carbon effect would constitute climate-change mitigation.

The DEIS's equivocation about relevant timescale is also misleading and contrary to prevailing scientific consensus. The DEIS's analysis is explicitly premised on the treatment of timescale as an undefined variable, the uncertainty of which renders the Plan alternatives' net carbon effects unknowable. As the DEIS explains, "[e]stimating the effects of the proposed alternatives on climate change and carbon sequestration is complex," because, *inter alia*:

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[151] I. Thompson *et al.*, FOREST RESILIENCE, BIODIVERSITY AND CLIMATE CHANGE, A SYNTHESIS OF THE BIODIVERSITY/RESILIENCE/STABILITY RELATIONSHIP IN FOREST ECOSYSTEMS, Secretariat of the Convention on Biological Diversity, Montreal, Technical Series No. 43 at 7, 21, 39, 41 (2009).

[152] Luyssaert at 213.

[153] C. B. Field & J. Kaduk, *The Carbon Balance of an Old-Growth Forest: Building Across Approaches*, 7 ECOSYSTEMS 525, 532 (2004).

[154] Harmon at 699.

[155] *Id.*

[156] Krankina & Harmon at 85.

[157] A. Ingerson, *Wood Products and Carbon Storage: Can Increased Production Help Solve the Climate Crisis?* at 5 (2009).

[158] *See* DEIS at 3-15; PR 769_05_000072 (J. Perez-Garcia *et al.*, *An Assessment of Carbon Pools, Storage, and Wood Products Market Substitution Using Life-Cycle Analysis Results*, 37 WOOD AND FIBER SCIENCE 140 (2005)).

> [S]equestration and carbon release happen at different time scales. . . . [D]ifferent authors have reached widely different conclusions about net sequestration because of different assumptions about [*inter alia*] the timeframe of interest. . . . Each of the harvesting alternatives has the potential to either increase or decrease the amount of stored carbon, depending on the time scale of consideration.[159]

Studies affirm the determinative role timescale plays in assessing the net changes in the carbon cycle,[160] but explicitly state that the relevant timescale is under 100 years—a period shorter than the time required for a forest to reestablish equilibrium after disturbance. The Intergovernmental Panel on Climate Change's analysis indicates that in order to avoid a global average surface temperature increase (relative to pre-industrial levels) of 2°C, the atmospheric concentration of CO_2 in the year 2100 will have to be around 450 ppm.[161] Such a concentration can only be achieved, according to the IPCC, if "substantial cuts in anthropogenic GHG emissions" occur "by mid-century through large-scale changes in energy systems and potentially land use."[162] With regard to the timing of these necessary "large-scale changes," the IPCC found that "[d]elaying mitigation efforts . . . through 2030 is estimated to substantially increase the difficulty of the transition to low longer-term emissions levels and narrow the range of options consistent with maintaining temperature change below 2°C relative to pre-industrial levels."[163] The IPCC's projections are known to be conservative with respect to the estimated pace of global warming, so that it would be reasonable to believe that necessary large-scale changes

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[159] DEIS at 3-19 to 3-20.

[160] Janisch & Harmon at 77.

[161] Intergovernmental Panel on Climate Change, *Climate Change 2014: Mitigation of Climate Change*, Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change at 10 (O. Edenhofer *et al.* eds, 2014) (Climate Change 2014 Working Group III Report).

[162] *Id.*

[163] *Id.* at 12.

would in fact be needed well before 2030 in order for mitigation to succeed.[164] The relevant timescale of climate-change mitigation measures is thus the next 10 to 15 years.

In the FEIS, the Service must correct the DEIS's mistake of treating timescale as an undefined variable. The DEIS ignores scientific consensus that "large-scale changes" in land use must occur within 10 to 15 years. As the DEIS recognizes, after logging, a forest takes around 200 years to re-establish equilibrium.[165] The re-attainment of similar levels of carbon storage can take even longer.[166] Therefore the timescale to mitigate climate change is shorter—by more than an order of ten—than the period during which a disturbed old-growth stand can return to equivalent levels of carbon storage and sequestration.

IV. THE DEIS'S COMPARISON OF THE TONGASS'S CARBON BALANCE TO THE TOTAL GLOBAL CARBON CYCLE MISREPRESENTS THE CENTRALITY OF THE TONGASS TO THE UNITED STATES'S EFFORTS TO LEAD GLOBAL CLIMATE CHANGE POLICY.

The Federal Government has taken a leadership role in global efforts to mitigate climate change[167] and has recognized forest policy's importance in these efforts. Specifically, the Administration has highlighted the importance of "[c]onservation and sustainable management . . . to ensure our forests continue to remove carbon from the atmosphere while also improving soil and water quality, reducing wildfire risk, and otherwise managing forests to be more resilient in

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[164] K. Brysse*, et al.*, *Climate Change Prediction: Erring on the Side of Least Drama?*, 23 GLOBAL ENVT'L CHANGE 327 (2013); W. R. L. Anderegg *et al.*, *Awareness of Both Type 1 and 2 Errors in Climate Science and Assessment*, 95 BULLETIN OF THE AMERICAN METEOROLOGICAL SOCIETY 1445 (2014); *see also* C. Mooney, *The world's climate change watchdog may be underestimating global warming*, WASHINGTON POST (Oct. 30, 2014) https://www.washingtonpost.com/news/wonk/wp/2014/10/30/climate-scientists-arent-too-alarmist-theyre-too-conservative/ ("According to a number of scientific critics, the scientific consensus represented by the IPCC is a very conservative consensus. IPCC's reports, they say, often *underestimate* the severity of global warming, in a way that may actually confuse policymakers (or worse). . . . [I]n a new study just out in the *Bulletin of the American Meteorological Society*, another group of researchers echoes that point. In scientific parlance, they charge that the IPCC is focused on avoiding . . . false positive[s]—rather than on avoiding . . . false negative[s]. The consequence is that we do not always hear directly from the IPCC about how bad things could be."); G. Scherer, *Climate Science Predictions Prove Too Conservative*, SCIENTIFIC AMERICAN (Dec. 6, 2012) *available at* http://www.scientificamerican.com /article/climate-science-predictions-prove-too-conservative/ ("Across two decades and thousands of pages of reports, the world's most authoritative voice on climate science has consistently understated the rate and intensity of climate change and the danger those impacts represent, say a growing number of studies on the topic.").

[165] DEIS at 3-15; *see also* Janisch & Harmon at 85 ("Our data indicate that regrowth biomass approximates that found in old-growth forests by about 200 years after disturbance.").

[166] Harmon at 700 ("Harvest of old-growth forests reduced C storage for at least 250 years.").

[167] *See* President's Climate Action Plan at 5.

the face of climate change."[168] As part of its prioritization of policy to mitigate climate change, the Council on Environmental Quality (CEQ) has cautioned against agencies using comparisons of emissions from a government action to total global emissions. According to the CEQ's Draft Guidance for Greenhouse Gas Emissions and Climate Change Impacts, "these comparisons are not an appropriate method for characterizing the potential impacts associated with a proposed action . . . [because t]his approach does not reveal anything beyond the nature of the climate change challenge itself: the fact that diverse individual sources of emissions each make relatively small additions to global atmospheric GHG concentrations that collectively have huge impact."[169]

In this context, the DEIS's comparison of the Tongass's 650 million tons in above-ground carbon stores to the overall magnitude of the carbon cycle, purportedly "83,500,000 billion metric tons,"[170] is misguided for several reasons. First, the DEIS's figure for the overall size of the carbon cycle is orders of magnitude off the mark: according to the Intergovernmental Panel on Climate Change, the total magnitude of the carbon cycle is more like 44,750 billion metric tons.[171] If the Forest Service retains this comparison in the FEIS, it should correct this error.

Second, even as a matter of evaluating the Tongass's quantitative contribution to climate change mitigation, the DEIS's comparison is not illuminating. If a purely quantitative evaluation were sufficient, it would be more relevant to compare the Tongass's sequestration and storage levels and the specific increment of anthropogenic contributions to the carbon cycle that would raise global average temperature over 2°C above pre-industrial levels, i.e. the increment that must be avoided to mitigate the harms of anthropogenic climate change.

Most importantly, however, a quantitative evaluation of the emissions savings or losses attributable to the Plan is insufficient and off-the-mark as a general matter. As the CEQ's draft guidance describes, the DEIS's quantitative evaluation of the Tongass's importance to climate change mitigation is misleading and "does not reveal anything beyond the nature of the climate change challenge itself."[172] Moreover, a purely quantitative account obscures the Plan's impacts on the implementation of the Federal Government's heralded commitment to climate change mitigation. Having recognized that "America's forests play a critical role in addressing carbon pollution,"[173] and that the Tongass is America's most important carbon forest, the Government

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[168] *Id*. at 11.

[169] Council on Environmental Quality, Revised Draft Guidance for Greenhouse Gas Emissions and Climate Change Impacts at 9 (Dec. 18, 2014), *available at* https://www.whitehouse.gov/sites/default/files/docs/nepa_revised _draft_ghg_guidance_searchable.pdf (CEQ Revised Draft Guidance).

[170] DEIS at 3-14.

[171] *See* Intergovernmental Panel on Climate Change, Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change at 515 fig. 7.3 (S. Solomon *et al.* eds, 2007).

[172] *See* CEQ Revised Draft Guidance at 9.

[173] President's Climate Action Plan at 11.

would contradict its pronouncements in practice by adopting a Plan that chooses to unnecessarily prolong old-growth logging in the Tongass—releasing deep stores of above- and below-ground carbon into the atmosphere, and reducing the Tongass's resilience to climate change.

V. THE DEIS FAILS TO ADDRESS HOW BASELINE ESTIMATES OF CARBON STORED IN FORESTS AND WOOD PRODUCTS WILL CHANGE UNDER THE VARIOUS ALTERNATIVES.

The DEIS fails to consider the Forest Service's recent whitepaper, *Baseline Estimates of Carbon Stocks in Forests and Harvested Wood Products for National Forest System Units, Alaska Region.*[174] In this report, the agency provides baseline carbon stocks and trends for seven different forest ecosystem carbon pools: above-ground live tree, below-ground live tree, standing dead, understory, down dead wood, forest floor and soil organic carbon."[175] It also provides estimates of carbon stored in wood products over longer time periods.[176] Inexplicably, the DEIS relies on a similar whitepaper prepared for the Rocky Mountain Region.[177] The FEIS should explain how the various alternatives, including the selected alternative, affect the amount of stored carbon on the Tongass.[178]

In lieu of an explanation, the DEIS simply states that "[e]stimating the effects of the proposed alternatives on climate change and carbon sequestration is complex."[179] The agency acknowledges "there is a substantial amount of recent literature about the effects of forest management on carbon stores," but the DEIS fails to include reference to any of those publications or explain agency's the conclusions regarding those scientific opinions.[180] Instead, it blandly states:

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[174] U.S. Forest Service, Baseline Estimates of Carbon Stocks in Forests and Harvested Wood Products for National Forest System Units; Alaska Region (March 2015), *available at* http://www.fs.fed.us./climatechange/documents/Alaska RegionCarbonAssessment.pdf (USFS Baseline Carbon Stocks).

[175] *Id*. at PDF 6.

[176] *Id*.

[177] *See* DEIS at 6-47.

[178] *See* USFS Baseline Carbon Stocks at PDF 14 (Figure 4).

[179] DEIS at 3-19.

[180] DEIS at 3-19.

> Each of the harvesting alternatives has the potential to either increase or decrease the amount of stored carbon, depending on the time scale of consideration; how much of the wood is removed/harvested from the Forest; how the wood is used; and how much carbon is released in cutting, yarding, transporting, and processing the wood as well as in soil carbon and woody debris decomposition (see discussion above, in the "Affected Environment" section).[181]

NEPA does not allow the Forest Service to avoid analyzing important issues, simply because the agency concludes they are "complex."

At a minimum, when the agency confronts incomplete or unavailable information as part of the environmental review process, NEPA regulations dictate how the agency must address that information.[182] "[T]he agency shall include the information in the environmental impact statement," if the missing information is: (1) "relevant to reasonably foreseeable significant adverse impacts;" (2) "essential to a reasoned choice among alternatives;" and (3) "the overall costs of obtaining it are not exorbitant."[183] The Council on Environmental Quality (CEQ) has explained that "[t]he evaluation of impacts under § 1502.22 is an integral part of an EIS and should be treated in the same manner as those impacts normally analyzed in an EIS."[184]

VI. THE DEIS IGNORES CHANGES TO THE TONGASS'S ABILITY TO SEQUESTER AND STORE CARBON THAT ARE EXPECTED TO RESULT FROM CLIMATE CHANGE.

The DEIS recognizes that "climate change and its projected warming trend may . . . affect subsurface carbon sequestration in Southeast Alaska."[185] However, the DEIS fails to consider specific processes relevant to the Tongass's carbon contribution that could be changed as a result of climate change.

A. The DEIS Fails to Account for Accelerated Decomposition Rates.

Although recognizing the importance of current low decomposition rates to the Tongass's carbon balance, the DEIS neglects the changes to these rates that will result from warming average temperatures. In its description of the Tongass's contributions to carbon storage, the DEIS states that cool conditions in the Tongass inhibit decomposition, and hence slow release of CO_2 to the atmosphere.[186] The DEIS also recognizes that, generally, carbon stored in the soil "may be

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[181] DEIS at 3-20.

[182] *See Mont. Wilderness Ass'n v. McAllister*, 666 F.3d 549, 559-561 (9th Cir. 2011).

[183] 40 C.F.R § 1502.22(a).

[184] 51 Fed. Reg. 15,618, 15,621 (Apr. 25, 1986).

[185] DEIS at 3-15.

[186] DEIS at 3-14.

released to the atmosphere in the form of carbon dioxide or methane as the climate warms."[187] However, the DEIS does not address the extent of carbon release that might occur in the Tongass, particularly in instances of old-growth logging, when rising temperatures accelerate decomposition processes. Specifically, the DEIS does not acknowledge the scientific literature's conclusion that the magnitude of carbon release from accelerated decomposition in circumstances of climate change is highly unpredictable, and hence a great risk. Studies indicate "that assumptions of stability are wrong, and a large potential for changes in carbon storage exists," but that, due to the lack of baseline information to inform models, "[t]he impacts of changing soil processing rates and hydrology pose an enormous challenge to scientists and managers."[188] The Forest Service itself has emphasized this lack of stability in other publications.[189]

In its FEIS, the Service should connect the recognition of the decomposition rate's importance in Tongass carbon flux, with the recognition that rising temperature accelerates decomposition. Its assessment of the carbon effects of each plan alternative should take into account the uncertainty of changes in decomposition rates, especially with regard to old-growth logging, which—all else equal—involves more carbon release via decomposition.

B. The DEIS Fails to Address Old-Growth Forest's Differential Resilience to Climate Change.

In addition to failing to fully describe the differential ability of old-growth forest to store and sequester carbon, *see supra* pp. 31-35, the DEIS also fails to describe the difference in resilience to climate change between old-growth and young forest. This difference is significant because it bears upon, *inter alia*, the extent to which the Tongass's carbon flux will persist depending on the plan alternatives' differing preservations of old-growth stands.

Scientific literature indicates that in the face of climate change, old-growth has a relative advantage as compared to young forest. The warming of average temperatures is likely to reduce forests' abilities to store carbon and regenerate following disturbance relative to their abilities before climate change.[190] "Rising temperatures . . . may lead to forests becoming a weaker sink or a net carbon source before the end of the century."[191] In the face of these changes, however, in general "[p]rimary forests tend to be more resilient to climate change and other human-induced environmental changes than secondary forests and plantations."[192] Studies have found

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[187] DEIS at 3-15.

[188] D'Amore & Edwards at 1.

[189] Carbon Cycle Science at 1 ("[T]he vast carbon store within [coastal temperate forest] may not be stable under a warming climate").

[190] Ryan at 13.

[191] Climate Change 2014 Working Group III Report at 845.

[192] *Id.* at 846; *see also* B. Mackey *et al.*, *Policy Options for the World's Primary Forests in Multilateral Environmental Agreements*, 8 CONSERVATION LETTERS 139, Supp. 14, Table s4 (2015).

that North America's Pacific Coastal Rainforests, especially the Tongass, may be particularly resilient to climate change.[193]

The FEIS should take these studies into account, and, given their findings, evaluate how the Plan alternatives' varying preservation of old-growth stands bears on overall forest resilience, and, *inter alia*, the Tongass's resulting future net carbon flux.

INTEGRITY OF THE CONSERVATION STRATEGY

This section of the comments addresses the consequences of the Forest Service's current transition proposal outlined in the Draft Forest Plan. At this point, the agency appears poised to rely on illusory transition out of industrial-scale old-growth logging that might or might not take place over the next 10, 15, or 20 years (if it ever happens at all). Additionally, the agency is proposing fundamental changes to the bedrock conservation principles that have supported the 1997 Forest Plan and the 2008 Amended Forest Plan by allow logging in OGRs, beach buffers, and riparian areas. If the Forest Service stays the course and adopts an amendment that fails to transition out of industrial-scale old-growth logging quickly and forever compromises the agency's long-held conservation strategy, then, as explained below, such an aggressive and scientifically indefensible position raises a wide-range of NEPA and NFMA infirmities.

If, however, the agency adopted a forest plan amendment that ended industrial-scale old-growth logging in less than five years and respected the scientific underpinnings of the conservation strategy, then the agency could avoid many of the concerns outlined in this section. One thing is clear; the agency's choice has serious implications that strike at the heart of the amendment.

I. THE DRAFT FOREST PLAN COMPROMISES THE CONSERVATION STRATEGY.

In the mid-1990s, the Forest Service confronted the possibility that the dwindling numbers of Alexander Archipelago wolves and Queen Charlotte Goshawks in the Tongass might require that those species be listed for protection under the Endangered Species Act (the ESA). The Forest Service's desire to avoid an ESA listing led the agency to adopt the old-growth forest

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[193] D. A. DellaSala *et al.*, *Climate Change May Trigger Broad Shifts in North America's Pacific Coastal Rainforests*, REFERENCE MODULE IN EARTH SYSTEMS AND ENVIRONMENTAL SCIENCES at 9 (2015) ("At broad spatial scales, northern coastal regions and their protected areas (BC, Alaska) may be more resilient to climate change than southern areas that are highly fragmented and more vulnerable to edge effects. . . . That pattern holds true for coastal regions compared to interior drier regions . . . perhaps because of climatic buffering of maritime climates. Our results therefore are important for maintaining ecological integrity and climate resilience in high priority conservation areas from north to south such as the Tongass Rainforest of Alaska . . . . Notably, ecological integrity and climate resilience are emphasized in the 2012 National Forest Planning Rule and climate resilience is emphasized in President Obama's Climate Action Plan (Executive Office of the President, 2013). Thus, the largely intact nature of the Tongass National Forest should provide important opportunities for meeting both policy objectives and for the northward expansion of rainforest communities in the face of climate change.").

conservation strategy as well as specific measures aimed at the conservation of these species and other old-growth dependent species.[194]

In the 2008 FEIS, the Forest Service explained that it developed its "integrated science-based old-growth forest habitat conservation strategy" during the 1997 Forest Plan Revision process.[195] The strategy has two basic components. The first is a forest-wide reserve network that "protects the integrity of the old-growth forest by retaining blocks of intact, largely undisturbed habitat."[196] The Forest Service made it clear that the efficacy of the reserve system "relies on blocks of intact, largely undisturbed habitats," which included but was not limited to old-growth forests.[197] The agency explained that reserves need to be "of the appropriate size, spacing, and composition to meet a desired design that will maintain viable well-distributed populations of one more species."[198] The second component is "management of the matrix, e.g., the lands with LUD allocations where commercial timber harvest may occur."[199] In those areas, "components of the old-growth ecosystem are maintained by standards and guidelines to protect important areas and provide old-growth forest habitat connectivity."[200]

The Draft Forest Plan proposes wholesale changes that detrimentally affect the long-standing wildlife conservation strategy.[201] The DEIS, however, does not acknowledge the broad scope of those changes or analyze the resulting adverse environmental impacts, including the ability to ensure the continued viability of fish and wildlife species.

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[194] *See* 2008 TLMP AR 603_1127 at 1 (Iverson, C., Letter to Mary Friberg Re: Review of Conservation Strategy Review Summary (Mar. 17, 2006)) (explaining that "an overriding motivation for the development of the [1997 Forest Plan's] conservation strategy was to respond to the petitions to list the Queen Charlotte Goshawk and Alexander Archipelago Wolf to incorporate necessary conservation measures such that listing under ESA was not necessary"). The undersigned groups understand that the Forest Service's planning records for the 1997 Tongass Land Management Plan and the 2008 Amended Tongass Land Management Plan are part of the agency's planning record for this plan amendment. Thus, they cite to the document numbers for relevant documents from those decision records where appropriate.

[195] 2008 FEIS at D-2.

[196] 2008 FEIS at D-2.

[197] 2008 FEIS at D-3.

[198] 2008 FEIS at D-3.

[199] 2008 FEIS at D-2.

[200] 2008 FEIS at D-2.

[201] *See generally* W. Smith, Comments on the Wildlife Conservation Strategy as represented in the Proposed Land and Resource Management Plan (Feb. 2016) (Smith Conservation Strategy Comments). The undersigned groups incorporate the Smith Conservation Strategy Comments into this letter in their entirety, including all cited publications.

The agency, moreover, is ignoring relevant science that has developed since the 2008 Amended Forest Plan was adopted, including science that questions many of the assumptions of the conservation strategy itself. The last time the Forest Service amended the forest plan, the agency conducted a conservation strategy review to assess new scientific information arising since 1997.[202] The Forest Service's review explained:

> The conservation strategy was based on careful analysis and integration of the best scientific information available at that time. The 1997 Forest Plan Record of Decision committed the USDA Forest Service to conduct an interagency review of the conservation strategy. The basic approach and central focus for the interagency review of the conservation strategy consists of a structured information assessment. This assessment examines pertinent new information accumulated since 1997, evaluates what it tells us about the conservation strategy, and identifies what additional information may be needed to better understand the strategy's effectiveness. Assessment results may be used to make adjustments to the strategy and to prioritize information needs for future investment.[203]

Yet, in this amendment process the Forest Service has inexplicably abandoned its commitment to basing its decision on "careful analysis and integration of the best scientific information available at that time." As explained below, the agency is deliberately ignoring relevant information.

All of these considerations are exacerbated by the fact that the Draft Forest Plan dramatically changes the agency's approach to resolving conflict between plan provisions. Under the Draft Plan, Chapter 5 would prevail over the rest of the plan in the event of conflict or discrepancies between directions.[204] The specific management prescriptions for each land use designation in Chapter 3 would prevail over the forest-wide standards and guidelines provided in Chapter 4.[205] The DEIS, however, fails to address the consequences of this dramatic change in management.

A. <u>The Length of the Transition Out of Industrial-Scale Old-Growth Logging is Too Long.</u>

As an initial matter, none of the alternatives analyzed in the DEIS contemplates a fast enough transition out of old-growth despite the growing chorus of concern from the nation's preeminent scientists. In 2003, two former Chiefs of the U.S. Forest Service, Jack Ward Thomas and Mike

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[202] *See* PR 769_05_000836 (Tongass National Forest Land and Resource Management Plan, *Interagency Conservation Strategy Review: An Assessment of New Information Since 1997*, Workshop Summary Report (2008).

[203] *Id*. at PDF 3.

[204] Draft Forest Plan at 1-4.

[205] *Id*.

Dombeck, cautioned that "harvest of old-growth from the national forests should come to an end . . . ."[206] In 2014, more than 275 scientists from academia and government urged the Forest Service to end the clear-cutting of the nation's remaining old-growth forests including the Tongass.[207] In January 2015, seven of North America's most prestigious scientific societies (representing a combined membership of over 30,000 scientists and natural resource professionals) called for an end to clear-cut logging of old-growth on the Tongass within the next three years.[208] Yet, the preferred alternative continues old-growth clear-cutting for at least 16 years and likely much longer, but the DEIS fails to examine the consequences of that transition.

The wildlife concerns on the Tongass, which are described below in detail, could be dramatically minimized if the Forest Service adopted a more rapid transition out of old-growth. Indeed, the troubling decline of numerous species could be minimized if the agency adopted a responsible approach to management of the nation's largest national forest. Conserving the Tongass's old-growth, rather than clear-cutting it for another 15, or 20, or 25 years as allowed under the Draft Forest Plan would assure that the Tongass Forest will continue to maintain a diverse forest ecosystem with abundant natural resources for future generations.

B. The Need to Concentrate Logging on Small Isolated Portions of the Tongass Compromises the Conservation Strategy.

The Draft Forest Plan also compromises the conservation strategy because the logging (both old-growth and now second-growth) will be occurring in very concentrated portions of the Tongass. This agency's proposal exacerbates an already troubling situation.

In the 2008 FEIS, the Forest Service analyzed the anticipated economic benefits from logging, which reflect the present value of expected Forest Service revenues from the timber sale program.[209] The agency projected future timber sale revenues using projected logging for each of the various alternatives analyzed in the 2008 FEIS.[210] The Forest Service recognized that:

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[206] M. Dombeck & J. W. Thomas, *P-I Focus: Declare harvest of old-growth forests off-limits and move on*, Seattle P-I Op Ed (Aug. 23, 2003) at PDF 3.

[207] *See* Thomas, J.W. *et al.*, Letter to the President by 78 North American Scientists calling for a national old-growth policy to protect the remaining old-growth on national forest lands throughout the US (June 25, 2014); Wilson, E.O. *et al.*, Letter to Secretary Vilsack from 200+ North American Scientists Re: Scientific support for completion of old-growth logging transition on the Tongass rainforest by the end of the Obama Administration (Oct.15, 2014).

[208] American Fisheries Society *et al.*, Letter to Secretary Vilsack Re: Old-growth logging transition on the Tongass National Forest (Jan. 20, 2015).

[209] 2008 FEIS at 3-546.

[210] *Id*.

> [U]nder current market conditions stumpage values for some stands would be negative . . . In other words, the estimated costs of harvesting and transporting the timber exceed the current value of the timber at the mill . . . and, as a result, the volume from these stands would be unlikely to sell.[211]

In the 2008 planning record, the Forest Service calculated average stumpage values per value comparison unit across the Tongass.[212] The agency considered both old-growth and second-growth. The agency then depicted the average stumpage values per acre on maps of the Tongass.[213]

According to the agency's own analysis, the vast majority of the stands in the Tongass result in negative stumpage value and the isolated areas that reflect positive stumpage values are located in a very concentrated portion of the Tongass. As explained below, these areas are of critical importance to several old-growth dependent species (e.g., wolves, goshawks, etc.). This means the Forest Service knew that it would have to log these areas of important habitat almost exclusively to develop economically profitable timber sales.

In the DEIS, the Forest Service acknowledges that regularly enacted Congressional Appropriation Act provisions prohibit the Tongass National Forest from offering timber sales that do not appraise positively.[214] It also explains "that individual timber sales offered under any of the alternatives in the first 25 years of the planning period will likely need to include a mix of old-growth and young-growth to appraise positive as required by Public Law 112-74, House Report 2055-257, Section 414."[215] Yet the DEIS never discloses the average stumpage values across the Tongass or the location of the only stands that appraise positively.[216] The analysis also does not appear in the planning record.

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[211] *Id*. at 3-547.

[212] *See* 2008 TLMP AR 603_Spectrum_Files\9_4_ROD\9_4_2_Stumpage 2451– Summary-OG-only; 2008 TLMP AR 603_Spectrum_Files\9_4_ROD\9_4_2_Stumpage 2456– TotalValByVCU-plusSummary-Phase 3; 2008 TLMP AR 603_Spectrum_Files\9_4_ROD\9_4_2_Stumpage 2457 – Summary-OG&YG.

[213] *See* 2008 TLMP AR 603_Spectrum_Files\9_4_ROD\9_4_2_Stumpage 2445 – TotalValby; 2008 TLMP AR 603_Spectrum_Files\9_4_ROD\9_4_2_Stumpage 2447 – TotalValbyVCU-OGonly; 2008 TLMP AR 603_Spectrum_Files\9_4_ROD\9_4_2_Stumpage 2448 – NICValbyVCU-OGonly; 2008 TLMP AR 603_Spectrum_Files\9_4_ROD\9_4_2_Stumpage 2449 – PositiveValbyVCU; 2008 TLMP AR 603_Spectrum_Files\9_4_ROD\9_4_2_Stumpage 2454 – PositiveValbyVCU-OGonly; 2008 TLMP AR 603_Spectrum_Files\9_4_ROD\9_4_2_Stumpage 2455 – TotalValbyVCU; 2008 TLMP AR 603_Spectrum_Files\9_4_ROD\9_4_2_Stumpage 2458 – NICValbyVCU.

[214] DEIS at 3-481; *see also* 2008 FEIS at 3-508.

[215] DEIS at 3-481.

[216] *See generally* DEIS at App. B.

Additionally, the agency knows that these same areas of the Tongass have already suffered the most adverse effects from the historical logging operations.[217] For example, Big Thorne is located in an area that historically contained more productive old-growth forest than any other part of the Tongass, but it has also experienced the highest amounts of logging.[218] Indeed, approximately 94 percent of the contiguous high-volume forest already has been logged on northern Prince of Wales Island.[219] As a result, there are approximately 50,261 acres of second-growth trees in the Big Thorne project area, of which 49,594 acres are a result of prior logging of old-growth forest.[220] Almost 40 percent of those young-growth stands (19,227 acres) are 25 years old or younger, in the stand initiation stage; the remaining are older and already in the stem exclusion stage.[221] Because of succession debt,[222] the full impacts of the 40 percent of stands still under 25 years of age have not yet been felt. Habitat quality will worsen as these stands reach stem exclusion. Yet these same areas are the only places on the Tongass the agency will be able provide timber sales that appraise positively.

The DEIS's failure to examine this information is unlawful. The agency knows, but fails to disclose relevant information regarding the overlap of positive timber sales and old-growth habitat. The FEIS must correct this failing and explain the adverse impacts associated with a timber program that can only operate on a handful of islands across the Tongass.

C. Logging in Buffers, Beach Fringe, & OGRs Compromises the Conservation Strategy.

The Forest Service's preferred alternative would allow second-growth logging, including clear-cuts of up to 10 acres, in OGRs, riparian management areas, and beach fringe buffers.[223] These areas comprise some of the most productive lands on the Tongass National Forest and their protection from logging is an essential component of the agency's conservation strategy. Clear-cutting on these protected lands compromises the conservation strategy and risks the viability of

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[217] D. M. Albert & J. W. Schoen, *Use of Historical Logging Patterns to Identify Disproportionately Logged Ecosystems Within Temperate Rainforests of Southeastern Alaska*, CONSERVATION BIOLOGY 27(4): 774-784 at PDF 6 (Table 1) (2013) (Albert & Schoen).

[218] *See* Big Thorne FEIS at 3-103; Albert & Schoen at PDF 7 (Fig. 2(b)); 2008 FEIS at 3-137 to 3-138 (explaining the largest trees in the Tongass, as well as those at lower elevations, were logged disproportionately decades ago due to their high productivity and the ease of access).

[219] Albert & Schoen at PDF 7.

[220] Big Thorne FEIS at 3-105.

[221] *Id*.

[222] *See* 2008 FEIS at 3-266; D. K. Person & T. J. Brinkman, Chapter 6: *Succession Debt and Roads, Short- and Long-Term Effects of Timber Harvest on a Large Mammal Predator-Prey Community in Southeast Alaska*, *in* NORTH PACIFIC TEMPERATE RAINFORESTS at 144, 155-160 (G. Orians & J. Schoen eds, 2013) (explaining consequences for deer and wolf dynamics).

[223] DEIS at 2-31.

many of the Tongass's most at-risk wildlife species. The DEIS entirely fails to analyze the myriad potential effects of this fundamental change in Tongass management. The undersigned groups request that the Forest Service retain current protections for OGRs, riparian management areas, and beach fringe buffers and reject any alternative that allows commercial logging in these areas.

The Tongass conservation strategy relies on a system of old-growth reserves, along with a set of protections for lands outside of the reserves that includes areas set aside as wildlife corridors to connect the reserves. Riparian management areas and beach fringe buffers provide essential connectivity between old-growth reserves for a wide variety of Tongass species. Although this system has not been proven sufficient to protect old-growth-dependent wildlife in the Tongass,[224] it provides a protective floor that should not be compromised. As the Pacific Northwest Research Station cautioned after its independent scientific peer review of the Forest Service's conservation strategy:

> Because of the unusually long time needed for succession to achieve climax in these temperate rain forests, a rotating block design for timber harvesting is not appropriate. Therefore, sufficient biodiversity *reserves must be established, and these must be permanently unavailable for timber harvest*. Otherwise blocks must be allowed to serve as source areas for many years beyond the age at rotation.[225]

The Forest Service has concluded that the "beach fringe was a very key feature of the overall Tongass conservation strategy."[226] Beach and riparian buffers are essential to maintaining viable populations of flying squirrels and other small mammals,[227] eagles, goshawks, deer, marten, otters, bears, shorebirds, waterfowl, bald eagles and other marine-associated species.[228] They also provide essential protections for salmon. According to the 2008 Amended Forest Plan FEIS:

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[224] *See* W. Smith, Proposed Forest Plan Amendment Further Compromises Established Conservation Measures to Sustain Viable Populations of Endemic Small Mammals (Feb. 2016) at PDF 9 (Smith Small Mammals Comments).

[225] 2008 TLMP AR 603_0009 (A. Ross Kiester & C. Eckhardt, *Review of Wildlife Management and Conservation Biology on the Tongass National Forest: A Synthesis with Recommendations* (Mar. 1994)) at 17 (PDF 24) (emphasis added).

[226] 2008 TLMP AR 603_1127 at 2.

[227] Smith Small Mammals Comments at PDF 9.

[228] 2008 Amended Forest Plan at 4-5.

> Beach and estuary fringe, and riparian habitats, have special importance as components of old-growth forests, serving as wildlife travel corridors, providing unique wildlife habitats, and providing a forest interface with marine or riverine influences that may distinguish them as separate ecosystems within the larger old-growth forest ecosystem . . . . In conjunction with riparian areas, which provide connectivity within watersheds, the beach fringe is thought to be a component of the major travel corridor system used by many resident wildlife species . . . . Accordingly, the Forest Plan establishes a Beach and Estuary Fringe Forest-wide Standard and Guideline that prevents timber harvest within 1,000 feet inland from mean high tide…Together, the beach and riparian habitat management features and the mapping of small reserves represented a substantial response to the landscape linkage element of conservation planning and significantly contributed to management of the overall matrix among habitat reserves.[229]

The Draft Forest Plan would allow clear-cutting of second-growth on up to 10-acre blocks. There is no science showing that these protected areas can be logged in economically viable ways without compromising the conservation values of the areas. To the contrary, scientists reviewing the TAC recommendations are highly critical of this strategy. For example, a group of ten independent scientists with significant Tongass experience stated that "we find no empirical data to support the contention that one can log 60-80 year old young-growth in ways that are economically viable *and* achieve desired wildlife benefits."[230] The scientists' letter reviews the current science on second-growth logging[231] and concludes, "[b]ased on the current science, the

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[229] 2008 FEIS at D-10 to D-11.

[230] Alaback, P. *et al.*, Letter to Deputy Forest Supervisor Jason Anderson at PDF 2 (May 12, 2015) (Alaback).

[231] Alaback, P.B. and J.C. Tappeiner II, *Response of western hemlock (Tsuga heterophylla) and early huckleberry (Vaccinium ovalifolium) seedlings to forest windthrow*, Can. J. For. Res. 21: 534-539 (1991); 2008 TLMP AR 603_0806 (DellaSala, D.A. et al., *Effects of silvicultural modifications of temperate rainforest on breeding and wintering bird communities, Prince of Wales Island, southeast Alaska*, Condor 98:706-721 (1996)); 2008 TLMP AR 603_0182 (Hanley, T.A., *Potential management of young-growth stands for understory vegetation and wildlife habitat in southeastern Alaska*, Landscape and Urban Planning 72:95-112 (2005)); PR 769_05_000797 (Hanley, T.A. *et al.*, *Precommercial thinning: Implications of early results from the Tongass-Wide Young-Growth Studies experiments for deer habitat in southeast Alaska*, Res. Pap. PNW-RP-593, USDA Forest Service, Pacific Northwest Research Station, Portland, OR (2013)); Matsuoka, S. *et al.*, *Succession of bird communities in young temperate rainforests following thinning*, J. Wildlife Management 76(5):919-931 (2012); Van Horne, B., *Density as a misleading indicator of habitat quality*, The Journal of Wildlife Management, 893-901 (1983); Zaborske, R.R and M.H. McClellan, *Understory vegetation development following commercial thinning in Southeast Alaska: preliminary results from the second-growth management area demonstration project*, *in* Beyond 2001: A Silvicultural odyssey to sustaining terrestrial and

prospects of achieving old-growth forest characteristics by placing small clear-cuts in mature young-growth stands is extremely low. We anticipate these ecologically important areas will be deferred from logging until that science changes." [232] Similarly, in his comments on the TAC recommendations, Matthew Kirchhoff was highly critical of the assertion that the Forest Service could allow 10-acre cuts in protected areas and still maintain their conservation value:

> What you are recognizing is we made a mistake when we clear-cut such ecologically valuable areas in the first place. What you are doing now is sentencing them to an infinite cycle of more logging—out of the misguided notion that calling it restoration makes it good. More logging will create more clear-cuts, which will demand more logging of inevitable second-growth. How and when will these stands be allowed to become old-growth—real old-growth that doesn't need our constant tree-cutting intervention to minimally function?[233]

Mr. Kirchhoff concluded that the Forest Service "should take beach fringe and OGRs off the table, except possibly, in very limited research-oriented applications."[234]

The Forest Service has not properly accounted for the major impacts that would result from relaxing protections for these areas. Given that the Forest Service and independent scientists have concluded that old-growth reserves, beach fringe buffers, and riparian management areas are essential to maintaining the viability of a wide array of wildlife species on the Tongass, the Forest Service must not allow commercial logging in these areas. The DEIS's preferred alternative would compromise the integrity of the Tongass conservation strategy and is therefore not an acceptable alternative.

II. THE FOREST SERVICE HAS NOT PROVIDED ITS ASSESSMENT OF WILDLIFE VIABILITY OR DISCLOSED ITS REASONING AND CONCLUSIONS IN THE DEIS.

The Tongass is the first national forest to amend a plan completed under the 1982 Planning Rule using the 2012 Planning Rule.[235] In so doing, the Forest Service appears to suggest that although Chapters 2, 3, and 4 of the Draft Forest Plan continue to be governed by the 1982 Planning Rule the agency does not have to address the impacts of the planning amendment effort on the ability

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aquatic ecosystems (S. Parker & S. Hummel eds, 2002), Pages 74-82, Proceedings of a workshop in 2001 in Hood River, OR. U.S. Forest Service, Gen. Tech. Rep. PNW-GTR-546.

[232] Alaback at PDF 2.

[233] Kirchoff, M., Comments on Draft Tongass Advisory Committee Recommendations at 9 Cmt. 48 (Apr. 19, 2015).

[234] *Id.* at 2, Cmt. 6.

[235] U.S. Forest Service Proposed Plan Amendment, http://www.fs.usda.gov/detail/tongass/landmanagement/?cid=stelprd3801708 (Feb. 18, 2016).

of the Tongass to maintain viable, well-distributed fish and wildlife populations. To the contrary, given that Chapters 2, 3, and 4 are governed by the 1982 Planning Rule the agency must be adhere to those NFMA requirements.

As the Forest Service acknowledges,"[t]he 2008 [Amended] Forest Plan was developed under the 1982 Planning Rule, but most changes to that Plan are made under the 2012 Planning Rule."[236] The DEIS explains that "[a]ll deletions would be done in Chapters 1 to 4 of the proposed Forest Plan, and any substantial changes or additions to the 2008 [Amended] Forest Plan management direction would be incorporated into Chapter 5 of the proposed Forest Plan."[237] More specifically, the agency is proposing that Chapter 5 will be governed by the 2012 Planning Rule, but that Chapters 2, 3, and 4 will be continue to governed by the 1982 Planning Rule.[238]

The National Forest Management Act requires that the Forest Service provide for the diversity of plants and animals, based on the suitability and capability of each National Forest, as part of meeting overall multiple-use objectives.[239] The Forest Service in turn adopted the 1982 Planning Rule, which provides: "Fish and wildlife habitat shall be managed to maintain viable populations of existing native and desired non-native vertebrate species in the planning area."[240] The agency characterizes a viable population, for planning purposes, "as one which has the estimated numbers and distribution of reproductive individuals to insure its continued existence is well distributed in the planning area."[241] This means, with regard to a forest plan, to "insure that viable populations will be maintained, habitat must be provided to support, at least, a minimum number of reproductive individuals and that habitat must be well distributed so that those individuals can interact with others in the planning area."[242] Stated more directly, "the forest plan must comply with substantive requirements of the Forest Act designed to ensure continued diversity of plant and animal communities and the continued viability of wildlife in the forest, including the requirement that wildlife habitat shall be managed to maintain viable populations of existing native and desired non-native vertebrate species in the planning area."[243]

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[236] DEIS at 2-10; *see also* Draft Forest Plan at 1-1.

[237] DEIS at 2-10.

[238] Draft Forest Plan at 1-4.

[239] 16 U.S.C. § 1604(g)(3)(B).

[240] 36 C.F.R. § 219.19 (2000).

[241] *Id*.

[242] *Id*.

[243] *Idaho Sporting Cong., Inc. v. Rittenhouse*, 305 F.3d 957, 961 (9th Cir. 2002).

Like the 2008 Amended Forest Plan,[244] the Draft Forest Plan explicitly incorporates these viability obligations.[245] Indeed, the DEIS explains "NFMA directs the Forest to manage wildlife habitat to maintain viable and well distributed populations to ensure continued existence in the planning area."[246] The agency explained that for its analysis, "the evaluation of viability includes considerations of the island archipelago environment as well as the best available science related to each species."[247]

Yet, the DEIS's discussion of direct and indirect species-specific impacts never discusses or describes the environmental effects in terms of impacts on the ability to retain viable, well-distributed fish and wildlife populations.[248] The cumulative impacts analysis offers only one mention of viability: "Activities that occur on other land ownerships within and adjacent to the Tongass have the potential to affect the overall context within which effects of Forest management on wildlife population distribution and viability are considered."[249]

Given the continued emphasis on industrial-scale old-growth logging in the Draft Forest Plan, and the changes to wildlife conservation elements of the plan, the Forest Service must comply with the 1982 Planning Rule and must explain its rationale regarding the ability to manage the Tongass habitat (both old-growth and second-growth) in such a way as to ensure well-distributed, viable populations of existing native and desired non-native vertebrate species. To do otherwise, violates NFMA's requirements.

Additionally, the DEIS fails to examine and disclose impacts on fish and wildlife viability. The DEIS states:

> The scope of this analysis is the individual proposed modifications the contributing elements of the conservation strategy and the associated potential to affect the functioning of the conservation strategy. The proposed Forest Plan amendment does not propose changes to the framework of the conservation strategy or the size or spacing of OGRs (with one exception resulting from land adjustments in the Carl Levin and Howard P. "Buck" Mckeon National Defense Authorization Act for Fiscal Year 2015 [hereafter referred to as the National Defense Authorization Act for Fiscal Year 2015]). The proposed OGR modifications

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[244] 2008 Amended Forest Plan at 4-89 ("Provide the abundance and distribution of habitat necessary to maintain viable populations of existing native and desirable introduced species well-distributed in the planning area (i.e., the Tongass National Forest).").

[245] *See* Draft Forest Plan at 4-82.

[246] DEIS at 3-235.

[247] DEIS at 3-235.

[248] *See* DEIS at 3-239- to 3-260.

[249] DEIS at 3-260.

> compensate for portions of individual OGRs that were located on National Forest System (NFS) lands that were conveyed to the Sealaska Native Corporation (see below). Therefore, this analysis is not intended to be a review of the underpinnings of the conservation strategy or its effectiveness as a whole. Such an evaluation is outside the scope of this proposed Forest Plan amendment and would be more appropriately conducted in the context of a Forest Plan revision, which under the 2012 Planning Rule requires an assessment of ecological sustainability and diversity of plant and animal communities.[250]

This statement is incorrect and must be eliminated. Significant portions of the Draft Forest Plan continue to be governed by the 1982 Planning Rule and its viability requirements. As explained throughout these comments, the Forest Service is proposing dramatic changes to the 2008 Amended Forest Plan and the agency's conservation strategy and proposes to continue industrial-scale old-growth logging for more than a decade to come. As a result, the FEIS must examine both the adverse impacts and necessary mitigation measures associated with each alternative's ability to manage habitat in such a manner as to ensure well-distributed, viable wildlife populations on the Tongass.

III. THE FOREST SERVICE CANNOT IGNORE RELEVANT SCIENCE.

A. The Forest Service Has Not Complied With NFMA's Regulatory Requirements to Base its Decision on the Best Available Science.

During the forest planning process, the Forest Service must comply with substantive obligations to base its decision-making on the best available science. Despite these obligations, the Forest Service has, in many instances, ignored relevant science and, in others, attempted to shield itself from those obligations.

Under the 2012 Planning Rule, "[t]he responsible official shall use the best available scientific information to inform the planning process required by this subpart."[251] More specifically, the agency "shall determine what information is the most accurate, reliable, and relevant to the issues being considered."[252] The regulations require the agency to "document how the best available scientific information was used to inform . . . the plan decision . . . ."[253] The agency also "must[] [i]dentify what information was determined to be the best available scientific information, explain the basis for that determination, and explain how the information was applied to the issues considered."[254]

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[250] DEIS at D-3.

[251] 36 C.F.R. § 219.3.

[252] *Id.*

[253] *Id.*

[254] *Id.*

Throughout this comment letter, the undersigned groups have identified numerous instances in which the Forest Service has either ignored or failed to account for the best available science in the manner prescribed by 36 C.F.R. § 219.3. In so doing, the agency has violated its obligations under the 2012 Planning Rule.

B. The DEIS Fails to Include or Address Relevant Science.

Throughout these comments, the undersigned groups have identified numerous scientific publications the Forest Service apparently ignored or failed to consider. In fact, the DEIS acknowledges that numerous scientific publications have raised serious questions regarding the efficacy of the agency's wildlife management, but the agency refuses to consider the science. The DEIS states, for example,

> Since 2008, there have been research publications that address some of these species including goshawks (Smith 2013), wolves (Person and Russell 2008, 2009; Weckworth et al. 2010, 2011; ADF&G 2012; Person and Logan 2012), brown bears (Flynn et al. 2009), marten (Flynn and Schumacher 2009, Pauli et al. 2015), deer (White et al. 2009) and flying squirrels (Flaherty et al.2008, 2010; Pyare et al. 2010; Smith et al. 2011) that may warrant an assessment of the efficacy of the original conservation strategy design criteria. This type of assessment is outside of the scope of the proposed Forest Plan amendment, and would be more appropriately conducted in the context of a Forest Plan revision.[255]

NEPA also does not allow the agency to ignore relevant scientific opinion given the continuing focus on old-growth logging in the Draft Forest Plan. The FEIS must remedy this pervasive problem.

IV. THE FOREST SERVICE MUST EXAMINE THE ADVERSE EFFECTS OF SECOND-GROWTH LOGGING ON THE OVERALL CONSERVATION STRATEGY.

The Draft Forest Plan is predicated on logging second-growth stands on the earliest-logged and highest-productivity sites. Many, but not all, of these sites, are located primarily in the southern Tongass, and are several decades into the transition back to an old-growth state, which if allowed to continue, would eventually provide high habitat value. The Forest Service has not examined how, when, and whether logging particular areas will adversely affect Tongass ecosystems or the agency's conservation strategy.

The Forest Service knew when it adopted the 1997 Forest Plan that the second-growth issues raised important considerations that the agency did not attempt to address at the time. For example, in response to the Interagency Viable Population Committee (VPOP) report, the Pacific Northwest Research Station conducted an independent scientific peer review of the Forest

[255] DEIS at D-4.

Service's conservation strategy.[256] The peer review raised specific concerns regarding second-growth and the reserve system:

> *The amount, kind and spatial distribution of second-growth depends on the patterns of regeneration that occur on the Tongass National Forest*. These patterns of regeneration are only poorly understood at this time and yet will be crucial to any landscape design strategy (see Section 7.4.4). *This is an important point where wildlife considerations interact with other parts of the planning effort*. Silviculture and harvesting methods will dictate much of the future pattern of second-growth.[257]

As the comments provided by NRDC and Geos explain, not all second-growth stands have the same ecological value. The agency must consider and disclose the ecological considerations in deciding whether, how, and when to log second-growth stands. Stated differently, the Forest Service cannot reset the clock on the most productive portions of the Tongass without examining the enormous set-back to the conservation strategy writ-large. The FEIS must examine the environmental impacts of logging these second-growth stands and how those decisions affect the forest ecosystem and the conservation strategy.

FISH & WILDLIFE IMPACTS

Like the preceding section addressing the integrity of the conservation strategy, this section of our comments addresses the consequences of the Forest Service's current transition proposal to rely on an unenforceable—and hence functionally illusory—transition out of industrial-scale old-growth logging and make fundamental changes to the wildlife conservation strategy by allow logging in OGRs, beach buffers, and riparian areas. Many of the fish and wildlife impacts arise because the agency is proposing not to safeguard essential habitat from logging. In so doing, the agency's approach raises a wide-range of NEPA and NFMA infirmities. To avoid most if not all of these problems, the agency should adopt a forest plan amendment that ends industrial-scale old-growth logging in less than five years and respects the scientific underpinnings of the conservation strategy.

I. WOLVES & DEER

To comply with its NFMA obligations, and as is true for other species as well, the Forest Service must explain how and whether it has concluded the Draft Forest Plan will manage habitat in such a way as to ensure the continued presence of a well-distributed, viable population of wolves in the Tongass. To do otherwise, the agency would violate its NFMA obligations. The DEIS fails to supply this explanation and the agency must correct this failing in the FEIS.

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[256] 2008 FEIS at D-3 to D-4.

[257] 2008 TLMP AR 603_0009 (A. Ross Kiester & C. Eckhardt, *Review of Wildlife Management and Conservation Biology on the Tongass National Forest: A Synthesis with Recommendations* (Mar. 1994)) at PDF 23 (emphasis added).

A. Tongass Wolves are in Serious Decline.

Only a concentrated part of the Tongass has ever been capable of supporting permanent wolf populations. There are, for example, no wolves on the northern islands of Admiralty, Baranof, or Chichagof or their associated smaller islands.[258] Only the largest southern islands, "including Prince of Wales, Kuiu, Kupreanof, Mitkof, Etolin, Revillagigedo, Kosciusko, Zarembo, and Dall islands, are thought to support persistent wolf populations."[259]

But the large islands south of Frederick Sound are critical to the future of the wolf not just on the Tongass, but throughout all of Southeast Alaska. They "support approximately 60 to 70 percent of the total wolf population in Southeast Alaska."[260] In fact, in the mid-1990s the wolves on Prince of Wales Island and the smaller surrounding islands alone likely represented almost 40 percent of all of the wolves in Southeast Alaska.[261]

Yet according to the Forest Service, there has been a "[d]ramatic decline of wolf population on Prince of Wales Island."[262] The State of Alaska's most recent 2014 estimate for a study area within Game Management Unit 2 is 9.9 wolves per 1000 square kilometers, which Alaska Fish and Games admits is "significantly lower than the autumn 2013 estimate of 24.5 wolves/1000 km^2."[263] Thus, the most recent estimate by the State of Alaska for Game Management Unit 2 is only 9.9 wolves per 386 square miles or 0.025 wolves per square mile.

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[258] 2008 FEIS at 3-284 (Table 3.10-9 n.2).

[259] 2008 FEIS at 3-236 to 3-237 (internal reference omitted); *see also id*. at 1-4 (map showing Prince of Wales, Etolin, Revillagigedo, and Zarembo islands); *id*. at 3-132 (map showing Kuiu, Kupreanof, Miktof, and Dall islands); 2008 TLMP AR 603_0290 at C2-420 (U.S Forest Service, Tongass Land Management Plan Revision, Draft Supplemental Environmental Impact Statement, Roadless Area Evaluation for Wilderness Recommendations, Volume II: Appendix C - Part 2 (May 2002)) (describing location of Kosciusko Island "near the northwest end of Prince of Wales Island"); 2008 TLMP AR 603_1603 (Roadless Area Inventory Map) (showing Kosciusko Island by identifying Roadless Area 515 on the island).

[260] 2008 FEIS at 3-237 (internal references omitted); *see also* Tongass National Forest Land and Resource Management Plan: Land Use Designations (Jan. 2008) (map showing location of Frederick Sound).

[261] PR 769_05_000523 (D. Person, *et al.*, *The Alexander Archipelago Wolf: A Conservation Assessment*, U.S. Forest Service at PDF 20 (Nov. 1996)) (Wolf Conservation Assessment).

[262] U.S. Forest Service Briefing Paper: Dramatic Decline of Wolf Population on Prince of Wales Island, Tongass National Forest at PDF 1 (May 29, 2015).

[263] *See* State of Alaska, Department of Fish and Game Division of Wildlife Conservation, Gretchen Roffler Memorandum to Ryan Scott at PDF 1 (June 16, 2015).

These troubling findings bolster the concerns Dr. David Person expressed when he participated in the Big Thorne environmental review.[264] He cautioned that wolves are "already facing the possibility of extinction on Prince of Wales Island."[265] He noted that Big Thorne logging "will remove the most important remaining deer winter habitat . . . [and] [a]s a result, the predator-prey relationship between wolves and deer on Prince of Wales is likely to collapse."[266] He opined that the "combined effects of Big Thorne and the other logging on wolves within the Prince of Wales Archipelago likely will be the collapse of a sustainable and resilient predator-prey ecological community."[267]

The Forest Service must address this decline in the FEIS. Additionally, it must explain whether and how the Draft Forest Plan fulfills the agency's obligations under NFMA to ensure the viability of the wolf.

B. The Draft Forest Plan and the DEIS Suffer From the Same Infirmities Addressed in *In Re: Big Thorne Project and 2008 Tongass Forest Plan*, No. 15-35244 (9th Circuit).

The Draft Forest Plan directly violates the agency's substantive obligation under the National Forest Management Act to adopt a forest plan that ensures the wolf remains viable in the Tongass. The 2008 Amended Forest Plan's critical mechanism for meeting this requirement is, the agency concedes, discretionary and non-binding and, as a result, the plan does not require the agency to maintain the necessary old-growth habitat to "insure [the wolf's] continued existence."[268] The Draft Forest Plan fails to correct this failing.[269]

Pursuant to the National Forest Management Act regulations, "wildlife habitat shall be managed to maintain viable populations of existing native and desired non-native vertebrate species in the planning area."[270] "[T]o insure that viable populations will be maintained," a forest plan must manage habitat in such a way as "to support, at least, a minimum number of reproductive individuals and that habitat must be well distributed so that those individuals can interact with

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[264] *See generally* Statement of David K. Person Regarding the Big Thorne Project, Prince of Wales Island (Aug. 15, 2013) (Person Statement); David K. Person Comments Regarding the Draft Supplemental Information Report for the Big Thorne Project (June 23, 2014).

[265] Person Statement at PDF 15-16.

[266] Person Statement at PDF 16; *see also id*. at PDF 1 (explaining Big Thorne will remove what remains of "the last high quality winter range for deer"); Audubon Alaska, Correction to Big Thorne SIR Comments, Updated USFS Deer Habitat Capability Map (Aug. 7, 2014).

[267] Person Statement at PDF 6.

[268] 36 C.F.R. § 219.19 (2000).

[269] *See* Draft Forest Plan at 4-88.

[270] 36 C.F.R. § 219.19 (2000); *see also* 16 U.S.C. § 1604(g)(3)(B).

others in the planning area." As described above, both the Draft Forest Plan and the 2008 Amended Forest Plan explicitly incorporate these obligations.[271]

In adopting the 2008 Amended Forest Plan, the Forest Service concluded that if the agency managed habitat in such a way that maintained sustainable wolf populations, it would by necessity maintain viable wolf populations. According to the Forest Service, the Wolf Conservation Assessment provided the best available information regarding wolf viability and it concluded that it was not scientifically defensible to identify what minimum wolf population would insure the wolf's continued existence. As a result, the Forest Service accepted its experts' recommendation that the agency meet its viability obligations by minimizing the risk of dropping below that unidentified viability floor by maintaining sufficient old-growth habitat to support the higher level of sustainable wolf populations (which accounts for deer hunting and wolf hunting and trapping). The Forest Service, however, concedes the 2008 Amended Forest Plan does not require the agency to maintain sufficient habitat to support sustainable wolf populations.[272]

Additionally, the DEIS does not examine or disclose how much of the Tongass is not expected to meet its generally recognized level of habitat capability of 18 deer per square mile to support a functioning dynamic between sustainable wolf populations and current human deer hunting as a result of the amount and concentrated location of the logging contemplated by the Draft Forest Plan. The agency does not explained how far below it expects individual wildlife analysis areas, biogeographic provinces, or even larger geographic areas to fall below this level of habitat capability. The DEIS fails to disclose this information or examine the consequences for the wolves and human deer hunters.

The DEIS also never examines the impacts on wolves and human deer hunters given the intersection of limited habitat availability and the fact that the economically viable timber sales are almost certain to come from wolf habitat. The agency knows that 60-70 percent of the wolves in Southeast Alaska inhabit areas that will largely or exclusively have to be logged to achieve the timber goals set out in the Draft Forest Plan. In 2008, the Forest Service provided maps that examined the stumpage values across the Tongass. Here, the agency fails to disclose any of the relevant information regarding the overlap of positive timber sales and wolf habitat. The agency must correct that failing in the FEIS.

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[271] *See supra* pp. 48-51.

[272] *See generally* Forest Plan Appellants' Opening Br. (July 2, 2015) (Doc. 19), Answering Brief of the Federal Defendants (Aug. 20, 2015) (Doc. 37-1), Forest Plan Appellants' Reply Brief (Sept. 8, 2015) (Doc. 45) in *In Re: Big Thorne Project and 2008 Tongass Forest Plan*, No. 15-35244 (9th Circuit). The briefing in this case, including the arguments advanced by the Forest Service, and the supporting record citations are incorporated by reference into this comment letter in their entirety.

C. The USFWS Wolf Finding Raises Important and Unanswered Questions Regarding the Forest Service's Ability to Maintain Viable Well-Distributed Wolves on the Tongass.

The Forest Service also must address the very troubling information from the U.S. Fish and Wildlife Service regarding the status of the wolf on the Tongass.[273] The U.S. Fish and Wildlife Service Service, for example, concluded that Game Management Unit 2 only has 50-159 wolves, representing six percent of range.[274] In 1995, the wolf population in that area was 300-350 animals.[275] Thus, although the U.S. Fish and Wildlife Service ultimately concluded that ESA listing was not warranted (given the overall geographic range of the wolf), it acknowledged that there was reasonable risk that wolves could be significantly reduced, or perhaps even extirpated, from Prince of Wales Island and the smaller surrounding islands as a result of declining prey abundance and increasing density of roads and subsequent human-induced mortality risk to wolves.[276]

As the Forest Service acknowledged in the 2008 FEIS:

> Recent research (*Alexander Archipelago Wolf*, presented at the Tongass Conservation Strategy Review Workshop 2006) has shown that the population on [Prince of Wales] Island is genetically isolated from other Tongass populations, *which presents profound implications for maintaining well-distributed wolf populations* in light of local declines, given that these populations are are [*sic*] more sensitive to human activity and habitat disturbance than wolf populations elsewhere in the state (Schoen and Person 2007).[277]

The Forest Service must explain its viability conclusions in light of the important components of genetic biodiversity the wolves on Prince of Wales and surrounding islands provide to the larger Tongass population. And the FEIS must disclose and explain the agency's assessment of the relevant science and explain its conclusions regarding the impacts to wolves.

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[273] *See generally* 81 Fed. Reg. 435 (Jan. 6, 2016).

[274] *Id*. at 440.

[275] Person Statement at PDF 9.

[276] 81 Fed. Reg. at 440, 452, 455-56, 458.

[277] 2008 FEIS at 3-281 (emphasis added); *see also* 2008 TLMP AR 603_0879 (B.V. Weckworth *et al.*, *A Signal for Independent Coastal and Continental histories among North American wolves*, MOLECULAR ECOLOGY 14: 917-931 (2005)); PR 769_05_000489 (B.V. Weckworth *et al.*, *Phylogeography of wolves (Canis lupus) in the Pacific Northwest*, JOURNAL OF MAMMALOGY, 91(2):363-375 (2010)); B.V. Weckworth *et al.*, *Genetic distinctiveness of Alexander Archipelago wolves (Canis lupus ligoni): Reply to Cronin et al. (2015)*, JOURNAL OF HEREDITY 1-3 (2015); E. A. Lacey, Ph.D. President, American Society of Mammalogists, Letter to Dr. Kimberley Titus, Alaska Department of Fish and Game (Nov. 1, 2015).

Additionally, the amendment process gives the Forest Service the opportunity to strengthen the plan provisions governing wolves. For example, management of road densities to limit hunter and trapper access should be required, rather than recommended, at least in the areas of greatest concern for wolves. The wolf management program should not require unanimous agreement. The U.S. Fish and Wildlife Service, the Forest Service, and others have repeatedly expressed concerns wolf mortality, but no such program has been developed because of the current plan provision.

II. GOSHAWKS

Goshawks are associated with, and well-adapted to, specific forest environments, and the prey that inhabit them. They "have broad short wings and a long tail, which enable rapid acceleration and agile maneuverability . . . ."[278] They "hunt by alternating short flights with a period of watching from a perch. Once prey is spotted, an attack is launched from the perch (Squires and Reynolds 1997). This method of hunting relies on cover to conceal the predator's approach, perches from which to observe and attack, adequate visibility for spotting prey, and adequate space between trees to allow for flying between perches and attacking prey (Reynolds et al. 1992)."[279] Canopy cover also protects goshawks and their nestlings from avian predators such as great horned and barred owls, and bald eagles.[280]

Importantly, in Southeast Alaska, goshawks are associated with a very particular forest type: very highly to moderately productive old-growth (250 years old or older).[281] Nests are "typically located in tall trees, within high-volume forest stands with relatively high canopy cover."[282] They spend significantly less time in low productivity forest (approximately 10 percent)[283] and actively avoid clear-cuts, nonforested areas, and mature sawtimber (approximately 75- to 150-year old stands).[284] "The amount and distribution of productive old-growth forest (especially the moderate to very high volume components), mature sawtimber, and riparian and beach zones are likely to set a limit on goshawk distribution and abundance."[285]

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[278] PR 769_05_000487 (U.S. Fish and Wildlife Service, Alaska Region, Juneau Fish and Wildlife Office, Queen Charlotte Goshawk Status Review) at 35 (Apr. 25, 2007)) (USFWS Goshawk Review).

[279] *Id.* at 64.

[280] *Id.* at 67, 107-08.

[281] *See* 2008 TLMP AR 603_0150 at PDF 43 (G. C. Iverson *et al.*, *Conservation Assessment for the Northern Goshawk in Southeast Alaska* (Nov. 1996)) (finding "a combined 58 percent of all habitat use occurring in these cover types") (Goshawk Conservation Assessment).

[282] The Shipley Group, Goshawk Survey Soule River Watershed Southeast Alaska at 8 (2009).

[283] Goshawk Conservation Assessment at PDF 43 (Table 9).

[284] *Id*. at PDF 42, 40, 43.

[285] *Id*. at PDF 69.

More specifically, as noted goshawk expert, Dr. Winston Smith explains in his comments on the Draft Forest Plan, "three critical spatial components of the nesting home range have been characterized: nest area, post-fledging family area (PFA), and foraging area."[286] Goshawk pairs have multiple nest areas (two to eight) and use nest areas for more than a year, but they can be used intermittently for decades.[287] Experts recommended that each goshawk pair have at least three nest areas, as well as three replacement nest areas.[288] Post-fledging family areas portions of the breeding home ranges used by adults and juveniles after young birds leave the nest.[289] These areas must afford young birds protection from predators and sufficient prey to sustain them as they develop.[290] These areas "should have overstories with at least 50 percent canopy cover and well-developed herb and shrub understories, as well as key habitat features essential to the life histories of the goshawk prey species."[291] Goshawks' foraging habitat must support a variety of prey species, requiring "an uneven-aged silvicultural system, which produces a mosaic of different-aged stands . . . rather than regeneration harvest, such as clear-cuts, that remove the entire canopy and result in a single, uniform and dense canopy for decades following harvest."[292] Goshawks foraging habitat reflects the majority of breeding home range.[293] Each of these three types of habitat "needs to be considered simultaneously in land-use planning or mitigation."[294]

A. A Multitude of Factors Threatens the Viability of Goshawks Throughout Southeast Alaska.

Goshawks in Southeast Alaska are at risk from both natural and anthropogenic factors, described below, that have resulted in extremely large territories and presumed increased risk factors, along with low goshawk numbers as follows.

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[286] W. Smith, Proposed Forest Plan Amendment Further Compromises Established Conservation Measures to Sustain Viable Northern Goshawk Populations at PDF 3 (Feb. 2016) (citing Reynolds et al. 1992) (Smith Goshawk Comments). The undersigned groups incorporate the Smith Goshawk Comments into this letter in their entirety, including all cited publications.

[287] *Id*. at PDF 3 (citing Reynolds et al. 1992).

[288] *Id*. (citing Reynolds et al. 1992).

[289] *Id*. (citing Reynolds et al. 1992).

[290] *Id*.

[291] *Id*. (citing Reynolds et al. 1992).

[292] *Id*. at PDF 4 (citing Nowacki & Kramer 1997).

[293] *Id*. at PDF 6.

[294] *Id*. at PDF 5 (citing Reynolds et al. 2006, Northern Goshawk *Accipiter gentilis laingi* Recovery Team 2008).

1. Goshawks have come to depend upon large territories in the region.

In Southeast Alaska, a combination of factors, including low prey abundance, natural fragmentation (by ice fields, muskeg bogs, steep terrain, and scattered islands), and past "highgrading" (disproportionately cutting down higher volume forest stands) has forced goshawks into larger foraging territories than anywhere else in North America. Breeding-season home ranges average 4,500 hectares (11,120 acres) for females, and 5,900 hectares (14,580 acres) for males.[295] One male breeding season use area was radio-tracked at more than 19,000 hectares (47,000 acres).[296] By contrast, in the rest of North America, breeding-season ranges average between 570 and 3,500 hectares.[297] Year-round use areas in Southeast Alaska are vast, averaging more than 15,000 hectares (up to 67,000 hectares) for males, and more than 50,000 hectares (up to 180,000 hectares) for females.[298]

As the U.S. Fish and Wildlife Service has noted, the energy expenditure associated with having to seek prey over such enormous areas poses a serious threat to goshawks in Southeast Alaska:

> Physiologically, foraging is a trade-off between the energy expended to acquire food and energy derived from its acquisition. The energetic demands of foraging increase with distance traveled. The thresholds for individual survival and for supplying food to nestlings and a brooding mate in this energy balance are unknown, but habitat alteration that decreases foraging efficiency will push individuals and broods toward that threshold.[299]

As foraging ranges increase during the breeding season, the likelihood of reproductive success is adversely affected. "Longer travel distances during foraging increase ... the probability that adults may abandon nests."[300] A Forest Service report concluded more than 15 years ago that "The very large areas used by goshawks in southeast Alaska may lead to high energy expenditure during daily movements. . . . [P]opulations of individuals requiring large ranges may be energetically stressed, have lower reproductive success, and be less resilient to further stress . . . ."[301] Outside of the breeding season, range expansion is associated with increased risk

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[295] 2008 TLMP AR 603_0815 at PDF 30-31 (C. Flatten, *et al.,* Alaska Department of Fish and Game, Northern Goshawk Monitoring, Population Ecology and Diet on the Tongass National Forest (Nov. 2001)) (Flatten).

[296] *Id.* at PDF 31.

[297] USFWS Goshawk Review at 34.

[298] Flatten at PDF 30-31; *see also* Goshawk Conservation Assessment at PDF 37-38 (recording multiple males and females using areas larger than 400,000 acres (162,000 hectares)).

[299] USFWS Goshawk Review at 66.

[300] *Id.*

[301] Goshawk Conservation Assessment at PDF 71.

of death. "Mortality of both male and female adult goshawks in Southeast Alaska was highest in late winter, when food availability is lowest" (and ranges were at their largest).[302]

2. Tongass Goshawks are a small, isolated, and declining population.

A second major threat to goshawks in Southeast Alaska, partially related to the phenomena discussed above, is the fact that they comprise a small, potentially genetically isolated, and almost certainly declining population. As USFWS has noted, Queen Charlotte goshawks exist in an inherently precarious status, highly vulnerable to any further stresses. "Given the small populations, low survival or reproductive rates could not be sustained long before viability of the subspecies would be at risk."[303]

USFWS has estimated that there are only approximately 300 to 400 pairs of goshawks remaining in the region (about the same number as in British Columbia, which USFWS determined to be a distinct population segment and threatened within the meaning of the Endangered Species Act).[304] USFWS has also determined that the Southeast Alaska population is largely isolated, because it appears to be cut off from both the Queen Charlotte Islands to the south (by open ocean), and the British Columbia mainland to the east (by the Coast Range mountains). And the agency has concluded that Tongass birds may be genetically diverse from Canadian specimens as well.[305]

In fact, the USFWS figure likely overstates the number of reproductive pairs in Southeast Alaska, putting local birds in an even worse position than USFWS reports. USFWS based its population estimate on studies done by Schempf and Woods (2000) and Flatten *et al.* (2001).[306] Schempf and Woods estimated that the Tongass National Forest contained between 580 and 747 nesting territories.[307] Flatten *et al.* (2001) used broadcast and telemetry surveys to determine that, on average, approximately 45 percent of nesting territories are occupied in any given year.[308] Applying Flatten's territory occupancy rate to Schempf and Woods' total territories, the USFWS estimated that, as of 2000, there were approximately 261 to 336 breeding pairs in the

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[302] USFWS Goshawk Review at 33, 41; *see also id.* at 55 ("Most adult mortality in Southeast Alaska and on Vancouver Island occurs in late winter (Titus et al. 2002, McClaren 2003a), when prey densities are lowest and snow or other factors may limit prey availability. Dead birds recovered were emaciated or in areas with limited prey, and food stress or starvation was suspected (Titus et al. 2002, McClaren 2003a)").

[303] USFWS Goshawk Review at 8-9.

[304] *See* 77 Fed. Reg. 45870, 45887-88 (Aug. 1, 2012); 72 Fed. Reg. 63123, 63,128 (Nov. 8, 2007).

[305] 72 Fed. Reg at 63,135.

[306] *Id*. at 63,127-28; *see also* P. Schempf & T. Woods, *Summary of Status of Queen Charlotte Goshawk Remand* (Feb. 23, 2000) (Schempf & Woods); Flatten.

[307] 72 Fed. Reg. at 63,127.

[308] 72 Fed. Reg. at 63,127-28.

Tongass National Forest.[309] Though it did not explain its methodology, the USFWS then extrapolated this range to estimate that approximately 300 to 400 pairs of goshawks occupied Southeast Alaska.[310]

However, much of the area that Schempf and Woods categorized as goshawk habitat, very likely would not support breeding birds. They considered hypothetical territories with as little as 20 percent of the land area in old-growth forest as "suitable."[311] In contrast, both the Goshawk Conservation Assessment and Doyle concluded that, to be suitable, at least half of a bird's territory probably needed to be covered in old-growth forest.[312] The Forest Service appears to accept these conclusions in principle.[313]

Thus, applying a more realistic habitat parameter would necessarily have led Schempf and Woods to a much lower estimate of the theoretical maximum number of suitable territories, and the USFWS to a much lower estimate of actual breeding pairs. A lower number would also be more consistent with a recent USFWS estimate for breeding pairs in Canada. In discussing its decision to list Queen Charlotte goshawks in British Columbia as threatened, USFWS estimated that, as of 2008, there were about 352 to 374 pairs of goshawks throughout the British Columbia (B.C.) distinct population segment (DPS).[314] However, the B.C. DPS inhabits twice as much productive old-growth forest (5.7 million ha) as the Southeast Alaska DPS (2.2 million ha).[315] It is not biologically plausible that half as much suitable habitat, in Southeast Alaska, would support as many pairs of the same subspecies as are found in neighboring and ecologically similar B.C. A substantially lower reproductive population would also be more consistent with

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[309] 72 Fed. Reg. at 63,127-28.

[310] *Id*.

[311] Schempf & Woods at 6 (explaining that "[c]ells with less than 20 percent of the land area in old-growth forest were excluded," but cells with more than that were counted).

[312] *See* Goshawk Conservation Assessment at PDF 43 ("a strong pattern for selection of very high to moderately productive old-growth forest with a combined 58 percent of all habitat use occurring in these cover types"); F. Doyle, *Breeding Success of the Goshawk (A. g. laingi) on Haida Gwaii/Queen Charlotte Islands: Is the Population Continuing to Decline* at 33 (Mar. 2005) (Doyle) ("[T]here appears to be a requirement for >41% and more typically >60% of the area to be in mature-old-growth forest, before goshawk breeding is detected in a landscape"). Doyle defined "mature" forest as 81-250 years old, and "old" forest as more than 250 years old. Doyle at 7.

[313] *See* Big Thorne FEIS at 3-132 to 3-133.

[314] 77 Fed. Reg. at 45,887.

[315] *See* U.S. Fish & Wildlife Service, Alaska Region, Juneau Fish and Wildlife Office, Updated Appendices Queen Charlotte Goshawk Status Review at 7 Table A-9 (May 2010).

Crocker-Bedford's 1994 estimate that there were at the time 100-200 breeding pairs of goshawks in all of Southeast Alaska.[316]

Further, due to continued logging of old-growth, it is highly probable that the Southeast Alaska population getting smaller. The Tongass National Forest reported having logged 6,996 acres between fiscal years 2007 and 2012.[317] And in addition to federal logging, habitat liquidation on other holdings has been considerable.[318] Compounding this problem, even if all old-growth logging in all of Southeast Alaska stopped today, the goshawk population would probably still continue to decline given lag times in population responses, particularly if a viability threshold has been crossed. As USFWS has explained, "goshawk populations may continue to decline for several years after logging of old-growth forests has ceased and timber harvest is restricted to second-growth stands because it is likely to take several generations for the populations to equilibrate with their modified environments."[319]

B. Goshawks on Prince of Wales Island Are More Vulnerable Than Elsewhere in the Tongass, Because of Natural Lack of Prey, Intensive Logging, Huge Territories, and Low Productivity.

1. Prince of Wales Island lacks important prey species for goshawks.

For a number of reasons, goshawks on Prince of Wales Island are more stressed, more sensitive, and more vulnerable to continued logging of old-growth forest than goshawks anywhere else in Southeast Alaska. Even without logging impacts, Prince of Wales Island and surrounding islands naturally lack important prey species. Red squirrels, which are significant prey for goshawks elsewhere in Southeast Alaska, are absent.[320] There is a species of flying squirrel in the area, but it is nocturnal, and essentially unavailable to goshawks (particularly during critical winter months).[321] Moreover, the long-term viability of flying squirrels themselves is in doubt on Prince of Wales Island, because small habitat reserves are too small to sustain their

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[316] D. Crocker-Bedford, Interagency Viable Population Committee for Tongass Land Management Planning, Conservation of the Queen Charlotte Goshawk in Southeast Alaska at 4 (May 5, 1994) (Crocker-Bedford).

[317] *See* U.S. Forest Service, 2012 Annual & Five Year Monitoring and Evaluation Report at 8 (May 2013).

[318] *See, e.g.*, USFWS Goshawk Review at 81 ("Intensive clear-cutting on large areas of corporation land [in Southeast Alaska] has converted many watersheds to very low quality habitat, or non-habitat, for goshawks. Loss of this habitat has likely contributed to at least local declines in goshawk populations").

[319] 72 Fed. Reg. at 63,136.

[320] Goshawk Conservation Assessment at PDF 65; Big Thorne FEIS at 3-133.

[321] USFWS Goshawk Review at 39.

populations in the absence of immigration, and the species' ability to disperse adequately across intervening spaces, whether naturally unforested or logged, is in doubt.[322]

Prince of Wales Island and the surrounding islands also lack sooty (blue) grouse, another key food source for goshawks elsewhere in Southeast Alaska.[323] Although spruce grouse inhabit the area, they are only about half the size of sooty grouse.[324] Further, logging has reduced the availability of spruce grouse to goshawks, because spruce grouse avoid clear-cuts, instead inhabiting 15- to 35-year-old second-growth,[325] where goshawks are at a distinct disadvantage because of the density of the tree stems, which "interfere with flight lines and decrease hunting success."[326] As a result, "[r]esearchers have identified food stress as a limitation for goshawks on Prince of Wales Island and surrounding islands in southern Southeast Alaska . . . ."[327] For example, "[m]ost females that died [of starvation] during Flatten et al.'s (2002) study were from the Prince of Wales area, which lacks red squirrels, hares and sooty grouse to support goshawks during winter (Titus et al. 2002)."[328]

2. Aggressive logging has affected disproportionately Prince of Wales Island.

Goshawks are also particularly at risk on Prince of Wales Island, because its forests have been disproportionately targeted by the timber industry:

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[322] *See* W.P. Smith & D. K. Person, *Estimated persistence of northern flying squirrel populations in temperate rain forest fragments of Southeast Alaska*, Biological Conservation 137:626-636 (2007) at PDF 1; *see also* PR 769_05_000465 at PDF 9 (E. Flaherty, *et al.*, *Diet and Food Availability: Implications for Foraging and Dispersal of Prince of Wales Northern Flying Squirrels Across Managed Landscapes*, JOURNAL OF MAMMOLOGY 91(1):79-91 (2010) at PDF 9 ("Our results suggest low availability of potentially critical food items in managed habitats, which may constrain dispersal of [flying squirrels on Prince of Wales Island]").

[323] USFWS Goshawk Review at 39; Goshawk Conservation Assessment at PDF 65.

[324] USFWS Goshawk Review at 39.

[325] *Id.* at 63.

[326] *Id.* at 36.

[327] 72 Fed. Reg. at 63136.

[328] USFWS Goshawk Review at 41.

> Timber harvest has not been evenly distributed across the Tongass NF. There are 21 biogeographic provinces within the Tongass NF (USDA Forest Service 1996a), and several have had little or no harvest (e.g., Admiralty Island and the mainland provinces). Other provinces have had substantial timber harvest activity (e.g., northeast Chichagof and Prince of Wales Islands).[329]

Already by 1995, more than 20 percent of the old-growth forest of northern Prince of Wales Island had been logged.[330] This percentage was almost twice as high as the next most-logged biogeographic provinces.[331] For context, the Goshawk Conservation Assessment concluded that "[h]arvest rates exceeding 13 percent [by 1995] . . . represent[ed] increased risk to long-term goshawk persistence."[332]

Subsequent logging on both federal and non-federal lands has substantially worsened the situation by contributing to cumulative effects. In 2007, the USFWS reported that:

> Threats to the Queen Charlotte goshawk in Southeast Alaska are greatest on Prince of Wales Island and the surrounding smaller islands at the southern end of the DPS. Timber harvest on both the Tongass National Forest and native corporation lands has been intensive in some parts of this area. Approximately 26 percent of the productive forest on Prince of Wales and the surrounding islands has been harvested, including some of the most productive forest lands in Southeast Alaska (Albert and Schoen 2006, pp. 15-18).[333]

3. Goshawks maintain larger territories and experience lower productivity on Prince of Wales Island than anywhere else in Southeast Alaska.

Loss of goshawk habitat and lower prey abundance on Prince of Wales Island (and nearby islands) has had a serious adverse impact on territory size and nesting productivity. USFWS reports that goshawks on Prince of Wales Island experience "comparatively low goshawk nesting

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[329] Goshawk Conservation Assessment at PDF 13; *see also* 2008 FEIS at 3-201 (indicating that North Central Prince of Wales Island has been logged far more heavily than any other Biogeographic Province); 72 Fed. Reg. at 63,131 ("Corporate lands, which cover only 3 percent of the total area of Southeast Alaska but include 7 percent of the region's 6.4 million ac (2.6 million ha) of productive forest, are distributed throughout Southeast Alaska, with concentrations on and near Prince of Wales Island . . .").

[330] Goshawk Conservation Assessment at PDF 80, Table 26.

[331] *Id.*

[332] *Id.*

[333] 72 Fed. Reg. at 63,136.

densities and lower reproductive success than elsewhere in the DPS."[334] Similarly, the Goshawk Conservation Assessment noted that goshawk territories on Prince of Wales Island are larger than elsewhere in Southeast Alaska.[335] Indeed, the very large individual home ranges mentioned above are from Prince of Wales Island.

Loss of habitat, in particular, affects goshawk densities (i.e., territory sizes) and reproductive success through several mechanisms:

> Several factors may contribute to decreased productivity and density in goshawk populations following particular changes in forest structure and composition: (1) increased predation on adults and young goshawks as hiding cover is reduced and potential predator populations increase (e.g., great horned owls); (2) loss of cool thermal conditions at nest sites; (3) reduced prey abundance or availability, or both; (4) increased competition as predators that adapt to more open forest become abundant; and (5) increased disturbance and human-caused mortality due to increased access from the timber harvest road network.[336]

C. <u>The DEIS Does Not Disclose the Precarious State of and Serious Risks to the Viability of Goshawks on the Tongass.</u>

The DEIS wholly fails to disclose or examine the serious risks to goshawks on the Tongass, either the pre-existing risks or the ways in which the various alternatives would aggravate them, in any fashion that would alert the public or decision-makers to them. In fact, it offers only four paragraphs to describe the goshawks' biological and ecological needs.[337] It fails to reference most of the scientific literature discussed in this section of the comment letter.[338] The impacts analysis is a paltry six paragraphs and never even addresses the fundamental changes proposed in the Draft Forest Plan.[339]

For example, the 2008 Amended Forest Plan classifies areas within the beach[340] and estuary fringe[341] "as unsuitable for timber harvest."[342] It establishes several important objectives regarding these areas:

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[334] 72 Fed. Reg. at 63,136-37.

[335] Goshawk Conservation Assessment at PDF 74.

[336] *Id.* at PDF 27.

[337] *See* DEIS at 3-211 to 3-212.

[338] *See id*.

[339] *See id*. at 3-240 to 3-241.

[340] The beach fringe is an area of approximately 1,000 feet slope distance inland from mean high tide around all marine coastlines. 2008 Amended Forest Plan at 4-4.

> 4. To maintain an approximate 1,000-foot-wide beach fringe of mostly unmodified forest to provide important habitats, corridors, and connectivity of habitat for eagles, goshawks, deer, marten, otter, bear, and other wildlife species associated with the maritime-influenced habitat. Old-growth forests are managed for near-natural habitat conditions (including natural disturbances) with little evidence of human-induced influence on the ecosystem.
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> 5. To maintain an approximate 1,000-foot-wide estuary fringe of mostly undisturbed forest that contributes to maintenance of the ecological integrity of the biologically rich tidal and intertidal estuary zone. Habitats for shorebirds, waterfowl, bald eagles, goshawks, and other marine-associated species are emphasized. Old-growth conifer stands, grasslands, wetlands, and other natural habitats associated with estuary areas above the mean high tide line are managed for near-natural habitat conditions with little evidence of human-induced disturbance.[343]

The Forest Service concluded that the "beach fringe was a very key feature of the overall Tongass conservation strategy," particularly with regard to goshawks.[344] As Chris Iverson, the author of Appendix N to the 1997 FEIS, explained:

> The most compelling argument for this extended beach fringe is that this zone of 1000 feet from the shoreline is a landscape region significantly selected by goshawks, for foraging we presumed, during our habitat selection analysis (see Goshawk [Conservation] Assessment, Figure 9, pages 52-53). When the leadership (Forest Supervisors, RF, IDT Leader) were presented with this graph and statistic -the decision was made to extend the beach fringe to 1000' to provide additional risk reduction and confidence in goshawk conservation to contribute to a not warranted decision by the FWS for the listing petition that they were considering at the time.[345]

The Draft Forest Plan deletes the portions of the standards and guidelines that prevented logging in the beach (and estuary) fringe.[346] In their place, the agency proposes a Forest-wide standard

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[341] The estuary fringe is an area of approximately 1,000 feet slope distance around all identified estuaries. *Id*. at 4-4.

[342] *Id*. at 4-5.

[343] *Id*. at 4-4.

[344] 2008 TLMP 603_1127 at 2.

[345] *Id.*

[346] PR 769_01_000088 (Redlined Version of Draft Forest Plan) at 4-5.

that prevents most old-growth logging in these areas (with several exceptions that do not count towards the projected timber sale quantity),[347] but the proposed plan now allows second-growth logging.[348] The DEIS, however, fails to examine the impacts of this change on the underlying conclusions regarding goshawk viability. In so doing, the agency violates NFMA and NEPA.

The DEIS also never examines how the proposed alternative will adversely affect the "three critical spatial components of the nesting home range:"[349] nest area, post-fledging family area, and foraging area.[350] As Dr. Winston Smith made clear in his comments, it also fails to examine the risks to goshawks in light of more recent science:

> First, spatially explicit analyses of contributions to northern goshawk breeding-season habitat revealed that conservation measures of the Tongass Land and Resource Management Plan contribute about half the secure habitat recommended for post-fledging areas of breeding pairs in the southern portion of this species range (Reynolds et al. 1992) and was less than half the relative amount of habitat documented in nest areas in Southeast Alaska. A similar conclusion was obtained for the broader landscape (21 km2) that surrounded each nest. This is because much of the habitat across the landscape has been clear-cut-logged and half the remaining choice habitat is in the Development land-use designation available for timber harvest.
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> Secondly, guidelines developed for northern goshawk populations in the southwestern United States may underestimate habitat needed by breeding pairs in Southeast Alaska.
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> . . .
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> In Southeast Alaska, the predominant (frequency and biomass) prey items during the breeding season (Lewis et al. 2006) are bird and mammal species that are most abundant, or occur exclusively, in productive old-growth forests (Iverson et al. 1996,Smith et al. 2001, 2004, 2005). Consider further that the mammal fauna of Southeast Alaska is depauperate (Smith 2005); few mammal species exclusively occur in low-volume or managed forests of Southeast Alaska (Smith et al. 2001, Smith and Nichols 2004); and the structure of dense second-growth stands effectively renders

[347] Draft Forest Plan at 5-17 (S-BEACH-01)

[348] *See* Draft Forest Plan at 5-8.

[349] Smith Goshawk Comments at 3.

[350] *See generally* DEIS at 3-240 to 3-241.

> prey unavailable to foraging goshawks (Reynolds et al 1992, 2006). Avian communities in managed forests include few, if any, additional prey for northern goshawks (Smith et al. 2001). Thus, breeding pairs in managed landscapes of Southeast Alaska likely rely almost entirely on productive old-growth forests as foraging and nesting habitat. That breeding pairs in managed landscapes of Southeast Alaska depend on productive old-growth forests to meet life-history needs was reflected in the findings of compositional analyses and radio-telemetry studies, both of which determined that northern goshawks strongly selected medium- and high-volume old-growth forests, and avoided recently managed or non-forested habitats [Goshawk Conservation Assessment].[351]

These challenges are exacerbated by the Forest Service's choices for second-growth management. "The potential for second-growth stands to become useable habitat over the Tongass planning horizon is limited because unmanaged second-growth typically requires at least 300 years following disturbance to develop old-forest features (Nowacki and Kramer 1998)."[352] Rather than manage second-growth in a way that returns it to old-growth characteristics, the Forest Service is targeting second-growth for commercial purposes in critical old-growth reserves, Beach-Estuary Fringe, and Riparian Management Areas. As explained above, these areas were set aside as reserves because they represented important habitat and they were considered critical to the long-term viability of many wildlife species across the forest, including goshawks.

To the extent the Forest Service suggests that adverse impacts are ameliorated because the clear-cutting of these areas will be in 10-acre or smaller blocks, the position is utterly unsubstantiated. The DEIS offers no scientific support for its suggestion that a ten-acre clear-cut does not set back the ecological structure and function of old-growth in ways that adversely affect not just goshawks, but other old-growth dependent species.[353]

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Further, the DEIS fails to analyze the adverse impacts on goshawks caused by clear-cutting old-growth forests for at least another 16 years and likely much longer. The Forest Service has not evaluated the impacts caused by the loss of habitat quality or taken into account all available information on differential utilization of various forest types and structures, and cumulative effects of past and foreseeable activities affecting habitat. The Forest Service historically based its viability conclusions on the fact that habitat was maintained and viability ensured by forest plan standards and guidelines governing, among other factors, riparian and beach buffers, old-growth reserves, etc. The Draft Forest Plan overturns that paradigm, allowing logging in the very areas that the agency's own experts considered the essential minimums of the agency's

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[351] Smith Goshawk Comments at PDF 9.

[352] *Id*. at PDF 9-10 (emphasis omitted).

[353] *See* DEIS at 3-240 to 241.

conservation strategy. To do so without sufficient scientific basis and explanation violates NFMA and its regulatory requirements. To perpetuate such a position in an environmental impact statement would violate NEPA. The Forest Service must correct these fundamental failings and chart a course that ensures the continued viability of goshawks.

III. NORTHERN FLYING SQUIRREL

A. The Tongass is Home to Multiple Subspecies, Including the Endemic Prince of Wales Flying Squirrel.

"The northern flying squirrel itself has multiple subspecies represented in Southeast Alaska, including the endemic Prince of Wales flying squirrel, whose geographic range is limited to Prince of Wales Island and near-shore islands (Prince of Wales Island Biogeographic Province; Smith 2005)."[354]

Even as a single taxon across Southeast Alaska, *G. sabrinus* was ranked second after northern goshawk with respect to concerns over viability.[355] Revised analysis by the VPOP committee ranked northern flying squirrel as a species of concern.[356] Northern flying squirrel was a "design" species for small size old-growth reserves (<10,000 acres [4,050 hectares]) in the 1997 TLMP because of their assumed "dependency on the forested habitats".[357]

The squirrel on Prince of Wales (*G.s. griseifrons*) has been listed as a subspecies of ecological concern in the Tongass National Forest and as potentially endangered in the Status Survey and

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[354] W. Smith, Proposed Forest Plan Amendment Further Compromises Established Conservation Measures to Sustain Viable Populations of Endemic Small Mammals at PDF 2-3 (Feb. 2016) (Smith Small Mammals Comments); *see also* 2008 TLMP AR 603_1419

(Bidlack, A.L. & J.A. Cook, *Reduced genetic variation in insular northern flying squirrels (Glaucomys sabrinus) along the North Pacific coast*, ANIMAL CONSERVATION 4:283-290 (2001)); 2008 TLMP AR 603_1420 (Bidlack, A.L. & J.A. Cook, *A nuclear perspective on endemism in northern flying squirrels (Glaucomys sabrinus) of the Alexander Archipelago, Alaska,* Conservation Genetics 3:247-259 (2002)); 2008 TLMP AR 603_0947 (Demboski, J. R. *et al.*, *Implications of cytochrome b sequence variation for biogeography and conservation of the northern flying squirrels (Glaucomys sabrinus) of the Alexander Archipelago, Alaska,* Can. J. Zool. 46:1771-1777 (1998)). The undersigned groups incorporate the Smith Small Mammals Comments into this letter in their entirety, including all cited publications.

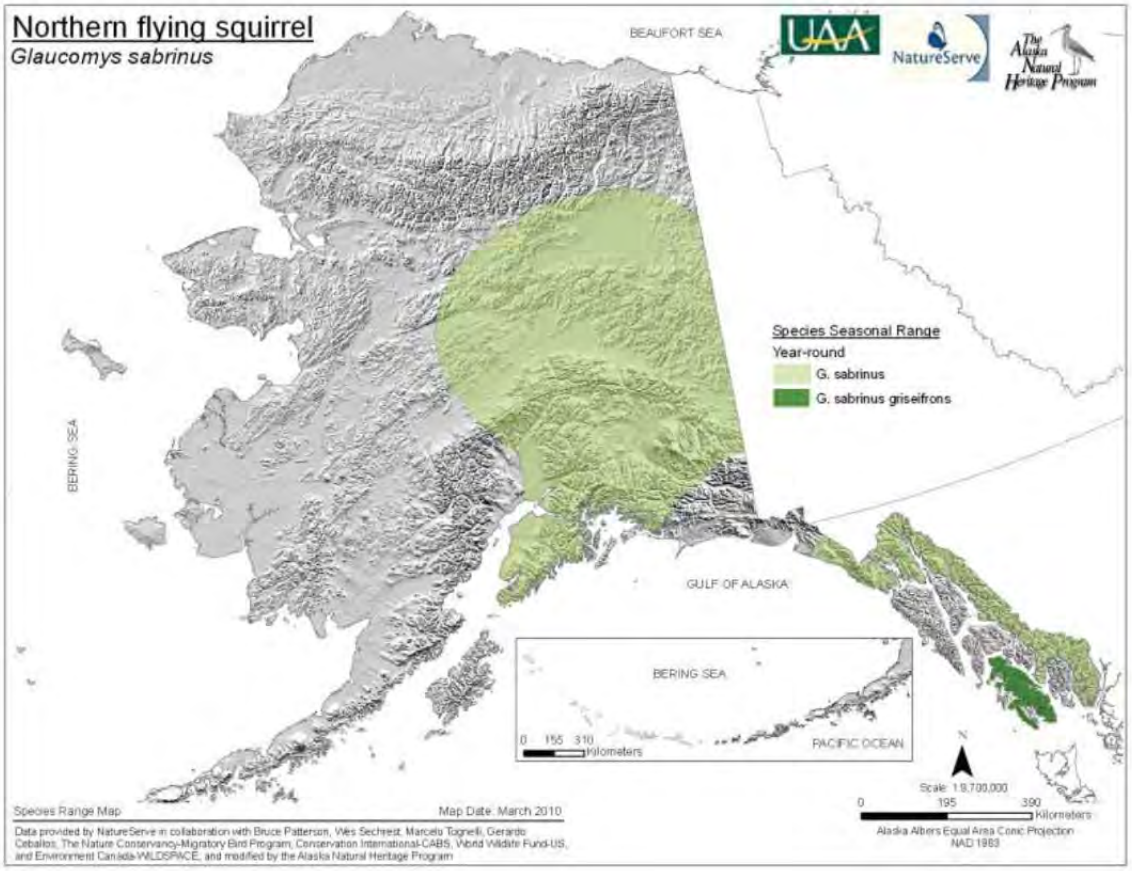
[355] 2008 TLMP AR 603_0008 at PDF 56 (L. Suring *et al.*, *A proposed strategy for maintaining well-distributed, viable populations of wildlife associated with old-growth forest in Southeast Alaska* (1993)) (Suring et al. 1993).

[356] *Id.*

[357] Schoen, J. *et al.*, *Northern Flying Squirrel (Glaucomys sabrinus), in* Southeast Alaska Conservation Assessment (2007) at 2 (citing Suring et al. 1993).

Conservation Action Plan for North American Rodents prepared by the International Union for the Conservation of Nature.[358]

The following map demonstrates the general pattern of isolation of insular flying squirrels in Southeast Alaska:[359]



The northern flying squirrel not only achieves its highest densities in old-growth forest, "it is a sentinel of ecological processes at multiple spatial scales."[360] As Dr. Smith explains:

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[358] *Id.*; *see also* Demboski, J. R. *et al.*, *Glaucomys sabrinus (Shaw 1801), Northern flying squirrel, in* North American Rodents: Status Survey and Conservation Action Plan, IUCN/SSC Rodent Specialist Group 37 (D. J. Hafner et al. eds., 1998) at PDF 7-8; NatureServe, Comprehensive Report Species - Glaucomys sabrinus griseifrons and National, State, and Provincial Conservation Status Rank Definitions, www.natureserve.org at PDF 1, 7 (last visited Feb. 19, 2016).

[359] Walton, K. *et al.*, Alaska Species Ranking System Summary Report - Northern flying squirrel, Prince of Wales (Jan. 1, 2013) at PDF 3.

[360] Smith Small Mammal Comments at PDF 3 (citations omitted).

> [H]ealthy northern flying squirrel populations throughout a watershed indicate a healthy forest ecosystem with functioning ecological communities. In temperate rainforests, northern flying squirrels facilitate an obligate symbiotic relationship between young conifer seedlings and belowground fungi, such as truffles. Young seedlings (and thus the rainforest) depend on truffles for nitrogen-fixing bacteria and other nutrients to develop. Flying squirrels, which forage on truffles, distribute the spores of truffles in their fecal pellets as they glide around through the forest nightly on foraging forays searching for fungal communities. Thus, the northern flying squirrel essentially contributes to the development of its own habitat. In Southeast Alaska, it is important prey for American marten and northern goshawks.[361]

This relationship to the larger overall ecosystem manifests itself in the flying squirrels' demonstrated sensitivity to clear-cut logging.[362] For example, clear-cuts and second-growth adversely affect squirrel orientation.[363] "Dispersing squirrels likely will not venture into managed habitats because logging creates clear-cuts larger than the perceptual range of these mammals." [364] The "[c]ost of quadrupedal locomotion, the mode of transport adopted by squirrels in clear-cut and [seco]nd-growth stands, is higher than expected . . . . Thus, unless flying squirrels are able to replenish their depleted energy stores when dispersing across high-cost managed stands, successful dispersal and viable metapopulations are unlikely (Smith and Person 2007; Smith et al., in press)."[365] As a result, second-growth and clear-cut stands are "energetically expensive habitats."[366]

For these reasons, "[r]estoring old-growth forest structure, especially large trees or snags, is fundamental to sustaining female productivity and overall population density because of the reliance upon cavities as natal dens." [367] Studies by Pyare and Smith have showed that Prince of Wales flying squirrels moved with least resistance "across large expanses of old-growth and old-

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[361] Smith Small Mammal Comments at PDF 3 (citations omitted).

[362] *See, e.g.*, G. Holloway & W. Smith, *A Meta-Analysis of Forest Age and Structure Effects on Northern Flying Squirrel Densities*, Journal of Wildlife Management 75(3): 668-674 (2011) (Holloway & Smith 2011).

[363] *Id.*

[364] PR 769_05_000464 at 1 (Flaherty, E.A. *et al.*, *Experimental trials of the northern flying squirrel (Glaucomys sabrinus) traversing managed rainforest landscapes: perceptual range and fine-scale movements,* Can. J. Zool. 86:1050-1058 (2008)).

[365] PR 769_05_000465 at 1(Flaherty, E. *et al.*, *Diet and Food Availability: Implications for Foraging and Dispersal of Prince of Wales Northern Flying Squirrels Across Managed Landscapes*, JOURNAL OF MAMMOLOGY 91(1):79-91 (2010)) (Flaherty 2010).

[366] Flaherty, *Diet and Food availability*.

[367] PR 769_05_000478 at 9 (Pyare, S. *et al.*, *Den use and selection by northern flying squirrels in fragmented landscapes*, JOURNAL OF MAMMALOGY, 91(4):886-896 (2010)).

growth fragments, followed by second-growth up to 100 meters wide, with the most resistance in regenerating clear-cuts and young second-growth greater than 100 meters wide. Notably, while males appeared to have a high movement potential in fragmented landscapes, females and juveniles did not."[368]

B. The DEIS All But Ignores Impacts to Flying Squirrels.

The DEIS offers a single paragraph to explain the impacts of the proposed changes to the forest plan to flying squirrels.[369] The DEIS ignores virtually all contemporary science regarding flying squirrels. For example, the agency ignores almost every publication Dr. Smith identifies in his comments.

The DEIS also perpetuates the mistaken suggestion that because logging on the Tongass as a whole has taken place at lower levels than those originally contemplated, then wildlife viability must be ensured. To the contrary, as the agency knows, logging on several parts of the Tongass are approaching maximum implementation, including Prince of Wales Island and especially the Big Thorne project area.[370] In fact, "more than 50% of the most productive old-growth forests have been clear-cut logged on [Prince of Wales Island], the majority of which occurred across north central [Prince of Wales], substantially reducing and fragmenting the most productive habitat of the [Prince of Wales] flying squirrel (Smith and Nichols 2003, Smith and Person 2007, Smith et al. 2011)."[371]

To be clear, as the Smith Mammals Comments explain:

> [F]urther loss of old-growth forests and active management of second-growth stands, especially in existing riparian buffers or other conservation elements, will increase the risk to viability of northern flying squirrels and other endemic small mammals. Additional regeneration harvests (i.e., clear-cuts) will set back succession toward suitable breeding habitat and create a barrier to dispersal . . . Therefore, any forest plan amendment or revision that proposes to continue the harvest of old-growth forest or impose clear-cuts in buffers or other conservation elements without including a comprehensive analysis of the Wildlife Conservation Strategy is imprudent and irresponsible as it ignores the best available credible science.[372]

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[368] 2008 TLMP FEIS at 3-222 to 3-323 (citing Pyare and Smith (2005, 2006)).

[369] DEIS at 3-259.

[370] Big Thorne Project Final Supplemental Information Report at PDF 8 (Aug. 2014).

[371] Smith Small Mammals Comments at PDF 5.

[372] Smith Small Mammals Comments at PDF 9.

To the extent the Forest Service continues to pursue a transition alternative that has the agency logging old-growth for another 10 to 15 years or more, the Forest Service must address, for the first time in most instances, the science questioning the efficacy of the agency's flying squirrel management. The Forest Service has not evaluated the impacts caused by the continued loss of habitat quality, especially on Prince of Wales, or taken into account all available information on differential utilization of various forest types and structures, and cumulative effects of past and foreseeable activities affecting habitat and the resulting ability to ensure the continued viability of flying squirrels on the Tongass. To do otherwise and without sufficient scientific basis and explanation, the agency will violate NFMA and its regulatory requirements. To present misleading or incomplete information in an environmental impact statement would violate NEPA.

IV. SALMON

The Draft Forest Plan poses risks to salmon, which depend on intact habitat in the Tongass during much of their lifecycles. Salmon are keystone species in aquatic ecosystems and provide a food base for terrestrial animals.[373] Local communities rely on healthy and abundant populations for commercial, recreational, and subsistence harvests.[374] Given these fishes' importance to Tongass ecosystems, traditional ways of life, and the local economy, it is crucial that the DEIS present a complete picture of potential impacts on salmon.

A. Logging Adversely Affects Salmon Species.

The destruction of forest stands, particularly old-growth, can have immediate and lasting impacts on streams flowing through the Tongass. The Forest Service's preferred alternative would leave some 260,000 acres of old-growth available for timber production and might achieve a transition to predominantly second-growth harvest only after 16 years, but likely much longer.[375] It would also significantly weaken current protections of forested areas along streams and fringes of beaches and estuaries[376]—the areas nearest fish habitat. It is therefore essential that the DEIS fully disclose the harms logging could cause salmon populations.

Old-growth logging near streams may have more-serious long-term impacts on salmon than second-growth logging. A recent study that analyzed data from stream sampling throughout the Tongass over two decades found that juvenile coho salmon were denser in old-growth riparian zones than in new growth zones.[377] It also noted long-term declines of coho salmon in second-

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[373] PR 769_05_000051 at PDF 16 (Haufler, J.B. *et al.*, *Climate change: anticipated effects on ecosystem services and potential actions by the Alaska Region*, U.S. Forest Service, Ecosystem Management Research Institute (2010)) (Haufler).

[374] Haufler at PDF 16.

[375] DEIS at 2-32; *id.* at 2-43, Table 2-18.

[376] DEIS at 2-31; *id.* at 2-42, Table 2-17.

[377] PR 769_05_000816 at PDF 5, 14 (Bryant, M.D. & B.E. Wright, *An analysis of juvenile salmonid densities from a diverse long-term data set*, Manuscript)) (Bryant & Wright). Juvenile

growth sites, a trend possibly attributable to reductions in large wood and pools because of riparian logging.[378] The authors acknowledged that their results were based on logging under forest management practices less protective than those in place,[379] and the Forest Service thus downplays the study's results.[380] Yet it raises important questions about the Draft Forest Plan's treatment of riparian zones.

The 2008 Amended Forest Plan includes riparian management areas generally free of logging, designed to protect sensitive ecological functions along both fish streams and fishless headwater streams.[381] Alternative 5, by contrast, would allow patch cuts of up to ten acres and 35 percent of the stand outside a statutorily required 100-foot buffer in riparian management areas and a 200-foot buffer on shorelines.[382] Although the Tongass Timber Reform Act (TTRA) prohibits commercial logging within 100 feet of all Class I streams (habitat for anadromous fish) and Class II streams (habitat for non-anadromous fish) that flow directly into Class I streams, it does not prevent logging immediately adjacent to Class III or Class IV streams that may flow into higher-class waters.[383] Such "[s]mall perennial and intermittent non-fish streams are especially important in routing water, sediment, and nutrients to downstream fish habitats."[384]

1. Logging results in elevated stream temperatures.

One way logging harms salmon is by elevating stream temperatures. The Service recognizes that heating streams lowers dissolved oxygen levels in streams and reduces salmon production and survival.[385] Fry emerge from eggs earlier in warmer water —up to 22 days earlier, even in streams with buffer strips—exposing them to floods in early spring.[386] In turn, premature smolt emigration may reduce adult yield because plankton food sources are less abundant earlier in the

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salmonid communities may also be more diverse in basins subject to a lower percentage of harvest. *See* G. H. Reeves *et al.*, *Fish Communities*, *in* River Ecology and Management: Lessons from the Pacific Coastal Ecoregion 200 (R. Naiman *et al.* eds., 1998), at PDF 23.

[378] Bryant & Wright at PDF 16.

[379] *Id.* at PDF 20.

[380] DEIS at 3-105.

[381] DEIS at 3-58.

[382] DEIS at 2-31.

[383] *See* 16 U.S.C. § 539d(e); DEIS at 3-50.

[384] PR 769_05_000150 at PDF 78 (Murphy, M.L., *Forestry impacts on freshwater habitat of anadromous salmonids in the Pacific Northwest and Alaska-requirements for protection and restoration*, NOAA Coastal Ocean Program Decision Analysis Series No. 7 (Oct. 1995)) (Murphy 1995).

[385] DEIS at 3-101.

[386] PR 769_05_000243 at PDF 11-12 (Murphy, M.L., & A.M. Milner, *Alaska Timber Harvest and Fish Habitat*, *in* Freshwaters of Alaska, Ecological Syntheses (A.M. Milner & M.W. Oswood eds,. 1997)) (Murphy 1997).

year.[387] The Service minimizes these potential impacts, citing Southeast Alaska's cool climate.[388] As discussed below, however, the climate is changing; and, even at present, "temperature sensitive" streams in the region can become excessively warm when shade canopy is removed.[389]

Although the DEIS acknowledges that logging can raise water temperatures by reducing streamside shade, it points to an industry study that found that streams in previously logged Tongass watersheds were not significantly warmer than in unlogged areas.[390] It also cites a study suggesting that relatively narrow (10 to 30 meters) riparian buffers may prevent temperature increases.[391] That study notes, however, that the buffers observed may have been especially effective because they shaded north-south streams, and that the results "contrast with those from some other studies, where substantial postharvest warming was observed despite retention of riparian buffers."[392] A review of the literature concluded that loss of canopy density may elevate stream temperatures, especially in summer when stream flow is low and air temperatures high.[393]

The total lack of buffers along Class III and IV streams under Alternative 5 would allow temperature increases in those streams, which may account for more than half of the total channel length in many watersheds in Alaska.[394] In light of the potentially significant increases in water temperatures in downstream salmon habitat, the Forest Service must analyze the impacts of failing to retain or improve existing buffers along all water bodies.

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[387] Murphy 1997 at PDF 16.

[388] DEIS at 3-101.

[389] Murphy 1997 at PDF 11.

[390] DEIS at 3-53; PR 769_05_000136 at PDF 33-35 (Konopacky Environmental, *Water turbidity in streams, related to the installation of road culverts of various diameters for timber harvest activities on the Tongass National forest, Southeast Alaska, during 1994 and 1995*, Presentation Synopsis (Feb. 1996)).

[391] DEIS at 3-53; *see also* Murphy 1995 at PDF 78, 81 (noting that trees about as far from a stream as they are tall provide shade, and that a buffer of about 30 meters may suffice).

[392] PR 769_05_000118 at PDF 8 (Gomi, T. *et al.*, *Headwater stream temperature response to clear-cut harvesting with different riparian treatments, coastal British Columbia, Canada*, Water Resources Research 42:W08437 (2006)).

[393] PR 769_05_000104 at PDF 17 (Belt, G.H. *et al.*, *Design of forest riparian buffer strips for the protection of water quality: Analysis of scientific literature*, Idaho Forest, Wildlife and Range Policy Analysis Group Report No. 8 (June 1992)) (Belt).

[394] Murphy 1995 at PDF 78.

2. *Logging increases stream flow, which is detrimental to fish.*

High water velocities can overturn streambeds, killing eggs that are incubating there.[395] For this reason, higher peak flows are generally considered detrimental to fish.[396]

Logging elevates peak stream flow by compacting soil and accelerating snowmelt.[397] The results may be long lasting: one study found only minor reductions in increased peak flow caused by timber harvest after 30 years.[398] The DEIS acknowledges that logging could increase peak flows in combination with roads.[399] Yet it offers no estimate of these changes, instead only listing maximum logging acres by alternative under the Draft Forest Plan, which "provides a general indication to assess projected effects on water quantity."[400] It is largely silent on the potential effects of higher peak flow on fishes.[401]

It may be true that stream flows in particular subwatersheds can only be estimated during project planning, given the complex interactions among logging, road construction, and local hydrological characteristics.[402] Nevertheless, given the potentially serious impacts that excessive stream flows may have on salmon, the Forest Service must consider and disclose information on this issue in the FEIS, potentially including modeling of representative watersheds.

3. *Logging increases sediment input in streams.*

Logging can also degrade salmon habitat by increasing sediment input. Fine sediment reduces egg-to-fry survival, food production, summer rearing area, and winter survival.[403] On a more granular level, sediment covers spawning gravel, smothering eggs or fry, and eliminating suitable rearing habitat and substrates suitable for macroinvertebrates.[404] In extreme cases, it can

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[395] Murphy 1997 at PDF 13.

[396] Murphy 1997 at PDF 13.

[397] Murphy 1997 at PDF 13.

[398] C. A. Troendle & R. M. King, *The Effect of Timber Harvest on the Fool Creek Watershed, 30 Years Later*, 21 Waters Resources Res. 1915 (1985), at PDF 8.

[399] DEIS at 3-62 to 64.

[400] DEIS at 3-64 & Table 3.4-3.

[401] *See* DEIS at 3-101 to 104; *id.* at 3-115 to 116.

[402] *See* DEIS at 3-64 to 65.

[403] Murphy 1995 at PDF 63.

[404] PR 769_05_000213 at PDF 7-8 (Furniss, M.J. *et al.*, *Road Construction and Maintenance*, *in* Influences of Forest Rangeland Management on Salmonid Fishes and Their Habitat, American Fisheries Society Special Publication 19:297-323 (W.R. Meehan ed., 1991)) (Furniss).

reduce egg survival or smolt yield by more than 80 percent.[405] Suspended materials (turbidity) also cause stress and avoidance behavior in salmon, even at low levels.[406] Larger, coarse sediment such as gravel and cobbles can clog stream channels and scour streambeds.[407]

Logging can add sediment to streams through a number of mechanisms. Small landslides triggered on clear-cut slopes—more than four times more frequent after logging[408]—add large quantities of sediment to streams.[409] The DEIS estimates that the alternatives considered may cause an additional 13 landslides over 25 years, compared to the current forest plan.[410] Erosion from felling and yarding along stream banks may also cause significant scour and deposition of sediment in streams.[411] The DEIS admits that "harvest in [riparian management areas] along Class III channels . . . may add sediment to streams that could be transported to fish streams, impacting rearing and spawning habitat," and that "harvest along lakeshores could also contribute to sediment increases, potentially reducing lake fish production."[412]

Scientists have recommended a 100-foot buffer to filter sediment,[413] yet the preferred alternative would allow logging up to the edge of Class III and IV streams that may flow into salmon habitat. It would also allow logging immediately adjacent to lakes.[414] To reduce harmful effects, we urge the Forest Service to retain or improve existing protections in riparian and shoreline areas. Further, although we support a shift away from old-growth logging, the transition should not come at the expense of salmon habitat through ill-considered second-growth logging and resultant landslides.

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[405] Murphy 1995 at PDF 63; *see also* D. W. Jensen *et al.*, *Impact of Fine Sediment on Egg-to-Fry Survival of Pacific Salmon: A Meta-Analysis of Published Studies*, Reviews in Fisheries Sci. 17(3):348 (2009), at PDF 9 (noting a "steep decrease in [egg-to-fry survival of Chinook salmon and steelhead], leveling out at less than 10% when fines were greater than 25%").

[406] D. S. Lloyd, *Turbidity as a Water Quality Standard for Salmonid Habitats in Alaska*, 7 N. Am. J. of Fisheries Mgmt. 34 (1987), at PDF 2-4.

[407] B. J. Hicks, *Gravel Galore: Impacts of Clear-Cut Logging on Salmon and Their Habitats*, *in* Ghost Runs: The Future of Wild Salmon on North and Central Coasts of British Columbia (B. Harvey & M. MacDuffee eds., 2002), at PDF 106-07.

[408] Murphy 1997at PDF 9.

[409] Murphy 1995 at PDF 60.

[410] DEIS at 3-69.

[411] Murphy 1997 at PDF 9.

[412] DEIS at 3-118.

[413] Murphy 1995 at PDF 81.

[414] DEIS at 3-73.

4. *Logging reduces large woody debris.*

The DEIS cites numerous studies documenting the various benefits large woody debris provides to fishes.[415] Large woody debris generally enhances fish habitat by providing cover and creating pools where juvenile salmon rear and adult migrating salmon rest.[416] It is especially important to salmon reproduction, as it forms gravel deposits free of fine sediments conducive to spawning.[417]

Logging within a riparian buffer directly reduces recruitment of large woody debris and adds floatable debris that can dislodge debris already present in the stream.[418] It may take 50 years or more for nearby trees to grow large enough to provide an adequate long-term supply of large woody debris.[419] Logging also indirectly removes in-stream large woody debris by increasing water velocities during peak stream flow and flooding.[420] The Service notes that the Tongass has significantly lower large woody debris in streams flowing through previously logged areas than through unlogged areas.[421] It discounts the importance of these findings, however, because "most of these watersheds were intensively logged under conditions that had no buffer strips on streams."[422]

Although the vast majority of large woody debris comes from within 100 feet of the stream, the DEIS suggests that even more-distant trees—perhaps as much as 140 feet away—may contribute and notes that some reduction in debris in stream channels "could occur over time if harvest occurs in [riparian management areas] outside of the TTRA buffer."[423] The Service further acknowledges that Class III streams also require large woody debris to function properly and may contribute to downstream fish habitat.[424]

The 100-foot TTRA buffer will not fully protect Class I streams from loss of large woody debris, and the ineffectual harvest limits along Class III and IV streams would not ensure delivery of this important habitat feature to salmon-bearing streams. Supply of large woody debris is thus another reason to keep existing buffers along water bodies in place, or expand them.

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[415] DEIS at 3-102.

[416] Belt at PDF 18; Murphy 1997 at PDF 14, 24.

[417] Murphy 1997 at PDF 13.

[418] Belt at PDF 18.

[419] Belt at PDF 18.

[420] Murphy 1997 at PDF 14.

[421] DEIS at 3-103.

[422] DEIS at 3-103.

[423] DEIS at 3-72; *see also* Murphy 1995 at PDF 78 (explaining large woody debris is "derived from a distance of about 1 tree height."), 82.

[424] DEIS at 3-115.

The FEIS must examine and account for these considerations.

5. Logging adversely affects food for salmon.

Logging also influences the availability of food to salmon. As streams are exposed to greater sunlight, algae production typically increases and invertebrates become more abundant.[425] In some Southeast Alaska watersheds, though, streams may be nutrient-limited and not respond to logging in this way.[426] Or, increased soil erosion and sedimentation ultimately obliterate such biological responses.[427] Where higher production does occur, after the first 20 years denser second-growth canopy can stifle production for a century.[428]

If lower production results from logging along Class III and IV streams in the Tongass, downstream salmon may suffer from a lack of food. The DEIS recognizes that "[s]mall streams, many of which are not fish-bearing, . . . supply nutrients that contribute substantially to larger streams," and "[s]mall streams in Alaska have . . . been found to substantially contribute food for fish to larger streams through downstream transport of terrestrial and aquatic prey directly and detritus resources indirectly for fish."[429]

Without adequate protection of Class III and IV streams, salmon may find less food in the streams of the Tongass. The Forest Service must protect riparian and shoreline buffers for this reason as well. Additionally, the FEIS must account for these concerns.

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Salmon populations in Southeast Alaska currently appear to be healthy,[430] but continued logging at the scales proposed in the Draft Forest Plan represents an ongoing threat. In addition, opening up shorelines and riparian areas to logging will expose these species to new, potentially significant harms. Alternative 5's already narrow fringe zones, separated from larger tracts of forest, may themselves deteriorate over time.[431] And, somewhat bafflingly, marine shorelines receive greater protection than stream banks even though sediment in marine environments

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[425] Murphy 1997 at PDF 20.

[426] *Id*. at PDF 20.

[427] *Id*. at PDF 23.

[428] *Id*. at PDF 23.

[429] DEIS at 3-104; *see also* PR 769_05_000280 at PDF 10 (M.S. & D.P. Gregovich, *Export of invertebrates and detritus from fishless headwater streams in southeastern Alaska: implications for downstream salmonid production*, Freshwater Biology 47:957-967 (2002) (Wipfli) ("Because fishless headwaters are so abundant in . . . coastal temperate rainforest, they may contribute substantially to the overall energy budgets of the fish-bearing habitats they flow into.").

[430] DEIS at 3-106.

[431] Murphy 1995 at PDF 85.

would be diluted, large woody debris is less important there, and shading is unlikely to affect water temperature.[432] The Forest Service must retain or improve existing protections in both riparian and shoreline areas to ensure the continued population viability of salmon, one of the Tongass' world-class values.

B. Roads Result in Adverse Effects on Salmon.

1. Road construction increases stream temperature.

As noted above, warmer water temperatures often hamper salmon reproduction and survival by decreasing the level of dissolved oxygen in the water and altering the timing of key life events.[433]

Roads constructed adjacent to streams may increase water temperatures, just as logging does.[434]

To the extent that roads are necessary for patch cutting along Class III and IV streams,[435] they will likely contribute to temperature increases in waters that may account for more than half of the total channel length in some watersheds.[436] To safeguard downstream salmon habitat from this detrimental effect, the Forest Service should consider not allowing logging near any water bodies.

2. Road construction increases stream flow.

As the DEIS notes, higher peak flow due to roads may cause channel erosion, bed scour, and streambed and bank instability, which generally degrades fish habitat.[437] Overturning the streambed also destroys fish eggs.[438]

The DEIS cites a substantial body of literature indicating that roads contribute to peak flows in various ways, often compounding the effects of logging.[439] For example, roads compact soil and reduce permeability; they also collect surface runoff in ditches and intercept subsurface flow.[440]

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[432] DEIS at 3-116 to 3-117.

[433] *See supra* notes 385-94 and accompanying text.

[434] DEIS at 3-53.

[435] DEIS at 3-67, Table 3.4-6 (listing miles of new and reconstructed roads in beach and estuary fringe and riparian management areas by alternative).

[436] Murphy 1995 at PDF 78.

[437] DEIS at 3-65.

[438] Murphy 1997 at PDF 13.

[439] DEIS at 3-65.

[440] Murphy 1997 at PDF 13.

When combined with clear-cutting, roads speed the delivery of water to channels during storm events and lead to much greater peak discharges than clear-cutting would by itself.[441]

To avert the harmful impacts of excessive stream flow on salmon, the Forest Service should adopt a plan that would involve fewer miles of newly constructed roads. Additionally, the FEIS must examine the adverse consequences associated with these concerns.

3. Roads increase sediment entering streams.

As discussed above, fine sediment introduced into streams can be particularly damaging to vulnerable salmon eggs and juveniles.[442] Excessive sedimentation or landslides caused by roads may also impede salmon migration.[443]

Roads add sediment to streams in a number of ways. Construction in steep terrain markedly increases soil erosion, and landslides in construction areas can be 300 times more frequent than in undisturbed forest and continue for decades after roads are built.[444] During use, road surfaces add large amounts of fine sediments to streams, with chronic sediment inputs nearly equal to those during construction.[445] The prolonged nature of the inputs and finer particle size can make sediment from surface erosion as harmful to stream biota as inputs from mass soil movements like landslides[446]; and the effects of sediment delivery are long-lasting, cumulative, and often difficult to remedy.[447] Finally, if stream crossings fail, extensive local scour and sediment deposition can occur, especially where stream flow is altered.[448]

The DEIS acknowledges that a 300-foot riparian buffer may be needed to control sediment delivery from roads to water bodies,[449] yet the preferred alternative would only establish a 200-foot buffer.[450] Again, the Forest Service must analyze the consequences of failing to retain or improve current buffers along water bodies to help mitigate the adverse impacts of

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[441] PR 769_05_000130 at PDF 970 (Jones, J.A. & Grant, G.E., *Peak Flow Responses to Clear-cutting and Roads in Small and Large Basins, Western Cascades, Oregon*, Water Resources Journal 32(4):956-974 (Apr. 1996)).

[442] *See supra* notes 403-13 & accompanying text.

[443] Furniss at PDF 7.

[444] Murphy 1995 at PDF 57-58.

[445] Murphy 1995 at PDF 57, 60.

[446] Furniss at PDF 3.

[447] Stephen C. Trombulak & Christopher A. Frissell, *Review of Ecological Effects of Roads on Terrestrial and Aquatic Communities*, 14 Conservation Biology 18 (1999), at PDF 5.

[448] Furniss at PDF 4.

[449] DEIS at 3-55; *see also* Belt at PDF 22.

[450] DEIS at 2-31.

sedimentations from roads on salmon habitat. The Service should consider options that would reduce the number of roads constructed by limiting logging to levels below those proposed. Although the DEIS notes that "current standards and guidelines . . . recommend avoiding building roads on slopes greater than 67 percent," it predicts that each alternative would result in such hazardous construction.[451] The FEIS must account for and examine the environmental consequences of the agency's decisions in this regard.

4. Habitat access and passage must be considered.

By definition, anadromous salmonids migrate over their lifecycles, frequently over long distances.[452] Even resident salmon commonly swim upstream and downstream in search of food and optimal habitat.[453]

The Forest Service recognizes that improperly located, installed, or maintained stream-crossing structures along roads may block fish migration.[454] According to the DEIS, the most common obstacles include vertical barriers, debris blockages, and high water velocities that may prevent smaller or juvenile fish from passing.[455] Additional difficulties include turbulent flow patterns that may disorient fish and a lack of resting pools below culverts.[456] "A single poorly installed culvert can eliminate the fish population of an entire stream."[457]

The DEIS discloses that an assessment of about 90 percent of fish stream road crossings in the Tongass revealed about 35 percent—mostly culverts—did not meet juvenile fish passage criteria.[458] The Forest Service's efforts to identify and remove, replace, or fix these crossing structures are commendable,[459] but the problem persists: only about half of recently installed culverts have been monitored for fish passage suitability.[460] The preferred alternative could lead to the construction of almost 2,000 stream crossings, with more than 400 on Class I streams.[461] Given the current limitations on monitoring, it is only reasonable to expect that many of these new crossings will not be regularly assessed for compliance with fish passage criteria. The

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[451] DEIS at 3-69 & tbl. 3.4-7.

[452] Furniss at PDF 17.

[453] Furniss at PDF 17.

[454] DEIS at 3-109.

[455] DEIS at 3-109.

[456] Furniss at PDF 17.

[457] Murphy 1995 at PDF 67.

[458] DEIS at 3-109.

[459] DEIS at 3-109 to 110.

[460] DEIS at 3-109.

[461] DEIS at 3-115, Table 3.6-4.

Forest Service should reduce the number of new stream crossings, by quickly transitioning out of industrial-scale old-growth logging in less than five years.

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The construction, use, and ongoing existence of forest roads can have severe, irremediable impacts on salmon habitat. Moreover, roads are known to compound the effects of the logging they facilitate. Alternative 5 would lead to the construction of 942 new miles of roads.[462] The Forest Service must consider options that would lessen this invasive activity and protect salmon from its serious, varied harms. The FEIS must examine and disclose the agency's rationale regarding its road construction decisions.

C. Hydroelectric Projects Harm Salmon.

Inappropriately designed or located hydroelectric dams may harm salmon at every stage of their lifecycle. Stream flow alterations from the operations can cause adult salmon to abandon their nests, and can kill eggs and larval fish.[463] Smolt migration may be delayed by more than a month where dams slow overall river flow.[464] Hydroelectric dams have been estimated to kill as many as 30 percent of emigrating smolt through direct mortality in turbines and increased predation on stunned fish if there is not adequate spillage.[465] Conversely, when spillage is high, fish may die from supersaturation of water with atmospheric gases.[466] Finally, salmon migrating upstream to spawn frequently have difficulty surmounting dams, even where fishway passages exist.[467]

The Service notes, without substantiation, that hydroelectric projects can affect fishes in several ways, including by: modifying stream flow, temperature, dissolved oxygen, or nutrients; stranding fish, preventing access to habitat, or impeding migration; killing fish when they pass through turbines; and degrading habitat through construction of associated transmission lines and roads.[468]

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[462] DEIS at 3-66, Table 3.4-5.

[463] R. A. Harnish *et al.*, *Effect of Hydroelectric Dam Operations on the Freshwater Productivity of a Columbia River Fall Chinook Salmon Population*, 71 Can. J. of Fisheries & Aquatic Sci. 602 (2013), at PDF 1.

[464] H. L. Raymond, *Effects of Hydroelectric Development and Fisheries Enhancement on Spring and Summer Chinook Salmon and Steelhead in the Columbia River Basin*, 8 N. Am. J. of Fisheries Mgmt. 1 (1988), at PDF 11.

[465] *Id.* at PDF 10-11.

[466] *Id.* at PDF 12.

[467] D. W. Roscoe *et al.*, *Fishway Passage and Post-Passage Mortality of Up-River Migrating Sockeye Salmon in the Seton River, British Columbia*, River Research & Applications (2010), at PDF 0, 8.

[468] DEIS at 3-120.

The DEIS acknowledges that renewable energy projects can already be developed throughout the forest under the current forest plan, and that "Alterntives [*sic*] 2 throught [*sic*] 5 would eliminate 'avoidance areas' which could increase the efficiency and likelyhood [*sic*] of developing these project [*sic*]."[469] The DEIS makes no effort to connect currently proposed renewable projects with possible impacts to salmon, noting only that they are "widely distributed across the Forest," which would "likely reduce cumulative effects of these action [*sic*] on any specific basin area."[470] This limited discussion does not satisfy NEPA's requirements.[471] The FEIS should disclose the potential harms of planned and potential renewable energy projects on salmon in greater detail.

D. The Forest Service Should Consult With the National Marine Fisheries Service on Essential Fish Habitat.

The DEIS notes that streams, lakes, ponds, and wetlands currently or historically accessible to salmon within the Tongass, as well as marine waters adjacent to forest lands, are considered essential fish habitat (EFH) for salmon under the Magnuson-Stevens Fishery Conservation and Management Act.[472] Although the Service completed a comprehensive review of impacts to EFH from non-fishing activities—including timber logging—in Alaska in 2011,[473] the DEIS states that it will not consult with the National Marine Fisheries Service (NMFS) on EFH impacts of the current forest plan revision.[474] Instead, the Service intends to defer consultation until specific actions take place under the plan.[475]

The Magnuson-Stevens Act requires a federal agency to enter into consultation for any action that "may adversely affect any [EFH]."[476] NMFS regulations provide for programmatic consultation for "large-scale planning efforts" that involve "a potentially large number of individual actions that may adversely affect EFH,"[477] and the relevant guidance lists forest

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[469] DEIS at 3-120 to 3-121.

[470] DEIS at 3-121.

[471] *See* 40 C.F.R. § 1502.16; *California v. Block*, 690 F.2d 753, 761 (9th Cir. 1982) (noting that an EIS must contain "a reasonably thorough discussion of the significant aspects of the probable environmental consequences" of an action (internal quotation marks and citation omitted)).

[472] DEIS at 3-111 to 3-112.

[473] PR 769_05_000248 at PDF 13-15 (National Marine Fisheries Service, *Impacts to Essential Fish Habitat from Non-fishing Activities in Alaska* (Nov. 2011)).

[474] DEIS at 3-125.

[475] DEIS at 3-125.

[476] *See* 16 U.S.C. § 1855(b)(2); 50 C.F.R. § 600.920(a)(1); *Souza v. Cal. Dep't of Transp.*, No. C 13-4407 PJH, 2014 WL 793644, at *2 (N.D. Cal. Feb. 26, 2014).

[477] 50 C.F.R. 600.920(j)(1).

planning as one type of action for which programmatic consultation may be appropriate.[478] Given the wide-ranging and potentially serious impacts to salmon habitat that may occur under the revised plan, the Forest Service should consult with NMFS on EFH before deciding it will not transition out of industrial-scale old-growth logging in less than five years and instead continuing this destructive and controversial practice for another decade or more.

E. Salmon Are Already Suffering Adverse Effects From Climate Change.

The Draft Forest Plan's impacts on salmon would be more keenly felt by fishes already struggling with the physiological and environmental changes caused by cumulative climate change effects. Without a comprehensive understanding of potential effects of climate change on salmon's lifecycles, any assessment of a plan's impacts remains inadequate.

The Forest Service's 2010 assessment of climate change in Alaska observed that ambient warming could increase stream temperatures,[479] and the DEIS largely reiterates that expectation.[480] Yet the range of potential impacts of higher water temperatures goes beyond the limited discussion in the DEIS.[481] For example, Chinook salmon swim far upstream to spawn —into the interior of Canada—and higher temperatures will increase metabolic costs during migration.[482] They may also abandon a strategy of remaining upstream for several years after hatching, which could decrease their chances of surviving in the ocean.[483] The DEIS states that higher lake temperatures "may . . . be of benefit for lake-rearing species because growth may increase in these systems, perhaps most noticeable in sockeye salmon."[484] Yet it omits the fact that juveniles' faster growth rates may cause greater food needs to go unmet, and that higher water temperatures can stress adults.[485]

As temperatures increase and more precipitation falls as rain than snow, stream flows will increase in fall and winter and decrease in the summer, especially in snowmelt- and rain-fed basins.[486] These changes could help or harm salmon, according to the DEIS, by increasing overwinter habitat or decreasing summer-rearing habitat, as well as causing egg scour during

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[478] National Marine Fisheries Service, Essential Fish Habitat Consultation Guidance at PDF 28 (Apr. 2004).

[479] Haufler at PDF 16.

[480] DEIS at 3-128.

[481] DEIS at 3-128 to 3-129.

[482] PR 769_05_000200 at PDF 10 (Bryant, M.D, *Global Climate Change and Potential Effects on Pacific Salmonids in Freshwater Ecosystems of Southeast Alaska*, Climate Change (2009) (Bryant).

[483] Bryant at PDF 11.

[484] DEIS at 3-129.

[485] Bryant at PDF 9.

[486] DEIS at 3-128.

peak flows in the fall and winter.[487] Yet low flow may harm salmon in several other ways, as well: sockeye salmon may be unable to access lakes and spawning streams;[488] juvenile coho salmon may find fewer invertebrates to eat;[489] and adult salmon may encounter barriers to migration during the summer.[490]

The DEIS also does not adequately address how warming's effects on snowpack and glaciers may influence the quality of salmon habitat. While the DEIS notes that peak flows will likely increase in fall and winter and decrease in summer, primarily in snowmelt-fed basins, it does not discuss how melting earlier in the spring may interfere with salmon lifecycles.[491] Similarly, although it recognizes that climate change will likely accelerate the melting of glaciers, it does not explain how that process would affect stream flows and temperatures in the short and long terms.[492] Melting glaciers will eventually disappear, decreasing stream flows and potentially exacerbating the effects of logging, road construction, and hydroelectric projects on salmon habitat.[493]

In addition, the DEIS acknowledges that sea level rise may affect fishes by inundating estuarine rearing areas but omits other potential impacts from this phenomenon.[494] For example, seawater may intrude into lakes at lower elevations, decreasing their productivity and harming sockeye salmon.[495] Pink and chum salmon that spawn in the intertidal or lower reaches of rivers and streams may be displaced by shifting sediment deposition zones.[496] These species "will be susceptible to changes in sea level and tidal influence even with relatively small changes."[497] Coho and other salmon species use similar habitat, which may be affected by less than one meter of sea level rise.[498] Thus, the possibility that land mass rise will partially offset dramatic sea level rise in Southeast Alaska[499] provides little assurance that pink and chum salmon will not be harmed.

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[487] DEIS at 3-128.

[488] Bryant at PDF 9.

[489] Bryant at PDF 13.

[490] Bryant at PDF 13.

[491] DEIS at 3-128 to 129.

[492] DEIS at 3-128.

[493] Haufler at PDF 16.

[494] *See* DEIS at 3-129.

[495] Bryant at PDF 10.

[496] Bryant at PDF 9.

[497] Bryant at PDF 9.

[498] Bryant at PDF 13.

[499] DEIS at 3-129.

Finally, climate change will likely increase the frequency of high-intensity precipitation events that could cause landslides, debris avalanches, and mass soil movement.[500] These natural disasters can wreak havoc on salmon eggs and embryos in streambeds through scour and deposition of fine sediment.[501] They may also affect the rearing habitat of sockeye and coho salmon in lakes and streams where sediment may settle.[502]

The FEIS must address these climate impacts, which do not receive mention in the DEIS. Climate change poses a serious threat to salmon and could exacerbate the impacts of the Draft Forest Plan, discussed above, particularly in light of the agency's emphasis on logging in OGRs, beach fringe, and provision of smaller stream buffers. The FEIS must disclose in detail the full range potential climate harms associated with continuing industrial-scale old-growth logging.

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Although salmon populations in Southeast Alaska are currently doing well, ecologists emphasize that their continued success depends on maintaining intact watersheds.[503] The proposed streamside and shoreline buffers would not adequately protect riparian function, especially along lakes and Class III and IV streams. And even a single improperly constructed culvert or poorly operated dam can drastically reduce salmon habitat and survival. The Forest Service must retain or expand existing buffers along all water bodies in the Tongass; end industrial-scale old-growth logging in less than five years; fully disclose the potential impacts of renewable energy projects on salmon; consult with the National Marine Fisheries Service on the effects of the Draft Forest Plan to essential fish habitat; and present a fuller picture of the threats climate change poses for salmon. Salmon are an integral part of Tongass ecosystems, traditional ways of life, and Southeast Alaska's economy. The Forest Service must have a complete understanding of the plan's possible impacts on salmon to make an informed decision—and it should ultimately select an alternative that will secure their future.

V. AMPHIBIANS

Amphibians, though frequently inconspicuous, often account for the greatest amount of vertebrate biomass in a forest[504] and support other vertebrates by converting small invertebrates into larger prey.[505] They also serve as indicators of local ecosystem health, as they are exposed to both aquatic and terrestrial environments and confine themselves to small areas of the

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[500] Bryant at PDF 8-9.

[501] Bryant at PDF 9, 13.

[502] Bryant at PDF 9, 13.

[503] Bryant at PDF 16.

[504] 2008 TLMP AR 603_1944 at PDF 11 (Carey, A. B., *Active Intentional Management (AIM) for Biodiversity and Other Forest Values*, *in* International Workshop on Balancing Ecosystem Values - Innovative Experiments for Sustainable Forestry Proceedings (1998)) (Carey).

[505] *Id.* at PDF 11.

forest.[506] Amphibians are sensitive to changes in the environment[507] and susceptible to a wide array of stressors, putting them at greater risk of extinction than other species in many regions.[508] It is therefore critical that the Forest Service consider impacts to amphibians in light of its proposal to weaken riparian area protections and continuing old-growth logging for an extended period.

Eight amphibian species are known to occur in Southeast Alaska, including two that have been introduced by humans[509]:

- The northwestern salamander (*Ambystoma gracile*) has been observed on Mary Island and Chichagof Island and appears to prefer habitat in old-growth forests.[510] Its distribution and status in Alaska are unknown.[511]

- The long-toed salamander (*Ambystoma macrodactylum*) has been found in the Stikine and Taku watersheds, as well as on several islands.[512] Although it is common in most of its range, in Southeast Alaska its distribution is limited and its status unknown.[513]

- The roughskin newt (*Taricha granulosa*) is the most common tailed amphibian in Southeast Alaska and occurs in much of the Alexander Archipelago.[514] They are probably most dense in mature and old-growth forests.[515]

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[506] *Id.* at PDF 11; *see also id.* at PDF 42; PR 769_05_000460 (MacDonald, S.O. & J.A. Cook, *Mammals and Amphibians of Southeast Alaska*, Special Publication No. 8, Museum of Southwestern Biology (2008) (MacDonald 2008)), at PDF 155 ("Amphibians, in particular, seem to be excellent sentinels of ecosystem health.").

[507] Carey at PDF 42.

[508] A. R. Blaustein *et al.*, *Direct and Indirect Effects of Climate Change on Amphibian Populations*, 2 Diversity 281 (2010), at PDF 2 (Blaustein).

[509] MacDonald 2008 at PDF 131-32.

[510] *Id.* at PDF 133.

[511] *Id.* at PDF 133

[512] *Id.* at PDF 134.

[513] *Id.* at PDF 134

[514] *Id.* at PDF 135-36.

[515] *Id.* at PDF 136.

- The western toad (*Bufo boreas*) is found throughout the Alexander Archipelago and on the coast as far north as Prince William Sound.[516] Southeast Alaska residents have reported sharp declines in the toad's numbers, and in recent years it has been considered near-threatened.[517]

- The Columbia spotted frog (*Rana luteiventris*) occurs primarily on mainland Southeast Alaska from the Salmon River near Hyder to the Taku River, but has also has been documented on Vank and Mitkof Islands.[518] Populations are likely declining in the region.[519]

- The wood frog (*Rana sylvatica*) is widespread in Southeast Alaska, mostly on the mainland.[520] Its range extends into central Alaska as well.[521]

- The Pacific chorus frog (*Pseudacris regilla*) is an exotic species that naturally ranges from southern British Columbia to Baja California but has been observed near Ward Lake on Revillagigedo Island in the Alexander Archipelago.[522] There it may compete with the western toad and roughskin newt, although native species have bred successfully in the same pond as the chorus frog.[523]

- The red-legged frog (*Rana aurora*) is another introduced species, which is native to western North America but has established populations on Chichagof Island and may be rapidly expanding.[524] Populations in California and Oregon, by contrast, have declined or disappeared, and in recent years the species has been considered near-threatened.[525]

The DEIS mentions amphibians only to note that their endemism to Southeast Alaska is in question[526] and that the effects of the two invasive species on native amphibians are either

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[516] *Id.* at PDF 137-38.

[517] *Id.* at PDF 137; *see also* Alaska Dep't of Fish & Game, Western Toad (*Bufo boreas*): Species Profile (ADFG, Western Toad), at PDF 2.

[518] MacDonald 2008 at PDF 141.

[519] Alaska Dep't of Fish & Game, Columbia Spotted Frog (*Rana luteiventris*): Species Profile (ADFG, Columbia Spotted Frog), at PDF 3.

[520] MacDonald 2008 at PDF 143.

[521] *Id.* at PDF 142-43.

[522] *Id.* at PDF 139.

[523] *Id.* at PDF 139.

[524] *Id.* at PDF 140.

[525] *Id.* at PDF 140.

[526] DEIS at 3-231.

unknown or minimal.[527] The Forest Service must include in the FEIS a more thorough analysis of potential impacts from its proposed continued industrial-scale old-growth logging to amphibians.

A. Logging Adversely Affects Amphibians.

1. Logging impairs and fragments habitat.

Studies have documented significantly lower abundances of amphibians in logged areas of the forest than in non-logged areas.[528] The primary causes of these declines are likely increased temperatures and rapid water loss,[529] and community recovery following clear cutting may take up to 50 years.[530] Logging also alters the hydrology, temperature, and substrate characteristics of amphibians' aquatic habitat, which may interfere with reproduction and foraging.[531] One study found that logging reduced fecundity and altered reproductive demographics.[532] Pristine old-growth forest, by contrast, may promote the abundance of amphibians and species diversity.[533]

Logging also harms amphibians through habitat fragmentation, "one of the key issues in the conservation" of these species.[534] Juvenile amphibians avoid clear cut areas to a greater degree than adults,[535] and these individuals play an especially important role in dispersal and population connectivity.[536] In Alaska, clear-cut logging destroys the terrestrial habitat and migration

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[527] *Id.* at 3-235; *see also* Blaustein at PDF 10 (noting that some frogs avoid clear cut areas when precipitation is low).

[528] Carey at PDF 15; *see also* B.D. Todd *et al.*, *Effects of Timber Harvesting on Terrestrial Survival of Pond-Breeding Amphibians*, 313 Forest Ecology & Mgmt. 123 (2014), at PDF 129 (observing that partial cutting harms juvenile salamanders just as much as clear cutting).

[529] R. D. Semlitsch *et al.*, *Effects of Timber Harvest on Amphibian Populations: Understanding Mechanisms from Forest Experiments*, 59(10) BioScience 853 (2009), at PDF 5-6.

[530] *See* at W. Mark Ford *et al.*, *Stand Age and Habitat Influences on Salamanders in Appalachian Cove Hardwood Forests*, 155 Forest Ecology & Mgmt. 131 (2002), at PDF 8 (Ford).

[531] U.S. Forest Service, *Forest Ecosystem Management: An Ecological, Economic, and Social Assessment*, U.S. Government Printing Office, 1993-793-071 (July 1993), at PDF 445, 563.

[532] Jessica A. Homyack & Carola A. Haas, *Long-Term Effects of Experimental Forest Harvesting on Abundance and Reproductive Demography of Terrestrial Salamanders*, 142 Biological Conservation 110 (2009), at PDF 119.

[533] *See* Ford at PDF 8.

[534] Samuel A. Cushman, *Effects of Habitat Loss and Fragmentation on Amphibians: A Review and Prospectus*, 128 Biological Conservation 231 (2006), at PDF 2 (Cushman).

[535] D. A. Patrick, *et al.*, *Effects of Experimental Forestry Treatments on a Maine Amphibian Community*, 234 Forest Ecology & Mgmt. 323 (2006), at PDF 330.

[536] Cushman at PDF 3.

corridors of the roughskin newt[537] and the habitat of the northwestern salamander.[538] Outside the Tongass, wood frogs have been observed to avoid clear-cut areas when migrating.[539]

The DEIS does not discuss the impacts of logging on amphibians specifically. It simply states that logging on wetland sites could lead to "reductions in available wildlife habitat."[540] The Forest Service must move more rapidly away from old-growth logging, but to the extent it continues framing the alternatives in terms of allowing old-growth logging for a decade or more to come, then the FEIS must assess the potentially destructive consequences of continued industrial-scale old-growth logging for amphibians. Indeed, some have recommend that logging that removes more than 40 percent of the canopy should be prohibited in areas in which amphibian diversity is known to be high.[541]

2. Logging results in direct mortality of amphibians.

Where amphibians are unable to disperse out of logged areas, they are especially vulnerable logging activities. The effect is more pronounced for this group than for other animals because amphibians are unable to move out of the way of logging equipment.[542]

The DEIS does not mention this potential impact of logging on amphibians, which stands as yet another reason to limit old-growth logging. The FEIS must correct this inadequacy.

3. Amphibians are especially susceptible to air pollution.

Because amphibians' skin is permeable, they are especially susceptible to the harmful effects of air pollution.[543] Atmospheric contaminants such as acidic compounds may also be deposited in aquatic ecosystems where amphibians live.[544] In Southeast Alaska, air pollution from local or global sources may be contributing to a decline in amphibians' numbers.[545]

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[537] MacDonald 2008 at PDF 136.

[538] S. O. MacDonald, THE AMPHIBIANS AND REPTILES OF ALASKA: A FIELD HANDBOOK (2010), at PDF 17 (MacDonald 2010).

[539] Tracy A. G. Rittenhouse & Raymond D. Semlitsch, *Behavioral Response of Migrating Wood Frogs to Experimental Timber Harvest Surrounding Wetlands*, 87 Can. J. Zoology 618 (2009), at PDF 6.

[540] DEIS at 3-86.

[541] R. D. Semlitsch *et al.*, *Effects of Timber Harvest on Amphibian Populations: Understanding Mechanisms from Forest Experiments*, BioScience 59(10): 853 (2009), at PDF 8.

[542] *See* Z. Burivalova *et al.*, *Thresholds of Logging Intensity to Maintain Tropical Forest Biodiversity*, 24 Current Biology 1 (2014), at PDF 3.

[543] 2008 TLMP AR 603_0690 (B.P. Kelly *et al.*, *Climate Change, Predicted Impacts on Juneau, Scientific Panel on Climate Change City and Borough of Juneau* (Apr. 2007) (Kelly), at PDF 60.

[544] Blaustein at PDF 19.

[545] Kelly at PDF 60.

The DEIS predicts that "direct effects on air quality from forest management activities would be temporary and limited in nature" and may include emissions from industrial processing sites and firewood burning.[546] It also notes that, cumulatively, air pollution from wood processing could "increase somewhat if more wood is burned to produce energy."[547] The Forest Service should consider the potential consequences of worsened air quality for amphibians in the FEIS, in light of their sensitivity to this kind of pollution.

B. Roads Adversely Affect Amphibians.

1. Roads impair and fragment habitat.

Roads add sediment to streams, potentially degrading habitat for amphibians[548] such as the Columbia spotted frog that depends on permanent water and various riparian habitats for breeding, foraging, and overwintering.[549] Roads may also contribute chemical contaminants to habitat, which may cause skeletal abnormalities in wood frogs—although introduction of predators, parasites, or pathogens by roads could also explain this phenomenon.[550] In areas immediately adjacent to roads, amphibians suffer from "edge effects," including drier, warmer soils, and increased light and wind.[551] The construction of roads can also destroy habitat outright: in Southeast Alaska, loss of wetland habitat threatens the Columbia spotted frog and the long-toed salamander.[552]

Apart from degrading habitat, roads can fragment habitat, and thus populations, by forming travel barriers for amphibians. Studies have shown that salamanders avoid forest roads, and that even a single road can reduce their movement by more than 50 percent.[553] Multiple roads could

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[546] DEIS at 3-18 to 3-19.

[547] *Id.*at 3-20 to 3-21.

[548] J. MacNeil *et al.*, Forest Management for Reptiles and Amphibians (2013), at PDF 8 (citing C. E. Moorman, *et al.*, *Reptile and Amphibian Response to Hardwood Forest Management and Early Successional Habitats*, *in* Sustaining Young Forest Communities: Ecology and Management of Early Successional Habitats in the Central Hardwood Region, USA (C. H. Greenberg, *et al*., eds., 2011)).

[549] MacDonald 2010 at PDF 25; ADFG, Columbia Spotted Frog at PDF at 3.

[550] *See* M. K. Reeves *et al.*, *Road Proximity Increases Risk of Skeletal Abnormalities in Wood Frogs from National Wildlife Refuges in Alaska*, Envtl. Health Perspectives 116(8): 1009 (2008), at PDF 5.

[551] *See, e.g.,* D. M. Marsh & N. G. Beckman, *Effects of Forest Roads on the Abundance and Activity of Terrestrial Salamanders*, Ecological Applications 14(6): 1882 (2004), at PDF 1-2.

[552] ADFG, Columbia Spotted Frog at PDF 3; Alaska Dep't of Fish & Game, Long-Toed Salamander (ADFG, Long-Toed Salamander), at PDF 2; MacDonald 2008 at PDF 134.

[553] D. M. Marsh *et al.*, *Forest Roads as Partial Barriers to Terrestrial Salamander Movement*, Conservation Biology 19(6): 2004 (2004), at PDF 3-4.

impede movement even more dramatically, potentially decreasing gene flow enough to cause a population to go extinct.[554] In Alaska, road construction can fragment terrestrial habitat and ponds where long-toed salamanders breed.[555]

The DEIS acknowledges that crossing wetlands with roads could lead to sedimentation or changes in hydrologic patterns and connectivity, but it does not connect these consequences to amphibians or discuss the fragmentation of their habitat.[556] The Forest Service should consider these impacts before selecting an alternative that continues industrial-scale old-growth logging for years to come.

2. Road result in direct mortality of amphibians.

The Forest Service has previously recognized that roads pose a greater threat to amphibians than other animals because amphibians are small and slow-moving and often migrate across roads.[557] Even narrow forest roads present a hazard,[558] and entire populations may be placed at risk.[559] Mortality rates vary by species, but slight variations in traffic can have significant impacts.[560]

As with other effects on amphibians, the DEIS omits any mention of the threat of road traffic to local populations. The Service should disclose and weigh these impacts in choosing a forest plan alternative.

C. Amphibians are Already Suffering From the Adverse Effects of Climate Change.

Because climate change could exacerbate many of the harms discussed above, it is essential that the Service understand and disclose the possible impacts of a warming environment on amphibians in conjunction with any continuing old-growth logging.

The effects of climate change on amphibians in Southeast Alaska are affecting amphibians in a variety of ways.[561] For example, prolonged dry weather in Southeast Alaska in the spring of 2003 dewatered some ponds and delayed or prevented western toads from spawning in them.[562]

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[554] *Id.* at PDF 4.

[555] ADFG, Long-Toed Salamander, at PDF 2.

[556] *See* DEIS at 3-87 to 3-88.

[557] U.S. Forest Service, *Forest Roads: A Synthesis of Scientific Information*, Pacific Northwest Research Station, General Technical Report PNW-GTR-509 (May 2001) at PDF 8, 48.

[558] *Id.* at PDF 48.

[559] *Id.* at PDF 47, 113.

[560] *See* M. J. Mazerolle, *Amphibian Road Mortality in Response to Nightly Variations in Traffic Intensity*, Herpetologica 60(1): 45 (2004), at PDF 7.

[561] Kelly at PDF 60.

[562] ADFG, Western Toad at PDF 3.

If the Tongass follows patterns observed in other rainforests, mist could rise, making low-lying habitat less suitable for amphibians.[563] Meanwhile, decreased snow depths in warming winters may expose toads to freezing conditions.[564]

Sea level rise could also harm the Tongass's amphibians. The Forest Service acknowledges that this phenomenon might occur in Southeast Alaska and inundate estuaries where young fish rear.[565] It could have similar effects on amphibian habitat by causing storm-related saltwater intrusion of coastal wetlands,[566] and it may introduce predators such as fishes into these habitats.[567]

It is possible that climate change will shift cooler waters toward the thermal optimum for chytridiomycosis — "an unusual fungal infection caused by a parasitic water mold that attacks keratin in amphibian skin"[568] — and cause outbreaks of this fungal disease in amphibian populations.[569] An extensive survey of amphibian habitat use in Southeast Alaska suggests that declines in toad populations are attributable to the disease and may be exacerbated by climate change.[570] Apart from creating conditions conducive to the fungus that causes it, warming can also make amphibians more vulnerable to it by decreasing their immune function.[571]

Amphibians are susceptible to changes in temperature because it controls many physiological processes, including "oxygen uptake, heart rate, locomotion, water balance, digestion, developmental rate, sex determination, and immune function.[572] Warming, as well as the resultant lower levels of dissolved oxygen in water, can harm developing embryos and larvae.[573] Higher temperatures in winter may decrease body size, number of eggs laid, and survival of

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[563] Kelly at PDF 61.

[564] ADFG, Western Toad at PDF 3.

[565] DEIS at 3-129.

[566] Blaustein at PDF 12.

[567] *Id.*

[568] R. Carstensen *et al.*, *Habitat Use of Amphibians in Northern Southeast Alaska* (2003) (Carstensen), at PDF 5.

[569] *Id.* at PDF 17.

[570] Carstensen at PDF 5; *see also id.* at 68 (fungus is a "potential cause of widespread declines [of toads] in Southeast Alaska"); MacDonald 2010 at PDF 23 (noting that five out of nine toads in the Dyea area tested positive for the fungus in 2005).

[571] Blaustein at PDF 6, 7, 15, 18.

[572] *Id.* at PDF 6.

[573] *Id.* at PDF 6-9.

female toads.[574] And sex ratios can become skewed as populations are masculinized or feminized, depending on the species.[575]

At the level of the individual, changes in ambient temperature alter amphibian behavior, including the timing of their breeding, hibernation, and foraging.[576] When temperatures rise, amphibians seek refugia and spend less time on activities such as foraging.[577] Warming may also lessen dispersal distances, which can reduce gene flow between populations and expose isolated populations to increased extinction risks.[578]

The above discussion offers only a glimpse of the numerous, acute threats climate change poses to amphibians. These harms and others could compound the impacts of old-growth logging and other development under the Draft Forest Plan on amphibians. The DEIS outlines possible climate effects for fishes, but not for amphibians.[579] The Service must consider and disclose these effects before selecting an alternative that continues industrial-scale old-growth logging.

* * *

Amphibians are an integral component of the forest ecosystem in Southeast Alaska and provide an important indicator of forest integrity. Several indigenous species are currently declining, and all face serious threats from a warming climate. If the agency is intent on adopting a forest plan amendment, based on continuing industrial-scale old-growth for another decade or more, the FEIS must carefully weigh—and disclose to the public—the potential impacts that logging, road construction, and renewable energy projects could have on these sensitive forest inhabitants. It also must examine and explain whether such a plan would ensure the continued viability of amphibians on the Tongass. To do otherwise, would run afoul of NFMA's requirements

VI. PRINCE OF WALES SPRUCE GROUSE

The Prince of Wales spruce grouse (*Falcipennis canadensis isleibi*) is a subspecies of spruce grouse endemic to the Alexander Archipelago that serves as an important prey species for the Queen Charlotte goshawk. The Prince of Wales spruce grouse possesses a variety of unique biological characteristics that make it especially vulnerable to old-growth logging, and its entire range is within parts of the Tongass that have been disproportionately altered by past logging. The DEIS fails to grapple with the potentially serious effects on the grouse of continued old-growth logging and weakening of the conservation strategy under the Draft Forest Plan.

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[574] *Id.* at PDF 7.

[575] *Id.* at PDF 9.

[576] *Id.* at PDF 5.

[577] *Id.* at PDF 10.

[578] *Id.*

[579] *See* DEIS at 3-128 to 129.

A. Prince of Wales Spruce Grouses' Unique Biological Characteristics Make Them Vulnerable to Extinction.

The Prince of Wales spruce grouse (*F. c. isleibi*) is endemic to the Alexander Archipelago in Southeast Alaska.[580] It has a known distribution only on Prince of Wales Island, Heceta, Suemez, Warren, Kosciusko, Zarembo, and Mitkof Island.[581] Conversely, "[t]he northern and eastern islands in the [Alexander] archipelago, the Queen Charlotte Islands, and the adjacent mainland all appear to be void of spruce grouse."[582] Prince of Wales spruce grouse are "geographically disjunct,"[583] or "isolated from spruce grouse elsewhere in North America, [and] appear morphologically and behaviorally distinct."[584] Prince of Wales spruce grouse is not only genetically divergent from other spruce grouse, but even the populations on Prince of Wales and Zarembo islands have "high levels of allele frequency divergence"[585] and "may be reproductively isolated from each other or could be in the near future."[586] According to Barry and Tallmon (2010), the genetic divergence between islands is likely attributable to geographic isolation: "In order for individuals to disperse from [Prince of Wales] to [Zarembo Island], they would have to travel ~3.2km through a series of small islands. Although it is not inconceivable that gene flow occurs, at least in a stepping-stone fashion, the ability of these small islands to sustain populations of Spruce Grouse is unknown, and our results suggest that any influence of dispersal is extremely low."[587]

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[580] S.J. Williamson *et al.*, *Spruce Grouse Continental Conservation Plan*, Association of Fish & Wildlife Agencies (2008) at 7 (Williamson); 769_05_000540 (U.S. Fish & Wildlife Service, Species Assessment and Listing Priority Assignment Form [*F. c. isleibi*] (2010)) (USFWS 2010), at 5; DEIS at 3-231 (Table 3.10-7), 3-233.

[581] DEIS at 3-231 (Table 3.10-7), 3-233. *See also* B.D. Barry & Tallmon, D.A., *Genetic Differentiation of a Subspecies of Spruce Grouse (Falcipennis canadensis) in an Endemism Hotspot*, Auk 127(3):617-625 (Barry & Tallmon), at 622 (listing eight islands, including San Fernando) (citing Dickerman & Gustafson 1996). According to U.S. Fish and Wildlife Service, there are "records" of spruce grouse on 11 islands, including Dall, Grindall, and Tuxecan. USFWS 2010 at 10. For a map of these islands, *see* USFWS 2010 at 11, Fig. 3.

[582] Williamson at 27.

[583] A.L. Russell, *Habitat relationships of spruce grouse in Southeast Alaska*, Texas Tech University, M.S. thesis (1999) at vii (Russell).

[584] Williamson at 26. *See also* R.W. Dickerman & J. Gustafson 1996, *The Prince of Wales Spruce Grouse: A New Subspecies From Southeastern Alaska*, Western Birds 27:41-47 (1996) at 46 (*F. c. isleibi* differs from other subspecies of *F. canadensis* (plumage) and from *F. c. franklinii* (size & plumage); USFWS 2010 at 2-3 & Fig. 1.

[585] Barry & Tallmon at 622.

[586] USFWS 2010 at 5.

[587] Barry & Tallmon at 622-23.

Prince of Wales spruce grouse likely occur at low densities,[588] and are considered rare.[589] Their density has been estimated at 2.5 birds "per 100 ha of potentially occupied habitat"[590] or 10,500 total spruce grouse."[591] Williamson et al. (2008), however, concluded that the "accuracy of this estimate is unknown but, at first appearance, seems high,"[592] and therefore assumed the Prince of Wales spruce grouse population to be less than 8,300 birds.[593]

Spruce grouse have very limited dispersal capabilities. They are sedentary birds with a preference for walking rather than flying.[594] They are poor long-distance flyers.[595] They exhibit limited seasonal migratory movement, typically less than a mile.[596] "Where diverse stand compositions (e.g., horizontal diversity) are present, seasonal movements may not be necessary (Bouta 1991)."[597] They are reluctant to disperse or move freely across open areas or thorough areas of unsuitable habitat.[598]

Prince of Wales spruce grouse are old-growth dependent. They are associated with muskegs, complex, high-volume old-growth, and mixed conifers (scrub).[599] They "appear most associated with Sitka spruce, western hemlock, and shore pine in an area with annual precipitation of up to 5 m."[600] Russell (1999) found Prince of Wales spruce grouse to prefer forest and muskegs, to use scrub and second-growth as available, and to avoid clear-cuts.[601] Spruce grouse frequently use old-growth for foraging, and mid-canopy also offers good concealment from avian predators.[602]

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[588] Williamson at 11.

[589] Russell at 13; USFWS 2010 at 12; A. R. Nelson, *Ecology of Prince of Wales Spruce Grouse*, Master of Science Thesis, University of Alaska Fairbanks (2010) (Nelson), at 7 (spruce grouce are infrequently observed on Prince of Wales Island).

[590] Williamson at 17, 20 Table 5 (citing Russell 1999).

[591] *Id.* at 27.

[592] *Id.* *See also* Barry & Tallmon at 623 (population sizes are considered low) (citing Storch 2007); USFWS 2010 at 13 (accuracy of estimates is unknown).

[593] USFWS 2010 at 13.

[594] DEIS at 3-233.

[595] *Id.*

[596] *Id.* (citing Dickerman & Gustafson 1996; Boag & Schroeder 1992; Williamson et al. 2008).

[597] Russell at 40.

[598] USFWS 2010 at 9.

[599] Williamson at 9 (citing Russell 1999); Russell at vii, 34; USFWS 2010 at 7.

[600] Williamson at 26. *See also* USFWS 2010 at 7.

[601] Russell at 26-42. *See also* USFWS 2010 at 7.

[602] Russell at 39.

B. Further Logging in Spruce Grouse Habitat Threatens the Species' Viability.

Further old-growth logging on the Tongass threatens the viability of Prince of Wales spruce grouse because clear-cuts and roads create barriers to dispersal and further isolate this already small and isolated population. The problem is compounded by the fact that the Prince of Wales spruce grouse's habitat has already been disproportionately logged.

Clear-cut forests are mostly unsuitable for spruce grouse for more than 100 years.[603] Clear-cuts create travel and dispersal barriers for Prince of Wales spruce grouse; the birds move "around rather than through" clear-cuts.[604] Clear-cuts inhibit spruce grouse movement (logging debris or slash), increase exposure to predators, and do not provide food.[605] "Timber harvest and associated fragmentation may lead to population declines if open areas are too large or forested patches are spread too far apart to enable spruce grouse to move between them (greater than 1 mile). Clear-cuts may also present a dispersal barrier to this species due to the thick logging debris often present which could inhibit walking, this species preferred method of movement (Russell 1999)."[606] Moreover, "fragmentation due to timber harvest can result in the isolation of local spruce grouse populations (i.e., if open areas are too large or forested patches are spread too far apart to enable spruce grouse to move between them)."[607] Forest connectivity may be especially important to juvenile spruce grouse. Juveniles "living in, and dispersing from, the low connectivity sites had significantly lower survival rates due to increased predation risk from raptors. Modelling these empirical data revealed that . . . further loss of high connectivity in the study area would lead to population decline."[608] In fragmented forest, both juveniles and adults make larger movements and have poorer survival (Whitcomb et. al. 1996a, Harrison 2001). "The poorer survival appears to be due to fragmentation rather than longer movements of individuals."[609] As a result, "[c]onnectivity of scrub and high-volume old-growth forest within the matrix of managed forest stands may be an important factor in minimizing negative effects to spruce grouse populations in southeast Alaska."[610]

Logging-associated roads also create dispersal barriers and lead to a high proportion of grouse mortality. For radio-marked Prince of Wales spruce grouse, 42-44% of mortalities were road-

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[603] *Id.* at 41

[604] *Id.* at 34.

[605] *Id.* at 35; DEIS at 3-233, 3-259 to 3-260.

[606] DEIS at 3-233.

[607] *Id.* at 3-260

[608] I. Storch, *Grouse: Status Survey and Conservation Action Plan 2006-2010*, IUCN, Gland, Switzerland, and Cambridge, United Kingdom, and World Pheasant Association, Fordingbridge, United Kingdom (2007) at 26.

[609] Williamson at 8.

[610] Russell at 34.

related.[611] Of these mortalities, about half were from road strikes and half were from hunting.[612] Spruce grouse are "particularly vulnerable to hunting along road systems, and thus are susceptible to overexploitation near roads and human populations (Williamson et al. 2008; Rabe 2009)."[613]

Threats to spruce grouse from logging on the Tongass are compounded by the fact that the Prince of Wales spruce grouse's range has already been disproportionately logged and roaded:

> Within Southeast Alaska, timber harvest has occurred disproportionately in the range of the [Prince of Wales] spruce grouse. On [Prince of Wales Island], in particular, there has been extensive and intensive, broad-scale clear-cutting, fragmenting the island into a patchwork of uniform-aged, forest stands less than 50 years old and a widespread road system. Between 1954 and 2005, approximately 20% of the total land area and 25% of the original productive old-growth on [Prince of Wales] and surrounding islands was logged (Brinkman et al. 2009). The majority (32%) of old-growth was taken from northern [Prince of Wales] Island compared to a smaller portion (10%) taken from southern [Prince of Wales] Island (Albert and Schoen 2007). Although this geographic distinction is not biologically meaningful to the [Prince of Wales] spruce grouse, logging did not occur uniformly across the island and therefore, any impacts from logging would be more prevalent in the northern part of the island. Similar logging practices have occurred on surrounding islands where [Prince of Wales] spruce grouse are known or expected to occur (productive old-growth logged: 20% on the Dall Island complex, 13% on the Outside Islands (Kosciusko, Warren, Heceta, Tuxekan, San Fernando, and Suemez Islands), and 16% on Etolin and Zarembo Islands; Fig. 3; Albert and Schoen 2007).[614]

Moreover, the amount of roads on Prince of Wales and surrounding islands, within Prince of Wales spruce grouse range, is estimated to be around 2,500 miles, most of which are open to public.[615]

Because Prince of Wales spruce grouse are isolated and have limited dispersal ability, clear-cuts significantly increase their risk of extinction. Further, the "lower genetic variation observed in both [Prince of Wales] and [Zarembo Island] compared with mainland populations reflects small

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[611] USFWS 2010 at 16; Nelson at 6.

[612] USFWS 2010 at 17.

[613] DEIS at 3-233.

[614] *Id.* at 15.

[615] *Id.* at 16.

effective population sizes, which could diminish their capacity to respond to a changing environment. The potentially low level of dispersal into these insular populations indicates that there may be little opportunity for the demographic rescue that many mainland grouse populations rely on to persist (Storch 2007)."[616] All these factors have led scientists to conclude that the viability of Prince of Wales spruce grouse is a high conservation concern.[617]

The Forest Service must explain and the FEIS must disclose whether and how the Draft Forest Plan will ensure the continued viability of Prince of Wales spruce grouse. The agency would violate the law if it did otherwise.

C. The DEIS Does Not Disclose the Draft Forest Plan's Risks to the Spruce Grouse.

The DEIS dedicates about one page to Prince of Wales spruce grouse and does not attempt to grapple with how the various alternatives could affect this key Tongass species.[618] Although the DEIS briefly mentions that continued old-growth logging and increased road densities could adversely affect spruce grouse, it dismisses these concerns by asserting that, "[l]ocalized increases in road densities would be managed through road closures and storage or decommissioning which would minimize the potential for increased harvest risk for spruce grouse over the long term," and "[t]he Forest Plan Conservation Strategy would continue to suitable [*sic.*] habitat and provide landscape connectivity for these species."[619] This is inadequate. The Forest Service must disclose and grapple with the consequences to Prince of Wales spruce grouse of continued old-growth logging over the next 16 years. Moreover, it may not rely on the conservation strategy to avoid such analysis, as the Draft Forest Plan significantly weakens the conservation strategy. *See supra* pp. 40-53.

VII. MARTEN

The marten is an important carnivore that is associated with old-growth forests. Currently, the best available science on the taxonomic status of marten (*Martes spp*.) in southeast Alaska indicates the presence of two distinct species: the Pacific marten (*Martes caurina*) and the American marten (*Martes Americana*).[620] These two species have discrepancies in their range overlap, with the American marten occurring primarily on coastal mainland sites and nearshore islands in the Alexander Archipelago.[621] The Pacific marten is a coastal species with a restricted

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[616] Barry & Tallmon at 623.

[617] *See* Suring et al. 1993.

[618] DEIS at 3-233, 3-259-60.

[619] *Id*. at 260.

[620] N.G. Dawson & J.A. Cook, J. A. Cook, *Behind the Genes: Diversification of North American Martens (Martes americana and M. caurina)*, *in* Biology and Conservation of Martens, Sables, and Fishers: A New Synthesis (K.B. Aubry *et al.* eds., 2012).

[621] Alaska Dep't of Fish & Game, American Marten (*Martes americana*): Species Profile, at PDF 2.

range on Kuiu Island and Admiralty Island.[622] Kuiu Island is particularly significant because it maintains a hybridization region for these two species of marten. Research on hybridization events indicates these can be areas of incredible biodiversity value.[623] The continental form occurs elsewhere in the species' range.[624]

As an initial matter, the Forest Service needs to address the status of the Pacific marten and explain how the Draft Forest Plan affects its viability in the Tongass. It has an extremely restricted known range of only two islands in southeast Alaska, and on Kuiu Island its population is thought to be in decline.[625]

The Forest Service also needs to address martens' varying habitat needs. "American martens are strongly associated with late successional forests of a variety of vegetation types, as long as the forests contain sufficient vertical diversity and structure."[626]

"American martens have been reported to prefer riparian areas (Anthony et al. 2003, Buskirk et al. 1989, Martin 1987) and sites close to water (Bull et al. 2005, Hargis and McCullough 1984, Simon 1980, Spencer et al. 1983)."[627] Indeed, the DEIS acknowledges that "[c]oastal habitats (beach fringe) and riparian areas have the highest habitat value for marten. . . ."[628] It also explains that "OGRs and other non-development LUDs provide refugia for marten from trapping pressure."[629] "Legacy Forest Structure standards and guidelines, in combination with the beach fringe and riparian buffers, aid in providing habitat and connectivity for marten on National Forest System (NFS) lands (USDA Forest Service 2008a)."[630] Yet, the DEIS fails to examine in any meaningful way the impacts associated with the Draft Forest Plan's proposal to log in OGRs, beach fringe, and riparian buffers. The agency also fails to examine the ability to ensure the viability of the marten given these changes.

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[622] Alaska Dep't of Fish & Game, Pacific Marten (*Martes caurina*): Species Profile, at PDF 2.

[623] B.M, Fitzpatrick, *et al.*, *Hybridization and the species problem in conservation*, Current Zoology 61: 206-216, 207 (2015).

[624] Small at 90 fig. 1.

[625] *See* Federal Subsistence Board, News Release, Federal Subsistence Board Closes Marten Trapping on Federal Lands on Kuiu Island (Dec. 16, 2013).

[626] M. Goldstein *et al*., Chp. 10 - Developing a Habitat Monitoring Program: Three Examples from National Forest Planning, IN A TECHNICAL GUIDE FOR MONITORING WILDLIFE HABITAT at PDF 23 (2013) (Goldstein).

[627] *Id*.

[628] DEIS at 3-220.

[629] DEIS at 3-222.

[630] DEIS at 3-222.

Goldstein identified "six landscape attributes and two site attributes that would be important to monitor to ensure that habitat of American martens is maintained. . . ."[631] Goldstein then translated the habitat attributes into measureable metrics.[632] The Forest Service should consider this analysis in deciding how to amend the forest plan.

VIII. CROSSBILLS

Crossbills are small birds belonging to the finch family whose bills are specially adapted to extract seeds from conifer cones.[633] Both white-winged crossbills (*Loxia leucoptera*) and red crossbills (*Loxia curvirostra*) live in Southeast Alaska; however, statewide, red crossbills' numbers have been estimated to be less than half of those of white-winged crossbills.[634] The red crossbill can also be found throughout coastal Southeast Alaska, whereas the white-winged species is largely restricted to the mainland.[635] For these reasons, the red crossbill is of greater concern on the Tongass.

The red crossbill complex in North America is divided into several groups or "call types" with different vocalizations,[636] morphology, ecology, and foraging habits.[637] Though various call types may nest in the same area, for the most part they remain reproductively isolated.[638] This lack of interbreeding may result from cultural evolution of distinctive vocalizations and positive

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[631] Goldstein at PDF 26-27.

[632] *Id*. at 27.

[633] Audubon, White-Winged Crossbill: *Loxia leucoptera*, Audubon Field Guide at PDF 1; Audubon, Red Crossbill: *Loxia curvirostra* (Audubon, Red Crossbill) Audubon Field Guide, at PDF 1; U.S. Forest Service, Birds: Red Crossbill (USFS, Red Crossbill).

[634] *Compare* K. Walton *et al.*, Alaska Species Ranking System Summary Report – Red Crossbill (2013) (Walton, Red Crossbill), at PDF 1 (statewide population of red crossbill is estimated at 809,300), *with* K. Walton *et al.*, Alaska Species Ranking System Summary Report – White-winged Crossbill (2013) (Walton White-winged Crossbill) (statewide population of white-winged crossbill is estimated at 2,340,000).

[635] *Compare* Walton, Red Crossbill at PDF 1, *with* Walton White-winged Crossbill.

[636] K. Irwin, *A New and Cryptic Call Type of the Red Crossbill*, 41 Western Birds 10 (2010), at PDF 10 (Irvin). Some consider them to be separate biological species. *Id.*; C.W. Benkman, *Logging, Conifers, and the Conservation of Crossbills*, Conservation Biology 7(3):473 (1993) (Benkman), at PDF 1; *see also* U.S. Forest Service, Birds: Red Crossbill (USFS, Red Crossbill), at PDF 1.

[637] Benkman at PDF 1.

[638] *See, e.g.*, Commission on the Status of Endangered Wildlife in Canada, COSEWIC Assessment and Status Report on the Red Crossbill (*Percna*) in Canada 2004 (2012), at PDF 2; Benkman at PDF 1.

assortative mating between individuals of the same vocal type.[639] Two call types can be found in the Tongass: the western hemlock crossbill (*L. c. minor*) and the Sitka spruce crossbill (*L. c. sitkensis*).[640]

Red crossbills are a "Species of Greatest Conservation Need" and have a category II (red) conservation ranking in Alaska.[641] While the statewide population is estimated to comprise 809,300 individuals,[642] the "actual status of nomadic crossbill populations is difficult to assess and large declines may go undetected."[643] Moreover, because red crossbills' sole food source (conifer seeds) intermittently fails, they are considered "extinction prone"; populations elsewhere have been pushed to the brink of extinction by logging.[644]

A. Red Crossbills Depend on Old-Growth in the Tongass for Their Continued Survival.

Red crossbills are specially adapted to remove seeds from the cones of the trees in the area where they live. The western hemlock crossbill, for example, has a palatal groove perfectly sized to extract seeds from the cone of the hemlock tree.[645] Given annual variations in the seed production capacity of the forests they depend on, red crossbills are limited to only a small amount of their range at any given time.[646] They typically move from forest to forest as "nomadic" populations, seeking out abundant conifer seeds.[647]

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[639] *See* Galis, F. & J. J. M. van Alphen, *How Fast Do Crossbills Speciate? On Assortative Mating and Vocalizations*, Trends In Ecology & Evolution 15(9): 356 (2000), at PDF 2; *see also* L. K. Snowberg & C. W. Benkman, *Mate Choice Based on a Key Ecological Performance Trait*, 22 J. Evolutionary Biology 762 (2009), at PDF 1 (noting a study in which 99 percent of individual crossbills paired assortatively by call type).

[640] M. Young, North American Red Crossbill Types: Status and Flight Call Identification, eBird (2012), at PDF 6-7, 11-12; *see also* Irwin at PDF 11; USFS, Red Crossbill at PDF 1.

[641] Walton, Red Crossbill at PDF 1.

[642] *Id.*

[643] Benkman at PDF 2.

[644] W. C. Holimon *et al.*, *The Importance of Mature Conifers to Red Crossbills in Southeast Alaska*, 102 Forest Ecology & Mgmt. 167 (1998) (Holimon), at PDF 1 (internal quotation marks omitted).

[645] USFS, Red Crossbill at PDF 1.

[646] Benkman at PDF 3.

[647] Audubon, Red Crossbill at PDF 2; *see also* W. C. Holimon, *et al.*, *The Importance of Mature Conifers to Red Crossbills in Southeast Alaska*, 102 Forest Ecology & Mgmt. 167 (1998), at PDF 1 ("Crossbills wander nomadically to exploit spatiotemporally varying confer seed crops.") (citations omitted).

"Dependency on a single resource makes crossbills extremely vulnerable to habitat loss and alteration."[648] Additionally, "protecting nomadic species such as crossbills represents a . . . formidable challenge because critical habitats are more difficult to recognize since they may be used only once every several years, with different areas crucial during different years."[649]

The two call types of red crossbills in Southeast Alaska feed primarily on the seeds of western hemlock and Sitka spruce.[650] They selectively forage in mature stands where cone production is higher than in young stands, in part to save energy on travel and in part to maximize both size of crop and seeds per cone.[651] Indeed:

> During a year of poor cone production, red crossbills did not forage in young stands. Within mature stands, crossbills tended to forage preferentially on trees with large cone crops. These favored trees were also the larger and older trees because cone production increases with tree size. The avoidance of young stands is especially pronounced during poor seed years and may be related to the absence of mature trees containing numerous cones with many seeds. . . . [M]ature stands [may be] critical for maintaining crossbill populations.[652]

It is therefore unsurprising that older forests "tend to support more crossbills than do younger forests, and as the proportion of the landscape containing older forests declines, crossbills decrease disproportionately in abundance."[653]

Red crossbills in the Tongass, being nomadic populations, also require large swaths of mature forest. In any given year they may find cones only within a small area, as Sitka spruce and western hemlocks can go eight years between good cone productions.[654]

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[648] Benkman at PDF 2.

[649] *Id.* at PDF 4.

[650] USFS, Red Crossbill; Holimon at PDF 1.

[651] Holimon at PDF 4.

[652] *Id.* at PDF 1.

[653] Benkman at PDF 4. One Forest Service survey associated Tongass red crossbills with closed-canopy forest dominated by large-diameter hemlock or spruce associations and a 20-year seral stage. *See* 2008 TLMP AR 603_0466 (W. P. Smith *et al.*, Bird, Mammal, and Vegetation Community Surveys of Research Natural Areas in the Tongass National Forest (2001)) at PDF 27, Table 11.

[654] USFS, Red Crossbill at PDF 1.

B. Logging in the Tongass Could Have Serious Impacts on Red Crossbills.

Logging that replaces mature or old-growth forests with younger forests harms red crossbill populations by reducing cone and seed production. Conifers generally do not bear cones until they are several decades old, and even then younger conifers are less productive than older conifers.[655] Sitka spruce may not produce cones before 40 years of age, and it produces sizeable cone crops only when it is older.[656] Some trees, such as western hemlock, are underrepresented in second-growth, so an old-growth stand may produce more than 200 times the number of seeds that the trees in a second-growth stand would.[657] Mature forests also produce cones more consistently over longer periods than regularly logged forests.[658]

This forest ecology means that logging in old-growth disproportionately affects cone production and has more severe effects on red crossbills than second-growth logging.[659] Logging may also target forests at lower elevations that produce more seeds than forests higher up and on steeper slopes.[660] When forests do not produce enough cones for crossbills to meet minimum energy requirements, "cone failures" occur.[661] And, even where cone failures do not decimate a population, smaller cone crops reduce breeding in crossbills and slow a population's recovery.[662]

As noted, crossbills follow a nomadic life strategy because they must find areas of abundant cones to survive, and because cone production varies annually. Years of prolific cone production in an area are often followed by years of little to no cone production, and "even the most regular cone-producing conifers have occasional cone failures."[663] Entire regions may fail to produce seeds for several years in a row.[664] Crossbills respond by moving out of these areas and concentrating where food is more plentiful.[665] As a result, "only a fraction of the total potential range can support crossbills," and "a reserve or system of reserves encompassing only a restricted geographic area would be inadequate to support nomadic populations of crossbills continuously."[666]

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[655] Benkman at PDF 2.

[656] USFS, Red Crossbill at PDF 1.

[657] Benkman at PDF 3.

[658] *Id.* at PDF 2-3.

[659] *See id.*; *see also id.* at PDF 4 (noting that one study in Finland found that, as old-growth forests diminished by 27 percent from clear cutting, red crossbills declined by 75 percent).

[660] *See id.* at PDF 3-4.

[661] *Id.* at PDF 3.

[662] *Id.*

[663] *Id.*

[664] *Id.*

[665] *Id.*

[666] *Id.*

Crossbills' nomadic nature also exposes them to habitat fragmentation, as suitable patches of forest become less and less available for local populations to colonize.[667] "Because a nomadic crossbill may need to colonize many patches during its lifetime, even slight declines in colonization rates can be important."[668] As local individual populations vanish, the species or metapopulation moves closer to extinction.[669]

Climate change can complicate matters for crossbills further by synchronizing cone failures, perhaps even in mountainous regions where stands would ordinarily produce at different times.[670] Spruce bark beetle infestations—which the DEIS notes will likely increase with warming[671]—may also eliminate a food source and reduce crossbill populations.[672] Thus, climate change will likely compound the harmful effects of logging on the red crossbill.

C. The DEIS Does Not Adequately Address the Consequences of the Draft Forest Plan for Red Crossbills.

The DEIS mentions crossbills only in passing, grouping it with a handful of other species that "use hemlock/spruce/cedar forest (both old-growth and young-growth)" as habitat.[673] It does not recognize the designation of the red crossbill as a "Species of Greatest Conservation Need" [674] or otherwise identify it as deserving special attention. Nor does it specifically assess the potential impacts of the Draft Forest Plan on red crossbill habitat.[675]

In the general discussion of environmental consequences on wildlife resources, the DEIS notes that the preferred alternative would open up the most productive old-growth forest to logging over 100 years of any of the action alternatives.[676] It also observes that the cumulative effect of road construction associated with logging "has the potential to impact wildlife species through habitat fragmentation (especially migratory birds and other interior-forest associated species)."[677] Finally, it predicts that climate change will "result in changes to vegetation and thus, the

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[667] *Id.* at PDF 4.

[668] *Id.*

[669] *Id.*

[670] *See id.* at PDF 3.

[671] DEIS at 3-162.

[672] Walton, Red Crossbill at PDF 2.

[673] DEIS at 3-228.

[674] Walton, Red Crossbill at PDF 1.

[675] *See* DEIS at 3-257 ("All of the alternatives would result in a reduction of perching, foraging, and potential nesting habitat and the increase in fragmentation associated with timber harvest and road building;").

[676] *Id.* at 3-235 to 3-236 & Table 3.10-8.

[677] *Id.* at 3-262.

suitability of wildlife habitat, among other impacts," including "insect infestations."[678] It does not attempt to connect these impacts to the red crossbill. Given the species' precarious status and vulnerable natural history, the Forest Service must include a more detailed discussion of the possible harms red crossbills could suffer from logging and other activities.

The Service also should consider the needs of the red crossbill as a factor in deciding whether to continue industrial-scale old-growth logging for decade or more to come. The Forest Service should adopt a forest plan that ends such logging in less than five years and establishes reserves in productive forests located in varied climatological regions, to avoid synchronous cone failures and other catastrophes for crossbills.[679] Only meaningful conservation efforts and thoughtful planning can ensure the red crossbill's continued existence in the Tongass. If the agency is intent on continuing industrial-scale old-growth logging in the manner contemplated by the Draft Forest Plan, then the agency must then explain how its chosen plan will ensure the continued viability of the red crossbill.

IX. BATS

There are five species of bats in the Alexander Archipelago, all of which may be vulnerable to logging on the Tongass National Forest. One of these bat species, the Keen's myotis (*Myotis keenii*), has an extremely restricted range and it roosts in old-growth trees, which means management decisions that affect its habitat on the Tongass can have long-term population-level consequences for the species. The DEIS does not disclose the Draft Forest Plan's effects on the Keen's myotis or other bat species and is therefore deficient.

A. The Tongass is Home to Five Bat Species, Including Keen's Myotis.

Five species of bat have been documented in Southeast Alaska: little brown myotis (*Myotis lucifugus*), California myotis (*M. californicus*), long-legged myotis (*M. volans*), Keen's myotis (*M. keenii*), and the silver-haired bat (*Lasionycteris noctivagans*).[680] Southeast Alaska is thought to be the northern distributional limit for each of these species except the little brown myotis, which occurs in the interior of Alaska.[681]

Among these, the Keen's myotis may have the most restricted range of any species of bat in North America and occurs within a narrow strip of coastal coniferous rain forest from Juneau in

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[678] *Id.* at 3-262.

[679] Benkman at PDF 4-5.

[680] D. I. Parker *et al.*, *Distributional limits of bats in Alaska*, Arctic (1997): 256-265; J. L. Boland et al., *Survey of bats in Southeast Alaska with emphasis on Keen's Myotis (Myotis keenii)*, Northwest Science 83.3: 169-179 (2009) (Boland et al. 2009a).

[681] *Id*.

Southeast Alaska through southwestern British Columbia and into northwestern Washington.[682] Its population in Alaska is unknown, but it is considered small and rare.[683]

Although there is not enough data to conclusively determine habitat associations for all of the bat species in Southeast Alaska, many of them are primarily associated with forested habitats in other parts of their range.[684] The long-legged myotis and Keen's myotis are primarily associated with coniferous forests.[685] Keen's Myotis are also generally associated with cool, wet, coastal montane forests and karst features.[686] In Alaska, female Keen's myotis on Prince of Wales Island appear to primarily select day roost sites in old-growth forests.[687] The Keen's bat's "small size, low wing-loading ratio, and very low intensity echolocation call makes it well adapted for flying and foraging within structurally complex old forest."[688]

B. Keen's Myotis and Other Bat Species are Vulnerable to Logging Impacts.

Keen's myotis and other bat species are vulnerable to logging impacts. Because Keen's myotis preferentially roost in large diameter trees in old-growth forests, scientists have concluded that old-growth logging poses a threat to the species:

> Removal of large-diameter trees during timber harvest can reduce the number of potential roosts available to bats, and harvesting forests under short rotations can inhibit the development of suitable roosts over time. Our findings suggest that maintaining coniferous forests with a diversity of decay stages and high proportions of large diameter trees with defects in close proximity to riparian habitats provides critical roosting habitat for female Keen's myotis on Prince of Wales Island.[689]

In fact, studies show that all five species of bats in Southeast Alaska prefer unlogged old-growth forests and riparian zones and avoid clear-cuts and second-growth:

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[682] Boland et al. 2009a.

[683] U.S. Forest Service, *2009 Forest Service Alaska Region Sensitive Species List: Assessment and Proposed Revisions to the 2002 List* (2009) (USFS 2009), PDF 32-34.

[684] *Id*.

[685] *Id*.

[686] USFS 2009 at PDF 32-34.

[687] Boland, J.L. *et al*., *Selection of day-roosts by Keen's Myotis (Myotis keenii) at multiple spatial scales*, Journal of Mammalogy 90.1: 222-234 (2009) (Boland et al. 2009b) at PDF 1, 10.

[688] USFS 2009 at PDF 33.

[689] Boland et al. 2009b at PDF 10.

> Our study suggests that old-growth forests and riparian areas provide roosting and foraging habitat for the five bat species in southeastern Alaska. . . . Our data also indicate that clear-cuts are not important bat habitat, perhaps because clear-cuts do not provide roosting structure. . . . . Closed-canopy second-growth is not used by bats in southeastern Alaska.[690]

In addition to their preference for old-growth habitat, bats in Southeast Alaska possess biological traits that increase their vulnerability to logging: "[m]any factors contribute to species declines and risk of extinction. Intrinsic biological and ecological factors such as low reproductive rate and rarity may exacerbate consequences of extrinsic factors such as habitat loss."[691] In particular, "Keen's Myotis are long-lived with low reproductive rates. Low densities in conjunction with these life history traits may make populations of forest-dwelling bats in Southeast Alaska vulnerable to decline due to habitat perturbations, climate change, or a combination of factors."[692]

For all of these reasons, NatureServe ranks Keen's myotis globally as vulnerable, and ranks the species in Alaska as critically endangered to vulnerable.[693] Keen's myotis are managed as a sensitive species on national forests in Washington, and are listed as a candidate species for the state's threatened, endangered or sensitive classification. In the Forest Service's 2009 Alaska Sensitive Species Assessment, the Service did not recommend Keen's myotis for listing, primarily because of a lack of data, but it recommended that the species be closely monitored and re-evaluated when new information becomes available.[694]

C. The DEIS Fails to Consider or Disclose Impacts on Keen's Myotis and Other Bat Species.

The Forest Service must consider how the Draft Forest Plan will affect Keen's myotis and other bat species on the Tongass National Forest. Its own 2009 Alaska Sensitive Species Assessment relied in part on the 2008 Amended Forest Plan to decline listing, indicating that management of the forest drives the bat's status.[695] Moreover, scientists who study bats in the Tongass have warned that current and future logging threatens the species:

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[690] D. I. Parker *et al*., *Effects of timber harvest on bat activity in southeastern Alaska's temperate rainforests*, Bats and Forests Symposium, British Columbia Ministry of Forests, Victoria, BC, Canada (1996).

[691] Boland et al. 2009a at PDF 10.

[692] USFS 2009 at PDF 10.

[693] *See* NatureServe, *Comprehensive Report Species - Myotis keenii* and *National, State, and Provincial Conservation Status Rank Definitions*, www.natureserve.org (last visited Feb. 19, 2016).

[694] USFS 2009 at PDF 33-34.

[695] *Id*. at PDF 34.

> Extensive past and future harvesting in southeastern Alaska suggests a significant impact on these species. . . . Old-growth characteristics that provide suitable roost sites for bats do not develop until at least 150 years after harvesting. . . . Although questions remain about how habitat modification in southeastern Alaska affects bat populations, this study strongly suggests that present levels of timber harvesting will have a detrimental effect on these bat populations.[696]

Rather than addressing how the Draft Forest Plan affects bats, the Forest Service has entirely omitted them from consideration in the DEIS. The Forest Service must confront how an additional 15 years of old-growth logging will affect Keen's myotis and other bats in the Tongass National Forest. Moreover, because bats rely heavily on riparian areas in old-growth forests, the Forest Service must analyze how the Draft Forest Plan's weakening of protections for riparian management areas will affect the viability of the species.

TRANSPORTATION

This section of our comments addresses the Draft Forest Plan's changes to the transportation regime on the Tongass. The 2008 Amended Forest Plan places varying levels of restriction on road projects. This section describes the existing restrictions and their justifications. It then explains how the Forest Service must clarify the Draft Forest Plan so as to avoid the conclusion that the plan is a radical and unjustified departure from the status quo.

I. IN THE 2008 AMENDED FOREST PLAN, ROADS ARE PROHIBITED OR RESTRICTED IN MANY LUDS.

In the 2008 Amended Forest Plan, there are many LUDs in which roads are prohibited or seriously restricted. For planning purposes, the 2008 Forest Plan distinguishes between: 1) significant transportation projects designed to meet public transportation needs, and 2) all other roads. The first of these are categorized as "Transportation and Utility Systems," also called TUS, for which there is a special overlay LUD as well as management prescriptions specific to each regular LUD.[697] Transportation and Utility Systems include "state and federal highways, railroads, public hydroelectric power projects and associated facilities, powerlines 66kV or greater, and pipelines 10 inches or greater in diameter."[698] "For planning purposes, potential and proposed [Transportation and Utility Systems] corridors are depicted on the Plan map to show approximate corridor routes and widths."[699] Other roads are a broader group including all

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[696] Parker 1996.

[697] *See* 2008 Amended Forest Plan at 7-44 (definition); *id.* at 3-2 (LUD allocations); *id.* at 3-129 (TUS LUD); *id.* at 3-59 (TUS management prescriptions in Old-Growth Habitat LUD).

[698] *Id.* at 3-128.

[699] *Id.* at 7-6.

"motor vehicle route[s] over 50 inches wide, unless identified and managed as a trail."[700] They are not governed by their own LUD, but by management prescriptions for each regular LUD.

A. Thirteen LUDs Comprising 62 Percent of the Forest are "Avoidance Areas" for Transportation and Utility Systems.

The management prescription standards and guidelines for each LUD, which are the highest priority directions in the 2008 Forest Plan, classify it as either a Transportation and Utility Systems "avoidance area" or "window."[701] Transportation and Utility Systems can only be located in avoidance area LUDs if the Service determines there is no other feasible[702] route that would avoid the LUD.[703] A Transportation and Utility Systems window, by contrast, is "[a]n area potentially available for the location of transportation or utility corridors and sites."[704] In the 2008 Amended Forest Plan, there are thirteen avoidance area LUDs covering 10,379,771 acres, or 62% of the Tongass.[705]

B. When Transportation and Utility Systems are Constructed, an Overlay LUD is Activated That Includes Certain Environmental Protections.

In the 2008 Amended Forest Plan, there is a special Transportation and Utility Systems "overlay" LUD on the entire Tongass.[706] The overlay LUD takes effect after the location of a Transportation or Utility System is approved, when construction begins.[707] At that time its management prescriptions replace those of the underlying LUD in the applicable Transportation or Utility System corridor.[708] It describes how Transportation and Utility Systems should be designed and managed.[709]

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[700] *Id.* at 7-34.

[701] *Id.* at 1-3, 4-32; *see also, e.g.*, *id.* at 3-111 (designating the Modified Landscape LUD as a "window"); *id.* at 3-59 (designating the Old-growth Habitat LUD as an "avoidance area").

[702] "Feasible" means "[c]apable of being accomplished in a successful manner within a reasonable period of time, taking into account economic, environmental, technical, and safety factors." *Id.* at 7-12.

[703] *Id.* at 7-44 to 7-45; *see also, e.g., id.* at 3-59.

[704] *Id.* at 7-45; *see also, e.g., id.* at 3-111.

[705] *See* DEIS at 3-281 to 3-282, Table 3.12b-2.

[706] *See* 2008 Amended Forest Plan at 3-2 n.2 & n.5; *id.* at 3-128.

[707] *Id.* at 3-128.

[708] *Id.*

[709] *See generally id.* at 3-128 to 3-133.

The overlay LUD includes certain environmental protections. For example, the Forest Service will "[r]educe impacts to wildlife habitat and populations to the maximum extent feasible" within a Transportation and Utility Systems corridor.[710] It will also "[e]stablish a baseline [wildlife habitat] inventory . . . preceding or coinciding with Transportation Utility Systems development."[711]

C. Ten of the Thirteen "Avoidance Area" LUDs Also Prohibit or Seriously Restrict Other Kinds of Road Construction.

Roads that are not Transportation and Utility Systems are either allowed, restricted, or prohibited by the management prescription standards and guidelines for each regular LUD.[712] Of the thirteen LUDs that are "avoidance areas" for Transportation and Utility Systems, ten also prohibit or severely restrict construction of other kinds of roads. The management prescription standards and guidelines for the Research Natural Area, Wild River, Non-wilderness National Monument, Remote Recreation, Wilderness, and Wilderness National Monument Wilderness LUDs generally prohibit road construction.[713] They typically state that "new roads are not permitted," with specified exceptions such as when necessary to access valid mining claims.[714] The management prescription standards and guidelines for the Municipal Watershed, Special Interest Area, Land Use Designation II, and Old-growth Habitat LUDs seriously restrict roads.[715] They typically allow roads only under limited circumstances or "if no feasible alternative is available."[716]

D. The 2008 Amended Forest Plan's Restrictions on Roads Protect Wildlife and Other Forest Values.

In general, Transportation and Utility Systems avoidance areas "represent LUD's [sic] where . . . a [transportation or utility corridor] designation would not be desirable."[717] Other types of roads

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[710] *Id.* at 3-133.

[711] *Id.*

[712] *See generally id.*, Chapter 3.

[713] *See id.* at 3-34, 3-36 to 3-37, 3-38 (Research Natural Area); *id.* at 3-80 (Wild River); *id.* at 3-33 (Nonwilderness National Monument); *id.* at 3-49 (Remote Recreation); *id.* at 3-22 (Wilderness and National Monument Wilderness).

[714] *E.g., id.* at 3-33.

[715] *See id.* at 3-51, 3-56 (Municipal Watershed); *id.* at 3-40, 3-44 (Special Interest Area); *id.* at 3-69, 3-73 (Land Use Designation II); *id.* at 3-57, 3-61 (Old-growth Habitat).

[716] *E.g,*, *id.* at 3-61.

[717] 1997 FEIS at 3-88; *accord* DEIS at 3-21 ("A TUS 'avoidance area' is an area where the establishment and use of transportation or utility corridors and sites is not desirable given the LUD emphasis.").

are similarly undesirable in most of the avoidance area LUDs.[718] The assumption that roads generally will not be built in non-development LUDs is a basic component of the agency's conservation strategy:

> The framework of the old-growth conservation strategy relative to wildlife viability is now further described as two basic components: 1) the reserve system in terms of its ability to effectively maintain the integrity of the old-growth forest ecosystem through non-development LUDs such as Wilderness Areas, Research Natural Areas, Remote and Semi-Remote Recreation, and Old- Growth Habitat, among others, and 2) matrix lands where development, such as timber harvest and *road building*, is permitted that will alter the old-growth forest ecosystem on a portion of the lands.[719]

That is, when the 2008 Amended Forest Plan describes the conservation strategy, non-development LUDs are contrasted with areas where road building is permitted.

Although not fully adequate, the road-limiting measures in the 2008 Amended Forest Plan do provide some level of protection for wildlife and other forest values through the avoidance area safeguard and individual LUD restrictions. "[R]oads are one of the greatest risk factors to fish habitat, water quality, and to some wildlife species."[720] Through habitat degradation and fragmentation, "[r]oads pose *the* greatest risk to fish resources on the Tongass."[721] For species such as the brown bear, black bear, wolf, spruce grouse, mountain goat, and marten, roads increase the risk of excessive hunting and trapping by improving human access.[722] To illustrate this point, there is a direct positive relationship between roads and brown bear mortality.[723]

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[718] *See*, *e.g.*, 2008 Amended Forest Plan at 3-61 ("New road construction is generally inconsistent with Old-growth Habitat LUD objectives . . . ."); *id.* at 3-69 (listing as a goal for LUD II, to "[m]anage these areas in a roadless state to retain their wildland character"); *id.* at 3-34 (listing as an objective for the Research Natural Area LUD, to "[m]aintain the natural, undisturbed character of each area by . . . [p]ermitting no permanent facilities, and no roads or trails except for research purposes or as otherwise provided by law"); 1997 FEIS at L-160; *id.* at L-161.

[719] 2008 FEIS at 3-256 (emphasis added).

[720] 1997 FEIS at L-157.

[721] 2003 FEIS, Vol. I at 3-25 (emphasis added); 1997 FEIS at 3-64 ("The greatest risk to the fish resource is caused by roads."); *see also* DEIS at 3-114; 1997 FEIS at 3-65.

[722] *See* 2008 FEIS at 3-280 to 3-281 (brown bear); *id.* at 3-237 to 3-238, 3-384 (wolf); *id.* at 3-243 (spruce grouse); 3-268 (black bear); *id.* at 3-278 (marten).

[723] 1997 FEIS at 3-416.

II. THE FOREST SERVICE MUST CLARIFY THE DRAFT FOREST PLAN'S CHANGES TO TRANSPORTATION TO PREVENT POTENTIAL MAJOR ADVERSE IMPACTS TO FOREST RESOURCES.

The Draft Forest Plan and DEIS describe significant changes to the current forest plan that could, depending on how they are interpreted, result in fundamental changes that substantially weaken resource protection standards on the Tongass. To eliminate this problem, in the FEIS and adopted plan, there are three points that the Forest Service must make clear as to the scope and effect of the transportation-related changes included in this amendment. First, the Forest Service must specify that the Chapter 5 Transportation Systems Corridors direction in the Draft Forest Plan only applies to major roads, just like the 2008 Forest Plan's Transportation and Utility Systems LUD that the Draft Forest Plan deletes, to eliminate any suggestion that under the amended plan the LUD restrictions for roads to not fully apply to logging and other forest management roads. Second, the Service must ensure that in the absence of the 2008 Forest Plan's "avoidance area" protections, the Chapter 3 LUD restrictions for ordinary roads apply to major roads. Third, the Service must give a detailed account of the circumstances under which Chapter 5's Transportation Systems Corridors direction could supersede Chapter 3 LUD road restrictions, specifically addressing a scenario in which the LUD road restrictions would prohibit a major road. These clarifications are needed to ensure that the amendment does not unjustifiably reduce environmental protections governing roads.

A. The Forest Service Should Clarify that the New "Transportation Systems Corridors Direction" Only Applies to Major Roads.

The Draft Forest Plan's new Transportation Systems Corridors direction replaces the 2008 Forest Plan's Transportation and Utility Systems LUD direction as it applies to major roads. The Transportation and Utility Systems LUD in the 2008 Forest Plan governs both public energy projects and major roads.[724] The Forest Service deletes this LUD from the Draft Forest Plan and replaces its direction as to public energy projects with new direction on renewable energy,[725] which was one of the four significant issues the Forest Service identified during scoping.[726] Although the Forest Service has not identified a similar need for new direction as to major roads,[727] deletion of the Transportation and Utility Systems LUD leaves a gap in that major public roads would otherwise receive special consideration under the 2008 Forest Plan. The Draft Forest Plan fills this gap with new Transportation Systems Corridors direction. Like the 2008 Transportation and Utility Systems LUD, the new Transportation Systems Corridors direction is meant to "facilitate the development of[] existing and future major public Transportation . . . Systems."[728]

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[724] *See* 2008 Forest Plan at 3-129 (TUS LUD).

[725] DEIS at 3-21.

[726] DEIS at 1-9.

[727] *See supra* p. 9.

[728] 2008 Forest Plan at 3-128 (emphasis added); *accord* Draft Forest Plan at 5-13 (the purpose of the Transportation Systems Corridors direction "is to facilitate . . . existing and future transportation systems").

Unfortunately, the DEIS and Draft Forest Plan describe the scope of this change quite differently and with potentially much different effects. The DEIS explains that these "[Transportation Systems Corridor] plan components apply *only* to major road systems such as state and federal highways, railroads, and those identified by the State of Alaska in the current version of the SATP and applicable laws (for example, Section 4407 of Public Law 109-59, Title XI of the Alaska National Interest Lands Conservation Act, Public Law 96-487)."[729] Yet, due to drafting defects in the Draft Forest Plan, it is possible to conclude that the new Transportation Systems Corridors direction applies to roads of all types, including logging roads. In the glossary, "Transportation Systems Corridor" is defined as "[e]xisting and future transportation systems . . . ."[730] The term "transportation system" appears in the glossary, but its entry reads "[s]ee forest transportation system."[731] "Forest transportation system" is in turn defined as "[t]he system of National Forest System (NFS) roads, trails, and airfields on NFS lands."[732] Therefore, "transportation systems corridor" could erroneously be equated with "road," with the result that the Draft Forest Plan revises the direction for all roads.[733]

The Forest Service has not analyzed, justified, or even disclosed in the DEIS that it is rewriting the direction for all roads on the forest. Nor has it determined whether such a sweeping change would comport with its NFMA obligations. As discussed below, if the Draft Forest Plan's Transportation Systems Corridors direction applied to all roads, a further ambiguity in that direction could lead to the absurd conclusion that the entire forest is being opened to roads of any kind, including logging roads.

The Forest Service should clarify, in both the FEIS and the plan it adopts, that any new transportation direction such as the Transportation Systems Corridors direction in Chapter 5 of the Draft Forest Plan applies only to major roads.

B. The Forest Service Should Clarify that Chapter 3 LUD Road Restrictions Apply to Major Roads In Place of Former "Avoidance Area" Protections.

Even with the clarification that the new transportation direction in Chapter 5 applies only to new major public highways and not to logging or other Forest Service roads, the Draft Forest Plan does not clearly indicate how the new direction for major roads is to be applied with other LUD restrictions governing roads. The Forest Service should make clear that in the absence of "avoidance area" designations, the road restrictions for individual LUDs apply to major roads. Under the 2008 Forest Plan, major roads and utility projects could not be built in the majority of

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[729] DEIS at 3-277.

[730] *Id.* at 7-74.

[731] *Id.* at 7-74.

[732] *Id.* at 7-24.

[733] *See also* Draft Forest Plan at 5-13 (explaining that the purpose of Chapter 5 transportation direction is to facilitate "existing and future transportation systems *such as*" major roads, without limiting its applicability to major roads (emphasis added)).

the Tongass unless there was no feasible alternative in a Transportation and Utility Systems "window."[734] The Draft Forest Plan deletes "avoidance area" and "window" designations from the management prescriptions for each underlying LUD.[735] The DEIS repeatedly acknowledges that deleting avoidance areas will make it easier to build renewable energy projects.[736] The DEIS does not contain similar acknowledgements about major roads—nor would they be accurate as to major roads, because as explained above, most avoidance area LUDs still prohibit or seriously restrict roads even after the avoidance area direction is removed.[737] However, because under the 2008 Forest Plan major roads receive special treatment and therefore are not subject to Chapter 3 LUD restrictions for regular roads, the fact that under the Draft Forest Plan they *are* subject to the Chapter 3 LUD road restrictions is less than ideally clear. The Forest Service should make it explicit that major roads are subject to Chapter 3 road restrictions for individual LUDs in the Draft Forest Plan.

If, instead, the Forest Service intends that Chapter 3 LUD road restrictions should not apply to major roads, then deleting avoidance area status eliminates a protective barrier to siting major roads on the majority of the Tongass. The DEIS does not disclose such a sweeping change,[738] let alone analyze its significant environmental effects or compliance with NFMA.

C. The Forest Service Should Clearly Describe What, If Anything, the Transportation System Corridors Direction Will Supersede.

In addition to clarifying that the new direction is limited to major roads and that major roads are subject to Chapter 3 LUD road restrictions, the Forest Service must make the new transportation direction itself clearer. Specifically, the transportation direction in Chapter 5 will supersede direction in Chapters 3 and 4 "[s]hould a conflict or discrepancy in direction occur."[739] The Forest Service must thoroughly explain what will constitute a conflict in this context, providing examples of the circumstances under which Chapter 5 direction would or would not supersede Chapters 3 and 4.

On the one hand, no provision of the new transportation direction in Chapter 5 directly conflicts with the limitations on road construction in the LUDs found in Chapters 3 and 4, and thus the best interpretation of the changes may be that the Forest Service is to apply all of the standards

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[734] *See supra* pp. 111-12.

[735] Draft Forest Plan at 5-3, 5-13.

[736] DEIS at 3-92, 3-119, 3-120 to 3-121, 3-145, 3-148, 3-149, 3-151, 3-152, 3-155, 3-156, 3-485.

[737] *See supra* p. 113.

[738] *Cf.* DEIS at 3-92, 3-119, 3-120 to 3-121, 3-145, 3-148, 3-149, 3-151, 3-152, 3-155, 3-156, 3-485 (admitting that renewable energy projects will be easier to build under the Draft Forest Plan).

[739] Draft Forest Plan at 1-4; *id.* at 5-1; DEIS at 3-277 ("TSC plan components, e.g., standards and guidelines to the Forest Plan, would take precedence over other forest-wide and LUD-specific standards and guidelines (subject to applicable laws) where TSC are proposed or exist.").

and guidelines consistently. But there is language in the new direction that could be interpreted differently and could signal a major change in the restrictions governing even major roads. If, for example, Chapter 3 LUD road restrictions were interpreted to conflict with the Chapter 5 transportation direction because the purpose of that new direction is to "facilitate the availability of" Tongass land for major roads, then the Draft Forest Plan could be read to emasculate all major road restrictions outside of Chapter 5.[740] That would undoubtedly have significant environmental effects, as the DEIS is replete with descriptions of how roads harm species.[741] But the Forest Service has not said that the Draft Forest Plan makes it easier to build major roads on the Tongass, let alone that it potentially guts all Chapter 3 LUD road restrictions. The DEIS does not analyze or disclose this result.

In the FEIS and adopted plan, the Forest Service must describe and cabin the extent of potential conflicts between the Draft Forest Plan's Chapter 5 transportation direction and Chapter 3 LUD road restrictions. To name just one example, the Service must clarify that in the Old Growth Habitat LUD from which Chapter 3 precludes roads unless there is no feasible alternative,[742] Chapter 5 will not enable the Forest Service to build a road for which there is a feasible alternative.[743] If the Forest Service intends otherwise, it has failed to discharge its obligations under NEPA and NFMA with respect to the transportation changes.

RENEWABLE ENERGY

Commenters commend the Forest Service for proposing, in the Draft Forest Plan, a policy that promotes renewable energy on the Tongass. The Forest Service is right to encourage renewable energy projects because of their comparatively beneficial effects on climate and the environment, and because they may provide more affordable energy. Nonetheless, the new renewable energy direction in the Draft Forest Plan needs revision in two respects. First, it inappropriately removes and trumps environmentally protective management prescriptions in the 2008 Forest Plan, leading to unnecessary risks and putting the plan in tension with applicable laws. Second, it promotes biomass energy, which accelerates climate change and is harmful to the environment and human health.

I. THE DRAFT FOREST PLAN LIBERALIZES RENEWABLE ENERGY SITE PLACEMENT AND DIRECTS THE FOREST SERVICE TO ENCOURAGE RENEWABLE ENERGY PROJECTS.

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[740] And if Chapter 5 is not limited to major roads as it should be, *see supra* pp. 115-16, the Draft Forest Plan could open the entire forest to all roads, including logging roads. The DEIS does not even hint at this extreme result, which the Forest Service certainly has not justified or evaluated.

[741] *See* DEIS at 3-114 (admitting that "[r]oads pose the greatest risk to fish resources on the Tongass"); *id.* at 3-246 to 3-247 (explaining the adverse effects of road access on mountain goats); *id.* at 3-247 to 3-248 (black bears); *id.* at 3-251 (marten); *id.* at 3-252 (brown bear and wolf); *id.* at 3-260 (spruce grouse); 3-257 (migratory birds).

[742] Draft Forest Plan at 3-60.

[743] *See also supra* p. 113.

The Draft Forest Plan makes renewable energy a high priority. It includes a broad new objective to "encourage renewable energy production."[744] For purposes of the objective, the plan does not discriminate between types of renewable energy, except to prioritize energy that would be used locally over energy that would be exported.[745] The Draft Forest Plan makes the entire forest available: "[a]ll [National Forest System] lands may be suitable for renewable energy sites on a case-by-case basis in consideration of the LUD, ecological and social values, and benefit to Southeast Alaska communities."[746] In case of a conflict, the new renewable energy direction will take precedence over both management prescriptions for individual LUDs and forest-wide standards and guidelines.[747] The "avoidance area" designation in the 2008 Amended Forest Plan is removed.[748]

II. ALTHOUGH THE FOREST SERVICE SHOULD ENCOURAGE RENEWABLE ENERGY, IT SHOULD NOT DO SO AT THE EXPENSE OF ESSENTIAL ENVIRONMENTAL PROTECTIONS.

The Forest Service deserves praise for emphasizing renewable energy and for incorporating a goal to encourage renewable energy into the Draft Forest Plan. However, the Draft Forest Plan takes this policy too far to the extreme by ignoring its prior determination that 62% of the forest should be off limits to hydroelectric projects unless there is no feasible alternative location and by prioritizing the new renewable energy plan components such that they may override legally required and essential environmentally protective measures.

Even a renewable technology like hydroelectric energy that has the potential to yield net benefits when well-planned can be disastrous to salmon or other resources in the wrong place, at the wrong scale, or in the absence of protective standards and guidelines.[749] The Forest Service already made judgments in the 2008 Forest Plan about what are generally the wrong places to put hydroelectric facilities, establishing 62% of the forest as "avoidance areas" off limits to new hydroelectric facilities unless there is no feasible alternative location.[750] And even if there is no feasible alternative, transportation and utility systems LUD protections take effect that aim to minimize the project's environmental impact.[751]

In the Draft Forest Plan, by contrast, none of the forest is off limits to hydroelectric projects, and many of the transportation and utility systems LUD protections that currently apply to

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[744] Draft Forest Plan at 5-12.

[745] *See id.*

[746] *Id.*; *accord id.* ("Beach and estuary fringe is suitable for renewable energy sites.").

[747] *Id.* at 5-12.

[748] *Id.* at 5-3.

[749] *See supra* pp. 84-85.

[750] *See supra* pp. 111-12.

[751] *See* 2008 Forest Plan at 3-128 to 3-133.

hydroelectric projects have simply been deleted, with no corollary included in the new management direction.[752] Regardless of site-specific environmental review, these changes dramatically weaken the Draft Forest Plan's restrictions on hydroelectric power in a way that compromises important salmon habitat and other environmental values.

In addition, there are several places in the Draft Forest Plan where protective management prescriptions for individual LUDs could potentially conflict with, and therefore be overridden by, the new renewable energy direction's unqualified objective to encourage renewable energy and universal suitability determination.

One example of a possible conflict is between the new renewable energy direction and the management prescriptions for the Wild, Scenic, and Recreational Rivers LUDs. In these LUDs, the Draft Forest Plan requires that the Forest Service "not authorize development of hydroelectric power facilities" for projects on rivers designated under the National Wild and Scenic Rivers Act, as that Act requires.[753] A second potential conflict is between the new direction and management prescriptions designed to protect native species as required by the National Forest Management Act.[754] These prescriptions include everything from the general prescription that the Forest Service "[m]aintain contiguous blocks of old-growth forest habitat in a forest-wide system of old-growth reserves"[755] to the specific, such as "[a]void the placement of facilities near the Pike Lakes that would increase harvest pressure to the point where the viability of [the Northern Pike] is affected."[756]

That the Draft Forest Plan's current priority of direction might override important environmentally protective plan provisions, whether or not they are required by law, counsels against its adoption. Especially so, given that neither the DEIS nor Draft Forest Plan even attempts to justify in its current form the priority of direction with respect to renewable energy. These documents do not, for example, suggest that the proposed priority is needed to fulfill renewable energy goals at the expense of other important resource protection needs.

The Forest Service also has not explained why it is necessary to locate renewable energy sites in what are currently "avoidance" areas, let alone highly sensitive areas like beach and estuary fringe.[757] These and other vulnerable locations should be not be opened to development of

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[752] *Compare* Draft Forest Plan at 5-12 to 5-13 *with* 2008 Forest Plan at 3-128 to 3-133.

[753] Draft Forest Plan at 3-77, 3-83 to 3-84, 3-90 to 3-91; 16 U.S.C. § 1278 (providing *inter alia* that "no department or agency of the United States shall assist by loan, grant, license, or otherwise in the construction of any water resources project that would have a direct and adverse effect on the values for which such [designated national wild and scenic] river was established").

[754] *See* 16 U.S.C. § 1604(g)(3)(B); 36 C.F.R. § 219.9(b)(1).

[755] Draft Forest Plan at 3-61.

[756] *Id.* at 4-13; *see also id.* at 4-89 (for mountain goats, "[l]ocate facilities and concentrated human activities as far from important wintering and kidding habitat as feasible").

[757] Draft Forest Plan at 5-12.

renewable energy wholesale, and certainly not while the Draft Forest Plan includes destructive technologies in its broad definition of "renewable energy."[758]

The DEIS does not disclose the potential environmental impacts of opening the entire forest to renewable energy sites while encouraging renewable energy. It is not even internally consistent about whether the new direction will result in more renewable energy projects, saying variously that "the new components are likely to result in more energy project development over the long term"[759] and "the new Renewable Energy Plan Components . . . would simplify the process for projects, but would not necessarily result in an increase in the number of projects developed."[760] Nor is it consistent in describing the priority of direction that will apply under the Draft Forest Plan, saying both that "[c]onsistent with the 2008 Forest Plan, renewable energy projects need to be consistent with the standards and guidelines for the respective LUDs affected by energy development"[761] and that "should there be a conflict in direction, the proposed plan components in Chapter 5 would take priority over forest-wide and LUD-specific standards and guidelines (subject to applicable laws)."[762] Despite this confusion in the DEIS, the clear result of making it easier to build renewable energy projects is that more of such projects may be built. And the clear priority of direction in the Draft Forest Plan is that in case of a conflict, Chapter 5 governs, such that protections for individual LUDs that would otherwise apply could be overruled.[763] The DEIS's brief, vague and conclusory statements about how the renewable energy direction in the Draft Forest Plan will affect the risks to forest resources do not satisfy NEPA.[764]

The pursuit of renewable energy, while important, should not trump other provisions essential for environmental protection. The Forest Service should adopt a forest plan in which the management direction for renewable energy is subject to protective forest-wide standards and guidelines and management prescriptions for individual LUDs, not the other way around.

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[758] *See infra* pp. 122-23.

[759] DEIS at 3-379.

[760] *Id.* at 3-289; *see also id.* at 3-509 ("[t]he revised components may affect the timing and rate that new projects are proposed and developed on National Forest System (NFS) lands"); *id.* at 3-119 ("the proposed new direction under Alterntives [*sic*] 2 throught [*sic*] 5 would eliminate 'avoidance areas' which could increase the efficiency and likelihood [*sic*] of developing these project [*sic*]").

[761] *Id.* at 3-289.

[762] *Id.* at 3-288.

[763] Draft Forest Plan at 1-4.

[764] *See, e.g.*, DEIS at 3-145 (the renewable energy changes "could result in greater impacts to sensitive and rare plant species"); *id.* at 3-148 ("would result in increased risk of potential adverse effects to moosewort fern and its habitat from renewable energy development"); *id.* at 3-149 ("may result in a slightly increased risk of potential adverse effects to Macoun's thistle and its habitat"); *id.* at 3-121 ("would likely have little additional adverse effects to fish resources relative to current conditions").

III. THE FOREST SERVICE SHOULD NOT PROMOTE OR LIBERALIZE THE MOST DAMAGING RENEWABLE TECHNOLOGIES.

The Forest Service is right to encourage most kinds of renewable energy, but its encouragement should not extend indiscriminately to all renewable energy technologies, regardless of their environmental effects. For example, as explained above, poorly designed and located hydropower projects can adversely affect salmon, which are critical to the Southeast Alaskan way of life. *See supra* pp. 84-85.

Not all energy that is technically renewable is also worth what it costs the government, local communities and the forest. For that matter, the Draft Forest plan's definition of renewable energy is so broad as to be useless. It includes any energy resource that is "naturally replenishing but . . . limited in the amount of energy available per unit of time."[765] That definition is not helpful for two reasons. First, it does not differentiate between energy resources whose use will force climate change, such as biomass, and those whose use is essentially climate-neutral or even climate beneficial, such as wave action. Second, it does not exclude resources that replenish too slowly to matter for purposes of human civilization. As a result, old-growth forests that may take hundreds or thousands of years to replenish, if they can ever truly be replenished, are absurdly classified as "renewable energy." On a long enough timescale, even some fossil fuels might be replenished, but that does not mean the Forest Service should encourage their exploitation. The Forest Service should encourage renewable energy technologies for their climate, environmental, social, and economic benefits—not simply because they exploit resources that will be replenished at some point in the future. The Forest Service should discard its overbroad definition and adopt one that focuses on beneficial technologies.

If "renewable energy" was appropriately defined along those lines, it would not include biomass energy, or the burning of wood and wood-based products to produce energy.[766] Far from encouraging biomass energy, the Forest Service should eschew it. This technology's effect on climate change, its potential demands on forest resources that far exceed the demands of other renewable energies, its harmful emissions, and the fact that Tongass biofuels are not competitive in the market all make this form of energy a mistake to pursue. Moreover, the Forest Service has not considered these issues in the DEIS, despite the fact that they are integral to understanding the environmental effects of the Draft Forest Plan's new renewable energy direction.

There is no question that logging old-growth forest has a net climate forcing effect because of the superlative climate-protective functions of this kind of forest.[767] If old-growth forest is logged and then burned for energy, the net effect on the climate will only be worse than the already extremely detrimental effect of converting old-growth into wood products. But the Draft Forest Plan directs the Forest Service to encourage biomass without any limitation on the kinds of wood

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[765] Draft Forest Plan at 7-55.

[766] It is possible to generate biomass energy with materials other than wood, *see id.* at 7-6, but these comments focus on wood-based energy.

[767] *See supra* pp. 31-35.

that can be burned, and so could result in destroying centuries-old forest in the name of "renewable energy." Doing so would unequivocally advance climate change, and the Forest Service should not adopt any plan that could conceivably require the agency to encourage it.

However, even if only second-growth is burned, biomass is likely to be climate forcing. Even if no old-growth trees are burned for biomass, the use of second-growth trees and other Tongass wood for biomass could drive carbon-releasing increases in old-growth logging by making it more economical. The Forest Service appears to acknowledge this, projecting in the DEIS that developing biomass would generate demand for low- and utility-grade logs without specifying their origin.[768] And young growth and slash have their own climate protective effects that would be negated by burning, albeit less impressive than those of old growth.[769]

Ultimately, though, the climate risks of using Tongass second growth for biomass are demonstrated by the widespread scientific consensus that replenishing carbon stores by growing new trees in the place of those that are logged can take tens or hundreds of years in *any* forest, delaying biomass's alleged carbon neutrality until well past the time when carbon reductions are essential to prevent catastrophic climate change.[770] For natural forests like the Tongass as opposed to plantation forests, it is estimated to take thousands of years.[771] Some scientists even conclude that when modeled realistically as a series of logging events rather than just a one-off, biomass never achieves carbon neutrality.[772] And carbon recapture is all the more drawn-out in

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[768] DEIS at G-5 to G-6; *id.* at 3-459.

[769] A. Repo *et al.* concluded that in a young growth forest with a constant logging rotation, removing just forest residues for biomass energy would have a warming impact on climate for 62 years. *Can we produce carbon and climate neutral forest bioenergy?*, 7 GCB BIOENERGY 253, 253, 255, 257 (2015) (removing "branches, unmercantable tops and stumps . . . decreases the carbon input to dead wood, litter and soil carbon pools, and consequently decreases forest carbon stock").

[770] For example, T. Buchholz *et al.* found that among 123 peer-reviewed studies that calculated the "payback period," or length of time necessary to resequester the carbon released through burning for biomass, the average payback period regardless of forest type was estimated at 300 years. *A global meta-analysis of forest bioenergy greenhouse gas emission accounting studies*, 8 GCB BIOENERGY 281, 285, Fig. 3 (2015) (Buchholz 2015); *see also* J. S. Gunn *et al.*, *Biogenic vs. geologic carbon emissions and forest biomass energy production*, 4 GCB BIOENERGY 239, 240 (2012) (Gunn) ("If alternatives to fossil fuels include use of forests where C is emitted and resides in the atmosphere for long periods of time (e.g. decades or longer), a reduction of atmospheric concentrations of CO2 (e.g. to 350 ppm) will be difficult to achieve and may contribute to some degree of irreversible climate change." (citations omitted))

[771] Buchholz 2015 at 284 (payback periods in natural forests averaged 2,495 years and ranged up to 4,500 years).

[772] B. Holtsmark, *The outcome is in the assumptions: analyzing the effects on atmospheric CO2 levels of increased use of bioenergy from forest biomass*, 5 GCB BIOENERGY 467 (2013).

the Tongass because its trees are slow-growing.[773] Meanwhile, biomass generation spews carbon into the atmosphere at a higher rate per unit of energy than fossil fuel technologies.[774]

Under no circumstances does it make sense to burn any Tongass trees for the sake of the climate. "Close examination shows that the reverse is true: logging and burning trees will produce a near-term surge in carbon releases – greater than from burning coal – while eroding for decades the forests' ability to recapture those emissions."[775] Not surprisingly, the Forest Service does not cite any evidence in the DEIS or Draft Forest Plan that biomass energy will yield climate benefits if used on the Tongass, although a Forest Service employee whose jobs is to develop biomass energy asserts, without support, that biomass's carbon emissions compare favorably to those of fossil fuels.[776] For all the reasons explained above, that is incorrect.

Biomass energy also has harmful non-climate environmental effects. Unlike other renewable energy technologies, it requires more than a static area of the forest. By burning wood and increasing the demand for timber, it consumes an ever-larger portion of land, at growing costs to wildlife habitat and other non-timber uses.[777] And burning biomass emits air pollutants such as carbon monoxide, fine particulate matter, nitrogen dioxide, sulfur dioxide and volatile organic compounds that can increase the risk of cancer, cardiovascular disease, and adverse reproductive effects.[778] Because "the combustion of wood . . . poses a significant threat to human health," the American Lung Association opposes biomass, grouping it with coal and oil as a technology policymakers should transition away from "to cleaner alternatives."[779]

Biomass energy fuel products, or biofuels, such as wood pellets are a poor investment on the Tongass. There is currently little demand for Tongass biofuels and only one wood pellet producer in the forest.[780] Tongass biofuels are not economically competitive because wood

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[773] DEIS at 3-19 (noting that sequestration potential in the Tongass is limited by the fact that most stands have a relatively low growth rate).

[774] M.S. Booth & R. Wiles, *Clear cut Disaster: Carbon Loophole Threatens U.S. Forests* at 9 (2010); Gunn at 239 ("switching from fossil fuels to wood energy could actually result in increased levels of atmospheric GHGs, at least over a period of decades" (citations omitted)).

[775] Booth 2010 at 1.

[776] 769_05_000349 at PDF 1.

[777] *See* DEIS at 3-459.

[778] J. Lewtas*, Air pollution combustion emissions: Characterization of causative agents and mechanisms associated with cancer, reproductive, and cardiovascular effects*, 636 MUTATION RESEARCH 95 (2007); T. Jayarathne *et al.*, *Emissions of Fine Particle Fluoride from Biomass Burning*, 48 Envtl. Sci. & Tech. 12,636 (2014); H. Wang *et al.*, *Source Profiles of Volatile Organic Compounds from Biomass Burning in Yangtze River Delta, China*, 14 AEROSOL AND AIR QUALITY RESEARCH 818 (2014).

[779] Am. Lung Ass'n, Public Policy Position at 4, 5, 6 (June 25, 2015).

[780] AK Ctr. for Energy & Power, *A Review of Commercial-Scale Wood Pellet Boilers in Southeast Alaska* at 9-10 (2013).

pellets from Southeast Alaska are both more expensive and lower quality than wood pellets from British Columbia and Washington State.[781] As for biomass energy generation, the Forest Service cites no analysis suggesting that it can outperform other energy sources from an economic perspective. The Forest Service should not waste resources that could be invested in clean renewable energy to develop dirty, potentially climate forcing biomass energy and uneconomic biofuels.

Finally, the Forest Service has not considered the environmental effects of promoting biomass energy as NEPA requires. The DEIS says little about the climate effects of biomass energy, except to note that burning low-value logs for energy would reduce carbon storage compared to using these logs for lumber or building materials.[782] As to the non-climate environmental and health effects of Tongass biomass for energy and the odds that Tongass biomass energy products could become economically competitive, the DEIS is silent.[783] The Forest Service must consider all of these issues before it adopts forest plan components that promote biomass energy, such as those in the Draft Forest Plan.

In sum, the Forest Service should except biomass from its otherwise laudable goal of promoting renewable energy because it is a poor investment with serious detrimental environmental effects that the Forest Service has not adequately considered or disclosed. The DEIS mentions a Forest Service goal of converting 30% of fuel oil heating to biomass.[784] No such goal is incorporated into the Draft Forest Plan.[785] Nor should it be part of the plan, for all of the above reasons. To avoid confusion, the Forest Service should remove references to this goal from the FEIS.

KARST

The DEIS does not adequately disclose the effects of changes to the degree and methods of logging permitted in karst lands as between the Draft Forest Plan and 2008 Amended Forest Plan. These changes are substantial. The 2008 Amended Forest Plan permits thinning on medium and low vulnerability karst only within specific guidelines "when the karst management objectives can be met."[786] In contrast, the Draft Forest Plan allows even-aged management in low vulnerability karst and limited clear-cutting in medium vulnerability karst.[787] Other restrictions on second-growth logging in karst lands are removed.[788] Overall, the Draft Forest

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[781] *Id.*

[782] DEIS at 3-20.

[783] Although the DEIS discloses environmental effects of logging, it does not explain how the increased demand for timber due to biomass energy development could amplify those effects.

[784] DEIS at 3-459, 3-463, 3-475.

[785] *See generally* Draft Forest Plan.

[786] 2008 Amended Forest Plan at 4-24 to 4-25; *id.* at H-8.

[787] Draft Forest Plan at 5-9.

[788] *Compare* 2008 Amended Forest Plan at 4-24 to 4-25; *id.* at H-8 *with* Draft Forest Plan at 4-24; *id.* at H-8.

Plan is projected to allow a nearly 12 percent increase in acres logged from karst lands compared to the 2008 Amended Forest Plan.[789]

Although the DEIS discusses the types of effects that can result from logging in karst lands in general terms and estimates total maximum acres logged in karst lands under each alternative, it does not explain the overall environmental effects of the Draft Forest Plan's ongoing and increased logging in karst.[790] For example, the DEIS does not analyze the cumulative effects of logging 54,755 acres of karst on the productive plant, animal, and aquatic communities that rely on characteristics unique to valuable karst habitat.[791] Nor does it examine how a shift from limited commercial thinning to clear-cutting could change the kinds of effects associated with logging in karst.[792] It is not enough to say how many acres of karst could be logged under the proposed action alternative and note the kinds of effects that could result on those acres.[793] The Forest Service must translate those data into information that allows a meaningful comparison of the alternatives, such as relative risks to forest resources like plants, animals, soil and caves.

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[789] DEIS at 3-32 (showing 48,906 total acres to be logged on karst lands under alternative 1, the no action alternative, and 54,755 acres under alternative 5, the preferred alternative).

[790] *See generally* DEIS at 3-31 to 3-34.

[791] DEIS at 3-26 to 3-28 (describing the unique productivity of karst lands); *id.* at 3-32 to 3-33 (citing no study analyzing the effects of the proposed changes and no source less than eight years old other than monitoring reports from logged areas).

[792] *See* K. A. Harding & D. C. Ford, *Impacts of primary deforestation upon limestone slopes in northern Vancouver Island, British Columbia*, 21 ENVTL. GEOLOGY 137, 142-43 (1993) (documenting effects of clearcutting on karst lands, including loss of soil, moss, and litter, and in some cases "desertification that, in terms of human history, is permanent"); K. Allred, *Some carbonate erosion rates of Southeast Alaska*, 66 J. of Cave and Karst Studies 89, 96 (2004) (finding higher carbonate erosion rates in clearcuts); M. D. Bryant *et al.*, *Coho Salmon Populations in the Karst Landscape of North Prince of Wales Island, Southeast Alaska*, 127 Transactions of the Am. Fisheries Soc'y 425, 426 (1998) (noting that clearcutting can "confound the responses of stream geochemistry and salmonid populations to karst geology").

[793] *See* DEIS at 3-31 to 3-34.

For the reasons stated above, the Draft Forest Plan needlessly mires Southeast Alaska in the destructive and controversial practices of industrial-scale old-growth logging for another 16 years or more to come. The Forest Service should make changes in the Draft Forest Plan to ensure an end to industrial old-growth logging within five years and correct the deficiencies described above.

Respectfully,

Kristen Miller
Conservation Director
ALASKA WILDERNESS LEAGUE

Pete Nelson
Senior Policy Advisor for Federal Lands
DEFENDERS OF WILDLIFE

Dominick A. DellaSala, Ph.D
President, Chief Scientist
GEOS INSTITUTE

Alli Harvey
Our Wild America Alaska Campaign Rep
SIERRA CLUB

Miyoko Sakashita
Senior Counsel
CENTER FOR BIOLOGICAL DIVERSITY

Holly Harris
Staff Attorney
EARTHJUSTICE

Niel Lawrence
Alaska Director
NATURAL RESOURCES DEFENSE COUNCIL

Emily Ferry
Interim Executive Director
SOUTHEAST ALASKA CONSERVATION COUNCIL

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Conservation
Economics
Institute

Tongass Roadless Rule DEIS Economic Review

December 16, 2019

Alaska Roadless Rule,
USDA Forest Service, Alaska Region
Ecosystem Planning and Budget Staff,
P.O. Box 21628, Juneau, Alaska 99802– 1628.

FROM: Evan Hjerpe, Ph.D., Forest Economist

RE: Alaska Roadless Rule Draft Environmental Impact Statement (DEIS) Review of Economics

To whom it concerns,

I am submitting the following economic comments on the Tongass Roadless Rule DEIS. I am a forest economist with over a decade of professional experience researching the economic values of forest management in the U.S. and internationally. I have a Ph.D. in forest management and economics from Northern Arizona University. I spent five years working in the Tongass National Forest, researching economic forestry solutions that benefited southeast Alaskan communities. With the Draft Environmental Impact Statement (DEIS) to exempt the Tongass National Forest from the Roadless Rule, I am compelled to illustrate the shortfalls in economic analysis contained in the DEIS. My experience in forest economics and on the Tongass makes me highly qualified for reviewing the economic components of the DEIS.

In these comments, I detail how all economic valuations and trends associated with Tongass timber production and roadless protections clearly indicate that both national and Southeast Alaskan residents will incur greater benefits by keeping the Roadless Rule in place in Alaska. In fact, removing roadless protections from the Tongass will result in tremendous costs and damages to other economic sectors, national taxpayers, ecosystem services, and biodiversity. Because of the obvious economic perils and government waste that would result from removing Tongass roadless protections, the only reasonable alternative is the No Action alternative.

The Tongass Roadless Rule DEIS, released on October 17, 2019, is lacking credible economic analysis and falls well short of appropriate NEPA economic requirements. In the DEIS, USDA has ignored the best available economic science, which clearly illustrates that from almost every economic angle, the U.S. and southeast Alaskans are better off keeping the Roadless Rule intact. Not only has USDA ignored the best available science, they also did not provide any economic analysis to show how exempting the Roadless Rule on the Tongass would help Alaska or the nation. The disregard for incorporating the best available science, combined with providing no supporting economic analysis, undermines the validity of

appropriate NEPA analysis. These economic issues were flagged in detailed scoping comments,[1] yet were not addressed in the DEIS.

The overarching economic theme presented in the DEIS is that Tongass roadless timber production can occur in a vacuum without damaging the primary economic drivers of the region or the ecological integrity of the Tongass, and without additional costs to the agency. This is perhaps the biggest flaw of the NEPA analysis and illustrates a poor understanding of real-world economics. With numerous deficiencies, USDA's economic analysis in the DEIS does not accord with economic theory and does not meet the acceptable standard for economic analysis on public lands as mandated by NEPA, the NFMA, and appropriate forest planning. With such a paucity of credible economic analysis, this DEIS and its management direction, is fatally flawed and must be withdrawn.

The major deficiencies regarding economics in the DEIS include:

- USDA did not validate the State of Alaska's claims of economic harm from the Roadless Rule, which are meritless and unsupported.
- USDA's purpose and need are irrational, and they have provided no logical rationale, economic or otherwise, to justify the proposed rule.
- USDA's distributional effects analysis shows the proposed rule will result in zero increases in regional employment, output, or income. USDA has thus verified that there is no logical rationale for the proposed rule, as the entire rationale is predicated on providing further economic development to Southeast Alaska.
- The Cost-Benefit Assessment required for this rulemaking is does not pass scientific or legal muster and does not accord with standard economic theory.
- USDA included timber harvesting costs in Tongass IRAs that are erroneously projected to decrease under the proposed rule, but inexplicably did not include any increased road construction, decommissioning, or maintenance costs.
- In the Cost-Benefit assessment, USDA has mistaken distributional effects of changes in industry revenues for costs and benefits to be used in economic efficiency analysis.
- USDA has not quantified any costs or benefits to the US Forest Service (USFS) or society at large, despite numerous cost increases that will result from the proposed rule.
- USDA's net present valuation (NPV) of costs and benefits appears to be wildly inaccurate.
- USDA has provided almost no supporting economic data to support their claims of harvest cost savings, nor any supporting engineering or economic analysis to project road needs and costs for timber production in Tongass IRAs.
- USDA has omitted most of the Tongass economics literature illustrating the severe economic inefficiency of Tongass timber production and peer-reviewed research illustrating conservation benefits for protecting Tongass old growth.
- When including increased road costs and lost conservation benefits, credible cost-benefit analysis illustrates that the proposed rule will result in losses ranging from $26 million to $48 million, at a minimum.
- USDA has not included synthesized economic research showing that the Tongass timber program has an average cost-benefit ratio of 25. That is, for every $1 million received by the

[1] See Tongass Roadless Rule scoping comments submitted by Dr. Evan Hjerpe on 10/15/18.

U.S. Treasury for stumpage fees, U.S. taxpayers pay $25 million in federal agency costs to subsidize timber harvests.

- In the Agency and Regulatory Costs section, USDA has failed to quantify a single cost to the agency, despite many costs to choose from for analysis.
- USDA did not provide an ecosystem services perspective of the proposed rule, despite its current prominence as the USFS's dominant management paradigm.

1. Introduction

The Tongass National Forest (hereafter, the Tongass) is renowned for its pristine old growth rainforests. Tongass roadless forests provide the iconic backdrop to numerous cruise ships that show tourists the Inside Passage. Tongass roadless forests also provide habitat for spawning salmon and directly support one of the biggest economic drivers in the region---commercial fishing. By providing the economic goods and services required to produce the primary regional economic activities of tourism, commercial and sport fishing, and subsistence, Tongass roadless forests are critical to the provision of widespread economic benefits and impacts to Southeast Alaska. Nationally, Tongass roadless forests are also a major source of economic benefit through carbon sequestration and by providing immense passive use value in the form or option, bequest, and existence values held for scarce and pristine coastal temperate old growth rainforests.

Opening Tongass roadless areas to development will result in tremendous economic losses for the American public and residents of Southeast Alaska. Removing roadless protections in the Tongass is certainly NOT in the best interest of Alaskan residents, nor is it in the best interest of national residents. Jeopardizing such valuable landscapes with irreversible environmental damages is extremely short-sighted and will result in damages to every industry except the timber industry. Exempting the Tongass from the Roadless Rule will also perpetuate a corporate welfare program where taxpayers are forced to subsidize a damaging industry to the tune of $30 million a year. The proposed Rule in the DEIS is both fiscally and ecologically irresponsible.

Inventoried Roadless Areas (IRAs) recognized by the 2001 Roadless Rule provide for numerous economic benefits and impacts to adjacent communities and the nation. By keeping roadless areas undeveloped, nature is allowed to provide high quality ecosystem services, or benefits to mankind. The most notable and obvious ecosystem services protected by the Roadless Rule center on water quality and supply, biodiversity, and carbon storage. Roadless forests in the U.S. contain many headwaters, pristine forests, and critical fish and wildlife habitat. While the majority of ecosystem services produced by roadless forests are not traded in financial markets, there are non-timber forest products such as mushrooms, berries, firewood, and wild game and fish that are marketed or act as monetary supplements for grocery budgets. This is especially true of economies that include a high rate of subsistence activities such as Southeast Alaska.

Much like Wilderness areas and other protected lands, roadless forests are a critical component of our national conservation lands. This reserve of conservation lands is akin to a bank account of nature that continually collects interest and becomes more valuable into the future. Natural areas are rapidly diminishing world-wide and in the U.S. As natural landscapes, and their associated natural disturbance

regimes, continue to shrink and scarcity increases, remaining natural areas increase in value.[2] These natural areas hold tremendous option value for the future, including potential medicinal cures, refuge for climate change-affected species, and chemical compounds for agriculture and manufacturing. Eliminating the protection for roadless areas could eliminate future options associated with these natural areas.

Nationally, roadless areas are important for regional economic benefits and impacts. Recent research[3] illustrated the overall economic value of Lower 48 roadless areas. Results show over 11 million annual visits to roadless areas in the Lower 48, that provided for $500 million of economic benefit, or personal value to recreationists, and millions of dollars in regional economic impact from outdoor recreationists who purchase goods and services in the small towns adjacent to roadless areas. Roadless areas are also highly regarded for their passive use values such as existence, option, and bequest values, estimated at $8 billion annually in the continental U.S.[4]

A major reason for implementing the original Roadless Rule was to help prevent wasteful government spending, saying that, "budget constraints prevent the Forest Service from adequately maintaining the existing road system."[5] The original Roadless Rule also indicated that a national rule was necessary because the Forest Service has "the responsibility to consider the 'whole picture' regarding the management of the National Forest System, including inventoried roadless areas" and "[l]ocal land management planning efforts may not always recognize the national significance of inventoried roadless areas and the values they represent in an increasingly developed landscape."[6]

This is very true of the Tongass, where local land management has failed to recognize, or account for, the national significance of Tongass roadless areas and has shown disregard for the "whole picture" as related to total economic benefits provided. Likewise, Alaska has a $68 million Forest Service road maintenance backlog; nationally the USFS road maintenance backlog is estimated at $3.2 billion.[7] With such an extensive backlog of road needs, why is USDA attempting to increase this deficit? Where is the collective national taxpayer voice in this process?

Given that roadless areas are important for wildlife, water quality, and recreation, there is a tremendous need to have the economic values of roadless areas on the Tongass and elsewhere documented in the public record during the NEPA process. There are over nine million acres of IRAs in the Tongass. These roadless areas gained protection under the 2001 Roadless Rule but would be exempted and opened up for extractive development under the proposed rule. Opening Tongass roadless areas to timber harvest will result in significant environmental consequences---effects that must be, but are not, disclosed in the DEIS.

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[2] Holmes, T. P., Bowker, J. M., Englin, J., Hjerpe, E., Loomis, J. B., Phillips, S., & Richardson, R. (2015). A synthesis of the economic values of wilderness. *Journal of Forestry*, *114*(3), 320-328.

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[4] *Ibid.*

[5] 36 C.F.R. §§ 294 (2001), Federal Register pp. 3245-3246.

[6] *Ibid*.

[7] USFS responses to Rep. Mike Quigley.

The best way to visualize the economic value of roadless forests is to examine landscapes that have been developed for timber production or mining. That is, what will be lost when roadless forests lose their protection? What is the economic cost associated with land degradation and pollution? These are the costs that need to be clearly assessed in the EIS process but are lacking in the DEIS. What are the benefits of keeping roadless protections, such as avoided costs of pollution and resource damages? A full assessment of the trade-offs associated with this rulemaking needs to be clearly delineated. The current DEIS is insufficient and must be withdrawn and redone.

2. There is No Purpose and Need for a Tongass Roadless Exemption

USDA and the State of Alaska have not come up with any rational purpose and need for a new rulemaking process. Throughout the DEIS, USDA indicates that the impetus for this rulemaking comes from a petition from the State of Alaska. The State of Alaska (SOA)'s petition (DEIS: Appendix A) to USDA for this rulemaking claims the 2001 Roadless Rule has resulted in "extensive damage… to the economic and social fabric of Southeast Alaska…" The Petition also states that a roadless exemption is needed for the socioeconomic well-being of Tongass residents. However, the State of Alaska provides zero evidence of economic damages coming from the 2001 Roadless Rule, and zero evidence that exempting the Tongass from the Roadless Rule will improve the socioeconomic well-being of Tongass residents. In fact, there is overwhelming economic evidence to the contrary. USDA apparently did not verify the economic rationale from the State's petition, nor did they provide any logical economic reasoning for the preferred alternative in the DEIS. In lieu of evidence-based research from the State of Alaska or USDA, I will first illustrate why the purpose and need for this rulemaking are faulty and then provide economic explanations of the trade-offs associated with exempting the Tongass from the Roadless Rule.

2.1 The Rationale from the State of Alaska's Petition is Inaccurate

The rationale throughout SOA's petition is crystal clear: they are asking the USDA "…to support a diverse and robust forest products sector in Southeast Alaska." It is also crystal clear that the SOA wants a forest products sector based strictly on clearcutting old growth forests. Not only does the SOA's petition request USDA to revise the Roadless Rule on the Tongass, it also requests that USDA revise the 2016 TLMP Amendment *and* revise the established transition from old growth to young growth harvests (DEIS: A-4).

The SOA's petition suggests that the Tongass Roadless Rule is an unnecessary protective policy layer, stating that these roadless areas would be protected with or without the Roadless Rule. Not as clearly stated, but deduced by the content of the entire petition, is that the SOA is seeking access and funding to harvest the most accessible and productive old growth stands currently protected by the Roadless Rule---the 165,000 acres of old growth that the preferred Alternative (6) in the DEIS would convert from unsuitable for timber production to suitable. Is the Tongass Roadless Rule unnecessary and duplicative? No--- especially not for the most accessible and productive old growth, areas with some of the greatest ecosystem service production, that would be on the chopping block.

The SOA claims of economic harm from the Roadless Rule are meritless, as are their claims that regional economic and timber industry conditions are the same as 2003. The SOA petitions states:

> "Addressing the serious socioeconomic consequences to Alaskans and complying with ANILCA and TTRA are all compelling rationale for a Tongass Exemption today, as they were in 2003….The State respectfully submits this petition for a rulemaking to exempt the Tongass from the Roadless Rule in the interest of the socioeconomic well-being of its residents." (DEIS: A7-A8).

These "serious socioeconomic consequences" of the Roadless Rule are never specified. How exempting the Tongass from the Roadless Rule will be in the best "interest of the socioeconomic well-being of its residents" is never detailed. Despite an overwhelming lack of evidence to back up these claims, USDA rests its entire purpose and need on the SOA's petition. Additionally, economic conditions in Southeast Alaska have changed substantially since 2003. TTRA "market demand" is down to 46 million board feet and Asian export markets are waning due to tariff and trade war effects. Mill capacity is a fraction of that in 2003 and the transition to Tongass second growth has commenced. Regional Tongass timber employment currently represents less than one percent of regional employment.[8] The two largest private industrial sectors in Southeast Alaska are tourism and commercial fisheries, making up about 15% and 10% of regional private employment respectively.[9] These two industries, tourism and seafood production, are the real drivers of the regional economy and are directly dependent upon the protected roadless forests of the Tongass. The economics question is, why would the federal government remove roadless protections to boost a dying industry (i.e., logging) while irreversibly damaging the natural resources that the rest of the regional economy depends on?

The SOA claims that a Tongass exemption from the Roadless Rule is needed for economic development (i.e, clearcutting old growth), but has provided zero economic evidence for this need, nor any details of how societal benefits would outweigh the costs of development. In summary, the only purpose or need for this rulemaking is to direct greater federal tax dollars to build roads to clearcut old growth forests in Southeast Alaska. This is illogical from almost all perspectives and is an insufficient purpose and need for such a damaging rulemaking.

2.2 The DEIS Purpose and Need is Irrational

Likewise, the Forest Service makes bold claims about supposed economic benefits of the proposed rule:

> "The proposed rule is expected to yield a range of benefits (or cost reductions) derived from greater flexibility and a positive net benefit (USDA Forest Service 2019b) and economic opportunities for small business. For example, greater flexibility is provided for the selection of future timber sale areas and sale design (depending on the planning areas selected); and could, in turn, potentially improve the Forest Service's ability to offer economic sales that meet the needs of industry." (RIA: 26)

Upon closer examination, only one benefit has been illustrated---cost reduction in felling, yarding, and loading harvest costs. As shown below, this is not a benefit nor is it accurately calculated. There simply is no positive net benefit from the proposed rule. The citation provided to supposedly show positive net benefit contains no document that illustrates increased net benefit. To provide a useless citation as the source for demonstrating improved economics from the rulemaking is suspect and indicates that there

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[8] Alexander, B. and R. Gorte. 2014. The Tongass National Forest and the Transition Framework: A New Path Forward? Bozeman, MT: Headwaters Economics, 32p.
[9] *Ibid*.

are actually no positive benefits to report. Finally, the only effects on small businesses demonstrated in the DEIS are adverse effects on small tourism guides and outfitters.

The real purpose and need given in the DEIS is that:

> "The USDA and Forest Service believe the 2001 Roadless Rule prohibitions on timber harvest and road construction/reconstruction can be adjusted for the Tongass in a manner that meaningfully addresses local economic and development concerns and roadless area conservation needs." (DEIS: ES-2).

While this might be a noble intention, the DEIS tells us that neither economic development concerns, nor roadless area conservation needs, will be meaningfully addressed. In the DEIS, it is estimated that there will be no changes in regional employment and no changes in overall timber production. The proposed rule would eliminate all Roadless protections from 9.2 million acres; this certainly does not address roadless conservation needs. How exactly does the preferred rule help the regional economy or address roadless area conservation needs?

We know that timber harvest and road construction miles in Tongass roadless areas will be greater than zero. Otherwise, there is no purpose for this rulemaking. "Alternative 6 is the preferred alternative and provides maximum additional timber harvest opportunity as the full exemption alternative, which was requested by the State of Alaska's petition." (DEIS: 2-16) We also know that a single mile of constructed roads, and a single acre of clearcut old growth, has adverse environmental consequences on water quality, wildlife habitat, fish, deer, and carbon storage. We know that timber harvests in roadless areas will require more road construction than harvests in the roaded timber base. We know a mile of new roads costs substantial amounts of taxpayer dollars, as does road decommissioning. We know that U.S. taxpayers subsidize timber production on the Tongass at a rate from approximately $500--$1,100 per thousand board feet of timber.[10] These average costs, benefits, and damages are not disclosed in the DEIS. USDA must, at a minimum, cite this information and ultimately this requires major revisions to the DEIS.

Preferably, these quantified economic and ecological values should be incorporated into sensitivity analysis that illustrates overall average costs for anticipated small, medium, and large incursions into Tongass roadless areas (e.g., see section 4.2 later in this document). This is not difficult and would be much more reasonable than acting as if the preferred alternative will have zero repercussions on the ground. If nothing will happen from this rulemaking, there is no need for it. Providing rough averages and details of obvious implications of the various alternatives is required by NEPA. That the DEIS simply ignores critical environmental consequences is a fatal flaw rendering the DEIS unusable. A new DEIS must be conducted to account for these fatal flaws.

3. The Economic Reality Ignored in the DEIS

USDA is hitching their horse to old growth timber harvesting on the Tongass, which is about as economically and environmentally prudent as subsidizing antiquated extractive industries like coal

[10] Alexander, B. and R. Gorte. 2014. The Tongass National Forest and the Transition Framework: A New Path Forward? Bozeman, MT: Headwaters Economics, 32p.

mining. With excessive subsidies already required for any Tongass timber production, opening Tongass roadless areas to timber development will only increase total subsidies. Exempting the Tongass from the Roadless Rule will also cause economic harm to Southeast Alaska residents by threatening tourism, commercial fishing, sport fishing, and subsistence economies. Corporate welfare provided to the timber industry comes at the costs to southeast Alaskan residents, Alaskan residents, and U.S. residents. USDA must incorporate the best available science to come to a decision in this rulemaking process. If the rulemaking process is actually bound by NEPA guidelines, utilizes the best available science, and maintains integrity owed to the public, the obvious conclusion would have been selecting the no-action alternative as the preferred alternative.

3.1 Economic Trends of Tongass Timber Production Will Not be Reversed by Opening Tongass Roadless Areas for Development

Large-scale timber production in Alaska has never been sustainable, nor has it ever been profitable. In total, U.S. taxpayers have paid billions of dollars to fund Tongass old growth logging. Southeast Alaska has suffered from the resource curse, where communities propped up by subsidized resource extraction, are left worse off after the experiment ends. The jobs were never sustainable, and the remaining residents are stuck with heavily damaged forests and watersheds from logging. The Tongass has always been a "last in, first out" supplier of wood due to exorbitant production costs, extreme isolation from markets, and a lower quality of wood.[11] These multiple factors make it impossible to have large-scale industrial timber production on the Tongass *without* massive taxpayer subsidies.

The Tongass is simply too remote and too mountainous to ever be profitable in large-scale timber production.[12] Most of the Tongass includes low-value trees, which has been exacerbated by a legacy of high-grading. The biggest and the best trees have already been cut. With extreme isolation and ruggedness, the Tongass has the highest logging and processing costs anywhere. Compared to British Columbia and the Pacific Northwest (PNW) region of the continental U.S., Southeast Alaska has the highest timber manufacturing costs and the lowest stumpage prices, with logging costs being 66% greater than in the PNW.[13]

While the Tongass has always been the most inefficient timber production region in the U.S, the absurdity of perpetuating Tongass old growth logging is that Tongass timber production is only getting more and more inefficient. Recent research[14] shows that for the last six years, Tongass timber expenses by the USFS are $122.5 million while stumpage received, or revenue, is $3.4 million. With costs exceeding benefits by 36 times, the Tongass timber program is an incredibly wasteful federal program.

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[11] Robertson, G. and D. Brooks. 2001. Assessment of the competitive position of the forest products sector in southeast Alaska, 1985–94. Gen. Tech. Rep. PNW-GTR-504. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 29 p.

[12] Crone, L. 2005. Southeast Alaska economics: A resource-abundant region competing in a global marketplace. Landscape and urban planning 72: 215-233.

[13] Robertson, G. and D. Brooks. 2001. Assessment of the competitive position of the forest products sector in southeast Alaska, 1985–94. Gen. Tech. Rep. PNW-GTR-504. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 29 p.

[14] Taxpayers for Common Sense. 2019. Cutting Our Losses: 20 Years of Money-Losing Timber Sales in the Tongass. Available at https://www.taxpayer.net/wp-content/uploads/2019/09/TCS-Cutting-Our-Losses-2019-.pdf

The inevitable decline in Tongass timber production is obvious. Employment from Tongass timber production currently sits at an all-time low of approximately 61 jobs. (DEIS: 3-28) The economic trends for timber production and jobs have been steadily decreasing since the closure of the region's pulp mills for both Tongass NF associated production and the entire southeast Alaska region. Due to the exorbitant production costs, isolation from markets, and long-term high grading discussed above, a large-scale timber industry from Tongass production simply isn't feasible. The economic trends will not be reversed by removing roadless protections, nor will it be overcome by subsidizing the industry $30 million a year. A visual projection of Tongass timber jobs clearly illustrates the futility of wasting taxpayers' dollars on corporate welfare.

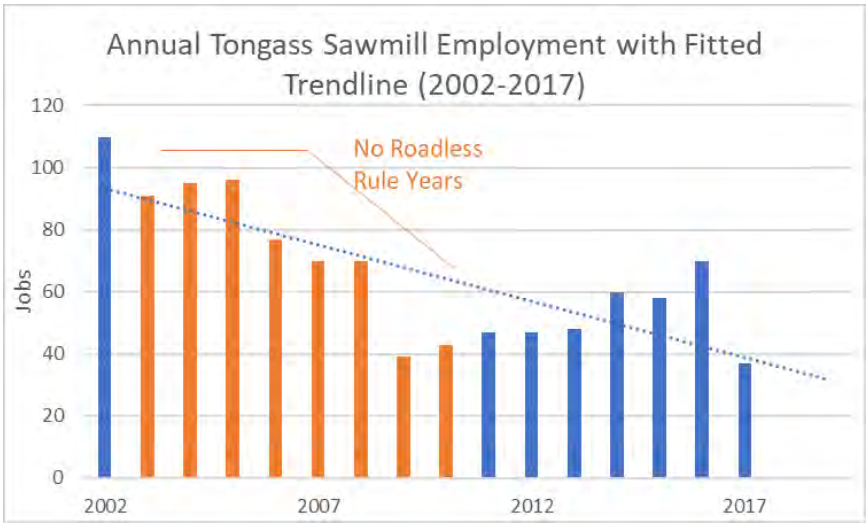

Figure 1: Source for employment data--DEIS Table 3.2-2: 3-28.

Figures 1, 2, and 3 show the steady decline of timber related employment in southeast Alaska and the Tongass. It is important to remember that the last 20 years included multiple Administrations, eight exempt years from the Roadless Rule, and consistent and heavy federal subsidization of logging roads and timber production. For the SOA to claim that the same economic rationale for exempting the Tongass from the Roadless Rule in 2003 exists today is blatantly ignoring all market evidence since then.

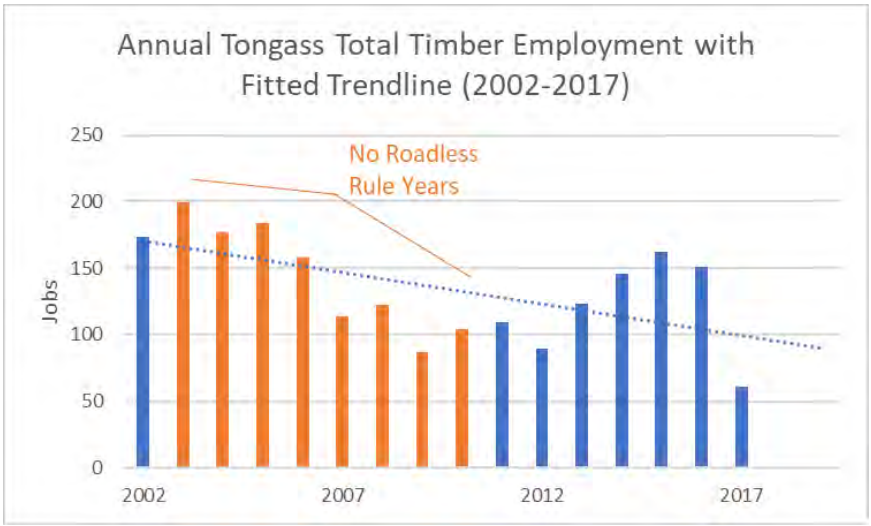

Figure 2: Source for employment data---DEIS Table 3.2-2: 3-28.

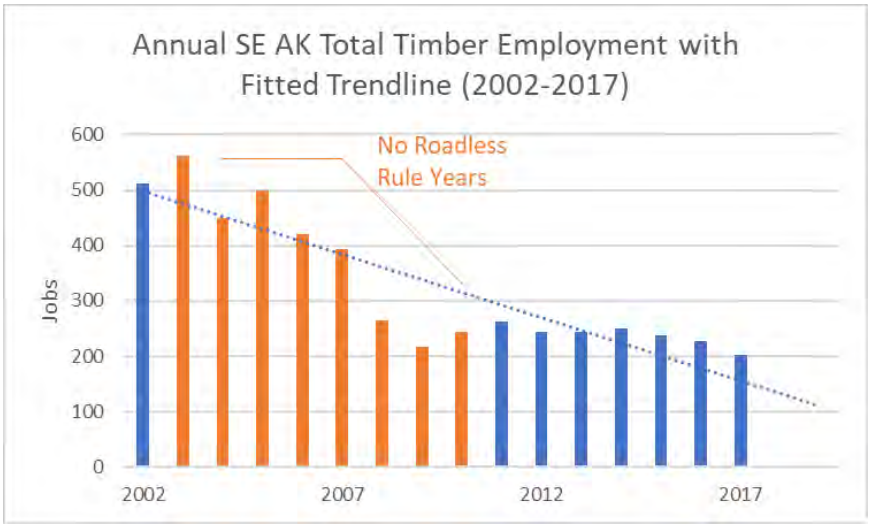

Figure 3: Source for employment data---DEIS Table 3.2-2: 3-28.

The timber industry and Alaska politicians hope that increasing federal timber supply on the Tongass will revive a shell of an industry. However, these hopes are unfounded. Wood supply has never been the problem for the lack of profitability for Tongass timber production. A 2004 legal decision (U.S. Federal Court of Claims, No, 95-153C, Alaska Pulp Corporation (APC) v. United States of America) showed that APC was unprofitable regardless of provisions associated with the Tongass Timber Reform Act (TTRA). Furthermore, recent Tongass timber sales demonstrate that wood supply is still not an issue. From 2000-2010, a period largely exempt from the Roadless Rule, nearly 50 % of Tongass timber sales offered were not bid on at all; of the timber that did sell, approximately 40 % of that supply was defaulted on by the

operators or mutually cancelled.[15] The most recent efforts to increase logging by subsidizing prohibitive road construction and timber sale planning costs are still proving to be impossible to overcome the severe lack of profitability. The North Kuiu timber sale has been offered in 2016 and again in 2018. Despite extreme subsidies attached to the timber sale, the sale has received zero bids.[16]

The clear trends indicate that markets and employment for Tongass timber are dying because industrial logging on the Tongass is just too expensive, with or without the Roadless Rule. How can USDA defend its purpose and need for expanding timber opportunities when all evidence shows that exempting the Roadless Rule will not be enough to overcome the dismal economics that characterize southeast Alaskan timber production? Where is the support for SOA's claims in their petition? Where is the scientific evidence in the DEIS? Why is the best available science regarding the lack of economic viability for Tongass timber excluded from the DEIS? This undermines the entire DEIS.

4. The Regulatory Impact Assessment and Cost-Benefit Assessment are Not Credible and are Legally Inadequate

USDA has determined that this rulemaking is a significant rule, per the direction of Executive Orders 13563 and 12866. These orders require federal agencies to conduct a regulatory analysis for economically significant regulatory actions, or those that have an annual economic effect greater than $100 million or adversely affect the economy or economic sectors. As USDA estimates the economic effects of the proposed rule to be less than $100 million annually, the reasoning for determining this as a significant rule is due to anticipated adverse effects to the economy and individual economic sectors. Executive Orders 13563 and 12866 mandate cost-benefit analysis of significant rules and instructs the agency to choose regulatory approaches that "maximize net benefits." From the RIA:

> "Executive Orders 13563 and 12866 direct agencies to assess costs and benefits of available regulatory alternatives and, if regulation is necessary, to select regulatory approaches that maximize net benefits (including potential economic, environmental, public health and safety effects, distributive impacts, and equity). Executive Order 13563 emphasizes the importance of quantifying both costs and benefits, of reducing costs, of harmonizing rules, and of promoting flexibility." (RIA: 4)

Despite the noted importance of quantifying both costs and benefits, USDA has failed to quantify any real costs or benefits in the RIA. One cost and one benefit are quantified---savings in timber harvest costs (which are not really savings) and revenue losses for outfitters and guides. The purported savings in harvest costs are inaccurate and neither one of the costs are appropriate inputs for cost-benefit assessment and should have been included as distributional effects. In addition to inappropriate and inaccurate inputs, the actual calculations of the Net Present Valuation (NPV) appear to be wrong.

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[15] Hjerpe, E. 2011. Seeing the Tongass for the Trees: The Economics of Transitioning to Sustainable Forest Management. Washington: The Wilderness Society, 61p. Available at https://www.researchgate.net/publication/301553259_The_Economics_of_a_Tongass_Transition

[16] https://www.juneauempire.com/news/controversial-timber-sale-cant-find-a-bidder/

In this section, I illustrate the numerous shortfalls in the RIA. These shortfalls are fatal flaws for the entire DEIS, as the economic analysis contained in the RIA is the basis for the environmental consequences in the DEIS. After a review of the analysis included in the RIA, I provide a credible, economically rigorous cost-benefit assessment of the proposed rule that should be used in a revised DEIS. The results of the new CBA clearly show, with reasoned determination, that the benefits of the exempting the Tongass from the Roadless Rule are much less than the costs. That is, the preferred rule is clearly not in the best interest of the public. To meet legal standards for cost-benefit assessments for public lands, USDA must withdraw the net present valuation (NPV) presented in Table 6 (RIA: 35) and incorporate the credible CBA presented at the end of this section in a revised DEIS.

4.1 The Cost-Benefit Assessment is Rudimentary and Wholly Inaccurate

Cost-benefit assessment is used to compare projected management scenarios such as the preferred Alternative to a baseline. In this case, the baseline is the no-action Alternative of keeping the Tongass Roadless Rule in place. The baseline comparison is similar to a with/without analysis. That is, what are the projected changes in costs and benefits if the preferred Alternative (6) is adopted?

Furthermore, it is important to understand the role of Executive Order 13463 and federal guidelines for cost-benefit assessment. Quantifying costs and benefits are emphasized, but if only a few costs and benefits can be quantified, it is still imperative to include "reasoned determination" that the benefits of regulatory actions justify the costs…. "Executive Order 13563 recognizes that a quantifiable analysis is not always possible, but must include a reasoned determination that the benefits justify the regulatory costs." (RIA: 22)

The question being asked at the beginning of the RIA should be "do the benefits justify the costs?" Instead of starting with an open question and working to a reasoned determination, it appears as if USDA worked backwards from a pre-determined answer regarding costs and benefits. There is no other way to account for the errors and lack of economic rigor in the presented CBA. In no way does the RIA and CBA pass scientific or legal muster.

4.1.1 Timber Harvesting Costs Will Increase in Roadless Areas, Not Decrease

Trendlines over time for three timber harvesting costs, out of many, are used to suggest that harvest costs will be reduced by exempting the Tongass from the Roadless Rule. This is the only quantified "benefit" in the CBA. In short, this estimate and the use of this estimate as the only quantified benefit in the CBA, does not constitute credible economic analysis.

For starters, the harvesting costs isolated (felling, yarding, and loading) are just three components of a number of total timber harvest costs and have little meaning when presented in isolation, or cherry picked as done in the DEIS and the RIA. USDA openly admits that the harvest costs presented in the CBA are one set of many harvest costs. For example:

> "In practice, many factors can influence the cost of timber harvest, adding economic risks for potential purchasers and affecting the ability of the Forest Service to offer timber sales. Road construction, helicopter yarding, complex silvicultural prescriptions, setting size, and other factors may increase costs, which then decrease the value of the offering." (RIA: 29).

How the USDA doesn't also include road construction, road decommissioning, and road maintenance costs in this CBA is bewildering, especially as they admit that timber road costs will increase in roadless

areas. Furthermore, road construction for timber sales are a cost to the USFS and the public taxpayer[17] whereas felling, yarding, and loading are typically costs covered by timber sale purchasers. USDA has extensive data on the need for more roads per unit of wood harvest in roadless areas, as well as extensive data on road costs. Why were these not included? Why were just a few harvesting costs cherry-picked?

Additionally, the point estimate of reduced harvest costs has no data to back it up and ultimately has nothing to do with the Roadless Rule. Changes in harvesting cost will not be the result of efficiency gains. The USDA's cost-benefit assessment (Table 6, RIA: 35) presents supposed reductions in harvest costs to the timber industry as benefits by comparing timber data for the eight years of the Roadless Rule exemption (2003-2010) to the eight most recent years with the Roadless Rule in place (2011-2018). A simple comparison of the years with and without the Roadless Rule is completely arbitrary and USDA has not made any case on why harvesting costs would change with and without the Roadless Rule.

Science involves developing a hypothesis first, then testing it. But USDA has provided no logical explanation of why harvest costs would be reduced in roadless areas---because they will not be reduced. Tongass timber harvesting costs have been continually increasing for decades, as high grading forces every subsequent timber sale farther up the watershed and towards less valuable wood (greater defect and smaller trees) and less accessible wood (i.e., steeper slopes). While shifting 165,000 acres of Roadless old growth to the suitable timber base may open a few low-elevation, high-volume stands for harvest,[18] overall harvest costs throughout roadless areas will increase due to increased road construction. Felling, yarding, and loading harvest costs are also likely to increase, rather than decrease, when estimating harvest costs across the entire 165,000 acres of old growth---as opposed to just estimating felling, yarding, and loading costs on the most productive 18,000 acres.[19]

Ultimately, USDA has confused the results of a short-term comparison of all timber sales for years before and after the Roadless Rule was back in place (2011), with efficiencies in harvesting costs due to Roadless designation. This is a critical error that undermines the entire CBA. USDA needs to withdraw the current DEIS and revise it to include long-term harvest cost trends dating back to at least the 1980s and adjusted for inflation. Long term harvesting cost trends would likely show continual increases in harvestings costs over time. USDA must also include full engineering and linear programming (e.g. Woodstock model analysis) reports estimating road construction needs, road costs, and other harvesting costs for Roadless old growth forests shifted to the suitable timber base.

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[17] Road construction needed for Tongass timber harvests are fully subsidized by the USFS. When timber sale appraisals are negative, the USFS will tend to pre-road to help sell the timber. Tongass timber managers are directed to reclassify roads as Public Works when dealing with negatively appraised sales (e.g., see R10 Timber Appraisal direction documents). In other sales, the USFS will give purchaser road credits to timber sale purchasers for the estimated cost of road construction. These purchaser road credits are then used for stumpage fee reductions on that sale or other future timber sales, resulting in losses in stumpage revenue to the U.S. Treasury and full subsidization of timber roads by U.S. taxpayers. The budget line item CMRD (Roads Capital Improvements and Maintenance) is used to pay for timber roads and averaged 50% of overall timber budgets from 2001-2008 and 40% of overall timber budgets from 1999-2018.

[18] USDA estimates that only 59,000 acres of the proposed 165,000 acres of Roadless old growth are high-volume.

[19] USDA claims that the projected harvest on the 165,000 acres of Roadless timber for the preferred alternative (6) will only be 18,000 acres over 100 years (DEIS: 3-19). If this is the case, there is no reason to choose Alternative 6 as the preferred alternative. If converting 165,000 acres of Roadless old growth to suitable for timber production, harvesting costs must be estimated for the potential harvest of all acres and must be presented in a revised DEIS.

4.1.2 USDA Has Mistaken Distributional Effects for Costs and Benefits

Another fatal flaw in the CBA is that USDA has only quantified a market impact to the timber industry and a market impact to the tourism industry, not a benefit or cost to the federal government or the public. That is, if there is a reduction of harvest costs (not likely), it will not save the federal government any money; rather, it will save the timber industry a few bucks. This also critically undermines the CBA.

At the beginning of the CBA, USDA states that:

> "Benefits and costs are divided into two parts: 1) those which are realized by any organization or individual, and 2) those realized by the Forest Service. Financial considerations include revenues and costs from the perspective of the Forest Service or other government agencies." (RIA: 22)

Upon review of the RIA, however, there are zero quantified costs and benefits presented for the Forest Service or for the taxpayer at large. Additionally, changes to timber industry harvest costs and costs to the tourism industry are not treated as benefits or costs to individuals or organizations when conducting economic efficiency analysis---they are market changes that are classified as distributional effects according to economic theory. Federal regulatory effects on public lands can have implications for specific industry sectors. While these should be examined, they should **not** be included in cost-benefit assessments. Market impacts are considered distributional effects by economists. Distributional effects do not go into the net present valuation (NPV) calculations---NPV is only for the CBA which is part of the "economic efficiency" analysis required in RIAs and DEISs. Some qualitative effects on individual regional industries are described in the Distributional Effects of the RIA (p. 39) where they belong. But using estimated distributional effects for the timber and tourism industries in the CBA is faulty economics and illustrates a lack of economic rigor in the DEIS and RIA.

The numerous footnotes associated with the damage estimates for the tourism industry in the DEIS and RIA help illustrate why industry effects are not considered as costs or benefits, but rather represent redistributions of wealth. This footnote in the RIA refers to the estimates of lost outfitter and guide revenues due to exempting the Tongass from the Roadless Rule:

> "These estimates provide an upper-bound ceiling for consideration of potential lost revenue, alongside cost savings to the timber industry, and should not be used as precise estimates of roadless area visitor expenditures or losses. Expenses incurred by visitors are not necessarily lost but subject to displacement related changes. While some businesses may lose revenues, if visitors choose not to travel to Southeast Alaska, others may see increases in revenues if visitors choose to stay longer or travel to substitute sites within Southeast Alaska." (RIA: 39)

"Displacement related changes" in markets is the very definition of distributional effects. The recreation industry losses are distributional effects, not a societal cost, even though they represent clear economic harm to the southeast Alaskan tourism industry. This is because visitors will spend their money elsewhere. Despite a lack of footnotes in the RIA explaining this same concept for the timber industry, it should be noted that decreases or increases in timber harvest costs paid for by timber companies are also distributional effects.

4.1.3 Cost-Benefit and Net Present Valuation Calculations Make No Sense

The poor economic analyses presented in the RIA is compounded by illogical Net Present Valuation (NPV) calculations and a lack of clarity. Table 6 in the RIA (p. 35) presents USDA's final CBA as

represented by 20-year discounted NPV estimates. Table 6 is difficult to understand for a few reasons. First, costs attributed to the recreation/tourism industry need to be shown as negative numbers, as opposed to currently being presented as positive. Second, a footnote tells the reader that OMB Regulatory Analysis requires the use of two discount rates (3% and 7%). But Table 6 presents NPV estimates under only one discount rate----but which one is not noted. Do the final NPV estimates represent valuation under a 3% discount rate or a 7% discount rate?

Finally, the NPV estimates appear to be wildly inaccurate. Even with incorrect theoretical assumptions for CBA (using distributional effects as costs and benefits), even with arbitrary inputs (incorrect interpretation of reduced harvest costs), the presented NPV estimates still cannot be replicated. If using industry harvest cost reductions of $1 million -- $2 million per year as a benefit in CBA, discounted over 20 years, how does this result in $91 million in NPV for the 46mmbf level and $30 million for the 24mmbf level? From the RIA:

> "Applying cost averages before and after the federal court decision in 2011 ($220 and $265 per MBF, respectively) indicates the proposed rule and Alternatives 2 through 5 could provide approximately $2 million dollars in annual savings at the harvest ceiling of 46 MMBF under the 2016 Forest Plan FEIS." (RIA: 31)

At the upper-bound harvest ceiling of 46 mmbf, harvesting costs reductions are said to be $2 million annually. Specifically, $45/mbf of savings multiplied by 46mmbf equals $2,070,000 of purported savings. To correctly estimate NPV over 20 years, the $2 million in annual savings (or $2.07 million) should be entered as positive cash flow for each of the 20 years. With no discount rate, the NPV would simply be the sum of all 20 years of cash flow, or $41.4 million. However, utilizing a discount rate to account for inflation and a preference for money today as opposed to next year, the $41.4 million NPV is reduced. At a 3% discount rate, the NPV for the 46mmbf level is $30.8 million. At 7% discount rate, the NPV for the 46mmbf level is $21.9 million. The Forest Products Industry—cost savings NPV estimate in Table 6 for the 46mmbf level is $91 million for all alternatives except the No-Action alternative. This estimate is more than three times the real estimate.[20]

The same issues are present for the lower-bound harvest cost NPVs, the Recreation/Tourism cost NPVs, and the final NPVs. USDA needs to clearly articulate how the NPV estimates were calculated. As of now, Table 6 appears to present wildly inaccurate NPV estimates throughout the entire table. Given the numerous problems in the RIA, none of the analysis is to be trusted. None of the analysis is scientifically or legally credible.

4.2 A Credible Regulatory Impact Assessment and Cost-Benefit Assessment

What should have been USDA's approach for assessing costs and benefits of the Tongass Roadless exemption as required in RIAs? That is, what would an economically credible cost-benefit assessment for the Tongass Roadless Rule look like? Below, I detail a scientifically appropriate CBA for the Tongass Roadless Rule to illustrate what is lacking in the DEIS and RIA, and to offer a blueprint for USDA to utilize in a revised DEIS.

[20] For example, at a 7% discount rate, it would require about $8.5 million in annual savings over 20 years to generate an NPV of $91 million. The USDA has attributed an extra $6.5 million in annual cost savings that do not exist, on top of $1-$2 million of annual harvest cost savings that also don't really exist.

If we are to assume that the preferred Alternative will not increase the overall annual Tongass harvest levels and the Projected Timber Sale Quantity (PTSQ), as stated numerous times in the DEIS, we still know that exempting the Tongass from the Roadless Rule will directly lead to timber harvest in current roadless areas (if there will be no harvest in roadless areas, then there is no need for this rulemaking). Expanding Tongass timber production into roadless areas, even if overall forest harvest levels remain the same, will spur a number of economic costs above and beyond the status quo Tongass timber harvest program. These costs include increased road and overall harvesting costs, decreased conservation values, and damages to the quantity and quality of ecosystem services being produced by intact Tongass roadless areas.

USDA has quantifiable secondary data on increased road costs in Tongass roadless areas. Tongass timber road costs, as opposed to felling, yarding, and loading, are primarily paid for by the USFS. Harvest costs of felling, yarding, and loading are the responsibility of the timber sale purchaser and represent costs to the timber industry, not to the USFS or the public. As discussed above, this means that any effects to the timber industry should be placed under the Distributional Effects section. But, changes in costs to the USFS, such as increased roading costs, are changes in societal wealth and need to be included in the cost-benefit assessment. In fact, agency roads for Tongass timber production are the biggest cost contributor for timber budgets, estimated to be 50% of overall agency timber costs from 2001-2008,[21] and over 40% of overall agency timber costs from 1999-2018.[22]

As discussed above, USDA presented no logical reasoning on why harvesting costs would decrease in roadless areas. Harvest costs, including road costs, on the Tongass steadily increase over time as timber sales are continually pushed higher into watersheds and into less economical timber (this includes roadless areas, as they would have already been harvested prior to the 2001 Roadless Rule if they compared to the productivity of stands that have been harvested since the 1950s). Examining harvest costs based only on a comparison of the eight years without the Roadless Rule (2003-2010) to the eight years with the Roadless Rule (2011-2018) is a poor methodology as these years do not actually correlate well with roaded/roadless logging or include all the relevant costs. Few roadless areas were actually harvested from 2003-2010. The USFS has long term data on harvests occurring in roadless areas and should have engineering reports and estimates on the amount and cost of road construction for the 165,000 acres of Roadless old growth that is shifted into the suitable timber base by the proposed rule.[23] Why has this data not been provided in the DEIS? The only logical conclusion is that USDA has not included this essential data because it illustrates that roadless areas will require much greater timber

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[21] Hjerpe, E. 2011. Seeing the Tongass for the Trees: The Economics of Transitioning to Sustainable Forest Management. Washington: The Wilderness Society, 61p. Available at https://www.researchgate.net/publication/301553259_The_Economics_of_a_Tongass_Transition

[22] Taxpayers for Common Sense. 2019. Cutting Our Losses: 20 Years of Money-Losing Timber Sales in the Tongass. Available at https://www.taxpayer.net/wp-content/uploads/2019/09/TCS-Cutting-Our-Losses-2019-.pdf

[23] For example, forest planning analysis in the 2016 TLMP Amendment estimates that one mile of new road construction will be needed for every 150 acres of old growth harvest (2016 TLMP EIS: B-27). Extrapolated to the 165,000 acres of old growth that will become suitable under the proposed rule, the preferred alternative may lead to the construction of 1,100 miles of new road, requiring a quarter billion dollars of taxpayer funds. USDA claims that the projected harvest on the 165,000 acres of Roadless timber for the preferred alternative (6) will only be 18,000 acres over 100 years (DEIS: 3-19) ---which would still lead to 120 miles of new road. This leads to the question of why Alternative 6 was chosen as the preferred alternative if only 11% of the Roadless old growth acres will be harvested. Thus, USDA's claims are out of alignment and are suspect.

roads, and thus much greater road construction costs, as compared to harvests in the roaded timber base.

With most of the essential data missing from the DEIS, I use the only data provided by USDA on harvesting costs.[24] These data were received only upon request, as the provided reference in the DEIS was a dead end. Harvesting cost data used by USDA have clear findings of inefficiencies in overall timber harvesting costs----specifically increased timber road costs in Tongass roadless areas. The data reveals that the Roadless exemption period (2003-2010) required, on average, three times the miles of new road construction for timber sales as compared to the period when the Roadless Rule was back in place (2011-2018). An annual average of 15.3 miles of new roads were built during Roadless exemption years, but only 5.0 miles of new road were annually constructed with the Roadless Rule in place despite only slightly greater acres harvested in Roadless exemption years (an annual average of 1,700 acres harvested vs 1,400 acres harvested per year from 2011-2018). Clearly, extending timber sales into Tongass Roadless will require many more miles of road as compared to keeping timber sales out of Roadless.

Furthermore, the timber sale information before and after the Roadless Rule change in 2011 show that for every million board feet (mmbf) of Tongass timber harvested during the Roadless Rule exemption, twice as many miles of new roads were constructed as were for every million board feet harvested with the Roadless Rule in place. From 2003-2010, .42 miles of new road were constructed for every million board feet harvested. From 2011-2018, only .21 miles of new road were required for every million board feet harvested, indicating that harvest in Tongass roadless areas will require, on average, .21 more miles of new road construction for each million board feet harvested. While overall Tongass harvest levels may stay the same under the current Roadless rulemaking, overall agency road costs are bound to increase, decreasing Tongass timber production efficiency and increasing federal subsidies. And, as stated earlier, the 2003-2010 to 2011-2018 roadless rule dichotomy is a poor estimate of harvest costs, and the actual increase in roads and road costs for Roadless timber are likely much greater than the data used in this analysis.

While there is variance in Tongass road costs, especially in regard to slope angle, average forest-wide road construction costs were estimated at $185,000 per mile and $50,000 per mile for maintenance in the 2008 TLMP.[25] Adjusting for inflation reveals that current Tongass road construction costs are approximately $225,000 per mile. By incorporating projected timber sale incursions into roadless areas and the increase in average road construction costs, a credible cost-benefit assessment can be conducted. Because the Tongass timber sale program loses substantial amounts of money (i.e., stumpage fees are a fraction of agency timber costs) and damages all other resources and industries, there are no benefits to include in a cost-benefit assessment of exempting the Tongass from the Roadless Rule.

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[24] The USDA reference for Tongass timber harvesting costs presented in the RIA (e.g., p. 29) is presented as "USDA Forest Service 2019b. Timber Sale Summary Reports and Accomplishments, Region 10 RV (Residual Value) Appraisals of Record (1+ MMBF, from 2003 to 2018) available at: https://www.fs.usda.gov/detail/r10/landmanagement/resourcemanagement/." No document with the referenced data was available at the listed website. Upon requesting the source for harvesting cost information, USDA released an Excel spreadsheet titled "Region 10 timber sales before and after 2011 Roadless ip 112219."

[25] 2008 FEIS, TLMP Amendment, Vol. II, App. B at B-11.

While the rulemaking does not authorize site-specific activities, it does authorize a shifting of Tongass timber sales to roadless areas. To provide a general overview of costs and benefits of increasing Tongass timber sales in roadless areas, I examine three potential outcomes: small, medium, and large timber sale incursions into roadless areas.[26] Utilizing the same upper and lower bound of anticipated Tongass timber harvest as presented in USDA's cost-benefit assessment (Table 6 of the RIA, p. 35), I conducted a net present valuation of shifting 25% (small), 50% (medium), and 75% (large) of projected timber sales into roadless areas. Table 1 shows the harvest scenarios and the annual timber road cost increases associated with each scenario.

Table 1: Roadless Timber Incursion Scenarios for Tongass Roadless Rule Cost-Benefit Assessment

Type of Roadless Timber Incursion	Harvest Scenario	MMBF Harvested in Roadless Areas	Additional Roads /mmbf	Addtional Miles of New Road Needed	Road Costs/mile	Annual Timber Road Cost Increase
	Upper-bound--46MMBF harvest ceiling					
Small	25% of timber sales shifted to Roadless	11.5	0.21	2.42	$ 225,000	$ 543,375
Medium	50% of timber sales shifted to Roadless	23	0.21	4.83	$ 225,000	$ 1,086,750
Large	75% of timber sales shifted to Roadless	34.5	0.21	7.25	$ 225,000	$ 1,630,125
	Lower-bound--24MMBF					
Small	25% of timber sales shifted to Roadless	6	0.21	1.26	$ 225,000	$ 283,500
Medium	50% of timber sales shifted to Roadless	12	0.21	2.52	$ 225,000	$ 567,000
Large	75% of timber sales shifted to Roadless	18	0.21	3.78	$ 225,000	$ 850,500

With a range of Roadless harvest scenarios, sensitivity analysis can be conducted to provide a realistic range of road construction cost increases expected under the proposed rule. Table 2 illustrates the NPV for road construction cost increases under two discount rates. These NPVs show tremendous costs over the next 20 years, losses ranging from $3 million to $24 million.

[26] The Roadless timber sale incursion scenarios utilize the USFS-provided data on harvesting and road needs that are based on the eight years with no Roadless Rule (2003-2010) compared to the eight years with the Roadless Rule (2011-2018). As discussed in the text, this short-term comparison of the road/roadless dichotomy is a poor substitute for data focused strictly on timber sales and harvest in roadless areas. The USFS data also only include estimates of new road miles needed per mmbf of harvest. Yet, numerous miles of timber roads have been constructed for sales that go unsold and for large portions of timber sales that are regularly defaulted on. The result is that the presented increase in annual road construction and road costs in this section is likely to be vastly underestimated.

Table 2: Net Present Valuation of 20-Year Discounted Agency Timber Road Cost Increases for Proposed Rule (Alternative 6)[1]

Full Exemption Alternative Scenarios for Proposed Rule		Increased Annual Agency Road Cost	NPV @ 3%	NPV @ 7%
Upper-bound--46MMBF harvest ceiling				
Small	25% of timber sales shifted to Roadless	($543,375)	($8,084,000)	($5,757,000)
Medium	50% of timber sales shifted to Roadless	($1,086,750)	($16,168,000)	($11,513,000)
Large	75% of timber sales shifted to Roadless	($1,630,125)	($24,252,000)	($17,270,000)
Lower-bound--24MMBF				
Small	25% of timber sales shifted to Roadless	($283,500)	($4,218,000)	($3,003,000)
Medium	50% of timber sales shifted to Roadless	($567,000)	($8,436,000)	($6,007,000)
Large	75% of timber sales shifted to Roadless	($850,500)	($12,653,000)	($9,010,000)

[1]OMB Circular A-4 - Regulatory Analysis (Sep 17, 2003) requires use of two discount rates (both 3 and 7 percent).

Economic efficiency analysis, or cost-benefit assessment, on public lands must also include the opportunity costs, or benefits foregone, of choosing a preferred land management alternative.[27] That is, what benefits will be foregone by removing Tongass Roadless protections? In the case of the proposed full exemption, the greatest loss of benefits occur to societal conservation values held for pristine and protected Tongass forests. Conservation values are comprised of both use and passive use values held for intact Tongass old growth forests such as those contained in roadless areas. Combined, use and passive use values are known as Total Economic Value (TEV). Use values include direct and indirect use values, such as consumer surplus for recreation and benefits received from ecosystem services produced by roadless areas such as clean drinking water. Passive use values include option, bequest, and existence values held for Tongass roadless areas. Because conservation values are largely comprised of non-market values, they are not as easy to quantify as board feet of timber harvested and typically require direct survey techniques focused on willingness to pay for conservation.

Agencies are traditionally forced to use existing secondary data for analysis as the costs and time for gathering primary data for all regulatory actions is prohibitive. Many national forests may not have existing secondary data on willingness to pay values for forest conservation, to use for quantifying opportunity costs in cost-benefit assessment of development activities. Fortunately, the Tongass does. Recent peer-reviewed research[28] describes a choice experiment quantifying regional Alaska residents' willingness to pay for conserving Tongass old growth forests as opposed to harvesting them. Econometric analysis shows that Alaska residents are willing to pay $150 per acre for the conservation, not harvest, of Tongass old growth in the suitable timber base.[29] As the preferred Alternative (6) shifts

[27] See for example, Freeman, A. M. (2003). The Measurement of Environmental and Resource Values. *Resources for the Future. Washington DC,* p.202; and Hjerpe, E. E., & Hussain, A. (2016). Willingness to pay for ecosystem conservation in Alaska's Tongass National Forest: a choice modeling study. *Ecology and Society*, *21*(2).
[28] Hjerpe, E. E., & Hussain, A. (2016). Willingness to pay for ecosystem conservation in Alaska's Tongass National Forest: a choice modeling study. *Ecology and Society*, *21*(2). Available at https://www.ecologyandsociety.org/vol21/iss2/art8/
[29] *Ibid.*

165,000 acres of Roadless old growth to suitable for timber production, a total conservation benefit value of $24,750,000 (165,000 acres X $150/acre) is lost. These foregone benefits are a one-time societal cost, regardless of how many acres are eventually harvested. Just knowing that these once pristine and roadless areas are open to timber development and have lost their protective status results in the substantial losses in ecosystem conservation value.

Adding in lost conservation benefits[30] as opportunity costs associated with the proposed rule allows for a complete cost-benefit assessment to be estimated, one that appropriately accords with economic theory. Table 3 shows a full cost-benefit assessment of the proposed rule under two discount rates and under six Roadless timber incursion scenarios.

Table 3: Net Present Valuation of 20-Year Discounted Costs and Benefits for Proposed Rule (Alternative 6) Under Six Roadless Incursion Scenarios[1,2,3]

Full Exemption Alternative Scenarios for Proposed Rule		Increased Annual Agency Road Cost	Lost Conservation Benefits*	NPV @ 3%	NPV @ 7%
Upper-bound--46MMBF harvest ceiling					
Small	25% of timber shifted to Roadless	($543,375)	($24,750,000)	($32,113,000)	($28,887,000)
Medium	50% of timber shifted to Roadless	($1,086,750)	($24,750,000)	($40,197,000)	($34,644,000)
Large	75% of timber shifted to Roadless	($1,630,125)	($24,750,000)	($48,281,000)	($40,400,000)
	Lower-bound--24MMBF				
Small	25% of timber shifted to Roadless	($283,500)	($24,750,000)	($28,247,000)	($26,134,000)
Medium	50% of timber shifted to Roadless	($567,000)	($24,750,000)	($32,465,000)	($29,138,000)
Large	75% of timber shifted to Roadless	($850,500)	($24,750,000)	($36,682,000)	($32,141,000)

*Loss of conservation benefits are entered as a one-time cost in Year 1 only.
[1]The baseline for comparison is a continuation of the 2001 Roadless Rule (No-Action Alternative). The No-Action Alternative would produce zero increased costs and has an NPV of $0, a substantially higher NPV than the preferred Alternative.
[2]As there are no economic benefits for exempting the Tongass from the Roadless Rule, the cost-benefit analysis is comprised only of costs and all scenarios result in negative NPV estimates.
[3]OMB Circular A-4 - Regulatory Analysis (Sep 17, 2003) requires use of two discount rates (both 3 and 7 percent).

[30] WTP estimates in Hjerpe and Hussain (2016) are comprised of both use and passive use values. Choice experiment respondents were provided information on environmental damages from Tongass timber production, along with tradeoffs such as providing for timber employment. The average WTP for conserving Tongass old growth includes the societal value for passive use, such as bequest values for knowing that our children will have pristine old growth forests. It also includes the value society holds for avoiding the associated environmental damage that comes from clearcutting old growth. This value includes direct and indirect use values such as damages to subsistence ability (direct use value) and damages to carbon sequestration that affects climate change (indirect use value). The overall WTP is a value known in economics as Total Economic Value (TEV) and is currently the best way to capture societal value held for avoiding environmental damages and foregone conservation benefits when protected areas lose their protective designation.

The cost-benefit assessment shows that all scenarios result in negative NPV, losses ranging from $26 million to $48 million. But the reality is that adoption of the proposed rule will result in much greater losses than illustrated in Table 3. The presented cost-benefit assessment is conservative in every analyzed cost. Not included in this assessment are the additional road decommissioning costs that will be spurred by greater road construction in roadless areas along with greater road maintenance costs. If nearly all of the newly constructed timber roads in roadless areas will be decommissioned, as stated in the DEIS,[31] then the additional roads needed for timber harvest in roadless areas will also result in additional road decommissioning needs. Roads that are not decommissioned will require maintenance. Road maintenance and decommissioning will likely increase overall road costs used in this analysis by 50%. It is strongly recommended that USDA incorporate increased road decommissioning costs in a revised DEIS.

Administrative costs associated with increased road building in roadless areas are estimated to be 40% of overall timber program costs.[32] Additional road construction needs in roadless areas will lead to much greater USFS administration, contracts, and site development needs. Instead of working on restoration projects and land management activities that benefit fish and wildlife, more USFS employees will be needed to administer road construction in roadless areas. In the revised DEIS, USDA should include additional taxpayer costs in indirect and overhead expenses associated with additional road construction that will be spurred by the proposed rule.

Mean willingness to pay estimates used for determining lost conservation benefits were only extrapolated to Alaskan residents[33] and are also very conservative estimates. Given the importance and uniqueness of the Tongass as spectacular public lands, we know that old growth conservation values for the Tongass extend to some degree throughout the rest of the U.S. Given its vast carbon stores, Tongass roadless areas are valued for their conservation throughout the entire world. Research[34] has shown that willingness to pay values, especially for the conservation of iconic and scarce landscapes such as coastal temperate rainforests, extend thousands of miles from the valuation site. USDA should model lost Tongass conservation benefits for the entire U.S. in the revised DEIS.

Finally, other opportunity costs of a Tongass Roadless Rule exemption are omitted in the RIA and DEIS. Increased agency costs from the proposed rule could be used for other, more sustainable, Tongass opportunities instead. For example, if it is "jobs-in-the-woods" that USDA and the State of Alaska are seeking, the focus should not be on developing Tongass roadless areas. The focus and subsidies should

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[31] Page 43 of the RIA… "Nearly all new roads constructed under the regulatory alternatives would be closed following harvest."

[32] Hjerpe, E. 2011. Seeing the Tongass for the Trees: The Economics of Transitioning to Sustainable Forest Management. Washington: The Wilderness Society, 61p. Available at https://www.researchgate.net/publication/301553259_The_Economics_of_a_Tongass_Transition

[33] See Hjerpe, E. E., & Hussain, A. (2016). Willingness to pay for ecosystem conservation in Alaska's Tongass National Forest: a choice modeling study. *Ecology and Society*, *21*(2).

[34] For dam removal and salmon WTP, Loomis (1996) found that the rest of the U.S. households reflected 97% of the benefits. For protecting California old growth forests from fire, Loomis and Gonzalez-Caban (1996) found that nonresidents WTP declined by only 1% for each 1000-mile increase. See: Loomis, J. B. 1996. How large is the extent of the market for public goods: evidence from a nationwide contingent valuation survey? *Applied Economics* 28:779-782 and Loomis, J. B., and A. Gonzalez-Caban. 1996. The importance of the market area determination for estimating aggregate benefits of public goods: testing differences in resident and nonresident willingness to pay. *Agricultural and Resource Economics Review* 25:161-169.

be directed at vastly increasing Tongass recreation and restoration budgets. The Forest Service estimates that there are 500 miles of Tongass fish streams in need of in-channel restoration, along with 15,000 acres of riparian second growth in need of thinning (USFS 2006).[35] There are also 2,300 miles of closed roads on the Tongass, over 500 miles of which represent opportunities for improving water quality and fish habitat (USFS 2006).[36] USDA must focus on community stability and job creation that *help* maintain and restore environmental functions, as opposed to coming at the *cost* of the environment.

4.3 Cost-Benefit Assessment of the Tongass Timber Program

In addition to the cost-benefit of the proposed rulemaking that must be presented in the RIA, USDA should also include overall cost-benefit assessments of increasing Tongass logging. That is, because the proposed rule without limitation on future plan amendments opens the door to substantially greater roadless intrusions over time, the RIA needs to consider the potential costs and benefits of longer range and broader scale old growth logging and road construction than it currently does. An overview of costs and benefits for the Tongass timber program, above and beyond the Roadless rulemaking, would properly frame the significant taxpayer losses associated with any rulemaking aimed at maintaining or increasing Tongass timber harvests. Illustrating the huge subsidies required to produce Tongass timber would more clearly demonstrate that any federal policy that will boost the timber industry will exacerbate existing economic inefficiencies (i.e., increase the costs to benefits ratio).

From a societal perspective, timber production on federal lands have costs associated with preparing timber sales and lost conservation values. Correlating benefits are associated with revenue, or stumpage fees paid by private corporations to the USFS for access to publicly owned timber. In the last decade, there have been four studies that have quantified overall costs and benefits of the Tongass timber program. Interestingly, only one of these studies is acknowledged in the DEIS and RIA. To fill the gap in the Tongass timber program cost-benefit literature, I present the details of the four studies in Table 4.

Table 4: Research on Cost-Benefit Ratios for Tongass Timber Program

Source	Years	Costs	Benefits (Revenue)	Cost-Benefit Ratio	Notes
Taxpayers for Common Sense (2019)	1999-2018	$632 million	$33.8 million	18.7	Includes road costs
Hjerpe and Hussain (2016)	2008, 2012	$108.5/mbf	$7.12/mbf	15.2	Does **not** include road costs; includes lost conservation benefits
GAO (2016)	2005-2014	$12.5 million/year	$1.1 million/year	11.4	Does **not** include road costs
Hjerpe (2011)	2001-2008	$255 million	$7 million	36.4	Includes road costs

Sources: Taxpayers for Common Sense. (2019). Cutting Our Losses: 20 Years of Money-Losing Timber Sales in the Tongass; Hjerpe, E. E., & Hussain, A. (2016). Willingness to pay for ecosystem conservation in Alaska's Tongass National Forest: a choice modeling study. *Ecology and Society*, *21*(2); USGAO. (2016). Tongass National Forest: Forest Service's Actions Related to Its Planned Timber Program Transition. GAO-16-456; Hjerpe, E. (2011). Seeing the Tongass for the Trees: The Economics of Transitioning to Sustainable Forest Management. Washington: The Wilderness Society, 61p.

[35] USFS. 2006. Investing in habitat improvements vital for ecological sustainability, local economies, subsistence users. Alaska Region Newsletter, June 2006. 2p.
[36] *Ibid.*

From 1999 to 2018, USFS spending on roads in the Tongass made up more than 40% of all timber sale expenses.[37] To compare similar cost-benefit ratios, I update two of the estimates that previously did not include road costs by adding in road costs at 40% of total costs. Table 5 illustrates complete cost-benefit ratios for the Tongass.

Table 5: Research on Full Cost-Benefit Ratios for Tongass Timber Program

Source	Years	Total Costs	Total Benefits (Revenue)	Cost-Benefit Ratio
Taxpayers for Common Sense (2019)	1999-2018	$632 million	$33.8 million	**18.7**
Hjerpe and Hussain (2016)	2008, 2012	$181/mbf	$7.12/mbf	**25.4**
GAO (2016)	2005-2014	$20.8 million/year	$1.1 million/year	**18.9**
Hjerpe (2011)	2001-2008	$255 million	$7 million	**36.4**

Throughout different periods over the last 20 years, the Tongass timber program has a total cost-benefit ratio ranging from 18.7—36.4, with an average cost-benefit ratio of 25. That is, on average, for every $1,000,000 received by the U.S. Treasury in Tongass timber stumpage fees, U.S. taxpayers pay $25,000,000 in federal agency costs to subsidize timber harvest. It is important to note that these timber program costs do not include associated indirect and overhead expenses which were estimated at 40% of total costs for Tongass timber from 2001—2008.[38]

With costs exceeding benefits by 25, and only 61 total Tongass timber jobs supported by millions in taxpayer dollars, the Tongass timber program makes zero economic sense. Over the years, the federal government loses billions of dollars while causing substantial ecological damage. The original Roadless Rule was put in place to eliminate this exact government waste and to avoid this exact ecological destruction. Any federal rulemaking related to Tongass timber production, and specifically the current proposed Roadless exemption, should start with economic facts that clearly show that any attempts to maintain or increase Tongass timber production will only create greater societal losses. Only when dealing with the economic facts can a reasonable determination be made that exempting the Tongass from the Roadless Rule is bad business.

4.4 Agency Costs and Control of Regulatory Costs

In the RIA section "Agency Costs including Control of Regulatory Costs" (RIA: 37), USDA provides no evidence that agency costs will be reduced. USDA conflates incorrectly presumed timber industry savings in reduced harvest costs to reduced agency costs. This is incorrect. The economic reality is that

[37] Taxpayers for Common Sense. 2019. Cutting Our Losses: 20 Years of Money-Losing Timber Sales in the Tongass. Available at Available at https://www.taxpayer.net/wp-content/uploads/2019/09/TCS-Cutting-Our-Losses-2019-.pdf

[38] Hjerpe, E. 2011. Seeing the Tongass for the Trees: The Economics of Transitioning to Sustainable Forest Management. Washington: The Wilderness Society, 61p. Available at https://www.researchgate.net/publication/301553259_The_Economics_of_a_Tongass_Transition

agency costs will sharply increase under the proposed rule, especially for road construction, road decommissioning, and road maintenance.

The Introduction section of the RIA states:

> "If costs from potential displacement of recreationists accrued they would occur alongside cost reduction from more acres of land available for timber harvest. Timber harvest levels on the Tongass NF are set by the 2016 Forest Plan (USDA Forest Service 2016) and continual timber demand monitoring, currently 46 million board feet (MMBF). The propose rule (Alternatives 6) would increase flexibility for timber managers for designing timber sales that appraise positive." (RIA: 6)

How would agency "cost reductions" occur by making more acres of land available for timber harvest? This is nonsensical. Expanding the Tongass woodshed will increase costs and will certainly **not** lead to cost reductions.

The section continues:

> "Cost savings from improved flexibility could, in turn, potentially improve the Forest Service's ability to offer economic sales that meet the needs of industry. Areas closer to markets, either a mill or export facility, are also more likely to offer more economic timber sale options. More distant areas would be relatively expensive to harvest and less likely to be accessed." (RIA: 6)

Cost savings for the agency are never divulged in the RIA or DEIS. What are these costs savings and how would they occur? By the USDA's own admission, agency costs for timber production would substantially increase in roadless areas. Purported harvest costs reductions accrue to the timber industry only. These are not savings for the agency, nor public taxpayer savings. This statement is wholly inaccurate and is simply is not supported in the RIA. This appears to be an attempt to mislead the public and an attempt to satisfy Executive Order 13771. Contrary to this concluding statement, the CBA and RIA conducted by USDA fails the most basic tests for economic rigor and have certainly not illustrated a maximization of net benefits, nor that benefits would outweigh the extremely expensive costs of exempting the Tongass from the Roadless Rule.

The concluding paragraph in the Agency Costs section states:

> "Cost savings from improved flexibility for the agency and timber industry would accrue alongside other benefits, displayed in Table 5 and discussed above; including reduced cost for leasable mineral availability, renewable energy development potential, potential for development of state roads and other transportation projects, and benefits to Alaska native customary and traditional uses." (RIA: 45).

No essential energy or transportation projects have been stopped by the Tongass Roadless Rule. Other purported benefits are also red herrings. As stated above, these "cost savings" are never divulged because they won't actually occur. Expanding the Tongass timber footprint into roadless areas will universally increase all agency costs. Other presumed benefits in this statement are questionable to say the least. It is difficult to even understand what is meant by "reduced cost for leasable mineral availability?" Regardless, the Environmental Effects analysis shows that there are no leasable minerals on the Tongasss (DEIS: 3-155). So how will non-existent costs be reduced?

As shown in the previous section, the agency costs for Tongass timber are already out of control and will escalate under the preferred Alternative (6). Alaska USFS already has a road management backlog estimated at $68 million.[39] Part of the annual Tongass timber agency costs, and miniscule benefits, are acknowledged in the RIA:

> "On average, the Forest Service spent approximately $12.5 million per year to administer Tongass timber sales from 2005-2014, excluding road building costs, and received approximately $1.1 million in revenue per year (GAO 2016)." (RIA: 38)

However, none of the increased road costs that will occur under the proposed rule are presented in the Agency Costs section. At a minimum, per mile road construction, decommissioning, and maintenance costs for the Tongass must be presented in a revised DEIS. Furthermore, USDA must address current road maintenance backlogs in the Tongass, necessary culvert replacements, and watershed restoration needs and estimate current costs. Also not reported in the Agency Costs section, is the estimated $5 million dollar price tag for conducting this rulemaking, much of which has gone to the State of Alaska and to the Alaska timber industry lobby group (Alaska Forest Association).[40] With no rational purpose or need, and without verifying any of the state's claims of economic harm, this rulemaking should have never been initiated and is a large waste of taxpayer dollars. Excessive agency spending will be required if the Tongass is exempted from the Roadless rule. How are these myriad costs not acknowledged in the Agency Cost section?

4.5 The Distributional Effects Show the Preferred Rule will Have Zero Impact on Regional Employment

If the proposed rule is "intended to provide for economic development opportunities in Southeast Alaska" (RIA: 45), then surely the DEIS and RIA would contain economic analysis showing how the rule would increase economic activity and increase regional employment. However, the RIA and the Distributional Effects section does not project any increased economic activity or employment from the proposed rule. This is the case even for the timber industry: "Thus no change in timber related employment or income is expected as a result of the proposed rule or other regulatory alternatives." (RIA: 40)

The primary component of distributional effects used in the NEPA process for public lands rulemaking is economic impact analysis. Economic impact analysis, also known as economic contribution analysis, measures the resulting market impacts associated with a change in regional final demand resulting from a changed land policy. Economic impacts are part of distributional effects because they represent shifts in regional wealth. This shift in final demand results in distributional effects that have a greater impact on industries favored by the rulemaking.

That the Distributional Effects section of the RIA shows zero changes in market impacts or regional employment is clear evidence that the entire purpose and need for this rulemaking is not legitimate. Rhetoric and propaganda from Alaska politicians and the current Administration hold no water. For example, Alaska Governor Michael Dunleavy stated on November 20th that, "Exempting the Tongass from the Roadless Rule will create new jobs and economic activity in a region hit hard by the misguided

[39] USFS responses to Rep. Mike Quigley.
[40] "Congressional Democrats ask for investigation into Alaska use of forest grant." Alaska Daily News, 12/1/19.

policies of a previous administration."[41] Apparently Gov. Dunleavy and his staff have not read the DEIS and the Distributional Effects section in the RIA, where it is clearly divulged that there will be NO new jobs and NO new economic activity spurred by the proposed rule.

5. An Ecosystem Service Valuation for Tongass Roadless Areas

When viewed through an ecosystem services lens, it becomes abundantly clear that exempting the Tongass from the Roadless Rule will damage all other aspects of the Southeast Alaska economy in order to prop up the timber industry. Ecosystem services broadly represent nature's benefits to humans and can be classified into regulating, supporting, provisioning, and cultural. Expanding the Tongass timber harvesting woodshed by increasing the suitable timber base will have adverse environmental consequences on a bevy of ecosystem services currently protected by the Roadless Rule. Because the DEIS does not include an ecosystem service perspective and has failed to even qualitatively describe the full environmental consequences from removing the Tongass Roadless Rule, despite explicit flagging of these economic issues in the scoping process, the DEIS must be withdrawn and a new one must be produced.

Even if one were to accept USDA's insistence that Tongass harvest levels will not increase with the removal of the Roadless Rule, as stated numerous times in the DEIS, the Tongass timber footprint will greatly expand. The habitat fragmentation, sediment alterations, and stream damage from new incursions into Tongass roadless areas will come at a steep price. Unfortunately, secondary data is often missing for quantifying natural resource damages incurred due to expanding the timber footprint into Tongass roadless areas. These resource damages stemming from timber development include reduced water quality, loss of wildlife habitat, and increased carbon emissions. Despite not having easily transferable quantified economic values for damages to ecosystem services, adverse effects from timber production need to be included in a revised DEIS.

A review of the scientific literature paints a very clear picture: Tongass roadless forests provide much greater economic value than the logged over forests. Developing roadless forests for clearcutting of Tongass old growth has been shown to have adverse effects on critical regulating and supporting ecosystem services by increasing erosion[42] and sedimentation of salmon streams.[43] Tongass timber harvests alter hydrologic processes through erosion[44] and reduce large woody debris recruitment to streams resulting in degraded salmon habitat.[45] Road construction needed to access timber also limits

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[41] *Ibid*.

[42] Kahklen, K and W. Hartsog. 1998. Results of road erosion studies on the Tongass National Forest. Unpublished report for USDA Forest Service, Juneau Forestry Sciences Laboratory. 47p.

[43] Tiegs, S., D. Chaloner, P. Levi, J. Ruegg, J. Tank, and G. Lambert. 2008. Timber harvest transforms ecological roles of salmon in southeast Alaska rainforest streams. Ecological Applications 18(1): 4-11.

[44] Gomi, T, R. Sidle, and D. Swanston. 2004. Hydrogeomorphic linkages of sediment transport in headwater streams, Maybeso Experimental Forest, southeast Alaska. Hydrological Processes 18: 667-683.

[45] Heifetz, J., M. Murphy, and K. Koski. 1986. Effects of logging on winter habitat of juvenile salmonids in Alaskan streams. North American Journal of Fisheries Management 6(1): 52-58.

fish passage due to perched culverts.[46] Edges of Tongass clearcuts lose wind firmness, increasing blow down near harvest sites and leading to unraveling of stream buffers.[47]

These adverse effects on regulating and supporting Tongass ecosystem services associated with opening up roadless forests for development will have cascading effects on all biodiversity, particularly salmon. Ultimately, developing Tongass roadless areas will result in fewer salmon for recreational fishing, guided sport fishing, subsistence fishing, and commercial fisheries. This will, in turn, negatively affect economic activities and employment in industries that are much more important to Southeast Alaska than the timber industry. These are the economic trade-offs that must be analyzed and acknowledged in a revised DEIS.

The damage to streams and rivers is just part of the ecosystem service degradation legacy left by Tongass logging. Clearcutting Tongass old growth is also very problematic for forest structure and wildlife habitat. Due to extended decades of stem exclusion phases after clearcut regeneration, Tongass second growth becomes a liability to wildlife dependent on understory forbs and plants. This is particularly problematic for Sitka black-tailed deer,[48] but has cascading adverse effects on wolves and biological regulation functions of the forest.[49] Overstory bird species, such as goshawks and murrelets, also face declining habitat whenever Tongass old growth is clearcut.[50]

Tongass roadless forests represent vast carbon reservoirs. If these forests are opened to timber harvest, carbon will be released contributing to increased climate change. Since Tongass old growth forests have been estimated to contain about eight percent of the coterminous U.S.' carbon stocks[51] and some of the last old growth temperate rainforest in the world,[52] the Tongass also holds tremendous global value and is a critical component in helping mitigate climate change. Using international carbon markets,

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[46] Dunlap, R. 1997. Summary of the 1997 Fish Habitat Risk Assessment Panel, Tongass National Forest, Juneau, Alaska. May 7, 1997.

[47] Harris, A. 1999. Wind in the forests of southeast Alaska and guides for reducing damage. Gen. Tech. Rep. PNW-GTR-244. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 63 p.

[48] Schoen, J., M. Kirchhoff, and M. Thomas. 1985. Seasonal distribution and habitat use by sitka blacktailed deer in southeastern Alaska. Division of Game, Alaska Department of Fish and Game, Juneau, AK; Mazza, R. 2003. Hunter demand for deer on Prince of Wales Island, Alaska: an analysis of influencing factors. Gen. Tech. Rep. PNW-GTR-581. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 21 p.

[49] Person, D., C. Darimont, P. Paquet, and R. Bowyer. 2001. Succesion debt: effects of clear-cut logging on wolf-deer predator-prey dynamics in coastal British Columbia and Southeast Alaska. Paper presented at Canid Biology and Conservation: An International Conference. Oxford University.

[50] Flatten, C., K. Titus, and R. Lowell. 2001. Northern goshawk population monitoring, population ecology and diet on the Tongass National Forest. Alaska Department of Fish and Game. Juneau, AK; Cotter, P. and M. Kirchoff. 2007. Marbled Murrelet. In J. Schoen and E. Dovichin, eds. 2007. The coastal forests and mountain ecoregion of southeastern Alaska and the Tongass National Forest. Audubon Alaska and The Nature Conservancy, Anchorage, Alaska.

[51] Leighty, W., S. Hamburg, and J. Caouette. 2006. Effects of management on carbon sequestration in forest biomass in southeast Alaska. Ecosystems 9: 1051-1065.

[52] DellaSala, D. A., Moola, F., Alaback, P., Paquet, P. C., Schoen, J. W., & Noss, R. F. (2011). Temperate and boreal rainforests of the Pacific Coast of North America. In *Temperate and boreal rainforests of the world: ecology and conservation* (pp. 42-81). Island Press, Washington, DC.

researchers have estimated a market value of $3 to $7 million a year for stopping Tongass old growth harvesting.[53]

5.1 Ecosystem Service Damage from Roadless Timber Development Adversely Impacts the Regional Economy

Resource damage from pursuing Tongass timber production in roadless areas manifests in various economic forms and leaves a natural capital debt for future generations. Reduced black-tailed deer populations and reduced salmon have a direct economic effect on Southeast Alaska's largest private sector industries of tourism and seafood production. Subsistence activities, which comprise a large share of many Alaska residents' annual food budgets, are also degraded by reducing the number of animals for harvest and increasing the time and resources needed to fill the freezer. These resource damages cause economic harm to residents. The natural resource damages must be accounted for in the rulemaking process and must be countered with mitigation efforts.

For example, the latest economic impacts assessment for Tongass recreation shows that recreational visits are increasing and are at almost three million visits per year.[54] Visitors spend money in Tongass gateway communities for transportation services, food, gear, and lodging. In total, Tongass visitor expenditures are estimated at about $400 million annually, resulting in over $100 million in personal income for Southeast Alaska residents. Tongass recreational expenditures support approximately 4,000 direct local jobs, and over 5,000 jobs when including multiplier effects.[55]

The primary appeal for recreating on the Tongass is to enjoy its wildness and ecologically intact attributes that result in abundant native fish and wildlife. Sport fishing and hunting adventures in Southeast Alaska are considered to be "bucket list" trips for avid fisherman and hunters. Wildlife viewing and the ability to view unspoiled old growth forests, glaciers, and mountains spur numerous cruise ship passengers and do-it-yourselfers to pay substantial money to visit the Tongass.

The pristine nature of the Tongass *is* its comparative economic advantage for attracting tourists and recreationists from the Lower 48. If Tongass roadless areas are opened up for greater road building and clearcutting, this comparative advantage is decreased and will diminish the attractiveness of the Tongass for recreation. Are a handful of new subsidized logging jobs worth damaging the vibrant recreation industry on the Tongass? Are a handful of subsidized sawmill jobs worth damaging the vibrant commercial and sport fisheries associated with the Tongass? Economic theory indicates the answer is a resounding no.

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[53] Leighty, W., S. Hamburg, and J. Caouette. 2006. Effects of management on carbon sequestration in forest biomass in southeast Alaska. Ecosystems 9: 1051-1065.
[54] USDA. 2017. Economic effects of national forest recreation in Alaska. Alaska Region Briefing Paper, March 2017.
[55] *Ibid.*

The Washington Post
Democracy Dies in Darkness

Trump pushes to allow new logging in Alaska's Tongass National Forest

By **Juliet Eilperin** and **Josh Dawsey**

August 27, 2019 at 1:29 p.m. AKDT

President Trump has instructed Agriculture Secretary Sonny Perdue to exempt Alaska's 16.7-million-acre Tongass National Forest from logging restrictions imposed nearly 20 years ago, according to three people briefed on the issue, after privately discussing the matter with the state's governor aboard Air Force One.

The move would affect more than half of the world's largest intact temperate rainforest, opening it to potential logging, energy and mining projects. It would undercut a sweeping Clinton administration policy known as the "roadless rule," which has survived a decades-long legal assault.

Trump has taken a personal interest in "forest management," a term he told a group of lawmakers last year he has "redefined" since taking office.

AD

Politicians have tussled for years over the fate of the Tongass, a massive stretch of southeastern Alaska replete with old-growth spruce, hemlock and cedar, rivers running with salmon, and dramatic fjords. President Bill Clinton put more than half of it off limits to logging just days before leaving office in 2001, when he barred the construction of roads in 58.5 million acres of undeveloped national forest across the country. President George W. Bush sought to reverse that policy, holding a handful of timber sales in the Tongass before a federal judge reinstated the Clinton rule.

Trump's decision to weigh in, at a time when Forest Service officials had planned much more modest changes to managing the agency's single largest holding, revives a battle that the previous administration had aimed to settle.

In 2016, the agency finalized a plan to phase out old-growth logging in the Tongass within a decade. Congress has designated more than 5.7 million acres of the forest as wilderness, which must remain undeveloped under any circumstances. If Trump's plan succeeds, it could affect 9.5 million acres.

AD

Timber provides a small fraction of southeastern Alaska's jobs — just under 1 percent, according to the regional development organization Southeast Conference, compared with seafood processing's 8 percent and tourism's 17 percent.

But Alaskans, including Gov. Mike Dunleavy (R) and Sen. Lisa Murkowski (R), have pressed Trump to exempt their state from the rule, which does not allow roads except when the Forest Service approves specific projects. It bars commercial logging.

In a statement, Murkowski said Alaska's entire congressional delegation and the governor have sought to block the roadless rule.

"It should never have been applied to our state, and it is harming our ability to develop a sustainable, year-round economy for the Southeast region, where less than one percent of the land is privately held," she said. "The timber industry has declined precipitously, and it is astonishing that the few remaining mills in our nation's largest national forest have to constantly worry about running out of supply."

AD

Alaskan leaders have found a powerful ally in the president. Speaking to reporters on June 26, after meeting with Trump during a refueling stop at Elmendorf Air Force Base, Dunleavy said of the president, "He really believes in the opportunities here in Alaska, and he's done everything he can to work with us on our mining concerns, timber concerns; we talked about tariffs as well. We're working on a whole bunch of things together, but the president does care very much about the state of Alaska."

Trump expressed support for exempting the Tongass from the roadless rule during that conversation with Dunleavy, according to three people who spoke on the condition of anonymity to discuss internal deliberations. Earlier this month, Trump told Perdue to issue a plan to that effect this fall, these individuals said.

It is unclear how much logging would take place in the Tongass if federal restrictions were lifted because the Forest Service would have to amend its management plan to hold a new timber sale. The 2016 plan identified 962,000 acres as suitable for commercial timber and suggested no more than 568,000 acres of that should be logged.

AD

John Schoen, a retired wildlife ecologist who worked in the Tongass for the Alaska Department of Fish and Game, co-authored a 2013 research paper finding that roughly half of the forest's large old-growth trees had been logged last century. The remaining big trees provide critical habitat for brown bears, Sitka black-tailed deer, a bird of prey called the Northern Goshawk and other species, he added.

Trump has frequently talked with his advisers about how to manage the nation's forests and signed an executive order last year aimed at increasing logging by streamlining federal environmental reviews of these projects. The president was widely ridiculed after suggesting during a visit to Paradise, the California community devastated by a 2018 wildfire, that the United States could curb such disasters by following Finland's model, claiming that nation spends "a lot of time on raking and cleaning and doing things, and they don't have any problem."

The president has peppered Perdue with questions about forest management and has indicated that he wants to weigh in on any major forestry decision, according to current and former aides. Trump wanted to deprive California of federal funds in retaliation for the way officials managed the state's forests, but he did not follow up on the plan.

AD

One former Trump staffer, who spoke on the condition of anonymity to avoid retaliation, said forest policy has become "an obsession of his."

White House and Agriculture Department officials referred questions this week to the Forest Service, which declined to comment. But the three people who spoke on the condition of anonymity said it was forging ahead with an exemption at Perdue's instructions.

Chris Wood, president of the environmental group Trout Unlimited, joined with local business owners and conservation and outdoors organizations in urging federal officials to make more limited changes to the rule. He said the shift could jeopardize the region's commercial, sport and subsistence salmon fishing industry.

AD

About 40 percent of wild salmon that make their way down the West Coast spawn in the Tongass: The Forest Service estimates that the salmon industry generates $986 million annually. Returning salmon bring nutrients that sustain forest growth, while intact stands of trees keep streams cool and trap sediment.

Wood, who worked on the Clinton rule while at the Forest Service, said that in recent years, agency officials have "realized the golden goose is the salmon, not the trees."

"They need to keep the trees standing in order to keep the fish in the creeks," Wood said.

The question of what sort of roads should be built in the United States' remaining wild forests sparked intense battles in the 1990s, culminating in the 2001 rule affecting a third of the Forest Service's holdings in a dozen states. Some Western governors, including in Idaho and Wyoming, challenged the restrictions.

AD

In some cases, conservationists and developers have forged compromises. A decade ago, Idaho officials opened up roughly 400,000 acres of roadless areas to ease operations for a phosphate mine while protecting 8.9 million acres in exchange.

But in Alaska, consensus has been more elusive, with many state officials arguing that the limits have hampered development.

The Forest Service has approved at least 55 projects in roadless areas, according to the agency, including 36 for mining and 10 related to the power sector. Most win approval "within a month of submission," according to an agency fact sheet.

But Robert Venables, executive director of the Southeast Conference, said permitting for some projects has taken years and made them too costly to complete. A proposal that would have lowered electricity costs in the Alaskan community of Kake by connecting its supply to neighboring Petersburg, he said, won approval only after a lengthy review, which imposed requirements that boosted the price tag into the tens of millions.

AD

"The roadless rule has shown itself to be very arbitrary and cumbersome," Venables said in a phone interview. "Many projects have proven to be uneconomic because of the constraints here."

A number of businesses operating in the region back the current restrictions, arguing that the forest's rugged landscapes, abundant wildlife and pristine terrain draw visitors.

Dan Blanchard, owner and CEO of the adventure travel firm UnCruise Adventures , said in an interview that when he was working as a boat captain in the 1980s, "we had a difficult time avoiding clear cuts in southeast Alaska."

"The forest has come back," said Blanchard, who has 350 employees and brings 7,000 guests to Alaska each year. "The demand for wilderness and uncut areas have just dramatically increased. Our view here is, there are very few places in the world that are wild. Here we have one, in southeast Alaska, and it's being put at risk."

Juliet Eilperin
Juliet Eilperin is The Washington Post's senior national affairs correspondent, covering the transformation of federal environmental policy. She's authored two books, "Demon Fish: Travels Through The Hidden World of Sharks" and "Fight Club Politics: How Partisanship is Poisoning the House of Representatives." She has worked for The Post since 1998. Follow

Josh Dawsey
Josh Dawsey is a White House reporter for The Washington Post. He joined the paper in 2017. He previously covered the White House for Politico, and New York City Hall and New Jersey Gov. Chris Christie for the Wall Street Journal. Follow

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Comments

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mmcd2 3 months ago

You can take the boy out of Queens, but you cannot take the Queens out of the boy.

Can no one just confine this vulgar churl?

Like 1 Link Report

poil_brun 3 months ago

The argument from businesses is pretty simple: "Please make it easier for us to destroy the environment so that we can make more money".

There's nothing suprising about this demand or about Trump being receptive to it. The real issue is whether the population is happy about it or not.

Like 1 Link Report

Rockfish1 3 months ago

Damn him.

Like 4 Link Report

LarryBens 3 months ago

How can we ever convince Brazil or Indonesia to save much of their extensive tropical rainforests if Trump trashes one of world's rare temperate rainforests? It's doubtful that our overweight President, whose idea of enjoying nature is to ride in a cart around one of his golf courses, has ever hiked into one of the small remaining stands of California's once vast Coastal Redwood forests to get an idea of what would be lost in Alaska?

Like 3 Link Report

Cat 3 months ago

And his sons' idea of enjoying the outdoors is to kill wild and exotic animals. Ugh. Ugly family in thought and deed.

Like 5 Link Report

In memory of Thomas Francis Meagher 3 months ago *(Edited)*

It is always the case: whatever Trump touches dies. Can Planet Earth survive 4 more years of satan? Can we take the chance? There is no planet B!!!

Like 6 Link Report

ohiolawyer 3 months ago

My comment on the 50th anniversary of the Wilderness Act in 2014: In this golden anniversary year of the Wilderness Act, my hope is that we all leave this celebration not only proud of what has been accomplished, but rededicated to meet the enormous challenges of the future. I hope we leave this celebration prepared to not only protect more wild places on a map– important as that is – but to also meet the enormous challenge of climate disruption, to make our environmental movement and our society more broad, more just, and more inclusive, and to leave for future generations the beautiful, wild and livable planet that is their birthright.

Like 4 Link Report

Maricaibo 3 months ago

Trump knows as much about logging as he does about military service to America and Democracy

America and Democracy.

Like 3 Link Report

Time Was 3 months ago

Having been born and raised in the Tongass Forest I can tell you this: logging was always subsidized by the Feds and never a viable sustainable industry in Southeast Alaska. The number of jobs produced, especially now that all the pulp mills have close (thank god for that) are miniscule in comparison to fishing and tourism. All the Republicans care about is the profit they can eek out of the State - when they push the Pebble Mine through, and log the Tongass, what will be left for the next generation of Alaskans? Polluted streams and dead salmon runs. If Alaska's economy weren't so terribly managed by the Republicans they wouldn't find themselves in this predicament. Remember the "good old days" when the Alaska pipeline was new and they couldn't decide what to do with all the money? Now they want to cut the Alaska Ferry Service and other ridiculous means to save money. Not that they care that is one of the only means of transportation the citizens of Southeast have. And they just keep voting the bastards in... such stupidity and a waste.

Like 16 Link Report

Maricaibo 3 months ago

Those hardy, independent Alaskans have had it pretty rough since Ted Stevens died and stopped the DC-to-Juneau gravy train. The state has ALWAYS depended on the Feds.

Yet the idiots still support trump? Jeesh...

Like 2 Link Report

jettatdi 3 months ago *(Edited)*

The only thing sacred to tRump and the entire republiCON party is money.

Like 4 Link Report

res ipsa loquitur 3 months ago

Only T-rump's stupidity exceeds his arrogance.

Like 6 Link Report

View More Comments

Community Association of Elfin Cove
Resolution 19-01
A RESOLUTION BY THE COMMUNITY OF ELFIN COVE COMMENTING ON THE PROPOSED ALASKA ROADLESS RULE

WHEREAS, the community of Elfin Cove is a unique community that prospers by and through the protection of its natural resources; and,

WHEREAS, the sustainability of the future of the Elfin Cove economy and the lifestyles of our citizens depend on nurturing and sustaining the natural resources supported by our forests, rivers, and ocean, both locally and regionally in Southeast Alaska; and,

WHEREAS, many Elfin Cove residents obtain their livelihoods from economic activities including tourism, sport fishing lodges, and commercial fishing, which are highly dependent on a pristine and productive natural setting; and,

WHEREAS, salmon populations are facing increased challenges from changing ocean conditions, climate change, ocean acidification, rising water temperatures in streams, and ocean warming events in the Pacific Ocean, and freshwater habitat development in the contiguous United States; and,

WHEREAS, Southeast Alaska is the most visited region in Alaska, with two thirds of all visitors to the State coming to this region and therefore acting as a significant contribution; and,

WHEREAS, the Roadless Rule was first adopted in 2001 to protect 58 million acres of our nation's most essential and intact roadless areas in the National Forest system; and,

WHEREAS, the State of Alaska has been against the inclusion of the Tongass in the Roadless Rule since it was promulgated in 2001, despite support for the rule from a wide variety of constituents living in Southeast Alaska; and,

WHEREAS, during the scoping process for an Alaska Specific Roadless Rule in 2018, communities throughout Southeast Alaska spoke in support of the 2001 Roadless Rule, as well as the heavy majority of the official scoping comments received strongly supported maintaining the 2001 Roadless Rule on the Tongass; and,

WHEREAS, the State of Alaska invested time, staff, and money to convene a stakeholder-led Citizen Advisory Committee in 2018 that provided recommendations and perspectives from different interest groups in Southeast Alaska on what a workable compromise for an Alaska Roadless Rule could look like; and,

WHEREAS, the pursuit of a full exemption from the 2001 Roadless Rule is highly divisive among local populations and stakeholders and increased conflict on the Tongass will tarnish the reputation of the region nationwide, with negative impacts for the tourism industry; and,

WHEREAS, roadless areas on the Tongass are essential to Southeast Alaska's way of life and represent some of the most spectacular and unique roadless areas anywhere in the National Forest System and support hunting, fishing, customary and traditional uses, unparalleled outdoor recreation opportunities, and opportunities for businesses; and,

WHEREAS, the amount of carbon stored in the intact old growth forests and soils of the Tongass National Forest represent one of the highest carbon stores in the world, and the conservation of intact roadless areas and old growth forests on the Tongass are essential for maintaining local and national climate resilience and slowing down climate change throughout the world; and,

WHEREAS, the Tongass provides a wide range of ecosystem services beyond carbon sequestration to residents and visitors alike on a 24/7 basis naturally and at no cost to the taxpayer, and;

WHEREAS, increased logging of old growth forests in the existing roadless areas of the Tongass seriously impact the ability of the forest to provide these vital ecosystem services, such as clean water, clean air, stream temperature regulation, and healthy salmon and wildlife habitat; and,

WHEREAS, increased logging of old growth forests in existing roadless areas would incur a significant cost to taxpayers, as new reports estimate that the subsidization of industrial-scale logging on the Tongass has cost taxpayers over $600 million over the past twenty years; and,

WHEREAS, according to the State of Alaska's own economic experts and commissioned reports, Tongass timber is uncompetitive due to fundamental, permanent changes that have occurred in global markets, high labor costs, distance from markets, and the availability of less-expensive substitutes; and,

WHEREAS, the sport fishing, commercial fishing, and visitor industries that provide the economic backbone of Southeast Alaska depend on pristine, scenic vistas and healthy, productive watersheds; and,

WHEREAS, opening up roadless areas for future development industrial-scale logging will actively work against the Tongass Transition and the 2016 Tongass Land Management Plan Amendment, which was created after years of collaborative work between a diverse group of stakeholders;

THEREFORE, BE IT RESOLVED, that given long-lasting, fiscally irresponsible, and environmentally damaging impacts of any reduction in current Roadless Rule protections on the Tongass, the community of Elfin Cove strongly supports lasting protection for all inventoried roadless areas on the Tongass National Forest as provided for in the 2001 Roadless Rule.

There should be no exemption from the Roadless Rule for Alaska and there should be no Alaska-specific roadless rule; and,

BE IT FURTHER RESOLVED, that the economic livelihoods of Elfin Cove residents would be irreparably harmed by the effects of a full exemption from the Roadless Rule on the Tongass, which would lead to increased climate change, increased threats to fish habitat, and development occurring in pristine natural areas that showcase the tourism potential and natural beauty of the Tongass National Forest; and,

BE IT FINALLY RESOLVED, that the community of Elfin Cove encourages the USFS to invest in programs, workforce development, and employment opportunities for Southeast Alaskan residents to restore the health and productivity of the Tongass National Forest's degraded watersheds and streams, as a means of supporting vital ecosystem services such as carbon sequestration, small-scale timber production, fish and wildlife populations, natural vistas, visitor amenities, subsistence, hydrological resources, and more.

The Community of Elfin Cove submits the attached letter with expanded and specific comments for our local area.

Passed and Approved by the Community of Elfin Cove this 30th of October, 2019.

Wayne Stauffer (electronically signed)

Community of Elfin Cove, Chairman

The Effects of Forest Management on Erosion and Soil Productivity

William J. Elliot
Project Leader

Deborah Page-Dumroese
Research Soil Scientist

Peter R. Robichaud
Research Engineer

Intermountain Research Station, USDA Forest Service, Moscow, Idaho 83843

An invited paper presented at the Symposium
on Soil Quality and Erosion Interaction
sponsored by
The Soil and Water Conservation Society of America
held at
Keystone, Colorado

July 7th, 1996

Currently under Review for inclusion in Proceedings

Introduction

For many years, research has been ongoing to relate soil erosion to productivity. Most research has focused on agricultural or rangeland conditions. Pierce (1991) presents an overview including over 60 references summarizing past research on impacts of erosion on agricultural production. He concluded that exact relationships between erosion and productivity are unclear, and considerable research is necessary over a wide range of soil and plant conditions to define any such relationship. Research on the effects of soil erosion on forest productivity is limited. This paper will provide an overview of current knowledge on the effects of forest management on soil erosion, and related onsite impacts, and the effects of those impacts on forest productivity.

Soil erosion in an undisturbed forest is extremely low, generally under 1 t ha^{-1} yr^{-1}. Disturbances, however, can dramatically increase soil erosion to levels exceeding 100 t ha^{-1} yr^{-1}. These disturbances include natural events such as wild fires and mass movements, and human-induced disturbances such as road construction and timber harvesting. Soil erosion, combined with other impacts from forest disturbance, such as soil compaction, can reduce forest sustainability and soil productivity. Research is ongoing to quantify the effects of disturbance on soil erosion and soil productivity.

Forest Practices

Soil erosion in forests generally follows a disturbance, such as road construction , logging operations, or fire. In undisturbed forests, erosion is generally due to epochal events associated with fire cycles, land slides, and geologic gully incision.

Ground cover by forest litter, duff and organic material is the most important component of the forest environment for protecting the mineral soil from erosion and provides most of the nutrients needed for sustainable forestry. Ground cover amounts can be reduced by the logging operation (harvesting and site preparation) and burning either by wildfire or prescribed fire. For example, skidder traffic on skid trails can reduce ground cover from 100 to 10-65 percent. Burning can reduce ground cover from 100 percent to 10-90 percent depending on the fire severity.

Roads

In most managed forest watersheds, most eroded sediment comes from roads. Roads have no vegetative protection, and tend to have low hydraulic conductivities leading to much

greater runoff and erosion rates than in the surrounding forests (Elliot et al., 1994a). Numerous researchers, including Swift (1988) and Bilby et al. (1989), have quantified the major role of roads on sedimentation in forests. In addition to erosion, roads also reduce forest productivity by the land that they occupy. A kilometer of road in 1 km^2 of forest represents a 0.5 percent loss in area and removal from productivity. In some areas, forest roads can occupy up to 10 percent of the forest area if there is a past history of intensive logging. Roads are assumed to be unproductive in forest plans, regardless of any erosion impacts.

Currently, the USDA Forest Service has a major program to close roads. Closure methods can vary from locking a gate to completely removing the road prism in an effort to reduce sedimentation and related hydrologic problems. The productivity of closed or removed roads has not been directly measured, but frequently, additional mitigation measures such as ripping and replanting are included in any closure scenario to encourage maximum regrowth rates (Moll, 1996).

Timber Management

Traditionally, forest management practices focus on fire suppression and clear-cut logging methods. With an increased understanding of forest ecosystems, the USDA Forest Service is applying ecosystem management principles to forest management. These principles include partial cut management systems and increased use of prescribed fires. Such practices, however, require more frequent operations in the forest environment.

Harvesting Effects

Harvesting methods vary in degree of disturbance. On steeper slopes (generally > 35 percent slope) helicopter, skyline, or ground-cable logging systems are common. Trees are felled and removed with full suspension of logs via a helicopter or cable system and carried to landing sites. With a ground cable system, one end of the log is suspended and the other end is slid on the ground to a landing area. On less steep slopes (generally < 35 percent slope), wheeled or tracked forwarders or skidders remove felled trees. A forwarder loads and carries trees to a landing area in one operation, or a skidder drags the logs to the landing generally on designated skid trails. Skid trails cause the most disturbance by displacing the ground cover and compacting the mineral soil. Additional disturbance is caused by skidder tires loosening the soil, especially on slopes of 20-35 percent.

Even though timber harvest operations usually cause less erosion per unit area than roads, the area of timber harvest is usually large relative to roads so that the total erosion from timber

harvest operations may approach that from roads (Megahan 1986). Tree cutting by itself does not cause significant erosion, although the resulting decrease in evapotranspiration contributes to increased subsurface flow, streamflow and channel erosion. However, soil disturbance caused by the harvesting operation results in reduced infiltration capacities and increased surface runoff which promotes surface erosion (Yoho 1980). Accelerated erosion caused by timber harvesting activity may result in deterioration of soil physical properties, nutrient loss, and degraded stream water quality from sediment, herbicides and plant nutrient inputs. (Douglas and Goodwin 1980).

Nutrient Impacts

Harvesting trees removes nutrients from a generally nutrient deficient environment (Miller et al., 1989). Table 1 shows the effect of tree harvest on nitrogen availability. Increasing harvest intensity from bole only through whole tree and complete biomass harvesting doubled nitrogen loss on the average quality site, but more than tripled loss at the poor quality site. Leaching losses are also greater on the poorer site. Researchers generally agree that harvesting the bole only will not greatly deplete nutrient reserves, but shorter rotations and whole tree harvesting removes more nutrients than can be replaced in a rotation. Harvesting crowns is undesirable because they contain a large portion of the stand nutrient content.

Fire Effects

The most common method of site preparation in the U.S. is prescribed burning. Although mechanical methods are commonly used in southern forests to physically destroy or remove unwanted vegetation from the site and to facilitate machine planting. Burning is conducted alone, and in combination with other treatments, to dispose of slash, reduce the risk of insects and fire hazards, prepare seedbeds, and suppress plant competition for natural and artificial regeneration. Current research is finding that fire helps maintain forest health. Fire has long been a natural component of forests ecosystems (Agee 1993). The use of prescribed fire will increase as ecosystem management strategies include a greater use of fire, and fewer clearcuts, and more partial cuts.

Erosion following fires can vary from extensive to minimal, depending on the fire severity and areal extent (Robichaud and Waldrop, 1994). Fire severity refers to the effect of the fire on some component of the forest ecosystem, such as nutrient loss or amount of organic material consumed (litter and duff). Erosion from high severity fires can be cover large areas.

and. fires may create hydrophobic or water repellent conditions. Erosion from low severity fires may be minimal to none (Robichaud et al. 1993b; Robichaud and Waldrop 1994).

Erosion Modeling

Since the late 1950s, soil erosion models have provided natural resource managers with tools to predict the impacts of management practices on soil erosion. Earlier models tended to focus on midwest and southeast agricultural conditions where erosion was considered to be a severe problem associated with farming practices. Models for range lands and forest lands have only recently been receiving wide-spread interest as managers have begun to focus as much on off-site sediment impacts as on onsite erosion rates.

Sediment Yield Models

Most of the early models, which culminated in the Universal Soil Loss Equation (USLE), focused on upland soil erosion rates (Wischmeier and Smith, 1978). The USLE was developed to predict soil erosion from small, relatively homogenous plots (Mutchler et al., 1994). Forest environments tend to have much greater spatial variability in vegetation and soils (Elliot et al., 1996), making the application of the USLE difficult. Dissmeyer and Foster (1985) developed a subfactor approach to predict soil erosion from forest conditions for areas where intensive operations such as tillage are carried out, and harvest areas can be considered similar to intensively managed farming systems. The erosion-productivity impact calculator (EPIC) model was developed to apply the USLE prediction technology to long-term productivity impact predictions (Williams et al., 1984). The EPIC model, however, was developed for applications to croplands only.

Forest service specialists have developed watershed models to aid in predicting the cumulative effects of road and harvest area erosion on stream sedimentation (like WATSED, Range, Air, Watershed and Ecology Staff Unit, 1991). The strength of these models is in allowing assessment of cumulative effects on stream sedimentation in a large watershed. WATSED, however, was not developed to predict site-specific effects.

More recent physically-based soil erosion models, including the Chemicals Runoff and Erosion from Agricultural Management Systems (CREAMS) model (Knisel, 1980) and the Water Erosion Prediction Project (WEPP) model (Laflen et al., 1991) have also provided estimates of sedimentation for predicting both onsite and offsite impacts. The WEPP model, in particular, has shown considerable promise as a tool to assist in predicting soil erosion and sediment yields in a forest environment (Elliot et al., 1996).

The physically-based WEPP model allows predicting upland erosion and offsite effects from erosion events influenced by management activities (Laflen et al., 1991). Erodibility values have been measured on forest roads and disturbed harvest areas, and validation activities with the WEPP model for forests have been encouraging (Elliot et al., 1994). The WEPP model not only predicts erosion, but also predicts the textural and organic composition of the eroded sediment.

Productivity Response to Management

A coordinated national research effort is being implemented on a broad spectrum of benchmark sites across the nation (Powers et al., 1990). These sites were relatively undisturbed prior to study installation. An extensive range of pre- and post-harvest measurements are being taken. This study alters site organic matter and total soil porosity over a range of intensities encompassing a range of possible management scenarios and creates a network of comparable experiments producing nil to severe soil disturbance and physiological stress in vegetation over a broad range of soils and climates. Establishing and monitoring this network directly addresses the needs of National Forest Systems, and creates a research opportunity of unusual scope and significance. Early results indicate that immediate post-harvest biomass declines are most likely caused by compaction and not organic matter removal whereas long-term productivity changes will be more dependent on organic matter losses.

Erosion Loss

The close tie between surface organic matter and forest soil productivity is clear (Jurgensen et al., 1996). As a rooting medium for higher plants, soils provide the essentials of water, structural support, nutrients, and soil biota. Mixing and/or short-distance displacement of topsoil and surface organic matter from a site can decrease productivity. Soil disturbance by logging is generally less than 30% of the total harvested area (Rice et al., 1972; Miller and Sirois 1986), but the impact can be severe. Erosion can further damage site productivity. First, erosion reduces crop productivity mainly by decreasing the soil water availability; this is a result of changing the water holding capacity and thickness of the root zone (Swanson et al. 1989). Second, erosion also removes plant available nutrients. Fertilizer applications can partly offset these losses, but they greatly increase costs and are uncommon. Third, erosion reduces productivity by degrading soil structure. Removal of the loose, organic surface materials promotes surface sealing and crusting which decreases infiltration capacity and may increase erosion (Childs et al. 1989). Fourth, erosion results in loss of important soil biota,

such as mycorrhizal fungi, which facilitate nutrient uptake by plants (Amaranthus et al., 1989, 1996).

Surface erosion proceeds downward from the O horizons. Because the highest concentrations of nutrients and biota and the maximum water-holding capacity are in the uppermost soil horizons, incremental removals of soil nearer the surface are more damaging than those of subsoils. Productivity may inevitably decline on most shallow forest soils as erosion causes root-restricting layers to be nearer the surface and as organic matter is washed away. Consequently, the largest declines in productivity are most likely to occur in marginal, dry environments.

Assessing the effects of erosion on site productivity is often difficult. Erosion rates are poor indicators of loss in productivity because most soil is redistributed within a watershed and not necessarily lost to production. Soils differ in their tolerance to erosion loss. For instance, Andisols have relatively high water-holding capacity and natural fertility. Erosion may be severe on these sites, but productivity may decline little. In contrast, Spodosols frequently lose productivity because they are commonly highly leached and naturally infertile, they retain fertilizers poorly, and have low water-holding capacity.

Compaction Impacts

Field research has also found that timber harvesting systems tend to compact the soil. Compaction increases soil erosion, and adversely impacts forest productivity. Most erosion comes from skid trails on timber harvested units (Robichaud et al. 1993b). This is due to the low infiltration rates and disturbance to the organic layer.

Compaction of forest soil is a serious concern for managers because of the use of heavy equipment to harvest timber and prepare a site for planting. Usually, the more porous the soil initially, the greater the compaction depth. For example, volcanic ash soils of the western U.S. are highly productive in their undisturbed condition, but are prone to compaction because they have a low volume weight (weight-to-volume ratio) and relatively few coarse fragments (Geist and Cochran 1991). Once these sites have been disturbed through timber harvest activities and site preparation, porosity (Dickerson 1976) and hydraulic conductivity declines (Gent et al., 1984). Compaction depth can exceed 450 mm (Page-Dumroese 1996).

Compaction is a reduction in total porosity. Macro porosity is reduced while micro porosity increases as large pores are compacted into smaller ones. An increase in micro

porosity can lead to greater available water-holding capacity throughout a site, but this increase is usually at the expense of aeration and drainage (Incerti et al., 1987).

There is little doubt that compaction reduces productivity (Greacen and Sands 1980; Froehlich and McNabb 1984). Reduction in root growth, height, and timber volume have been observed (Froehlich and McNabb 1984) and may be produced by a single pass of logging equipment across a site (Wronski 1984). Productivity losses have been documented for whole sites (Wert and Thomas 1981) and for individual trees (Froehlich 1979; Helms and Hipkin, 1986). Decreases in important microbial populations have been observed in compacted soils (Amaranthus et al., 1996). In general, however, the environmental degradation observed in the field result from both compaction and disturbance or removal of surface organic horizons (Childs et al., 1989).

Soil compaction may also increase surface runoff because of reduced infiltration (Greacen and Sands 1980). However, because of increased soil strength, compacted soils may have lower erodibility, and consequently suffer less erosion for the same amount of runoff (Liew 1974). A significant amount of erosion after harvest activities has been attributed to compaction, but may be attributable to both compaction and the removal of vegetative cover (Dickerson 1976).

Predicted Erosion Rates and Productivity

A series of WEPP runs were carried out on a productivity study site in central Idaho to allow comparison of a range of management effects on soil erosion. The predicted effects on erosion from wildfires were compared to prescribed fires, partial cuts, and clear cuts to better understand the interactions among natural events, human activities, soil erosion, soil productivity, and ultimately forest ecosystem sustainability.

Harvesting Impacts

For the modeling study, a slope length of 100 m, with a steepness of 61 percent was modeled, typical of the site Soil properties of the site are presented in Table 2. The WEPP management file described a forest in the first year, a disturbance in the second year, and regeneration of forest in eight subsequent years as described by Elliot et al. (1996). The biomass reduction due to harvest effects was described in the residue management and harvest index (harvest index = biomass removed/biomass present) values in the management files. The values assumed are presented in Table 3. The climate for the simulations was stochastically

generated with the CLIGEN generator (Flanagan and Livingston, 1995) using the Deadwood Dam, ID climate statistics (mean annual precipitation = 830 mm).

Tables 4 and 5 present the predicted runoff and erosion rates for different treatments. Continuing field research will collect field data from the productivity treatments. The WEPP predictions are generally logical. More compaction leads to greater runoff and greater erosion. The effect of removing greater amounts of vegetation also leads to greater erosion rates. The complete removal of biomass was modeled as removing 100 percent of the surface residue, which resulted in a small increase in runoff, but a doubling of erosion rates. The role of surface residue is critical in controlling erosion in forests just as it is in agriculture.

An additional WEPP run was made with no disturbance. In this scenario, there was no runoff and no erosion. With the amount of residue cover and litter accumulation typical of forests, WEPP seldom predicts erosion. Our field observations generally confirm this, with most sediment from undisturbed watersheds coming from eroding ephemeral channels or landslides.

In order to compare the productivity impacts of soil erosion, an estimation was made of the nitrogen losses associated with the above erosion rates. It was assumed that the typical forest soil contains 4 percent organic matter, and that organic matter is 2 percent nitrogen. The resulting nitrogen losses for 8 years of predicted erosion are presented in Table 6. The values in Table 6 can be compared to Table 1 to see that nutrient losses due to erosion are significant, greater than observed leaching losses, but not as great as losses due to vegetation removal. In a generally nutrient-deficient environment, these nitrogen losses will have a significant impact on future productivity.

Natural Fire Impacts

To model a severe fire, 100 percent of the residue was burned, and half of the remaining biomass was harvested in the autumn. This is generally much more severe than observed in the field, but allows comparison of the extreme events. Generally, even "severe" fires do not remove more than 90 to 95 percent of the residue, and the remaining residue can reduce the predicted erosion rates by more than 90 percent. If the soil hydraulic conductivity remained unchanged, there was little change in either runoff or erosion from the values predicted for the severe compaction, bole removal treatment. If the hydraulic conductivity was reduced to 4 mm/hr to reflect hydrophobic soil conditions that sometimes occurs after severe fires, then the predicted runoff was doubled to 65 mm per year. The predicted erosion was 11.6 t ha^{-1}, greater than the bole and crown removal treatments, but still somewhat less than the predicted rates on

sites with complete biomass removal. As the soil hydrologically recovers following a severe fire, the runoff and erosion rates would decline, a characteristic that WEPP is currently not capable of modeling continuously. Such a scenario could be developed with a series of one-year runs with a different conductivity for each year.

Summary/Conclusions

We have presented an overview of the impacts of forest management activities on soil erosion and productivity. Erosion alone is seldom the cause of greatly reduced site productivity. However, erosion in combination with other site factors, work to degrade productivity on the scale of decades and centuries. Extreme disturbances, such as wildfire or tractor logging, cause the loss of nutrients, mycorrhizae, and organic matter. These combined losses reduce long-term site productivity and may lead to sustained periods of extended erosion which could exacerbate degradation.

From a management perspective, we should be concerned with harvesting impacts, site preparation disturbances, amount of tree that is removed, and the accumulation of fuel from fire suppression. On erosion-sensitive sites, we need to carefully evaluate such management factors.

Prescribed fire is generally an excellent tool in preparing sites for regeneration, for reducing fuel loads, and for returning sites to a more natural condition. Burning conducted under correct conditions will reduce the fire hazard, make planting easier, and retain the lower duff material to protect the mineral soil and conserve nutrients to sustain forest productivity.

The WEPP model can describe various impacts due to harvesting, but further work is required to model fire effects and the subsequent temporal and spatial variation in soil hydraulic conductivity and ground cover effects. From field observations and the modeling exercise, it appears that disturbances caused by harvest activities will lead to increases in erosion and runoff rates, much greater than natural conditions, even when extreme wild fire effects are considered.

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Table 1 Comparison of height, diameter, and nitrogen pools after harvest treatments of varying intensities from two sites of differing site quality, Pack Forest, WA[1]

Harvest treatment	Height growth (m)	Diameter growth (cm)	Total N	Harvest loss	3-yr leaching loss	% loss
			- - - - - - - - - - - - kg/ha - -- - - - - - -			
Average site quality						
Bole only	1.7	2.9	2,935	470	4.4	16
Whole tree	1.9	3.2	2,827	678	0.5	24
Complete[2]	1.8	3.6	2,719	870	0.7	32
Poor site quality						
Bole only	1.4	2.2	984	157	2.1	16
Whole tree	1.1	1.6	903	289	4.7	32
Complete[2]	1.1	1.5	934	486	5.5	53

[1] From Miller et al. 1989.

[2]Complete removal of all above ground biomass.

Table 2. Soil properties assumed for the WEPP model computer simulations

Soil Property	Value	Units
Sand content	40	percent
Silt content	45	percent
Clay content	15	percent
Interrill erodibility	2100	kg s m^{-4}
Rill erodibility	0.008	s m^{-1}
Critical shear	3	Pa
Saturated hydraulic conductivity		
uncompacted	20	mm hr^{-1}
moderate compaction	15	mm hr^{-1}
severe compaction	8	mm hr^{-1}

Table 3. Values describing the effects of timber harvest in the WEPP model computer simulations

Treatment	Residue Management	Harvest Index
Complete biomass removal	100 percent surface residue removed	0.9
Bole and crown removed	No surface residue removed	0.8
Bole only removed	No residue management	0.4

Table 4. Average annual runoff (mm) from rainfall from the WEPP simulations for five simulated forest conditions

	Compaction		
Treatment	None	Moderate	Severe
	- - - mm - - -		
Undisturbed	0.0	--	--
Complete biomass removal	12.8	18.8	35.6
Bole & Crown removed	9.2	15.4	32.4
Bole only removed	9.1	16.1	32.7
Severe wild fire	65.0	--	--

Table 5. Average annual soil loss (t ha^{-1}) from the WEPP simulations for five simulated forest conditions

Treatment	Compaction		
	None	Moderate	Severe
	- - - - t ha^{-1} - - -		
Undisturbed	0.0	--	--
Complete biomass removal	4.5	7.4	14.4
Bole & crown removed	2.0	3.3	7.2
Bole only removed	2.0	3.5	7.2
Severe wild fire	11.6	--	--

Table 6. Predicted nitrogen loss due to erosion in the first 8 years of regrowth following harvest

Treatment	Compaction		
	None	Moderate	Severe
	- - kg ha^{-1} - -		
Undisturbed	0.0	--	--
Complete biomass removal	28.8	47.4	92.2
Bole & crown removed	12.8	21.1	46.1
Bole only removed	12.8	22.4	46.1
Severe wild fire	74.2	--	--

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Rocky Mountain Research Station
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1221 South Main Street
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RESEARCH ARTICLE

Biotic factors influencing the unexpected distribution of a Humboldt marten (*Martes caurina humboldtensis*) population in a young coastal forest

Charlotte E. Eriksson[1]*, **Katie M. Moriarty**[2¤], **Mark A. Linnell**[3], **Taal Levi**[1]

1 Department of Fisheries and Wildlife, Oregon State University, Corvallis, OR, United States of America,
2 Pacific Northwest Research Station, USDA Forest Service, Olympia, WA, United States of America,
3 Department of Forest Engineering, Resources, and Management, Oregon State University, Corvallis, OR, United States of America

¤ Current address: National Council for Air and Stream Improvement, Inc (NCASI), Corvallis OR, United States of America
* charlotte.eriksson@oregonstate.edu

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Abstract

Pacific martens (*Martes caurina*) are often associated with mature forests with complex structure for denning, resting, and efficient hunting. Nonetheless, a small isolated population of the Humboldt subspecies of Pacific martens (*Martes caurina humboldtensis*) occupies a narrow strip of young, coastal forest (< 70 years old) but not inland mature forest in the central Oregon Coast Range. We examined factors contributing to this unexpected distribution of martens by 1) analyzing marten diets using DNA metabarcoding to assess 90 scats, 2) using camera traps to assess differences in the relative abundances of prey, competitors, and predators across a coastal to inland gradient of vegetation types, and 3) quantifying differences in extent of fruit-producing shrubs and vegetation structure within vegetation types. Diets of martens were diverse (12, 10, and 3 species of birds, mammals, and amphibians respectively), and most fall and winter scats contained fruit. Voles, mice, and varied thrushes (*Ixoreus naevius*) were dominant prey items. Voles, mice, and most birds, but not varied thrushes, were more commonly observed in the coastal shrub-dominated forest than in inland forest. The coastal shrub-dominated forest had the highest diversity of vertebrates and potential prey overall. Bobcats (*Lynx rufus*), a key potential predator, were more commonly detected in inland forest. Of potential competitors, western spotted skunks (*Spilogale gracilis*) were more commonly detected in inland forest, with gray foxes (*Urocyon cinereoargenteus*) and raccoons (*Procyon lotor*) detected almost exclusively in coastal forests. Vegetation in coastal forests appears to provide, at least seasonally, more prey and fruit, and more overhead shrub cover compared with inland forest. Remaining plausible hypotheses for the restricted distribution of marten to coastal forests include increased prey, fruit, and overhead cover, and reduced predation risk from bobcats.

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Introduction

Understanding the drivers that shape a species' distribution is a fundamental aspect of conservation. These drivers may be abiotic conditions including climate, altitude, and elevation [1–4] or biotic factors such as prey availability, competition and the influence of predators [2,4–6]. Although examples exist with clear causal mechanisms that facilitate presence or recovery of species, commonly we are limited in the availability of natural history data to establish these linkages. A lack of information regarding key prey species, competitors, and predators restricts inferences about whether species interactions influence species' distribution [7]. This may be particularly relevant for mid-trophic taxa such as small carnivores that are subject to predation but may also be limited by prey availability [8]. Here we focus on potential effects of species interactions on the distribution of a poorly studied small carnivore, the Humboldt marten (*Martes caurina humboldtensis*) subspecies in the Pacific Northwest of North America.

Humboldt martens in the central coast of Oregon were withdrawn from listing consideration under the United States Endangered Species Act in 2015 [9]. The withdrawal concluded that large areas of mature forests present within their historical range provided sufficient habitat for the species to persist [9,10]. After the listing decision, subsequent distributional surveys have suggested that despite extensive older and mature inland forests within the central coast range of Oregon, these forests did not currently support a marten population [11]. Instead, a Humboldt marten population appears isolated to a narrow strip of young coastal forests (< 70 years old) growing on sandy soils on the margin of the Pacific Ocean [11], contrasting prevailing observations that martens require old (e.g., > 200 years) structurally complex forests [12]. The population estimate for this remnant marten population was 41–87 individuals living at unusually high density [13]. Evidence that the current populations of Humboldt martens appear small and isolated have contributed to a recent proposal to reconsider for listing the coastal Distinct Population Segment as Threatened [14]; however, little information is available describing predicted habitat and distribution of these coastal martens.

North American marten (*M. caurina*, *M. americana*) densities have been correlated with the abundance of prey species such as voles [15], mice, and squirrels [16]. In comparison to other carnivores, martens have high metabolism and limited fat reserves [17]. A typical marten must consume approximately 25% of its body weight daily, the equivalent of 7 red-backed voles (25 g each) (*Myodes* spp.), a common prey item across the range of these species [18]. Due to their relatively high metabolic rate, martens could be limited by bottom-up resource availability, suggesting that differences in prey availability could drive the distribution of marten. However, being a small-sized carnivore, martens must balance their high nutritional requirements with predation risk by larger mammalian carnivores such as bobcats (*Lynx rufus*), coyotes (*Canis latrans*), fishers (*Pekania pennanti*), and avian predators [19,20]. Martens are often associated with dense overstory cover and forests with structural complexity that are thought to reduce predation risk [21]. Differences in structural complexity and/or the abundance of their predators are thus plausible hypotheses to at least partially explain the observed Humboldt marten distribution. Similarly, other carnivores of similar body size and diet composition, such as gray fox (*Urocyon cinereoargenteus*), raccoon (*Procyon lotor*), and western spotted skunk (*Spilogale gracilis*) could also play a role in Humboldt marten distribution in coastal Oregon.

Structurally-complex mature forests, typified by logs, legacy trees, and snags >200 years old [22], are thought to fulfill the life-history requirements of North American martens. Such forests provide thermally efficient resting and denning sites [23,24], increased prey availability [16,25], foraging success [26], and predator protection. It is unclear whether and how the young coastal forest in Central Oregon provides for the life history requirements of Humboldt

martens (henceforth, martens), but better understanding of the ecological context may be critical to marten conservation.

The apparent contradiction between habitat associations based on forest age, and a broad lack of natural history information about this marten population motivated our study. Our objectives were to assess factors contributing to the observed restricted distribution of martens to coastal forests compared to inland forest by 1) quantifying the diet of martens to determine nutritional resources consumed in the coastal forests, 2) quantifying differences in the relative abundance of prey, predators, or competitors, and 3) assessing differences in fruit availability and vegetation structure. We quantified marten diet with DNA metabarcoding of scats followed by mechanical sorting of a subset of scats to assess seed and invertebrate composition. Because it was unclear *a priori* whether a particular vegetation type might support key prey, predator or competitor species influencing the persistence of martens, we used grids of camera traps to characterize relative abundance of these species across our vegetation type gradient. Finally, we conducted vegetation surveys at all camera stations to quantify ground cover, shrub, and overstory characteristics, and cover of fruit-producing shrubs.

Materials and methods

Study area

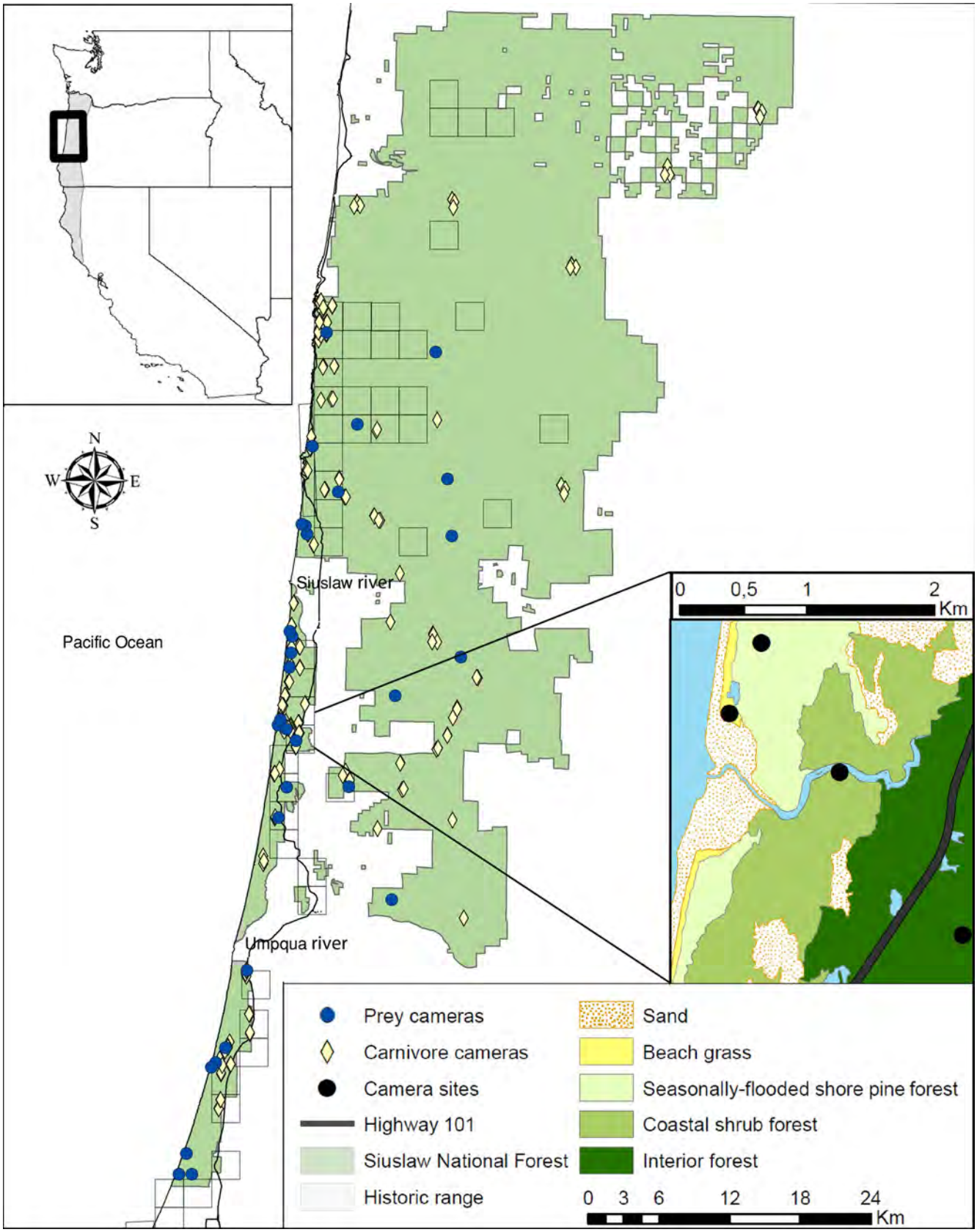
Our study occurred in the Siuslaw National Forest on the central Oregon coast, United States, with most coastal forests located in the Oregon Dunes National Recreation Area, hereafter Oregon Dunes (43˚42' N, 124˚10'W) (Fig 1). This region had a mild Pacific maritime climate with mean minimum and maximum temperatures of 10.1˚C and 20.3˚C in summer and 3.2˚C and 10.2˚C in winter. Total rainfall per year was approximately 1800 mm (Western Regional Climate Center 2017; https://wrcc.dri.edu/).

We stratified the study area into 4 vegetation types (listed from coast to inland) based on species dominance adapted from [27]: 1) foredune beach grass, hereafter "beach grass" 2) shore pine (*Pinus contorta*)/slough sedge (*Carex obnupta*) seasonally flooded forests located in the deflation plains, hereafter "seasonally-flooded shore pine forest", 3) shore pine-sitka spruce (*Picea sitchensis*) forest with a dense ericaceous understory dominated by evergreen huckleberry (*Vaccinium ovatum*) and salal (*Gaultheria shallon*), hereafter "coastal shrub forest" and 4) Douglas-fir (*Pseudotsuga menziesii*) and sitka spruce dominated forests, inland of the coastal zone, hereafter "interior forest". The 4 vegetation types were located within close proximity to each other, occasionally with all 4 types spaced within 2.5 km from the Pacific ocean (Fig 1).

Marten diet

We quantified diet from marten scats using DNA metabarcoding. We acquired scats in 3 ways: using scent detection dog teams (2015–2017), by observers opportunistically when trapping martens (2015–2016), and while identifying locations where martens rested (Winter 2016), during a separate study [28] (S1 Document). Detection dog surveys either saturated accessible young coastal forests in the central Oregon coast or were placed using a stratified random design in the interior forest in 3 x 3 km sample units with searches > 6 hours (see [29] for methods).

We dried scats in the field and processed them in the lab. We extracted DNA for species and diet composition in a laboratory dedicated to processing degraded DNA using the QIAamp DNA Stool Mini Kit (Qiagen, USA). We included an extraction blank with every batch of extractions as a negative control to monitor for cross-contamination. We used slightly modified universal vertebrate primers 12SV5F (`TTAGATACCCCACTATGC`) and 12SV5R (`YAGAACAGGCTCCTCTAG`) to amplify the ribosomal mitochondrial 12S gene region (adapted


N
W
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Pacific Ocean
Siuslaw river
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0 0,5 1 2
Km
Prey cameras
Carnivore cameras
Camera sites
Highway 101
Siuslaw National Forest
Historic range
Sand
Beach grass
Seasonally-flooded shore pine forest
Coastal shrub forest
Interior forest
0 3 6 12 18 24
Km

Fig 1. Map of Humboldt marten (*Martes caurina humboldtensis*) study area and survey locations on the central coast, Oregon, USA. We collected data on potential marten prey (n = 31 grids with 20 baited cameras per grid, 620 camera stations (circles). For potential competitors and predators, we used data from 259 sample units (n = 843 camera stations including the cameras on the grids) in forest east of the coastal highway (n = 86 baited, 55 unbaited trail camera stations) and coastal forests (n = 74 baited, 44 unbaited trail) (diamonds). Six marten detections (n = 4 genetically verified scats, 2 remote camera photographs) were within 500–700 m from the Pacific Ocean, with the remainder (n = 86 genetically verified scats, 5876 remote camera photographs) within the Oregon Dunes National Recreation Area. Our inset (upper left) shows the Humboldt marten range (grey) and study area location where the inset on the lower right depicts an example of the prey grid locations amongst the 4 vegetation types.

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from [30]). We performed 3 PCR replicates per scat with Qiagen Multiplex PCR Kit (Qiagen, USA). Each reaction was amplified with identical unique 8 base pair tags on the 5' end of the forward and reverse primer to identify individual scat samples after pooling and to prevent false assignment due to tag jumping [31]. PCR reactions were carried out in a volume of 20 μL using the following reagent mixtures: 10 μL Qiagen Multiplex PCR Master Mix (Qiagen, USA), 0.4 μL of each primer for a final primer concentration of 200 nM, 0.2 μL bovine serum albumin (BSA), 6.2 μL of water, and 1 μL final DNA extract elution (including extraction blanks and PCR no template controls). Following a 15 minutes initial denaturation at 95˚C, the cycling conditions were: 40 cycles of 94˚C for 30 seconds, 58˚C for 90 seconds, 72˚C for 90 seconds, and a final extension at 72˚C for 10 minutes. We normalized and pooled the PCR products and used NEBNext Ultra II Library Prep Kit (New England BioLabs, USA) to adapt the library pools into Illumina sequencing libraries (Illumina Inc, USA). Libraries were purified using the Silica Bead method (Aline Biosciences, USA) and sent for 150 bp paired-end sequencing on the Illumina HiSeq 3000 at the Center for Genome Research and Biocomputing, Oregon State University. The raw reads were paired using *PEAR* [32] and demultiplexed based on the unique 8bp-index sequences using a custom shell script. Unique reads from each sample replicate were counted and taxonomically assigned using BLAST (www.ncbi.nlm.nih.gov/blast), against all 12S vertebrate sequences in Genbank and against a custom 12S library created for vertebrates not present in Genbank (S2 Document). We used the negative controls to set filtering read thresholds and assigned species if present in at least 2 out of the 3 replicates.

We manually sorted a random subset of scats from each season (~40%) to determine the presence of seeds and invertebrates. Each scat was soaked over a fine mesh, dried and then mechanically sorted using a dissecting microscope. We identified seeds to species and invertebrates to order. We recorded all dietary items using relative frequency of occurrence (number of occurrences of each species/number scats) by season.

Relative abundance of potential prey species

To inform availability of marten prey, we quantified an index of relative prey abundance using grids of remote cameras randomly placed within each of our 4 vegetation types during October to December 2015 (Fig 1). We defined prey as vertebrates that we detected in marten diet, or vertebrates morphologically similar to prey previously recorded in marten diet studies [33] and mountain beaver (*Aplodontia rufa*), which we included as potential prey due to its locally endemic distribution and body mass within the range of marten prey items [34]. We followed previous research that used track plates [35] to estimate an index of small mammal abundance, but used remote cameras rather than tracks to identify occurrence. We recorded occurrence of vertebrates within prey camera grids consisting of 20 remote cameras (Bushnell Aggressor, model 119776, Overland Park, MI) spaced 30 m apart in 2 parallel transects. A trap spacing of 20 m has been recommended for cricetid rodents for mark-recapture density estimates [36]. However, we chose 30 m to balance detections of cricetid rodents with species that have larger home ranges (e.g. sciurids). We placed cameras 20 cm high, 1.5 m from bait. Bait consisted of

15 grams each of sunflower seeds, peanut butter, oats, commercial rabbit food and strawberry jam placed within a pocket (20 x 10 cm) of fine metal mesh nailed to the ground. The mesh ensured that bait was released slowly such that it was present for the duration of camera deployment. Remote cameras monitored bait for 7 to 15 consecutive days and were programmed to take 3 photos per trigger with 1 min delay between triggers. We defined our index of relative abundance as number of cameras detecting a species on a camera grid within a 7-day period for all taxa, except deer mouse (*Peromyscus maniculatus*), which was assessed within a two-day window to avoid detection saturation at all cameras. Our purpose here was to estimate relative abundances within species among 4 vegetation types. We also note that our approach cannot be used to compare relative abundances across species, due to differences in movement, foraging, and spacing patterns that influence probabilities of detection.

For each camera, we assigned a 1 if a species was detected at least once and 0 if it was not detected within 7 days (or 2 days for deer mice). The number of cameras per grid detecting a given species relative to the number of functional cameras (20, unless a camera malfunctioned) was treated as a binomial response variable (K successes in N trials) in a generalized linear mixed-effects model (R function *glmer* from the lme4 package [37]). We included a random effect at the grid level to account for overdispersion as recommended by [38]. For each species, we included a factor level for each vegetation type and used Tukey's honest significant difference (HSD) to test for differences in relative abundance across all pairwise combinations of vegetation types (see R code in S3 Document). We excluded species detected in fewer than 3 grids due to inadequate sample sizes.

We examined trends in species diversity among vegetation types by calculating the reciprocal of Simpson's index per grid and averaging it across vegetation types [39]. We assessed the difference in diversity in 3 categories: total vertebrate diversity (all vertebrate species detected), potential marten prey diversity (small mammal and bird species), and carnivore species diversity. As voles and shrews were not identified to species, diversity indices for all vertebrate species and prey species are potentially biased lower.

Relative abundance of competitors and predators

We estimated an index of abundance of carnivores, which we assumed could be predators or competitors of martens. Because carnivore home ranges (> 2 km^2) could readily overlap the beach grass, seasonally-flooded shore pine forest, and coastal shrub forest vegetation types, we combined them into "coastal forests". Thus, for our carnivore analysis we compared coastal forests with interior forest. We used 4 camera data sources to create our carnivore abundance index: 1) We paired 2 camera stations with each prey grid > 75 m away and separated by > 50 m following the protocol in [29]. We baited 1 station with chicken and an olfactory lure, hereafter a "baited" station (Gusto, Minnesota Trapline Products, USA), and 1 was left unbaited on a trail, hereafter "unbaited trail" station. Cameras ($n = 40$ baited, 40 unbaited trail) were deployed for 7–15 days. 2) We used data within our study area from [29] ($n = 79$ baited, 50 unbaited trail cameras) deployed May–October 2015. These cameras were placed in areas 5–50 km from historic marten detections using a stratified random sampling design to ensure even distribution of samples in early to mature forest age classes. We deployed cameras for a minimum of 21 days, baited with Gusto and chicken or cat food (for more details see [29]. 3) We used camera data from our marten surveys in the coastal forests ($n = 9$ baited, 10 trail), which were deployed for an average of 35 days October 2015 –February 2016, and were baited with chicken, cat food, and strawberry jam [13]. 4) Because our camera grids designed to identify prey frequently also detected carnivores, we pooled observations from all cameras on the grid, which we then treated as a single baited camera station ($n = 31$ camera grids, 620 camera

stations). Combined, we included data from 259 sample units (n = 843 camera stations including the camera stations on the grids). For analyses, our sample sizes were similar between the interior (n = 86 baited, 55 unbaited trail camera stations) and coastal forests (n = 74 baited, 44 unbaited trail) May 2015 –October 2016 (Fig 1). We excluded carnivore species that were detected with fewer than 5 cameras from further analysis due to inadequate sample sizes.

We used the camera trapping rate as an index of abundance, but to avoid over-weighting repeat visits within a short time interval, we treated each day, rather than photo, with a species detection as an encounter out of N total potential encounters, where N was the number of sampling days. For each carnivore or potential competitor, we used a binomial generalized linear mixed-effects model (R function *glmer* from the lme4 package; [37]) with the number of days detecting a species relative to the number of monitoring days per camera as the binomial dependent variable. We included forest (coastal vs interior) as a covariate to test for differences in relative abundance between the forests, and a random effect of sample unit. To account for differences in probability of detection by camera set type, we included binary factors (fixed effects) for unbaited trail camera vs. baited camera, and prey camera grid vs. single camera. By accounting for the assumed higher probability of detection on the prey camera grid, or when using bait, we were able to incorporate all available data in our analyses. If we found significant differences in the probability of carnivore detection between the coastal forests and interior forest, we assessed whether significance (p-value) was inflated by pseudoreplication due to spatial autocorrelation using a Moran's I test on the residuals [40]. If the Moran's I test was significant, we included a spatial lag autocovariate term in the model and report the Moran's I p-value in the more inclusive model [40].

All camera surveys followed protocols approved by the USDA Forest Service, Pacific Northwest Research Station, and surveys were conducted under Oregon Department of Fish and Wildlife Scientific Take Permits (119–15, 128–16).

Vegetation surveys

To compare vegetation among our 4 vegetation types, we performed ocular vegetation surveys at each camera station within the prey camera grids. We estimated percentage cover of the canopy, shrub and ground layers, shrub height (m), tree height (m), and tree diameter (cm) at breast height within a 10 m radius circular plot centered on the camera. All vegetation surveys were performed by 1 observer, and we calculated averages per vegetation community using Kruskal-Wallis tests and adjusted for multiple comparisons using the False Discovery Rate method [41]. We estimated approximate forest age of the camera grids using gradient nearest-neighbor maps (2012 version accessed from: https://lemma.forestry.oregonstate.edu/data). Gradient nearest-neighbor techniques combine data from on the ground vegetation plots from the Forest Inventory and Analysis National Program with remotely sensed spectral data to infer forest quality and characteristics in areas without vegetation plots [42].

Results

Diet analysis

We collected 108 scats during 3 seasons, of which 98 were genetically confirmed to be from marten, 5 gray fox, 1 raccoon, 1 bobcat and 3 scat samples contained only prey DNA. Four marten scats were located within the interior forest approximately 200–300 m east of the coastal forests where the remainder of the scats were found.

Eight marten scats yielded marten DNA exclusively, leaving 90 scats for diet analysis. We identified 25 vertebrate taxa (Table 1), 3 plant species, and 5 insect orders. Martens fed extensively on berries during fall (100% of scats) and winter (86% of scats) (Table 2). Invertebrates

Table 1. Percent occurrence of vertebrate food items in spring, fall and winter of Humboldt martens (*Martes caurina humboldtensis*) in the central Oregon coast from April 2015—March 2016. Data were based on 90 genetically confirmed marten scats collected using scent detection dog surveys (n = 30), live-trapping and from known marten rest structures (n = 60). The samples were grouped per season (spring: March–May 34 scats, fall: September–November 31 scats, and winter: December–February 25 scats). We quantified vertebrate diet composition using DNA metabarcoding.

		Spring (%)	Fall (%)	Winter (%)	Total (%)
Prey		**(n = 34)**	**(n = 31)**	**(n = 25)**	**(n = 90)**
Sample size		34	31	25	90
Birds		38.2	41.9	88	53.3
Corvidae	Steller's jay (*Cyanocitta stelleri*)	2.9	0	0	1.1
Parulidae	Yellow-rumped warbler (*Setophaga coronata*)	0	3.2	8	3.3
Phasianidae	Ruffed grouse (*Bonasa umbellus*)	2.9	0	0	1.1
Picidae	Northern flicker (*Colaptes auratus*)	0	0	8	2.2
Rallidae	American coot (*Fulica americana*)	0	3.2	0	1.1
Regulidae	Golden-crowned kinglet (*Regulus satrapa*)	0	6.5	12	5.6
Sylviidae	Wrentit (*Chamaea fasciata*)	5.9	6.5	0	4.4
Throchilidae	Rufous hummingbird (*Selasphorus rufus*)	2.9	0	0	1.1
Troglodytidae	Pacific wren (*Troglodytes pacificus*)	0	0	4	1.1
Turdidae	American robin (*Turdus migratorius*)	0	0	8	2.2
	Swainson's thrush (*Catharus ustulatus*)	0	0	4	1.1
	Varied thrush (*Ixoreus naevius*)	26.5	35.5	84	45.6
Mammals		70.6	87.1	80	80
Cervidae	Black-tailed deer *(Odocoileus hemionus)*	0	35.5	0	12.2
Cricetidae	White-footed vole (*Arborimus albipes*)	11.8	22.6	4	13.3
	Red tree vole (*Arborimus longicaudus*)	29.4	0	0	11.1
	Western red-backed vole (*Myodes californicus*)	26.5	22.6	52	32.2
	Creeping vole (*Microtus oregoni*)	0	3.2	20	6.7
	Deer mouse (*Peromyscus maniculatus*)	3.4	25.8	28	17.8
Leporidae	Brush rabbit (*Sylvilagus bachmani*)	0	0	4	1.1
Sciuridae	Townsend's chipmunk (*Tamias townsendii*)	0	3.2	0	2.2
	Douglas squirrel (*Tamiasciurus douglasii*)	11.8	0	0	4.4
Soricidae	Trowbridge's shrew (*Sorex trowbridgii*)	5.9	0	0	2.2
Amphibians		8.9	3.2	4	5.6
Ambystomatidae	Coastal giant salamander (*Dicamptodon tenebrosus*)	2.9	0	0	1.1
Hylidae	Pacific chorus frog (*Pseudacris regilla*)	0	3.2	4	2.2
Plethodontidae	Wandering salamander (*Aneides vagrans*)	5.9	0	0	2.2

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were eaten in small amounts across all seasons and comprised mainly of beetles (Coleoptera), bees and ants (Hymenoptera), but were not the dominant component of any scats (Table 2). Mammals featured consistently in the diet across spring, fall and winter with voles being the most frequent prey group (55%; Table 1). Western red-backed vole (*Myodes californicus*) and deer mouse were common food items (32.2% and 17.8%, respectively). Eleven percent of the scats contained red tree vole (*Arborimus longicaudus*), a strictly arboreal species. In addition, white-footed vole (*Arborimus albipes*), a little known arboreal species [43], occurred in 13.3%

Table 2. Percent occurrence of berries and invertebrates from a subsample of Humboldt marten (*Martes caurina humboldtensis*) scats collected in the central Oregon coast April 2015 –March 2016. We manually sorted a random subset of marten scats (n = 35) and examined contents using a dissecting microscope because we were unable to detect berries and invertebrates using our vertebrate primers during DNA metabarcoding. We grouped samples by season (spring: March–May, fall: September–November, and winter: December–February).

Food group		Spring (%) n = 11	Fall (%) n = 10	Winter (%) n = 14	Total (%) n = 35
Fruit		0	100	85.7	62.9
Ericaceae	Evergreen huckleberry *(Vaccinium ovatum)*	0	90	85.7	54.5
Rosaceae	*Rubus* spp.	0	20	53.8	50
Unknown spp.		0	0	35.7	35.7
Invertebrates		18.2	45.5	50	38.9
	Coleoptera	9	9	14	18.2
	Diptera	0	0	7	4.5
	Hymenoptera	0	9	28.6	22.7
	Oribatida	0	9	0	4.5
	Trichoptera	0	9	0	4.5
	Unknown insect	9	9	0	9

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of the scats. We documented birds more frequently in scats from winter (88.0%) compared to fall (41.9%; β = -3.24, $P < 0.01$) and spring (38.2%; β = -3.48, $P < 0.01$). Birds occurred on average in 53.3% of marten scats with varied thrush being the overall most frequently consumed species (45.6%) (Table 1).

Relative abundance indices–potential prey

We deployed 620 camera stations across 31 grids in 4 vegetation types to quantify relative prey abundance (interior forest n = 12 grids, coastal shrub forest n = 7 grids, seasonally-flooded shore pine forest n = 8 grids, and beach grass n = 4 grids) (Fig 1). Two grids had 1 camera malfunction each, and 1 grid had 3, leaving 615 camera stations. We detected 34 potential prey species (16 small mammal and 18 bird species). Because shrews and voles were only identified to genus (*Sorex*) and subfamily (*Arvicolinae*), respectively, the actual number of small mammal species is likely higher because there are 5 shrew and 6 vole species present in this region [34]. Deer mice were documented at every grid across all vegetation types but were less abundant in interior forest compared to coastal shrub forest (β = -4.24, $P < 0.001$). Similarly, voles were observed in all vegetation types but were less abundant in interior forest compared to other vegetation types (β = -2.94, P = 0.02). In contrast, mountain beaver and California ground squirrel (*Otospermophilus beecheyi*) were only detected in interior forest and beach grass, respectively. The other squirrel species were observed in all forested vegetation types with Townsend's chipmunk (*Tamias townsendii*) being less abundant in the interior forest (β = -2.99, P = 0.01) (Fig 2).

The most commonly detected bird species were varied thrush, fox sparrow (*Passerella iliaca*), song sparrow (*Melospiza melodia*), and spotted towhee (*Pipilo maculatus*, Fig 3). Five out of 11 bird species were detected significantly more in the coastal shrub forest or seasonally-flooded shore pine forest compared to interior forest (Fig 3). See S1 Table for a complete list of all potential prey species detected and S1 Fig for photograph examples.

Relative abundance indices–potential predators and competitors

Composition and relative abundance of carnivore species varied between coastal forests and interior forest (Fig 4). Martens (β = -5.93, $P < 0.001$), raccoons (*Procyon lotor*) (β = -2.52,

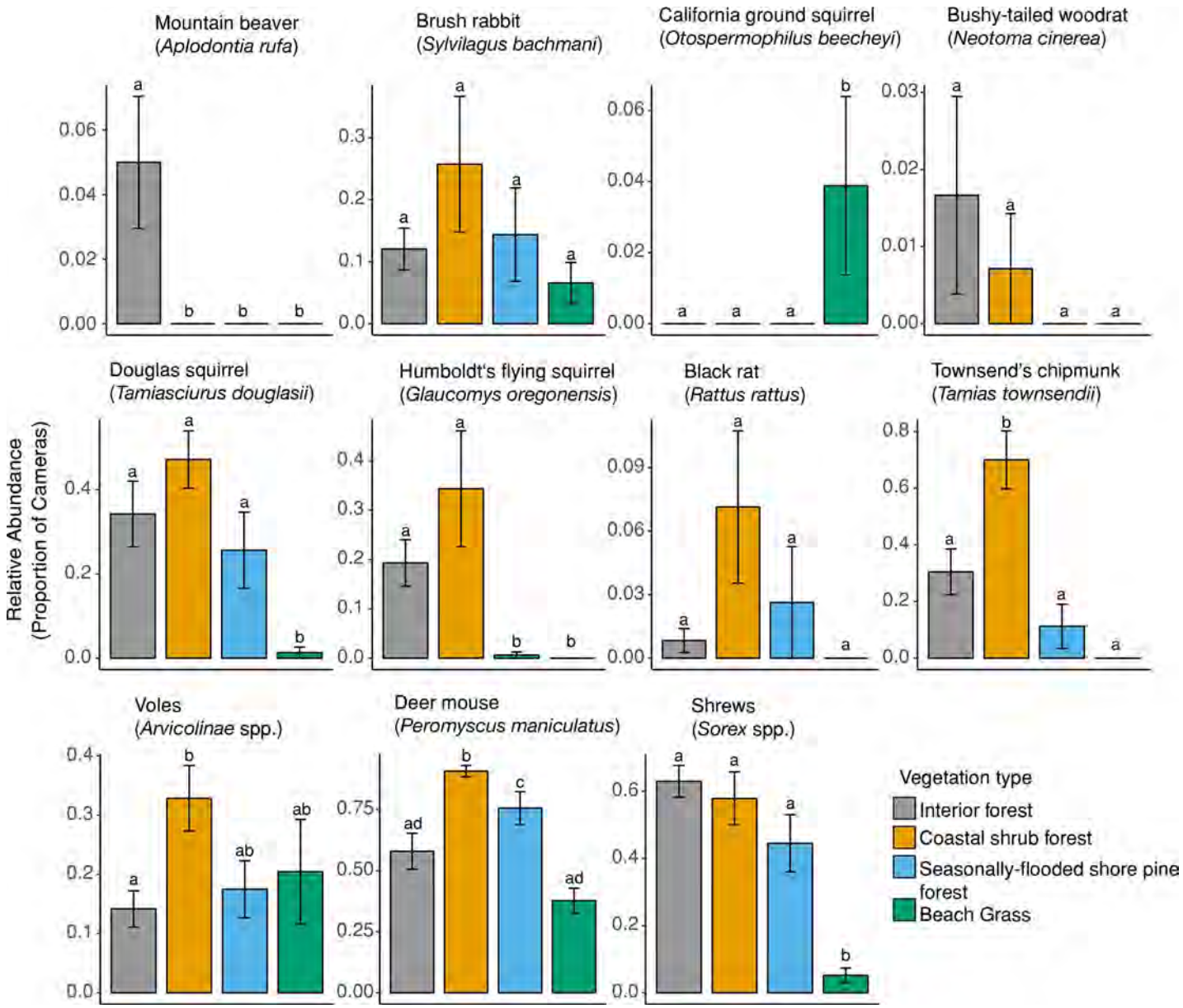

Fig 2. Relative abundance ($\bar{x} \pm SE$) of small mammals by vegetation type in the central Oregon coast during October–December 2015. We defined relative abundance as the proportion of cameras that detected a species per vegetation community. We deployed 620 camera stations across 31 grids in 4 vegetation types (interior forest $n = 12$ grids, coastal shrub forest $n = 7$ grids, seasonally-flooded shore pine forest $n = 8$ grids, and beach grass $n = 4$ grids). Species plots are arranged based on average body mass from heavier to lighter (left to right). Different letters denote significant difference ($P \leq 0.05$) among vegetation types after Tukey's HSD adjustment for multiple comparisons. Plots without any letters indicate that there was no significant difference between any vegetation types.

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$P < 0.001$), and gray foxes (*Urocyon cinereoargenteus*) (β = -4.47, $P < 0.001$) had significantly fewer detections in interior forest compared to coastal forests. In contrast, bobcats and western spotted skunks had significantly higher detections in the interior forest ($\beta = 2.29$, $P < 0.001$) and ($\beta = 2.24$, $P < 0.001$), respectively (Fig 4; Table 3). The prey camera setup with 20 cameras per grid significantly increased detection rates relative to the baited and unbaited trail cameras for all carnivore species except martens, long-tailed weasels (*Mustela frenata*), and cougars (*Puma concolor*) (Table 3). Unbaited trail cameras had significantly higher detection rates of

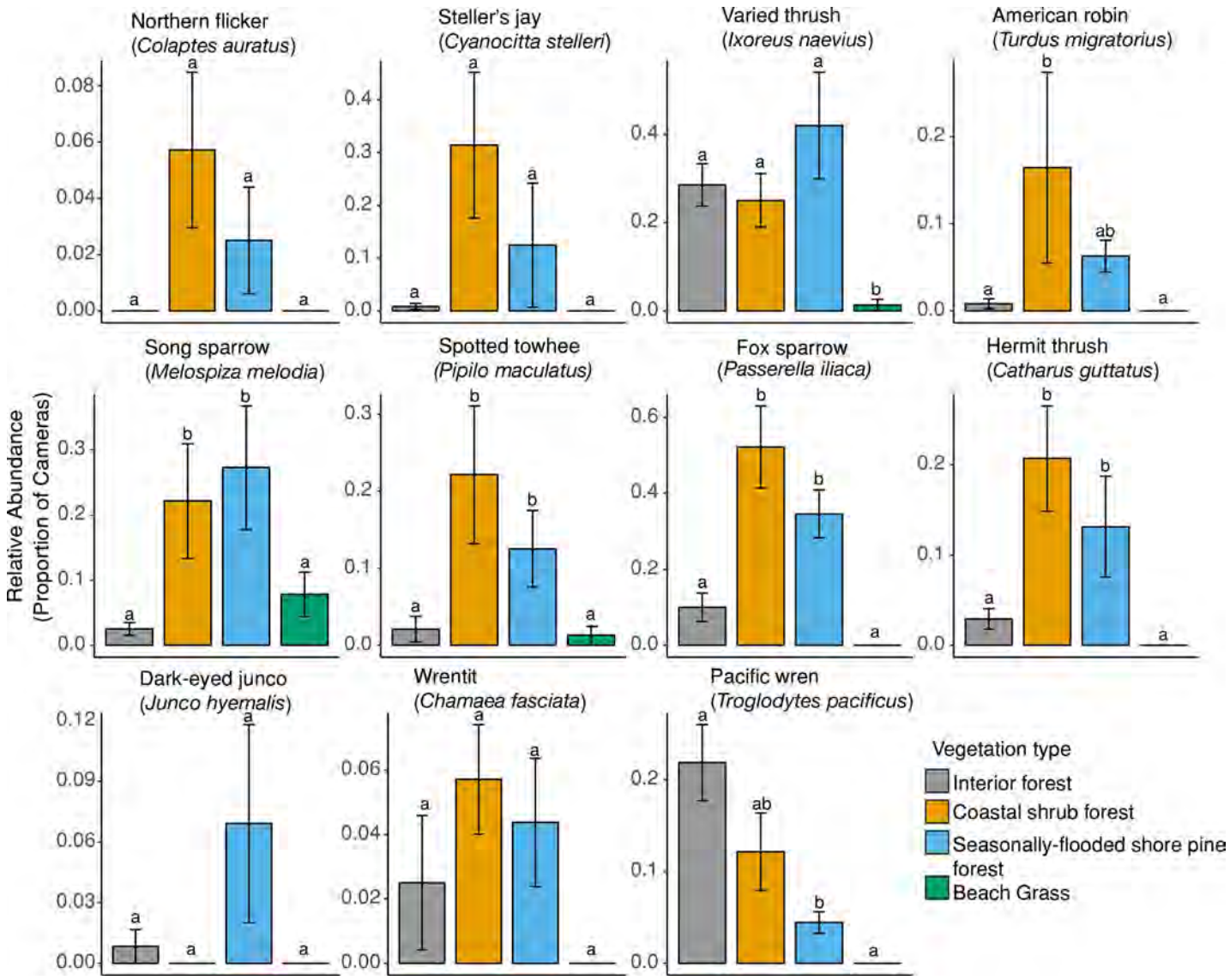

Fig 3. Relative abundance ($\bar{x} \pm SE$) of birds per vegetation type in the central Oregon coast during October–December 2015. We defined relative abundance as the proportion of cameras that detected a species per vegetation type and calculated standard error of the difference between the vegetation types. We deployed 620 camera stations across 31 grids with 20 cameras each (interior forest $n = 12$ grids, coastal shrub forest $n = 7$ grids, seasonally-flooded shore pine forest $n = 8$ grids, and beach grass $n = 4$ grids). Species plots are arranged based on average body mass from heavier to lighter (left to right). Different letters denote significant difference ($P \leq 0.05$) among vegetation types after Tukey's HSD adjustment for multiple comparisons. Plots without any letter indicate that there was no significant difference between any vegetation types.

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bobcats ($\beta = 1.12$, $P < 0.001$) compared to baited cameras and we observed the opposite relationship for western spotted skunks ($\beta = -0.38$, $P < 0.001$) (Table 3). Among the species with detection rates that significantly differed between the vegetation types, we found spatial autocorrelation for bobcats (Moran's I test, $P < 0.001$), cougars ($P = 0.03$), raccoons ($P < 0.001$) and western spotted skunks ($P < 0.001$) which we then accounted for in our models. After adding a spatial lag autocovariate, no additional spatial autocorrelation was found for bobcats (Moran's I test, $P = 0.53$), cougars ($P = 0.80$), raccoons ($P = 0.73$), or western spotted skunks ($P = 0.08$) (Table 3). All species detected are included in S1 Table.

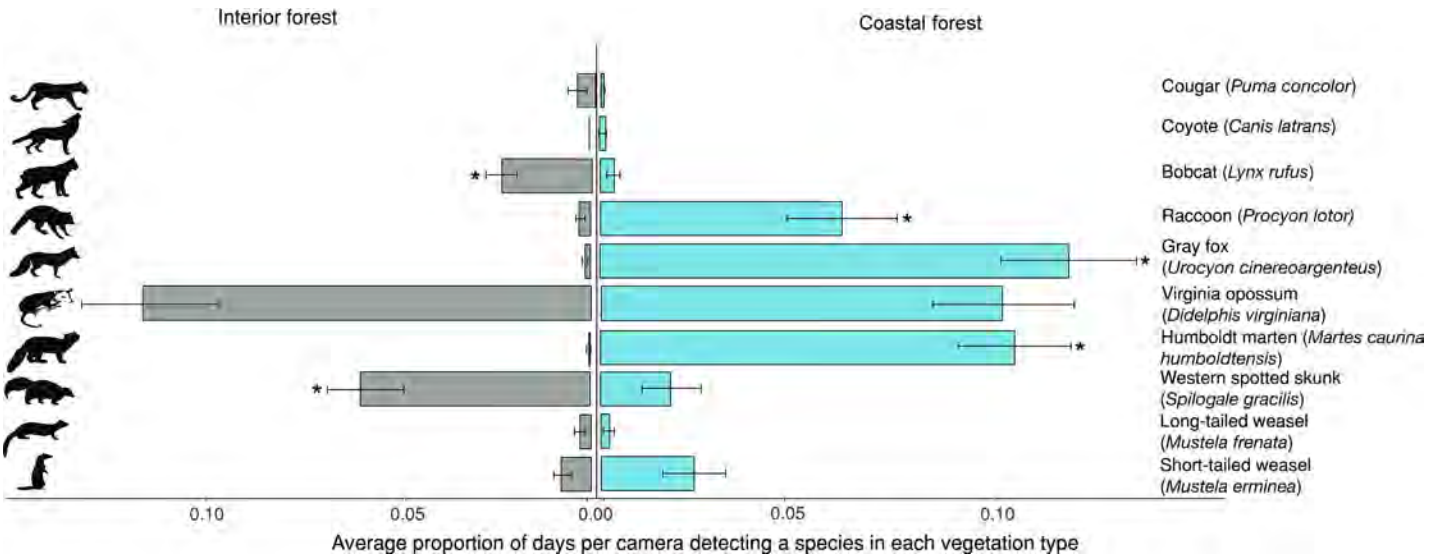

Fig 4. Relative abundance defined as the average proportion of days per camera detecting a species ($\bar{x} \pm SE$) of carnivores and potential competitors in the interior forest (on the left) and coastal forests (on the right) during an investigation of Humboldt marten (*Martes caurina humboldtensis*) occurrence in the central Oregon coast. We used 843 cameras combined into 259 spatially distinct sample units (n = 160 baited sample units, including the 31 prey camera grids with 20 cameras each) and 99 unbaited camera stations). Species bars are arranged from heavier to lighter species (top to bottom). Stars indicate significant difference in relative abundance between the interior forest and the coastal forests.

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Vertebrate diversity was significantly lower in interior forest compared to the coastal shrub forest (β = -3.85, P = <0.0001) (Fig 5A). Similarly, diversity of potential prey species was significantly lower in interior forest compared to the coastal shrub forest (β = -2.91, P = 0.0002) but similar to seasonally-flooded shore pine forest (β = -0.07, P = 0.999). The diversity of carnivore species was higher in coastal forests compared to interior forest (β = 0.634, P < .0001) (Fig 5C).

Vegetation types

Vegetation composition and cover varied among our 4 vegetation types (Fig 6). Estimated percent total shrub cover and fruit-producing shrub cover increased from beach grass to seasonally-flooded shore pine forest and reached its maximum in coastal shrub forest. Shrub cover in coastal shrub forest was significantly higher than in the interior forest (Fig 6A).

Estimated shrub height was similar between the interior forest and seasonally-flooded shore pine forest and tallest in coastal shrub forest (Fig 6D). Our estimates of canopy cover were similar between the interior and seasonally-flooded forest but lower in the coastal shrub forest (Fig 6A). Estimated average tree height (Fig 6B) and diameter (Fig 6C) were the highest in the interior forest. Understory vegetation within the 4 vegetation types were highly distinct with a few species dominating. The beach grass vegetation type consisted mainly of European beach grass (*Ammophila arenaria*) while the seasonally-flooded shore pine forest was dominated by slough sedge (*Carex obnupta*) within the understory. The interior forest understory was slightly more variable but consisted often of sword fern (*Polystichum munitum*). Dense shrub cover in the coastal shrub forest permitted little sun to reach the ground and was almost devoid of ground cover (Fig 6A). Approximate forest age of our randomly selected grids varied widely within vegetation types. The remotely sensed database suggested the seasonally-flooded shore pine community ranged 15–62 years old (n = 8, 37.2 ± 6.4, average ± SE), coastal shrub forest 33–70 years old (n = 7, 53 ± 5.7) and interior forest 23–194 years old (n = 12, 80.7 ± 17).

Table 3. We evaluated potential predator or competitor detections, defined as the proportion of days detecting a species, in relation to two vegetation classes during a Humboldt marten (*Martes caurina humboldtensis*) study in central Oregon.

Species	Estimate	Standard error	p-value
Bobcat (*Lynx rufus*)*			
Interior forest	2.296	0.632	<0.001
Unbaited trail camera	1.118	0.307	<0.001
Prey camera setup	-0.251	0.824	0.761
Spatial lag variable	-0.704	0.221	0.001
Moran's I p-value	0.532		
Cougar (*Puma concolor*)*			
Interior forest	1.168	1.73	0.499
Unbaited trail camera	0.523	0.698	0.454
Prey camera setup	3.447	1.292	0.007
Moran's I p-value	0.80		
Humboldt marten (*Martes caurina humboldtensis*)			
Interior forest	-5.928	1.138	<0.001
Unbaited trail camera	-1.021	0.221	<0.001
Prey camera setup	0.065	0.32	0.839
Gray fox (*Urocyon cinereoargenteus*)*			
Interior forest	-4.471	0.737	<0.001
Unbaited trail camera	-0.313	0.215	0.145
Prey camera setup	1.266	0.317	<0.001
Spatial lag variable	-0.449	0.14	0.001
Moran's I p-value	0.212		
Long-tailed weasel (*Mustela frenata*)			
Interior forest	-0.534	2.239	0.812
Unbaited trail camera	-0.246	0.952	0.796
Prey camera setup	0.04	1.023	0.969
Raccoon (*Procyon lotor*)*			
Interior forest	-2.522	0.584	<0.001
Unbaited trail camera	0.301	0.313	0.336
Prey camera setup	2.288	0.411	<0.001
Spatial lag variable	-0.777	0.178	<0.001
Moran's I p-value	0.732		
Short-tailed weasel (*Mustela erminea*)			
Interior forest	-0.309	0.532	0.56
Unbaited trail camera	-0.42	0.614	0.494
Prey camera setup	3.249	0.509	<0.001
Virginia opossum (*Didelphis virginiana*)			
Interior forest	-0.009	0.522	0.986
Unbaited trail camera	-0.012	0.142	0.933
Prey camera setup	1.442	0.236	<0.001
Western spotted skunk (*Spilogale gracilis*)*			
Interior forest	2.238	0.521	<0.001
Unbaited trail camera	-0.384	0.174	0.027
Prey camera setup	1.636	0.358	<0.001
Spatial lag variable	-0.294	0.049	<0.001

(*Continued*)

Table 3. (Continued)

Species	Estimate	Standard error	p-value
Moran's I p-value	0.082		

We used generalized linear mixed-effects models to examine the influence of vegetation type (interior forest relative to coastal forests), camera type (unbaited trail cameras compared to baited camera), and survey type (prey camera setup versus carnivore camera setup) for each species. If we found significant difference in abundance between the 2 vegetation types, we assessed whether *p*-values were inflated by pseudoreplication due to spatial autocorrelation using a Moran's I test on the residuals. If present, spatial autocorrelation (indicated by *) was accounted for by adding a spatial lag variable to the model and we report our Moran's I test on the residuals.

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Discussion

Although Humboldt martens in northern California have been associated with mature forests [13], our data support the conclusion of previous distribution surveys that suggest Humboldt martens in the central Oregon Coast Range currently appear to be restricted to a narrow band of young coastal forest (< 70 years) on the Oregon Dunes rather than the adjacent, older, interior forest [11]. Not only are martens seemingly absent from the historically occupied interior forest [11,44], but density estimates from the small area of adjacent young coastal forest are higher than all documented North American marten populations [13]. To explore the cause of this puzzling distribution we quantified marten diet and tested a series of plausible hypotheses. We found that 1) food availability (small vertebrate and fruit resources) was greater in the young coastal forests, 2) that potential predators, particularly bobcats, were less common in the coastal forests, 3) potential competitors were more common in coastal forests compared to interior forests, and 4) vegetation in the coastal forests provided more fruit and overhead shrub cover. This rapid assessment of potential mechanisms responsible for the apparent restricted distribution of martens was facilitated by a novel use of camera traps on replicated grids to produce an index of prey abundance and DNA metabarcoding to provide fine taxonomic resolution of marten diet.

Marten diet

Marten diets were highly diverse but consisted primarily of 3 main categories: voles, birds and berries. The dominant consumption of voles was consistent with North American marten diet studies [33,45,46] but we detected substantially lower frequency of occurrence of squirrel species [45,47] and shrews [48]. Humboldt marten diet in northern California has been shown to heavily rely on chipmunks [49] which contrasts our study, where chipmunks were only detected in 2% of the marten scats despite being highly abundant in the coastal forests (Fig 2). Frequency of birds in the diet (53.3%) was much higher than the 10–20% typically described in North American martens' diet [33]. Diet diversification [50] and increased bird consumption [51] has been suggested to result from low availability of voles; however, voles were abundant in the coastal forests and were consumed frequently throughout the year. High consumption of birds in this study may result from mild temperatures and seasonally high occurrence of fruits, which in combination support an abundance of resident and migratory birds as a prey resource for martens through the fall and winter. Alternatively, high bird consumption may result from increased hunting success due to structurally complex understory vegetation [26]. Dense ericaceous shrub cover in coastal forests may attract frugivorous birds by providing food and cover from avian predators. Yet, nearly impenetrable shrub layer may also reduce the ability of birds to detect predators such as martens that are actively foraging underneath the

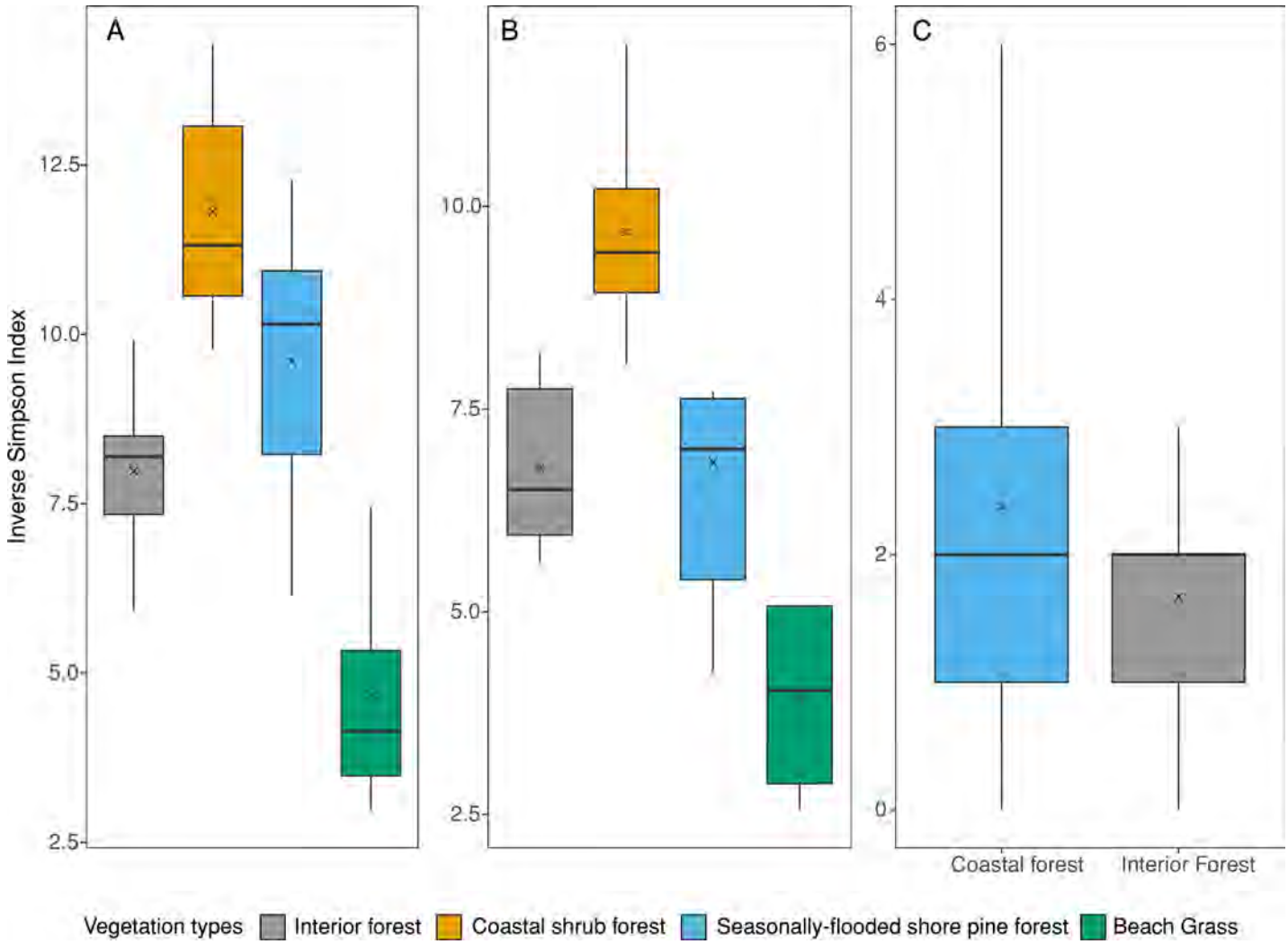

Fig 5. Boxplots of inverse Simpson diversity index for A) all vertebrate species, B) potential Humboldt marten (*Martes caurina humboldtensis*) prey species (small mammals and birds) and C) carnivores and potential competitors, detected with camera traps across 4 vegetation types in coastal Oregon. The black lines denote the medians, *x* the average, and boxes the 25% and 50% quartiles.

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shrubs and limiting vertical escape routes [52]. In addition, ericaceous shrubs provide fruit that are readily consumed by martens, a result (also observed using stable isotope assessment of diet [53]), potentially allowing them to opportunistically prey on vertebrates while foraging for fruit. Diet of North American marten populations appear to vary [49,53], suggesting generalizing across populations may not be valid even within this subspecies.

This was the first study on the diet of a *Martes* species using DNA metabarcoding, which enabled us to present a taxonomic precision previously not achievable using mechanical sorting that earlier studies have relied on [47–49] but see [54]. Avian resolution in the diet was previously restricted to 'birds' [33,47] or 'small birds' versus 'large birds' [48,49]. Here, we were able to demonstrate a broad variety of passerines consumed. Importantly, shorebirds including the Federally Threatened snowy plover (*Charadrius nivosus*) were not identified as consumed, tentatively suggesting martens were not a threat to this species which is associated with open beaches (see also [53]).

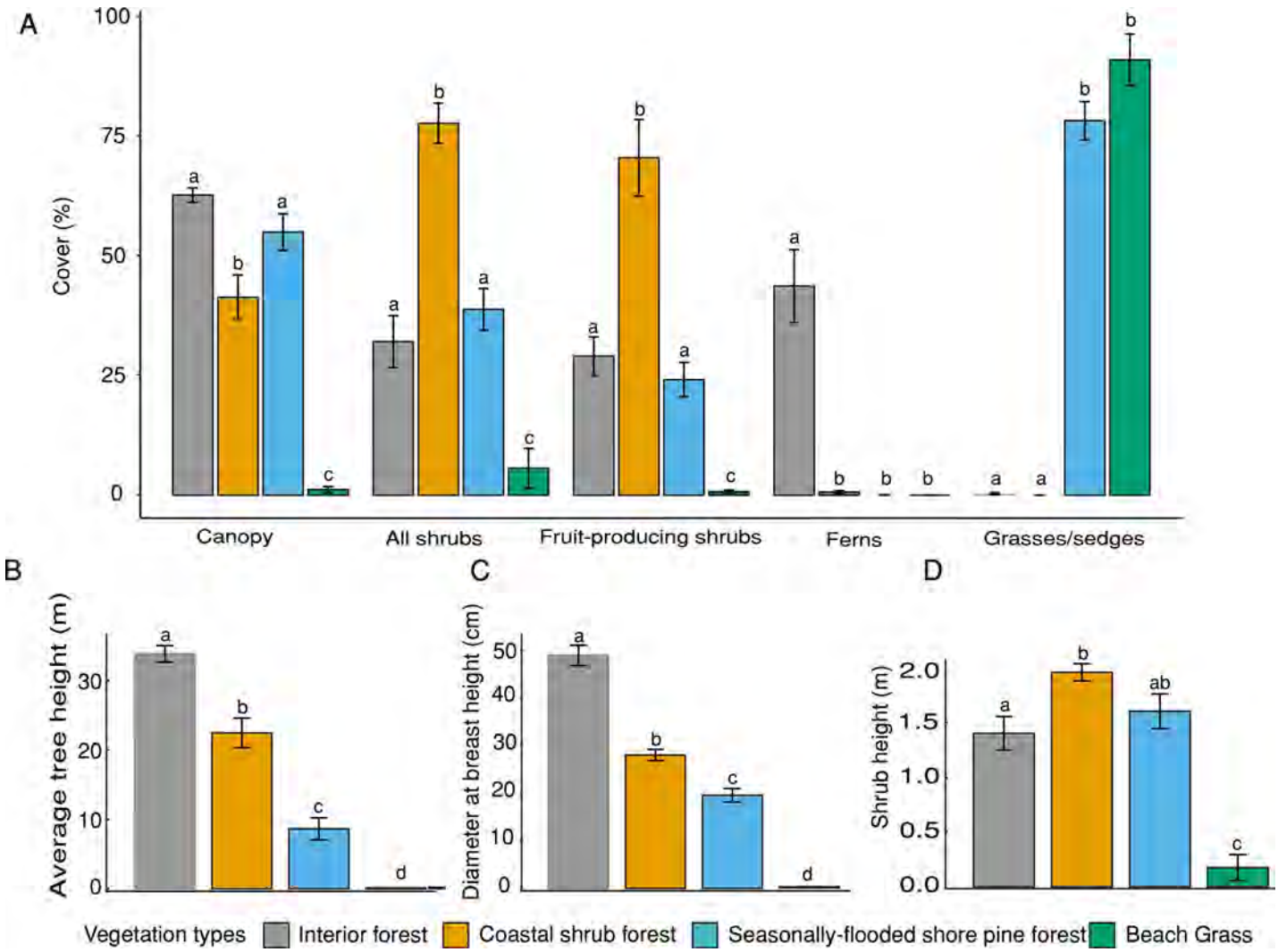

Fig 6. Vegetation data was collected at each prey camera station, averaged across camera stations within a grid and averaged between grids in different vegetation types (interior forest *n* = 12 grids, coastal shrub forest *n* = 7 grids, seasonally-flooded shore pine forest *n* = 8 grids, and beach grass *n* = 4 grids). We depict percent cover ($\bar{x} \pm SE$) of the canopy, total shrub and fruit-producing shrub cover (A), B) average tree height, C) average diameter at breast height, and D) average shrub height. Different letters denote significant difference among the vegetation types after Tukey's HSD adjustment for multiple comparisons.

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Prey abundance

We found evidence of increased food resources in the coastal shrub forest and the seasonally-flooded shore pine forest compared to the interior forest. In particular, voles, deer mice and fruit, which were all highly represented in marten diets, were significantly more abundant in coastal shrub forests. While previous studies have found a positive or neutral association between small mammals and mature forests [55–57], our findings suggest that younger coastal forests had a higher relative abundance of several small mammal species (Fig 2). Increased forest age and small mammal abundance has been positively associated with fungi and fruit, known food sources for many small mammals [58,59]. During our study, the coastal shrub forest contained more fruit-producing shrubs, and although not investigated here, potentially increased fungi. Ashkannejhad and Horton [59] found a higher diversity of fungi in coastal forests compared to interior forest, including species commonly consumed by small mammals

[55,60]. In addition, highly abundant ericaceous species present in coastal forests such as salal, Pacific rhododendron (*Rhododendron macrophyllum*) and huckleberries (*Vaccinium* spp), host a range of ectomycorrhizal fungi [61] potentially supplying small mammals with fungi in addition to fruit compared to interior forest. Thus, mechanisms for increased small mammal abundance in our study may be correlated with several factors: 1) abundant fungi present in coastal forests, 2) small mammal species in our study area were supported by fruits and seeds from high relative abundance of fruit-producing shrubs, or 3) these small mammal species are top-down limited, and predator protection in the form of a dense shrub layer has resulted in increased prey abundance.

Bird abundance was higher in coastal forests compared to interior forest. Fox sparrows, song sparrows, spotted towhees, hermit thrushes (*Catharus guttatus*) and Steller's jays (*Cyanocitta stelleri*) were significantly more abundant in the coastal forests, but of these, only Steller's jays were consumed by martens during our study (Table 1). Varied thrush was the most frequently consumed prey species by marten and abundant in all forests. In addition, martens preyed upon several bird species that we failed to detect with our camera-based surveys such as yellow-rumped warblers (*Setophaga coronata*), American coot (*Fulica americana*), and golden-crowned kinglets (*Regulus satrapa*). Failure to detect these species with our cameras may reflect the bird's ecology (aquatic vs terrestrial habitat) or small body size, which could limit detectability [62]. It is likely that the fruit-producing shrubs (e.g., huckleberries, salal) attract frugivorous birds [63] which may explain the higher abundance of some birds in the coastal forests (Fig 3). Whatever the mechanism, our hypothesis of increased prey availability in the coastal forest was supported and remains a viable explanation for determining the observed distribution of martens.

Competition and predation

Gray foxes, raccoons, and western spotted skunks have an omnivorous diet and may include diet items similar to martens [34], potentially making these species competitors for nutritional resources. Our camera-based surveys revealed that gray foxes and raccoons were, like martens, detected almost exclusively in the coastal forests. This observation suggests that release from competition with these species was not a plausible hypothesis for why martens appear to be restricted to the coastal forests. Nonetheless, western spotted skunks, the most similar species in terms of body size, phylogeny, and small cavity utilization, were relatively rare in the coastal forests and common in the interior forest. It is thus plausible that marten face some degree of competition with spotted skunks in the interior forest, although unlikely because the extent of diet overlap is unknown, they have divergent diel activity patterns, spotted skunks are smaller-bodied, and there is no evidence of interference competition between these species. In addition, we could not dismiss the competitive release hypothesis as similar bodied avian competitors were not surveyed for but could provide substantial competition for martens in addition to being their direct predators [19].

Bobcats, an important mammalian predator of martens [19,64], were significantly more common in the interior forest. The extensive shrub cover in the coastal shrub forest is likely to facilitate escape from larger-bodied bobcats and provide overhead cover to avoid avian predation. However, martens were frequently detected in the seasonally-flooded shore pine forest, where sedges provide some understory structure, but shrub cover was not distinct from the interior forest. Although shrub cover was lower in seasonally-flooded shore pine forest, seasonal flooding may have contributed to the relative absence of bobcats and other competitors there, perhaps conferring some competitive advantage to the scansorial marten if they are able to travel through the canopy more readily when forests are flooded (Fig 6). Although shrubs

were not prevalent in all forests used by martens, other studies in the Pacific states have shown strong association with shrub cover and therefore further mechanistic studies examining whether shrubs provide protective cover may be warranted [12,65].

Rapid community assessment for imperiled species

The causal mechanisms contributing to the rarity of species are often challenging to discern, and manipulative experiments are often infeasible or unethical. In addition, when conservation measures are urgently needed, as is the case with Humboldt martens in Oregon, a framework for rapid identification of potential stressors limiting population growth and spread is warranted. The conservation of most species is hampered by a general lack of natural history and community ecology information. This includes limited information about distribution, abundance, and diet of species across trophic levels. For example, in our study area, we had no information on what martens were eating, relative abundance of their prey, potential predators and competitors across vegetation types, and we still lack basic information on avian predators and competitors, and cause-specific mortality of martens. In addition, we are deficient in basic information about the base of the food web. For instance, small mammals are a basal resource, but the degree to which each species is energetically supported by fungi, lichen, fruit, seeds, or insects is poorly known. The lack of information about seasonally important prey and drivers of prey abundance limits our ability to predict how differences in vegetation affect marten populations.

We benefited from new technologies, such as DNA metabarcoding, and the availability of affordable and efficient camera traps to broadly survey the vertebrate community. By using replicated and distinct camera survey methods, we provided insights into biases present when using only 1 set or bait type, especially if the goal is to broadly describe the vertebrate community. For instance, marten detections were strongly associated with stations baited with chicken and lure, whereas bobcats were primarily detected on unbaited trail cameras (Table 3). Gray foxes, raccoons, short-tailed weasels (*Mustela erminea*), opossums (*Didelphis virginiana*), and spotted skunks were detected more frequently at our small mammal grids baited with oats, seeds, nuts, and jams (Table 3) than at carnivore cameras. Our study was not designed to evaluate survey efficiency, but differing bait types and camera density per sample unit (i.e., 1 vs. 20 cameras in a grid) allowed us to have a higher probability of detecting a suite of vertebrate species.

Our approach to producing an index of abundance assumed that animals are detected more frequently when abundant. Although this is largely self-evident, if detectability of animals was highly variable across vegetation types for other reasons, such as behavioral changes that result in differential attraction to baits, then our inference about abundance may be biased. For example, if voles were consistently less willing to approach bait in the interior forest and trigger the camera, then our inference that they were less abundant there is not necessarily robust. Although unlikely, this is an important caveat, but more invasive methods to estimate abundance, such as mark-recapture, have their own limitations. In particular, such approaches are taxonomically limited to certain species, whereas cameras detect a suite of species including birds and mammals across a range of body sizes from small shrews to large-bodied carnivores. This taxonomic limitation is key when assessing the prey base of a species with such a diverse diet. In addition, mark-recapture approaches are time-consuming, and it is logistically challenging to implement high levels of replication on landscape scales. With these considerations in mind, the use of these rapid multi-species methods can be a valuable tool for the conservation of imperiled or relatively unknown species.

Conclusion

Our results provide insight into the importance of complex and cohesive understory vegetation, which provide protection and food availability for marten and their prey. We found a higher relative prey abundance and fewer detections of bobcats in the coastal forests compared to the interior forest. Several small-bodied carnivores were common in the coastal forests, therefore release from competition from terrestrial competitors did not seem a plausible explanation for the restricted distribution of martens, but we could not dismiss avian competition as we did not adequately survey for raptors.

A potential explanation for the absence of martens in the interior forest is that the mature forest of the Siuslaw National Forest, much of which was removed by fires in the 19th century and timber harvest in the 20th century, may not yet be old enough to provide the complex understory and decay related features, often used for resting and denning [23,66]. Complex understories are characteristic of very old forests in the Pacific Northwest (e.g. > 300 years), which contain extensive downed wood and shrub cover in long-lived tree-fall gaps [22]. Our results suggest that direct and indirect effects of understory shrubs may play a key role in the distribution of martens. Fruit-producing shrubs provide a direct nutritional resource that was a key component of marten diets, and dense shrub cover can reduce predation risk from larger mammalian predators that cannot navigate through shrubs and avian predators that cannot access the area beneath the shrub layer.

Based on our data and literature on marten movement [67], maintaining connected forests within the Oregon Dunes National Recreation Area in conjunction with habitat restoration to promote recruitment of a dense, tall fruit-producing shrub layer within interior forest would likely benefit Humboldt marten populations. Our study was limited to publicly owned forests leaving some private landholdings unstudied. Future survey efforts could focus on filling in these gaps to more evenly cover the interior forest. In addition, further studies to facilitate marten conservation could include identification of structures used as rest and den sites within the young coastal forests, causes of mortality, threats of disease and rodenticides, and opportunities for dispersal into the interior forest [68]. Based on historic records and knowledge of marten habitat elsewhere [44], we know that parts of the interior forest were once occupied by martens and have the potential to be occupied again if conditions improve. If the fruit-producing shrubs in the coastal forests are increasing prey diversity and abundance, or reducing predation risk for martens (and their prey), future restoration efforts in the interior forest could focus on fruit-producing shrub recruitment of similar density and height to coastal shrub forests.

Supporting information

S1 Document. Humboldt marten (*Martes caurina humboldtensis*) scat collection methods.
(DOCX)

S2 Document. Potential Humboldt marten (*Martes caurina humboldtensis*) prey species sequencing methods.
(DOCX)

S3 Document. R code for all camera and vegetation data analyses during Humboldt marten (*Martes caurina humboldtensis*) study in the central Oregon coast.
(DOCX)

S1 Fig. Study area depiction and examples of photos obtained with remote cameras across our four vegetation types during a study of Humboldt marten (*Martes caurina humboldtensis*)

occurrence in the central Oregon coast: A) Beach grass, B) Seasonally-flooded shore pine forest, C) Coastal shrub forest, D) Interior forest.
(DOCX)

S1 Table. List of all species detected with camera traps in four vegetation types during a study of Humboldt marten (*Martes caurina humboldtensis*) distribution. We combined data into 4 vegetation types: Beach grass, Seasonally-flooded shore pine forest, Coastal shrub forest and Interior forest. We depict the number of photographs obtained for each species type.
(DOCX)

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Author Contributions

Conceptualization: Charlotte E. Eriksson, Katie M. Moriarty, Mark A. Linnell, Taal Levi.

Data curation: Charlotte E. Eriksson, Katie M. Moriarty, Mark A. Linnell.

Formal analysis: Charlotte E. Eriksson, Taal Levi.

Funding acquisition: Katie M. Moriarty, Taal Levi.

Investigation: Charlotte E. Eriksson.

Methodology: Charlotte E. Eriksson.

Project administration: Mark A. Linnell.

Resources: Katie M. Moriarty, Mark A. Linnell.

Supervision: Katie M. Moriarty, Taal Levi.

Writing – original draft: Charlotte E. Eriksson.

Writing – review & editing: Charlotte E. Eriksson, Katie M. Moriarty, Mark A. Linnell, Taal Levi.

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Fred H. Everest, Douglas N. Swanston, Charles G. Shaw III,
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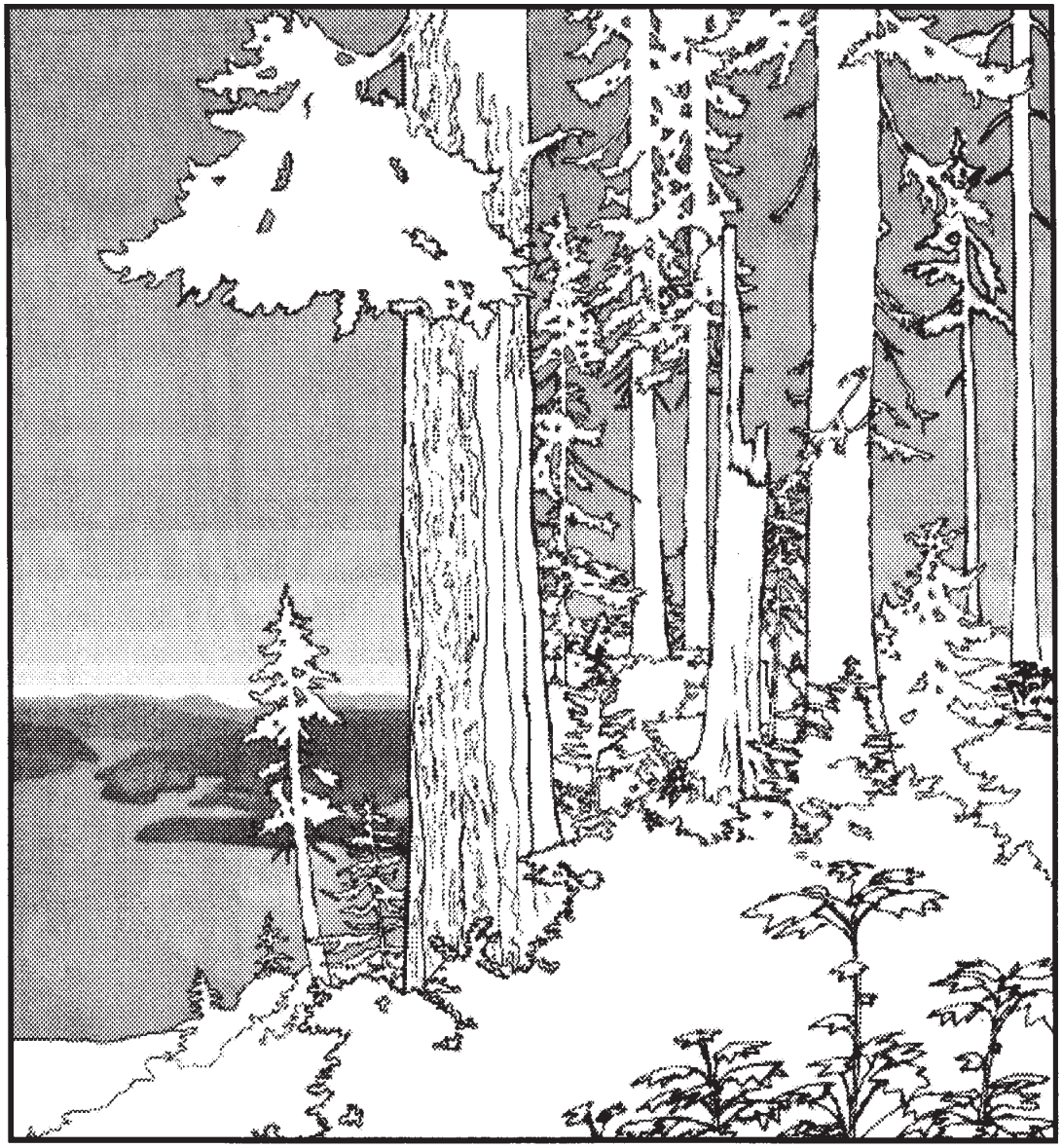

Authors

FRED H. EVEREST is a research fisheries biologist and research policy coordinator for the Tongass Land Management Planning Team, DOUGLAS N. SWANSTON is a research geologist and coleader of the Tongass Land Management Planning Team, CHARLES G. SHAW III is a research plant pathologist and science manager for the Tongass Land Management Planning Team, WINSTON P. SMITH is a research wildlife biologist, KENT R. JULIN is a research forest ecologist, and STEWART D. ALLEN is a research social scientist for the Tongass Land Management Planning Team, U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 2770 Sherwood Lane, Suite 2A, Juneau, AK 99801-8545.

Conservation and Resource Assessments for the Tongass Land Management Plan Revision

Charles G. Shaw III, Technical Coordinator

Evaluation of the Use of Scientific Information in Developing the 1997 Forest Plan for the Tongass National Forest

Fred H. Everest, Douglas N. Swanston, Charles G. Shaw III,
Winston P. Smith, Kent R. Julin, and Stewart D. Allen

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Abstract

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The Tongass National Forest is the largest remaining relatively unaltered coastal temperate rain forest in the world. The Forest consists of 16.9 million acres of land distributed across more that 22,000 islands and a narrow strip of mainland in southeast Alaska. The Forest contains abundant timber, wildlife, fisheries, mineral, and scenic resources.

The authors participated as scientists on the Tongass Land Management Planning Team from 1995 to 1997. We joined the planning team as full members but maintained separate and distinct roles from National Forest System members. We were asked to assure that credible, value-neutral, scientific information was developed independently without reference to management decisions. We also displayed the likely levels of risk to resources and society associated with various management options.

We examined how scientific information was used in making management decisions and evaluated whether the decisions were consistent with the available information. We developed and used a set of criteria to evaluate the way in which managers used scientific information in formulating decisions. This evaluation began while the final alternative was in the formative stages so that managers could alter their management approach, if they so desired, before the Forest plan was finalized. Many management decisions were altered during this “adaptive decisionmaking process” in which changes were made concurrent with iterations of this paper. Our conclusion was that the final management decisions made in developing the 1997 Forest plan achieved a high degree of consistency with the available scientific information. This paper does not consider any information gathered after the signing of the record of decision on May 23, 1997, or deal with subsequent implementation of the 1997 Tongass Forest plan.

Keywords: Tongass National Forest, forest management, land management planning, science evaluation, science policy.

Preface

Any reasoned decision about the management of natural resources must be based on a sound foundation of scientific information. The complexity of natural systems and their importance to people depending on them demand this.

Though an essential consideration, science information alone does not "make" a decision: Decisionmakers make decisions. They make decisions after they complete what is essentially a value-oriented integration of the good and bad features of the outcomes of alternative management paths.

Because science information does not make decisions, what part should scientists play in validating and evaluating a decision? Scientists should not advocate any particular outcome or decision. They should, however, advocate that the relevant scientific information be considered when a decision is made. They should determine whether the decision is *consistent* with the science information.

The authors of this report charted new ground in the application of scientific information to natural resource decisions. This science consistency check was valuable in three ways. First, early drafts of this check communicated the science information to the land management decisionmakers more effectively than separate science reports would have. Second, the science consistency check helped counteract those who alleged that science in fact did "make" decisions. Third, it helped the scientists better clarify and stay within their science role. In the process, the authors helped us all better craft the proper role of scientific information and of scientists in decisions about the management of the Nation's natural resources.

Thomas J. Mills
Station Director
Pacific Northwest
Research Station

Summary

The authors participated as scientists on the Tongass Land Management Planning (TLMP) Team from 1995 to 1997. We joined the planning team as full members but maintained separate and distinct roles from National Forest System members. We worked in cooperation with other resource experts from the Forest Service, state and other Federal agencies, and universities to assemble the most complete base of information ever developed for Forest planning in the Tongass National Forest. We were asked to assure that credible, value-neutral, scientific information was developed independently without reference to management decisions. Emphasis was placed on acquisition, assessment, and synthesis of available information. We displayed options and the likely levels of risk to resources and society associated with various decisions.

The information, with a scientific interpretation of it, was provided to the managers responsible for deciding the content and direction of the Tongass land management plan revision. Managers considered this information as they developed an array of draft alternatives and, subsequently, a draft of the preferred alternative for management of the Tongass. After further analysis and review of public comment, the managers modified the draft of the preferred alternative into the Forest plan.

One of our final responsibilities as members of the TLMP Team was to examine how the available scientific information was used in making management decisions and to evaluate whether the decisions were consistent with that information. We developed and used a specific set of criteria to evaluate consistency of management decisions with scientific information. The evaluation of how scientific information was used began while the final alternative was still in the formative stages.

Management decisions were evaluated as they were made in an iterative process that resulted in more than 20 drafts of this paper, and a like number of changes in management decisions, before the emerging preferred alternative was finalized. In the final analysis, we consider that this paper, as much as any other aspect of the planning process, helped assure that the Forest plan was scientifically credible, legally defensible, and resource sustainable in the long term.

This evaluation does not consider any information gathered after the signing of the record of decision on May 23, 1997, or deal with subsequent implementation of the 1997 Tongass Forest plan.

A clear understanding of the different roles that science and management play in natural resource decisionmaking is necessary to establish criteria for assessing the consistency of management decisions relative to available scientific information. Scientists provide managers and policymakers with the foundational information for making reasoned decisions, but policy issues, not science, dictate the decisions. Scientists objectively follow rigid scientific protocols in developing new information, integrating and synthesizing existing and new information, assuring that information is interpreted correctly, and assessing the probable consequences associated with various proposed management actions.

Managers and policymakers, on the other hand, use scientific information, legal mandates, societal desires, and other factors to make decisions about, for example, allocation of natural resources on Federal lands. All policy decisions concerning the use of natural resources contain some level of risk to resources as a result of long-term implementation. Potential risks associated with decisions can be numerous and might affect, for example, community stability, wildlife viability, or long-term sustainability of resources. When making decisions, managers strive to balance the array of

risks associated with their decisions with the values of goods and services flowing to society from National Forest lands. Such management decisions almost always include compromises for one or more resources.

The appropriate level of risk to accept in management of the National Forests is a policy decision determined by managers. It is not an issue that can be answered by the scientific method. If scientists attempt to participate in or personally influence such decisions, then their objectivity may be compromised as they bring their personal values regarding levels of acceptable risk to bear on the decision.

To address how scientific information was used by managers making decisions about revision of the Tongass land management plan, we developed the following criteria:

A. A management decision was considered to be *consistent* with available scientific information if the following three conditions were met:

   1. All relevant scientific information made available to managers was considered in the decision.

   2. Scientific information was understood and correctly interpreted.

   3. Resource risks associated with decisions were acknowledged and documented.

   All three criteria had to be met before a decision could receive a summary rating of being *consistent*, in our assessment, with available scientific information.

B. A management decision was considered to be *inconsistent* with available scientific information if *any* of the following circumstances occurred:

   1. Managers misrepresented or reinterpreted information in ways not supported by the original information.

   2. Managers selectively used information such that a different decision was reached than would have been made if all available information had been used.

   3. Decisions were stated and documented in such a way that implementation effects could not be predicted.

   4. Projected consequences of management actions were not consistent with scientific information.

   Failure to meet any of these criteria resulted in a summary rating of being *inconsistent*, in our assessment, with available scientific information.

The following example addressing brown bear habitat illustrates how these criteria were used to assess consistency of decisions with available scientific information.

Example: Radio-telemetry data from a brown bear study indicate that areas along principal salmon spawning streams are key brown bear feeding habitat and that bear-use of these zones extends about 500 feet on either side of the streams. The long-term health of bear populations is tied to maintenance of old-growth forest habitat in these areas, which provides hiding cover from humans and isolates feeding and resting bears from each other. In this example, we consider three possible approaches for management of brown bear feeding areas along salmon spawning streams.

1. Managers develop standards and guidelines for managing brown bear feeding areas along salmon spawning streams that protect about 500-foot widths of forest habitat along each side of the streams. They acknowledge, and risk assessment panels verify, that the risk to bear populations would be low from using this management approach.

   We would consider this decision to be consistent with available scientific information because managers were provided relevant information, it appears that they understood it by developing management directions that fully protected this key bear habitat, and they documented the risk to bear populations.

2. Managers develop no standards and guidelines that specifically protect brown bear feeding areas along salmon spawning streams. Managers, however, document in the final environmental impact statement (FEIS) that research indicates key brown bear feeding areas extend about 500 feet on each side of principal salmon spawning streams. They also state that they considered riparian standards and guidelines prescribed for these streams, which protect an average 300-foot width along each side and provide an acceptable compromise between maintenance of bear habitat and timber production. They acknowledge that the risk to bear populations could be increased by this decision and commit to further studies to assess the extent of risk.

   We also would consider this decision consistent with available scientific information. The managers based their decision on a reasonable interpretation of relevant scientific information, with consideration of public desires for resources and, perhaps, legal mandates. Managers acknowledge and document that some increased risk to brown bear populations will result from their management direction.

3. Managers develop no standards and guidelines specifically to protect brown bear feeding areas along salmon spawning streams. They state in the FEIS that they considered riparian standards and guidelines prescribed for these streams, which protect an average 300-foot width along each side, adequate to protect the principal brown bear feeding areas along salmon spawning streams.

   We would consider this decision to be inconsistent with available scientific information. Even though all relevant scientific information was provided to managers, they failed to acknowledge its existence or to incorporate it into their decision, and they also failed to acknowledge and document the increased risk to brown bear feeding areas associated with the decision.

   Certain types of decisions cannot be subjected to this type of evaluation. Any scientific document containing recommendations for managers may be in this category. Most recommendations are not scientific conclusions but, rather, a combination of scientific information integrated with professional experience and personal values that embody a level of risk that those making the recommendations thought appropriate. Recommendations, even those made by scientists, were not considered in this evaluation of the use of scientific information.

Using the criteria as defined above, many of the major components of the emerging final alternative were evaluated for consistency with the available information base. Results of the evaluation are detailed in table 1 for riparian and fish sustainability, wildlife viability, karst and caves protection, slope stability, timber resources, social and economic effects, and monitoring. Overall, while we remain value-neutral on the Forest plan, our evaluations indicated that the decisions made in developing the plan achieved a high degree of consistency with the available scientific information.

Table 1—Summary of whether information on key issues in the Tongass Forest plan was considered, understood, and risks revealed and documented, and whether decisions are consistent with available scientific information

	Criteria for decision							
	Was all information considered?		Was information understood and interpreted correctly?		Were risks revealed and documented?		Was decision consistent?	
Issue	Yes	No	Yes	No	Yes	No	Yes	No
Aquatic:								
Watershed analysis	X		X		X		X	
Cumulative effects	X		X		X		X	
Hydrologic function	X		X		X		X	
Sediment	X		X		X		X	
Water quality	X		X		X		X	
Riparian vegetation	X		X		X		X	
Stability of complex channels	X		X		X		X	
Woody debris	X		X		X		X	
Slope stability	X		X		X		X	
Fish access	X		X		X		X	
Anadromous fish habitat	X		X		X		X	
Resident fish habitat	X		X		X		X	
Wildlife:								
Viability	X		X		X		X	
Goshawk	X		X		X		X	
Wolf	X		X		X		X	
Deer	X		X		X		X	
Brown bear	X		X		X		X	
Terrestrial mammals	X		X		X		X	
Beach/estuary	X		X		X		X	
Matrix lands	X		X		X		X	
Reserves	X		X		X		X	
Road management	X		X		X		X	

Table 1—Summary of whether information on key issues in the Tongass Forest plan was considered, understood, and risks revealed and documented, and whether decisions are consistent with available scientific information (continued)

	Criteria for decision							
	Was all information considered?		Was information understood and interpreted correctly?		Were risks revealed and documented?		Was decision consistent?	
Issue	Yes	No	Yes	No	Yes	No	Yes	No
Karst and caves	X		X		X		X	
Slope stability	X		X		X		X	
Timber:								
Estimation of old-growth timber volume	X		X		X		X	
Modeling implementation reduction factors (MIRFS)	X		X		X		X	
Estimation of young-growth timber volume	X		X		X		X	
Noninter-changable components (NICs)	X		X		X		X	
Management of forested wetlands	X		X		X		X	
Windthrow	X		X		X		X	
Socioeconomic concerns	X		X		X		X	
Monitoring and adaptive management	X		X		X		X	

Contents

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Introduction

The Tongass National Forest is the largest remaining relatively unaltered coastal temperate rain forest in the world. The Forest consists of 16.9 million acres of land distributed across more than 22,000 islands and a narrow strip of mainland, which extends about 540 miles north from the southern end of the Alexander Archipelago (fig. 1 located in inside back cover). Geographic extent and isolation, physiographic variation, and elevational and climatic gradients combine to produce numerous spatially and temporally heterogeneous environments that support a diverse indigenous biota including several endemic plants and animals. There are 21 ecological provinces that stratify the Forest according to various configurations of physical, climatic, and biotic features. About 10 million acres is timberland; the remainder is fen, sphagnum bogs, rock, glaciers, ice fields, and water. Before large-scale industrial logging began (ca. 1954), there were about 5.4 million acres of productive old-growth forest[1] with nearly half in high-volume old-growth forest.[2] During the last half of the 20th century, about 400,000 acres of primarily high-volume old-growth forest were clearcut logged, which further fragmented naturally dissected landscapes and forest habitats.

Of the 5 million plus acres of productive old-growth forest that remain in 1997, 4.2 million acres are withdrawn from timber production, including 1.7 million acres of legislated wilderness areas. The more productive timberlands are in the southern half of the Forest where a world-class karst and cave resource, sensitive to timber harvest activities, is prevalent. The lakes, streams, and surrounding marine waters support one of the most diverse and productive fisheries for wild anadromous salmonids in the world. Although there are expanding recreational and tourist interests that complement a commercial fishery, mining, and timber based economy, subsistence is still a significant component of the lifestyle of many residents.

Balancing the levels of protection and use of natural resources across such a vast and diverse landscape has been an ongoing challenge for USDA Forest Service officials in Alaska. Under the direction of the Alaska Regional Forester and the three Tongass National Forest Supervisors, the Tongass land management plan revision was completed on May 23, 1997. A basic principle of National Forest System planning requires that resources be managed in a sustainable fashion (U.S. Laws, Statutes 1976:sect. 6(e)(1)). Sustained yield is "...the achievement and maintenance in perpetuity of a high-level of annual or periodic output of the various renewable resources... without impairment of the productivity of the land" (USDA Forest Service 1982).

In accordance with the principles of National Forest System planning, the authors participated as scientists and full members of the 1995-97 Tongass Land Management Planning (TLMP) Team but maintained roles distinct from those of the National Forest System members. We independently developed credible, value-neutral, scientific information without reference to management decisions. Emphasis was placed on

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[1] Productive old-growth forest is operationally defined as a previously unharvested forested area of trees more than 150 years old with greater than 10 percent tree cover, and an accumulated net saw-log volume of more than 8,000 board feet per acre (mbf/acre) (Julin and Caouette 1997).

[2] High-volume old-growth forest in the Tongass averages 34,600 board feet per acre (Julin and Caouette 1997).

acquisition, assessment, and synthesis of available information, including new information generated by ongoing studies. We displayed options and the likely levels of risk to resources and society associated with various decisions. We did not suggest or concur on what level of risk was acceptable because that is a public policy decision to be determined by policymakers, not scientists. We also did not make management decisions.

We collaborated with other resource experts and scientists from the Forest Service, state and other Federal agencies, and universities to assemble the most complete information ever developed for forest planning in the Tongass National Forest. The principle components of the information base included:

- Scientific assessments for Queen Charlotte goshawk (*Accipiter gentilis laingi*) (Iverson and others 1996), Alexander Archipelago wolf (*Canis lupus ligoni*) (Person and others 1996), marbled murrelet (*Brachyrampus marmoratus*) (DeGange 1996), anadromous fish habitat (Anadromous Fish Habitat Assessment [AFHA] Team 1995), karst and cave resources (Baichtal and Swanston 1996), and socioeconomic effects (Allen and others, in press).
- Resource analyses on forested wetlands (Julin and Meade 1997), slope stability (Swanston 1997), old-growth timber volume strata (Julin and Caouette 1997), wind disturbance (Nowacki and Kramer, in prep.), a synthesis of wildlife viability (Iverson and René 1997), and timber demand (Brooks and Haynes 1997).
- Expert panels commissioned to evaluate the risk to resources from implementation of an array of proposed draft alternatives for managing the Tongass National Forest. Panels examined potential effects of proposed alternatives on brown bear (*Ursus arctos*), Queen Charlotte goshawk, American marten (*Martes americana*), marbled murrelet, Alexander Archipelago wolf, Sitka black-tailed deer (*Odocoileus hemionus sitkensis*), other terrestrial mammals, fish and riparian conditions, old-growth ecosystems, subsistence, and socioeconomic issues (Shaw, in prep.; Swanston and others 1996). As the emerging final alternative was nearing completion, a second set of risk assessment panels was conducted to evaluate risk to brown bear, Queen Charlotte goshawk, American marten, Alexander Archipelago wolf, other terrestrial mammals, and fish and riparian conditions (Shaw, in prep.).
- A document that distilled key findings from the information components listed above (Swanston and others 1996).

We provided the information listed above, and a scientific interpretation of it, to the managers responsible for deciding the content and direction of the Tongass land management plan revision. The managers considered this information as they developed and modified a draft of the preferred alternative for management of the Tongass and, after further analysis and review of public comment on this draft, modified it into the Forest plan.

"Forest plan" refers to the selected alternative as presented in the final environmental impact statement (FEIS) (USDA Forest Service 1997b) and documented in the record of decision (ROD) (USDA Forest Service 1997c). "Emerging final alternative" refers to the alternative that was being built by the planning team for several months and was "finalized" into the Forest plan. This term is not to be confused with the "preferred alternative" in the revised supplemental draft environmental impact statement (RSDEIS) (USDA Forest Service 1996b), which contains some different management strategies from the final alternative in the FEIS.

One of our final responsibilities as members of the TLMP Team was to examine how the available scientific information was used in making management decisions and to evaluate whether the decisions were consistent with that information. We developed and used a specific set of criteria to evaluate consistency of management decisions with scientific information. The evaluation of how scientific information was used began while the final alternative was still in the formative stages.

Management decisions were evaluated as they were made in an iterative process that resulted in more than 20 drafts of this paper, and a like number of changes in management decisions, before the emerging preferred alternative was finalized. In the final analysis, we consider that this paper, as much as any other aspect of the planning process, helped produce a Forest plan that was scientifically credible, legally defensible, and resource sustainable in the long term.

Methods

A clear understanding of the different roles that science and management play in natural resource decisionmaking is necessary to establish criteria for assessing the consistency of management decisions relative to available scientific information. Scientists provide managers and policymakers with the foundational information for making reasoned decisions, but policy issues, not science, dictate the decisions. Scientists objectively follow rigid scientific protocols in developing new information, integrating and synthesizing existing and new information, assuring that information is interpreted correctly, and assessing the probable consequences associated with various proposed management actions.

Managers and policymakers, on the other hand, use scientific information, legal mandates, societal desires, and other factors to make decisions on, for example, allocation of natural resources on Federal lands. All policy decisions concerning the use of natural resources contain some level of risk to resources as a result of long-term implementation. Potential risks associated with decisions can be numerous and might affect, for example, community stability, wildlife viability, or long-term sustainability of resources. When making decisions, managers strive to balance the array of risks associated with their decisions with the values of goods and services flowing to society from National Forest lands, while still meeting legal mandates. Such decisionmaking almost always includes compromises for one or more resources.

The appropriate level of risk to accept is a policy decision determined by managers. It is not an issue that can be answered by the scientific method. If scientists attempt to participate in or personally influence such decisions, then their objectivity may be compromised as they bring their personal values regarding levels of acceptable risk to bear on the decision.

To address how scientific information was used by managers making decisions about revision to the Tongass land management plan, we developed the following criteria:

A. A management decision was considered to be *consistent* with available scientific information if the following three conditions were met:

1. All relevant scientific information made available to managers was considered in the decision.
2. Scientific information was understood and correctly interpreted.
3. Resource risks associated with decisions were acknowledged and documented.

All three criteria had to be met before a decision could receive a summary rating of being *consistent*, in our assessment, with available scientific information.

B. A management decision was considered to be *inconsistent* with available scientific information if *any* of the following circumstances occurred:

1. Managers misrepresented or reinterpreted information in ways not supported by the original information.
2. Managers selectively used information such that a different decision was reached than would have been made if all available information had been used.
3. Decisions were stated and documented in such a way that implementation effects could not be predicted.
4. Projected consequences of management actions were not consistent with scientific information.

Failure to meet any of these criteria resulted in a summary rating of being *inconsistent*, in our assessment, with available scientific information.

Two examples, one related to brown bear habitat and another related to slope stability, are used to illustrate how the criteria were applied to management decisions.

Example 1: Radio-telemetry data from a brown bear study indicate that areas along principal salmon spawning streams are key brown bear feeding habitat and that bear-use of these zones extends about 500 feet on either side of the streams. The long-term health of bear populations is tied to maintenance of old-growth forest habitat in these areas, which provides hiding cover from humans and isolates feeding and resting bears from each other. In this example, we consider three possible approaches for management of brown bear feeding areas along salmon spawning streams.

1. Managers develop standards and guidelines for managing brown bear feeding areas along salmon spawning streams that protect about 500-foot widths of forest habitat along each side of the streams. They acknowledge, and risk assessment panels verify, that the risk to bear populations would be low from using this management approach.

> We would consider this decision to be consistent with available scientific information because managers were provided relevant information, it appears that they understood it by developing management directions that fully protected this key bear habitat, and they documented the risk to bear populations.

2. Managers develop no standards and guidelines that specifically protect brown bear feeding areas along salmon spawning streams. Managers, however, document in the final environmental impact statement (FEIS) that research indicates key brown bear feeding areas extend about 500 feet on each side of principal salmon spawning streams. They also state that they considered riparian standards and guidelines prescribed for these streams, which protect an average 300-foot width along each side and provide an acceptable compromise between maintenance of bear habitat and timber production. They acknowledge that the risk to bear populations could be increased by this decision and commit to further studies to assess the extent of risk.

> We also would consider this decision consistent with available scientific information. The managers based their decision on a reasonable interpretation of relevant scientific information, with consideration of public desires for resources and, perhaps, legal mandates. Managers acknowledge and document that some increased risk to brown bear populations will result from their management direction.

3. Managers develop no standards and guidelines specifically to protect brown bear feeding areas along salmon spawning streams. They state in the FEIS that they considered riparian standards and guidelines prescribed for these streams, which protect an average 300-foot width along each side, adequate to protect the principal brown bear feeding areas along salmon spawning streams.

> We would consider this decision to be inconsistent with available scientific information. Even though all relevant scientific information was provided to managers, they failed to acknowledge its existence or to incorporate it into their decision, and they also failed to acknowledge and document the increased risk to brown bear feeding areas associated with the decision.

Example 2: Roads and timber harvest directly influence stability of slopes across the Tongass National Forest through disturbance of the soil mantle and alteration of natural drainages. The resulting landslides remove soil from forest slopes and are major contributors of sediment and large woody debris to streams used by anadromous fish. Managers recognized the significance of this relation and developed a mass failure hazard (landslide) assessment protocol in 1987 based on the information available at the time. This protocol defined a set of variables controlling stability on steep forested slopes and identified the critical slope gradient above which no timber harvesting should occur, because of the high potential for landslides.

Inspection of the 1987 protocol indicated that the data on which the protocol was based were out of date, and there was a lack of consistency in identification, measurement, and interpretation of variables across Administrative Areas (Ketchikan, Stikine, and Chatham) of the Forest. A reanalysis and assessment of slope stability literature and the most current data, including recently developed field information from southeast Alaska, identified the need to redefine and lower the critical slope gradient identifying high-hazard soils from 75 percent to 72 percent and revise and standardize the protocol to provide for consistent measurement and application of variables across Administrative Areas of the Forest.

In this example, we consider three possible management directions for addressing slope stability.

1. Managers acknowledge and document in the FEIS the new information provided in the analyses and assessments, adopt the revised mass failure hazard protocol, lower the critical slope gradient from 75 percent to 72 percent, develop soil and water standards and guidelines for managing potentially unstable terrain that reflect these changes, and acknowledge that the risk of accelerated mass erosion is reduced when these changes are applied to timber sales.

> We would consider this decision to be consistent with available scientific information, because managers considered all available information, understood it, and developed management direction that maximized protection of high-hazard soils and reduced the potential for accelerated delivery of sediment and large woody debris to anadromous fish streams. Managers also acknowledged the risk associated with their decision.

2. Managers do not adopt the revised protocol, do not lower the critical slope gradient from 75 percent to 72 percent, and do not alter existing standards and guidelines. Managers, however, do acknowledge and document in the FEIS the new information provided in the analyses and assessments. They state that even though the new information is valuable and will be incorporated into the Forest plan where practical, they consider that the existing standards and guidelines and current high-hazard soil boundary provide more management flexibility and represent an acceptable compromise between protection of unstable slopes and timber production. They acknowledge that the risk of accelerated landslides could be increased by this decision and commit to a monitoring and assessment program to evaluate the protocol and determine the extent of risk.

> This decision is based on correct interpretation of available scientific information, with consideration of public desires and resources. Managers acknowledge and reveal that some increased risk of landslides as the result of forest management is inherent in the decision. We would consider this decision to be consistent with available scientific information.

3. Managers do not adopt the revised protocol, do not lower the critical slope gradient from 75 percent to 72 percent, and do not alter existing standards and guidelines. They state in the FEIS that they consider that the existing protocol and critical slope gradient of 75 percent are adequate to protect high-hazard soils and reduce the potential for accelerated delivery of sediment and large woody debris to anadromous fish streams.

> This decision does not consider all available scientific information and fails to acknowledge and reveal the increased risk of landslides and accelerated sediment delivery associated with the decision. We would consider this decision to be inconsistent with available scientific information.

Certain types of decisions cannot be subjected to this type of evaluation. Any scientific document containing recommendations for managers may be in this category. Recommendations are not scientific conclusions but, rather, a combination of scientific information integrated with professional experience and personal values that embody a level of risk that those making the recommendations think appropriate. Recommendations, even those made by scientists, were not considered in this science consistency check.

Evaluation of Decisions Related to Specific Resources

Decisions related to some resource areas, for example fish and wildlife issues, better lend themselves to a science consistency check of this type than, for example, socio-economic and monitoring issues, which may be based on less quantitative science. In resource areas where information is more qualitative, a greater degree of uncertainty surrounds a science consistency evaluation, and it is thus more difficult to conduct.

Aquatic, Riparian, and Fisheries Sustainability

The Tongass National Forest lies in the northern sector of the coastal temperate rain forest along the west coast of North America. The high annual precipitation along this coast and fairly even distribution of precipitation in time and space creates and maintains an extensive variety of stream, river, lake, wetland, and riparian habitats across the Forest. These abundant physical features of the landscape provide essential habitats for a wide variety of fish and wildlife species of special interest to sport, commercial, and subsistence users.

Protection of aquatic and riparian habitats was identified as one of the key issues in the 1995-97 revision of the Tongass land management plan. The focus on aquatic and riparian habitats was related largely to the importance of anadromous salmonids (primarily Pacific salmon and steelhead [*Oncorhynchus mykiss*] produced annually in these habitats, although resident fish and other species also are of interest and concern to managers. The management standard for anadromous fish in the current Forest plan is maintenance of current habitat productivity sufficient to provide for undiminished future harvestable populations.

The stream and lake habitats of the Tongass National Forest are unique in their complexity and productivity. Islands of the Alexander Archipelago and the narrow strip of mainland southeast Alaska contain:

- Numerous small creeks and streams on islands
- Some large streams and rivers on islands
- Some large transboundary mainland rivers
- Numerous small coastal mainland rivers
- Numerous stream and lake complexes in all types of systems

All these aquatic systems are subjected to variable climatic conditions that result in streams dominated by glaciers, interior continental climate, coastal maritime climate, climate of the transition snow zone, and local orographic effects.

The aquatic and fisheries resources of the Tongass are adapted to this diversity of habitats and climatic conditions. Consequently, numerous species and discrete spawning populations of anadromous and resident salmonids exist in the region. Mixed assemblages of wild salmonids occur in nearly all the different habitat types. Most watersheds of the Tongass contain five or more species of salmonids—a common mix includes coho salmon (*O. kisutch*), chum salmon (*O. keta*), and pink salmon (*O. gorbuscha*), steelhead, sea-run cutthroat trout (*O. clarki*), and Dolly Varden char (*Salvelinus malma*). These species may all reside together in the same watershed and share the same stream reaches, but each uses available habitat in a slightly different way. Although all species and age classes in fresh water are vulnerable to habitat disturbances, the most vulnerable are those whose juveniles reside in fresh

water the longest before migrating to sea (e.g., steelhead and cutthroat trout; Reeves and others 1993). Any major changes in these habitats can have immediate effects on the productivity and community structure of salmonids (Reeves and others 1995) and other aquatic species.

Anadromous salmonids represent one of the preeminent natural resources in southeast Alaska. The Alexander Archipelago and the mainland of southeast Alaska support one of the most productive and highly valued fisheries for wild salmon in the world. Commercial salmon fisheries yield 160 million pounds (average annual production from the Tongass) worth about $250 million annually and provide more than 5,000 direct jobs (AFHA 1995). The sport fishing industry is smaller in economic and employment effects but is growing at an average of 10 percent per year (AFHA 1995). Sport fishing now provides more than 1,200 direct, full-time job equivalents with over $28 million in earnings and 250,000 angler days. Sport fishers spend more than $90 for each salmon caught. The annual subsistence harvest of salmon is in excess of 1.2 million pounds. Harvesting salmon in traditional areas is important to sustaining the Tlingit, Haida, and Tsimshian cultures. Activities associated with salmon annually support the highest number of natural resource-related jobs in southeast Alaska (Allen and others, in press).

The variety of freshwater habitats where anadromous fish breed and reside for variable periods before entering the marine phase of their life cycles are essential to the long-term sustainability of their populations. The same is true for resident fish and other aquatic species.

Factors influencing aquatic, riparian, and fish habitats—Salmonids have exacting freshwater habitat requirements and are sensitive to habitat disturbances (Everest and others 1985). All salmonids require relatively pristine freshwater habitats during all or part of their life cycles (Meehan 1991). Consequently, salmonids can be used as effective indicators of the health of aquatic systems (Sonstegard and Leatherland 1984). If salmonid populations are healthy in watersheds, then populations of other aquatic species generally are healthy as well.

Salmonids in fresh water are vulnerable to natural and human-caused changes in habitat quality. The quality of habitat for fish or other aquatic species in any stream depends on the condition of the area that it drains. Thus, watersheds are the basic unit of forested landscapes, controlling the quality of aquatic habitats and to a large extent the populations of fish and other aquatic species present, as well as the numbers of fish that can be sustainably harvested (AFHA 1995).

Watersheds in southeast Alaska are disturbed by both natural events and human activities, but the effects of these disturbances on fish habitat are generally different. The primary differences are related to the frequency and extent of disturbances across forested landscapes and the potential for recovery of disturbed landscapes and fish habitats.

Natural disturbances (e.g., floods, landslides, windthrow of trees, insect outbreaks, and earthquakes) that create spatial and temporal variability in aquatic habitats across forested landscapes can negatively affect fish and other aquatic species. These disturbances, however, are infrequent and sporadic in time and space, such that entire watersheds rarely are affected by a single event (Reeves and others 1995). The recurrence interval for large natural disturbances, such as earthquakes and major landslides, is often in the range of centuries to millennia for any given site. Because only

a small component of the landscape usually is affected at any one time, and disturbances are infrequent, refugia usually are available to provide for survival of aquatic species until the disturbed area recovers naturally, which might take a century or more. Salmonids and other aquatic species have adaptive strategies for natural disturbances and, consequently, are rarely placed at risk of extinction from natural events.

Human disturbances typically are more frequent and widespread than natural disturbances. The most common forms of human disturbances in forested watersheds are logging, often in relatively large blocks, and the associated construction and maintenance of roads. There is no natural analog to either type of disturbance in southeast Alaska. Aquatic species in the region may not be able to adapt to the intensities of these disturbances before irrevocable population changes occur. Logging affects water quality, hydrologic function, and other characteristics of aquatic habitats in watersheds. Disturbance from logging is relatively frequent in both time and space across watersheds subjected to timber harvest. Roads, even where not associated with timber management, create major watershed disturbances with perennial effects on aquatic resources and watershed function (Furniss and others 1991). Roads interrupt natural hydrologic function in watersheds and generally increase both suspended and bedload sediments in streams. Clearly, widespread and frequent human disturbances, such as logging and road construction, increase the risk to aquatic ecosystems and associated species in southeast Alaska. Management actions can be applied to reduce risk, however (AFHA 1995).

Critical features of aquatic systems for salmonid and other species—Salmonids and other aquatic species have developed within a natural range of habitat variability in watersheds of southeast Alaska. Because of the economic and social importance of salmonids and their role as management indicator species, the critical features that combine to form and maintain productive salmonid habitat also are assumed to provide productive habitat for other aquatic species. Salmonid habitats have been intensively studied for decades, and a large body of information exists on their habitat needs and preferences. Several excellent synthesis papers have been prepared on the subject over the past few years. The following summary of critical features of salmonid habitat was derived primarily from Stouder and others (1997), Meehan (1991), Salo and Cundy (1987), and Everest and others (1985).

If the natural range of variability in critical habitat features can be maintained in the watersheds of southeast Alaska, then aquatic habitats for salmonids and other aquatic species generally will be at lower risk from human disturbance (AFHA 1995). The natural range of these features can be determined from inventory and monitoring of largely undisturbed watersheds.

Key features of aquatic habitats include:

- Hydrologic function (in time, space, and amplitude), including stream flow, ground water, and hyporheic flow (Gregory and Bisson 1997)
- Sediment supply and routing (Everest and others 1987)
- Water quality, including temperature (Beschta and others 1987), dissolved oxygen (Bjornn and Reiser 1991), and nutrients (Murphy and Meehan 1991)
- Riparian vegetation, including intact old-growth forests (Gregory and Bisson 1997)
- Relatively stable, complex stream channels (Sullivan and others 1987)

- Side-channel and off-channel habitats in flood plains (Gregory and Bisson 1997)
- Large woody debris in stream channels (Bisson and others 1987)
- Upstream access (Bjornn and Reiser 1991) and downstream access (Everest and others 1985) for migratory salmonids

Current condition of watersheds in the Tongass National Forest—The general condition of watersheds in the Tongass was assessed in 1992 by National Forest System personnel. Their findings were reported in AFHA (1995:15) as follows:

> Information from the 11 Ranger Districts of the Tongass was summarized by using four general types of land allocation: wilderness, roadless, variety of uses and outputs, and commodity emphasis. Each type was assigned into two categories: watershed condition currently healthy (conditions and functions generally in balance) or watershed condition not healthy. The biologists and hydrologists working on the assessment based their determinations on information about the natural conditions of the watershed, known changes in riparian conditions or stream habitat, and professional observations and experience (J. Christner, data on file at Tongass National Forest, Chatham Area, 204 Siginaka Way, Sitka, AK 99835). Stream class, channel type, management influences (for example, riparian area harvested, minerals activity, or roads), soil hazard rating, and results from a riparian harvest-effects model for coho salmon also influenced their assessments.

For lands within the Tongass National Forest boundary, including all ownerships, 77 percent of the watersheds were considered to have a healthy watershed function and condition and 23 percent had a reduced condition. In the categories “variety of uses and outputs” and “commodity emphasis,” 72 percent of these watersheds were classified as healthy, and 28 percent had conditions with reduced condition.

Most watersheds with reduced function and conditions were affected by commercial timber harvest operations during the 1950s to 1970s when Best Management Practices (USDA Forest Service 1993) were not used. Watersheds managed during the 1980s and 1990s were subject to Best Management Practices. In some places, the watersheds have become wilderness or legislated roadless [areas] since the logging or other activity occurred. What are currently considered Best Management Practices were not in use during the earlier periods when activities such as clearcutting to all channel banks, salvage and removal of natural large woody debris from streams, yarding logs across or through stream channels, extensive road construction in flood plains, inadequate road drainage structure designs, and limited erosion control measures were common.

Results of the watershed assessment indicated that most aquatic habitats in the Tongass remain in relatively undisturbed states, with exceptions in some areas heavily logged two to three decades ago. The strategy developed by managers during the 1995-97 TLMP revision was designed to reduce risk of any further degradation of watershed function in the Forest.

An overview of aquatic and riparian components in the Forest plan—The Forest plan (USDA Forest Service 1997a) draws heavily from scientific information in AFHA (1995), and other sources to develop a strategy for maintaining the productivity and integrity of aquatic, riparian, and fish habitats. A number of management actions are included that reduce risk to aquatic resources from human disturbances. Further

actions could have been taken and were not, presumably for other resource considerations. Whether the final risk to fish habitats, as estimated by expert opinion (Shaw, in prep.; table 2) is acceptable is a policy question and not a question that can be answered by science. In the Forest plan, key components of the strategy to protect aquatic, riparian, and fish habitat include:

1. The following land use designations (LUDs) that protect old-growth forest habitat and reduce the effects of human disturbance on watersheds, riparian areas, and streams:

- An array of lands totaling about 4.2 million acres that are legislatively withdrawn from timber production (USDA Forest Service 1997b:table 3-77).
- An old-growth forest conservation strategy that includes a network of large, medium, and small old-growth reserves, which in combination with various other LUDS protect an additional 3.5 million acres of old-growth habitat (Iverson 1997).
- The spatial distribution of at least one very large old-growth reserve in each of the 21 biogeographic provinces of the Tongass, and a small reserve in each of 237 large watersheds (USDA Forest Service 1997c).
- A 1,000-foot beach and estuary buffer that provides an additional 140,000 acres of protected old-growth habitat within timber production LUDs (Iverson 1997).

Table 2—Likelihood scores (100 points possible) for fish from 2d set of risk assessment panels, Tongass Forest plan[a]

Species	Scores
Chinook salmon	100
Coho salmon	87
Sockeye salmon	93
Chum salmon	94
Pink salmon	93
Steelhead	88
Cutthroat trout:	
Resident	84
Anadromous	85
Dolly Varden:	
Resident	84
Anadromous	89

[a] Panelists evaluated whether salmonid habitats suitable for maintenance of current levels of fish production would be met throughout most of the Tongass National Forest, and if little or no additional habitat degradation would occur, following 100 years of implementation of the emerging final alternative.

Source: Shaw, in prep.

- Riparian buffers along class I, class II, and most class III streams that protect approximately 224,000 additional acres of productive old-growth habitat within timber production LUDs (Iverson 1997).
- Approximately 500-foot buffers to protect principal brown bear feeding areas along salmon streams and also protect riparian and salmon habitat along those streams.

2. Standards and guidelines and other features of the Forest plan that contribute to protection of aquatic, riparian, and fish habitat:

- Direction to buffer all class I, class II, and most class III streams with appropriate widths of existing vegetation, which in most cases is old-growth forest (USDA Forest Service 1997b:4-53, 4-56 to 4-69).
- Direction to conduct watershed analysis (watershed-scale planning) before changing buffer widths designated by standards and guidelines for class I, II, and III streams (USDA Forest Service 1997b:4-51, app. J).
- Direction to implement Best Management Practices to protect beneficial uses of water (USDA Forest Service 1997b:4-53, app. C).
- Riparian management areas (those areas of particular value to watershed and stream system integrity) must be defined for all class I, II, and III streams before timber harvest begins (USDA Forest Service 1997b:4-51).
- Special protection is given to off-channel fish habitats of flood plain, glacial outwash, and alluvial fan process groups (for description of process groups see Paustian 1992; USDA Forest Service 1997b:4-53, 4-56 to 4-59).
- Special emphasis is given to design of windfirm buffers (USDA Forest Service 1997b:4-58 to 4-68).
- New definitions of class III and IV streams are developed (USDA Forest Service 1997b:4-7 to 4-8).
- Operating rules for mass erosion hazard areas are clarified and revised (USDA Forest Service 1997b:4-82).
- Quantitative objectives for fish habitat management are required (USDA Forest Service 1997b:4-8 to 4-9).
- Specific rules for salvage of timber from riparian areas, (USDA Forest Service 1997b:4-54).
- Specific rules for road design, layout, construction, and maintenance (USDA Forest Service 1997b:4-53, 4-57 to 4-71, 4-107).
- Maintain migratory access for anadromous salmonids (USDA Forest Service 1997b:4-57 to 4-69).

Habitat features absent from the Forest plan—The following additions to actions, prescriptions, and standards and guidelines in the Forest plan would likely provide incremental and cumulative benefits toward maintaining aquatic and riparian habitats and sustainability of fisheries in the Tongass. Absence of these features, however, is not an indication that the Forest plan is deficient, and including these or any other features does not ensure that a Forest plan meets acceptable levels of risk for maintaining sustainability of aquatic, riparian, and fishery resources as indicated previously.

1. Increased width of buffer strips along all class I, II, and III streams. Such action would further reduce the risk of windthrow in the core area of buffer strips and increase the likelihood that old-growth timber would remain standing near streams where it contributes incrementally to further gains in:

- Long-term recruitment of large woody debris to stream channels
- Stable and complex stream banks
- Stable summer and winter microhabitat conditions within buffer strips
- Natural summer and winter water temperature regimes
- Natural water quality and nutrient levels
- Stable habitats for anadromous and resident fish

2. Increased protection of class IV headwater streams. Vegetative buffer strips along class IV stream channels, or other measures such as directional timber felling and yarding used to reduce the risk of major disturbances to these channels, would likely reduce the risk of:

- Loss of habitat for old-growth-associated, riparian amphibians
- Acute and chronic mass erosion from headwater channels
- Long-term cumulative effects of sedimentation on water quality and fish habitat in downstream waters

3. More conservative engineering of road drainage structures (e.g., for 100-year storm events) to reduce the risk of road cut and fill failures during exceptional storm events. Drainage structures designed to accommodate larger storm events would reduce the risk of catastrophic sedimentation of fish habitats.

4. Longer timber rotation age in the matrix lands. This addition, coupled with a correspondingly reduced allowable sale quantity, would reduce watershed disturbance in time and space and reduce the risk of both accelerated stream sedimentation and buffer strip blowdown.

5. More comprehensive and intensive watershed analysis. A more comprehensive and detailed level of watershed analysis likely would reduce management-related risks to all resources in a watershed, including aquatic, riparian, and fishery resources.

Consistency in the use of aquatic- and riparian-related information—Sustainability of aquatic, riparian, and fishery resources was identified as a major focal issue to be addressed during the revision of the Tongass land management plan. Protection of fishery resources in particular has been a long-standing controversial issue for the Tongass as evidenced by the debate over passage of the Tongass Timber Reform Act (U.S. Public Laws, Statutes 1990), and the subsequent debate over AFHA (1995). Consequently, how information on aquatic, riparian, and fishery resources was used in the Forest plan is important. Relevant information in several publications synthesizing literature on salmonid habitats, and the anadromous fish habitat assessment report (AFHA 1995), was used in developing the strategy and standards and guidelines for protecting aquatic habitats in the Forest plan. A summary of how scientific

information on key issues related to sustainability of aquatic, riparian, and fishery resources was used in the Forest plan and how risks were acknowledged is provided in table 1 (see “Summary”). The actual risk ratings (likelihood scores), as estimated by expert opinion, for implementation of the emerging final alternative for 100 years, are explained by Shaw (in prep.) and summarized in table 2.

The Forest plan makes consistent use of available scientific information and provides a strong relation between key features of salmonid habitat (see p. 9-10) and Forest plan LUDs and standards and guidelines (see p. 10-12) to produce a strategy with a high likelihood of maintaining sustainable levels of aquatic, riparian, and fish habitats. Overall, the consistency criteria of using all available information, interpreting and considering the information correctly, and acknowledging and documenting risks associated with management decisions were met to a high degree.

A science consistency check on recommendations within the AFHA report (1995) is not appropriate, but a general comparison between AFHA recommendations and the Forest plan is made in appendix 1. The AFHA recommendations are a combination of scientific information, professional experience, and the personal values of those making the recommendations and therefore are not suitable for a science consistency check.

Wildlife Viability

The Tongass represents habitat for numerous indigenous vertebrates: 53 mammals, 231 birds, and 5 amphibians and reptiles (USDA Forest Service 1996a). The Forest encompasses the largest remaining largely unaltered block of coastal temperate rain forest in the world (Lawford and others 1996). It is comprised of the Alexander Archipelago and a narrow strip of North American mainland. Like the more than 22,000 islands of the archipelago, the mainland is largely isolated from other large, contiguous land masses because of mountains, glaciers, and icefields immediately to the east that create a significant barrier to movement by vertebrates. A few east-west river valleys dissect an otherwise snow-, ice-, or rock-covered mountainous landscape, each river valley providing only a constricted corridor for dispersal of wildlife. Because of these features, southeast Alaska supports a rich and varied flora and fauna including many unique organisms. Indeed, mammals alone are represented by 27 recognized endemic taxa (MacDonald and Cook 1996).

There are several uses and values associated with the indigenous fauna of the Forest, including commercial and noncommercial commodity values, subsistence use, maintaining Native cultural traditions, sport hunting, wildlife viewing, recreation and tourism, and ecosystem integrity. Thus, maintaining the diversity and viability of fauna within the Tongass meets many needs.

National Forest Management Act regulations state that “wildlife habitat shall be managed to maintain viable populations of native and desired non-native vertebrate species” (USDA Forest Service 1982:19). Unfortunately, there currently is not a clear definition in the scientific literature of what viability represents among all wildlife species. The National Forest Management Act regulations define a viable population for planning purposes as “one which has the estimated numbers and distribution of reproductive individuals to ensure its continued existence is well distributed in the planning area.” Furthermore, “habitat must be provided to support, at least, a minimum number of reproductive individuals and that habitat must be well distributed so that individuals can interact with others in the planning area.”

Concerns about long-term viability of wildlife in the Tongass led the leader of the TLMP Team to establish an interagency viable population committee (V-POP) in 1990. The committee's draft report (Suring and others 1993) provides recommendations for old-growth forest retention designed to protect long-term wildlife viability in the Tongass. The committee also made recommendations designed to maintain the viability of individual wildlife species. The committee's report was subsequently subjected to review (here called the peer review), at the request of the Regional Forester, by scientists affiliated with the Pacific Northwest Research Station (Kiester and Eckhardt 1994).

An attempt was made to reconcile differing recommendations made in the original V-POP report and the peer review to provide a consistent set of recommendations for an old-growth retention strategy to provide habitat sufficient to meet viability needs for all wildlife species in the Tongass. Preliminary recommendations for reconciliation were made (Suring and others 1994) but they were never subjected to scientific analysis or further review because the V-POP committee was disbanded by the Regional Forester.These documents articulated the need for a strategy to maintain habitat for old-growth-associated wildlife in the Tongass and made recommendations for ways to achieve that goal. The V-POP documents, and many others pertaining to wildlife viability, were provided to forest managers for consideration in revision of the Tongass land management plan.

The V-POP reports contained detailed recommendations on how, in the opinion of the authors, to achieve wildlife viability in the Tongass National Forest. The recommendations, however, are not science but rather a combination of scientific information integrated with professional experience and personal values embodying a level of risk the authors thought appropriate. Recommendations of this type cannot be subjected to a science consistency evaluation using the criteria defined in this paper. Because of interagency interest in the V-POP documents, however, recommendations in those documents are compared to the Forest plan in appendix 2. This exercise is simply a comparison of one possible strategy with another for managing wildlife habitat in the Tongass.

In general, any major disturbance to the landscape, especially in highly fragmented archipelago systems, poses some increased level of risk of local extripation to the resident biota (Burkey 1995). Generally, the more the disturbance processes deviate from the natural disturbance regime, the greater the likelihood of an increased risk to the viability of the resident biotic communities. When the areal extent, distribution, or frequency of a disturbance becomes great enough, and fragmentation of habitat extensive enough, then the ability of a species to persist across the landscape may be compromised. An estimate of temporal and spatial disturbance to habitat resulting from implementation of the Forest plan therefore is assessed in terms of risk to the viability of wildlife species of the Tongass and the probability of populations remaining well distributed across the planning area.

Factors influencing wildlife viability—Although population viability is difficult to define for many wildlife species, wildlife biologists clearly recognize the primary factors influencing viability (Soulé and Wilcox 1980). Numerous factors contribute to the viability of wildlife populations, not the least of which are the quantity, quality, and distribution of habitat, in both time and space. Habitat provides necessary resources such as shelter, food, and water, and habitat quality significantly influences susceptibility to

predation, disease, and severe climate. Ultimately, habitat determines reproductive success, recruitment, and demography, all important parameters influencing viability (Soulé and Wilcox 1980).

In addition to habitat, environmental features such as geologic history and natural processes, which affect regeneration and development of natural communities, establish the ecological and evolutionary context within which organisms must respond and ultimately adapt. More specifically, how the environment influences a species will differ and depends on several factors: trophic position (e.g., primary consumers like herbivores versus top level carnivores); dispersal ability; and any one of numerous life history traits including whether a species is migratory, whether a species is monogamous, generation time, age at first reproduction, number of litters or broods of young produced per year, and ecological affinities or interspecific relations.

Numerous species have very narrow habitat niches (specialists), or they may be involved in obligate relations with other species (e.g., predator-prey relations). For habitat specialists, the range of tolerance to habitat disturbance is narrow; for species dependent on others for food, viability is linked not only to its own habitat requirements but also to those of its prey. Also, whereas species occupying early forest successional seres (or other ephemeral habitats) have evolved dispersal ability and other traits that facilitate colonizing new habitat that becomes available, old-growth-associated species have instead evolved keen competitive abilities (Pianka 1988). The consequence is that old-growth-associated vertebrate species are typically poor dispersers and do not fare well at reoccupying disturbed habitat following extirpation (MacArthur 1972:87; MacArthur and Wilson 1967:81), even though the habitat may recover in 200 to 300 years.

This life history attribute is especially problematic in archipelago systems where habitat is already naturally fragmented and “source” populations are already isolated (Burkey 1995). Management activities that fragment habitats within existing land masses and contiguous forest habitat will further increase the likelihood that populations may not only become locally extirpated but also that fewer source populations may exist, and opportunities for colonizing any suitable second-growth forest habitat may be diminished because habitat connectivity has been interrupted.

For endemic vertebrates with restricted ranges (i.e., one or a few islands), the risk to viability, indeed extinction, is greater than for widely distributed species (Burkey 1995, Soulé and Wilcox 1980). Habitat requirements of many old-growth associates not only include microhabitat and macrohabitat dimensions but also are spatially explicit within a broader landscape context; that is, some endemic species have specific stand-level (and even finer scale) habitat requirements and are also “area-sensitive” (Soulé and Wilcox 1980). Thus, landscape context is as important as overall habitat quantity or quality. Small carnivores like marten, for example, have some of the largest mass-specific home ranges and need fairly large, contiguous, tracts of habitat (Chapman and Feldhamer 1982, Harlow 1994).

Land management planning for the Tongass National Forest occurs at the scale of millions of acres (USDA Forest Service 1996a), which is a broader scale than that used by wildlife populations that rely on habitat distributed at a spatial scale an order or more in magnitude finer than the Forest, or even an Administrative Area. Indeed,

many rare and endemic mammals have entire distributional ranges representing only thousands of acres. For example, the entire known geographic range of the Suemez Island ermine (*Mustela erminea seclusa*) is Suemez Island (MacDonald and Cook 1996), which is less than 40,000 acres.

Elements of Forest planning that affect the viability of old-growth-associated wildlife species are (1) total amount of habitat, habitat type representation, and habitat patch size; (2) extent of additional fragmentation of productive old-growth habitat; (3) connectivity of old-growth habitat; (4) extent, frequency, and intensity of disturbance within the matrix (e.g., rotation length, regeneration method, distribution in time and space); and (5) amount, distribution, and duration of roads and road access.

The draft alternatives and the emerging final alternative used different combinations of the above elements to address the issue of wildlife viability. The basic premise of this approach was that similar levels of protection can be achieved through wildlife management strategies employing different combinations of the elements. For example, similar levels of risk for a certain species may be achieved by using either a strategy emphasizing well-connected reserves, or one with no specific reserve strategy that uses a 300-year rotation across all lands scheduled for timber harvest, especially if harvest occurs in small blocks emulating natural disturbances.

An overview of wildlife components in the Forest plan—The Forest plan includes an array of components derived from available scientific information that managers have used to protect wildlife viability across the Tongass. These management actions reduce the risk to wildlife viability; additional actions could have been taken to further reduce risk and were not, presumably to gain other benefits. Whether the risk in the Forest plan is acceptable is a policy question, not one of science. The TLMP scientists evaluated the risk posed to maintaining habitat for well-distributed wildlife populations in the Tongass across the entire array of alternatives so that managers could see how the emerging final alternative compared to the draft alternatives (see appendix 3). The key components of the viability strategy in the Forest plan include:

- An array of lands totaling 4.2 million acres that is legislatively withdrawn from timber production (USDA Forest Service 1997b:table 3-77).
- An old-growth habitat conservation strategy that includes a network of large, medium, and small reserves, which in combination with various other LUDs protect an additional 3.5 million acres of old-growth forest habitat (Iverson 1997).
- Strategic location of reserves across the 21 biogeographic provinces of the Tongass to enhance their connectivity to other reserves or protected habitat; the system includes at least one very large old-growth reserve per ecological province and a small reserve in each of 237 large watersheds (USDA Forest Service 1997c).
- A mechanism for controlling human access to roads across the entire Forest, as necessary, to protect wolves and brown bears.
- A 1,000-foot beach and estuary buffer that provides an additional 140,000 acres of protected productive old-growth habitat within timber production LUDs (Iverson 1997) and connectivity among some reserves.

- Riparian buffers that protect an additional 223,800 acres of productive old-growth habitat within timber production LUDs (Iverson 1997) and provide connectivity among some reserves while providing protection for fish habitat.
- Approximately 500-foot buffers to protect principal brown bear feeding areas along salmon streams and provide connectivity with other old-growth habitats.
- Removal of all forested islands ≤1,000 acres (n=461) from the timber base to protect about 43,000 acres of productive old-growth habitat (Iverson 1997).
- A reevaluation of candidate species to include on the Regional Forester's sensitive species list.
- Strengthening the rigor of procedures and standardizing protocols to conduct biological evaluations for taxa on the Regional Forester's sensitive species list.
- Establishment of a "survey before management" approach for endemic mammals before beginning projects that substantially alter vegetation on islands ≤50,000 acres in size. Similar surveys on larger islands may be conducted if an initial assessment indicates a high likelihood that endemic mammals are present. Survey results will be used as necessary to modify project design to provide for the continued persistence of endemic taxa identified during surveys.
- Accelerated research on endemic mammal taxa in the Forest.
- A timber harvesting plan that schedules <43 percent of the acres of old-growth timber in the lands available for timber harvest between reserves (timber-emphasis LUDs, also called matrix lands) for harvest in a rotation, leaving after 100 years of implementation about 1 million acres (i.e., 57 percent) of the productive old growth in the matrix (Iverson 1997) and 84 percent of the Forest-wide productive old-growth habitat that was present in 1954 (USDA Forest Service 1997c).

After a full rotation of implementation (100 years), the Forest plan leaves about 57 percent of the old-growth forest within the portions of the Tongass National Forest designated for timber emphasis which are either protected by standards and guidelines or will be classified as too isolated or difficult to use for timber production. Scheduled timber harvests, however, contribute to additional fragmentation of forest habitat for old-growth-associated wildlife species.

Management actions that reduce risk to viability—The Forest plan includes several standards and guidelines that reduce risk to the viability of all wildlife species, including a subgroup called other terrestrial mammals.[3] Reduction in risk is measured from that associated with current practices in use on the Tongass; i.e., before the 1995-97 planning effort. These actions include many fundamental elements used in developing the array of Forest plan alternatives. These elements and their likely effect on each wildlife species are summarized in table 3.

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[3] The classification "other terrestrial mammals" includes a group designated as "wide ranging" and an endemic group. Several species are contained in each group (Shaw, in prep.).

Table 3—Synthesis derived from the 1st set of risk assessment panels on how various elements of the draft alternatives contribute to maintaining wildlife viability in the Tongass National Forest[a b]

		Alternative features				
Panels	Reserves	Rotation length in years	Harvest method [c]	VCU[d] harvest[e] thresholds	Old-growth retention[f]	Riparian options
Wolf	+	200>100	ul>2>es	+	+	1>2
Bear	+	0/-(differs by situation)	0	-		++1>2
Murrelet	++	200>100	ul>2>es	++	+	0 or -
Northern goshawk	+	200>100	ul>2>es	++	0/+	1>2
American marten	++e.g. Alternative 3	200>100 to clearcut	ul>2>es	+	+	1>2
Other terrestrial mammals	++	200>100 to clearcut	ul>2>es	+	+	1>2
Subsistence deer harvest	+ (community specific)[g]	200>100	ul>2>es	+	++	1>2
Old growth:						
Abundance and diversity	++	200>100	ul>2>es	++	1>2	
Connectivity	++	200>100	ul>2>es	+	1>2	
Process, structure, and function	+	200>100	ul>2>es	+	1>2	
Fisheries and riparian				+		

Table 3—Synthesis derived from the 1st set of risk assessment panels on how various elements of the draft alternatives contribute to maintaining wildlife viability in the Tongass National Forest[a b] (continued)

	Alternative features							
	Beach fringe 0-500 feet	Beach fringe 0-1000 feet	Estuary fringe	Deer winter range	Mixed matrix managed lands	Matrix and reserve[h]	Roads	Wild and scenic rivers
Wolf	+	+	+	+			-	
Bear	+	0	++	0	-	++	-	0
Murrelet	+	+	+	+			0	+
Northern goshawk	+	++	++		++	-	0	
American marten	+	++	++	+			0	
Other terrestrial mammals	+	+	+	+	-		-	
Subsistence								
deer harvest	+	++	++	++	0	0	-	0
Old growth:								
Abundance and diversity			+	+				0
Connectivity		+	+			-		
Process, structure, and function	+	+					-	
Fisheries and riparian							-	+

[a] Panel evaluation symbols:

0 neutral or benign

+ important positive feature

++ critical positive feature

- detriment

> better than

= comparable

No mark in a column indicates that panels made no comment.

[b] Swanston and others 1996.

[c] Harvest method: ul = uneven aged-management,rotation (≤200 yr); es = even-aged management, short rotation (100 yr); 2 = 2-aged management.

[d] VCU=value comparison unit, equivalent to an individual watershed.

[e] Refers to threshold to trigger “no further management” when VCU has experienced a predetermined percentage of harvesting (USDA Forest Service 1996a).

[f] Size and distribution of old-growth blocks was initially consistent across alternatives as determined by a 1995 GIS (geographic information system) inventory (USDA Forest Service 1995).

[g] Benefits derived for deer and other subsistence resources differ among communities.

[h] Refers to a management approach that uses a combination of alternatives to clearcutting in the matrix and a configuration of habitat reserves to achieve a desired condition for wildlife and other nontimber values (USDA Forest Service 1996a).

All wildlife species—Standards and guidelines and habitat elements designed to address all wildlife species excluding "other terrestrial mammals" include:

1. The identification and mapping of large, medium, and small old-growth reserves across the Tongass National Forest with consideration for historical distribution of habitat and connectivity among reserves. Reserves contain productive old-growth forest habitat types strategically distributed for wildlife purposes. After 100 years of implementing the Forest plan, about 3.5 million acres of the estimated 5.4 million acres of productive old growth present in 1954 will be retained through this old-growth habitat conservation strategy. There will be at least one large reserve (≥40,000 acres) in each of the 21 biogeographical provinces and one small reserve (≥1,600 acres) present in each of 237 large (>10,000 acres) watersheds. An additional 360,000 acres of productive old-growth habitat within timber production LUDs is protected by beach, estuary, and riparian buffers, and also contributes to landscape connectivity among reserves and other protected habitat (Iverson 1997).

2. A 1,000-foot, largely undisturbed beach fringe buffer, of which 60 percent is productive old-growth forest. Historically, limited partial logging and some clearcut logging occurred in the beach buffer; however, 93 percent of its total acreage and the productive old growth remain largely unaltered. Below are examples of wildlife species benefiting from this component.

   a. Queen Charlotte goshawks benefit because the beach fringe is highly utilized and preferred old-growth habitat (Iverson and others 1996).

   b. American marten benefit from the additional connectivity across lowland old-growth habitat provided by undisturbed beach fringe buffers as well as an addition of high-quality habitat.

   c. Marbled murrelets benefit from an increase in interior-forest nesting habitat across the landscape by adding undisturbed old-growth forest habitat near existing, contiguous old-growth habitat; murrelets also may nest within the beach fringe buffer.

   d. Beach fringe represents important winter range habitat for Sitka black-tailed deer, the predominant prey of the Alexander Archipelago wolf. Undisturbed beach fringe increases connectivity across lowland productive old-growth habitat.

   e. Breeding density of bald eagles (*Haliaeetus leucocephaius*) is reduced when nests are within about 1,000 feet of clearcuts; within the 1,000-foot proximity, some benefit is afforded to eagles that nest further from clearcuts (Gende and others, in press).

3. A 1,000-foot undisturbed estuary buffer benefits all old-growth-associated wildlife species. Below are some examples of wildlife species that would particularly benefit from this action.

   a. Queen Charlotte goshawks, because the estuary fringe is highly used and a preferred old-growth habitat (Iverson and others 1996).

   b. American marten, from the additional connectivity across lowland old-growth habitat provided by an undisturbed estuary fringe buffer as well as an addition of high-quality habitat.

c. Marbled murrelets, from an increase in interior-forest nesting habitat across the landscape by adding undisturbed old-growth forest habitat near existing, contiguous old-growth habitat; murrelets also may nest within the estuary fringe buffer.

d. Alexander Archipelago wolves because the estuary fringe represents important winter range habitat for Sitka black-tailed deer, the predominant prey of the Alexander Archipelago wolf. Undisturbed estuary fringe increases connectivity across lowland productive old-growth habitat and to upland habitat, which serves as summer or winter range, and lowland winter range.

e. Breeding density of bald eagles in southeast Alaska is reduced when nests are within about 1,000 feet of clearcuts; a 1,000-foot estuary buffer reduces the likelihood of some bald eagles nesting within 1,000 feet of a clearcut (Gende and others, in press).

4. Brown bears benefit from provisions to provide about 500-foot undisturbed buffers as needed along salmon streams, which function as principal feeding areas, and from protected estuary fringes, especially those associated with class I anadromous fish streams, which serve as principal feeding areas. These buffers also are potential corridors for wildlife movement among designated reserves.

5. Brown bears and wolves benefit from the interagency process to determine whether road access markedly contributes to mortality and from implementing an effective road management plan where necessary (USDA Forest Service 1997c).

6. The Forest plan protects 86 percent of the highest quality deer winter range in the Tongass National Forest, which is more than all draft alternatives except alternative 1—the no-additional-management alternative (USDA Forest Service 1997b:table 3-111). Winter range is included within the 1,000-foot beach fringe, riparian buffers, large, medium, and small old-growth habitat reserves, legislated reserves, and other large reserved areas such as south Cleveland Peninsula and South Kuiu Island that result in scheduling a relatively small proportion of old growth for timber harvest (474,000 acres of old-growth forest over the next 100 years).

7. Applying riparian standards and guidelines as consistent with the AFHA report (1995) provides some reduction of risk to wildlife viability. The benefits accrue primarily as corridors for movement because the buffers do not represent “habitat” for most old-growth-associated wildlife species, particularly area-sensitive species (e.g., hairy woodpecker [*Picoides villosus*], marbled murrelet). Species that would perhaps benefit from riparian movement corridors include American marten, brown bear, Sitka black-tailed deer, and the Alexander Archipelago wolf.

8. Several standards and guidelines under Wildlife and Threatened, Endangered, and Sensitive Species (USDA Forest Service 1997a: chapter 4) were added to strengthen the emerging final alternative in response to the 1997 risk assessment panel results and other comments. In general, these additions were made to improve habitat conditions for the northern goshawk and the American marten in programmed harvest units within high-priority habitat areas for each species. Guidelines detail various actions to maintain more old-growth character deemed important to these species in cutting units. Additional measures were included to further evaluate any increases in American marten mortality (e.g., as may occur from hunting and trapping) where such

mortality is identified as posing a serious threat to marten populations. Another addition is intended to provide additional assurance of maintaining connections among habitat blocks throughout the Tongass. Where existing conditions (e.g., riparian and beach buffers) are determined to not provide for adequate connectivity, additional habitat will be allocated to provide for connectivity of old-growth habitats. It is unclear how determinations will be made as to the sufficiency of existing conditions to provide adequate connectivity.

Wildlife habitat elements noted below likely would provide increased incremental and cumulative benefits toward maintaining wildlife viability but are absent from the Forest plan. The benefits afforded each species probably differ and are difficult to quantify. Absence of these elements is not an indication that the Forest plan is deficient, and the inclusion of these or any other elements does not ensure that a Forest plan meets acceptable levels of risk for maintaining wildlife viability. That is a policy decision. Appendix 3 includes an evaluation of the risks to wildlife viability of implementing the emerging final alternative for 100 years. The wildlife habitat elements are as follows:

1. Application of all riparian prescriptions from draft alternative 3, which included increased protection of specific, high-value watersheds identified by the Alaska Department of Fish and Game (option 1), as well as application of AFHA (1995) recommendations (option 2).

2. Riparian buffers and movement corridors that are wide enough (e.g., some biologists suggest corridors of 600 feet or more) to allow unrestricted movement of marten and other far-ranging wildlife species among old-growth forest blocks and provide sufficient forest-interior habitat to support breeding of area-sensitive species (e.g., hairy woodpecker). Buffers for protection of streams and riparian zones, as proposed, are generally too narrow to provide any substantial interior habitat on class I streams. Although additional measures to enhance connectivity for wildlife, particularly small mammals (USDA Forest Service 1997b:app. N, p. 10) and marten (USDA Forest Service 1997b:app. N, p. 14) seek to minimize edge and include interior-forest habitat, these measures lack clarity as to how the connectivity corridors will be designed to contain sufficient interior habitat to support successful reproduction of area-sensitive species. Broader corridors also would likely benefit wolves by increasing Sitka black-tailed deer winter range and by reducing human-caused mortality of wolves through an increase in escape cover along roadway buffers.

3. More productive riparian old-growth forest habitat than proposed in the Forest plan, a highly used, preferred habitat of goshawks. Riparian-associated forests are equally or more productive in supporting bird and mammal populations, which are important prey species for goshawks.

4. Broader riparian buffers than proposed in the Forest plan, along streams associated with estuaries near marine foraging habitat important for marbled murrelets. This would contribute important interior-forest habitat for nesting murrelets, which are vulnerable to increased nest predators associated with forest edge habitat.

5. Retention of all deer winter range in watersheds where deer harvest exceeds 20 percent of deer habitat capability (applying deer standard and guides from draft alternative 3; USDA Forst Service 1996a), and retention of important winter range where deer harvest is between 10 and 20 percent of deer habitat capability. This action would be particularly important in portions of the forest with wolves, where both wolf viability and human subsistence use of deer are focal issues.

6. Protection of more highly preferred lowland productive old-growth habitat than offered in the Forest plan, for American marten and goshawk nesting and foraging.

7. Retention of all deer winter range (e.g., productive old-growth forest) below 800 feet in elevation. Besides benefiting deer, this measure also would contribute to available interior-forest habitat for nesting murrelets and other area-sensitive species, which are vulnerable to increased nest predators associated with forest edge habitat.

8. Applying a greater proportion of longer rotation and uneven-age management within the forest matrix among reserves than that in the Forest plan would be especially beneficial to the old-growth-associated species noted in items a and b, below.

> a. Applying long rotations and additional uneven-age management in all matrix lands rather than only those in certain locations (USDA Forest Service 1997a, 1997b:app. N, p. 15) would more closely approach a dynamic landscape circumstance than current practices and thus likely reduce the viability risk to goshawks (Iverson and others 1996).
>
> b. Applying long rotations and additional uneven-age management in all matrix lands rather than only those in certain locations (USDA Forest Service 1997a,1997b:app. N, p. 15) would increase potential nesting habitat for marbled murrelets because it will allow for the development of a more contiguous, mature forest habitat with stems that are old and large enough to possess the features typical of marbled murrelet nest trees (Ralph and others 1995).

Other terrestrial mammals—There are 27 currently recognized “unique” taxa of mammals that occur in southeast Alaska, all of which are known from the Tongass National Forest (MacDonald and Cook 1996). The taxonomic diversity of terrestrial mammals (and presumably other faunal elements) mainly results from the high degree of naturally fragmented landscapes in the Tongass. Indeed, across the Alexander Archipelago, isolated habitat types historically existed within oceanic islands because of topography, geology, climate, and other environmental features. Because of this situation, both spatial and temporal heterogeneity occur in a manner rarely found elsewhere. A consequence of this environmental context is isolation of vertebrate populations, reduced gene flow, and ultimately through time, separate and distinct breeding populations that are genetically isolated and become recognized as unique taxa.

Many distinct taxa (e.g., Prince of Wales flying squirrel [*Glaucomys sabrinus griseifrons*]) have restricted geographic distributions; including one (Suemez Island ermine) known from only a single island. The smaller a population is, the more likely it will be negatively impacted by habitat disturbance and possibly become extirpated; the fewer the populations, the more likely a taxon will become extinct. Clearly, extinction rates are higher in archipelago systems (Burkey 1995).

Habitat disturbances like timber harvest, particularly clearcutting, can further fragment wildlife populations, which can reduce the size of contiguous populations and the number and distribution of larger populations, increase sensitivity to stochastic perturbations, and increase the likelihood that more isolated and smaller populations may become extirpated (Soulé and Wilcox 1980). What may represent a small proportional reduction in old-growth habitat across the Forest could represent a substantial portion, indeed perhaps the entire known habitat, of an endemic mammal’s range.

Both risk assessment panels for “other terrestrial mammals” (Shaw, in prep.; Swanston and others 1996) recognized the potential negative impacts of forest management on the viability of both “wide ranging” and “endemic” mammals in the Forest. Indeed, recognition by the first panel that some appreciable risk to viability already existed because of past management is reflected in the relatively high likelihood (67 percent) of at least one taxon “not being well distributed” in the alternative with essentially no further management (alternative 1). For that reason, the panel provided options for reducing risk to taxa already impacted by past management (e.g., Prince of Wales flying squirrel) and suggestions for reducing the risk to viability from future management actions in the Forest.

The habitat elements noted above under “All Wildlife Species” also benefit “Other Terrestrial Mammals” as a group, including endemic terrestrial mammals. Additional measures specific to endemic mammals were provided by the panel and an analysis of available science information. The following three items are included in the Forest plan.

1. Protection of all islands ≤1,000 acres (USDA Forest Service 1997c) will reduce risks to viability of some small endemic mammals (see Lidicker 1994) such as voles (*Clethrionomys* and *Microtus*), mice (*Peromyscus*), and shrews (*Sorex*). But a conservation strategy for maintaining these species should not rely just on protection of these small islands (Burkey 1995).

2. Evaluation and designation of rare and endemic vertebrates as sensitive species where appropriate. To reduce risk, designated sensitive species require a biological evaluation (USDA Forest Service 1997c) as part of the National Environmental Protection Act (NEPA) process before any project implementation. Such biological evaluations (USDA Forest Service 1997c) will include an on-the-ground survey conducted according to acceptable protocols. Additional taxa could be included as Alaska Region sensitive species.

3. Commitment to further research on the distribution, abundance, detectability and genotypic variability of small endemic mammals in the Tongass (see USDA Forest Service 1997c).

Results of the 1997 risk assessment panels indicated concern for small, endemic, and widely distributed mammals (Shaw, in prep.). The following additional conservation measures that reduce viability risk to small, endemic, and widely distributed mammals were added to the emerging final alternative to form the Forest plan:

a. Surveys for endemic mammals are to be completed before projects are begun that would substantially alter vegetation on islands of ≤50,000 acres. Surveys will be conducted on larger islands if initial assessment indicates high likelihood that endemic mammals occur there.

b. Where endemic taxa are detected by surveys, projects will be designed to provide for continued persistence of these taxa.

c. Ongoing research of endemic taxa within the Tongass will be accelerated.

d. Endemic and widely distributed small mammals benefit from conservation measures to maintain connectivity of large and small reserves and other nondevelopment LUDs.

e. Endemic and widely distributed small mammals benefit from guidelines to increase old-growth character for marten and goshawk habitat.

The following habitat elements likely would provide incremental and cumulative benefits toward maintaining viability of other terrestrial mammals but are absent from the Forest plan. These elements were identified in the viability panel assessments and by analysis of available science information. The benefits afforded each species in this group probably differ and are difficult to quantify. The absence of these elements is not an indication that the Forest plan is deficient, or does including these or other elements ensure that a Forest plan meets acceptable levels of risk for maintaining wildlife viability. That is a policy decision. Appendix 3 includes an evaluation of the risks to wildlife viability of implementing the emerging final alternative for 100 years. These elements include:

1. For endemic and rare taxa with restricted ranges, application of an old-growth-retention prescription across the entire range, rather than a survey and manage approach on islands ≤50,000 acres, until additional study concludes that the geographic distributions or habitat requirements do not warrant this degree of protection.

2. Restoration of habitat further fragmented because of management actions within the existing range of certain endemics (e.g., the Prince of Wales Island flying squirrel on northern Prince of Wales Island).

Restoration could include delaying, preventing, or possibly improving habitat through subsequent entries into clearcuts. Or it could limit future harvest to small group selections so that older (i.e., ≥250 years) forest habitat would be preferentially retained in landscapes where it currently coexists in a highly fragmented matrix of recent clearcuts, seedlings, young growth, or young sawtimber stands. This action would reduce the risk to maintaining well-distributed populations by increasing connectivity as well as by increasing the total amount of mature forest habitat (i.e., well beyond the understory reinitiation phase; Lawford and others 1996) within the range of the species.

Consistency in use of wildlife-related information—Wildlife viability was identified as a major focal issue to be addressed during revision of the Tongass land management plan. Consequently, the manner in which wildlife information was used in developing the emerging final alternative is critically important. New assessments completed for anadromous fish (AFHA 1995), marbled murrelet (DeGange 1996), Queen Charlotte goshawk (Iverson and others 1996), and Alexander Archipelago wolf (Person and others 1996), the existing V-POP documents (Kiester and Eckhardt 1994; Suring and others 1993, 1994) , and other documents (Iverson and René 1997) were used in developing the wildlife viability strategy in the Forest plan. A summary of how scientific information on key wildlife viability issues was used in the Forest plan and how risks were acknowledged and documented is provided in table 1 (see “Summary”).

Overall, the consistency criteria were met. The Forest plan makes consistent use of available scientific information and provides a strategy that reduces the risk of not maintaining well-distributed wildlife populations across the planning area to a level lower than any of the original draft alternatives, except draft alternatives 1 or 5, depending on the species.

A science consistency check of recommendations within V-POP documents (Kiester and Eckhardt 1994; Suring and others 1993, 1994) is not appropriate, but a general comparison between V-POP recommendations and the Forest plan is made in appendix 1. Recommendations from the V-POP reports were designed to produce a wildlife viability strategy with a reasonable probability of maintaining well-distributed, viable populations of all wildlife species across the planning area. These recommendations, as previously noted, are not science but a combination of scientific information, professional experience, and the personal values of those making the recommendations and therefore are not suitable for a science consistency check.

The Forest plan, which seeks to maintain wildlife viability through a somewhat different strategy than that recommended by the V-POP report (Suring and others 1993) and the peer review (Kiester and Eckhardt 1994), differs in many respects from the detailed species-specific recommendations of these V-POP documents (see appendix 1). The levels of risk to maintaining habitat for well-distributed populations of wildlife across the planning area, therefore, might be different. But whether the level of risk embodied in the Forest plan is acceptable is not an issue for science. It is a public policy decision, the making of which should benefit from available scientific information.

Karst and Cave Protection

Introduction—The Tongass National Forest contains the largest concentration of dissolution caves known in Alaska. The Forest also contains unique karst landform features, particularly epikarst (surface karst), concentrated primarily in the alpine and subalpine zones. The caves and the karst landform features are intimately related and a direct manifestation of chemical weathering of limestone and marble bedrock underlying approximately 437,400 acres of the Tongass in a northwest trending band extending across the Forest. These karst and cave resources are a newly discovered and recognized attribute of the Tongass National Forest. They are of national and international significance for many reasons, including their intensity and diversity of development; biological, mineralogical, cultural, and paleontological components; and recreational values (Aley and others 1993, Baichtal and Swanston 1996).

Karst landforms—The basic principles of karst development and cave formation are known internationally and are well documented (Ford and Williams 1994, White 1988, White and White 1989). The interactions of the variables controlling the rate and extent of karst landscape and cave development, however, are not fully understood, particularly as they relate to development of these features under the cool, moist, heavily forested conditions along the north Pacific coast. In southeast Alaska, karst landforms comprise a geologic terrain developed atop and within carbonate bedrock in which unique caverns, tunnels, and surface features have developed as the result of differential solution by acid surface waters and ground waters (Baichtal and Swanston 1996). The surface of the terrain is characterized by distinct erosional features such as deep shafts, crevasslike dissolved fissures, erosion or dissolution rills, and spikes and spires of nearly pure limestone and marble. Surface drainage in such terrain is infrequent, with surface water moving rapidly underground and into complex subsurface drainage networks associated with extensive cave and underground tunnel systems. Once the water and any associated sediment enters the underground drainage system, the breadth of distribution and points of discharge are unpredictable and may extend far beyond surface watershed boundaries.

Significant features—Field observations indicate a definite direct association between presence of karst terrain and the productivity of the western hemlock-Sitka spruce (*Tsuga heterophylla* (Raf.) Sarg.-*Picea stichensis* (Bong.) Carr.) forests found

in the Tongass (Baichtal and Swanston 1996). The controlling variables seem to be the nutrient richness of soils on the valley floors and along slopes below 400 feet in elevaton, well-developed subsurface drainage, and dissected bedrock surfaces that allow tree roots to hold fast and become windfirm. Exceptionally dense stands of very large-diameter western hemlock and Sitka spruce are characteristic of these low-elevation sites, and the old-growth forest cover provides a well-structured, multi-layered canopy important as winter habitat for Sitka black-tailed deer and other mammals.

Streams draining these karst areas support extremely productive aquatic communities. The geochemistry associated with karst development contributes to productivity of aquatic environments through its carbonate buffering capacity and carbon input dissolved from the limestone bedrock (Wissmar and others 1997). This action has significant downstream effects on the aquatic food chain and biotic community. Preliminary studies suggest that aquatic habitats associated with karst landscapes may be 8 to 10 times more productive than adjacent, nonkarst-dominated aquatic habitats (Bryant and others, in prep.). The karst-dominated aquatic habitats also appear to support a higher biodiversity than the noncarbonate-based systems, have higher growth rates for salmon smolts and resident fish, reflect less variable water temperatures and flow regimes, and contain unique habitats affecting species distribution, abundance, and adaptations.

Many wildlife species find the epikarst features, the stable environment, and the shelter provided within associated caves to be suitable habitat. Caves have been used as natal denning sites by northern river otter (*Lutra canadensis*) and as resting and denning sites for Sitka black-tailed deer, bear, wolf, and small furbearers (Baichtal 1993b, Baichtal and Swanston 1996). Sitka black-tailed deer are known to rest around cave entrances in summer, when the air coming from the caves is cooler, and in winter, when the cave entrance environment is warmer than surrounding terrain. Cave systems are known to provide critical roosting and hibernating habitat for bats (Baichtal 1993b, Baichtal and Swanston 1996, Parker 1996). Several birds such as the water ouzel (*Cinclus mexicanus*), thrush (Turdidae), and swallow (Hirundinidae) use cave entrances for nesting and feeding. Rookeries for sea birds, including cormorants (Phalacrocoracidae) and pigeon guillemots (*Cepphus coluba*), occur in some littoral caves (Baichtal 1993a, 1993b; Baichtal and Swanston 1996).

The potential cultural and paleontological significance of the karst landscapes also is high. The Pleistocene paleontology and late Pleistocene and Holocene history of the area associated with carbonate terrain in southeast Alaska is primarily known from cave and rock shelter deposits, which often are intimately related to significant archeological sites. The cool, stable, basic environmental conditions in the caves result in exceptionally good preservation of bone and organic materials (Aley and others 1993). Recent work on Prince of Wales and surrounding islands on the extensive karst resources, combined with botanical surveys of alpine areas and genetic studies on chum salmon populations (Kondzela and others 1994), strengthen the argument for a well-developed coastal refugium along the west coast of the southern end of southeast Alaska. The evidence sheds new light on our understanding of glacial chronology, climatic change, biogeography, and archeology along the western margin of North America (Autrey and Baichtal 1992, Dixon and others 1992, Heaton and Grady 1992).

The functions and biological and cultural significance of karst landscapes are only just beginning to be understood. Preliminary investigations suggest an increased productivity for plant, animal, and aquatic communities that develop on top of or otherwise

benefit from karst landscapes. Developing paleontological and archeological information, derived from bones, cultural artifacts, and sediment layers in caves and rock shelters, is providing growing evidence of a coastal refugium through which Pleistocene mammals and humans may have migrated south from the Bering Sea landbridge. Based on these preliminary findings alone, the significance of the karst and cave resources in the Tongass, and the importance of careful forest management on this fragile ecosystem, is apparent.

Quantitative information on the effects of forest harvest on karst ecosystems in southeast Alaska is limited. Recent sediment deposits and waterline marks in caves and underground systems suggest that timber harvesting has increased sediment and debris transport and flooding of underground passages that had not been flooded for centuries. Many cave entrances are filled or blocked by logging slash, sediment, and debris. Field observations and aerial-photo interpretation also show strong evidence of greatly increased surface runoff on karst landforms and adjacent surfaces after harvest, which increases sediment, nutrient, and debris transport capability of associated drainage networks. The vulnerability of a particular karst system to harvest activity is related to the openness of the system. The many solution-widened fissures, sinkholes, and cave entrances become deposition areas for slash, sediment, and debris and injection points into the complex subsurface drainage system. These fissures rapidly move water, sediment, and debris vertically downward into the underground lateral systems. Material transported from roads and disturbed lands may emerge unexpectedly at one or more distant springs or resurgence channels, altering the chemical, nutrient, and sediment loads of the receiving streams.

We and other Tongass land management planning team members analyzed and assessed the available literature and the most up-to-date technical information on karst development and solution cave formation, as well as current thinking on karst management. An independent assessment of karst and cave resource significance for the Ketchikan Area (Aley and others 1993) provided additional quantitative information on processes and controlling variables operating in the area of maximum karst development in the Tongass. Information from active research and monitoring programs designed to measure and analyze biological and cultural interactions with karst terrain and ongoing Alaska Region inventories of karst landscapes provided information used for defining important karst terrain characteristics and significant areas of management concern Forest-wide.

Results of these analyses and assessments and the emerging field information were used by managers to develop a detailed karst and cave management strategy for the Tongass that incorporates methodologies to identify sensitive terrain and assess risk from forest harvest activities. The methodology also provides standards and guidelines for overall management of karst landscapes.

Managers gave full consideration to the scientific information in developing the Forest plan. The entire management strategy developed by the team, with input from the PNW scientists, has been incorporated into the final plan (USDA Forest Service 1997a:4-17 to 4-19 and app. I; 1997b: 3-82 to 3-86). Incorporation of this strategy into the plan and subsequent discussions with managers and field personnel on application have demonstrated that the information is understood and correctly interpreted. Resource risks are acknowledged and accounted for in the application of the strategy. Overall, the three consistency criteria were met (table 1).

Slope Stability

The Tongass National Forest is characterized by steep slopes, shallow, permeable soils, and exceptionally high rainfall—factors contributing significantly to unstable conditions in this mountainous terrain. Because of high soil permeabilities, slope drainage is primarily by subsurface flow with little or no surface flow outside established channels. When surface flow does occur, the thick mat of forest humus and plant cover is adequate to protect the mineral soil from surface erosion. During major storms, high soil moisture levels, local areas of saturation, and temporary water table development are produced on the slope, greatly increasing the unstable character of the soils. This combination of events can cause mass erosion events (landslides) to occur.

Landslides, involving the downslope movement of soil, primarily under the force of gravity, are the principal processes of hillslope erosion and sediment transport to channel systems, and constitute a major management concern for the Tongass National Forest. A critical balance exists between slope stability and slope failure on these unstable lands. The anchoring effect of root growth through the thin soil and into joints and fractures in the bedrock probably increases the stability of these soils somewhat, as does the cohesion provided by organic colloids. The effectiveness of these stabilizing factors, however, may be reduced during periods of high rainfall by development of seepage and pore-water pressures associated with rising soil-water levels. Under these conditions, the soils are in their least stable condition, and only a small triggering force is required to cause total failure and rapid downslope movement of the soil mass. Such a force can be produced by sharp increases in soil-water content, rapid increase in soil mass, and direct destruction of stabilizing root masses.

Landslides are a natural disturbance process in southeast Alaska, but human activities can affect the rate, size, and location of landslide events. Roads and timber management directly influence stability of slopes across the Tongass National Forest through disturbance of the soil mantle. The result is an increase in soil mass movement or landslides, which remove productive soil layers on slopes and transport large quantities of sediment and large woody debris into anadromous fish streams.

Forest managers recognized the significance of this relation and developed a mass failure hazard indexing methodology in 1987 (Alexander 1987). The methodology was developed at two scales. At the larger scale, the index is used to assess risks related to the amount and methods of timber harvest and extent of road construction at the Area and project levels. At this scale, the methodology is primarily field oriented and is based on physical characteristics of soil units and on measured or estimated slope, drainage, and landform characteristics that control the stability of soils. At the smaller scale, the landscape or Forest planning level, index values based on geographic information system (GIS) soil mapping units and slope gradient values developed from digital elevation models (DEM) were used to identify lands with a potentially high hazard for landslide development. These lands were then removed from the “tentatively suitable” timber base.

Initial inspection of the 1987 methodology indicated that although the methodology is sound, the database from which the indexing values were derived was out of date and there was a lack of consistency in identification, measurement, and interpretation

of variables across Administrative Areas of the Forest. A careful reanalysis and assessment of slope stability literature and the most current data, as well as recently developed field information from southeast Alaska, have resulted in a revision of the methodology to provide a current and standardized base for Forest-wide stability hazard evaluation (Swanston 1997). The approach directly addresses a concern and recommendation of AFHA (1995). The analyses also have provided a reliable and defensible slope gradient boundary for defining highly unstable terrain.

Managers and resource specialists in the field have been closely involved with development of this revised methodology. The critical field gradient for soil stability has been redefined at 72 percent, based on these analyses. Slopes greater than this gradient are not suitable for timber harvest because of the high potential for landslides after harvest.

Digital elevation models analyzed with GIS can markedly underestimate actual slope steepness. At the Forest planning level, lands above this gradient (as defined by the DEM) therefore are removed from the tentatively suitable timber base with the caveat that withdrawal of these lands should be verified at the Area and project level by suitable field analysis. This criterion is in close agreement with the recommendations of AFHA (1995). Managers have used this information in developing the Forest plan. The redefinition of the high-hazard boundary as a 72-percent slope has been incorporated into the latest GIS analyses and has resulted in improved accuracy for delineation of tentatively suitable forest lands in the Tongass. The new information and revised methodology is understood and has been correctly interpreted in final plan development (USDA Forest Service 1997a:4-81 to 4-84; 1997b:3-197 to 3-201). The revised methodology clearly defines resource risk. Overall, the three consistency criteria were met (table 1).

Timber Resources

For purposes of the Forest plan, productive old growth within the Tongass National Forest is defined as a previously unharvested forested area of trees more than 150 years old with greater than 10 percent tree cover, and an accumulated net saw-log volume of more than 8,000 board feet per acre (bf/acre; Julin and Caouette 1997). The term “productive” refers to the utility of this forested area to provide wood products, although timber volume has been used in Forest planning as a surrogate for wildlife habitat capability. Many of the stands designated as productive old growth in the Tongass GIS database may be relatively young (i.e., 150 to 200 years old) as a result of natural windthrow cycles that prevent development of recognized old-growth attributes (Nowacki and Kramer, in prep.).

Within the boundaries of the 16.9-million-acre Tongass National Forest, there are about 10 million acres of forest land (>10 percent tree cover) and about 6.9 million acres of ice, rock, water, alpine meadow, fen, and sphagnum bog (USDA Forest Service 1997b). Forest conditions range from fens with scattered trees, to heavily timbered stands with volumes that sometimes exceed 70,000 bf/acre. An estimated 5 million acres of this forest is coniferous old-growth forest supporting accumulated levels of wood volume considered to be economical for harvest. About 580,000 acres of the National Forest supports young-growth stands regenerated after logging (69 percent) and natural disturbance (31 percent) (USDA Forest Service 1997b).

The species composition and spatial distribution of forested lands varies in a complex mosaic over the fragmented Alexander Archipelago landscape with changes in elevation, aspect, soil drainage, and patterns of disturbance. Western hemlock, Sitka spruce, Alaska-cedar (*Chamaecyparis nootkatensis* (D.Don) Spach), western redcedar

(*Thuja plicata* Donn ex D.Don), and mountain hemlock (*Tsuga mertensiana* (Bong.) Carr.) are harvested for a variety of traditional (Native and non-Native) and commercial uses. Western hemlock and Sitka spruce are the most abundant tree species, comprising 64 percent and 28 percent of the total growing stock, respectively (Harris and Farr 1974). Alaska-cedar is the most economically valuable, however, particularly in the export market.

Tlingit and Haida people have long used this timber for building houses, canoes, totem poles, and creating ceremonial, household, and personal objects. Non-Native timber use began with Russian colonization in the 1790s and was continued by the United States on a limited scale until the mid-1950s. Development of the Ketchikan Pulp Mill (1954), the Sitka Pulp Mill (1959), and their affiliated saw mills significantly increased timber harvest levels. From 1954 through 1995, we estimate that 17.2 billion board feet were harvested from the Tongass—an average of about 41.5 thousand board feet per acre (mbf/acre). The closure of the Sitka Pulp Mill and Wrangell saw mill in 1993 and the closure of the Ketchikan Pulp Mill in 1997 have markedly reduced capacity to locally process Tongass timber. Currently there are several saw mills and individuals operating in southeast Alaska producing dimension lumber, cedar shingles, specialty products (e.g., guitar faces and piano sounding boards), and cants for export to Pacific Rim markets.

The Forest plan schedules 676,000 acres of old- and young-growth timber for harvest over the next 100 years (USDA Forest Service 1997a). Young-growth stands are not programmed for harvest for another 50 to 60 years—coincident with the projected 95-percent culmination of maximum annual average stand growth rate. This schedule is a key factor controlling the rate at which remaining old growth is harvested. Under this management regime, the future commercial forest, if harvested as planned, may consist of relatively small-diameter trees more suited by current standards for fiber (e.g., pulp, fiber board) or laminated products than for dimension lumber.

A basic principle of National Forest System planning is that resources are to be managed in a sustainable fashion (U.S. Public Laws, Statutes 1960). Sustained yield is "the achievement and maintenance in perpetuity of a high-level of annual or periodic output of the various renewable resources...without impairment of the productivity of the land" (USDA Forest Service 1982). At its most general level, timber sustainability is determined by the availability of commercial timber and the rate of its use through time. Commercial timberland in the Tongass is first defined in terms of suitability for timber production. Once the extent of the timber base is defined, and the amount of available timber in young and old growth is estimated, harvest rate is defined in terms of allowable sale quantity, which must be below the long-term sustainability of the Forest. The actual rate of harvest is determined by numerous factors including but not limited to economics (timber value and operability), appropriated funds, and litigation.

The managers made many decisions concerning the availability, growth, and use of the timber resource in the Forest Plan. Below we discuss six important decisions concerning timber resources. These decisions include how (1) old-growth timber volume was estimated, (2) young-growth timber volume was estimated, (3) uncertainty in the size of the suitable land base was recognized, (4) allowable sale quantity was partitioned to promote economic sustainability (5) certain forested wetlands will be managed, and (6) effects of windthrow will be mitigated.

Decisions regarding the application of each feature of the linear programming model (FORPLAN; Johnson 1986) used in estimating Forest outputs are not fully explored here. Two reviews conducted at the Rocky Mountain Forest and Range Experiment Station raised various questions about the application of FORPLAN in the estimation of Forest plan outputs.[4] Some of these questions were addressed in appendix B of the FEIS (USDA Forest Service 1997b); other questions may warrant further attention. The managers recognized the uncertainty associated with FORPLAN estimates by prescribing a monitoring activity to review modeling assumptions used in FORPLAN every 5 years or whenever major changes in the assumptions occur.

Estimation of old-growth timber volume—The managers used a three-strata approach (low, medium, and high volume) to estimate the abundance of productive old-growth timber in the Forest (USDA Forest Service 1997b). In addition, they assumed that net growth over time in the old-growth component is equal to zero.

Brickell (1989) evaluated the spatial accuracy of the Tongass interpreted timber type data layer (TIMTYP map). This data layer contains mapped information about volume levels within the forested landscape. Using forest inventory data for validation, Brickell found no statistical basis for maintaining separate strata for the higher volume classes (volume classes 5, 6, and 7) on the TIMTYP map.

Julin and Caouette (1997) describe five options for estimating net saw-log old-growth volume in the Tongass. These options define mean timber volume strata by using the volume class item (volume classes 4-7) from the TIMTYP map alone and in combination with the forest type item from the TIMTYP map, and several items from the common land unit database (slope, hydric and nonhydric soils, and site index). Managers modified the low-, medium-, and high-volume strata developed by Julin and Caouette (1997) from a 16-foot net saw-log basis to a 32-foot net saw-log plus utility basis by applying their estimates for breakage, standing merchantable dead volume, and Tongass cut and sold reports (see footnote 4).

The managers seem to have correctly interpreted the available scientific information. The managers abandoned the use of the four original volume classes based on work by Brickell (1989) as confirmed by Julin and Caouette (1997). Instead, they used an approach providing differences among volume strata (low, medium, and high) that are statistically significant for each Administrative Area. The assumption regarding net growth equaling zero cannot be substantiated but seems to be reasonable.

Modeling implementation reduction factors (MIRFs)—Managers recognized limitations in their ability to accurately estimate timber output levels based on existing inventories and unforeseen circumstances encountered during project implementation. “Fall down” occurs when the number of acres actually harvested is less than the number of acres planned for harvest. Unmapped stream buffers, new eagle nests, and misclassified karst vulnerability lands, for example, result in fall down. To anticipate fall down, managers applied MIRFs to reduce the area of land suitable for timber production.

The methodologies used to define MIRFs are documented in appendix B of the FEIS (USDA Forest Service 1997b) and in the planning record (see footnote 4). These factors are based on professional judgment and some limited project experience. Discrepancies between the coefficients in the MIRF documentation and those in FORPLAN

[4] Planning record. On file with: Tongass Land Management Planning Office, 8465 Old Dairy Road, Juneau, AK 99801.

are noted in the planning record. These discrepancies are generally 1 to 2 percent, and planning team analysts predict that such differences will not noticeably alter model results. The application of these factors can markedly affect calculated levels of sustainable timber production. An estimated 731,000 acres are designated as suitable for timber harvest in the Forest plan.

Managers acknowledge the risk of overestimating the available timber resource by applying MIRFs. The accuracy of these estimates will be determined through monitoring and evaluation during project implementation.

Estimation of young-growth timber volume—Young-forest volume is estimated in the Forest plan from the SEAPROG model (Dixon and others 1992), which is the best model available for estimating tree growth in the Tongass. It was constructed from more than 11,000 tree records[5] from (1) the Tongass timber inventory (1982-84), (2) the PNW and Alaska Region stand density study, (3) Administrative Area stand data (stand exams, young-growth surveys, growing stock studies), (4) Makah Indian Reservation inventory data, and (5) Queen Charlotte Island forest inventory data. Tongass land management plan analysts assessed the relation of young-forest yield projections generated by using the SEAPROG model to yields from three normal yield tables for stand ages 30 to 150 years (Barnes 1962, Meyer 1937, Taylor 1934). A test for differences among regression coefficients (slope) revealed no significant differences between the yield table estimates and the SEAPROG projection. The data used to construct the SEAPROG model may not represent the range of stand conditions now subject to timber harvest—namely higher elevation and other poor quality sites (e.g., forested wetlands). An overestimation of the availability of the timber resource (where timber harvest levels may not be sustainable) could result from using a model derived generally from higher quality sites. With information lacking, sensitivity analyses could be beneficially used to examine this issue.

Noninterchangable components (NICs)—The managers partitioned the annual allowable sale quantity of 267 million board feet into two components to promote economic sustainability of the timber resource. This approach distinguishes portions of the timber supply at lower risk of attainment from those portions at higher risk of attainment owing to marginal economic conditions and possible changes in future demand. Volumes associated with each component are to be identified separately in annual harvest plans for the Forest and are not to be substituted (i.e., noninterchangeable) for volume from the other component to determine the allowable sale quantity.

The managers used logging operability maps to identify three logging feasibility categories: (1) normal operability—areas that are expected to be logged economically by using standard logging systems (e.g., tractor, standard cable, and short-distance helicopter); (2) difficult operability—areas that could be logged, but at significantly higher costs, by using long-span cable systems and medium-distance helicopter yarding; and (3) isolated operability—areas that would be extremely costly to log with long-distance helicopter yarding. These categories were developed in the late 1980s by logging engineers and timber planners at each Administrative Area and were based on existing and proposed roads, types of terrain, and professional judgment.

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[5] Data on file with: USDA Forest Service WO-TM Service Center, 3825 E. Mulberry St., Fort Collins, CO 80524.

The managers assigned low-volume stands of difficult operability and low- and medium-volume stands of isolated operability to a nonharvest prescription. The managers then created two noninterchangeable components based on operability: land of normal operability was designated NIC I (220 million board feet per year); all other land was designated as NIC II (47 million board feet per year).

Although this approach is intended to discourage harvest that is not economically sustainable, there is some uncertainty about the proportions of the allowable sale quantity that fall within each component. One analysis of the operability layer, that was considered by managers, indicated that the NIC I component may be overestimated by about 12 percent (USDA Forest Service 1997d). The operability layer was not modified on the basis of this analysis. Instead, managers acknowledged uncertainty of this data layer through their plans to "review the accuracy of the planning information used to allocate ASQ between the two NICs during the first year of Plan implementation" (USDA Forest Service 1997c). Using our criteria, the decision is thus consistent with the available information.

Forested wetlands—Managers decided to minimize timber harvest on certain forested wetlands (Kaikli, Kitkun, Karheen, and Maybeso Series) until more information related to the effects of timber harvesting in such locations is developed (USDA Forest Service 1997c).

Brock and Kissinger (1995) questioned the suitability of these forested wetlands for timber production, about 40,000 acres of which are designated as suitable for timber production in the Forest Plan. Brock and Kissinger's concern for harvesting timber on these soils is based on the following: (1) the few records available to categorize timber productivity following harvest—the four, 50-year site index records they cited ranged from 38 to 48 feet (Babik 1983, Farr 1984, Stephens and others 1968a); (2) Kissinger and others (1979) observed stunted, chlorotic forested wetlands on south Kupreanof Island; (3) the British Columbia Ministry of Forests recognizes forested wetland sites as "marginally merchantable" owing to drainage and aeration problems (Banner and others 1993); (4) Weetman and others (1989) report stunted forested wetland plantations in British Columbia; and (5) Moore (1982) reports that some forested wetlands in Europe have been converted to bogs following timber harvest, fire, and grazing.

Zaborske (1995) polled Alaska Region soil scientists and silviculturists about their concerns over harvesting timber from forested wetlands. Based on their work experience, most of these professionals (9 of 11) were not concerned about the effects of harvesting trees from these forested wetlands in terms of regeneration and growth response. The levels of experience with these particular forested wetlands (i.e., harvested acreage, field observations following harvest) are not documented.

Julin and Meade (1997) reviewed relevant background information—literature, plot records, and professional experience—concerning these forested wetlands. They found no information concerning damage resulting from timber harvest on these forested wetlands in southeast Alaska or coastal British Columbia; they found seven, 50-year site index records for these forested wetlands that project a range of heights from 38 to 65 feet (Babik 1983; Farr 1984; Stephens and others 1968a, 1968b). This overall lack of information can be attributed to our limited experience harvesting and studying growth and yield in low-volume stands.

It appears that the managers correctly interpreted the available scientific information on the forested wetlands in question. For example, when modeling growth and yield, they used 50-year site indexes ranging from 40 to 50 feet. These values are within the range known for forested wetlands in the Tongass.

The managers acknowledged the risk to forested wetlands by providing direction in the record of decision (USDA Forest Service, 1997c) to "...avoid harvesting on these four forested wetland soils." They also recognized that small inclusions (less than 2 acres) of these forested wetlands would be incidentally harvested owing to practical considerations. About 2,500 acres of these forested wetlands are scheduled for harvest in the next 60 years. They also commissioned a study to further clarify the status of young-growth stands on these soil types (Julin and McClellan 1996). Thus, using our criteria, the forested wetlands decision is consistent with the available information (table 1).

Windthrow—Managers applied standards and guidelines to reduce the risk of windthrow associated with timber harvest activities. This decision is based on the recognition that windthrow is a common form of disturbance in southeast Alaska (Harris 1989, Harris and Farr 1974) that can be exacerbated by timber harvest activities. Although catastrophic windstorms may be viewed as devastating to the timber resource, wind is considered an important ecological driver in that it creates tree canopy gaps that promote understory development (Alaback and Tappeiner 1991; Nowacki and Kramer, in prep.), and maintains soil productivity by uprooting trees that mix soil layers (Bormann and others 1995).

Harris (1989) describes physiographic circumstances where windthrow commonly occurs and provides guidelines for reducing windthrow damage. Nowacki and Kramer (in prep.) summarize several unpublished studies on wind disturbance at both small and large scales. They also present a model developed for Kuiu Island that predicts the probability of windthrow at a landscape level.

The managers seem to have correctly interpreted the available scientific literature and acknowledged risks by incorporating the following directions in to the Forest plan (USDA Forest Service 1997a):

- Riparian standards and guidelines: "Manage an appropriate distance beyond the no-harvest zone to provide a reasonable assurance of windfirmness of the Riparian Management Area."
- Timber standards and guidelines: "Where the chance of windthrow in adjacent stands is increased by timber harvest, measures should be taken to contain the windthrow within Land Use Designations where timber harvest is allowed."
- Modified landscape and timber production management prescriptions: "Seek to provide for a reasonable assurance of windfirm boundaries. To design for windfirmness, consider conditions such as soils, local wind patterns, tree height and size, and other site factors."

The managers also committed to using guidelines for reducing windthrow damage developed by Harris (1989) and other appropriate information to anticipate and mitigate windthrow damage described in the record of decision (USDA Forest Service 1997c). Timber harvest prescribed in the Forest plan will vary temporally and spatially over the landscape, and associated windthrow will be dispersed in time and space; it also will mimic (at a higher intensity) natural windthrow, which is considered an important disturbance agent in the Forest.

Summary of information related to timber issues—The use of information in the decisions considered here and whether risks were documented in development of the Forest plan are summarized in table 1. Overall, the timber-related decisions were consistent with the available information.

Social and Economic Effects

The Tongass National Forest, which comprises 85 percent of the land base of southeast Alaska, has profound effects on the quality of life for the nearly 75,000 residents. Management of the Tongass has provided or contributed to many types of employment in southeast Alaska. The Tongass historically has contributed nearly half of the timber harvested annually in southeast Alaska and, thus provided a substantial portion of the approximately 2,000 direct jobs in the industry. The industry, however, has experienced a decline of over 1,500 jobs since 1990. The Tongass provides habitat that supports the fishing and seafood processing industries, which together formed the region's largest private industry with an estimated 3,500 employees in 1994. The Forest also directly or indirectly contributes to the tourism and recreation industry, the fastest growing of the resource-dependent sectors, providing nearly 3,000 direct jobs in the region.

In addition to these employment opportunities and the related benefits that flow to local communities, the Forest provides habitat for many fish and wildlife species used for subsistence, a critical component of the lifestyles of Native and non-Native residents. Subsistence activities perpetuate cultural customs and traditions, supplement personal income, provide a major source of unprocessed food, and serve many other social, cultural, spiritual, and economic needs. In 1987, 85 percent of rural southeast Alaska households harvested subsistence food, and half of those reported harvesting more than 80 pounds of edible product per person. The Alaska Department of Fish and Game currently is updating subsistence-use information for southeast Alaska, which will result in a better understanding of current harvest levels.

The Tongass also provides opportunities for recreation for both residents and nonresidents. The unique wildlife-related opportunities and vast amount of scenic resources that are increasingly in demand are reflected in a steady increase in visits to the Tongass, which have more than doubled in the last 10 years.

The communities of southeast Alaska are affected in diverse ways by management of the Tongass. Many types of effects cannot be generalized from community to community, because the communities are unique in population size, access, economic structure and diversity, visions of the future, and local uses of the Tongass. Trends present at the regional or borough level manifest themselves in different ways in the communities, so data cannot be extrapolated to broader geographic areas or political boundaries. Analyses at multiple scales are necessary to understand social and economic conditions and trends in southeast Alaska.

The Tongass also has value to the entire Nation. As the largest National Forest in the United States and a key component of the largest remaining relatively unaltered coastal temperate rain forest on Earth, it has unique biodiversity values of concern to an increasing number of Americans. The tremendous increase in cruise ship visits also reflects the national and global interest in viewing and experiencing southeast Alaska's unique environment and culture. Alaska is typically used as an example of a place having high values apart from direct use, often referred to as "existence" or "preservation" values: people value just knowing it is there. These national values are

reflected by the number of comments about the Tongass Forest plan that came from outside Alaska—nearly two-thirds of the 21,000 responses received. Although the dollar value of nonmarket resources and opportunities in the Tongass was not estimated, there is little question that the amount is significant.

A substantial body of social and economic information is available to help decisionmakers identify effects of the Forest plan on residents of southeast Alaska and others who have an interest in management of the Tongass. An assessment of socioeconomic conditions and trends will be published soon (Allen and others, in press). Most of the data are regularly collected and compiled by the Alaska Department of Labor, the Alaska Department of Community Affairs, the Alaska Department of Fish and Game, and other agencies. These data include recreational opportunities and uses, subsistence use of wildlife and plants, structure and diversity of local, subregional, and regional economies, areas used by local residents, Federal revenue sharing payments, and socioeconomic panel ratings of the original RSDEIS alternatives (Swanston and others 1996).

Given the available information,[6] the following four questions can be used to gauge the consistency of decisions leading to the Forest plan:

1. Does the decision consider effects on the quality of life of southeast Alaska residents?

> The socioeconomic panel addressed potential effects on quality of life as one of nine indicators in its evaluation of the nine RSDEIS (USDA Forest Service 1996b) draft alternatives. Further analysis of panel ratings suggest that draft alternative 5 would provide overall the most beneficial effects on quality of life; panelists estimated that 15 communities would be beneficially affected, 4 negatively affected, and 9 unaffected (they disagreed on effects in 4 communities). Of lesser, but still positive benefit were draft alternatives 1, 4, and 6. Of these three alternatives, however, draft alternative 1 was predicted to have negative effects in twice as many communities as 4 and 6. Draft alternative 1 was anticipated to be either positive or negative for all but 4 communities, and draft alternative 4 was expected to have little effect in 16 communities. In contrast, draft alternative 7 was expected to have negative effects on quality of life in 24 communities, compared to positive effects in 3, and few effects either way in just 2 communities. The panel did not evaluate the emerging final alternative, which required the decisionmakers to extrapolate from the original judgments.
>
> Based on language contained in the ROD (USDA Forest Service 1997c), it seems that decisionmakers considered effects of the Forest plan on quality of life, including such components as opportunities for employment in Tongass-related industries, the availability of and access to resources used for subsistence, recreational opportunities, environmental quality, payments to local governments, and related aspects of quality of life that are directly affected by management of the Tongass.

[6] For purposes of our analysis, we assumed that socioeconomic information and analyses contained in the FEIS (USDA Forest Service 1997b) were incorporated into the decision.

2. Does the decision consider the diversity of southeast Alaska communities in terms of their economic diversity and structure, population size, subsistence use, community use areas, and other social and economic characteristics that could be affected by changes in management of the Tongass?

The FEIS (USDA Forest Service 1997b) contains detailed information on 32 southeast Alaska communities. Based on language in the ROD (USDA Forest Service 1997c), it seems that decisionmakers considered the differential effects of the Forest plan on southeast Alaska communities. Documentation could be improved, however, by including more of their reasoning in the ROD, along with discussion of differential risk carried by the final plan.

The decision did not appear to misrepresent or incorrectly interpret the scientific information available. At the broad scale, the apparent focus of the decision on timber industry employment seems to be consistent with the data available. The alternatives were projected to differ little in their short-term effects on employment in other resource-dependent industries, such as recreation and tourism, mining, and seafood processing, leaving timber industry employment as the key employment variable. Consistent with the science information available in the FEIS (USDA Forest Service 1997b), the ROD (USDA Forest Service 1997c) acknowledges uncertainty regarding long-term effects on employment in the fishing and seafood processing industries and the possible long-term risk to recreation and tourism employment. The ROD also explicitly acknowledges anticipated declines in employment in the timber industry, while still providing sustainable levels of annual timber harvest.

3. Does the decision consider effects on others who have a stake or concern in management of the Tongass?

Based on language in the ROD (USDA Forest Service 1997c), it seems that decisionmakers considered broader social and economic concerns extending beyond southeast Alaska communities, including the 65 percent of the 21,000 RSDEIS (USDA Forest Service 1996b) comments that came from nonresidents. Such concerns, apart from those discussed above, include ecosystem conditions and functioning, roadless areas, wild and scenic rivers, and the preservation of values providing benefits aside from actual use of Tongass resources. The discussion in the ROD incorporates both market and nonmarket benefits. The ROD also considers consistency of the Forest plan with national policy, such as the Forest Service long-term, national strategic plan, the Forest and Rangeland Renewable Resources Protection Act (RPA) (U.S. Public Laws, Statutes 1974), and the National Forest Management Act (U.S. Public Laws, Statutes 1976).

4. Does the decision contain a mechanism for addressing ongoing concerns about effects of Tongass management on the communities of southeast Alaska?

Whether impacts associated with risks actually occur depends on many variables—some within Forest Service control, such as where and how timber harvest and other activities occur, and some outside Forest Service control, such as what companies bid for contracts, where wood processing

occurs, and what happens to timber prices. Such uncertainties are one reason why it is difficult to predict the effects of a programmatic plan on individual communities.

In this situation, monitoring, mitigation, and an effective means for involving local residents in ongoing Forest Service activities becomes critical. Monitoring that includes tracking social and economic indicators at the community group scale would strengthen the plan. The spotted owl (*Strix occidentails*) decision in the Pacific Northwest, for example, lists a number of items to be monitored to assess the effects of the plan on rural economics and communities: demographics, employment in relevant sectors, government revenues, facilities and infrastructure, social service burden, Federal assistance programs, business trends, and taxes. Including quality of life and community resiliency information (how well communities are equipped to deal with change) as part of the monitoring plan would help to mitigate lack of quantitative prediction of socioeconomic impacts.

The ROD (USDA Forest Service 1997c) acknowledges the room for improvement in the plan's monitoring provisions, which could include emphasis on social and economic variables. Mitigation, which includes working with communities likely to be negatively affected (or simply with all communities), would help to identify ways of addressing impacts. The Forest plan contains an objective calling for such efforts, which presumably includes efforts such as the collaborative stewardship strategy currently being planned. Forest-wide standards and guidelines in the Forest plan call for consideration of local community needs in project plans, and for working with communities to jointly identify and develop natural resource opportunities. Finally, the "Information Needs" section of the Forest plan identifies social and economic research as a high priority, with the results inserted into the adaptive management feedback loop of the plan. This should help to ensure that additional scientific information on social and economic conditions will be collected and applied.

Table 1 summarizes how information on social and economic issues was used and how risks were documented in developing the Forest plan.

Monitoring

The Tongass land management plan identifies management direction for the Forest in terms of goals, objectives, and standards and guidelines. Within a logical planning framework, these elements are linked hierarchically and have specific roles. Goals describe desired conditions to be achieved some time in the future and are normally expressed in broad general terms. Objectives form the basis for further planning to define the precise steps to be taken and resources used in achieving identified goals (USDA Forest Service 1982). Objectives are concise, time-specific statements of measurable planned results that respond to goals. A standard is a limitation on management activities and a guideline is a description of a preferred or advisable course of action—both of which respond to objectives.

The Forest plan fits well within the planning framework outlined above. As a result, the monitoring plan will be effective as an adaptive management tool. Several of the key elements follow:

- Forest plan objectives are representative of Forest plan goals. For each of the 18 resource goals there is a corresponding objective (table 4).

Table 4—Status of objectives and monitoring plan items for each resource goal in the Tongass Forest Plan

Goals	Stated objectives	Measurable objectives	Monitoring plan items
Air	1	1	1
Biodiversity	2	2	4
Fish	2	2	3
Heritage resources	1	0	2
Karst and caves	2	2	2
Local and regional economies	2	2	2
Minerals and geology	1	0	1
Recreation and tourism	3	2	2
Research	1	1	1
Scenery	4	4	1
Soil and water	4	4	4
Subsistence	1	1	1
Timber	4	4	6
Transportation	3	3	1
Wetlands	2	2	2
Wild and scenic rivers	3	3	2
Wilderness	1	1	2
Wildlife	2	2	1
Total	39	36	38

- Most of the objectives are measurable. Of the 39 resource objectives identified, 36 are measurable (i.e., clear and quantifiable). Application of standards and guidelines was not considered to be a measurable objective.
- Monitoring items are representative of goals. All the resource goals have at least one monitoring item.

The Regional Forester recognized some limitations of the current monitoring plan and included language in the ROD (USDA Forest Service 1997c) to facilitate its improvement. The statement in the ROD is (p. 52):

> The monitoring plan does not specify the protocols to be used in gathering monitoring information. I would like to see these protocols developed, and monitoring accomplished, through a cooperative effort with our federal partners (i.e., the Environmental Protection Agency [EPA], Fish and Wildlife Service [FWS], and National Marine Fisheries Service [NMFS]) and the State of Alaska. Accordingly, I am directing the Forest Supervisors to convene an interagency group within 60 days to develop recommended monitoring protocols for the monitoring questions specified in chapter 6 of the Forest plan [USDA Forest Service 1997a]. I have requested the participation of the Pacific Northwest Research Station to assist in these efforts. I would also like private organizations, recognized tribes, and interested individuals to have an opportunity to participate in monitoring activities.

Table 1 summarizes how information on monitoring issues was used and how risks were documented in developing the Forest Plan.

Review of Overall Consistency

While we remain value-neutral on the Tongass Forest plan, our evalutions indicated that the plan achieved a high degree of overall consistency with the available scientific information. The major decisions related to management of riparian habitat, beach fringe habitat, and estuarine areas are consistent with the information base. Decisions on management of steep slopes and karst and cave resources also are consistent with available information. Decisions on development of an old-growth forest reserve strategy to provide habitat for well-distributed wildlife populations across the Tongass are consistent with available information. Treatment of timber resources and socioeconomic issues is consistent with available information, although decisions on these issues, especially socioeconomics, are subject to more uncertainty than the foregoing issues. And finally, the process now in place to monitor various issues, and accelerate research in high priority areas, helps assure that any remaining areas of uncertainty are subjected to ongoing reevaluation.

Metric Equivalents

When you know:	Multiply by:	To find:
feet	0.31	meters
miles	1.61	kilometers
acres	0.40	hectares
board feet per acre	0.017	cubic meters per hectare

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Appendix 1

This appendix is a comparison of the recommendations in AFHA (1995) with standards and guidelines, elements, and features in the Forest plan for protection of anadromous fish habitats in the Tongass National Forest.

Authors of the AFHA report (1995), in response to congressional directives, made numerous recommendations for management of aquatic and riparian habitats. These recommendations were a combination of scientific information and the professional experience and personal values of those making the recommendations. Correspondence from Regional Forester, Phil Janik, clearly states that he believed the AFHA recommendations, and their associated risk levels, were well founded and warranted careful consideration in the TLMP revision (Janik 1994). He suggested that AFHA (1995) should be the benchmark against which aquatic and riparian management decisions in the final TLMP alternative are compared and evaluated. We followed this suggestion and make that comparison here.

The AFHA is currently the definitive assessment for anadromous fish in the Tongass. The document was prepared in response to directives in the 1994 Congressional Appropriations Act Conference Committee Report (U.S. Public Laws, Statutes, 1994). The Alaska Region of the Forest Service was directed to:

- Proceed with stream analyses and studies and review procedures related to the PACFISH strategy in 1994 in order to study the effectiveness of current procedures (for protecting the habitat of anadromous salmonids).
- Determine if any additional protection (for anadromous fish habitat) is needed.

The AFHA report concluded that current procedures (those used in timber harvest activities after passage of the Tongass Timber Reform Act of 1990) were not fully effective in protecting anadromous fish habitat in the long term and made recommendations for improved protection.

Some of the AFHA recommendations could be accomplished by strengthening current direction for management of fish habitat and riparian zones on the Forest. The Regional Forester (Janik 1995) subsequently issued instructions to field units to accomplish the things necessary to comply with those specific recommendations from the AFHA report.

A second set of AFHA recommendations required new management direction, or changes in Forest policy, for implementation. Those recommendations were referred to TLMP by the Regional Forester for study and consideration in the current revision of the Tongass land management plan. The planning team addressed each of the recommendations, and the results were incorporated in the plan RSDEIS (USDA Forest Service 1996b), subsequently modified, and then included in the FEIS for the Forest Plan (USDA Forest Service 1997b).

The AFHA recommendations examined in the TLMP process were:

- To increase protection of headwater areas: steep slopes, high-hazard soils, and class III and IV streams.
- To modify streamside buffers on flood plains, alluvial fans, and confined alluvial channels.

- To clarify the current Tongass plan direction to "...preserve the biological productivity of all fish streams on the Tongass."
- To establish quantitative objectives for fish habitat capability.

Many AFHA recommendations on these issues were general. For example, recommendations in the fish habitat assessment team report from the expert field review (AFHA 1995:24) state:

> Stream Buffers: Buffer design should consider fish habitat, and water and sediment routing, rather than just the presence or absence of fish species. Buffers should also match the site topography, and those associated with flood plains managed to at least the borders of the active flood plain. Buffers should be designed to remain windfirm. Generally 100-foot buffers are not sufficient on larger, unconfined class I and II streams to meet these needs. Numerous suggestions were made on how to size the width of buffers to retain the buffer function, considering the amount of windthrow in southeast Alaska. Recommendations ranged from maintaining an additional 25% width of the needed "functional" buffer size, to incorporating entire flood plains plus 100 additional feet. Recommendations were also made that no harvest occur in areas of extreme windthrow hazard. Salvage of windthrow timber in buffers should not be allowed because the windthrown trees function to break-up the force of wind as it approaches the remaining standing timber and provide nurse logs for tree regeneration.

Other summary AFHA recommendations (AFHA 1995:38-43) were more specific:

> Additional protection of fish habitat requires two parallel efforts: the first is an ecosystem approach for evaluating and protecting watershed processes and functions at the landscape scale, and the second is the full implementation of Best Management Practices in planning and implementing activities that could affect aquatic ecosystems [p. 38].
>
> We recommend implementing watershed analysis using concepts presented in A Federal Agency Guide for Pilot Watershed Analysis (USDA Forest Service 1994) as a precursor to timber sales and other management activities that could significantly influence fish habitat [p. 39].
>
> We recommend as part of watershed analysis, that Riparian Management Areas be defined and the area within them managed to fully protect fish habitat in the long-term [p. 39].
>
> Because watershed analysis on all Tongass watersheds will not be implemented immediately, we propose the following interim recommendations:
>
> We recommend that riparian zones adjacent to unconfined alluvial flood plain channels, alluvial fan channels, and glacial outwash channels (Paustian 1992) should not be subject to timber harvest unless they are fully evaluated. The entire flood plain should be considered as the interim Riparian Management Area. In these channel types, riparian zones may extend beyond the minimum width specified under current procedures because the

stream is often dissected into a main low-flow channel with several side channels. These side channels are important fish habitat (Hartman and Brown 1987). Ecosystem-scale analysis should consider the whole riparian area, as defined by riparian soils and vegetation. Site specific harvest is only allowable when these riparian areas are fully evaluated and the purpose is consistent with the goal of full riparian protection [p. 39].

We recommend using a distance equivalent to the height of a site-potential tree to determine the Riparian Management Area width (assuming it is greater than 100 feet) for confined alluvial channel types of class I and II streams. In no case would Riparian Management Areas be less than 100 feet wide. Again, proposed harvest in these areas should fully protect riparian values. We had similar concerns about confined alluvial channel types that are class I and II streams. In these channels, debris recruitment may come from beyond a fixed distance, depending on bank slope, topography, and tree height of the dominant debris-producing trees [p. 39].

We recommend minimum 100-foot buffers on each side of class III streams until individual needs are identified during watershed analysis. Class III streams have important water quality values. These streams also are sources for woody debris recruitment and litter, and they deliver nutrient and sediment inputs into larger streams (FEMAT 1993:app. C.2; Gregory and others 1991; Hicks and others 1991). These streams are typically high-gradient streams often associated with steep, unstable terrain. Timber management opportunities within the buffers are evaluated case by case, considering mass movement hazards as debris, litter, nutrient, and sediment input [p. 39-43].

We recommend defining a new category, class IV streams for the intermittent or ephemeral colluvial channels and small, perennial spring-fed rill channels that are not dominant sediment transport channels. Streams that are currently unclassified and class III streams that are misclassified cause some confusion. They should be managed primarily to protect water quality. These streams are typically very small, high-gradient streams draining mountain slopes. They rarely need buffer strips, but often require special provisions for felling, yarding, and determining where to place landings and roads [p. 40].

We recommend that consistent Forest-wide definitions, inventory standards, and interpretations of mass-movement-hazard areas be developed, and that a full inventory and analysis of high-and very high-hazard soils be conducted [p. 40].

We recommend adopting the following additional management measures where steep slopes, high-hazard soil conditions, or both threaten fish habitat:

> High-hazard soils should not be clearcut or roaded before their mass-movement potential is assessed on site.
>
> No slopes greater than 84% should be clearcut.
>
> No coluvial hollows or highly dissected mountain slopes greater than 70% should be clearcut [p. 40].

The AFHA (1995:45) also states the following in regard to riparian management objectives:

> We recommend fish habitat objectives for large woody debris, width/depth ratio, and pool frequency based on currently undisturbed conditions. We also recommend further analysis and inventory to establish additional fish habitat objectives.

In regard to upslope stability:

> Areas of very high mass-movement hazard are removed from the timber base.

In regard to watershed analysis:

> Watershed analysis is recommended before timber harvest and other major land-disturbing activities. This recommendation is similar to the PACFISH (USDA Forest Service and USDI Bureau of Land Management 1994) strategy.

The foregoing recommendations indicate that the intent of AFHA (1995) is to increase protection of class I and II flood-plain systems, alluvial fan channels, and confined alluvial channels, provide buffer protection to class III streams, reduce the risk to fish habitat from mass erosion, and develop quantitative management objectives for aquatic systems. It is recommended that watershed analyses be conducted to ascertain the site-specific prescriptions needed to protect riparian and fish habitats before major human disturbances are conducted. The AFHA (1995:16) also states that without first conducting watershed and project analyses, the default buffer along class I and II flood-plain channels, alluvial fan channels, and glacial outwash channels is the entire flood plain; along class I and II confined alluvial channels, the buffer is the height of one site-potential tree; and along class III streams, the buffer is 100 feet of limited harvest.

These recommendations were forwarded to the TLMP team for further study, analysis, and modification as necessary to clarify and meet the intent of the AFHA (1995) report, and finally, incorporation into the TLMP revision process. In accordance with the AFHA recommendations, management of aquatic and riparian habitat in the Tongass has two goals: (1) the stated goal in the Tongass land management plan RSDEIS (USDA Forest Service 1996b) is to “maintain and restore the natural range and frequency of aquatic habitat conditions Forest-wide to sustain the diversity and production of fish and other freshwater organisms”; and, (2) it is the policy of the Alaska Region of the Forest Service to manage the Forest to avoid listing of fish stocks under the Endangered Species Act. These goals are clearly aimed at maintaining the biological productivity of the Forest’s fisheries and preventing any management that might lead eventually to the nonviability of fish stocks.

An interdisciplinary and interagency group of specialists took the AFHA (1995) recommendations and crafted a set of standards and guidelines (riparian option 2) to meet the intent of AFHA (USDA Forest Service 1996a:sec. 2.9). At the same time, two other sets of standards and guides were crafted. Riparian option 1, more restrictive than AFHA, is similar to the PACFISH (USDA Forest Service and USDI Bureau of Land Management 1994) strategy of the Pacific Northwest. Riparian option 3 is similar to procedures currently used in the Tongass to manage fish and riparian habitats. The standards and guides for these three riparian management options were differentially applied to the array of alternatives described in the Tongass RSDEIS (USDA Forest Service 1996b).

The riparian standards and guides in the Forest plan are not identical to riparian standards and guides in any of the draft alternatives described in the RSDEIS, although they contain elements from several of the draft alternatives. The Forest-wide riparian standards and guides for the final TLMP alternative are subdivided into general standards and guides, and those for each stream process group. In general, the standards and guides are written as direction for management of riparian habitat in the Tongass. The direction can be changed after watershed analysis, if the objectives established for each process group can be met. The guidelines provide well-defined defaults that will be used for riparian buffers and other management practices in the absence of watershed analysis. The riparian standards and guides developed for the Forest plan in the TLMP FEIS (USDA Forest Service 1997a) and key recommendations from the AFHA (1995) report are compared for consistency in table 5. With minor exceptions noted in the table, the riparian standards and guides of the Forest plan are consistent with the recommendations and intent of the AFHA (1995) report.

Table 5—Comparison of some key AFHA[a] recommendations with standards and guidelines in the Tongass Forest plan for management of riparian areas.

AFHA recommendation	Are final TLMP standards and guides consistent with AFHA?		
	Yes	No	Unknown
Use ecosystem approach for protecting watershed processes and functions	X		
Implement existing best management practices	X		
Implement watershed analysis	X[b]		
Define riparian management areas and manage to fully protect fish habitat in the long term	X		
No timber harvest in flood plains of glacial outwash, alluvial fan[c], and flood-plain process groups before watershed analysis	X		
Any timber harvest in flood plains of glacial outwash, alluvial fan, and flood-plain process (postwatershed analysis) must meet riparian protection goals	X		
Use distance equal to height of 1 site-potential tree to define riparian management areas along confined alluvial channels of class I and II streams	X		
Use minimum 100-foot buffers on each side of class III streams until individual needs are determined during watershed analysis	X		
Design buffers to remain windfirm	X		
Develop a new stream class (IV)	X		
Manage class IV streams to protect water quality	X		
Control timber harvest of steep and unstable slopes	X		
Develop consistent definitions, inventory standards, and interpretations of mass-movement hazard areas	X		
Develop objectives for fish habitat management	X		
No timber salvage in riparian areas unless approved by line officer in consultation with fisheries biologist or other resource experts	X		

[a] Anadromous Fish Habitat Assessment (1995).

[b] There are minor differences in use of watershed analysis but overall level of protection meets AFHA.

[c] Some unprogrammed timber harvest could be allowed in the alluvial fan process group before watershed analysis if objectives of the process group are met.

Appendix 2

This appendix is a comparison of the viable population committee reports and the final TLMP alternative.

Concerns about long-term viability of wildlife on the Tongass caused the leader of the Tongass land management planning team to establish an interagency viable population committee (V-POP) in 1990. The committee's report (Suring and others 1993) provided recommendations for an old-growth forest retention strategy designed to protect long-term wildlife viability in the Tongass. The committee also made recommendations designed to maintain the viability of individual wildlife species. The committee's report was subsequently subjected to peer review (referred to here as the "peer review") by a team of scientists commissioned by PNW (Kiester and Eckhardt 1994). An attempt was made to reconcile differing recommendations made in the original V-POP report and the peer review to provide a consistent set of recommendations for an old-growth-retention strategy to provide habitat sufficient to meet viability needs for all wildlife species in the Tongass. Preliminary recommendations for reconciliation were made in a report (Suring and others 1994) but were never subjected to scientific analysis or review. These documents articulate the need for a strategy to maintain habitat for old-growth-associated wildlife in the Tongass and make recommendations for one or more ways to achieve that goal.

The V-POP report and the peer review contain detailed recommendations on how to achieve wildlife viability on the Tongass National Forest. The recommendations, however, are not science but rather a combination of scientific information and interpretation, integrated with professional experience and personal values embodying a level of risk the authors thought appropriate. Recommendations of this type cannot be subjected to a science consistency check by using the criteria defined in this paper. Because of the interagency interest in the V-POP report and peer review, the recommendations are compared to the Forest plan in tables 6, 7, and 8. Remember, when reviewing the tables, that they do not represent a science consistency check but a comparison of one possible strategy for managing wildlife habitat in the Tongass with another.

Tables 6, 7, and 8 summarize, respectively, the degree of consistency between the Forest plan and the recommendations of the V-POP committee, the peer review, and the attempted reconciliation between the latter two.

Table 6—Comparison of V-POP[a] committee recommendations for wildlife viability and standards and guidelines for implementing the Tongass Forest plan

V-POP recommendations	Forest plan: Meets or exceeds V-POP	Forest plan: Addresses needs with another strategy[b]	Forest plan: Fails to address species needs
Old-growth habitat: Size, quality and distribution of large, medium, and small habitat conservation areas	X		
Matrix: Management of forested lands subject to timber harvest between reserves		X	
Great blue heron (*Ardea herodias*): Within 2 yr after nesting, no development within 1/8 mile of nest, or flights within 660 feet elevation (1/4 mile) of active nest, March 1-July 31.		X	
Goshawk: Inventory timber sales for active nests; estimate breeding pair home range and male core area; no disturbance in core, < 5% in home range		X	
Wolf and roads: Maintain open road density ≤1 mi/mi^2 including shorelines) in any 3 contiguous Alaska Department of Fish and Game wildlife analysis areas (WAAs) where wolves occur and roads are accessible to communities ≥1,000 people (≤1.25 mi/mi^2 in WAAs next to roadless or wilderness areas)		X	
Wolf and prey: Maintain habitat for ≥5 deer/mi^2 where deer are primary prey	X		
Brown bear: Locate human facilities ≥1 mile from seasonal brown bear concentrations; protect habitat and reduce human-bear conflicts; close roads in habitat conservation areas (HCAs) except during resource extraction; maintain 300-foot buffer along salmon streams that are important bear feeding areas; no roads within 300 feet of these streams	X		
River otter: 500-foot old-growth buffers along marine coastline and 1,000-foot buffers along estuaries	X		
Goats: Maintain 100% of potential winter habitat capability for populations of ≤50 goats as determined by models; protect kidding areas May 1-Aug. 1; keep development ≥1 mile from winter habitat and kidding areas; reduce human-goat conflicts via seasonal restrictions	X		

Table 6—Comparison of V-POP[a] committee recommendations for wildlife viability and standards and guidelines for implementing the Tongass Forest plan (continued)

V-POP recommendations	Forest plan: Meets or exceeds V-POP	Forest plan: Addresses needs with another strategy[b]	Forest plan: Fails to address species needs
Boreal owl (*Aegolius funereus*): Provide HCAs ≥5,000 acres ≤10 miles apart; protect nests outside HCAs with 0.5-mile buffer	X		
Flying squirrel: Maintain ≥1 patch of 1,000 acres of productive old growth (POG) per watershed; maintain corridors of POG between POG patches; corridor interruptions ≤65 feet; other details, see Suring and others (1993:255)	X		
Marten: Large, medium, and small HCAs recommended; consider connectivity among HCAs; minimize roads, HCAs, and corridors; employ effective road management techniques	X		

[a] Suring and others (1993).

[b] Level of risk likely differs from that embodied (but undefined) within V-POP recommendation.

Table 7—Comparison of peer review[a] recommendations for wildlife viability and standards and guidelines for implementing the Tongass Forest plan

Peer review recommendations	Forest plan: Meets or exceeds V-POP[b] peer	Forest plan: Addresses needs with another strategy[c]	Forest plan: Fails to address species needs
Vancouver Canada goose (*Branta canadensis*): Broaden the array of habitat features to evaluate habitat capability; include as a management indicator species	X		
Bald eagle: Standards and guidelines should be based on field information as well as habitat capability estimates	X		
Hairy woodpecker: Maintain specific micro-habitat features or mitigate for continued persistence across the forest, especially patches of recently dead trees; include as management indicator species	X		
Red-breasted sapsucker (*Sphyrapicus varis*): Develop habitat standards, especially for winter; include as management indicator species	X		
Brown creeper (*Certhia familiaris*): Use existing habitat capability index model to evaluate habitat; include as management indicator species	X		
Wolf: Needs more protected habitat than available in the Tongass National Forest, cooperate with other agencies and Canada to maintain habitat; provide effective road management	X		
Black bear (*Ursus americanus*): Require suitable matrix management with small clearcuts and alternative methods of harvesting; distribution and quality of habitat conservation areas (HCAs) reflect naturally fragmented nature of Tongass National Forest; emphasize HCAs and connectivity where needed most	X		
Brown bear: Preferred standards and guides meet V-POP and peer recommendations	X		
Marten: Need HCA strategy with different spacing than proposed by V-POP; matrix management better solution than in preferred alternative (P)	X		
River otter: No specific management recom-mendations beyond V-POP	X		
Deer: No additional specific guidelines related to management of this species	X		

Table 7—Comparison of peer review[a] recommendations for wildlife viability and standards and guidelines for implementing the Tongass Forest plan (continued)

Peer review recommendations	Forest plan: Meets or exceeds V-POP peer[b]	Forest plan: Addresses needs with another strategy[c]	Forest plan: Fails to address species needs
Mountain goat (*Oreamnos americanus*): Do not focus just on habitat for ≤50 animals; evaluate existing and additional fragmentation for this species		X	
Red squirrel (*Tamiasciurus hudsonicus*): Not included in V-POP species list; ensure availability of spruce cones and other habitat features at a spatial scale to maintain breeding populations and interactions among populations		X	
Great blue heron: Recommendations similar to those in V-POP report		X	
Goshawk: proposed under V-POP likely not adequate; matrix management likely critical to maintaining habitat capability		X[d]	
Boreal owl: Management of second growth next to HCAs critical in maintaining populations of this species		X	
Northern flying squirrel: No new management considerations; maintain corridors to facilitate movement		X	
Blue grouse (*Dendragapus obscurus*): No evidence that this species should be considered as a management indicator species	X		

[a] Keister and Eckhardt (1994).

[b] Suring and others (1993).

[c] Level of risk likely differs from that embodied (but undefined) within V-POP and peer review recommendations.

[d] See Iverson and others (1996).

Table 8—Comparison of reconciliation between V-POP[a] and peer review[b] recommendations for wildlife viability and standards and guidelines for implementing the Tongass Forest plan

V-POP and peer review reconciliation	Forest plan: Meets or exceeds V-POP reconciliation	Forest plan: Addresses needs with another strategy[c]	Forest plan: Fails to address species needs
Restrict logging and road building to areas other than volume classes 6 and 7 old-growth forest below 800 feet elevation		X	
Restrict logging, road building, and salvage sales to areas other than large and medium habitat conservation areas (HCAs)	X		
Restrict logging and road building to areas outside 3 largest old-growth forest patches within each ecological province		X	
Establish a 0.5- to 1-mile buffer around all large and medium HCAs as a special management zone in which road building and clearcutting are prohibited		X[d]	
Connect large and medium HCAs with corridors where logging is not allowed (1,600+feet wide between large HCAs; 1,000+feet wide between medium HCAs): keep corridors below 800 feet elevation. Place a 3,300-foot-wide "special management zone" along the coastline		X[d]	
Maintain old-growth forests that have been identified as important wildlife habitat through local knowledge or field experience (e.g., wildlife habitat retention areas mapped in the current plan[e])		X	

[a] Suring and others (1993).

[b] Keister and Eckhardt (1994).

[c] Level of risk likely differs from that embodied (but undefined) within V-POP and peer review recommendations.

[d] See Iverson and others 1996.

[e] USDA Forest Service 1979.

Appendix 3

Evaluation of Wildlife Viability in the Forest Plan Through Risk Assessment Panels

Scientists on the TLMP team used panels of subject matter experts, somewhat similar to the "Delphi method" (Linstone and Turoff 1975), to independently evaluate each of the original nine draft Forest plan alternatives for likely effects on wildlife populations (Shaw, in prep.; Swanston and others 1996). A second set of risk assessment panels was held early in 1997 to similarly evaluate the emerging final alternative (Shaw in prep.; USDA Forest Service 1997b:app. N). Members of each panel independently evaluated a selected species (or group) for the likelihood of obtaining specific outcomes regarding its status and distribution following implementation for 100 years of each draft Forest plan alternative. Each wildlife species (or group) was selected according to specific life history characteristics and habitat needs. Our intent was to select an array of old-growth associates whose individual life histories were dissimilar; yet, collectively their ecologies incorporated the breadth of forest habitat features and other environmental variation represented across the Tongass National Forest.

Six wildlife panels were conducted in the first set of panels: brown bear, marbled murrelet, Queen Charlotte goshawk, Alexander Archipelago wolf, American marten, and "other terrestrial mammals," which comprised a group of more wide ranging mammals and a group of endemic small mammals, some with known distributions in southeast Alaska that are restricted to a few islands or isolated portions of the mainland (MacDonald and Cook 1996). Except for marbled murrelet, the second set of panels considered the same species.

Each panel was comprised of subject matter experts who independently evaluated each draft Forest plan alternative (Shaw, in prep.; Swanston and others 1996). Each evaluator served on only one panel in each set, but most evaluators served on that species' panel in both sets. With one exception, none of the evaluators had previous involvement with the TLMP process. During the panel, each evaluator independently assigned 100 "likelihood" points across five outcomes for each alternative. Likelihood points assigned to each outcome were not probabilities in the empirical sense of frequencies; rather, they reflected extent of conviction, or uncertainty, according to available information and sound professional judgment in expected outcomes and were expressed through a probability-like scale. Individual ratings remained anonymous. A scribe recorded scores, computed a mean score for each alternative, and provided all results to each panel member for further discussion.

The "other terrestrial mammals" panel followed this general procedure with one notable exception: evaluators examined 14 species, most of them old-growth associates, but selected what they considered to be the most sensitive species or group of species to evaluate the effect of each alternative on the guild. For some alternatives, this group was comprised of a few or even one species. This, in part, reflected differences in geographic distribution of many island endemics and the variation in human-caused disturbance represented among alternatives. In many circumstances, however, several species were viewed as responding similarly to a selected alternative. Regardless, this guild was treated similarly to the single species in that 100 points were assigned for each alternative (Shaw, in prep.; Swanston and others 1996).

Outcomes reflected the expected population distribution and abundance of wildlife species relative to current condition of habitat in the Tongass, and to the projected condition of habitat following implementation of each draft alternative for 100 years (Shaw, in prep.; Swanston and others 1996).

The five outcomes were:

I Habitat is of sufficient quality, distribution, and abundance to allow the species to maintain well-distributed breeding populations across the Tongass National Forest. The concept of well distributed must be based on knowledge of the distributional range of the species and its life history.

II Habitat is of sufficient quality, distribution, and abundance to allow the species to maintain breeding populations across the Tongass National Forest. Some local populations, however, are more ephemeral because of reduced population levels and increased susceptibility to environmental extremes and stochastic events associated with reduced habitat abundance and distribution. Vacated habitats may become recolonized in the future.

III Habitat is of sufficient quality, distribution, and abundance to allow the species to maintain some breeding populations, but with significant gaps in the historic distribution on the Forest. These gaps likely are permanent and will result in some limitation of interaction among local populations. Significance of gaps must be judged relative to the distributional range of the species and its life history.

IV Habitat allows continued species existence only in refugia, with substantial limitations on interactions among local populations. Significance of extirpations across islands or regional landscapes must be evaluated relative to the distributional range of the species and its life history.

V Habitat conditions result in species extirpation from Federal land.

Limited discussions by the evaluators in the first set of panels (along with our appraisal of the notes from these sessions) related to outcomes meeting the condition of maintaining well-distributed populations across the planning area led us to derive two new outcomes from the original five: (1) populations likely will be well distributed—thus, mean likelihood scores from outcomes that likely will maintain well distributed populations (i.e., outcomes I and II in the previous list) for each species (or group) were summed to represent outcome I; and (2) populations likely will not be well distributed relative to their historic range—mean likelihood scores from the remaining outcomes (i.e., outcomes III, IV, and V) were summed to represent outcome II (Shaw, in prep.; Swanston and others 1996).

Each of the second set of panels had considerable, indepth discussion on this issue and on the differences, if any, between “viable” and “well-distributed” populations. Results of these discussions indicated that the panelists’ concepts of habitat conditions sufficient to support viable and well-distributed populations differed but, generally, fell within outcome III with well distributed being the more stringent criterion to meet.[1] Because of the indepth discussion of these topics in the second set of panels, we place more emphasis on their evaluation of where within the five habitat outcomes the concepts of habitat sufficient to support viable and well-distributed populations are described. As such, the boundaries of the two new outcomes changed such that only outcomes IV and V clearly represent (except for brown bear viability, which may be only outcome V; Shaw, in prep.) habitat conditions where populations likely will not be well distributed relative to their historic range.

[1] Evaluators indicated that well distributed equaled the sum of outcomes I and II for endemic mammals.

Mean likelihood scores among the five initial outcomes for the wildlife species common to both sets of panels appear in appendix N of the FEIS (USDA Forest Service 1997b). The means are presented with an understanding that there is a variance, often large, around these values (Swanston and others 1996). These tables also present the scores as summed for outcomes I and II combined, and outcomes I, II, and III combined to help evaluate the concept of habitat sufficient to support viable and well-distributed populations. Results of the first set of panels, including those for marbled murrelet, also appear in the wildlife section of chapter 3 of the FEIS (USDA Forest Service 1997b:351-430).

Evaluation of Wildlife Viability in the Forest Plan

Assessment—Before the second set of risk assessment panels occurred, the emerging final alternative had not been subjected to risk assessment panels. Because of the high interest in the likely risk to wildlife viability posed by the then-emerging final alternative, we reviewed the issue by examining substantive changes from draft alternative 3, the base from which the emerging final alternative was developed, and evaluated the direction (i.e., increase or decrease) of changes in risk to wildlife species. Particular emphasis on risk to Alexander Archipelago wolf and Queen Charlotte goshawk was evaluated because those species were being considered for listing under the Endangered Species Act.

To evaluate risk associated with the emerging final alternative, we examined the likelihood scores from the first set of panels for draft alternative 3, the draft alternative on which the final was based, and assessed effects of the departure of each design element from draft alternative 3.

The substantive departures from alternative 3 in the RSDEIS (USDA Forest Service 1996b) included:

1. Departures that decrease risk:

- A reduction in projected cut of old-growth acreage from about 571,000 to 474,000 acres over a 100-year timber rotation, with an associated reduction in building of roads.
- A mechanism for controlling road access when necessary to protect wolves and brown bears.
- An increase in spatially explicit reserves and the connectivity among reserves.
- More old-growth retention in timber LUDs.
- An increase in protected beach fringe from 500 to 1,000 feet.
- The removal of islands ≤1,000 acres from the timber base to increase protection of small endemic mammals.
- A survey and manage standard that requires biological evaluations for “sensitive” small mammals before timber harvest on islands ≤ 50,000 acres.
- An increase in protection of principal brown bear feeding areas along salmon streams.
- An increase in the number of wild and scenic rivers.
- Increased “firmness” in the language of standards and guidelines.

2. Departures that increase risk:

- A reduction in protection of riparian systems (i.e., from riparian options 1 and 2 applied to Forest Habitat Integrity Program[1] 1 and 2 watersheds, respectively, to a revised and improved option 2 Forest-wide).
- A reduction in the amount of timber harvested with prescriptions other then clear-cutting from matrix lands between reserves (i.e., from 100 percent to an unknown percentage in areas designated in green, yellow, and brown on the Forest plan map; fig. 1, inside back cover).
- A decrease in protection levels for specific deer winter ranges.

The risk to wildlife viability from implementing draft alternative 3 was represented by the sum of likelihood scores assigned to outcomes III, IV, and V combined in the first set of panels (Swanston and others 1996). Conceptually, this sum represented the view of the first set of panels that “not meeting well distributed” described populations across the Forest of each of the six species or groups evaluated. Expected effects on wildlife viability of implementing draft alternative 3 for 100 years differed among wildlife species but can be characterized through three discernible groups, based on the first set of risk assessment panel likelihood scores: the marbled murrelet and the Alexander Archipelago wolf, which each had >80-percent likelihood of maintaining well-distributed populations across the planning area; the brown bear, American marten, and northern goshawk, which had roughly a 50-50 distribution of mean likelihood scores; and endemic mammals, where at least one species had less than a 25-percent likelihood of maintaining well-distributed populations across the Tongass National Forest (Swanston and others 1996).

Each of the significant changes (made to draft alternative 3 to develop the emerging final alternative) independently affected risk to wildlife viability. How much each design element influences wildlife viability, or more importantly, the cumulative effect of these changes, is difficult to quantify.

Each projected departure in alternative elements differs in the manner and the extent of its influence on the risk to viability of individual species. Each projected departure also differs in its effect among different species. The change in riparian protection, for example, primarily reduces the number of headwater buffers for class III streams; narrow strips of riparian timber associated with this change are of limited value to old-growth-associated wildlife species. The change from two-age timber management in the matrix lands between reserves to primarily clearcutting may have some negative affect on overall biodiversity, but the isolated clumps of old growth left by two-age management likely would provide limited habitat features for some old-growth-associated species in the original alternative 3.

Although the TLMP scientists cannot ascribe numerical changes in risk (i.e., specific likelihood scores) to wildlife viability associated with each change, the direction of these changes in the emerging final alternative appeared to present an increase from draft alternative 3 in the likelihood of wildlife populations meeting criteria for being well-distributed. The reduction in total acres of old-growth forest scheduled to be cut in the next 100 years and the associated reduction in new roads, increased old-growth forest area dedicated to reserves, increased protection of brown bear feeding

[1] Forest Habitat Integrity Program (FHIP) defines highest quality watersheds for sport and commercial fish.

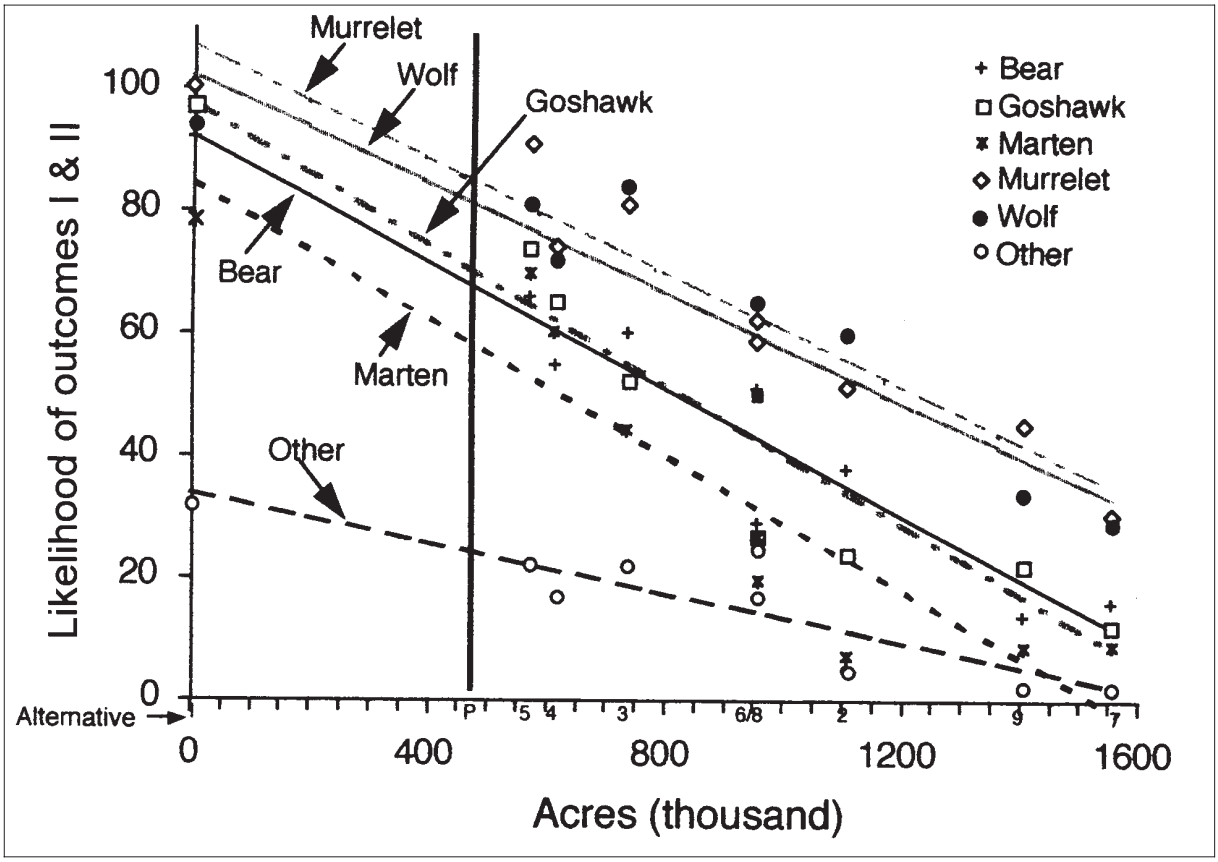

Figure 2—Analysis of mean scores from the first set of risk assessment panels indicated a negative correlation between acres of old-growth forest scheduled for harvest over the next 100 years in each draft alternative and the correspondingly assigned likelihood scores for maintaining habitat sufficient to support well-distributed populations of selected wildlife species across the Tongass. Partially from this relation, we estimated that the emerging final alternative, with 474,000 acres of productive old-growth scheduled for harvest over the next 100 years, would fall between draft alternatives 1 and 5 in its likeilhood of maintaining habitat sufficient to support well-distributed populations of various wildlife species.

areas, increased protection for small endemic mammals, and firmer standards and guidelines are important factors that decrease risk to wildlife viability. For example, reduction in total acres of productive old-growth scheduled to be harvested appeared to correspond with increases in assigned likelihood scores of maintaining well-distributed populations among the species reviewed by the first set of panels (fig. 2), probably because panelists used acres harvested as a surrogate for habitat loss during the evaluation process.

The lines in figure 2 are a regression plot of mean, first-set panel scores for each draft alternative. Without subjecting figure 2 to more analysis than is appropriate, and assuming that panelists would have applied the same criteria in evaluating the emerging final alternative as they did in evaluating draft alternatives, it is reasonable to assume that had panelists rated the emerging final alternative, their mean scores would have fallen between those assigned to draft alternatives 1 and 5. Following this logic, combined mean scores for outcomes I and II (i.e., meeting the concept of well-distributed as distilled from the first set of panels) likely would fall between 81 and 94

for the Alexander Archipelago wolf, and between 74 and 98 for the Queen Charlotte goshawk. Accordingly, we estimated that the emerging final alternative would have a lower risk to wildlife viability than all draft alternatives except draft alternative 1; i.e., risk would fall between alternatives 1 and 5.

The original nine draft alternatives were consistently discernible as three groups relative to their likelihood of "meeting well-distributed" criteria for each wildlife species or group (USDA Forest Service 1996b). Draft alternatives 1, 4, and 5 represented a group that generally had the greatest likelihood of meeting criteria for well distributed, whereas draft alternatives 2, 7, and 9 had the least likelihood of meeting these criteria. Draft alternative 3 was among a group, including alternatives 6 and 8, intermediate in their likelihood of meeting for criteria well distributed (USDA Forest Service 1996b). The emerging final alternative appears to fall within the first group that generally had the greatest likelihood of meeting well-distributed criteria.

Corroboration—Conducting the second set of risk assessment panels provided us with an unexpected opportunity to examine further our evaluation of the risk level associated with the emerging final alternative. Considering that the second set of panels further clarified their concepts of where within the outcomes habitat sufficient to support well-distributed populations is represented (generally within outcome III), our estimated range of scores within which the emerging final alternative would fall was similar to the results from the second set of risk assessment panels—the comparable panel scores for well-distributed populations would be between 83 and 97 for the Alexander Archipelago wolf and between 71 and 97 for the Queen Charlotte goshawk.

Our evaluation that the risk posed by the emerging final alternative to wildlife viability would be lower than all draft alternatives except alternative 1 was only partially corroborated by results from the second set of panels. This evaluation was consistent with panel scores for most species of fish, wolf, and endemic mammals—but not the other species paneled. Rather, the panel results more closely paralleled our third generalization; namely that the emerging final alternative fell within a group with original draft alternatives 1, 4, and 5—a group that had the greatest likelihood of meeting criteria for well distributed.

Additional measures were added to the Forest plan after the second set of panel evaluations. These measures were designed to further reduce the risk posed by the Forest plan to wildlife viability, most specifically for the northern (Queen Charlotte) goshawk and the American marten. Actions designed to improve connectivity among blocks of old-growth forest, where needed, also were included. These measures are discussed in the wildlife section in the body of this document, in the wildlife standards and guidelines for the Forest plan (USDA Forest Service 1997a), and in appendix N of the FEIS (USDA Forest Service 1997b).

Everest, Fred H.; Swanston, Douglas N.; Shaw, Charles G., III; Smith, Winston P.; Julin, Kent R.; Allen, Stewart D. 1997. Evaluation of the use of scientific information in developing the 1997 Forest plan for the Tongass National Forest. Gen. Tech. Rep. PNW-GTR-415. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 69 p. (Shaw, Charles G., III, tech. coord.; Conservation and resource assessments for the Tongass land management plan revision).

How scientific information was used in making management decisions relative to the Tongass land management plan was examined and evaluated on whether the decisions were consistent with the available information. The final conclusion was that the management decisions made in developing the final alternative achieved a relatively high degree of consistency with the available scientific information.

Keywords: Tongass National Forest, forest management, land management planning, science evaluation, science policy

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Federal Subsistence Management Program

Share

Wolf harvest season announced for GMU 2, new process explained

CRAIG, Alaska – Biologists with the Alaska Department of Fish and Game (ADF&G), in cooperation with the U.S. Forest Service (USFS), announce that the state and federal hunting and trapping seasons for wolf in Game Management Unit (GMU) 2 will close at 11:59 pm on Jan. 15, 2020.

10/30/2019

ADF&G, Tom Schumacher
Regional Supervisor, (907) 465-4359,
tom.schumacher@alaska.gov
U.S. Forest Service, Paul Robbins Jr.
Public Affairs Officer, (907) 228-6201,
paulrobbins@fs.fed.us

Management of wolf harvest on Prince of Wales and associated islands, collectively known as GMU 2, was based on a harvest quota and in-season harvest monitoring prior to 2019. When harvest approached the quota, ADF&G and the USFS would close the season by emergency order. This strategy resulted in unpredictable and often short trapping seasons. Trappers noted this strategy limited their flexibility to plan, and at times, has forced them to go out in unfavorable weather conditions to close their traplines in compliance with emergency orders.

ADF&G worked with the USFS, Fish and Game Advisory Committees, the Alaska Board of Game, the Federal Subsistence Regional Advisory Council, and trappers to develop a new strategy that would provide the flexibility and responsibility the trappers desired while sustainably managing harvest of this high-profile, wolf population.

The new strategy manages harvest of GMU 2 wolves by adjusting trapping season length based on ADF&G's most recent wolf population estimate and its relation to the established population objective. At their January 2019 meeting in Petersburg, the Alaska Board of Game altered harvest regulations to implement this strategy by establishing a GMU 2 wolf population objective of 150-200 wolves, endorsing ADF&G's harvest management plan, and aligning the opening dates for the state and federal trapping seasons (Nov. 15).

In August of this year, the Federal Subsistence Board (FSB) approved a temporary special action request by the Southeast Alaska Subsistence Regional Advisory Council to remove regulatory language referencing "a combined Federal-State harvest quota" for wolves in Unit 2 for the 2019-2020 regulatory year. The board also changed the wolf sealing period, the ADF&G monitoring process of placing a tag or seal on harvested animals, in Unit 2 to within 30 days of the end of the season. These FSB actions will promote coordinated management of wolves between ADF&G, and hunters and trappers using the new harvest strategy.

The new harvest management strategy is based on estimating the abundance of GMU 2 wolves. Because dense forest cover makes estimating wolf numbers from aerial surveys impractical, ADF&G, with support from the USFS, estimates wolf abundance in GMU 2 using a DNA-based mark-recapture technique. In fall 2018, ADF&G used the same large, northern and central Prince of Wales Island study area as in 2014-2017. ADF&G also collaborated with the Hydaburg Cooperative Association (HCA) and the Nature Conservancy (TNC) to establish an additional study area, monitored by HCA and TNC staff, adjacent to the southern and western boundary of the original study area. This collaboration expanded the study area to approximately 80% of Prince of Wales Island and over 60% of the land area of GMU 2.

Data collected from October through December 2018 resulted in a GMU 2-wide population estimate of 170 wolves, with high confidence that the actual number of wolves in GMU 2 prior to the autumn 2018 hunting and trapping seasons was within the range, 147 to 202 wolves. This is the most current population estimate. The autumn 2018 estimate was lower than estimates in fall 2016 (231 wolves) and fall 2017 (225 wolves), but well within the population

objective range of 150-200 wolves established by the Board of Game. Under the new harvest management plan, when the most current population estimate (170) is within the objective range (150-200) the trapping season may be up to two months long.

The new wolf harvest strategy, built on the cooperative spirit among the ADF&G, the Federal Subsistence Board, and GMU 2 hunters and trappers, offers the full two months of wolf trapping opportunity allowed under the management plan for the 2019-2020 season. State and federal trapping seasons will both open on Nov. 15, 2019, and close on Jan. 15, 2020. The federal wolf hunting season in GMU 2 opened on Sept. 1, 2019, but the state wolf hunting season will not open until Dec. 1, 2019. State and federal GMU 2 wolf hunting seasons will also close on Jan. 15, 2020. Hunters and trappers are reminded that the goal of the new GMU 2 wolf harvest management strategy is to maintain the fall wolf population within the range of 150-200 wolves.

Please call the ADF&G Ketchikan area office at 907-225-2475 for more information. For more information from the U.S. Forest Service, please call District Ranger Scot Shuler at 907-826-1600. Maps of Federal lands within GMU 2 are available at Forest Service offices. Maps and additional information on the Federal Subsistence Management Program can be found on the web at http://www.doi.gov/subsistence/index.cfm.

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-end-

PRESS RELEASE

Mountain goat harvest closed in Whale Bay Zone on Baranof Island in Unit 4

PRESS RELEASE

Federal Subsistence Board takes action on Temporary Wildlife Special Action Requests WSA19-07 (Mulchatna caribou) and WSA19-08 (Unit 13E ptarmigan)

PRESS RELEASE

Plans Announced for Winter Federal Subsistence Moose Hunt in a Portion of Unit 11

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RESEARCH FINAL PERFORMANCE REPORT

STATE: Alaska **STUDY:** SE-4-2-6

COOPERATOR: USDA Forest Service, US Fish & Wildlife Service

GRANT: SE-4

TITLE: Northern Goshawk monitoring, population ecology and diet on the Tongass National Forest

AUTHORS: Craig Flatten, Kimberly Titus, and Richard Lowell

PERIOD: 1 April 1991–30 September 2001

Editor's Note: Although the grant for this project began in 1995, in this final research report authors discuss nesting data as far back as 1991 during the program's onset when the USDA Forest Service and Alaska Department of Fish and Game funded this project. The title for Grant SE-4-2-6 includes additional focus on diet brought to this project through contract amendment.

SUMMARY

There is a need to evaluate the status, population, and habitat ecology of the northern goshawk (*Accipiter gentilis*) on the Tongass National Forest (Tongass). The northern goshawk was a key design species in the revision of the Tongass Land and Resource Management Plan (TLMP; US Forest Service 1997), and the US Fish and Wildlife Service reviewed this species for possible listing under the Endangered Species Act. TLMP has a number of requirements for monitoring goshawks on the Tongass and for maintaining goshawk habitat across the landscape. During 1991–1999, the Alaska Department of Fish and Game (ADF&G) and the US Forest Service (USFS) located and monitored goshawk nesting areas across the Tongass as part of an interagency study of the ecology and habitat relationships of this species in Southeast Alaska. Sixty-one nesting areas were documented within a study area approximately 77,000 km^2 (30,000 mi^2). Nesting areas were initially located during activities associated with timber sales (61%), recreation (10%), agency bird surveys (8%), or by tracking radiotagged adult goshawks (21%). Most nesting areas were located in remote areas and the majority (68%) used floatplanes for principal access; more than half (58%) required more than one type of motorized transportation for access. Nesting areas were searched annually 1 to >10 times during attempts to determine occupancy, nesting status, and productivity. Searches were aided by radiotelemetry or by standard goshawk detection methods, including broadcasting conspecific calls, watching for goshawks in flight above the forest canopy, listening for goshawk vocalizations, and accessing nest sites and surrounding areas by foot to search for clues of goshawks and nesting. We did not test the efficacy of these detection methods or apply them under strict protocol. Annual searches of nesting areas were conducted during a total of 283 nesting

area-years at 59 nesting areas. Across all years 1991–99, a mean of 83% (annual range = 71–96%) of known nesting areas were searched each year; a mean of 14% (annual range = 0–29%) of known nesting areas were searched each year by tracking radiotagged adults, and a mean of 70% (annual range = 53–80%) of known nesting areas were searched each year without the aid of radiotelemetry. Despite extensive efforts each breeding season to detect goshawks and nesting activity, our results from 9 years of nesting area monitoring indicate that nesting area searches done without the aid of radiotelemetry had limited success. This observation is supported by our general field experience and analysis of data from radiotagged goshawks concerning nesting area fidelity and annual movement between nests, both within and between nesting areas (see below). Various factors are believed to have affected the negative outcome of many nesting area searches done without radiotelemetry. These principally include 1) the dense rainforests and 2) frequently inclement weather of our study area, which hindered our ability to access nesting areas and detect goshawks, 3) the potentially large size of nesting areas (see below), and 4) the limited experience of some observers. Additionally, difficult and/or expensive logistics also limited access to many nesting areas and consequently the frequency and area extent of searches at these locations. We conclude that site conditions and logistical constraints in Southeast Alaska generally preclude efficient application of standard detection methods for searching goshawk nesting areas. Because searches of nesting areas among the dense rainforests and remote locations of Southeast Alaska are inconclusive when no goshawks or nests are detected using standard detection methods, and because we experienced low detection rates of goshawks and nests using these methods at known nesting areas, we further conclude that standard detection methods are not effective for reliably monitoring the annual or long-term status of most nesting areas in this region.

During 1992–99, 57 adult goshawks, including 26 females and 29 males, were radiotagged at 28 nesting areas and tracked year-round with airplanes; 88% of females and 69% of males were relocated in ≥ 1 subsequent breeding season. Distances moved between active nests in consecutive years by these birds ranged between 0.05 and 3.20 km within the same nesting area, and 3.67–152 km between nesting areas. We defined the maximum size of nesting areas in Southeast Alaska as 804 ha (1987 ac; 3.11 mi^2), based on the maximum distance of 3.2 km that a radiotagged pair moved between active nests in consecutive years within its home range. We observed that nest sites are generally 5–15 ha (12–37 ac) and that distances separating more than one nest site within the same nesting area can range from a few hundred meters to >3 km. Of active nests found with radiotelemetry within the same nesting areas in consecutive years, 54.2% (13 of 24) were located within a 0.359 km radius and 40.5 ha (100 ac) circular area of the year 1 nest; 79.2% (19 of 24) were located within a 1.0 km radius and 314.2 ha (776.4 ac) circular area of the year 1 nest. Based on these results and the inferred low rate of active nest detection we experienced when searching nesting areas without radiotelemetry, we recommend that the minimum size of "no commercial timber harvest" buffers around goshawks nests be increased beyond the 40.5 ha (100 ac) currently specified in the TLMP if a nest and nesting area based approach to goshawk management is to be used in the future. From our data and observations, we conclude that increasing the size of buffers around known nests will provide greater integrity to nesting areas by protecting more distant (0.359–3.2 km) alternate nests that have a low probability of detection without the aid or radiotelemetry.

Mean occupancy of nest sites based on the presence of an active nest detected with or without radiotelemetry was 28.4% ± 7.6 SE per year at Southeast Alaska nesting areas monitored ≥ 5 to 9 years but varied among management areas of the Tongass National Forest (Ketchikan = 13.0% ± 9.2 SE; Stikine = 20.0% ± 13.0 SE; Chatham = 53.2% ± 15.0 SE). Nesting area fidelity and mate fidelity were moderate for radiotagged females and high for radiotagged males. This difference between sexes can be explained in part by mate abandonment and movement to different nesting areas by some females, but not by males. Eleven females moved to a different nesting area in 35.7% of consecutive year events and remained at the same nesting area 64.3% of events. All adult males remained at the same nesting areas in consecutive year events. For radiotagged pairs, both members of a pair nested at the same nesting area in 55.2% of consecutive year events, and in 75.9% of events at least one member of the same pair was present at the same nesting area. Males retained the same mate in 81.0% of (male) consecutive year events; females retained the same mate in 54.8% of (female) consecutive year events. During 1991–99, a total of 223 fledglings were observed at 113 active nests in 55 nesting areas. Mean productivity across all years was 2.0 fledglings per active nest (annual range = 1.5–2.3). Mean rate of success (≥ 1 young fledged) of active nests across all years was 93% (annual range = 87%–100%). We captured and determined the sex of 49 fledglings that represented all young known to have fledged from 23 nests at 15 nesting areas. Male/female ratio of these birds was 1.04. A total of 81 fledglings, including 40 females and 41 males, were captured at 31 nesting areas during 1992–99. We banded all of these birds and 44, including 17 males and 27 females at 24 nesting areas, were also radiotagged with tail-mounted transmitters to study their dispersal movements and survival.

Our results are of interest to Tongass land managers who need to understand the implications of timber harvest on goshawk nesting areas. Our results indicate there is a high probability that not all active goshawk nests will be detected even when goshawk surveys are conducted before and during timber sale development. In addition, movements by goshawks to alternate nests in subsequent years confound survey difficulties. We conclude that a nest-based management approach to conserving goshawks would not be successful. A landscape approach, as adopted in TLMP, that includes both unknown nesting goshawks and sufficient foraging habitat is the cornerstone to a sound, long-term habitat management plan.

Key words: *Accipiter gentilis*, forest management, mate fidelity, nest area fidelity, nest productivity, nest success, northern goshawk, raptor, Queen Charlotte goshawk, Tongass National Forest, radiotelemetry.

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BACKGROUND

During 1991–99 the Alaska Department of Fish and Game (ADF&G) and the USDA Forest Service (USFS) conducted a study of northern goshawk (*Accipiter gentilis*) (hereafter "goshawk") ecology and habitat relationships on the Tongass National Forest in Southeast Alaska. This study was initiated from the need to focus applied ecological studies of the goshawk on the Tongass National Forest to meet requirements of The National Forest Management Act of 1976, which directs the USFS to manage wildlife habitats so that viable vertebrate populations are maintained in a well-distributed manner on National Forestlands. In the early 1990s, concerns about the effects of timber harvest on goshawk populations were first publicized in the southwestern U.S. (Crocker-Bedford 1990), and the U.S. Fish and Wildlife Service subsequently called for a nationwide status review of the species (Federal Register 1991 and 1992). Committees of biologists also

identified the goshawk as a species with population viability concerns in Southeast Alaska due to the high levels of timber harvest in this region (Crocker-Bedford 1992, Suring et al 1992). The goshawk was identified as a species of special interest and one in need of study on the Tongass National Forest. The Southeast Alaska population of the northern goshawk is listed as a species of special concern by the Alaska Department of Fish and Game. This species was chosen for study because of its affinity for forested landscapes, its association with larger and more mature forests in the Pacific Northwest (e.g., Reynolds et al. 1992, Squires and Reynolds 1997), its association with large habitat patches (Widen 1989), and the unique Queen Charlotte subspecies (*A.g. laingi*) that occurs in Southeast Alaska (Taverner 1940, American Ornithologists Union 1957, Palmer 1988, Webster 1988, Johnson 1989, Whaley and White 1994). Additionally, before the current study almost no information existed about the goshawk, its populations, or status in Southeast Alaska, and it was recognized that most knowledge of the species' relative abundance and habitat associations from other portions of its range is probably not applicable to the coastal rainforest environment of this region (ADF&G 1992).

Project goals and objectives were first identified in the 1992 ADF&G–USFS project study plan (ADF&G 1992). Objectives were broad in the beginning years of study when the base knowledge of goshawk natural history in Southeast Alaska was limited. For example, only 4 active nests at 10 documented historic and current nesting areas were known on the entire Tongass in 1992, the first full field season of effort. We accumulated some baseline data after several field seasons, allowing goals and objectives to be refocused and revised in updated study plans (ADF&G 1996).

OBJECTIVES

We focused on developing adequate samples of nesting areas and radiotagged goshawks and on determining the annual status and productivity of known nesting areas. We examined goshawk diet during the nesting season and assisted MS candidate Steve Lewis in his study of this objective. We also continued to collect and analyze morphometric data to assess the subspecific status of goshawks in Southeast Alaska.

Objective 1: Determine annual occupancy, nesting status, and productivity of known nesting areas.

Objective 2: Locate additional goshawk nesting areas to increase sample size.

Objective 3: Determine home ranges, habitat associations, and interyear movements of goshawks using radiotelemetry.

Objective 4: Evaluate goshawk diet during the nesting period.

Objective 5: Assess subspecific status of goshawks in Southeast Alaska.

STUDY AREA

Our study area was approximately 77,000 km^2 (about 160 x 480 km; 30,000 mi^2) and encompasses most of the Tongass National Forest in Southeast Alaska, including major portions of the Ketchikan, Stikine, and Chatham Management Areas and 8 of their 10 ranger districts. Goshawks occur in low densities in Southeast Alaska and are difficult to study in the dense temperate rainforests covering this vast region of island archipelagos and mountainous terrain (ADF&G 1993 and 1997, Schempf et al. 1995). Goshawks are also wide-ranging and secretive raptors and are essentially impossible to study away from the immediate vicinity of nest sites in Southeast Alaska without the aid of radiotelemetry (hereafter "telemetry"). In pursuing the goals and objectives of this project, we focused field efforts during 1991–1999 on locating active goshawk nests and developing and maintaining a sample of radiotagged adults over time. We derived nearly all study data from active nests, which allowed us to gather nest-based information on goshawk natural history and ecology and which served as loci where we could capture and radiotag adults and fledglings to investigate their movements, habitat use, and survival. New nesting areas were nearly always confirmed by the discovery of an active nest, and, once identified, these areas were added to the pool of known nesting areas monitored annually to assess nesting activity and productivity. At nesting areas monitored with telemetry, we were able to objectively assess annual occupancy, site and mate fidelity, and movement between alternate nests.

METHODS

Most nesting areas were identified from observations of goshawks and nests reported by USFS personnel conducting wildlife inventory, fisheries, stand exam, or engineering field activities related to timber sales. Other nesting areas were identified from observations by agency biologists conducting goshawk or songbird surveys, from observations by individuals engaged in recreational and other incidental activities, or from historic nest site records. We also tracked radiotagged adult goshawks to previously unknown nesting areas. We did not use random, systematic, or complete searches to identify new nesting areas (e.g., Reynolds and Joy 1998), nor did we attempt to test the efficacy of goshawk detection methods at known nest sites or nesting areas (e.g., Kennedy and Stalecker 1993, Watson et al. 1999).

Methods used to locate new nesting areas and to assess the status of known nesting areas were divided into 2 basic types: searches aided by telemetry and searches not aided by telemetry. Telemetry-aided searches were used whenever a radiotagged adult from a previous year could be tracked during a later breeding season. These efforts included aerial relocation followed by ground-based relocation of transmitters. Telemetry-aided searches located tagged adult females and males that moved between alternate nests within the same nest stand or nesting area, or that were present at the same nesting area but did not nest. These searches also located radiotagged adult females that moved to other distant areas where they did or did not nest. This is unlike many other goshawk studies that do not use telemetry and have much lower probability of locating nesting goshawks that may have

moved out of an intensively searched study area or moved a long distance to alternate nests within the same home range (e.g., Woodbridge and Detrich 1994, Reynolds et al. 1994).

Nesting area searches done without telemetry occurred when no adult goshawk wearing an active radio tag was present at a known or suspected nesting area. The method we used most frequently to assist detection of active nests was broadcasting recordings of conspecific vocalizations to elicit vocal or other responses from goshawks (e.g., Kennedy and Stalecker 1993). Other search methods included observing forested areas from vantage points to detect goshawks in flight above the canopy (e.g., Kostrzewa and Kostzewa 1990), listening for unsolicited goshawk vocalizations (e.g., Pentriani 1999), and accessing nesting areas and surrounding areas by foot (e.g., Fuller and Mosher 1987), vehicle, and boat to search for evidence of nesting. During searches done without telemetry at known nesting areas, all known nests were first inspected for activity. If these nests were determined inactive, variable effort was then applied to the surrounding area. Search effort without telemetry varied considerably between years, nesting areas, number of area visits, duration of visits, time in breeding season, number and experience of personnel, and extent of area searches. The number of visits made to a nesting area during a breeding season and the resulting degree of search effort was dictated largely by the remoteness of a location and logistical and cost considerations. We did not attempt to precisely document, summarize, or analyze effort or other method variables for nest searches done without telemetry. Access to many nesting areas we studied required travel by aircraft or boat and was therefore limited by unfavorable weather and cost. We summarized methods of transportation used to access field sites to examine how these logistics affected our ability to monitor nesting areas.

Annual Monitoring of Nesting Areas

ADF&G or USFS biologists and technicians made 1 to >10 visits to most documented nesting areas each year during the breeding season, March 1 to August 15, to determine occupancy, nesting status, and productivity and to collect site data. We summarized results of nesting area annual monitoring efforts for the period 1991–1999. Summary by year was done by tallying the number and proportion of nesting areas known from a previous year that were searched with and without telemetry and the outcome of these searches. Not included in this summary were nesting areas where no active nest had been located in any year, nesting areas where the status of both radiotagged adults from a previous year was “dead” or “unknown,” and nesting areas that were first documented in 1999. As noted, we did not attempt to directly compare search methods at nesting areas during monitoring efforts, nor did we attempt to first locate nests without telemetry when radiotagged adults were present at a nesting area.

Distances Moved Between Nests

We examined the distances moved between nests by identifying all occasions in which an active nest was located at the same nesting area in 2 consecutive years. We also identified all occasions that a radiotagged adult moved to a different nesting area and nested there in year 2 of consecutive breeding seasons. Distance between each pair of year 1 and year 2 active nests was measured using USGS topographic maps, aerial photos, or GIS map

software. Data was divided into 2 groups: second year nests located with telemetry and second year nests located without telemetry. We examined each data group separately and also made general comparisons between groups. For nests located with telemetry, the identities and home ranges of all individuals were known, and we were able to determine which second year nests were located within the same nesting area and home range and which were located in a different nesting area and home range. For nests located without telemetry, where no adult wore an active radio and many were unmarked and their identities unknown, we assumed that at least 1 member of the year 1 pair was present at the year 2 nest (i.e., the move to the year 2 nest was made by at least 1 adult from the year 1 nest). We also assumed that a second year nest located without telemetry was located within the same nesting area associated with the year 1 nest. See Results and Discussion for information supporting these assumptions.

Occupancy of Nesting Areas

During 1992–99 we monitored individual radiotagged adult goshawks year-round from <1 to 7 years. Radiotagging the adults allowed us to track movements of individuals and pairs and to determine their nesting status during 1 or more subsequent breeding seasons. To summarize adult nesting area occupancy rates, we identified all occasions in which a radiotagged individual was present in 1 or more breeding seasons at the same nesting area. Mean length of occupancy for adult females and males was then calculated for all nesting areas. We also summarized nesting area occupancy by any radiotagged adult. This nesting area-based method is different from bird-based occupancy because it also accounts for new birds replacing mates that died or left a nesting area.

Status of Adults and Pairs in Consecutive Breeding Seasons

The status of radiotagged adults and adult pairs at nesting areas during 1992–1999 was summarized using consecutive breeding seasons (see section *Distance Moved Between Nests* above). We identified all occasions in which an individual adult or an adult pair was known to be alive and nested in 1 breeding season, labeled year 1, and effort was made to determine its status during the following breeding season, labeled year 2. For individual adults, results were divided by sex and the following year into status types: 1) nested at same nesting area, 2) remained at same nesting area but did not nest, 3) moved to other nesting area and nested with a different mate, 4) moved to other nesting area and did not nest, 5) dead, and 6) unknown. Year 2 status types for pairs included 1) pair nested at same nesting area, 2) female moved to different nesting area; male remained and nested with new mate, 3) female moved to different nesting area; male remained and did not nest, 4) one mate died; remaining bird nested with new mate, 5) one mate died; remaining bird did not nest, and 6) status of 1 or both sexes unknown. We also used this information on the status of adults in consecutive years to summarize nesting area fidelity and mate fidelity.

RESULTS AND DISCUSSION

Objectives 1 and 2: Determine annual occupancy, nesting status, and productivity of known nesting areas. Locate additional goshawk nesting areas to increase sample size.

Annual Monitoring of Nesting Areas

We define *nest site* as the nest, nest tree, and forested area surrounding the nest that includes prey-handling areas, perches and roosts, and may contain ≥1 alternate nest. Nest sites in Southeast Alaska are approximately 5–15 ha. We define *nesting area* as the landscape area up to 804 ha (1987 ac; 3.11 mi^2) that includes all nest sites and alternate nests used by a goshawk pair or individual within its breeding home range. This definition is based on 8 years of our radiotelemetry data from adult goshawks in Southeast Alaska. We note that our telemetry-based definition of nesting area is analogous to the term *territory*, when used in other goshawk studies to describe the landscape area encompassing all known nests used by a pair (e.g., Woodbridge and Detrich 1994, Reynolds, et al. 1994). See Table 1 for definitions of these and other terms.

During 1991–1999, a total of 61 goshawk nesting areas were identified within an area approximately 77,000 km^2 (30,000 mi^2) in Southeast Alaska, including 4 nesting areas documented before 1991 and 57 nesting areas identified during project fieldwork in 1991–1999 (Table 2). A total of 56 nesting areas were located on Tongass National Forest land, including 14 on the Ketchikan, 26 on the Stikine, and 16 on the Chatham Management Areas; five other nesting areas were located on land of other ownership. The annual cumulative total of known nesting areas ranged from 7 in 1991 to 61 in 1999. Number of nesting areas known to have an active nest ranged from 3 in 1991 to 23 in 1999. Of the 61 nesting areas, 37 (61%) were identified through reports of goshawks and nests observed during activities associated with USFS or other timber sales. These included observations from wildlife inventory, fisheries, stand exam, or engineering crews. Six (10%) nesting areas were identified from reports from incidental observations or individuals engaged in recreational activities, and 5 (8%) were identified during agency bird surveys or goshawk project surveys. We identified 13 (21%) of the 61 nesting areas by tracking a radiotagged adult to a new nesting area. We monitored nesting areas annually by making 1 to >10 visits during the breeding season to assess occupancy and nesting status. Nesting area searches were aided by telemetry or by standard goshawk detection methods, including: broadcasting conspecific calls, watching for goshawks in flight above the canopy, listening for goshawk vocalizations, and accessing nest sites and surrounding areas on foot to search for visual clues of goshawks and nesting. We did not test the efficacy of these standard detection methods or apply them under strict protocol.

During 1991–1999, nesting areas were monitored during a total of 283 nesting area-years at 59 nesting areas. Two additional nesting areas, both located on private land, were visited by ADF&G or USFS personnel before 1991; one area contained an active nest site that was clearcut during logging activities. The other is near Skagway at the northern periphery of our study area. Nesting area monitoring included searches during 82 nesting area-years at 40 nesting areas where 1 or more radiotagged adult was present (including the year of radiotagging) and 201 nesting area-years at 51 nesting areas where no radiotagged adults

were present. Primary access to nesting areas from the closest ADF&G or USFS office was facilitated by various types of motorized transportation, and access to more than half (58%) of the nesting areas used or required more than 1 type of transportation (e.g., floatplane and road vehicle). For nesting areas monitored during 1991–1999, transportation types used on 1 or more occasions were floatplanes, 68% (40); helicopters, 24% (14); boats, 27% (16); and road vehicles, 51% (30). Of those nesting areas that we used only road vehicles to gain access, eight of twelve (75%) nesting areas were on the Juneau road system. In addition to 1 or more modes of transportation, access to all nesting areas and nest sites required traversing forested areas on foot from transportation departure points (e.g., beaches, muskegs, and roads). Distances traversed on foot to nesting areas ranged from 0.05 to >3km.

Summary by Year

The following results are from annual nesting area monitoring during 1991–1999.

Nesting area searches with and without telemetry. A mean of 83% (annual range = 70–96%) of all nesting areas known from a previous year were searched for nesting activity with or without telemetry on at least 1 occasion during the breeding season. An active nest was located at a mean of 25% (annual range = 19%–36%) of all nesting areas searched across all years. Additionally, a goshawk and other evidence of goshawk activity, such as recent prey remains or molted feathers, were observed and an active nest was not located, or a radiotagged goshawk was present and did not nest at 20% (annual range = 4–40%) of all nesting areas searched.

Nesting area searches with telemetry. A mean of 14% (annual range = 0–29%) of all nesting areas known from a previous year were searched with telemetry, and an active nest was located at 77% (annual range = 40–100%) of these nesting areas. Additionally, a radiotagged goshawk was present but did not nest at 23% (annual range = 0–50%) of these nesting areas. These latter observations involved 6 adult goshawks, including 1 female and 5 males that were radiotracked at 6 nesting areas. The movements of these individuals were focused on their respective nesting areas. These movements and ground checking of telemetry relocations indicated a high confidence that nesting did not occur. The 5 adult males (Blueberry 95–96 and Eagle Creek 93–94, Douglas Is.; East Bay of Pillars 95–96, Kuiu Is.; Lace River 94–95, Juneau mainland) were abandoned (after nesting) by their mates in 1 year and remained at the same nesting area the following year but did not nest. The one adult female (Pavlof River 95–97, Chichagof Island) remained in the vicinity of her nesting area for 2 successive years and did not nest either year. Her mate was not radiotagged. All these observations were made on the Chatham Area (Table 2).

Nesting area searches without telemetry. A mean of 70% (annual range = 53–80%) of all nesting areas known from a previous year were searched without telemetry and an active nest was located at 16% (annual range = 9–25%) of these nesting areas. A goshawk and/or other evidence of goshawk activity were observed, and an active nest was not located at 19% (annual range = 5–40%) of these nesting areas.

These results show that from 1991–1999 most annual monitoring of nesting areas was done without the aid of telemetry and that the mean proportion of nesting areas searched

with standard detection methods across all years was 70%, compared to 14% for searches with telemetry. Additionally, the mean proportion of nesting areas searched without telemetry and an active nest was located was 16%, compared to 77% for nesting areas searched with telemetry and an active nest was located. For nesting areas searched with telemetry and nesting areas searched without telemetry combined, the mean proportion across all years of all nesting areas searched was 83%, and an active nest was located at a mean proportion of 25% of these nesting areas.

We present this information as a record of extensive nesting area monitoring efforts that were conducted throughout Southeast Alaska over a 9-year period by ADF&G and USFS personnel. These results are presented by basic search types (with and without telemetry) to give a general accounting of these efforts. We caution against using these results to make direct and quantitative comparisons of the relative effectiveness of each search method for locating goshawk nests. For example, interpreting this information directly would incorrectly imply we observed a five-fold greater effectiveness of searches with telemetry (77% vs. 16%). Although a gross comparison of our field experiences with these search methods shows a relatively low success rate for nesting area searches done without telemetry (see below), we did not conduct a true test of the effectiveness of each search type; therefore, accurate comparisons of these search methods is not possible using this data set.

A primary focus of this project was to locate goshawk nests as the basis of ecological studies of this species. Our project goals and objectives did not include directly quantifying either goshawk detection rates or goshawk nesting densities. Though our data from annual monitoring of nesting areas provide a general summary of these efforts, more detailed interpretation of this data is confounded by limitations and variation in factors affecting the outcome of searches done without the aid of telemetry, such as variation in observer experience, number of nesting area visits, area extent of searches, and time in breeding season of searches. Additionally, we did not study nor attempt to account for fluctuation in natural factors, such as prey abundance and weather, which may have caused annual variation in nesting area occupancy and nesting status. Based on qualitative comparison of results from nesting area searches we conducted with and without telemetry, and analysis of data on nesting area fidelity and movement by radiotagged adults within and between nesting areas, we conclude that some significant proportion of goshawks and active nests were probably not detected during nesting area searches done without telemetry. Despite these shortcomings in our data from monitoring of nesting areas, we believe that these and other results from 9 years of study of goshawks and their nesting areas throughout Southeast Alaska nonetheless support our observations that goshawks are both uncommon in this region and nest here in densities lower than those reported for some other North American regions, such as Interior Alaska, Arizona, California, and Oregon (e.g., Squires and Reynolds 1997).

Logistical considerations were an important factor determining our ability to access and search nesting areas. Two-thirds of nesting areas we studied used floatplanes for access on 1 or more occasions, and many required more than 1 mode of motorized transportation. This is unlike most studies of goshawk nesting in other North American regions (e.g.,

southwestern US, California, Oregon, and Washington), where nesting area access is often entirely by vehicle and foot. Our extensive field experiences in Southeast Alaska support our belief that the temperate rainforests and remote island achipelagos of this region present one of the most challenging environments for the study of goshawks and that the dense forest structure and often inclement weather here make goshawk detection more difficult than in other regions having drier and more open forests. These considerations and our observations that goshawks appear to occur in lower densities in Southeast Alaska than in many other areas of North America indicate our data from nesting area monitoring is not directly comparable to similar data from other regions where goshawks occur in higher densities and more open forests and where nesting areas are accessible primarily by road and foot.

Objective 3: Determine home ranges, habitat associations, and interyear movements of goshawks using radiotelemetry.

Home Ranges

We evaluated the home range sizes of goshawks and found wide variation among individuals. These results were presented in detail in Iverson et al. 1996. Subsequent to 1996, we reanalyzed goshawk home range size using all the aerial radiotelemetry data from 1992–2000. This was done to: 1) increase the sample sizes presented in Iverson et al. (*op cit.*), 2) perform a more thorough data editing, and 3) provide additional home range estimates. These results are summarized in Tables 3 and 4.

Habitat Associations

Habitat associations were presented in Iverson et al. (1996), Alaska Department of Fish and Game (1994), and Pendleton et al. (1998).

Interyear movements

Nests Located with Telemetry

Twenty-four year 2 active nests were located with telemetry within the same nesting areas as their year 1 active nests by tracking radiotagged adults in consecutive breeding seasons. For radiotagged females and males combined, median distance moved between year 1 and year 2 nests at the same nesting area by adult goshawks was 0.23 km (inner quartile range = 0.17–0.68); mean distance was 0.73 km ± 1.00 SD (range = 0.50–3.20, n = 24 pairs of year 1 and year 2 nests used by 18 females and 11 males at 25 total nesting areas). The greatest distance that both members of a radiotagged pair moved in consecutive breeding seasons to an alternate nest within its documented home range and nesting area was 3.2 km (Margaret Lake, Revillagigedo Is. 1996–97; Table 2). We used a diameter of 3.2 km to describe the maximum area extent of nesting areas in Southeast Alaska (Table 1). A circular area having a diameter of 3.2 km is equal to 804 ha (1987 ac; 3.11 mi^2). This was also the greatest distance moved between nests in consecutive breeding seasons by a radiotagged adult male. We did not observe any radiotagged adult male move to and nest at another nesting area outside its documented breeding home range. In northern Arizona mean distance moved between alternate nests in the same territory was 0.489 km ± 0.541

SD (range = 0.021 km–3.41 km, *n* = 103 territories). Median distance moved between alternate nests was 0.285 km (Reynolds and Joy 1998).

We observed that all movement between active nests in consecutive breeding seasons greater than 3.2 km was made by radiotagged adult females that abandoned their year 1 mate and moved to a different nesting area in year 2 where they nested with a different mate. All movement by radiotagged females between alternate nests ≤ 3.2 km occurred within the same documented nesting area and home range. During 1992–99, 13 active nests were located by tracking an adult female to a different nesting area in year 2 of consecutive breeding seasons. Median distance moved by these birds from a respective year 1 nest was 18.50 km (inner quartile range = 7.80–36.10; range = 3.67–152 km, n = 13 nests; mean distance moved was not calculated due to the large variance of this data).

In northern California, distances moved to other territories (nesting areas) by banded adults in subsequent years averaged 9.8 km ± 2.7 SD (range = 5.5–12.9 km, *n* = 4) for females and 6.5 km ± 2.7 SD (range = 4.2–10.3 km, *n* = 3) for males (Detrich and Woodbridge 1994). In northern Arizona, distances moved to other territories by banded adults averaged 5.2 km ± 2.66 SD (range = 2.4–8.6 km, *n* = 5) for females and 2.8 km ± 1.06 SD (range = 2.0–3.5 km, *n* = 2) for males (Reynolds and Joy 1998). Unlike our study in Southeast Alaska, these studies observed that some adult males moved to and nested in areas considered different territories. As noted, nesting areas and home ranges in our study were defined with telemetry and we did not observe any adult male to move to and nest in a different nesting area outside of its documented home range. We are uncertain as to how the area extent of territories (home ranges) was defined in these other studies; however, comparing the maximum moved between nests by adult males in our study (3.2 km) with the range of distances moved by males to nests in other territories in California (4.2–10.3 km) and Arizona (2.0–3.5 km) shows that at least in California some males moved greater distances between nests than goshawks in Southeast Alaska. Similar to our study, banded adult females in these other studies were observed to move farther and more frequently than adult males. The observed range of distances and maximum distance moved by adult females between nesting areas, however, was considerably greater in Southeast Alaska than in these other regions (3.67–152 km vs. 5.5–12.9 km and 2.4–8.6 km, respectively). As with our comparisons of intra-nesting area movement between alternate nests, this difference may be explained largely by our use of telemetry to relocate adults both more consistently and at greater distances than is possible with birds marked with only bands.

The tendency for greater territory residency by males than females is widespread among bird species, including birds of prey (Greenwood 1980). Within *Accipiter* this pattern has been observed for goshawks in Southeast Alaska (this study), California (Detrich and Woodbridge 1994) and Arizona (Reynolds and Joy 1998), for Cooper's Hawks (*Accipiter cooperii*) in Wisconsin (Rosenfield and Bielefeldt 1996), and for sparrowhawks (*Accipiter nisus*) in Europe (Newton 1986). Mate abandonment and movement to other territories by females was observed in all of these studies; however, observations of this behavior in males are less consistent. Unlike Detrich and Woodbridge (*op cit.*) and Reynolds and Joy (*op cit.*), who studied goshawks 9 years and 6 years, respectively, we did not observe mate abandonment and movement to other territories or nesting areas by adult male goshawks.

Observations from our 8-year study in Southeast Alaska are consistent, however, with those for Cooper's hawks by Rosenfield and Bielefeldt (1996), who studied this species in Wisconsin over a 16-year period and observed that adult males remained at the same territories while some adult females moved to other territories. Newton (*op cit.*) studied sparrowhawks over a 14-year period and observed mate abandonment and movement to other territories by both males and females. He suggests that greater residency by males may be related to maintaining territories and observed that adult male sparrowhawks appear to be the prime defenders of nesting areas and home ranges where they procure food to support the female and young during the breeding season.

Nests Located without Telemetry

For the group of nests located without telemetry in year 2 of consecutive breeding seasons, all adults were 1) unmarked and their identities unknown, 2) banded and not wearing a radio tag, or 3) banded and wearing an inactive radio tag. Marked birds in this group could be positively identified only if captured. We assume that for nests located in year 2 without telemetry that at least 1 member of the year 1 pair was present at the year 2 nest and that the year 2 nest is located within the home range of the year 1 pair. That is, we assume that the year 1 and year 2 nests are within the same nesting area and occupied by at least 1 member of the same pair. These assumptions have been commonly used in other studies that examined nesting area occupancy and movement between alternate nests by unmarked goshawks (e.g., Reynolds and Wight 1978, Crocker-Bedford 1990). Additionally, in our study we believe these assumptions are supported in part by the observation that the maximum distance moved between nests by adults in this group (no telemetry) is within the maximum intra-nesting area distance moved between alternate nests by a radiotagged adult (1.6 and 3.2 km, respectively). Also, at some nesting areas where adults wore expired radio tags and/or were banded, we did confirm the presence of individuals at a nesting area in consecutive breeding seasons when they were recaptured. Though other data in our study indicate a relatively low success rate for nesting area searches done without telemetry (which weakens the validity of our data set representing the distances moved to year 2 nests located without telemetry), we present these results to allow general comparison of this data with data from the group of nests located with telemetry.

For the 19 active nests located without telemetry in year 2 of consecutive breeding seasons, median distance from a respective year 1 nest was 0.35 km (inner quartile range = 0.25–0.50 km); mean distance = 0.46 km ± 0.42 SD (range = 0 [same nest reused] –1.60 km). Of these 19 year 2 nests, 53% (10 of 19) were located within 0.359 km and a 40.5 ha (100 ac) circular area of their respective year 1 nests, and 84% (16 of 19) were located within 1.0 km and a 314.2 ha (776.4 ac) circular area of their respective year 1 nests. Comparison of year 2 nests located with and without telemetry shows that the maximum distance moved from respective year 1 nests at the same nesting area was 2 times as great (3.2 vs. 1.6 km, respectively) for radiotagged adults as for unmarked adults, banded only, or banded and wearing an inactive radio tag. Although year 2 nests located with telemetry within the same nesting area and >1.6 km from the previous year's nest represent only 17% (4 of 24) of this intra-nesting area group, these more distant moves to alternate nests emphasize both the mobility possible by goshawks within a nesting area and the potential

for more distant alternate nests to go undetected in searches without telemetry. Additionally, due to limitations in factors affecting the success of nest searches done without telemetry during 1991–99, such as experience of some observers, number of nesting area visits, area extent of searches, and time in breeding season of searches, we believe that some significant portion of active alternate nests were probably not located during year 2 searches using this method. We believe this is probably especially true for active alternate nests > 0.4 km from inactive known nests, which were most often the starting points for nest searches during annual monitoring efforts at nesting areas done without telemetry.

Occupancy of Nesting Areas

Radiotagged Adults

Telemetry allowed us to directly locate birds that moved to alternate nests within the same nesting area (0.05–3.2 km), birds that moved to other nesting areas (>3.2–152 km), and birds that were present at a nesting area but not nesting. We considered a nesting area occupied when a radiotagged adult was present on ≥ 1 occasion during the breeding season March 1–August 15, including the breeding season in which an individual was first radiotagged or first moved to a new nesting area. We counted 1 year of occupancy when a radiotagged adult was present during the breeding season in a year beginning March 1 and ending February 28 of the following calendar year. For each adult female and each male, we counted the total years of occupancy and then calculated mean occupancy for all nesting areas for each sex. This method of summarizing occupancy is similar to that used by Woodbridge and Detrich (1994), who counted the first breeding season of presence and each subsequent breeding season of presence at a nesting area as 1 "year" of territory occupancy. In our study of radiotagged goshawks, in most cases we monitored adults year-round and detected presence within a nesting area or home range at different times of the year. Though occupancy is expressed here in "years" of presence, because the measure used is actually presence at a nesting area during the breeding season, it may be more accurate to express these results as "breeding seasons" of presence.

During 1992–1999 mean nesting area occupancy by radiotagged adults was 1.6 years ± 1.0 SD (range = 1–6, n = 26 birds at 29 nesting areas) for females, and 1.9 years ± 1.4 SD (range = 1–7, *n* = 28 birds at 25 nesting areas) for males. These occupancy estimates include data from 11 females that made a total of 13 moves to other nesting areas (>3.2 km), where occupancy was counted anew. Mean nesting area occupancy by only adult females that remained at the same nesting area was 1.6 years ± 0.7 SD (range = 1–3, n = 26 birds at 19 nesting areas). We did not document any moves to other nest areas by adult males. In northern California, colored leg bands were used to identify adults at nest sites and assess annual territory (nesting area) occupancy. Territory occupancy averaged 1.8 years ± 1.3 SD (range = 1–7 years, *n* = 40 birds) for adult females, and 1.3 years ± 0.54 SD (range = 1–3 years, *n* = 27 birds) for adult males. Fifty-three percent of adult females and 60% of adult males were not relocated in years subsequent to the year of banding (Detrich and Woodbridge 1994). In northern Arizona, territory occupancy averaged 1.88 years for banded adult females (range = 1–6) and 1.42 years for banded adult males (range = 1–6) (Reynolds and Joy 1998).

Studies using banded birds tend to underestimate actual occupancy rates because of difficulties with locating birds that move to more distant alternate nests or that leave a nesting area and with locating birds present at a nesting area but not nesting. Banded adult males in these studies are also often more difficult to identify than their nest-attending mates because they are frequently foraging away from the nest. Males are also generally less aggressive and therefore less visible than females at the nest site.

Goshawk researchers have traditionally considered observation of 1 or more adults at a nesting area during the breeding season as minimum evidence of occupancy (e.g., Crocker-Bedford 1990). Although we also used this criterion in our study, we did not attempt to estimate nesting area occupancy using this information because we believe detection of goshawks in the rainforest environments of Southeast Alaska is sufficiently low when not aided by telemetry and, therefore, confidence in this estimate is precluded. We could not conclude with high confidence that a nesting area was unoccupied or inactive when no goshawk was detected. To further summarize our efforts to monitor nesting area activities over time, we examined the presence of any radiotagged adult at nesting areas in 1 or more breeding seasons. This method differs from the above analysis that considers occupancy by individual radiotagged adult females and males. At 28 nesting areas monitored during 1992–1999, we observed 33 exclusive occupancy events in which 1 or more radiotagged adults were present at the same nesting area during 1–7 sequential breeding seasons. Of these 33 events, presence by radiotagged adults at nesting areas was distributed as follows (1 = one breeding season, including season of initial radiotagging; 7 = seven sequential breeding seasons): 1 = 15 (45%), 2 = 10 (30%), 3 = 5 (15%), 4 = 1 (3%), 5 = 0 (0%), 6 = 1 (3%), and 7 = 1 (3%). We observed more than 1 single-year or multiple-year event, separated by 1 or more years of apparent inactivity, at only 2 nesting areas. Mean nesting area occupancy by any radiotagged adult during 1992–99 was 2.03 years ± 1.42 SD (range = 1–7 years, n = 56 nesting attempts and 6 occasions of presence only at 27 nesting areas).

Our results show that most (75% of events) nesting area occupancy by radiotagged adults occurred during the first 2 breeding seasons of sequential year use and that few nesting areas were occupied by radiotagged adults for more than 3 sequential breeding seasons. Goshawk studies in other North American regions have observed that most nest sites are occupied from 1 to 3 years and some much longer (Squires and Reynolds 1997). Various factors limited our ability to monitor many goshawks over multiple years. These included factors related to our study methods, such as transmitter failure or normal battery expiration (9–24+ mos., depending on type), loss of tail-mounted transmitters during molt, inability to recapture and retag some trap-shy birds, and inability to relocate some birds due to transmitter failure and/or movement outside of our study area. Other factors related to goshawk survival and behavior were mortality and movement by some adult females to other nesting areas. Consequently, presence of radiotagged adults at nesting areas decreased over time, and we were unable to assess occupancy and nesting activity with telemetry at most nesting areas for more than a few sequential breeding seasons. We estimated annual survival of radiotagged adult females and males (combined) to be 0.72, with a 95% CI of 0.56–0.88 (Iverson, et al. 1996). These results are comparable to those reported for northern Arizona where survival of banded adult females and males during 6 years of study was estimated at 0.866 and 0.688, respectively (Reynolds and Joy 1998).

Our data also show that adult females in Southeast Alaska experience higher survival rates than adult males. These results will be presented in future reports.

Adults without Radio Tags

We documented nesting by unmarked adults in multiple (>2) breeding seasons at the same nesting area in Southeast Alaska on only 1 occasion during 1991–1999. This occurred at the Duffield Peninsula, Baranof Island nesting area (see Table 2), which is also the only location where we observed reuse of the same nest in sequential years. At this nesting area, unmarked adults successfully used the same nest for 3 sequential years during 1994, 1995, and 1996. Goshawks typically alternate between 2 or more nests within the same nest stand or nesting area (e.g., Reynolds and Wight 1978). That we observed reuse of the same nest in only 2 of 51 (4%) occasions where an active nest was located at the same nesting area in consecutive breeding seasons emphasizes the importance of alternate nests in the nesting behavior of goshawks in Southeast Alaska (Table 2). The reason that goshawks alternate between nests within a nesting area is unknown; however, it is thought that nest-switching may reduce exposure to disease and parasites (Squires and Reynolds 1997).

We believe our inability to locate many nests of unmarked adults in years subsequent to documented nesting is due partly to the relatively low rate of success we experienced while conducting nest searches at known nesting areas without telemetry (see *Annual Monitoring of Nesting Areas*). Results of these searches were also affected by the relative preponderance of radiotagged adults at the known nesting areas we studied. This was due to emphasis on our objective to develop and maintain a sample of radiotagged adults, which sometimes reduced opportunities to search nesting areas in subsequent years without telemetry, especially during 1–3 years after initial detection of nesting. For example, during 1992–1999, ≥ 1 radiotagged adult was present during at least 1 breeding season at 63% (27 of 43) nesting areas that were first discovered without telemetry. Additionally, many nesting areas were occupied by radiotagged adults in more than 1 year because we retagged some individuals over periods of ≥ 2 years and newly tagged other individuals that replaced abandoned or dead mates. Consequently, in combination with the inferred low success for nest searches done without telemetry, one effect of our radiotagging efforts was that at many nesting areas the known presence of untagged, nesting adults was often limited to the breeding season of initial nest detection, followed by subsequent years of occupancy by 1 or more radiotagged adults. In some cases, however, we documented unmarked adults nesting at areas previously occupied by radiotagged birds. This occurred when a radiotagged adult female abandoned her mate and was replaced in the following year by a new untagged female that nested with the previous year's untagged adult male. In these instances we were successful at locating the untagged pair's active nest by simply checking a previously known nest or by broadcasting conspecific calls or listening for calls at dawn.

We were unable to quantitatively summarize nesting area occupancy and nesting status by unmarked adults over multiple years due to our inability to confidently make these determinations without telemetry. At some nesting areas we documented periods of apparent inactivity of 1 to 7 years that were both preceded and followed by discovery of an

active nest occupied by unmarked birds. At other nesting areas during these intermediate years, no activity was detected during some years, while during other years goshawks were detected and no nest was located (Table 2). Within the smaller area of a nest site in Southeast Alaska (typically no more than 15 ha, or 218.5 m radius; see *Definitions*), we believe the accuracy of our efforts to detect active nests without telemetry was high. For nesting areas monitored ≥ 5 to 9 years, we observed that mean occupancy (active nest present) of known nests sites was 28.4% ± 7.6 SE per year (n = 40 active nests located during 141 nest site-years at 23 nesting areas; nest sites searched with telemetry and without telemetry pooled). We note, however, that mean nest site occupancy varied considerably from south to north among the management areas of the Tongass National Forest: Ketchikan Area = 13.0% ± 9.2 SE (n = 7 active nests located during 54 nest site-years at 8 nesting areas); Stikine Area = 20.0% ± 13.0 SE (n = 8 active nests located during 40 nest site-years at 6 nesting areas); Chatham Area = 53.2% ± 15.0 SE (n = 25 nests located during 47 nest site-years at 9 nest areas). We cannot explain this variation, but note that in the Chatham Area, nest sites within 3 of 9 nesting areas, Blueberry Hill, Fish Creek, and Ready Bullion, Douglas Island, accounted for more than half of nest site-years in which an active nest was present and therefore inflated the mean occupancy estimate for this area. Additionally, one nesting area, Fish Creek, Douglas Island, was the location of the longest duration of sequential years at the same nesting area that we observed in Southeast Alaska (6 years at 2 nest sites). We also note that the Blueberry Hill, Fish Creek, and Ready Bullion, Douglas Island nesting areas are all located within an approximate 3500-ha area (8648 ac; = 13.5 mi^2) and represent the highest nesting area density we observed in Southeast Alaska during 1991–1999 (Table 2). We hypothesize that this relatively high nesting density may be related to higher prey abundance or prey availability in this region of Southeast Alaska (Lewis 2001). In northern California, nest stands in territories that were monitored ≥ 5 years were occupied an average of 46% ± 6 SE of the time (n = 71 nest stands; Woodbridge and Dietrich 1994). In Interior Alaska, annual nest site occupancy ranged from 6 to 56% (n =16 nest sites; McGowan 1975), and in Oregon mean occupancy of nest sites was 40% (n = 63 nest sites; Reynolds and Wight 1978).

Objective 4: Evaluate goshawk diet during the nesting period.

This objective was met and resulted in the M.S. thesis of Lewis (2001) and is summarized in the Appendix.

Objective 5: Assess subspecific status of goshawks in Southeast Alaska.

This objective was met and we conclude that the *Accipiter gentilis laingi* continues to warrant subspecific status. Abstracts of papers presented on the subspecific status of Southeast Alaska goshawks are presented in the Appendix.

CONCLUSIONS AND RECOMMENDATIONS

Our results from studying and monitoring nesting areas during 1991–1999 indicate that nesting area searches done without the aid of telemetry provided us with limited success in

accurately assessing goshawk occupancy and nesting status. This observation is supported by our field observations and quantitative data on nesting area fidelity and movement of radiotagged adults between nests and nesting areas. Nesting area searches we conducted without the aid of telemetry used standard methods for detecting goshawks: broadcasting conspecific calls, watching for goshawks in flight above the canopy, listening for goshawk vocalizations, and accessing nest sites and surrounding areas by foot. Although we did not test the efficacy of these methods or apply them under strict protocol, we conclude from 9 years of field experience that because nesting areas in Southeast Alaska occur in dense rainforest conditions, in large areas (up to 804 ha), and often in remote locations requiring expensive logistics, it is generally not possible to efficiently access and search these areas or detect goshawks and their nests here with consistency using these methods. An important objective of annual nesting area searches was to locate active nests where we could collect information on goshawk nesting ecology and where we could capture and radiotag goshawks. We were largely successful in meeting this objective, given the hindrances to effective goshawk detection and site access we encountered. However, because we were unable to confidently determine that nesting areas were unoccupied or inactive when no goshawks or active nests were detected during searches without the aid of telemetry, we conclude that it is not possible to accurately interpret results from these searches within the context of monitoring long-term trends in nesting area occupancy and nesting status.

Because our data indicate that annual nesting area searches that used standard goshawk detection methods were often ineffective, unreliable, and often expensive due to aircraft and other transportation requirements at many locations, we recommend that surveys based solely on standard goshawk detection methods be discontinued in future monitoring. If monitoring of goshawk nesting areas is to be included as part of future forest management plans, we suggest a more limited approach be taken that focuses on assessing only long-term status of known nest sites (5–15 ha), where confidence in the outcome of searches is high and more indicative of goshawk nesting area occupancy. Our data show that nesting areas were typically occupied by radiotagged adults for at least 2–3 sequential years (mean = 2.03 years ± 1.42 SD, range = 1–7 years), that a large proportion of adults present in the same nesting area in consecutive years nested, and that reoccupancy and nesting at a nest site can occur after a few to 7–8 years of inactivity. Based on this information, we believe that visiting nest sites every 2 or 3 years may be suitable for generally assessing the long-term status of nesting areas. Our field experience indicates that 1 visit per nest site by experienced observers during an optimal time in the breeding season (June–July) would be sufficient to accurately assess nesting status and productivity of known nest sites and to select adjacent areas. These 1-day visits would provide opportunities for attempting to detect goshawks and nests in the selected areas adjacent to known nest sites by broadcasting conspecific calls or making observations from vantage points. Any detections would be noted for future monitoring at these sites. Alternating nest site visits at different nesting areas in different years would allow reduction in annual logistics costs. Finally, we recommend that the size of “no commercial timber harvest” buffers around known nests be increased beyond the 40.5 ha (100 ac) minimum size currently specified in the Tongass National Forest management plan. Based on our data from radiotagged adults, we conclude that increasing the size of buffers around known nests will provide greater integrity to

nesting areas by protecting more distant (0.359–3.2 km) alternate nests that have a low probability of detection without the aid of radiotelemetry.

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PREPARED BY: **APPROVED BY:**

Craig Flatten
Wildlife Biologist I

Kimberly Titus
Regional Supervisor

Richard Lowell
Wildlife Biologist II

Wayne L Regelin, Director
Division of Wildlife Conservation

SUBMITTED BY:

Kimberly Titus
Regional Supervisor

Steven R Peterson, Senior Staff Biologist
Division of Wildlife Conservation

Table 1. Definitions of goshawk nesting, nest use and occupancy in Southeast Alaska.

Term	Definition/Comment
nest site	Nest, nest tree, and forested area surrounding the nest that includes prey handling areas, perches and roosts, and may contain ≥ 1 alternate nest. Nest sites in SE Alaska are approximately 5–15 ha.
nest stand	Nest site and the associated contiguous forested area where stand structure is relatively homogeneous. Nest stands in SE Alaska may vary in size from 5 to > 50 ha and include ≥ 1 goshawk nest site.
nesting area	Landscape area up to 804 ha (= 1987 ac; = 3.11 mi^2) that includes all nest sites and alternate nests used by a pair or individual within its breeding home range. Based on 8 years of radiotelemetry data (see *Interyear movements*). The nesting area includes individual alternate nests or clusters of alternate nests located within a few hundred meters of each other (e.g., Crocker-Bedford 1990, 1995), as well as other individual or clustered alternate nests at nest stands that may be separated by 1–3 km but are located within the normal movement patterns and breeding home range of a pair.
breeding home range	Landscape area encompassing all radiotelemetry relocations documented for an individual or pair during the breeding season, March 1 to August 15.
annual home range	Landscape area that includes all radiotelemetry relocations documented for an individual or pair during a year.
territory	Term not used in our analysis as it traditionally defines a use area based on intraspecific defensive behavior, which is largely unobservable for goshawks. In other N.A. goshawk studies (e.g., Woodbridge and Detrich 1994, Reynolds, et al. 1994) the term territory has been used to describe the landscape area encompassing all known nests used by a pair or individual. This use is synonymous with our definition of nesting area.
active nest	Any of the following: presence of defensive adult(s) at a nest, fresh greenery or other evidence of recent nest construction, eggs present in nest, young present in nest, pre-dispersal fledglings located in the vicinity of a nest that was determined active that year by the presence of fresh whitewash, goshawk feathers, prey remains, or pellets.
active nesting area	Any of the criteria for an active nest plus, when the physical nest could not be located – when only fledglings could be observed and other evidence such as prey remains or aggressive adults indicated that the active nest was nearby.
inactive	None of the active nest and active nesting area evidence could be found.
nesting area occupied	Any of the following: adult goshawk(s) present, recent prey remains, molted goshawk feathers located, or ≥ 1 breeding or nonbreeding radiotagged adult goshawk present in the nesting area during the breeding season.
nesting area unoccupied	Unable to determine with high degree of confidence between unoccupied or inactive in a given year. This is due to variability in the ability to detect goshawks in their breeding season home range.

Table 2. Annual status of goshawk nesting areas. Southeast Alaska, 1985–1999.

Nesting Area	80s	90	91	92	93	94	95	96	97	98	99
Ketchikan Area, Tongass N.F.											
Butterball Lake, Heceta Island					G	NAR	O	O	O	O	O
Carroll River, Revilla.Island								G	NA	G	G
Convenient Cove, Hassler Is.				G	G	NA	G	G	O	O	O
Derrumba Ridge, Heceta Is.									NO	O	O
Logjam Creek, P.O.W. Island					NAR	O	O	O	O	O	O
Margaret Lake, Revilla.Is.						NA	G	BR	BR	BR	O
McDonald Lake, Cleveland Pen.											NA
Port Refugio, Suemez Is.(89)	NA	O	O	G	G	B	G	G	O	O	O
Roberts/Cutthroat Crk, P.O.W. Is.							NAR	BR	BR	BR	O
Sarheen Creek, P.O.W. Island			GF	G	G	O	O	O	O	O	O
Sarkar Lake, P.O.W. Island				NAR	O	O	G	O	O	O	O
Timber Knob, Heceta Island								NA	B	C(96)R	BR
Traitors Creek, Revilla.Is.						NAR	BR	O	O	O	O
Twelvemile Arm, P.O.W. Island								NO	O	O	O

A = active nest first located.
B = active alternate nest located.
C = previously known active nest reused; ( ) = year first active.
F = fledgling(s) observed, active nest not located.
G = goshawk(s)/activity observed during breeding season, active nest not located.
N = nesting area documented this year.
O = no goshawk/activity observed, active nest not located.
R = adult(s) radiotagged and/or present.
T = radiotagged goshawk present but did not nest.
X = area not checked.

Table 2. Continued

Nesting Area	80s	90	91	92	93	94	95	96	97	98	99
Stikine Area, Tongass N.F.											
Big John Creek, Kupreanof Is.				NA	BR	O	O	O	O	O	O
Brown Cove, Petersburg Mainland										NA	BR
Camp Carl, Etolin Island									NAR	BR	BR
Cat Creek, Cape Fanshaw						NAR	O	O	X	O	O
Doughnut, Wrangell Island											NAR
Duncan Creek, Kupreanof Is.						NA	O	O	G	O	BR
East Bay of Pillars, Kuiu Island						NAR	BR	X	BR	O	O
Elena Bay, Kuiu Island										NAR	BR
Farragut Bay, Petersburg Mainland											NAR
Irish Lakes, Kupreanof Island								NA	G	O	O
Kadake Bay, Kuiu Island								NAR	O	O	O
Kake, Kupreanof Island (89)	NA	O	X area cut 89		X	X	X	X	X	X	X
Kuakan, Deer Island									NA	B	BR
Madan Bay, Wrangell Mainland										NGF	BR
Mitchell Creek, Kupreanof Is.						NAR	B	O	O	O	O
Mossman Inlet, Etolin Is.(86)	NA	X	X	O	O	X	X	X	X	X	O
Mountain Point, Kupreanof Is.						NAR	O	X	O	O	O
Negro Creek, Port Houghton						NA	G	O	O	O	O
Rowan Creek, Kuiu Island					NAR	RT	G	O	G	O	O
Sanborn Canal, Port Houghton						NA	O	O	X	G	O
Security Bay, Kuiu Island									NAR	O	O
Starfish, Etolin Island			NA	O	O	O	O	X	O	O	BR
Totem Camp, Kupreanof Island						NA	O	X	O	O	O
Tunehean Creek, Kupreanof Island										NA	BR
Upper Totem, Kupreanof Island					NO	O	O	X	G	O	O
West Bay of Pillars, Kuiu Island						NAR	BR	X	O	O	O
Zim Creek, Kupreanof Island											NA

A = active nest first located.
B = active alternate nest located.
C = previously known active nest reused; ( ) = year first active.
F = fledgling(s) observed, active nest not located.
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N = nesting area documented this year.
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R = adult(s) radiotagged and/or present.
T = radiotagged goshawk present but did not nest.
X = area not checked.

Table 2. Continued

Nesting Area	80s	90	91	92	93	94	95	96	97	98	99
Chatham Area, Tongass NF.											
Auke Bay, Juneau Mainland										NAR	O
Blueberry Hill, Douglas Island					NAR	BR	C(93)R	RT	C(94)R	BR	BR
Dewey Lk., Skagway (85)	NA	X	X	X	X	X	X	X	X	X	X
Distin Lake Trail, Admiralty Is.						NA	X	X	B	O	B
Duffield Peninsula, Baranof Is.						NA	C(94)	C(94)	O	X	B
Eagle Creek, Douglas Island					NAR	RT	O	O	O	O	O
Eagle River, Juneau Mainland									NGF	AR	BR
Fish Creek, Douglas Island						NAR	BR	BR	BR	C(96)R	BR
Florence Bay, Chichagof Island								NAR	O	O	O
Green Cove, Admiralty Island								NAR	BR	BR	C(96)R
Lace River, Berners Bay						NAR	RT	O	O	O	X
Mud Bay River, Chichagof Is.					NA	GF	O	X	O	C(93)	O
Nugget Crk., Juneau Mainland					NAR	BR	RT	O	C(93)R	C(94)R	O
Pavlof River, Chichagof Island							NAR	RT	RT	O	O
Point Bridget, Juneau Mainland				NA	BR	O	G	G	G	BR	B
Ready Bullion Crk., Douglas Is.			NA	BR	O	O	O	O	C(91)R	BR	C(92)R
Sitkoh River, Chichagof Island											NA
Tolch Rock, Juneau Mainland											NAR
Turner Lake, Juneau Mainland								NGF	A	G	GF
Whitestone, Chichagof Island							NGFR	AR	O	O	O

A = active nest first located.
B = active alternate nest located.
C = previously known active nest reused; ( ) = year first active.
F = fledgling(s) observed, active nest not located.
G = goshawk(s)/activity observed during breeding season, active nest not located.
N = nesting area documented this year.
O = no goshawk/activity observed, active nest not located.
R = adult(s) radiotagged and/or present.
T = radiotagged goshawk present but did not nest.
X = area not checked.

Table 3. Breeding (nesting) season, nonbreeding season, and year-round use areas for adult female northern goshawks, Southeast Alaska, 1992–1999. One hundred percent and ninety-five percent minimum convex polygons (MCP) from radiotelemetry locations.

Adult Females		Locations	100% MCP (ha)	95% MCP (ha)
Breeding season	Mean	31	4,549	4,153
	SD	18	2,465	2,423
(Mar. 1 to Aug. 15)	Median	27	4,304	4,223
	First quartile		2,648	2,108
n = 16 birds [a, b]	Third quartile		5,767	5,455
	Minimum	11	975	871
	Maximum	67	9,986	8,968
Nonbreeding season	Mean	26	33,839	32,961
	SD	14	42,134	42,950
(Aug. 16 to Feb. 29)	Median	24	14,718	12,602
	First quartile		5,630	4,144
n = 18 birds [a]	Third quartile		50,701	59,023
	Minimum	10	2,146	2,146
	Maximum	62	147,113	146,926
Year-round	Mean	43	54,218	49,465
	SD	29	61,756	60,360
(all months)	Median	35	16,619	11,688
	First quartile		9,852	9,048
n = 27 birds [a]	Third quartile		93,886	90,209
	Minimum	10	3,995	3,035
	Maximum	107	180,036	180,036

[a] Includes birds with ≥ 10 locations.

[b] Does not include 1 adult female that moved >44 km from her nesting area on August 3 and returned on August 7, resulting in a 100% MCP breeding season use area of 29,600 ha.

Table 4. Breeding (nesting) season, nonbreeding season, and year-round use areas for adult male goshawks, Southeast Alaska 1992–2000. One hundred percent and ninety-five percent minimum convex polygons (MCP) from radiotelemetry locations.

Adult Males		Locations	100% MCP (ha)	95% MCP (ha)
Breeding season	Mean	27	5,910	4,785
	SD	17	4,776	3,332
(Mar. 1 to Aug. 15)	Median	24	4,258	3,924
	First quartile		3,257	2,886
n = 21 birds [a]	Third quartile		6,579	5,518
	Minimum	10	1,229	1,229
	Maximum	73	19,469	15,361
Nonbreeding season	Mean	27	19,454	16,503
	SD	11	16,464	15,601
(Aug. 16 to Feb. 29)	Median	25	13,358	13,024
	First quartile		7,706	5,946
n = 14 birds [a, b]	Third quartile		24,257	19,684
	Minimum	17	5,996	3,702
	Maximum	57	63,738	63,513
Year-round	Mean	44	15,871	12,508
	SD	28	15,665	14,150
(all months)	Median	43	11,243	6,279
	First quartile		6,320	4,530
n = 22 birds [a, b]	Third quartile		20,261	14,441
	Minimum	13	1,949	1,949
	Maximum	117	67,444	63,908

[a] Includes birds with ≥ 10 locations.

[b] Does not include 1 adult male that dispersed >80 km from its nesting area during the nonbreeding season and whose nonbreeding season and year-round 100% MCPs use areas were 231,509 ha.

Appendix. Selected abstracts of papers presented at professional meetings and of thesis by Stephen B. Lewis on northern goshawk diet in Southeast Alaska.

Abstract of paper presented at annual meeting of the Raptor Research Foundation, Ogden, Utah, 1998.

Northern Goshawks (*Accipiter gentilis*) and Forest Management on the Tongass National Forest – Alaska

K. Titus, Alaska Department of Fish and Game, Division of Wildlife Conservation, Douglas, AK 99824 USA, G.C. Iverson, USDA Forest Service – Alaska Region, Juneau, AK 99802 USA, R.E. Lowell, ADF&G, Douglas, AK USA and C.J. Flatten, ADF&G, Ketchikan, AK USA

The Tongass National Forest (16.9 million acres; 68,000 km^2) contains some of the largest remaining tracts of old-growth temperate coastal rainforest in the world. ADF&G and the US Forest Service began cooperative studies of the Northern Goshawk in the early 1990's. By 1992 interim goshawk habitat management guidelines were issued for the Tongass National Forest and there was an attempt to develop a conservation strategy for maintaining habitats so that old-growth associated wildlife remained viable and well distributed across the Tongass. The 1979 forest plan and interim management guidelines for goshawks were identified as being inadequate to conserve goshawks across the Tongass. In 1994 the Fish and Wildlife Service was petitioned list the Queen Charlotte Goshawk as endangered under the Endangered Species Act. An interagency goshawk conservation assessment was prepared in 1996 to provide the Forest Service with the best available science-based information for decision-making. Study results indicated that goshawks had large use areas (approximate home ranges; 69 km^2 adult ♂ nesting season, n = 16) and that goshawks were selecting for old growth forest. Interpretation of results suggested that the probability of persistence of goshawks has declined over the past 50 years based on past and present forest management practices. The authors felt that a reserve system was an important but incomplete component of a long-term management strategy to maintain goshawks across the forest. Risk assessment panels were held during the forest plan revision in 1996 and 1997 to evaluate the likelihood that goshawk populations would remain viable and well distributed across the forest under alternative management scenarios. Goshawk panel experts suggested that the reserve system should be combined with other approaches including extended timber rotations, management of the intervening forest matrix where timber harvest would occur, and extended riparian and beach habitat protection buffers. The final Tongass forest plan contains elements resulting from the science-based goshawk information base.

Abstract of paper presented at annual meeting of the Raptor Research Foundation, LaPaz, Mexico, 1999

Monitoring, Territory Reoccupancy, and Interyear Movements of Adult Northern Goshawks (*Accipiter gentilis*) on the Tongass National Forest, Alaska: lessons from a long-term Radiotelemetry Study. KIMBERLY TITUS, Richard E. Lowell**,** Alaska Department of Fish and Game, Box 240020, Douglas, Alaska 99824 USA, and Craig J. Flatten, Alaska Department of Fish and Game, 2030 Sea Level Drive, Ketchikan, Alaska 99901 USA.

The Tongass National Forest contains some of the largest remaining tracts of old-growth temperate rainforest in the world. Management of these forestlands includes efforts to conserve and maintain habitats for a variety of wildlife, including goshawks. ADF&G and the US Forest Service began

cooperative studies in 1992 to understand the ecology of goshawks in an old-growth temperate forest ecosystem. As part of these efforts we have been monitoring goshawk nest sites and nest stands with the aid of radiotelemetry. Since 1992 we have captured 135 goshawks, and a total of 51 adults have been fitted with radio tags. We use radiotelemetry to track the interyear movements of adult goshawks. Between 1992 and 1998, 9 adult female goshawks moved to different nesting territories a total of 11 times and nested with different mates. These females move a mean of 34 km (range = 3.2–152 km). Of 26 adult male goshawks radiotagged, none have moved to a new nesting territory. Results from our radiotelemetry data suggest that annual monitoring of nest stands and checking old nest sites for occupancy by goshawks can provide misleading information. Depending on how the monitoring is actually designed, one could conclude that a raptor nesting population is declining simply because of interyear movements by nesting adults to sites that are unknown. This is especially true for studies in dense forests where large, complete censuses of all nesting pairs are impossible. Our radiotelemetry results also indicate that some home ranges are occupied by nonnesting goshawks, and that some pairs move 2–3 km to different nests between years, while maintaining the same home range as previous years. Hence it would be improper to suggest that these territories are "unoccupied," but this would be an often used interpretation in many raptor-monitoring projects.

Abstract of paper presented at annual meeting of American Ornithologists Union meeting – Seattle, 2001.

Color and Size of the Northern Goshawk in Southeast Alaska. CRAIG FLATTEN*, Alaska Dept. of Fish and Game, Ketchikan, AK; KIM TITUS, Alaska Dept. of Fish and Game, Douglas, AK; RICHARD LOWELL, Alaska Dept. of Fish and Game, Petersburg, AK.

The taxonomy of Northern Goshawk (*Accipiter gentilis*) subspecies in N.A. is currently a topic of interest and debate. Resource managers are required to maintain well-distributed, viable goshawk populations and to protect distinct population segments that may be threatened or endangered. The AOU recognizes two Northern Goshawk subspecies in N.A.: *A.g. atricapillus and A.g. laingi.* Some question the validity of Northern Goshawk subspecies that are based primarily on subtle color and size distinctions. The *laingi* subspecies has been described as a smaller and darker race inhabiting the coastal temperate rainforests of British Columbia and Southeast Alaska. Information on plumage coloration and body size was collected from 68 adult and 70 juvenile goshawks captured at nest sites in Southeast Alaska between 1992 and 2000. Phenotypes ranged from dark forms identified as *laingi* to lighter forms identified as *atricapillus*. Mean wing chords were smaller than those reported for Northern Goshawks from other regions of Alaska, but larger than those reported for *laingi* specimens from coastal British Columbia. Slight clinal variation in size within Southeast Alaska was detected in some age-sex classes with smaller birds occurring in the south. Results generally support the original description of *laingi*, noting the occurrence of some clinal variation and probable intergradation of subspecies within Southeast Alaska.

Abstract of paper presented at annual meeting of American Ornithologists Union meeting – Seattle, 2001.

Breeding dispersal of adult Northern Goshawks in Southeast Alaska: implications for conservation.

KIMBERLY TITUS*, Alaska Dept. Fish and Game, Juneau, AK, CRAIG FLATTEN, ADF&G, Ketchikan, AK; and RICHARD LOWELL, Petersburg, AK.

Northern Goshawks nest in the old-growth temperate rainforests of Southeast Alaska and are a conservation concern for forest management activities. We evaluated breeding dispersal (movement from one nest to another in consecutive years), nesting status, and fate of adult goshawks during 1992–99 by tracking them with radiotelemetry. We defined a nest area as a 3.2 km diameter area because this was the maximum distance a pair moved in consecutive years while maintaining the same home range. Multiyear movements were determined for 23 females and 21 males at 27 nest areas. Breeding dispersal was observed only for adult females and no adult male moved to a new home range or nest area. For 13 nests located by tracking adult females to a different nest area in consecutive years, the median distance moved was 18.5 km; maximum distance moved was152 km. In 55% of our consecutive year outcomes the goshawk pair nested in the same nest area as the previous year. Overall, 31% of adult females dispersed to a new home range in consecutive years and either nested with a different mate or did not nest. These complex dispersal patterns by adult female goshawks present challenges to those charged with monitoring goshawks and/or their nests.

Citation and abstract of thesis by Stephen Lewis on northern goshawk diet in Southeast Alaska.

Lewis, Stephen B. 2001. Breeding season diet of northern goshawks in Southeast Alaska with a comparison of techniques used to examine raptor diet. Thesis. Boise State University. Boise, Idaho. 125pp.

Chapter 1
A video surveillance system for monitoring raptor nests in a temperate rainforest environment.

Abstract: I used a video surveillance system to monitor northern goshawk (*Accipiter gentilis*) nests in the coastal temperate rainforest of Southeast Alaska to gather data on their diet. I maintained five systems during the goshawk nesting seasons in 1998 and 1999, installing the cameras an average of 10 days after hatching. At these 10 nests, cameras were maintained for an average of 33 days, recording 5834 hours of nest-time. I captured an average of 69.3% of the daylight hours available from hatching to the day nests were no longer used by juvenile northern goshawks. Technical difficulties associated with maintaining video cameras in this rainforest environment included electronic malfunctions, recurrent battery failure, and problems with the recorded image. However, these video surveillance systems effectively monitored northern goshawk nests and could be adapted for most rainforest raptors that nest on open platforms. I recommend testing the systems under field conditions in which they are to be used prior to deployment.

Chapter 2

Comparison of three techniques for assessing raptor diet during the breeding season.

Abstract: Video recording of prey deliveries at nests is a new technique for collecting data on diet and food habits that has not been compared with results from collections of prey remains and pellet. As part of a study of the breeding season diet of northern goshawks (*Accipiter gentilis*) in Southeast Alaska, I compared data from these three techniques to determine the relative merits of the different methods for assessing diet. I monitored 5 nests during the northern goshawk breeding seasons of 1998 and 1999 and identified 1,541 prey from deliveries, 209 prey from remains, and 209 prey from pellets. The proportions of birds and mammals varied among techniques, as did the relative proportions of prey groups and age groups. Analysis of prey deliveries gave the narrowest diet breadth of the three techniques. Prey remains and pellets gave the least similar diet descriptions. Over two-day intervals during which data was collected using all three techniques, prey deliveries gave more individual prey and prey categories than the other two techniques. I found that prey was not directly tracked through all three techniques. Analysis of prey deliveries collected by remote videography provided the most complete description of diet and I recommend that studies attempting to describe diet use this method or some other direct technique.

Continued on next page

Chapter 3

Breeding season diet of northern goshawks in Southeast Alaska.

Abstract: I provided the first systematic description and quantification of the nesting season diet of northern goshawks (*Accipiter gentilis*) in Southeast Alaska and examined how their diet varied within this island archipelago. I collected data on the diet of goshawks using three techniques. I used remote videography to record prey deliveries at nests in two spatially distinct locations of Southeast Alaska to describe the diet in detail and examine spatial variation in the diet. I used prey remains and pellets collected at nests throughout Southeast Alaska to describe the diet of the goshawk over a broader spatial scale. Goshawks delivered more birds than mammals overall of Southeast Alaska but delivered more birds in the Prince of Wales Island area than in other parts of Southeast Alaska. In northern Southeast Alaska, blue grouse (*Dendragapus obscurus*), red squirrels (*Tamiascurius hudsonicus*), Steller's jays (*Cyanocitta stelleri*), varied thrushes (*Ixoreus naevius*), northwestern crows (*Corvus caurina*) and unknown passerine birds were the prey that contributed the most to the diet. In southern Southeast Alaska, spruce grouse (*Falcipennis canadensis*), Steller's jay, ptarmigan species (*Lagopus* spp.), varied thrushes, and unknown passerine birds were the commonly eaten prey. Diet niche was narrower in the north than in the south and nests in these areas, on average, showed little overlap. The relative proportion of grouse and thrushes in the diet appeared to vary as the nesting season progressed, as did the relative proportion of different aged prey. Data from prey remains and pellets collected over all of Southeast Alaska provided similar results as that from remote videography. In Southeast Alaska, goshawks ate similar types of prey as seen in other locations. My data support the supposition that goshawks are generalist predators and show a certain amount of adaptability in their tolerance to varying prey bases. However, there appears to be a limit to this adaptability, which was apparent on Prince of Wales Island. In this area, an extremely restricted prey base in combination with extensive landscape alteration due to timber harvest appears to have affected goshawks' ability to successfully reproduce. Goshawks in Southeast Alaska rely on a few important prey species that can be affected by timber harvesting activities. Therefore, management should focus on conserving forests that structurally and functionally mimic those that historically covered this region.

Population Dynamics, Movements, and Habitat Selection of Martens on Kuiu Island, Southeast Alaska

Rodney W. Flynn[1], Carl H. Koch[1], Dave P. Gregovich[1], Grey W. Pendleton[1], and Natalie G. Dawson[2]

[1] Alaska Department of Fish and Game, Division of Wildlife Conservation,
P. O. Box 110024, Juneau, AK 99811, USA
rod.flynn@alaska.gov
907-465-4353

[2] College of Forestry and Conservation, University of Montana, Missoula, MT 59812
natalie.dawson@cfc.umt.edu
406-243-6956

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Introduction

To better manage marten (*Martes americana* and *M. caurina*) populations on Kuiu Island (Fig. 1), we initiated a 6-year research project in 2007. Previous research indicated that the marten population was low on Kuiu Island; among the lowest in Southeast Alaska (Flynn et al. 2004). We began by measuring marten catch rates, densities, and recruitment each autumn, and then measuring annual survival rates from radiocollared animals. We also monitored marten movements, assimilated diets, and habitat selection. In addition, we measured rodent abundance each autumn to determine whether their abundance affected marten diets and population dynamics.

In a previous interim report (Flynn et al. 2012), we provided preliminary information on live captures, catch rates, and densities of martens during 2007–2012. We have now completed the survival analysis through May 2013. Also, we computed how far each marten radio location was from the coast, suggesting the level of vulnerability each marten had to beach trapping. We have not done additional analyses on habitat selection, so it is not included here. As of October 2013, we had 24 active radiocollars on martens and we will continue monitoring them through May 2014.

Study Area

Kuiu Island (982 km^2), located about 75 km west of Petersburg (lat 56˚28'N, long 134˚1'W), was selected for the study area (Fig. 1). At 1,929 km^2, Kuiu Island is the 7th largest island in Southeast Alaska, forming part of the Alexander Archipelago. Our primary study area on the

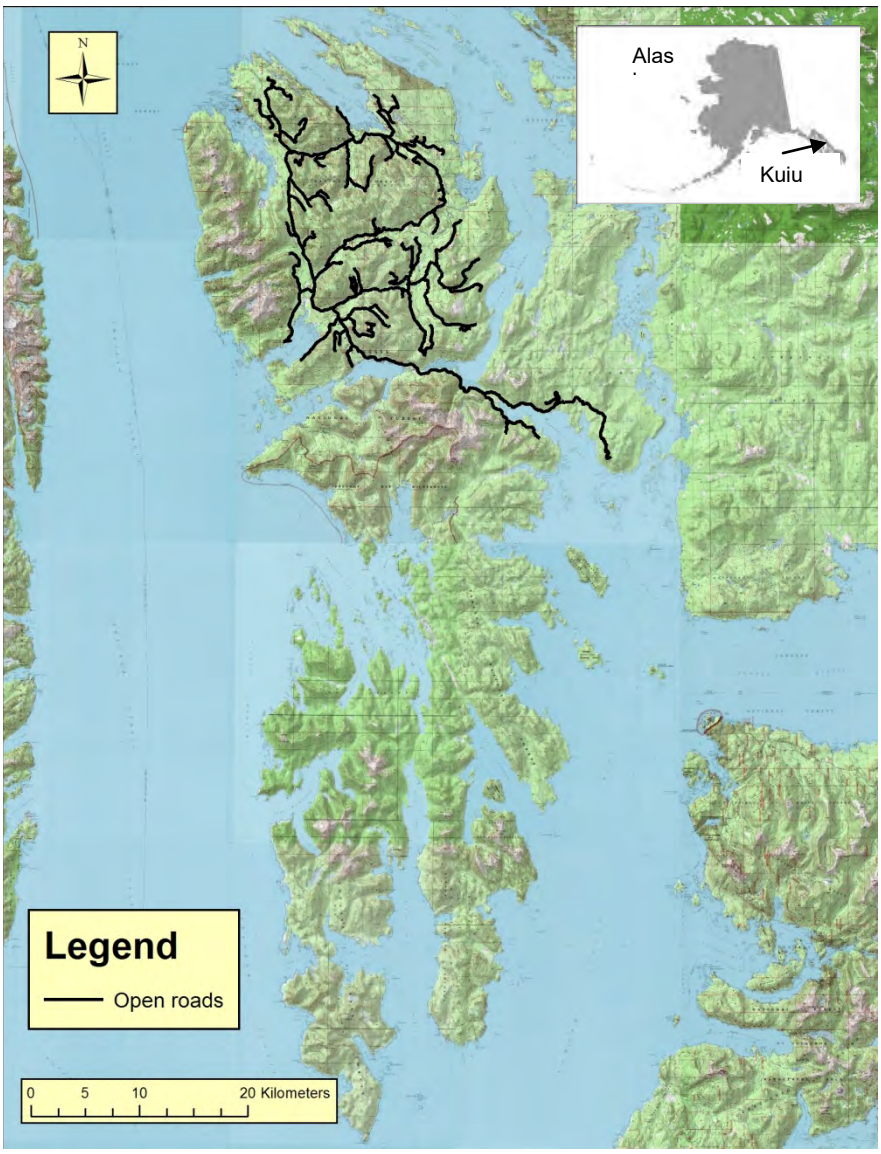

Figure 1. Study area on Kuiu Island, Southeast Alaska, showing the active logging roads.

northern portion of Kuiu represents about 50% of the island (Fig. 1). The land is mostly part of the Tongass National Forest under the jurisdiction of the U. S. Forest Service (USFS).

The Alexander Archipelago has a maritime climate; summers are cool and wet and winters are characterized by deep snow (2,360 mm annual precipitation). The snow-free period extends from early May to early November at lower elevations (< 1000 m). Vegetation at higher elevations is typically alpine tundra, and in lower elevations vegetation consists of coastal, old-growth forest of Sitka spruce (*Picea sitchensis*) and western hemlock (*Tsuga heterophylla*) with well-developed understory (mainly *Oplopanax horridum*, *Vaccinium* sp., *Menziesia ferruginea*, and *Rubus spp*.). Over 12% of the land area on our primary study area has been affected by clear-cut logging.

Mammal species of interest on Kuiu Island include red squirrel (*Tamiasciurus hudsonicus*), American beaver (*Castor canadensis*), long-tailed vole (*Microtus longicaudus*), northwestern deermouse (*Peromyscus keeni*), cinereous shrew (*Sorex cinereus),* dusky shrews (*Sorex monticolus*), gray wolf (*Canis lupus*), American black bear (*Ursus americanus*), river otters (*Lontra canadensis*), ermine (*Mustela erminea*), American mink (*Neovison vison*), moose (*Alces americanus*), and Sitka black-tailed deer (*Odocoileus hemionus sitkensis*). Although present, deer numbers have been low for many years.

Methods

CAPTURES

We captured martens in live traps placed at sites spaced about 500 m apart, located

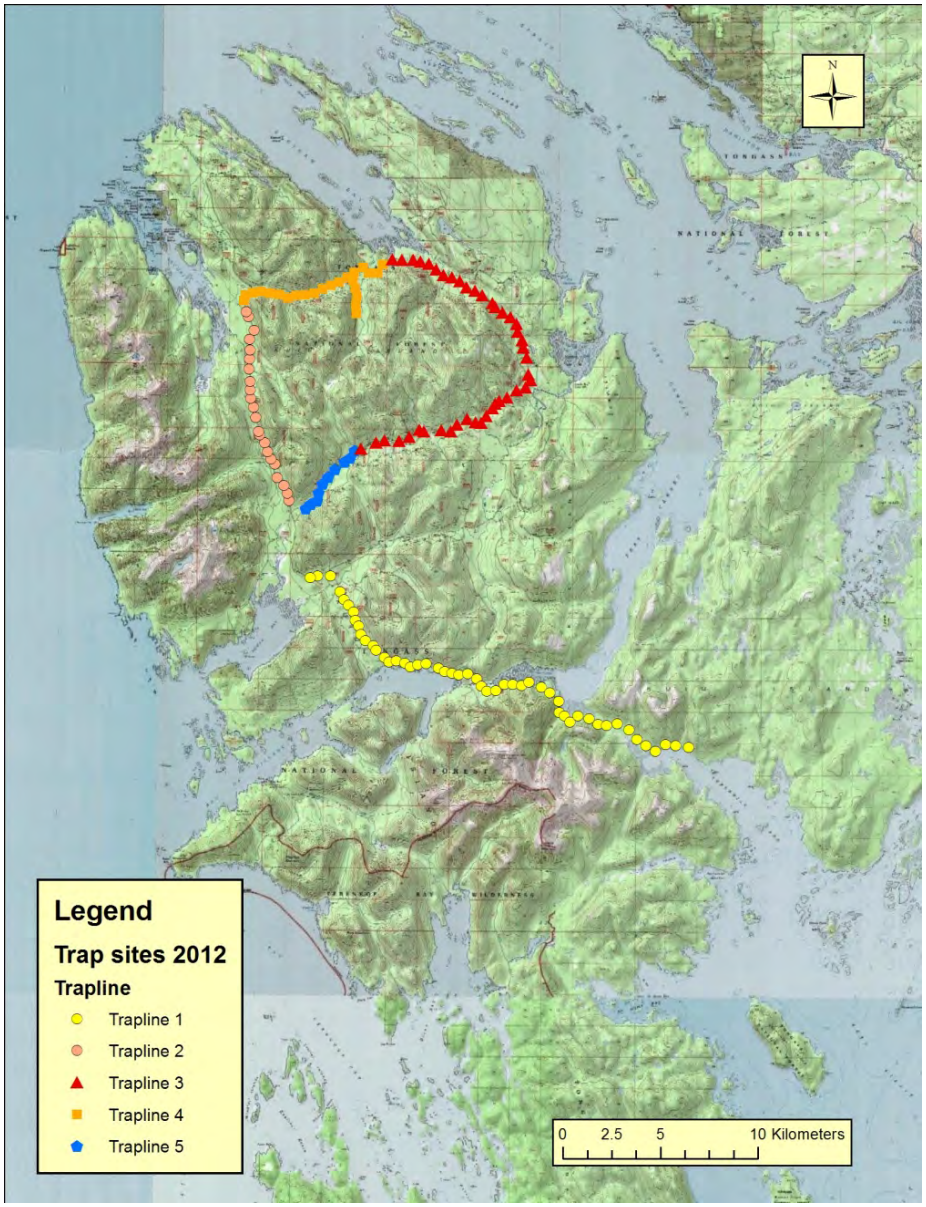

Figure 2. Study area on northern Kuiu Island, Southeast Alaska, showing marten live-trap sites during 2012. We added Trapline 4 and 5 for 2013 only.

systematically along 3 sections of the logging road system on northern Kuiu Island (Fig. 2). At each trap site, we placed a single live trap (single door Model 203, Tomahawk Live Trap Co., Tomahawk, WI). Traps were baited with strawberry jam, sardines, or venison scraps and placed under logs or the base of trees within 50 m of the road. Additionally, a commercial attractant marten lure (S. Stanley Hawbaker & Sons, Fort Loudon, PA) was used at each trap site. We covered traps with a piece of tarp to protect captured animals from the rain and hypothermia. We checked traps at least daily.

We placed one trapline (Port Camden) from 4.7 km east of Rowan Bay eastward 25 km along USFS Roads 6402 and 6434, ending on the north side of Threemile Arm, for 50 trap sites. Another trapline (Security Bay) started 7 km north of Rowan Bay on USFS Road 6402 and ended near Security Bay (22 trap sites). The third trapline (Kadake Bay) started near Saginaw Bay and extended along USFS Road 6415 to a point 5.2 km west of Kadake Bay (32 trap sites). During 2008-2011, we increased this trapline to include 11 more traps (43 trap sites) (Fig. 2). In 2012, we extended the northern trapline to include Trapline 4 (originally used for hair-snagged sites) and Trapline 5 (also originally used for hair-snagged sites) (Fig. 2).

The capture period was varied by year, but was typically from mid September to late October. We sedated captured martens with an injection of a mixture of 18.0 mg/kg ketamine hydrochloride (Ketaset, Fort Dodge Animal Health, Fort Dodge, IA) and 1.6 mg/kg xylazine hydrochloride (The Butler Company, Columbus, OH). All captured martens were eartagged (Size 1, Style 1005, National Band and Tag Co., Newport, KY), sexed, weighed, and measured (total, body, and tail length). Two first premolar teeth (upper or lower) were pulled for age determination by cementum analysis (Matson's Laboratory, Milltown, MT). We grouped martens into 2 age classes – juveniles (cementum age = 0) and adult (cementum ≥1).

Most of the captured martens were radiocollared. On female martens, we attached a radiocollar around the neck that weighed about 35 g with an expected life of 12 months (Telonics MOD-073, Telonics, Mesa, AZ). On males, we attached a 49-g collar (Telonics MOD-080, expected life of 12-18 months). After martens recovered from the immobilization, we fed them a teaspoon of jam for a short-term energy boost and released them at or near the capture sites. Martens recaptured during the same trapping session were released without additional processing. All capture and handling procedures were approved by the Animal Care and Use Committee of the Alaska Department of Fish and Game and were consistent with the animal care guidelines of the American Society of Mammalogists (Gannon et al. 2007).

LOCATIONS AND MOVEMENTS

We located radiocollared martens from small aircraft (Mech 1974) during daylight hours throughout the year at about monthly intervals to determine movement patterns, survival, and habitat selection. After we located an animal by circling in the aircraft, we determined the animal's geographic coordinates using a global positioning system (GPS) device. We found that our location accuracy averaged about 105 m (n = 15, SE = 33) based on comparisons of known locations with telemetry locations. Because the pulse rate of radiocollar transmitters change after the collar has not moved 6 hours (mortality mode), we determined where and when mortalities occurred.

We entered all location data into a geographic information system (GIS) file using ArcGIS 10 software (ESRI, Redlands, CA) on a personal computer. All spatial analyses were completed using GIS software, including spatial distribution of captures, subsequent relocations, home ranges, and distances traveled. We used only locations that were rated fair to excellent in our analyses.

Distance from the beach

We analyzed locations from animals that we had ≥10 telemetry locations for any time during the year (n = 71 martens). We analyzed proportional use by martens of terrain bands delineated by increasing distance from the shoreline of Kuiu Island based on telemetry locations collected in the years 2007-2013. The function **`gDistance`** from the R package **`rgeos`** was used to determine all distances (R Core Team 2013).

We computed the mean proportion of individual marten locations collected in each of a series of terrain bands delineated by distance from the shoreline in 250 m and 1000 m increments. We also determined the proportion of martens for which the minimum distance to shore of any telemetry location fell within each of these 250 m and 1000 m bands.

We analyzed the data for three different time frames: 1) a window roughly corresponding with the potential trapping season (December, January, and February); 2) all other months (March–November); and 3) all months combined.

POPULATION DYNAMICS

Abundance and density

Marten numbers were computed using a spatial capture-mark-recapture (CMR) approach (Efford et al. 2004) for the northern and southern traplines (Fig 2.) Martens were "captured" by live trapping or hair snagging. Capture histories were used to estimate population size and densities for each session and year. In addition, capture rates [unique captures per 100 trap-nights (TNs)] were computed for 3 traplines (Fig. 2). The mean capture rates were compared as a measure of population abundance (Flynn et al. 2004).

Recruitment

We computed percentages of juvenile martens that were live-trapped as a measure of recruitment.

Survival

We used known-fate analyses (program MARK, version 7.0; Cooch and White 2013) to estimate survival and examine the effects of age, sex, year, and season (1-3 month intervals) on survival. Survival probability (S) is the probability of surviving an interval between sampling occasions. We used a staged-entry design with censoring. Animals were added at live capture or when their radio signals were found after missing a time period. Animals were censured when we could not pick up the signal during a time period. Mortality was assumed to have occurred when a

mortality signal from the VHF collar was heard. We located the animal again on the next flight to confirm the mortality. We assigned all locations and mortalities to designated time categories as follows: 16 May – 30 September (summer), 1 October – 31 October (autumn), 1 November – 31 December (late autumn), 1 January – 28 February (winter), 1 March – 30 April (early spring), and 1 May – 15 May (late spring). We used sex, age class, year, and time period as covariates.

Generally, survival probability is the only parameter in known-fate models; differing from mark-resight models in that detection probability is assumed to be 1 (Cooch and White 2013). We incorporated that methodology with radio-tagged martens in this study. We considered 33 models with varying combinations of the predictor variables ranging from the simplest model that assumed constant survival, to complex models incorporating all 4 predictors. In separate models, year was modeled as either a categorical variable (i.e., each year having a separate survival) or continuous variable (i.e., the effect of year followed a linear or quadratic pattern over time). Model utility was evaluated using AICc and estimates were averaged across models based on AICc weights (Burnham and Anderson 2002). Annual survival estimates were produced as a product of the within-year seasonal estimates with variances of the annual estimates approximated using the delta method (Williams et al. 2001:736-737).

RODENT ABUNDANCE

We estimated the relative abundance of rodents during autumn using a snap-trap index similar to Calhoun (1948). We calculated the transect mean as a measure of rodent abundance. We originally established transects in 4 old-growth forest stands (conifer and mixed conifer) and 1 transect in nonforest along a beaver pond complex. Beginning in 2008, we additionally performed a transect in a riparian spruce stand. Each transect consisted of 25 trap stations with 2 Museum Special traps at each station placed at 15-m intervals. We baited traps with a mixture of peanut butter and rolled oats, and set them for 3 consecutive nights. Traps were checked once daily. All specimens were sent to the University of New Mexico Museum.

We used snap-trapping and transects because these approaches usually provide more captures of rodents than live-trapping and grids (Pendleton and Davison 1982, Pearson and Ruggiero 2003). Numbers of animals captured by species was expressed as the number of captures per 100 TNs. We used this index for relative rodent abundance because this method provides reliable data with a minimal investment of time and effort (Southern 1965). Also, Slade and Blair (2000) found that counts of rodents were proportional to population size estimated by mark-recapture methods. We kept capture probability similar by using a standardized trapping protocol: 1) a short sampling period; 2) sampling in similar habitats (productive old-growth forest); 3) and sampling during the same time of year (autumn).

Results and Discussion

CAPTURES

We live-trapped 160 individual martens (86 males, 74 females) 271 times over the course of the study. We captured some of the same martens multiple times in the same year and different years, so annual capture totals don't equal the number of captures by year. We captured 27 martens in 2007 (15 males, 12 females) 17 (10 males, 7 females) in 2008, 34 (21 males, 13

females) in 2009, 33 (20 males, 13 females) in 2010, 43 (19 males, 24 females) in 2011, and 41 (21 males, 20 females) in 2012. We radiocollared 140 martens (79 males, 61 females).

LOCATIONS AND MOVEMENTS

We located 140 radiocollared martens (79 males, 61 females) 1,455 times between 11 November 2007 and 15 May 2013. For 71 martens, we had ≥10 total telemetry locations across this time period.

Distance from the beach

Preliminary analyses (Fig. 3) showed a nearly identical spatial distribution of telemetry points in relation to the shoreline for the three time frames (trapping season, other months, and all months

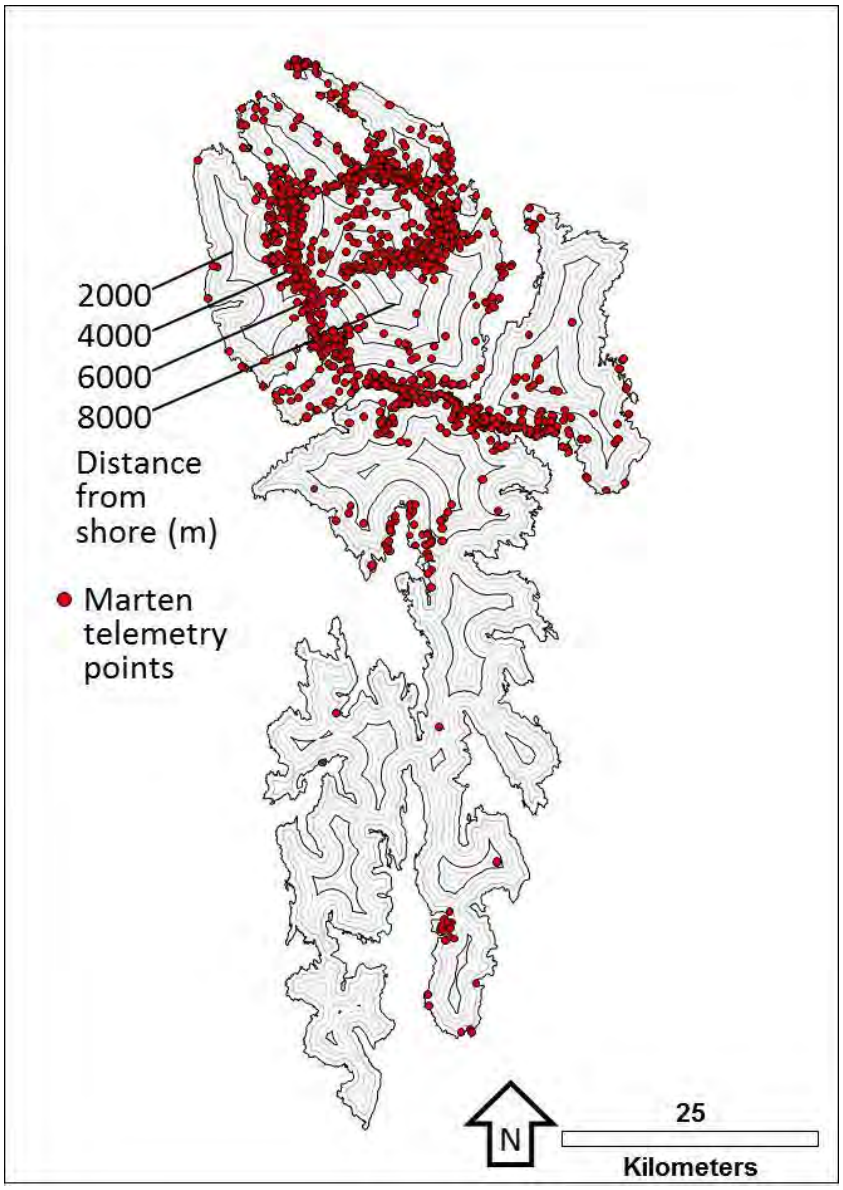

Figure 3. Marten telemetry points with distance bands used in analyses. Distance contours are shown at 250 (gray) and 1000 m (black) intervals.

combined), so results are presented for all months combined.

The probability of marten use decreased markedly with increasing distance from shore (Fig. 4A, 5A). The highest proportion of use was within 250 m of the shore (0.22), and was nearly twice that of the next most proximate (250-500 m) band (0.12). Roughly half (0.49) of martens had 20% or greater of their telemetry locations within 250 m of shore, and roughly half (0.55) of martens had 50% or greater of their telemetry locations within 1000 m of shore.

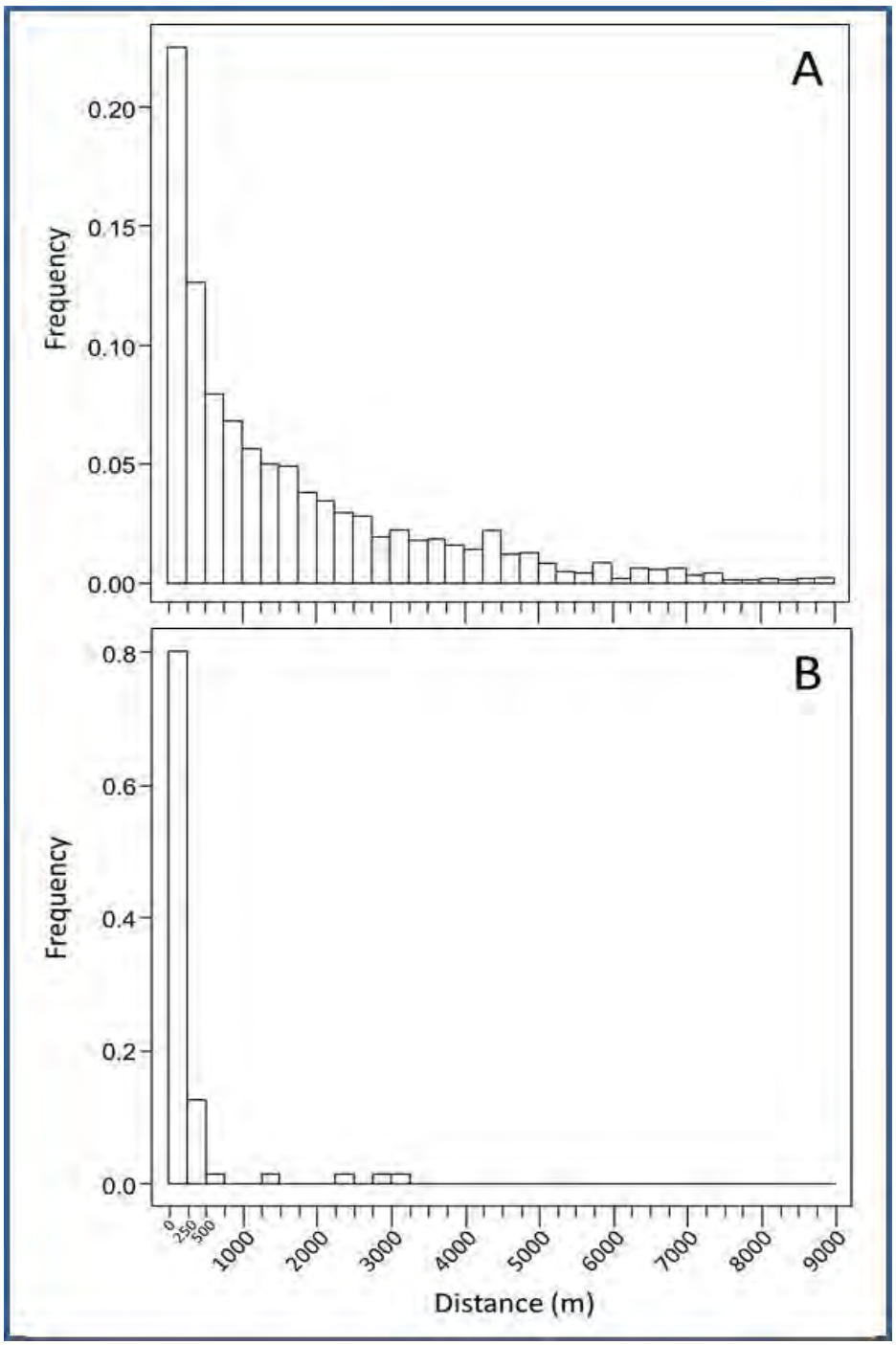

Figure 4. Histograms of the mean proportional use (A) and minimum distance from shore (B) of martens in terrain bands defined by 250 m increments.

Most martens used areas near the shoreline; 80.2% were located at some point in time within 250 m of the shore, and 94.3% were located at some point in time within 1,000 m of the shore (Fig. 4B, 5B). Because telemetry locations represent a vastly reduced subset of the actual locations visited by animals, it is likely that these percentages are biased low.

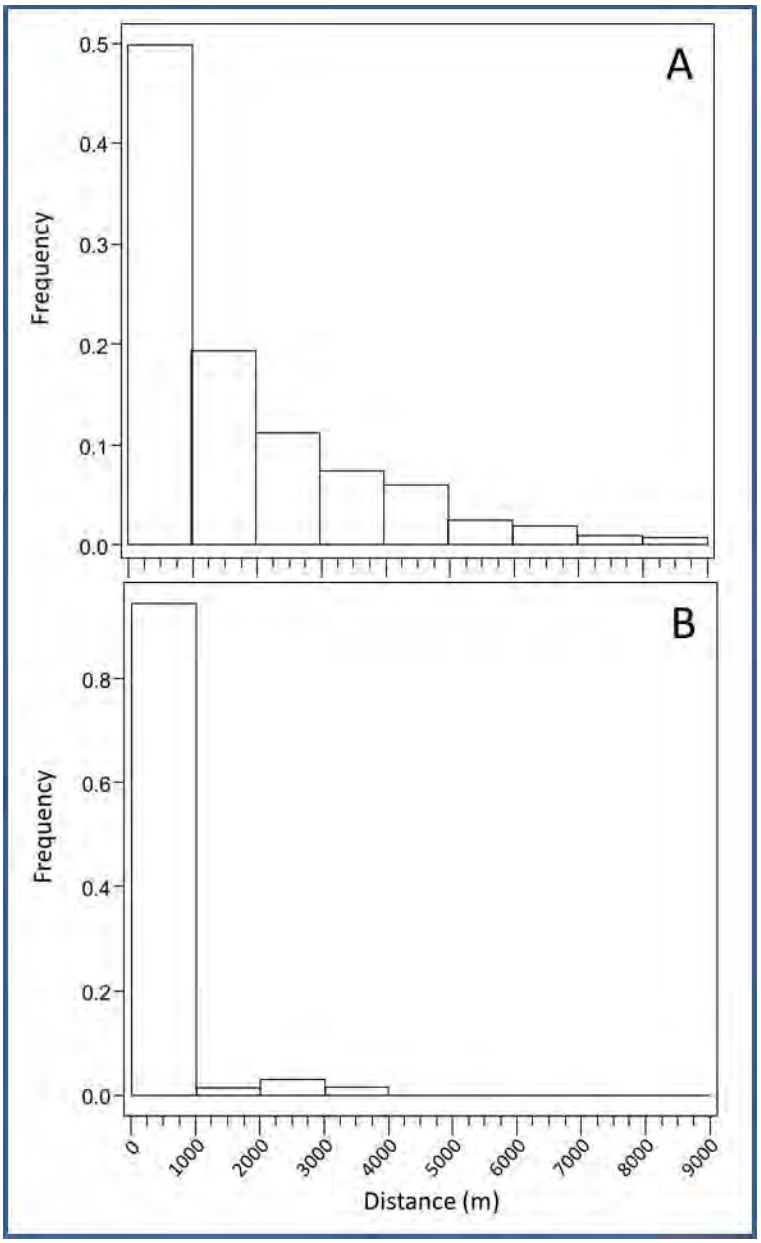

Figure 5. Histograms of the mean proportional use (A) and minimum distance from shore (B) of martens in terrain bands defined by 1000 m increments.

POPULATION DYNAMICS

Abundance and density

We summarized the abundance and density data in our 2012 progress report (Flynn el at. 2012). This information was summarized and included in this report(Table 1, Fig. 6, 7).

Table 1. Mean catch rates (captures/100 TNs) for martens and rodents for northern Kuiu Island, 2007–2012. We computed marten densities (mi^2) according to methods of Efford et al. (2004). These data were graphed in Figure 6. For 2008–2009, we had too few recaptures to estimate density precisely.

Year	Mean marten captures (captures/100 TNs)	Marten density (mi^2)	Lower CI	Upper CI
2007	2.0	0.48	0.18	1.29
2008	1.0			
2009	2.1			
2010	2.1	0.25	0.12	0.55
2011	3.2	0.65	0.31	1.35
2012	2.1	0.70	0.34	1.42

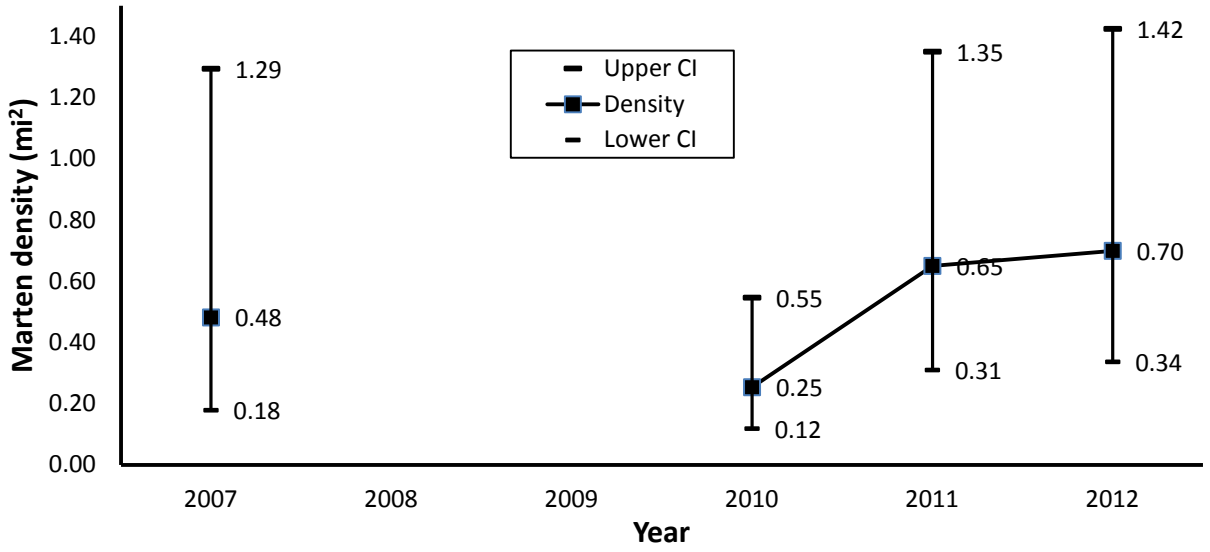

Figure 6. Preliminary density estimate for martens on northern Kuiu Island, Southeast Alaska, 2007–2012, including 95% confidence intervals. We combined Traplines 2–3 and added Traplines 4 and 5 in 2012. In 2008–2009, we didn't have enough recaptures to estimate density.

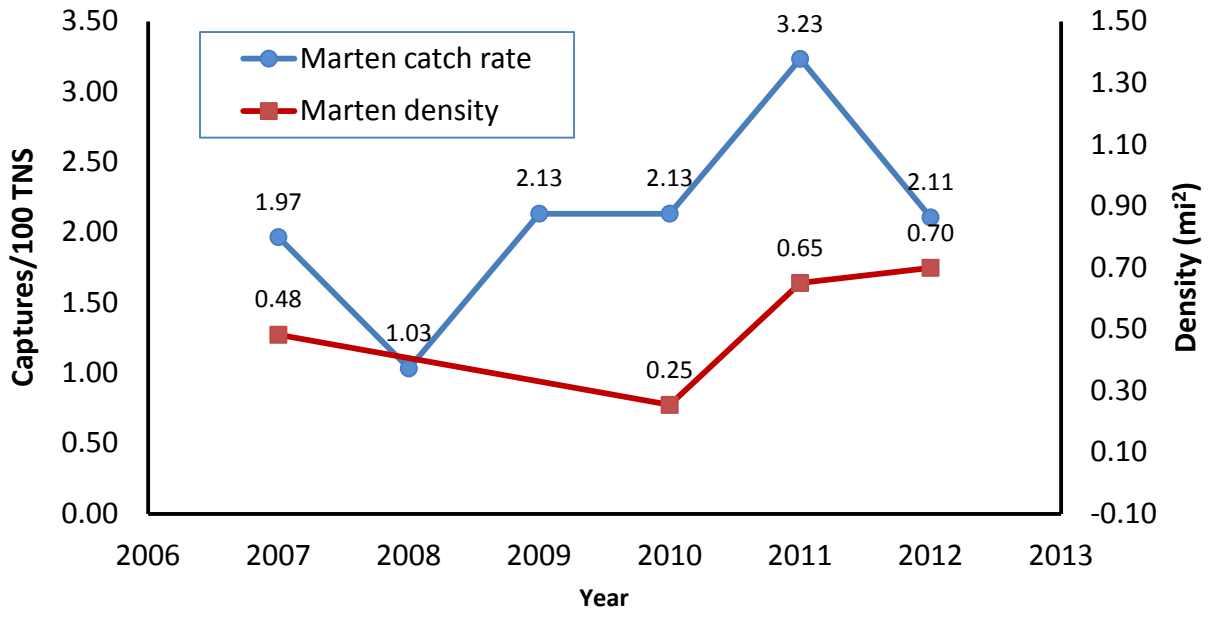

Figure 7. Relationship between marten densities and mean catch rates (Traplines 1-3) on northern Kuiu Island, Southeast Alaska, 2007–2012.

Recruitment

In 2011, we observed an increase in recruitment, but in 2012 recruitment declined to low levels, similar to 2007 and 2008 (Fig. 8).

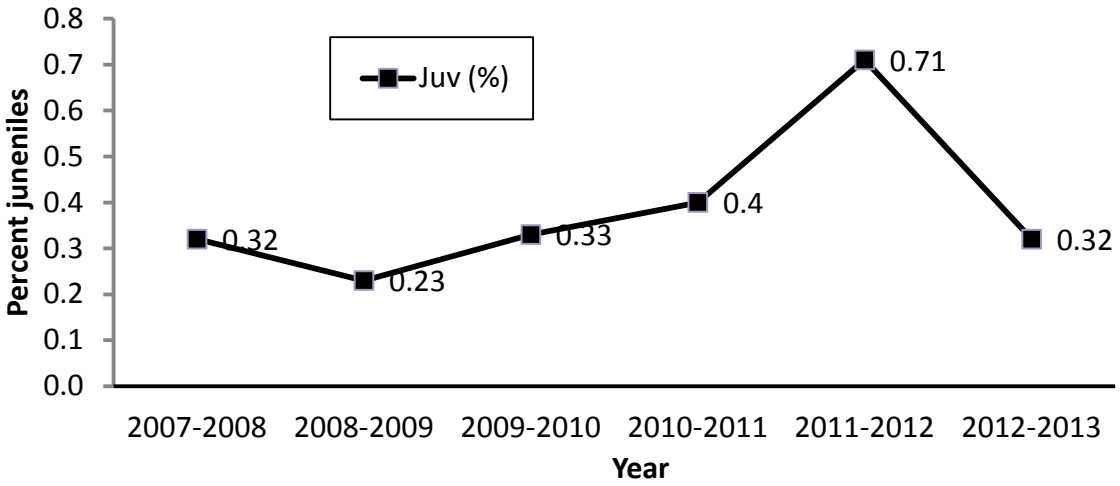

Figure 8. Percent juveniles of captured martens on Kuiu Island, Southeast Alaska, 2007–2012.

Survival

Marten survival was low (0.42) in 2007-2008 and increased steadily until 2010-2011 (Fig. 9). Beginning in 2011-2012, survival declined significantly (0.74) and continued to decline in 2012-2013. For all years, most of the mortality occurred in late winter and early spring. We found age

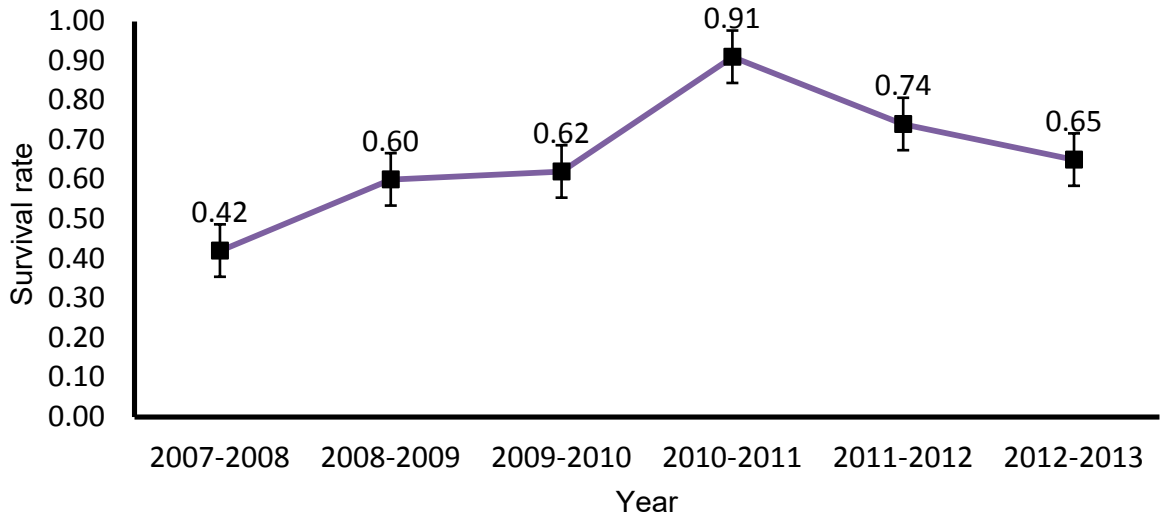

Figure 9. Annual survival rate (±SE) for the marten population on Kuiu Island, Southeast Alaska, 2007-2008 through 2012-2013.

class to be a significant covariate for survival. Juvenile martens have lower survival probably than adults.

RODENT ABUNDANCE

We summarized mean rodent abundance from 2007 to 2012 (Table 2). Rodents consisted mostly of deer mice.

Table 2. Mean abundance of rodents for 6 individual transects on northern Kuiu Island, Southeast Alaska, 2007–2012. The rodents were mostly deer mice. We did not include shrews.

Year	PEKE[a]	MILO[a]
2007	1.7	0.1
2008	4.4	0.0
2009	9.4	0.1
2010	6.8	0.2
2011	9.6	0.0
2012	6.7	0.0

[a] PEKE = *Peromyscus keeni,* MILO = *Microtus longicaudus*;

We compared rodent abundance to marten catch rates and densities (Fig. 10). Rodent abundance increased greatly from 2007 (1.8 rodents/100 TNs) until 2009 (9.3 rodents/100 TNs). From 2009–2012, rodent abundance has varied between 6.7–9.6 rodents/100 TNs.

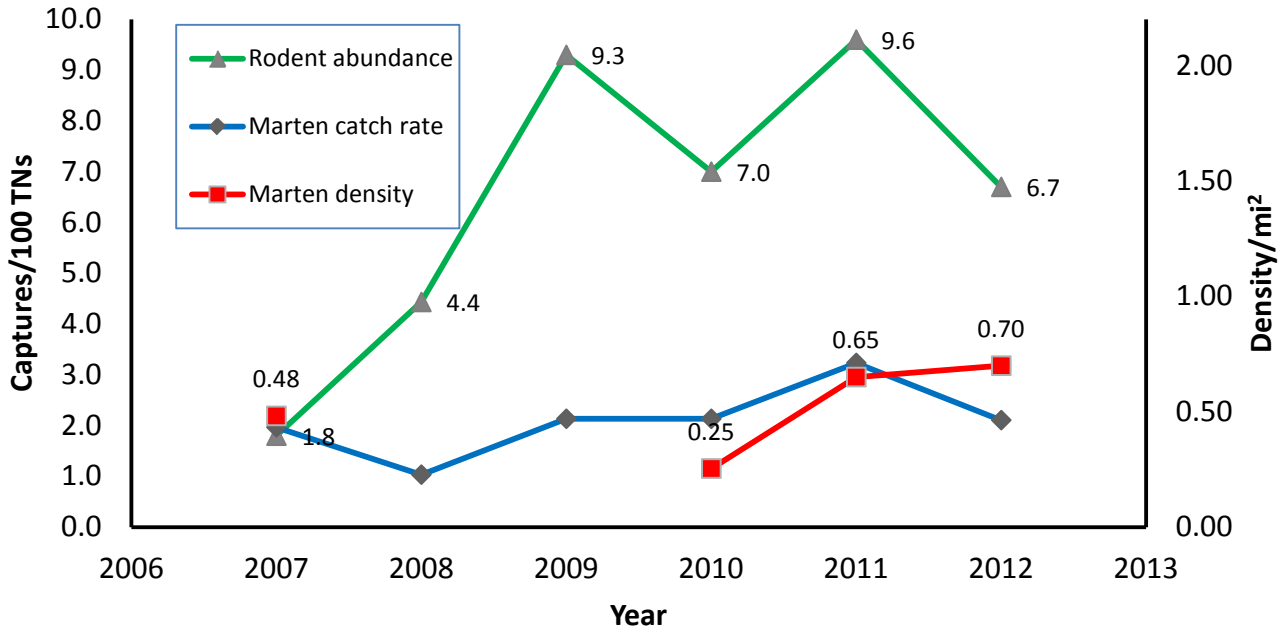

Figure 10. Relationship among rodent abundance and marten catch rate and densities for Kuiu Island, Southeast Alaska, 2007–2012.

Summary

The marten population on Kuiu Island remains low. Our density estimates ranged from 0.25-0.70 martens/ mi^2 over the course of the study. Although there is a slight increase in our marten density estimates in 2011-2012 relative to earlier study years, no density estimates over the course of the study differed significantly from each other.

In contrast to the densities observed on Kuiu Island, Flynn and Schumacher (2009) recorded a minimum density of 0.44 martens/mi^2 and a peak density of 1.42 martens/mi^2 on Chichagof Island. In the Yukon, Archibald and Jessup (1984) reported autumn densities of 1.55 martens/mi^2. Francis and Stephenson (1972) estimated densities of martens in their Ontario study area to be 1.2–1.9 martens/km^2 (3.1–4.9 martens/mi^2).

Our density estimates are relatively imprecise because we did not experience sufficient recaptures. Efford et al. (2004) recommends getting at least 10 recaptures in a trapping session. We had 6 recaptures in 2007, 12 in 2010, 12 in 2011, and 10 in 2012. Despite the imprecision in our marten density estimates, all estimates consistently indicated low numbers of martens relative to other areas in which they have been studied.

We also consider catch rates as an alternative to measuring abundance. Catch rates have varied over the past 6 years, ranging from 1.0 capture/100 TNs to 3.2 captures/100 TNs.

The Kuiu marten population has not responded to an increase in rodent numbers. However, the increase in rodents is mostly due to an increase in deer mice, which are not a preferred food resource of martens (Flynn et al. 2004). Deer numbers remain low on Kuiu. Numerous studies in other regions have indicated that deer and/or other ungulates, in the form of carrion, are an important component of marten diets (Buskirk and Ruggiero 1994).

In 2012, marten recruitment (% juveniles) was poor on Kuiu Island. Marten recruitment was low on Chichagof Island when food was low (Flynn and Schumacher 2009). During low food years, Flynn and Schumacher (2009) also documented that the ovulation rates for females was much lower.

Survival probability was low at the beginning of our study, but increased through 2010-2011 when most martens survived. In the last 2 years, survival has declined substantially, being relatively low in 2012-2013. Flynn and Schumacher (2009) recorded lower marten survival when food was scarce.

Apparently, survival and recruitment have not been adequate for the Kuiu Island marten population to increase since 2010. We have not seen any increase in catch rates or density since 2011. We have not done any actual population modeling to estimate lambda (net finite rate of increase) because it is beyond the scope of our current project. Nonetheless, we can assume that lambda is below 1 because we haven't seen any increase in population numbers from 2011-2013. For any harvest surplus, lambda has to be greater than 1 with no harvesting, unless the harvesting is entirely compensatory.

Most martens on Kuiu Island use the beach fringe to a high degree. Kuiu Island is a long, narrow island with substantial beach fringe habitat. More than 80% of the martens were located at some point in time within 250 m of the shoreline, and 94% were located at some point in time within 1,000 m of the shoreline. Because martens are relatively easy to catch (Flynn and Schumacher 2009), they would be highly vulnerable to trapping by boat-based trappers along the coast.

We used road access to captured martens for our study. Thus, every marten captured in this study had some segment of road in their home range and would be vulnerable to trapping along the road system. Martens are known to be easy to catch by trappers and the allowance of trapping along roads can result in high catch rates (Strickland and Douglas 1984, Flynn and Schumacher 2009).

Management Implications

Evidence suggests that the marten population on northern Kuiu Island is still low and probably below long-term sustainable levels. We are not sure whether the population will ever be very vigorous because of a chronic low food supply and habitat conditions.

Given these findings and uncertainties, we offer a couple of recommendations for ensuring the long-term conservation and sustainability of Kuiu Island martens: 1) retain the trapping season closure on Kuiu Island until the marten population has increased to a more sustainable level; or 2) create a refugia for martens in the center of Kuiu Island that would protect some animals from trapping. This could be accomplished by restricting trapping effort to coastal areas and disallowing trapping access along the island's road system during the trapping season. In conjunction with the road closure, we suggest shortening the trapping season to include only the month of December. In addition, we recommend requiring that trappers salvage skulls from trapped martens and provide them to our department for continued monitoring of the population's sex and age structure.

Future Activities

During the coming year (2013-2014), all radiocollared martens will be radio-tracked at least monthly to determine survival. All analyses will be completed in 2013-2014, and we plan to produce a final report in June 2014. Carl Koch will use data from this project for his Master of Science degree from the University of Wyoming, focusing on habitat selection, diets, and prey availabilities.

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This chapter provides procedural direction for the evaluation of timber and forest vegetation resources and the development of plan components in Forest, Grassland, and Prairie, or other comparable Administrative Unit plans. This handbook provides guidance to meet the requirements in 36 CFR 219.11 in the following order:

1. Identification of lands as not suited and suited for timber production.

2. Timber harvest for purposes of timber production.

3. Timber harvest for purposes other than timber production.

4. Limitations on timber harvest.

5. Land Management Guidance including display of forest vegetation management practices and timber harvest levels (36 CFR 219.7(f)(iv)).

This chapter provides guidance for developing land management plans, rather than guidance for individual projects. All projects and activities must be consistent with applicable plan components.

60.1 – Authority

The following law and regulation set forth the requirements for Forest Service planning:

1. Renewable Resource Planning Act of 1974 as amended by the National Forest Management Act of 1976 (16 U.S.C. 1600 et seq.). This act as amended sets forth the requirements for land management plans in the National Forest System. See FSM 1920 for specific requirements.

2. Title 36, Code of Federal Regulations, Part 219–Planning, Subpart A—National Forest System Land Management Planning (36 CFR part 219), published in the Federal Register on April 9, 2012 (77 FR 21162). This regulation provides direction on land management planning procedures on National Forest System (NFS) lands.

60.3 – Policy

Harvest of timber on National Forest System lands occurs for many different reasons, including ecological restoration, community protection in wildland-urban interfaces, habitat restoration, protection of municipal water supplies; and to contribute to economic sustainability through the production of timber, pulp for paper, specialty woods for furniture, and fuel as a renewable energy source. Timber harvest, whether for wood production, restoration, or other reasons, can support local businesses and employment.

Use this chapter in concert with Title 36, Code of Federal Regulations, part 219 (36 CFR part 219); the National Forest Management Act (NFMA), Title 16 U.S.C. 1604 and 1611; FSM 1900, and FSM 1920. Timber resource planning objectives, policies, and responsibilities are found in FSM 2410. Guidance on development of plan components for vegetation is found throughout chapter 20 of FSH 1909.12 and specifically for timber in section 23.22f.

60.5 – Definitions

Clearcutting. A regeneration harvest method that removes essentially all trees in a stand producing a fully exposed microclimate for the development of a new age class of trees. A clearcut may or may not have reserve trees left to attain goals other than regeneration.

Culmination of mean annual increment of growth. See "Mean annual increment of growth."

Even-aged stand. A stand of trees composed of a single age class (36 CFR 219.19).

Even-aged system. A planned sequence of treatments designed to maintain and regenerate a stand with predominantly one age class. The range of tree ages is usually less than 20 percent of the rotation (length in years). Treatments include clearcutting, seedtree, shelterwood, and coppice regeneration methods.

Final regeneration harvest. The final timber harvest in a sequence of harvests designed to regenerate a timber stand or release a regenerated stand. A final regeneration harvest could be a clearcut, removal cut of a shelterwood or seedtree system, or a selection cut.

Forest land. Land that is at least 10 percent occupied by forest trees of any size or formerly having had such tree cover and not currently developed for nonforest uses. Land developed for nonforest use includes areas for agricultural crops, improved pasture, residential or administrative areas, roads of any width and adjoining road clearing, and powerline clearing of any width (36 CFR 219.19).

Fuelwood. Wood used for conversion to some form of energy.

Growing stock. All trees growing in a forest or in a specified part of it, usually commercial species, meeting specified standards of size, quality, and vigor, and generally expressed in terms of trees per acre, density, or volume.

Land that may be suitable for timber production. A preliminary classification in the process of determining lands that are suited for timber production. This preliminary classification excludes National Forest System lands that are not suitable for timber production based on the factors identified in 36 CFR 219.11(a)(1)(i), (ii), (iv), (v), and

(vi), and is made prior to the consideration of the factor at 36 CFR 219.11(a)(iii), which identifies suitability based on objectives and desired conditions established by the plan for those lands.

Management practices (Vegetation management practices). Silvicultural practices such as reforestation, prescribed fire, thinning to reduce stand density, and other practices designed to facilitate growth and development of trees.

Management intensities. The set and schedule of management practices typically used for certain forest or timber types to achieve desired conditions that may include timber production.

Mean annual increment of growth and culmination of mean annual increment of growth. The mean annual increment of growth is the total increment of increase of volume of a stand (standing crop plus thinnings) up to a given age divided by that age. The culmination of mean annual increment of growth is the age in the growth cycle of an even-aged stand at which the average annual rate of increase of volume is at a maximum. In land management plans, the mean annual increment of growth is expressed in cubic measure and is based on the expected growth of stands according to intensities and utilization guidelines in the plan (36 CFR 219.19).

Nonforest land. Lands that do not meet the definition of forest land.

Projected timber sale quantity (PTSQ). The estimated quantity of timber meeting applicable utilization standards that is expected to be sold during the plan period. As a subset of the projected wood sale quantity (PWSQ), the projected timber sale quantity includes volume from timber harvest for any purpose from all lands in the plan area based on expected harvests that would be consistent with the plan components. The PTSQ is also based on the planning unit's fiscal capability and organizational capacity. PTSQ is not a target nor a limitation on harvest, and is not an objective unless the responsible official chooses to make it an objective in the plan.

Projected wood sale quantity (PWSQ). The estimated quantity of timber and all other wood products that is expected to be sold from the plan area for the plan period. The PWSQ consists of the projected timber sale quantity as well as other woody material such as fuelwood, firewood, or biomass that is also expected to be available for sale. The PWSQ includes volume from timber harvest for any purpose based on expected harvests that would be consistent with the plan components. The PWSQ is also based on the planning unit's fiscal capability and organizational capacity. PWSQ is not a target nor a limitation on harvest, and is not an objective unless the responsible official chooses to make it an objective in the plan.

Reasonable assurance. A judgment made by the Responsible Official based on best available scientific information and local professional experience that practices based on existing technology and knowledge are likely to deliver the intended results. Reasonable assurance applies to average and foreseeable conditions for the area and does not constitute a guarantee to achieve the intended results.

Regeneration harvest. Any removal of trees intended to assist in the regeneration of a new age class or to make regeneration of a new age class possible. Regeneration harvest may be through even-aged or uneven-aged methods.

Restocked. The condition of the growing space occupancy of trees to be achieved after a disturbance that has substantially altered the existing stocking (see definition of "Stocking").

Rotation. The number of years (including the regeneration period) required to establish and grow timber under an even-aged management system to a specified condition or maturity for regeneration harvest.

Salvage harvest. The removal of dead trees or trees damaged or dying because of injurious agents, other than competition, that recovers economic value that would otherwise be lost, or because the removal of the dead or damaged trees contributes to achieving plan desired conditions or objectives.

Stand. A contiguous group of trees sufficiently uniform in age class distribution, composition, and structure, and growing on a site of sufficiently uniform quality, to be a distinguishable unit, such as mixed, pure, even-aged, and uneven-aged stands.

Stocking. An indication of growing space occupancy of trees relative to plan-defined desired conditions for the stand or area. Common indices of stocking include the number of trees by size and spacing, percent occupancy, basal area, relative density or crown completion factor.

Suitability of lands. A determination made regarding the appropriateness of various lands within a plan area for various uses or activities, based on the desired conditions applicable to those lands. The terms suitable and suited and not suitable and not suited can be considered the same.

Sustained yield limit (SYL). The amount of timber, meeting applicable utilization standards, "which can be removed from [a] forest annually in perpetuity on a sustained-yield basis" (NFMA at section 11, 16 USC 1611; 36 CFR 219.11(d)(6))). It is the volume that could be produced in perpetuity on lands that *may be suitable* for timber production. Calculation of the limit includes volume from lands that may be deemed not suitable for timber production after further analysis during the planning process. The

calculation of the SYL is not limited by land management plan desired condition, other plan components, or the planning unit's fiscal capability and organizational capacity. The SYL is not a target but is a limitation on harvest, except when the plan allows for a departure.

Timber harvest. The removal of trees for wood fiber use and other multiple-use purposes (36 CFR 219.19).

Timber production. The purposeful growing, tending, harvesting, and regeneration of regulated crops of trees to be cut into logs, bolts, or other round sections for industrial or consumer use (36 CFR 219.19).

Two-aged system. A planned sequence of treatments designed to regenerate or maintain a timber stand with two age classes. A two-aged system is a form of even-aged management.

Uneven-aged stand. A stand of trees of three or more distinct age classes, either intimately mixed or in groups.

Uneven-aged system. A planned sequence of treatments designed to regenerate or maintain a timber stand with three or more age classes. Treatments include single-tree selection, and group selection regeneration methods.

Utilization standards. Utilization standards are specifications for merchantable forest products offered in a timber sale.

61 – IDENTIFICATION OF LANDS AS NOT SUITABLE AND SUITABLE FOR TIMBER PRODUCTION

When developing or revising a land management plan the Responsible Official shall review lands within the plan area to identify their suitability for timber production. For a plan amendment, the Responsible Official may conduct a review of lands suitable for timber production if relevant to the issues of the amendment. The Interdisciplinary Team shall summarize the results of this review in the plan or an appendix to the plan (sec. 61, ex. 01 of this Handbook).

Section 219.11 (a)(1) of the Planning Rule lists six factors to be used to identify lands in the plan area as not suited for timber production. The following excerpt from section 219.11(a)(1) of the Planning Rule presents each of these six factors, and next to each is a cross-reference to the section of this chapter containing more detail on the factor.

219.11 Timber requirements based on the NFMA.

***

(a) ***Lands not suited for timber production.*** **(1) The responsible official shall identify lands within the plan area as not suited for timber production if any one of the following factors applies:**

(i) Statute, Executive order, or regulation prohibits timber production on the land; (sec. 61.11 of this Handbook).

(ii) The Secretary of Agriculture or the Chief of the Forest Service has withdrawn the land from timber production; (sec. 61.11 of this Handbook).

(iii) Timber production would not be compatible with the achievement of desired conditions and objectives established by the plan for those lands; (sec. 61.2of this Handbook).

(iv) The technology is not currently available for conducting timber harvest without causing irreversible damage to soil, slope, or other watershed conditions; (sec. 61.12 of this Handbook).

(v) There is no reasonable assurance that such lands can be adequately restocked within 5 years after final regeneration harvest; or (sec. 61.13 of this Handbook).

(vi) The land is not forest land (sec. 61.14 of this Handbook).

To identify lands not suited for timber production, use the following two step approach:

1. Identify lands that are not suited based on legal and technical factors at 36 CFR 219.11 (a) (i), (ii), (iv), (v,) and (vi), further described in sections 61.11 to 61.14 of this Handbook. If any of these factors apply to the land, the land is not suited for timber production. These lands do not vary by alternative in the plan EIS and may be identified during the assessment or prior to the development of alternatives.

After subtracting the lands that are not suited from the total of National Forest System lands, the remaining lands are lands that *may be* suited for timber production, and are considered in step 2.

2. From the lands that *may* be suited for timber production (the remaining lands from step 1), identify lands that *are* suited for timber production, based on the compatibility of timber production with the desired conditions and objectives for those lands (sec. 61.2 of this Handbook). This second step process is done in the EIS for each alternative considered in plan development or plan revision, as the desired conditions, objectives, management areas and other plan components will vary among alternatives.

> Also in each alternative after subtracting the lands that are suited for timber production (from step 2) from the lands that *may* be suited for timber production (from step 1); the remaining lands are not suited for timber production because timber production is not compatible with the land area's desired conditions and objectives (36 CFR 219.11(a)(1)(iii)).

The plan must identify the lands that are suited and not suited for timber production using the table shown in exhibit 01. Exhibit 01 should be used in the plan to display the lands that, following step 1, *may* be suited for timber production, and the lands that, following step 2, have been determined to be suited and not suited for timber production.

Details such as the criteria and methods used, and lands identified for each category, should be kept in the planning record and summarized in an appendix to the plan EIS or appropriate environmental document. Information developed in the planning process on lands suited and not suited for timber production should be developed with sufficient breakdowns and detail in the planning record to be compatible with the national land suitability classification system for timber production (see FSM 2490 – Timber Management Information System). Exhibit 01 should be used in the plan to display the suitability of lands for timber production.

61 - Exhibit 01

Timber Production Suitability Classification

Land Classification Category	Acres
A. Total National Forest System lands in the plan area	1,000,000
B. Lands not suited for timber production due to legal or technical reasons	350,000
C. Lands that ***may* be suited** for timber production (A–B)	650,000
D. Total lands **suited** for timber production because timber production is compatible with the desired conditions and objectives established by the plan	400,000
E. Lands not suited for timber production because timber production is not compatible with the desired conditions and objectives established by the plan (C – D)	250,000
F. Total lands **not suited** for timber production (B+E)	600,000

61.1 – Lands Not Suitable for Timber Production Based on Legal and Technical Factors (Step 1)

Sections 61.11 – 61.14 of this Handbook provide guidance on how to apply the step 1 factors in the process of determining lands not suitable for timber production based on legal and technical factors (36 CFR 219.11(a)(i), (ii), (iv), (v) and (vi)).

61.11 – Lands on which Timber Production is Prohibited or Lands Withdrawn from Timber Production

Timber production may be prohibited on certain lands by statute, Executive order, regulation, or where the Secretary of Agriculture or the Chief of the Forest Service has withdrawn the land from timber production (36 CFR 219.11(a)(1)(i) and (ii), and FSM 1921.12). Examples include units of the National Wilderness Preservation System, designated wild river segments, research natural areas, and other designated areas (see 36 CFR 219.19 and FSH 1909.12, ch. 20) where timber production is specifically prohibited. Identify these lands as lands not suited for timber production.

61.12 – Lands on which Technology to Harvest Timber is Not Currently Available without Causing Irreversible Damage

The Responsible Official should develop criteria to identify lands within the plan area that are not suited for timber production, because technology to harvest timber without causing irreversible damage is not currently available. The criteria for identifying such lands should take into account information such as landforms, soil conditions, vegetation, and available technology for timber harvest in the plan area.

Apply the criteria to determine where it is not possible to carry out timber harvest activities without irreversible resource damage to soil, slope, or other watershed conditions in the plan area. Relevant information such as soil maps, geological maps, landslide maps, and remote sensing products, as well as information from the terrestrial ecological unit inventory or the soil resource inventory (FSM 2550), may be used to assess soil vulnerability to physical, chemical, and biological damage.

Identify these lands as not suited for timber production.

61.13 – Lands on Which There is No Reasonable Assurance that Lands Can be Adequately Restocked within 5 Years of Final Regeneration Harvest

The Responsible Official should identify criteria for what constitutes adequate restocking after final regeneration harvests for purposes of timber production (see the definition of stocking for possible indices to describe stocking). Such criteria could vary for the different forest types and

management systems in the plan area. Specific land types, soil types, and vegetative conditions should be evaluated for appropriate management systems to assess if reasonable assurance (see definition) exists that the lands can be regenerated to achieve adequate restocking 5 years after the final regeneration harvest of the appropriate management system.

Relevant information such as most recent soil maps, geological maps, terrestrial ecological unit inventory, monitoring results, and other best available scientific information may be used to identify lands within the planning area where there is no reasonable assurance that the land can be adequately restocked within 5 years after final regeneration harvest. Lands that do not have this reasonable assurance are not suited for timber production.

This is a technical determination that is not based on desired conditions and other plan components, but solely on the ability of the land to sustain timber production. This criteria is to be applied solely to determine suitability for timber production and is not to be confused with the criteria described in section 64.14 of this Handbook regarding assurance of restocking after harvest. Exhibit 01 shows an example of such criteria.

61.13 - Exhibit 01

Adequate Restocking Criteria for Timber Production in Northern Arizona

Trees per acre 5 years after final regeneration harvest

Management System/Final Regeneration Harvest	**Forest Type**	**Adequately Restocked**
Even aged/ Final shelterwood removal	Ponderosa Pine	>75 seedlings per acre
Even aged/ Final shelterwood removal	Mixed Conifer	>100 seedlings per acre
Uneven aged/ last group selection entry	Ponderosa Pine	>10 trees above 10 in. d.b.h. ≥50 seedlings
Uneven aged/ last group selection entry	Mixed Conifer	>15 trees above 10 in. d.b.h. ≥75 seedlings

Lands that do not have a reasonable assurance of achieving such adequate restocking within 5 years of final regeneration harvest are identified as not suitable for timber production.

61.14 – Land that Is Not Forest Land (Nonforest)

Nonforest lands are lands that do not meet the definition of forest lands contained in 36 CFR 219.19. Thus, nonforest lands are less than 10 percent occupied by forest trees of any size or that formerly had such tree cover and are currently developed for nonforest uses. Land developed for nonforest uses includes areas for agricultural crops, improved pasture, residential or administrative areas, improved roads of any width and adjoining road clearing, and powerline clearing of any width. Lands that were formerly occupied by tree cover, but do not presently have tree cover, should be identified as nonforest unless the land will be naturally or artificially regenerated into forest cover in the near future (example: clearcut lands).

Canopy cover of live forest trees at maturity occupying an area may be used to estimate if an area is at least 10 percent occupied by forest trees (FSM 1905).

For mapping purposes or calculating land area, unimproved roads, trails, intermittent or small perennial streams, and clearings in forest areas may be included as forestland if they are less than 120 feet in width. Calculations of forestland for determining timber volumes (sec. 64 of this Handbook) should adjust for the cumulative amount of such small inclusions of nonforest lands. Identify nonforest lands as not suitable for timber production.

61.2 – Lands Suited and Not Suited for Timber Production Based on Compatibility with Desired Conditions and Objectives (Step 2)

The second step is to determine for each alternative in the plan EIS which of the lands that *may* be suitable for timber production (identified in step 1) are suited for timber production based on compatibility with desired conditions and objectives. In making this determination, the Responsible Official should consider the following to determine if timber production is compatible with the desired conditions and objectives of the plan:

1. Timber production is a desired primary or secondary use of the land.

2. Timber production is anticipated to continue after desired conditions have been achieved.

3. A flow of timber can be planned and scheduled on a reasonably predictable basis.

4. Regeneration of the stand is intended.

5. Timber production is compatible with the desired conditions or objectives for the land designed to fulfill the requirements of 36 CFR 219.8 to 219.10.

Note that areas recommended for wilderness, wilderness study areas (see FSH 1909.12 ch. 20 sec. 24.41) and eligible or suitable wild river segments (see FSH 1909.12 ch. 20 sec 24.42 are not suitable for timber production to maintain the option for future designation.

The remaining lands after subtracting the lands that are suited from the lands that *may* be suited; are not suited for timber production, because timber production is not compatible with the land area's desired conditions and objectives (36 CFR 219.11(a)(1)(iii)). This is done in the EIS for each alternative considered in plan development or plan revision.

61.3 – Review of Lands Not Suited for Timber Production

> **The responsible official shall review lands identified in the plan as not suited for timber production at least once every 10 years or as otherwise prescribed by law, to determine whether conditions have changed so that they have become suitable for timber production. As a result of this 10-year review, the plan may be amended to identify such lands as suitable for timber production if warranted by changed conditions. (36 CFR 219.11(a)(2)).**

The Responsible Official should determine when and how to conduct this review. If the results of the review would lead to a change in the amount or location of lands suitable for timber production, the Responsible Official will determine the appropriate mechanism to change the plan.

62 – PLAN COMPONENTS FOR LANDS SUITABLE FOR TIMBER PRODUCTION

> **A plan that identifies lands as suitable for timber production must include plan components, including standards or guidelines, to guide timber harvest for timber production or other multiple use purposes on such lands.** (36 CFR 219.11(b)).

Plan components to meet the requirements of 36 CFR 219.8 -219.10 or to respond to public and governmental interest in the planning process directly or indirectly guide timber harvest on lands suitable for timber production. Plan components, including standards or guidelines that guide timber harvest activity, must at a minimum include the statutory and regulatory limitations on timber harvest as described in section 64 of this Handbook (16 USC 1604(g)(3)(E) and (F), 16 USC 1604(m), and 16 USC 1611; 36 CFR 219.11(d)).

On lands suitable for timber production, plan components may be designed to apply to all purposes for timber harvest, including harvest for timber production or harvest to protect multiple use values; or, plan components may be designed to apply separately as appropriate to each purpose. Plan components that apply to harvest to protect multiple use values may apply to lands not suitable for timber production as well as lands suitable for timber production, where appropriate (see sec. 63 of this Handbook).

63 – PLAN COMPONENTS FOR TIMBER HARVEST FOR PURPOSES OTHER THAN TIMBER PRODUCTION

Where timber harvest will be used as a tool for purposes other than timber production in order to protect other multiple use values, plans must provide appropriate plan components that allow and control the application of such timber harvest. Such plan components must identify the lands to which they apply. These lands may be identified by forest types, management or geographic areas, lands suited or not suited for timber production, or other criteria. On lands not suited for timber production, plan components may only allow timber harvest to occur to protect multiple use values other than timber production and for salvage, sanitation, public health, or safety, as provided by the rule at 36 CFR 219.11(c):

> **(c) *Timber harvest for purposes other than timber production.* Except as provided in paragraph (d) of this section, the plan may include plan components to allow for timber harvest for purposes other than timber production throughout the plan areas, or portions of the plan areas, as a tool to assist in achieving or maintaining one or more applicable desired conditions or objectives of the plan in order to protect other multiple-use values, and for salvage, sanitation, or public health or safety. Examples of using timber harvest to protect other multiple use values may include improving wildlife or fish habitat, thinning to reduce fire risk, or restoring meadow or savanna ecosystems where trees have invaded.** (36 CFR 219.11).

The plan must have appropriate plan components that establish permissible reasons for timber harvest for purposes other than timber production in order to protect other multiple use values on lands not suited for timber production. Desired conditions could describe conditions that are achievable through timber harvest. Objectives, identifying the desired rate of progress in achieving desired conditions, could do so in terms of planned outcomes in the plan period accomplished through timber harvest. Standards and guidelines could clarify what timber harvest methods can be used on certain lands and ensure that these activities are consistent with achieving the desired conditions of the plan and other requirements of the Planning Rule.

64 – LIMITATIONS ON TIMBER HARVEST

The National Forest Management Act (NFMA) requires the Planning Rule to include guidelines for land management plans to ensure that timber will be harvested from National Forest System lands only where certain limitations are met (16 U.S.C. 1604(g)(3)(E)(i-iv)) and to insure that evenaged cuts are only used on National Forest System lands consistent with other limitations (16 U.S.C. 1604(g)(3)(F)(i-v)). In addition, the NFMA at 16 U.S.C. 1604(m) requires that the Secretary establish standards to insure that prior to harvest, stands must generally have reached

the culmination of mean annual increment; and requires at U.S.C. 1611 that the Secretary of Agriculture shall limit the sale of timber from each national forest to a quantity equal to or less than a quantity that can be removed from such forest annually in perpetuity on a sustained-yield basis.

The Planning Rule at §219.11(d) requires:

> ***Limitations on timber harvest.*** **Whether timber harvest would be for the purposes of timber production or other purposes, plan components including standards and guidelines, must ensure the following:**

Paragraphs 1 through 7 within §219.11(d) identifies each limitation that must have appropriate plan components. Each limitation is covered in §219.11(d)(1)-(7) either by direct description in a specific paragraph or through reference to NFMA limitations set out in 219.11(d)(5). Exhibit 01 summarizes each limitation and indicates where in this chapter appropriate guidance is found on how plans can meet the limitation. The table is organized in three sections: limitations applicable to all timber harvest, limitations applicable to only even-aged timber harvest, and the limitation on volume that can be sold. Exhibit 01 also references the sections of U.S. Code that contain the limitations identified in the National Forest Management Act and the sections of the Planning Rule that contain or reference these limitations. Section 64 of this Handbook covers each of these limitations in the order identified in exhibit 01. The applicable text of both the NFMA and the Planning Rule are included in each section. If a National Forest Management Act requirement and a rule requirement overlap, both are described in the same section.

64 - Exhibit 01

Reference Table for Required Plan Components That Limit Timber Harvest

Chapter 60 Section	Requirement Summary	Title 16, U.S. Code Section	Subsection of 219.11
Requirements for all timber harvest			
64.11	No harvest for purposes of timber production on lands not suited for timber production.	1604 (k)	(d)(1)
64.12	Timber harvest would occur only where soil, slope, or watershed conditions would not be irreversibly damaged.	1604 (g)(3)(E)(i)	(d)(2)
64.13	Timber will be harvested only where protection is provided for streams, streambanks, shorelines, lakes, wetlands, and other bodies of water.	1604 (g)(3)(E)(iii)	(d)(5)
64.13	Timber harvest would be carried out consistent with the protection of soil, watershed, fish, wildlife, recreation, and aesthetic resources.		(d)(3)
64.14	Timber harvest will be harvested only where there is assurance that such lands can be adequately restocked within 5 years after harvest.	1604 (g)(3)(E)(ii)	(d)(5)
64.15	Timber will be harvested only where the harvesting system is not selected primarily because it will give the greatest dollar return or unit output of timber.	1604 (g)(3)(E)(iv)	(d)(5)
Requirements for only even-aged timber harvest			
64.21, 64.21a-c	Limits to the maximum size for openings that may be cut in one harvest operation and exceptions.	1604 (g)(3)(F)(iv)	(d)(4)
64.22	Clearcutting will be used only where determined to be the optimum method.	1604 (g)(3)(F)(i)	(d)(5)
64.22	Other cuts to regenerate an even-aged stand of timber will be used only where determined to be appropriate.	1604 (g)(3)(F)(i)	(d)(5)
64.23	Even-aged regeneration cutting will be used only where the interdisciplinary review has been completed.	1604 (g)(3)(F)(ii)	(d)(5)
64.24	Even-aged regeneration cutting will be used where cuts are shaped and blended with the natural terrain.	1604 (g)(3)(F)(iii)	(d)(5)
64.25	Timber harvest would be carried out consistent with the protection of soil, watershed, fish, wildlife, recreation, and aesthetic resources.	1604 (g)(3)(F)(v)	(d)(3)
64.26	Even-aged stands shall generally have reached culmination of mean annual increment to regeneration harvest and exceptions.	1604 (m)	(d)(7)
Limitation on volume that can be sold			
64.3, 64.31-34	The sale of timber from each national forest shall be limited to a quantity equal to or less than a quantity which can be removed from such forest annually in perpetuity on a sustained-yield basis and exceptions (departures, decade, salvage and sanitation harvests).	1611 (a) and (b)	(d)(6)

Sections 64.1 to 64.3 of this Handbook describe the guidance for the limitations required by 36 CFR 219.11(d)(1) through (d)(7). These limitations require plan components, including standards or guidelines. Standards or guidelines are required to provide for each limitation. In addition, other plan components may be used. Most of these sections reflect the expectation that standards will be used as the required plan component to meet the limitation, although Responsible Officials may use guidelines where appropriate. Thus the word "should" is often used consistent with FSH 1909.12, zero code, section 05.1, exhibit 01 to describe the expectation that standards will be used, but to allow Responsible Officials to use guidelines if there are justifiable reasons to do so. See the discussion on integration of plan components in FSH 1909.12, chapter 20, section 23.

In addition to meeting the requirements discussed in this section, the plan may include standards or guidelines that preclude or limit certain harvest methods that are not compatible with the desired conditions for a particular situation or area.

64.1 – Limitations Applicable to All Timber Harvest

No Timber Harvest for the Purpose of Timber Production May Occur on Lands Not Suited for Timber Production

NFMA directs that:

> **… the Secretary shall assure that except for salvage sales or sales necessitated to protect other multiple-use values, no timber harvesting shall occur on such lands [lands not suited for timber production] for a period of 10 years.** (U.S.C. 1604(k)).

The Planning Rule requires plan components, including standards or guidelines, to ensure that:

> **. . . (1) No timber harvest for the purpose of timber production may occur on lands not suited for timber production.** (36 CFR 219.11(d)).

As described in section 64 of this Handbook, plans should have standards that no timber harvest for the purpose of timber production may occur on lands not suited for timber production. This requirement does not preclude plans from having components that allow timber harvest on lands not suited for timber production to protect other multiple-use values, and for salvage, sanitation, public health, or safety (see 36 CFR 219.11(c) and sec. 63 of this Handbook).

64.12 – Timber Harvest May Not Occur if It Leads to Irreversible Damage

NFMA directs that plans:

> **(E) ensure that timber will be harvested from National Forest System lands only where—**
>
> **(i) soil, slope, or other watershed conditions will not be irreversibly damaged . . .** (16 U.S.C. 1604(g)(3)).

The Planning Rule requires plan components, including standards or guidelines, to ensure that:

> **. . . (2) Timber harvest would occur only where soil, slope, or other watershed conditions would not be irreversibly damaged;** (36 CFR 219.11(d)).

Plans must have standards to ensure that a project decision to harvest timber may only be made when it is determined the project would not cause irreversible damage to soil, slope, or watershed condition. The standards should require a site-specific finding that the timber harvest would not cause irreversible damage. A standard stating no timber harvest could occur on lands where technology to harvest timber is not currently available without causing irreversible damage (see sec. 61.12 of this Handbook) would remove lands with a risk of irreversible damage from potential harvest. For all other lands, Responsible Officials would have to consider additional, site-specific factors associated with any potential harvest to ensure meeting this limitation.

64.13 – Timber Harvest Must Be Consistent with Other Resource Protection

NFMA directs that plans:

> **(E) insure that timber will be harvested from National Forest System lands only where—**
>
> ***
>
> **(iii) protection is provided for streams, streambanks, shorelines, lakes, wetlands, and other bodies of water from detrimental changes in water temperatures, blockages of water courses, and deposits of sediment, where harvests are likely to seriously and adversely affect water conditions or fish habitat; . . .** (16 U.S.C. 1604(g)(3)).

The Planning Rule requires plan components, including standards or guidelines, to meet this limitation at § 219.11(d)(5) and to also ensure that:

> **. . . (3) Timber harvest would be carried out in a manner consistent with the protection of soil, watershed, fish, wildlife, recreation, and aesthetic resources. (36 CFR 219.11(d)).**

Plans must have plan components including standards and guidelines to address other aspects of the Planning Rule designed for the protection of streams and other water bodies, soil, watershed, fish, wildlife, recreation, and aesthetic resources. Such plan components may be sufficient to insure these resources are protected from timber harvest. The Responsible Official should review the plan components to determine if a timber harvest consistent with those plan components would also provide for appropriate protection of these resources. If not, additional plan components must be developed to ensure that all harvests meet this limitation.

64.14 – Assurance of Adequate Restocking within 5 Years after Harvest

NFMA directs that plans:

> **(E) insure that timber will be harvested from National Forest System lands only where . . .**
> ***
> **(ii) there is assurance that such lands can be adequately restocked within five years after harvest . . . (16 U.S.C. 1604(g)(3)).**

The Planning Rule requires plan components, including standards or guidelines, to meet this limitation at §219.11(d)(5).

As described in section 64 of this Handbook, plans should have standards that limit timber harvest to situations that have reasonable assurance (see definition) that the stand can be adequately restocked within 5 years after harvest. Such standards should be based on the different types of forests within the plan area, the harvest methods applied and the desired conditions and objectives of the plan. These standards can be organized either by having the plan determine what is adequate restocking in different harvest circumstances, or the plan can require that a determination of what is adequate restocking can be made at the project level. The Responsible Official may choose to use one of these two basic approaches or a third approach that combines the first two approaches.

1. In the first approach, the plan contains standards that limit timber harvest to situations with reasonable assurance (see definition) that the stand can be adequately restocked within 5 years of harvest. The standard also identifies what would constitute adequate restocking for specific harvest situations. The desired conditions of the plan may also identify "desired stocking conditions" for these same specific harvest situations.

   This approach is displayed in an example in exhibit 01.

   Plan documentation should support the determination that there is reasonable assurance (see definition) that the identified lands and harvest methods can be adequately restocked for the situations described. The determination of reasonable assurance should be based on best available scientific information.

   Timber harvest projects for these specific harvest situations can use the documentation for the plan to demonstrate reasonable assurance of adequate restocking. This approach should also require a finding that the land proposed for harvest can be adequately restocked based on the documentation for the land management plan.

64.14 - Exhibit 01

Example Table Showing Desired and Adequate Stocking Conditions

Stocking objectives (trees per acre) for ponderosa pine stands in Northern Arizona 5 years after harvest

Harvest method	Desired Stocking	Adequately Restocked
Shelterwood regeneration with overstory retention	5-10 trees above 10 in. d.b.h. 75-125 seedlings	>5 trees above 10 in. d.b.h. ≥75 seedlings
Intermediate thinning	10-20 trees above 10 in. d.b.h. 10-20 trees above 5 in. d.b.h. 25-60 seedlings	>10 trees above 10 in. d.b.h. >10 trees above 5 in. d.b.h. ≥25 seedlings
Salvage harvest following fire destroying canopy	5-10 trees above 10 in. d.b.h. 50-100 seedlings	>5 trees above 10 in. d.b.h. (if available) ≥50 seedlings
Harvest to create permanent meadow **	<10 trees of any size	No trees

** Any timber harvest that would not restock after harvest to create nonforest conditions must be based on desired conditions or objectives of the plan and be consistent with other plan components. After harvest, affected land should be classified as not suitable for timber production (sec. 61.13 of this Handbook) as it would no longer be forest land. This change can be made as part of the review of lands not suited for timber production (sec. 61.3 of this Handbook).

2. In the second approach, the plan also contains standards that limit timber harvest to situations with reasonable assurance (see definition) that the stand can be adequately restocked within 5 years of harvest. However, these standards require individual timber harvest projects to state findings *and* provide site-specific documentation that supports the reasonable assurance determination that lands can be adequately restocked within 5 years. The determination of restocking expectations is based on plan desired conditions and objectives applicable to the area and project, and consistent with all other applicable plan components.

This second approach may be appropriate where restocking criteria would vary across most of the plan area. It is also an option for the following types of timber harvests where removal of forest cover may be consistent with the desired conditions and objectives of the plan such as:

a. Harvest to restore habitat for a species requiring very open conditions.

b. Harvest to restore a previously impacted site, such as a timber plantation.

c. Harvest to eradicate an invasive tree that has dominated the site.

d. Harvest to restore an open cultural landscape associated with a historic property.

e. Harvest to create a parking lot for a trailhead.

3. In a third, mixed approach, the plan contains standards that limit timber harvest to situations that have reasonable assurance (see definition) that the stand can be adequately restocked within 5 years of harvest. These standards also identify of what would constitute adequate restocking as in the first approach. However, the standards explicitly allow for other situations or exceptions supported by a project-specific determination of adequate restocking. The Responsible Official in determining a finding of adequate restocking can rely on documentation in the plan record for the situations described in the plan, but must provide project-specific documentation for situations not included in the plan. This finding must be based on desired conditions and objectives and be consistent with all other applicable plan components.

64.15 - Selection of Harvesting System

NFMA directs that plans:

> **(E) insure that timber will be harvested from National Forest System lands only where . . .**
>
> **(iv) the harvesting system to be used is not selected primarily because it will give the greatest dollar return or the greatest unit output of timber . . .** (16 U.S.C. 1604(g)(3)).

The Planning Rule requires plan components, including standards or guidelines, to meet this limitation at §219.11(d)(5) .

As described in section 64 of this Handbook, plans should include a standard indicating that the harvesting system for a project must not be selected primarily for the greatest dollar return or output of timber.

64.2 – Limitations for Even-aged Harvest

The National Forest Management Act establishes limitations applicable to harvests designed to regenerate an even-aged stand of timber (16 U.S.C. 1604(g)(3)(F)):

> **(F) insure that clearcutting, seedtree cutting, shelterwood cutting, and other cuts designed to regenerate an even-aged stand of timber will be used as a cutting method on National Forest System lands only where—(limitations follow in 16 U.S.C. 1604(g)(3)(F)(i-v)).**

The balance of this section (sec. 64.21–64.26 of this Handbook) describes how plans should provide plan components for each of these five limitations.

64.21 – Limits on Maximum Size of Created Openings

The National Forest Management Act limits clearcutting and other even-aged harvest to situations where:

> **(iv) there are established according to geographic areas, forest types, or other suitable classifications the maximum size limits for areas to be cut in one harvest operation, including provision to exceed the established limits after appropriate public notice and review by the responsible Forest Service officer one level above the Forest Service officer who normally would approve the harvest proposal: Provided, That such limits shall not apply to the size of areas harvested as a result of natural catastrophic conditions such as fire, insect and disease attack, or windstorm . . .** (16 U.S.C. 1604(g)(3)(F)).

The regulation establishes maximum sized openings for types of tree species in specific geographic areas, and establishes a maximum size opening for all other tree species regardless of location. To meet this limitation, plans must have standards that establish size openings no larger than that allowed by the regulation except as described in the Planning Rule at §219.11(d)(4)(i)-(iii) and explained in sections 64.21a through 64.21c of this Handbook.

> **. . . (4) Where plan components will allow clearcutting, seedtree cutting, shelterwood cutting or other cuts designed to regenerate an even-aged stand of timber, the plan must include standards limiting the maximum size for openings that may be cut in one harvest operation according to geographic areas, forest types or other suitable classifications. Except as provided in paragraphs (d)(4)(i) through (iii) of this section, this limit may not exceed 60 acres for the Douglas-fir forest type of California, Oregon, and Washington; 80 acres for the southern yellow pine types of Alabama, Arkansas, Georgia, Florida, Louisiana, Mississippi, North Carolina, South Carolina, Oklahoma, and Texas; 100 acres for the hemlock-Sitka spruce forest type of coastal Alaska; and 40 acres for all other forest types. (36 CFR 219.11(d)(4)).**

64.21a – Standards for Exceptions to Exceed Opening Size Limits

> **(i) Plan standards may allow for openings larger than those specified in paragraph (d)(4) of this section to be cut in one harvest operation where the Responsible Official determines that larger harvest**

> **openings are necessary to help achieve desired ecological conditions in the plan area. If so, standards for exceptions shall include the particular conditions under which the larger size is permitted and must set a maximum size permitted under those conditions.** (36 CFR 219.11(d)(4)(i)).

The Planning Rule also provides that plans may specifically allow for larger openings than those established in the planning regulation. If a plan makes such exceptions, the standard limiting the size of created openings must clearly describe the particular conditions for the exception including the desired ecological conditions that the exception intends to achieve. Such an exception should be described in the standards that limit the size of openings described in section 64.21 of this Handbook

64.21b – Notification and Review to Exceed Opening Size Limits on an Individual Timber Sale Basis

> **(ii) Plan components may allow for size limits exceeding those established in paragraphs (d)(4) and (d)(4)(i) of this section on an individual timber sale basis after 60 days public notice and review by the regional forester.** (36 CFR 219.11(d)(4)).

The standards for opening size limits in section 64.21 of this Handbook and the exceptions identified in section 64.21a of this Handbook may also provide exceptions that would allow Responsible Officials to propose individual timber sales that exceed these opening size standards, where doing so would be consistent with other plan components. Such an exception must also provide that such a project may be authorized only after the Responsible Official provides 60 days' notice to the public and review by the Regional Forester.

64.21c – Catastrophic Conditions that Allow Exceeding Opening Size Limits

Plan standards that limit the size of created openings (sec. 64.21 of this Handbook) in one harvest operation do not apply to openings made by harvesting as a result of the catastrophic conditions such as fire, insect and disease attack, or windstorm (36 CFR 219.11(d)(4)).

> **(iii) The plan maximum size for openings to be cut in one harvest operation shall not apply to the size of openings harvested as a result of natural catastrophic conditions such as fire, insect and disease attack, or windstorm (16 U.S.C. 1604(g)(3)(F)(iv)).**
> (36 CFR 219.11(d)(4)).

Projects that exceed opening sizes because of natural catastrophic conditions must still be consistent with other plan components.

64.22 – Clearcutting and Other Even-aged Cutting Methods

The National Forest Management Act limits clearcutting and other even-aged harvest to situations where:

> **(i) for clearcutting, it is determined to be the optimum method, and for other such cuts it is determined to be appropriate, to meet the objectives and requirements of the relevant land management plan . . . (16 U.S.C. 1604(g)(3)(F)).**

The Planning Rule requires plan components, including standards or guidelines, to meet this limitation at §219.11(d)(5).

1. To meet the requirement that plan components may provide for clearcutting only when it is the optimum method, one of the following approaches may be used:

   a. The plan has standards that limit clearcutting to specific types of situations identified in the plan where clearcutting is the optimum method of harvest. The determination of optimality is based on the desired conditions or objectives for the plan area or relevant portion(s) thereof. The planning record must provide documentation to support this determination of clearcutting as the optimum method. The plan standards should also require a finding at the project level that clearcutting is the optimal method based on the documentation for the land management plan.

   b. The plan has a standard requiring, for each clearcutting project, that the Responsible Official make a finding that clearcutting is the optimum method for the project and document the rationale for that finding in the project record. The finding and documentation that clearcutting is the optimum method for the project must be based on site specific conditions and the desired conditions or objectives of the plan applicable to area proposed for clearcutting.

   c. The plan has a standard that combines approaches (a) and (b), limiting clearcutting to specific types of situations identified in the plan, but also allowing clearcutting in additional instances, upon a site-specific finding that clearcutting is the optimum method. The Responsible Official in determining a finding of optimality can rely on documentation in the plan record for the situations described in the plan, but must provide project specific documentation for situations not included in the plan.

2. To meet the requirement that plan components may provide for shelterwood, seedtree, and other types of even-aged cuts only where they are appropriate, one of the following approaches can be used:

a. The plan has a standard limiting certain types of even-aged harvests to the types of situations where the plan identifies such harvests are appropriate based on the desired conditions or objectives for the plan area or relevant portion(s) thereof. The plan may identify the specific even-age harvest method(s) appropriate to each type of situation, or may find that all even-age harvest methods (other than clearcutting) are appropriate. The planning record must provide documentation to support this determination that even-aged cuts are appropriate. The standard should also require a finding at the project level that the even-aged harvest is appropriate based on the documentation for the land management plan.

b. The plan has a standard requiring, for each even-aged harvest project, that the Responsible Official make a finding that such harvest is appropriate for the project, and document the rationale for that finding in the project record. The finding that the even-aged cuts for the project is appropriate must be based on site-specific conditions and the desired conditions or objectives of the plan applicable to area proposed for even-aged harvest.

c. The plan has a standard that combines approaches (a) and (b), limiting other types of even-aged cuts to specific types of situations identified in the plan, but also allowing for even-aged harvests in additional instances, upon a finding that the even-aged cut is appropriate. The Responsible Official in determining a finding of appropriateness can rely on documentation in the plan record for the situations described in the plan, but must provide project-specific documentation for situations not included in the plan.

64.23 – Interdisciplinary Review

The National Forest Management Act limits clearcutting and other even-aged harvest to situations where:

> **(ii) the interdisciplinary review as determined by the Secretary has been completed and the potential environmental, biological, esthetic, engineering, and economic impacts on each advertised sale area have been assessed, as well as the consistency of the sale with the multiple use of the general area . . .** (16 U.S.C. 1604(g)(3)(F)).

The Planning Rule requires plan components, including standards or guidelines, to meet this limitation at §219.11(d)(5).

As described in section 64 of this Handbook, plans should have standards requiring interdisciplinary review, assessments of the project's impacts through appropriate environmental documentation and a finding that the project is consistent with the multiple uses of the general area for any even-aged regeneration harvests.

64.24 – Cuts Shaped and Blended with Natural Terrain

The National Forest Management Act limits clearcutting and other even-aged harvest to situations where:

> **(iii) cut blocks, patches, or strips are shaped and blended to the extent practicable with the natural terrain . . .** (16 U.S.C. 1604(g)(3)(F)).

The Planning Rule requires plan components, including standards or guidelines, to meet this limitation in §219.11(d)(5).

Plan components related to scenic character usually provide for blending harvest units into the natural terrain (see FSH 1909.12, ch. 20, sec. 23.23(f)) and will likely suffice to ensure compliance with this timber harvest requirement. The Responsible Official should review plan components developed to provide for scenery to see if consistency with those plan components ensures that this requirement is also met. If not, additional plan components must be developed to ensure meeting this requirement.

64.25 – Consistency with Resource Protections

The National Forest Management Act limits clearcutting and other even-aged harvest to situations where:

> **(v) such cuts are carried out in a manner consistent with the protection of soil, watershed, fish, wildlife, recreation, and esthetic resources, and the regeneration of the timber resource.** (16 U.S.C. 1604(g)(3)(F)).

The Planning Rule requires plan components, including standards or guidelines, to meet these limitations at §219.11(d)(5). In addition, at 219.11 (d)(3), the rule applies these limitations (except the regeneration of timber) to all timber harvests.

Section 64.13 of this Handbook describes how plan components should be provided to meet the resource protections described in this section. Section 64.14 of this Handbook describes how plan components can be provided to provide for the regeneration of the timber resource.

64.26 – Culmination of Mean Annual Increment of Growth

The National Forest Management Act sets out a requirement to ensure that timber harvest must occur at the culmination of the mean annual increment of growth, but also provides for exceptions at 16 USC 1604(m):

> **(m) Establishment of standards to ensure culmination of mean annual increment of growth; silvicultural practices; salvage harvesting; exceptions.**

> **The Secretary shall establish —**
>
> **(1) standards to insure that, prior to harvest, stands of trees throughout the National Forest System shall generally have reached the culmination of mean annual increment of growth (calculated on the basis of cubic measurement or other methods of calculation at the discretion of the Secretary): *Provided*, That these standards shall not preclude the use of sound silvicultural practices, such as thinning or other stand improvement measures: *Provided further*, That these standards shall not preclude the Secretary from salvage or sanitation harvesting of timber stands which are substantially damaged by fire, windthrow or other catastrophe, or which are in imminent danger from insect or disease attack; and**
>
> **(2) exceptions to these standards for the harvest of particular species of trees in management units after consideration has been given to the multiple uses of the forest including, but not limited to, recreation, wildlife habitat, and range and after completion of public participation processes utilizing the procedures of subsection (d) of this section.**

The Planning Rule requires plan components, including standards or guidelines, to meet this limitation as described at §219.11 (d)(7).

> **(7) The regeneration harvest of even-aged stands of trees is limited to stands that generally have reached the culmination of mean annual increment (CMAI) of growth. This requirement would apply only to regeneration harvest of even-aged stands on lands identified as suited for timber production and where timber production is the primary purpose for the harvest. Plan components may allow for exceptions, set out in 16 U.S.C 1604(m), only if such harvest is consistent with the other plan components of the land management plan.**

Plans for national forests that have lands suitable for timber production must include plan components (typically standards and guidelines) to address these requirements.

A stand that "generally [has] reached the culmination of mean annual increment (CMAI) of growth" is the age at which the stand achieves at least 95 percent of the cubic foot volume at culmination.

1. When determining the culmination of the mean annual increment of growth, the Responsible Official shall:

   a. Calculate the culmination of the mean annual increment of growth in terms of cubic feet .

b. Base the determination of the culmination of mean annual increment of growth on the expected cumulative timber yield from regeneration harvest and additional timber yields from any planned intermediate harvests prior to the regeneration harvest. The minimum rotation age is the shortest length of time required to achieve volume production equivalent to at least 95 percent of the culmination of mean annual increment. To determine culmination of mean annual increment for two-age stands, the calculation should be based only on the age cohort that would be scheduled for regeneration harvest. See also the definition of "Mean annual increment" in section 60.5 of this Handbook.

2. Plan components that limit regeneration harvest to stands that have reached 95 percent of culmination of mean annual increment must clarify that these limitations do not apply to:

a. Thinning or other stand improvement treatments and uneven-aged systems that do not regenerate even-aged stands.

b. Salvage or sanitation harvesting of timber stands which are substantially damaged by fire, wind throw, or other catastrophe, or which are in imminent danger from insect or disease attack.

c. Harvesting of trees on lands not suited for timber production because the type and frequency of harvests are driven by the need to protect multiple use values other than timber production.

3. A plan may provide for exceptions to the culmination of mean annual increment requirement for expected situations where even-aged regeneration harvest at less than culmination of mean annual increment would contribute to the plan's desired conditions or objectives consistent with other plan components. For example, it may be appropriate for a plan to establish shorter even-aged rotations to maintain levels of early seral stages sufficient for wildlife that depend upon such habitat conditions.

64.3 – Limiting the Quantity of Timber that Can Be Removed

The National Forest Management Act requires the Forest Service limit the amount of timber that may be sold from each national forest (16 U.S.C. 1611).

(a) Limitations on removal; variations in allowable sale quantity; public participation.

The Secretary of Agriculture shall limit the sale of timber from each national forest to a quantity equal to or less than a quantity which can be removed from such forest annually in perpetuity on a sustained-yield basis: Provided, That, in order to meet overall multiple-use objectives, the Secretary may establish an allowable sale quantity for

any decade which departs from the projected long-term average sale quantity that would otherwise be established: Provided further, That any such planned departure must be consistent with the multiple-use management objectives of the land management plan. Plans for variations in the allowable sale quantity must be made with public participation as required by section 1604(d) of this title. In addition, within any decade, the Secretary may sell a quantity in excess of the annual allowable sale quantity established pursuant to this section in the case of any national forest so long as the average sale quantities of timber from such national forest over the decade covered by the plan do not exceed such quantity limitation. In those cases where a forest has less than two hundred thousand acres of commercial forest land, the Secretary may use two or more forests for purposes of determining the sustained yield.

(b) Salvage harvesting.

Nothing in subsection (a) of this section shall prohibit the Secretary from salvage or sanitation harvesting of timber stands which are substantially damaged by fire, windthrow, or other catastrophe, or which are in imminent danger from insect or disease attack. The Secretary may either substitute such timber for timber that would otherwise be sold under the plan or, if not feasible, sell such timber over and above the plan volume.

The Forest Service planning regulations at 36 CFR 219.11 (d) requires implementation of the statute as follows:

(d) Whether timber harvest would be for the purposes of timber production or other purposes, plan components, including standards or guidelines must ensure the following:

****

(6) The quantity of timber that may be sold from the national forest is limited to an amount equal to or less than that which can be removed from such forest annually in perpetuity on a sustained yield basis. This limit may be measured on a decadal basis.

(i) The plan may provide for departures from this limit as provided by the NFMA when departure would be consistent with the plan's desired conditions and objectives. Exceptions for departure from this limit on the quantity sold may be made only after a public review and comment period of at least 90 days.

(ii) This limit may be based upon increases in harvest levels based on intensified management practices, such as reforestation, thinning, and tree improvement if such practices justify increasing the harvests in accordance with the Multiple-Use Sustained-Yield Act of 1960. The plan must require that such harvest levels be decreased at the end of each planning period if such practices cannot be successfully implemented or funds are not received to permit such practices to continue substantially as planned.

(iii) The Chief must include in the Forest Service Directive System procedures for estimating the quantity of timber that can be removed annually in perpetuity on a sustained-yield basis, and exceptions, consistent with 16 U.S.C. 1611.

The plan for a national forest that intends to sell timber must identify the sustained yield limit (SYL). The sustained yield limit is the amount of timber that can be removed annually on a sustained yield basis. To meet the requirements that the volume sold cannot exceed the sustained yield limit, the plan must have either:

1. A standard that limits the quantity of timber that may be sold in a decade to less than or equal to 10 times the forest's annual sustained yield limit (see sec. 64.31 of this Handbook); or

2. A standard that limits the quantity of timber that may be sold in a decade to be less than or equal to 10 times the forest's departure limit. (See sec. 64.33 of this Handbook).

The sustained yield limit and the departure limit are calculated and displayed as an annual volume. However, in any given year, the Responsible Official may sell a quantity of timber in excess of the annual volume of the sustained yield limit or departure limit as long as the total quantity sold over a 10-year period does not exceed the applicable limit for the decade (U.S.C 1611(a)).

Neither the sustained yield limit nor the departure limit applies to the sale of volume from salvage or sanitation harvesting of timber stands substantially damaged by fire, windthrow, or other catastrophe, or that are in imminent danger from insect or disease attack (U.S.C 1611(b)).

Sections 64.31 and 64.32 of this Handbook describe how to identify the sustained yield limit, the projected wood sale quantity, the projected timber sale quantity and the amount of timber actually sold to ensure compliance with this standard. Each of these timber volumes must be measured in cubic feet. Section 64.35, exhibit 01 of this Handbook displays the different characteristics of these measures of timber volumes.

64.31 – Sustained Yield Limit

The Responsible Official shall identify the amount of timber that can be removed annually in perpetuity on a sustained-yield basis from the applicable national forest. This amount of timber is the forest's sustained yield limit (SYL). The sustained yield limit must be identified at the time of plan development and plan revision, and displayed as shown in section 65.2, exhibit 01 or exhibit 02 of this Handbook.

The Responsible Official shall determine of the sustained yield limit as the amount of timber that could be produced on all lands that *may be suitable* for timber production, assuming all of these lands were managed to produce timber without considering other multiple uses or fiscal or organizational capability. Assume the application of a management system (even-aged or uneven-aged) that is generally appropriate for the forest types and identify the potential flow of timber that could be reasonably planned and scheduled for these lands.

When determining the sustained yield limit:

1. Because the land that *may be suitable* for timber production does not vary by alternatives considered in the environmental impact statement for plan development or revision; the sustained yield limit calculation is a single constant for the applicable national forest. Because the sustained yield limit represents the potential of volume that could be harvested in perpetuity, it does not vary by decade or any other time period.

2. Volume from salvage and sanitation timber harvest is not included in calculating the sustained yield limit.

3. The sustained yield limit is measured in units of cubic feet that meet appropriate utilization standards (sec. 64.34 of this Handbook). As appropriate, the potential sustained yield may also be converted to board feet.

4. The analysis to determine the sustained yield limit should be done within reasonable cost and time commitments. The analysis can be based on standard growth and yield equations that estimate sustainable timber production from the lands that may be suitable for timber production. Calculations of the sustained yield limit should include all potential outputs of timber that would meet utilization standards (sec. 64.34 of this Handbook) for products sold from National Forest System lands. Regression equations and growth and yield simulation models such as Forest Vegetation Simulator (FVS) are acceptable tools for determining the sustained yield limit. Data used to develop sustained yield limit may include volume, basal area, number of trees, and average diameter at breast height (d.b.h.) by age class or successional stage. The Responsible Official shall determine the appropriate method to use for determining the sustained yield limit.

5. The calculation of the sustained yield limit may be based on application of intensified management practices, such as reforestation, thinning, and tree improvement.

64.32 – Projected Wood Sale Quantity, Projected Timber Sale Quantity, and Quantity of Timber Sold

The projected wood sale quantity (PWSQ), is an estimate of the volume of all timber and other wood products that is expected to be sold during the plan period from expected harvests for any purpose (except salvage harvest or sanitation harvest) on all lands in the plan area. The projected wood sale quantity includes all woody material likely to be sold from these harvests whether or not the woody material meets the utilization standards (sec. 64.34 of this Handbook).

The projected timber sale quantity is a subset of the projected wood sale quantity and is an estimate of the quantity of timber expected to be sold during the plan period. The volume in the projected timber sale quantity is the volume that meets utilization standards (sec. 64.34 of this Handbook). Except as provided in section 64.33 of this Handbook (departure from sustained yield limit), the projected timber sale quantity must be equal to or below the sustained yield limit for each decade of the plan.

The estimation of both the projected wood sale quantity and the projected timber sale quantity must take into account the fiscal capability of the planning unit and be consistent with all plan components. Both the projected wood sale quantity and the projected timber sale quantity should vary for each alternative considered in the environmental document. Estimates of the projected wood sale quantity and the projected timber sale quantity do not include any volumes anticipated from salvage or sanitation harvests.

See section 64.35, exhibit 01 of this Handbook for a comparison of the different characteristics between the sustained yield limit, the projected wood sale quantity, and the projected timber sale quantity.

The displays in section 65.2 of this Handbook for timber volumes must show volumes for both the first and second decade. Although the NFMA provides that the plan period is at least every 15 years, it limits the sale of timber to less the sustained yield limit for each *decade* of the plan (16 U.S.C. 1611). Providing estimates in the plan of the annual projected wood sale quantity and the annual projected timber sale quantity for the each of first two decades aligns with the NFMA decadal periods limiting the sale of timber, and provides estimates to cover a second decade if revision of the plan is delayed beyond the 15-year limit.

The display of the projected wood sale quantity and projected timber sale quantity is described in section 65.2 of this Handbook.

After development or revision of the plan, the Responsible Official shall track the quantity of woody material sold from the plan area (comparable to the projected wood sale quantity) and specifically the quantity of timber sold that meets utilization standards (comparable to the projected timber sale quantity) to evaluate compliance with the decadal timber limitation described in section 64.3 of this Handbook.

If intensified management practices were used to calculate the projected timber sale quantity, and such practices did not occur as planned, the projected timber sale quantity must be decreased at the end of the plan period by an amount attributable to the contribution of those intensified practices to the estimation of the projected timber sale quantity.

All data resulting from the guidance in this directive pertinent to timber volumes should be compiled according to the reporting requirements of the official Forest Service applications, the Timber Information Manager (TIM) and the Forest Service Activity Tracking System (FACTS), to report timber resource activities and timber sale accomplishment consistent with this directive (see FSM 2410 – Timber Resource Management Planning).

64.33 – Departure from Sustained Yield Limit

To meet overall multiple-use objectives and achieve the plan's desired conditions and objectives, the Responsible Official may decide to increase the expected sale of timber above the sustained yield limit (SYL) for the first decade of the plan, and for a second decade if necessary. In a departure the sustained yield limit is replaced by a departure limit that represents the maximum amount of timber meeting utilization standards that can be sold for the first or second decade of the plan. The departure limit can be different for each of these two decades. The departure limit is only applicable to a departure alternative considered in the appropriate environmental document. In all other respects, the assumptions for its calculation are the same as for the sustained yield limit.

Departure from the sustained yield limit must be designed for achieving the multiple-use management objectives of the land management plan, as reflected in the plan desired conditions, objectives and other plan components. The rationale for the departure must be explained in the plan decision document. Departures are expected to be used rarely to achieve desired conditions and objectives that cannot be achieved without the departure.

A departure above the sustained yield limit must be made with public participation after a public review and comment period of at least 90 days (16 USC 1604(d) and 1611(a); 36 CFR 219.11(d)(6)). The 90-day comment period for review of a proposed land management plan or plan revision meets this requirement (36 CFR 219.16(a)(2)).

The departure increment may exceed the sustained yield limit for one or more decades and subsequently lead to the projected timber sale quantity dropping below the sustained yield limit.

The Responsible Official may review the accomplishment and effects of the departure schedule and adjust as appropriate in the adaptive management framework.

64.34 – Utilization Standards

For purposes of the calculations and measurement of timber volumes described in section 64.35 of this Handbook and referenced in section 65.2, exhibits 01 and 02 of this Handbook; the plan must identify or reference the appropriate utilization standards that identify the standard types of timber products expected to be sold. These utilization standards may distinguish between sawtimber, pulpwood, and other products. Utilization standards for timber should not include branches, sawdust, fuelwood, firewood, biomass, or other woody material not consistently sold or measured on a cubic-volume basis. The woody material expected to be sold that does not meet timber utilization standards is not included in the projected timber sale quantity, but must be displayed as part of the projected wood sale quantity (sec. 65.1 of this Handbook). These utilization standards are not plan standards as described in 36 CFR 219.7(e)(1)(iii).

Regions must identify utilization standards in regional supplements to the directives to maintain a basis for consistent calculation and measurement of timber quantities identified in the plan and timber sold within the region. The plan must identify or reference the utilization standards used in developing the determination of the sustained yield limit and the estimation of the projected timber sale quantity.

64.35 – Summary of Measures of Timber Volumes

Exhibit 01 summarizes the characteristics of these different measures of timber volume described throughout section 65.2 of this Handbook.

64.35 - Exhibit 01

Characteristics of Timber Volume Measures

	Sustained Yield Limit (SYL)	Projected Wood Sale Quantity (PWSQ)	Projected Timber Sale Quantity (PTSQ)
Based on lands that *may* be suitable for timber production.	Yes	No	No
Based on quantity sold from all lands in plan area.	No	Yes	Yes
Based on the assumption that all lands that may be suitable for timber production are managed for timber production.	Yes	No	No
Limited by plan components, fiscal capability, and organizational capacity.	No	Yes	Yes
All volume meets utilization standards.	Yes	No	Yes
Includes salvage or sanitation harvest volume.	No	No	No
Varies by alternative in Plan EIS. *	No	Yes	Yes

* A departure limit shares the characteristics of the sustained yield limit, except that it is unique for an alternative that uses a departure.

65 – LAND MANGEMENT PLAN GUIDANCE

In addition to 36 CFR 219.11, sections 219.7, 219.8, 219.9, and 219.10 have substantive requirements that guide the development of plan components for vegetation.
FSH 1919.12, chapter 20, describes the plan development process and provides guidance for the development of plan components and other plan content.

The National Forest Management Act also requires plans to include specific information regarding timber management at 16 USC 1604 (e)(2) and (f)(2), as follows.

> **(e) Required assurances**
>
> **In developing, maintaining, and revising plans for units of the National Forest System pursuant to this section, the Secretary shall assure that such plans -**
>
> ***
>
> **(2) determine forest management systems, harvesting levels, and procedures in the light of all of the uses set forth in subsection (c)(1) of this section, the definition of the terms "multiple use" and "sustained yield" as provided in the Multiple-Use Sustained-Yield Act of 1960, and the availability of lands and their suitability for resource management**
>
> **(f) Required provisions**
>
> **Plans developed in accordance with this section shall -**
>
> ***
>
> **(2) be embodied in appropriate written material, including maps and other descriptive documents, reflecting proposed and possible actions, including the planned timber sale program and the proportion of probable methods of timber harvest within the unit necessary to fulfill the plan;**

The Planning Rule requires this information as plan content other than plan components, at 36 CFR 219.7:

> **(f) Other content in the plan. (1) Other required content in the plan. Every plan must . . .**
>
> ***
>
> **(iv) Contain information reflecting proposed and possible actions that may occur on the plan area during the life of the plan, including: the planned timber sale program; timber harvesting levels; and the proportion of probable methods of forest vegetation management**

> **practices expected to be used (16 U.S.C. 1604(e)(2) and (f)(2)). Such information is not a commitment to take any action and is not a ''proposal'' as defined by the Council on Environmental Quality regulations for implementing NEPA.**
> (40 CFR 1508.23, 42 U.S.C. 4322(2)(C)).

Section 65.1and 65.2 of this Handbook describe how to present in the land management plan the planned timber sale program. This includes the probable methods of forest vegetation management practices including timber harvest by the amount of land area planned for treatment; and the planned timber harvest levels in terms of the measures of timber volume discussed in 64.3 of this Handbook.

65.1 – Display of Forest Vegetation Management Practices

To meet the requirement for a display of the planned types of vegetation management practices including the planned timber sale program and the proportion of probable methods of timber harvest, use a table similar to that in exhibit 01. The table may also include other vegetation management practices such as prescribed fire, reforestation, or stand improvement. The table should display the estimated annual acreage of these practices planned for the first and second decades in the plan area, and the description of the table should acknowledge that the plan must be revised at least once every 15 years.

The planned practices are the estimated types of practices planned to achieve the outcomes described by the plan's desired conditions and objectives, consistent with the other plan components during the plan period. The estimated practices must be based on the fiscal capability of the planning unit. The estimated practices are not a commitment to take an action or a proposal for such action.

At a minimum, this identification of management practices must display or describe practices of even-aged and uneven-aged management systems planned for the plan area. The practices identified can be broken out by lands suited or not suited for timber production or any other land stratification deemed appropriate by the Responsible Official.

These estimated vegetation practices may also be plan objectives in the land management plan. The estimated practices usually derive from the analysis of the selected alternative in the environmental impact statement.

65.1 - Exhibit 01

Estimated Vegetation Management Practices

Forest-wide Vegetation Management Practices (acres) Annual Average per Decade		
Forest Cover Types/ Vegetation Management Practices	**Summary**	
	1st Decade	**2nd Decade**
Openings Maintenance Even Aged		
Prescribed Fire	0	1,040
Aspen Treatments		
Regeneration* (Even-aged harvest)	1,890	600
Thinning (Intermediate harvest)	960	390
Jack Pine Treatments		
Regeneration* (Even-aged harvest)	1,210	900
Thinning (Intermediate harvest)	230	30
Mid-seral Treatments		
Regeneration* (Even-aged harvest)	250	1,030
Thinning (Intermediate harvest)	3,730	2,510
Late-seral Treatments		
Regeneration* (Even-aged harvest)	1,770	3,560
Thinning (Intermediate harvest)	4,420	2,830
Improvement/Selection (Uneven-aged harvest)	3,820	3,920
Total Treatments		
Regeneration* (Even-aged harvest)	5,110	6,090
Thinning (Uneven-aged harvest)	9,340	5,760
Improvement/Selection (Uneven-aged harvest)	3,820	3,920

* Regeneration harvest treatment includes clearcuts, shelterwoods, shelterwood removal and seedtree methods.

65.2 – Display of Timber Harvest Levels

Display the sustained yield limit, the projected wood sale quantity, and the projected timber sale quantity, in a table with a format similar to that presented in exhibit 01 or exhibit 02. Exhibit 01 displays this information for a normal situation while exhibit 02 shows a similarly formatted table for a departure situation. All volumes in the table should be displayed as annual averages for the decade rather than decadal totals.

The following provides an explanation of the rows in the table at exhibit 02:

The first line shows the **sustained yield limit** in millions of cubic feet per year.

Row A displays the estimated volumes of timber meeting timber product utilization standards expected to be sold from lands suitable for timber production. Line A1 shows the amount of sawtimber volume in cubic feet and board feet for each decade. Line A2 shows the amount of non-sawtimber products expected to be sold that meet utilization standards for sale.

Row B shows the estimated volumes of timber meeting timber product utilization standards expected to be sold from lands not suitable for timber production if the plan anticipates timber harvest on those lands. If the plan anticipates timber harvest for protection of multiple use values on lands not suitable for timber production, separate harvest volume breakdowns should be provided for lands suitable and not suitable for timber production. The **B** lines repeat the same format and sequence as the **A** lines.

Row C sums these quantities from rows A1, A2, B1 and B2, to indicate the **projected timber sale quantity** for the plan area for the first two decades in both cubic and board foot measurements. The cubic foot total must be less than the sustained yield limit for each decade.

Row D shows any estimates of woody material not meeting utilization standards, such as fuelwood, firewood, or woody biomass, that are expected to be sold in the first two decades. If it is useful to do so, this row could show different types of wood products anticipated; only fuelwood is shown in the exhibit. Volumes should be displayed in both cubic feet and tons.

Finally, row **E** provides the **projected wood sale quantity** that includes all wood products and materials expected to be sold. This quantity must be displayed in cubic feet, but may also be converted into other units of measure.

For a departure schedule, the format in exhibit 02 should be used. This table has the same format as the table in exhibit 01, except that the departure increment and the departure limit are added to the table. Exhibit 02 shows a departure only for the first decade for restoration activity on lands that are not suitable for timber production. In a departure decade, the projected timber sale quantity must be less than or equal to the departure limit. In the second decade, there is no departure and the projected timber sale quantity must be less than or equal to the sustained yield limit.

These estimated outputs may be displayed in the plan in this or any supplemental table by softwoods and hardwoods, for different product types, species or any other desired breakdown.

65.2 - Exhibit 01

Planned Timber Sale Program

Annual Average Volume Outputs for 1st and 2nd Decade

Sustained Yield Limit (SYL)	**50.0 MMCF**			
	First Decade		**Second Decade**	
	MMCF	**MMBF**	**MMCF**	**MMBF**
Timber Products	Volumes other than salvage or sanitation volumes that meet timber product utilization standards			
Lands suitable for timber production				
A1. Sawtimber	18.0	90.0	18.0	90.0
A2. Other products	1.7		1.7	
Lands not suitable for timber production				
B1. Sawtimber	7.0	35.0	5.0	25.0
B2. Other products	1.3		1.3	
C. **Projected Timber Sale Quantity (PTSQ)** (A1+A2+B1+B2)	**28.0**	125.0	**26.0**	115.0
Other Estimated Wood Products	Fuelwood, biomass, and other volumes that do not meet timber product utilization standards			
	MMCF	**Tons**	**MMCF**	**Tons**
D. Fuelwood	3.0	45.0	2.5	37.5
E. **Projected Wood Sale Quantity (PWSQ)** (C+D)	**31.0**		**28.5**	

MMCF: Millions of cubic feet
MMBF: Millions of board feet

65.1 - Exhibit 02

Planned Timber Sale Program

With a Departure in the First Decade

Annual Average Volume Outputs for 1st and 2nd Decade

Sustained Yield Limit (SYL)	**50.0 MMCF**			
	First Decade		**Second Decade**	
	MMCF	**MMBF**	**MMCF**	**MMBF**
Departure increment	10.0		0	
Departure Limit	**60.0**		**50.0**	
Timber Products	Volumes other than salvage or sanitation volumes that meet timber product utilization standards			
Lands suitable for timber production				
A1. Sawtimber	40	200.0	40.0	200.0
A2. Other products	8.0		8.0	
Lands not suitable for timber production				
B1. Sawtimber	10.0	50.0	1.0	5.0
B2. Other products	2.0		0.2	
C. **Projected Timber Sale Quantity (PTSQ)** (A1+A2+B1+B2)	**60.0**	250.0	**49.2**	205.0
Other Estimated Wood Products	Fuelwood, biomass, and other volumes that do not meet timber product utilization standards			
	MMCF	**Tons**	**MMCF**	**Tons**
D. Fuelwood	6.0	90.0	5.0	75.0
E. **Projected Wood Sale Quantity (PWSQ)** (C+D)	**66.0**		**54.2**	

MMCF: Millions of cubic feet
MMBF: Millions of board feet

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Review

Genetics and extinction

Richard Frankham *

Key Centre for Biodiversity and Bioresources, Department of Biological Sciences, Macquarie University, NSW 2109, Australia
Australian Museum, 6 College Street, Sydney, NSW 2090, Australia

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Abstract

The role of genetic factors in extinction has been a controversial issue, especially since Lande's paper [Genetics and demography in biological conservation, Science 241 (1988) 1455–1460] paper in Science. Here I review the evidence on the contribution of genetic factors to extinction risk. Inbreeding depression, loss of genetic diversity and mutation accumulation have been hypothesised to increase extinction risk. There is now compelling evidence that inbreeding depression and loss of genetic diversity increase extinction risk in laboratory populations of naturally outbreeding species. There is now clear evidence for inbreeding depression in wild species of naturally outbreeding species and strong grounds from individual case studies and from computer projections for believing that this contributes to extinction risk. Further, most species are not driven to extinction before genetic factors have time to impact. The contributions of mutation accumulation to extinction risk in threatened taxa appear to be small and to require very many generations. Thus, there is now sufficient evidence to regard the controversies regarding the contribution of genetic factors to extinction risk as resolved. If genetic factors are ignored, extinction risk will be underestimated and inappropriate recovery strategies may be used.
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Keywords: Extinction risk; Genetic diversity; Inbreeding depression; Mutation accumulation; Self-incompatibility

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* Tel.: +61 02 9850 8186; fax: +61 02 9850 8245.
E-mail address: rfrankha@els.mq.edu.au.

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1. Introduction

There is ongoing controversy about the importance of genetic factors in extinction risk. Biological extinctions are due to the combined effects of deterministic (habitat loss, over exploitation, introduced species and pollution) and stochastic (demographic, environmental, genetic and catastrophic) factors (Shaffer, 1981). Prior to 1970, genetic factors were rarely mentioned as causes of extinction. Frankel (1970, 1974) was primarily responsible for proposing that loss of genetic diversity elevated extinction risk, especially by compromising evolutionary response to environmental change. Frankel and Soulé (1981) added the deleterious effects of inbreeding to the argument, and they presumed that genetic factors had an important role in causing extinctions. Later, the issue of mutation accumulation was introduced by Lande (1995) and Lynch et al. (1995). Outbreeding between diverged populations may also lead to deleterious effects on reproductive fitness (outbreeding depression), but its effects are clearly less important overall than that of inbreeding depression (Frankham et al., 2002). It is most probable when populations that are highly differentiated genetically are crossed. I will not consider this issue in detail here.

Genetic factors affect extinction risk because threatened species have small and/or declining populations (IUCN, 2004), and in such populations inbreeding and loss of genetic diversity are unavoidable (Frankham et al., 2002). The relationships between population size, loss of genetic diversity and inbreeding in closed random mating populations are described by the following equation:

$$H_t/H_0 = (1 - 1/[2N_e])^t = 1 - F, \quad (1)$$

where H_t is heterozygosity (Hardy–Weinberg expected heterozygosity, or gene diversity) at generation t, H_0 initial heterozygosity, N_e the genetically effective population size and F the inbreeding coefficient. Since the middle term in the equation is approximately $e^{-t/2N_e}$ this equation predicts an exponential decay of genetic diversity with generations that occurs at greater rates in smaller than larger populations (Fig. 1). The inbreeding coefficient equals the proportionate loss of genetic diversity. The rate of decay in genetic diversity and the increase in inbreeding depend upon the genetically effective population size, rather than the actual or census size. The effective size is typically much smaller than the number of potentially breeding adults in populations, averaging an order of magnitude lower than census population sizes (Frankham, 1995a).

A major controversy erupted over the role of genetic factors in extinction risk following Lande, 1988) paper in Science. He was interpreted as saying that most species are driven to extinction before genetic factors have time to impact them (the 'Lande scenario'; Pimm, 1991; Young, 1991; Wilson, 1992; Caro and Laurenson, 1994; Caughley, 1994; Dobson, 1999; Elgar and Clode, 2001). The effectiveness of natural selection in reducing the frequency of the deleterious alleles (purging) that cause inbreeding depression has been an important part of this controversy (Lande, 1988; Hedrick, 1994).

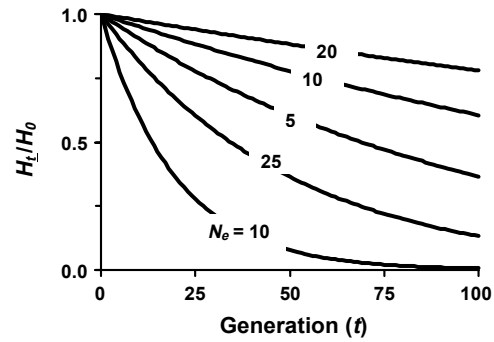

Fig. 1. Proportionate loss of genetic diversity (H_t/H_0) with generations in populations with different effective sizes (N_e).

A second controversy about the impact of inbreeding on reproductive fitness began in the late 1970s in relation to captive animals (see Ralls et al., 1979), and in the 1990s spread to scepticism about whether inbreeding depression affected species in wild habitats (see Caro and Laurenson, 1994; Caughley, 1994; Craig, 1994; Merola, 1994). A third controversy has erupted concerning the role of mutation accumulation in extinction risk for sexually reproducing species (Charlesworth et al., 1993; Lande, 1995; Lynch et al., 1995).

The purpose of this review is to examine evidence on the impacts of inbreeding depression, loss of genetic diversity and mutational accumulation on extinction risk.

2. Inbreeding depression

2.1. Effects of inbreeding on fitness in outbreeding species

Inbreeding has long been known to reduce reproduction and survival in naturally outbreeding species (inbreeding depression). Darwin (1876) provided the first compelling evidence on this, based on comparisons of the progeny of self and cross-fertilization in 57 species of plants. Selfing reduced seed production by an average of 41% and height by 13%. Not all species showed inbreeding depression for all characters studied, but virtually all showed it for most reproductive fitness characters. Subsequently, similar conclusions were found to apply for laboratory and domestic animals and plants (Charlesworth and Charlesworth, 1987; Falconer and Mackay, 1996; Lynch and Walsh, 1998). Inbreeding has deleterious consequences on all aspects of reproduction and survival, including sperm production, mating ability, female fecundity, juvenile survival, mothering

ability, age at sexual maturity and adult survival in animals, and in related components in plants (Frankham et al., 2002).

However, zoo keepers were sceptical that inbreeding depression applied to wildlife (Ralls et al., 1979). This controversy was resolved by Ralls and Ballou (1983) findings that juvenile mortality was higher in inbred than in contemporary non-inbred mammals for 41 of 44 populations studied. Further, progeny resulting from brother-sister (full-sib) mating had 33% higher juvenile mortality on average than found for non-inbred animals (Ralls et al., 1988). Deleterious effects of inbreeding on other reproductive and survival characters have also been reported for wildlife in captivity (Laikre and Ryman, 1991; Lacy et al., 1993; Laikre et al., 1996; Wilcken, 2001).

Controversy next erupted about the impacts of inbreeding on wild species in natural habitats (Caro and Laurenson, 1994; Caughley, 1994; Pimm, 1991; Wilson, 1992; Elgar and Clode, 2001). There is now ample evidence to resolve this issue. Of 157 valid data sets across 34 taxa compiled by Crnokrak and Roff (1999), 90% showed that inbred individuals had poorer reproductive fitness characters than non-inbred individuals. The few exceptions were small data sets and ones for which parentage had not been verified genetically. The effects are similar across a broad range of major taxa of animals and plants. Inbreeding depression has also been reported in at least another 15 wild populations (Frankham, 2000). The impacts of inbreeding depression are about seven-times greater in wild habitats than in captivity (Crnokrak and Roff, 1999). Lacy (1997) concluded that he was unaware of any well-studied species that did not show inbreeding depression. Overall, there is now compelling evidence that wild species in natural habitats show inbreeding depression.

2.2. Affect on extinction risk

Since inbreeding reduces reproduction and survival, it is expected to increase extinction risks under situations where other factors are controlled, or excluded. This has been verified in laboratory populations of *Drosophila*, houseflies and mice (Frankham, 1995b; Bijlsma et al., 1999, 2000; Reed and Bryant, 2000; Reed et al., 2002, 2003). While most of these studies have used full-sib inbreeding, slower inbreeding at rates within the range of many endangered species has also been shown to increase extinction risk (Reed and Bryant, 2000; Reed et al., 2003), as illustrated in Fig. 2.

While the situation in wild populations is more complex, inbreeding and loss of genetic diversity have been shown to contribute to extinction risk in the plant *Clarkia pulchella* (Newman and Pilson, 1997) and in Finnish populations of the Glanville fritillary butterfly (*Melitaea cinixia*: Saccheri et al., 1998). In the plant experiment,

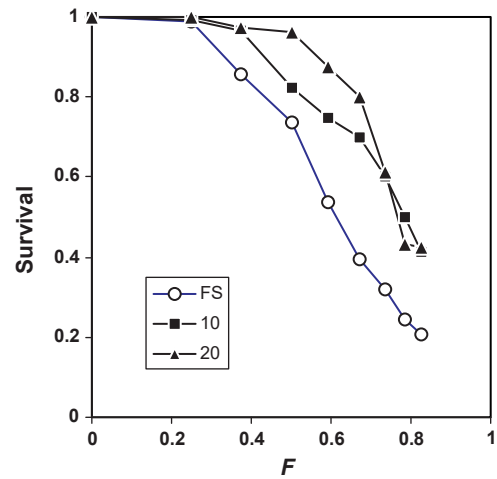

Fig. 2. Relationship between inbreeding (F) and proportion of replicate populations surviving for populations maintained using full-sib (FS) mating or effective sizes of 10 and 20 (Reed et al., 2003).

the extinction rate for less inbred populations ($F = 4\%$) was 25% while it was 69% for more inbred populations ($F = 8$–9%), a huge difference in extinction rates for a small difference in inbreeding coefficient. In the butterfly study, 42 populations were genotyped in 1995 and their survival to the following year determined. When all factors known to be affecting extinction risk in these populations were removed, level of inbreeding was still a significant predictor of extinction risk. Overall, inbreeding explained 26% of the variation in extinction risk.

Are these two cases exceptions or part of the general story concerning the contribution of inbreeding to extinction risk? Clearly many more field studies are needed, but we need an immediate resolution of this issue, as it affects management of threatened species. Using realistic computer projections for 20 threatened species, we compared median times to extinction for models with and without inbreeding depression (Brook et al., 2002). We applied inbreeding depression at a very conservative level of 3.14 diploid lethal equivalents for juvenile survival, as found by Ralls et al. (1988) for captive mammals, and included the effects of natural selection (purging) in our models. Inbreeding depression reduced median times to extinction by 25–31% for populations with initial sizes of 50, 250 and 1000, when all known or suspected demographic and stochastic threatening factors were included in the models. Other related studies on individual taxa have reached similar conclusions (Dobson et al., 1992; Oostermeijer, 2000), provided they carried the study for sufficient generations. With a more realistic level of inbreeding depression of 12 diploid lethal equivalents spread across the life cycle (Keller, 1998), stochastic computer projections for multiple taxa indicate even greater reductions in median time to extinction due to inbreeding depression (O'Grady et al. unpublished data). Thus, most naturally outbreeding taxa are likely to have their extinction risk elevated by inbreeding.

Populations with strongly negative population growth rates were less affected by inbreeding. They were driven to extinction so rapidly that genetic factors had insufficient time to impact, as Lande (1988) suggested. No impacts of inbreeding occurred in only two of 30 taxa, ones where the impacts of anthropogenic factors were very severe. Less direct evidence indicates that the proneness to extinction of island populations of vertebrates is partially due to inbreeding and loss of genetic diversity (Frankham, 1998).

2.3. Are species driven to extinction before genetic factors can impact?

While the Lande (1988) scenario has been promoted by many authors, it had not until recently been subjected to a comprehensive test. If threatened species are driven to extinction before genetic factors impact, they will have similar genetic diversity to taxonomically related non-threatened taxa. If not, threatened taxa will have lower genetic diversity and consequent higher levels of inbreeding (see Eq. 1) and lowered reproductive fitness than related non-threatened taxa (Reed and Frankham, 2003). In a comprehensive meta-analysis involving 170 paired comparisons, we found that the majority of threatened taxa (77%) exhibited reduced genetic diversity (Spielman et al., 2004), leading us to reject the 'Lande scenario' for most taxa. This effect was consistent across a broad array of major taxa and the median difference in heterozygosity was 40% (Fig. 3). We were not able to determine what caused the taxa to become threatened, but the results indicate that most threatened taxa are now suffering reduced ability to evolve, elevated inbreeding and consequent reduced reproductive fitness (Reed and Frankham, 2003) and that they will suffer elevated extinction risk into the future. Further, there is still time for genetic factors to impact upon taxa with normal levels of genetic diversity.

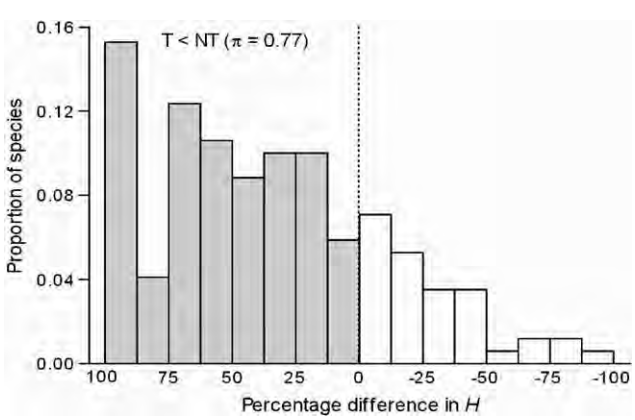

Fig. 3. Distribution of percentage differences in genetic diversity between threatened (T) and taxonomically related non-threatened (NT) taxa for 170 pairwise comparisons (Spielman et al., 2004).

For example, vulnerable taxa, the majority of our data set, have a probability of extinction of approximately 10% within 100 years (IUCN, 2004).

Why does not the 'Lande scenario' hold? New information since 1988 has changed our perceptions in four areas that all point to greater impacts of genetic factors on extinction risk. First, the effective population sizes of threatened species, the population size that determines genetic impacts (see Eq. 1), are lower than imagined in 1988. For most species, N_e is not known and must be estimated from the census size (N) and broadly based estimates of the N_e/N ratio. In 1988 and later, this ratio was thought to be 0.25–0.5 (Mace and Lande, 1991). Subsequently, I found in a meta-analysis that this ratio averaged approximately 0.1 when all relevant factors were considered (Frankham, 1995a). Thus, genetic factors impact sooner and at a greater rate than expected in 1988.

Second, information on the full impact of inbreeding depression was limited in 1988. Most people used the estimate of 3.14 lethal equivalents from Ralls et al. (1988), while current evidence now suggests inbreeding depression is around 12 lethal equivalents for the whole life cycle in the wild (Keller, 1998). Third, knowledge of the interactions between demographic and environmental stochasticity and catastrophes with inbreeding depression was limited in 1988. Subsequent information indicates strong interactions that increase the impacts of inbreeding on fitness in an extinction vortex (van Noordwijk, 1994; Tanaka, 2000).

Fourth, Lande (1988) suggested that purging was effective in removing the deleterious alleles that cause inbreeding depression. However, theoretical work indicates that purging will often have only modest effects in small populations, as deleterious alleles of small effect become effectively neutral and can drift to fixation (Hedrick, 1994; Thévenon and Couvet, 2002). Empirical evidence has typically found only moderate effects of purging (Ballou, 1997; Byers and Waller, 1999; Fowler and Whitlock, 1999; Frankham et al., 2001; Crnokrak and Barrett, 2002; Day et al., 2003; Reed et al., 2003). For example, extinction rates due to inbreeding increase more slowly with slower than faster inbreeding, but all treatments show elevated extinction rates with increasing F (Fig. 2).

A related issue concerns small surviving populations whose fitness is apparently normal (Craig, 1994; Elgar and Clode, 2001). For example, Chatham Island black robins (*Petroica traversi*), golden hamsters (*Mesocricetus auratus*) and Mauritius kestrels (*Falco punctatus*) all survived population bottlenecks of a single pair (Groombridge et al., 2000; Frankham et al., 2002). These and similar cases have led some authors to question the importance of inbreeding and loss of genetic diversity in population viability. However, the observations are highly selective, as they ignore the cases (the majority?) of small population that have gone extinct

(Laikre et al., 1997). Two unbiased studies both indicate that small surviving populations have reduced reproductive fitness. All 31 populations of *Drosophila* of size $N_e = 20$ that survived until $F = 0.8$ had lower fitness than the base population (Reed et al., 2003). Further, Briskie and Mackintosh (2004) showed that hatching failure was higher in bottlenecked than non-bottlenecked populations in New Zealand when the bottleneck size was less than 150 for 22 species of native birds and when the bottleneck was less than 600 for 15 species of introduced birds.

From the evidence above, inbreeding depression is likely to contribute to extinction risk for most threatened outbreeding taxa.

2.4. Recovery of small, inbred populations following outcrossing

Inbreeding is reduced following crossing of unrelated populations, even if those populations are themselves inbred. Outcrossing is well known to remove inbreeding depression in laboratory and domestic animals and plants (Spielman and Frankham, 1992; Falconer and Mackay, 1996). Such genetically stimulated recoveries have now been documented in the wild in populations of deer mice (*Peromyscus maniculatus*; Schwartz and Mills, 2005), gray wolf (*Canis lupus*; Vilà et al., 2003), greater prairie chicken (*Tymphanuchus cupido pinnatus*; Westemeier et al., 1998), adders (*Vipera berus*; Madsen et al., 1999, 2004), fish (*Poeciliopsis monacha*; Vrijenhoek, 1994), and plant species (*Ipomopsis aggregata*; Heschel and Paige, 1995; *Silene alba*; Richards, 2000).

2.5. Effects of inbreeding on fitness in species that inbreed

Inbreeding depression due to self-fertilization is typically less in species that inbreed naturally (23% reduction in mean fitness) than in natural outbreeders (53% reduction), based on a meta-analysis (Husband and Schemske, 1996). Given that the inbreeding coefficient increases from 0% to 50% to 75% in the first two generation of selfing, most of the difference can be explained if the species that inbreed naturally had one prior generation of selfing from an outbred base before their inbreeding depression was assessed.

I am unaware of any theoretical, simulation or empirical studies of the impact of inbreeding depression on extinction risk for naturally inbreeding taxa.

3. Loss of genetic diversity

3.1. Loss of evolutionary potential

Loss of genetic diversity in small populations is expected to increase extinction risk by adversely affecting the ability of populations to evolve to cope with environmental change (evolutionary potential). Environmental change is experienced by essentially all species, whether it be due to global climate change, new or changed diseases, pests and parasites, new predators, climatic cycles, etc. (Frankham and Kingslover, 2004). Evolutionary changes have been documented in many species in natural or human affected habitats (Briggs and Walters, 1997; Thompson, 1998; Mousseau et al., 2000; Bradshaw and Holzapfel, 2001; Umina et al., 2005).

The rate of evolutionary change (R) is determined primarily by quantitative genetic variation (Franklin, 1980) and is predicted by the breeders' equation below (Falconer and Mackay, 1996):

$$R = Sh^2, \tag{2}$$

where S the selection differential and h^2 the heritability, the proportion of the observed phenotypic variation that is due to additive genetic causes (V_A). The connection to genetic diversity is provided by the equation for V_A where it is seen to be a function of $2pq$, the expected heterozygosity in a random mating population (Falconer and Mackay, 1996), as follows:

$$V_A = \sum_{i=1,}^{\#\text{loci}} 2p_i q_i (a_i + d_i[q_i - p_i])^2, \tag{3}$$

where p and q are the allele frequencies, a twice the difference in mean phenotype between the two homozygotes and d is the deviation of the heterozygote phenotype from the mean of the two homozygotes.

Genetic diversity is widely assumed to exist for almost every trait in large wild outbreeding populations (Lewontin, 1974). However, there are exceptions as only some species of plants present in regions with heavy metal waste heaps have evolved resistance to heavy metals and only some plant species with exposure have evolved herbicide tolerance (Bradshaw, 1991). Further, Hoffmann et al. (2003) found that a population of a rainforest species of *Drosophila* near the limits of its distribution did not have genetic variation for desiccation resistance, in spite of having ample microsatellite genetic diversity.

Most information on genetic diversity in threatened species is for molecular variation, while quantitative genetic variation is the main determinant of the ability to evolve (Frankham et al., 2002). While both of these are related to heterozygosity, their correlation is low, and not different from zero for life history traits (Reed and Frankham, 2001). The low correlation could be due to differential natural selection on effectively neutral molecular versus adaptive quantitative genetic loci, to complications from non-additive genetic variation, to linkage disequilibrium, or to sampling variation. For characters peripheral to reproductive fitness with primarily additive genetic variation, the rates of loss of molecular and quantitative genetic variation in small populations do not differ significantly (Gilligan et al., 2005), pointing

to sampling variation as the major reasons for low correlations for such traits.

Characters associated with reproductive fitness exhibit higher levels of dominance and epistatic variation (together referred to as non-additive genetic variation) than characters more peripheral to fitness (Crnokrak and Roff, 1995). Following population bottlenecks, non-additive genetic variation may be converted into additive genetic variation (Robertson, 1952; Willis and Orr, 1993; Cheverud et al., 1999), leading to increases, rather than decreases in additive genetic variation. This has been observed empirically (Bryant et al., 1986; Lopez-Fanjul and Villaverde, 1989; Fernández et al., 1995; Wade et al., 1996). However, the effect is expected to be a relatively short-term one, as the relationship between additive genetic variation and inbreeding coefficient is curvilinear, but V_A is expected to be zero when $F = 1$. Further, in natural outbreeders, population size bottlenecks will lead to inbreeding and reduced fitness. As this reduces the selection differential (S in Eq. 2), it reduces the ability to evolve. Population bottlenecks have been shown to reduce the ability to evolve in tests where the combined effects of inbreeding depression and loss of genetic diversity were measured (Frankham et al., 1999, 2002, p. 235).

IUCN (McNeely et al., 1990) designate genetic diversity as one of three levels of biodiversity requiring conservation, but does inbreeding depression or loss of genetic diversity have the most important impact on extinction risk? Inbreeding depression has an immediate impact, while loss of genetic diversity typically impacts over the long-term, associated with environmental change. Vilas et al. (2005) found that inbreeding depression had significant impacts over a short time span in a plant species in Spain, whilst there was no detectable impact of genetic diversity.

3.2. Loss of alleles at self-incompatibility loci

In self-incompatible species, loss of genetic diversity reduces mate availability and population fitness due to an effect that is neither inbreeding depression, nor loss of evolutionary potential. Self-incompatibility in plants is controlled by one or more loci that have many alleles in large populations. However, such alleles are lost in small populations, reducing mate availability and thus reducing mean population fitness (Les et al., 1991; Richman and Kohn, 1996; Young et al., 2000). This effect may lead to extinction. A population of Lakeside daisy (*Hymenoxys acaulis* var. *glabra*) in Illinois was reduced to so few individuals that it did not reproduce for 15 years in spite of pollen flow (Demauro, 1993). It was only recovered by outcrossing to populations from elsewhere (Demauro, 1994). Reduced mate availability will eventually affect all threatened self-incompatible species with small populations.

In Hymenoptera, a related effect occurs at the sex determining locus of haplo-diploid species, due to the production of homozygous diploid males (Pamilo and Crozier, 1997). Even in mammals, preference for unrelated mates might reduce mate availability (Ryan and Lacy, 2003).

4. Mutation accumulation and meltdown

In large populations, deleterious alleles are kept at low frequencies due to the balance between mutation and natural selection. However, in small populations, selection is less effective and mildly deleterious alleles become selectively neutral, with their fate being determined by genetic drift (Lande, 1995; Lynch et al., 1995). Consequently, some of these mildly deleterious alleles increase in frequency and reduce reproductive fitness. Over long time spans, sufficient alleles could drift to fixation to cause negative population growth and a decline to extinction, termed mutational meltdown (Lynch et al., 1993).

The contribution of mutation accumulation to extinction risk is controversial, especially in naturally outbreeding species (Charlesworth et al., 1993; Lande, 1995; Lynch et al., 1995). Garcia-Dorado (2003) has shown that the effect of mutation accumulation depends critically on the distribution of mutational effects, being important when there are many mutations of predominantly small effect, but only minor when a substantial proportion of mutations have large effects. Favourable and compensatory mutations reduce the impacts of deleterious mutations on fitness and extinction risk (Lande, 1998; Whitlock, 2000; Whitlock et al., 2003). There is no consensus about the rate and distributions of effects of new mutations (Keightley and Lynch, 2003).

Empirical tests have also been controversial. Gilligan et al. (1997) found no evidence of greater mutational accumulation in smaller than larger populations in *Drosophila* populations with different sizes (N_e of 25, 50, 100, 250 and 500) maintained for 45–50 generations, leading us to conclude that mutation accumulation was of minor importance in normal time spans of conservation concern. While Shabalina et al. (1997) claimed to find deleterious effects of mutational accumulation in populations with N_e of 400 maintained under benign conditions, but whose fitness was measured under stressful conditions, their results appear to be due adaptations to the benign captive environment that were deleterious when moved to a stressful environment (Gilligan et al., 1997; Woodworth et al., 2002). Recently, Gu and Woodruff (personal communication) have found that mutation accumulation reduced fitness by 17% over 22 generations in populations maintained with four parents per generation, and increased the extinction risk, but no

changes were detected in populations of $N = 100$ (presumably N_e of ~ 30; Gilligan, 2001).

Asexually reproducing populations are expected to be more sensitive to mutation accumulation (Charlesworth et al., 1993). In yeast, one of 12 replicate populations went extinct due to mutation accumulations in populations with N_e of 250 over 2900 generations in a strain with an elevated mutation rate (Zeyl et al., 2001). No extinctions were observed in 12 control populations with normal mutation rates and their average population sizes did not change over time. Thus, the impact of mutational accumulation on extinction risk appears to be less important than the other genetic factors and to take very many generations.

5. Consequences of ignoring genetic factors

There is compelling evidence that inbreeding and loss of genetic variation contribute to extinction risk in captive populations, very strong evidence that they contribute in wild populations in nature, and evidence that most species are not driven to extinction before genetic factors impact them, as documented above. Thus, there is sufficient evidence to consider the controversies regarding the contribution of genetic factors to extinction risk as resolved.

However, does this matter in the context of the conservation management of threatened species? In what follows, I include additional issues relating to genetics and population viability that have not been addressed in this review (see Frankham et al., 2002). If genetic factors are ignored the following adverse effects are likely:

- Estimates of extinction risk will be substantially underestimated in many taxa (Brook et al., 2002).
- Inappropriate recovery strategies may be used. Attempts to recover the Illinois population of the greater prairie chicken by habitat restoration failed and it was only recovered following outcrossing to alleviate inbreeding depression (Westemeier et al., 1998).
- Inappropriate populations may be used for reintroductions. They may be adapted to the incorrect environment, as occurred when animals from Turkey and Sinai were added to a population of ibex in Czechoslovakia (Turcek, 1951). Alternatively, populations with low genetic diversity and high inbreeding may be used for reintroduction. Over 10,000 koalas have been restocked into southeastern Australia from an island population founded from 2–3 individuals. Not surprisingly, these restocked populations have low genetic diversity and lowered fitness, including lowered sperm concentrations and motility and an elevated frequency of testicular aplasia (Seymour et al., 2001).
- Fragmented populations may not be correctly diagnosed, and gene flow between them may not be correctly managed (Frankham et al., 2002).
- Problems associated with loss of self-incompatibility alleles in small populations will not be addressed (Young et al., 2000).
- Problems due to crossing of populations with different chromosome numbers will not be diagnosed and remedied. An endangered grassland daisy in southeastern Australia exists in both diploid and tetraploid forms. A restoration effort inadvertently placed plants of one ploidy near those of the other ploidy and some sterile triploid seeds resulted (Young and Murray, 2000).
- Different evolutionary significant units or taxa may be mixed and reduced fitness may result (outbreeding depression). Different sub-species of orangutans have been crossed in zoos and the hybrid animals (that are not suffering any reductions in fitness) are now not being bred and are using up spaces that could be used for captive breeding of this endangered species (Frankham et al., 2002). Similarly, African and Asiatic lions were inadvertently mixed in a captive breeding program. When this was discovered, the program was discontinued but not until a substantial amount of money had been wasted (Frankham et al., 2002).

Investigations of extinction risk, or recovery plans for threatened taxa that ignore genetic factors can no longer be considered scientifically credible.

6. Future directions

There are several areas where essential information is lacking, or controversial, or limited in scope, as follows:

- There is a great need for information on the impact of genetic factors on extinction risk for taxa that regularly inbreed or are asexual, haplo-diploid or polyploid.
- There is a need for much more information on the extent of inbreeding depression for the full life cycle for a broad range of outbreeding taxa in the wild. Additional information is required on the impacts of inbreeding depression on extinction risk in natural populations.
- Information is required on the relative contributions and interactions of inbreeding depression versus 'non-genetic' factors on extinction risk in wild populations, for example from realistic computer simulations.
- More information is required on the relative contributions of inbreeding depression, loss of genetic diversity and mutation accumulation to extinction risk, especially over the long term.

- Information on the extent of outbreeding depression and factors that influence it are sorely needed.
- Research is needed into the genetic impacts of habitat fragmentation and means for managing such situations. While there is need for field data, computer simulations should contribute in a major way to this in the short term, as many variables need to be evaluated simultaneously (Reed, 2004).

7. Conclusions

- Inbreeding and loss of genetic diversity contribute to extinction risk in small laboratory populations.
- Inbreeding depression contributes to extinction risk in most wild populations of naturally outbreeding species and loss of genetic diversity is expected to contribute in the long-term.
- Ignoring genetic factors may lead to inappropriate recovery strategies.
- Any study of extinction risk or minimum viable population sizes that excludes genetic factors will underestimate the true threat.

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A. Myrick Freeman III is William D. Shipman Professor of Economics Emeritus at Bowdoin College in Maine, USA, and a former Senior Fellow at Resources for the Future (RFF), Washington DC, USA.

Joseph A. Herriges is Professor of Economics at Iowa State University, USA. He was Managing Editor of the *Journal of Environmental Economics and Management* from 2001 to 2006.

Catherine L. Kling is Charles F. Curtiss Distinguished Professor at Iowa State University, USA, and the Director of the Center for Agricultural and Rural Development. She is a former President of the Association of Environmental and Resource Economists.

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Dedicated to our spouses:

Claire M. Darrow
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Foreword

I appreciate the opportunity to write the foreword for this new edition of *The Measurement of Environmental and Resource Values* because it allows me to be the first to comment on this revision to a classic. Over 35 years ago, when Freeman wrote the book that started his incredible legacy to environmental economists, the information available to economists about how people were affected by and responded to changes in environmental quality was limited. The primary data available were scarce. Surveys of outdoor recreation use (such as those conducted by the Bureau of Outdoor Recreation) and data describing average housing "prices" from the Census were the best we had. The norm for information allowing people's choices to be linked to environmental conditions was "bleak." Most data were from secondary sources. They were often aggregated in ways that compromised their ability to inform researchers about the importance of the differences in spatially delineated environmental services that people experienced as part of their everyday lives.

Today there appear to be few limits to what creative young environmental economists have been able to construct. We now have the ability to observe individuals and their households with considerable detail. Indeed, confidentiality concerns seem to be the single most important limitation to the granularity in the records this new generation has developed. We observe a host of market and non-market behaviors including how people use their time; and we can connect the environmental conditions that affect them in many diverse locations on the Earth. These records include variations between individuals at different locations at a point in time as well as over time. Often we can observe the same person (or household) responding to changes in environmental conditions over time. Economic analyses of what we can observe have also changed. Today there is greater integration of econometric methods and the economic theory describing how heterogeneous agents respond to changes in environmental services.

In the second edition of *Measurement*, Freeman noted that his book was intended to serve a complementary role to the treatment of econometric methods in other books. This new edition represents a significant departure from this strategy. In my view it is the first book in applied welfare economics to take seriously the need to integrate economics and micro-econometrics so that both contribute to

informing the decisions analysts must make in using the rich detail in this new information landscape to understand people's choices.

The result of this integration is important. We have the same clear explanations and attention to the role of assumptions in understanding the economic behaviors we wish to describe as Rick offered in his earlier editions. What is new is the direct link to the ways econometric models represent them. Some examples may help to illustrate my point. The discussion of the definitions and measures for changes in individual welfare in response to an exogenous change in the circumstances governing a person's choices now blends the careful treatment of how price, quantity, and quality changes are evaluated in Marshallian and Hicksian terms with an expansion in the discussion of how these distinctions are represented within discrete choice models. This blending of theory and empirical insights can also be seen in the discussion of how theory, experimental results, and new behavioral hypotheses all contribute to interpretations of the willingness to pay / willingness to accept disparity. There are many other examples of this integration throughout the new edition. The econometric treatment of data problems so often encountered in recreation demand modeling is recognized to be equally important to the measurement of consumer surplus with the models we estimate. This discussion is now able to highlight the modeling tradeoffs—simplicity in measuring tradeoffs versus strong restrictions to the data generating process. The discussion of modeling choices for describing multiple-site models is equally nuanced. The bottom line to these examples is that in each area where theory must be adapted to meet the complexities of real choice processes, the authors bring readers to the frontiers of our understanding. This is true for travel cost recreation demand, hedonic models, including locational equilibrium approaches, and stated preference surveys.

Ten years ago when I prepared the forward for Rick's second edition, I wondered (to myself) will this be the last revision? I wished there was a way to assure the Freeman legacy of providing a platform for clear access to what has been learned about measuring the economic values for changes in environmental services could be made sustainable. Rick and Resources for the Future answered my question. They recognized that many generations of environmental economists have been using those early editions of Freeman to learn and, as a result, appreciate the importance of maintaining it. This new edition adds two of the leaders in our field, Joe Herriges and Cathy Kling, to the team so that his legacy is sustained! By recruiting the best of his early "students" to help, RFF has assured the Freeman, Herriges, and Kling edition of *Measurement* will continue to help new generations of environmental economists understand what has been accomplished and build on it. In the process, I believe the addition of Joe and Cathy will enhance Rick's legacy by encouraging greater integration of economic theory and econometric methods in the ways we evaluate strategies for measuring environmental and resource values.

V. Kerry Smith, Cave Creek, AZ

Preface

Freeman has been very gratified by the reception that the second edition of this book received. He appreciates the many requests that he undertake another revision of the book to reflect recent developments in the field of nonmarket valuation. However, he recognizes that the field has been rapidly developing since his retirement from teaching some 13 years ago. He is immensely grateful that Joseph Herriges and Catherine Kling agreed to join him as coauthors of this, the third edition. Herriges and Kling, for their part, feel privileged to have been asked. As a graduate student, Kling used the first edition of Freeman's book and both Herriges and Kling taught for many years using the second edition. To be invited by the scholar that literally "wrote the book" on measuring environmental and natural resource values to participate in a revision of that book is truly an honor. Our goal was for the third edition to read as if written by a single author; consequently, we all contributed to all chapters, working and re-working the text together. We share equal responsibility for errors and oversights.

The objectives of this edition are essentially the same as those of the first two editions. These objectives are, first, to provide an introduction and overview of the principal methods and techniques of resource valuation to professional economists and graduate students who are not directly engaged in the field and, second, to give practitioners in the field an up-to-date reference on recent developments in the theory and methods underlying the practice of resource valuation. While we have tried to be comprehensive in our coverage of topics, this book is not a "how-to" manual. That kind of book would have to deal in much more detail with a host of econometric, data, and related technical issues and several excellent volumes are available. We do hope however that this book provides the necessary background in theory, basic models, data needs, and econometric overviews so that the reader will have a strong basis for diving into the task of undertaking a nonmarket valuation study.

What Is New

As the field of applied welfare economics and nonmarket valuation has become more mature, researchers have adapted and developed increasingly sophisticated

econometric tools. New theory and an increasing number of theoretical puzzles have also become apparent. A primary goal in this revision has been to update the text to reflect these advances and to freshen the examples with current empirical work on contemporary environmental issues. In addition to updates throughout the text, the chapter on stated preference methods has been completely rewritten and the recreation demand chapter significantly updated. In the chapter on property value models we have added a section on equilibrium sorting models. Also we have added discussions and references to recent work on behavioral economics and its implications for nonmarket valuation methods. Given the extraordinary creativity of environmental economists, we acknowledge that the text will quickly begin sliding out of date and we look forward to continuing to read and learn from our colleagues as this field continues to progress.

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A. Myrick Freeman III, Georgetown, ME
Catherine L. Kling, Ames, IA
Joseph A. Herriges, Ames, IA

Chapter 1

Resource Evaluation and Public Policy

The premise underlying this book is that estimates of the economic values of environmental and resource services can be a valuable part of the information base supporting resource and environmental management decisions. The importance of this premise is illustrated by a number of current environmental and resource policy issues, all of which involve in one way or another questions of economic values and tradeoffs. Consider these issues:

- Achieving the air and water pollution control objectives established by Congress requires massive expenditures on the part of both the public and private sectors. Is this diversion of resources from the production of other goods and services making us better off?
- Economists since A.C. Pigou (1929) have advocated placing taxes on emissions of air and water pollutants based on the damages they cause. What tax rate should be placed on these damages? How much do these rates vary across locations and time? An important related question is, are the gains from moving to pollution taxes greater than the costs of estimating the relevant marginal damages?
- The development of new reserves of petroleum and minerals is increasingly impinging on wild and natural areas that provide other environmental and resource services. Areas that might be affected include the Arctic National Wildlife Refuge, with its fragile habitat for caribou and other species, and the outer continental shelf, where commercial and recreational fisheries may be threatened by petroleum exploration and production. The 2010 oil spill in the Gulf of Mexico, and associated pictures on the nightly news of impacted wildlife, provided a cogent reminder that these same tradeoffs can occur in areas of existing production as well. Are restrictions on development in ecologically sensitive areas worth the costs they impose on society in the form of reduced availability of, and higher prices for, energy and minerals?
- The development and management of large river systems such as the Columbia River basin involves choosing among alternative combinations of hydroelectric power, water supply, and commercial and recreational fishing. There are also proposals to remove existing dams from many rivers. Are

the ecological and recreational benefits of removing a dam greater than the costs in the form of reduced power generation and water storage? Is it worthwhile to curb water withdrawals for irrigation or reduce discharges for power production in order to protect populations of salmon and other migratory fish?

- The commercial exploitation of some natural resource systems may be proceeding at unsustainable rates. Examples include some tropical forests and many of the world's fisheries. Shifting to sustainable rates of harvest may involve substantial short-term costs in the form of forgone incomes in order to achieve long-term increases in the flows of other ecological services. Are the long-term gains from achieving sustainable rates of harvest greater or less than the short-term costs?
- The scientific consensus is that substantial reductions in the emissions of greenhouse gasses will be required to slow or reverse the warming of the global climate. What degree of emissions reduction can be justified by the benefits of slowing or preventing global warming?
- Many people are now advocating that countries expand their system of national income accounts to include measures of the values of nonmarket environmental services, and deductions for the costs of environmental degradation and resource depletion. See, for example, Nordhaus and Kokkelenberg (1999).[1] How are these values and costs to be measured?

This book is about how, by providing measures of the economic values of the services of environmental and natural resource systems, economics as a discipline can contribute to answering questions such as these. We begin by introducing the idea of the natural environment as a set of assets or a kind of natural capital (Kareiva et al. 2011; Barbier 2011).

The Assets of Nature

Natural resources, such as forests and commercially exploitable fisheries, and environmental attributes, such as air quality, are valuable assets in that they yield flows of services to people. Public policies and the actions of individuals and firms can lead to changes in these service flows, thereby creating benefits and costs. Because of externalities and the common property and public good characteristics of at least some of these services, market forces can be relied on neither to guide them to their most highly valued uses nor to reveal prices that reflect their true social values. Externalities arise when a real variable (not a price) chosen by one economic agent enters the utility or production function of other economic agents. Inefficiencies can occur when there is no requirement to, or incentive for,

1 For information and references on green accounting, see the web page of the United Nations Environmental Program Green Accounting Resource Center at: www.unep.ch/etb/areas/VRC_index.php.

the first agent to take the effect on others into account when making choices. An example is the level of emissions of smoke chosen by an electric generating plant when that smoke causes ill health to people downwind of the plant. A public good is nonexcludable and nondepletable—that is, once the good has been provided to one individual, others cannot be prevented from making use of the good, and one person's use does not diminish the use that others can make of the good. It is the externalities and public good character of many environmental services that are responsible for the failure of the market system to allocate and price resource and environmental services correctly, and that create the need for economic measures of values to guide policymaking.

Benefit-cost analysis as the basis for making decisions about water resources investments came into its own more than 50 years ago. However, since the 1950s when the techniques of conventional benefit-cost analysis were being developed and refined, there have been significant changes in the nature of the problems being dealt with and the analytical tools that have become available. V. Kerry Smith called attention to these changes in his keynote lecture at Resources for the Future's 35th anniversary celebration in 1987. He went on to say that

> This expansion of applications has far-reaching implications for the techniques used and for the treatment of measures of the benefits and costs. Consequently, it has led me to argue for the use of a broader term, *resource evaluation*, to describe more adequately the amendments and expansions to benefit-cost methods in evaluating today's environmental and natural resource issues.
>
> (Smith 1988, 2)

One of the changes noted by Smith is the expanding range of resource and environmental management problems being subjected to economic analysis. As Smith pointed out, benefit-cost analysis was first developed to assess the net economic values of public works projects, especially water resource developments, that withdrew productive factor inputs (land, labor, capital, and materials) from the economy to produce tangible outputs (for example, hydroelectric power and transportation). Many of the outputs had market counterparts, so estimation of monetary values was relatively straightforward. For example, the savings in the monetary costs of repairing flood damages was taken to be a measure of the benefits of controlling floods. In contrast, today the effects of many public actions are much more subtle and wide-ranging. This is true for both the favorable effects (benefits) and unfavorable effects (costs and damages). What were once considered unquantifiable and perhaps relatively unimportant intangibles, such as improved recreation and visual amenities, are now recognized as significant sources of value. Also, consequences that were once unrecognized (for example, small changes in the risk of cancer) or were thought to lie outside the realm of economic analysis (say, loss of biodiversity and the preservation of endangered species and unique ecological systems), are often central issues in the analysis of policy choices today.

Another change is that the distinction between natural resources and the environment that has prevailed in the economics discipline for so long is often no longer meaningful. The objects of analysis for natural resource economists have typically been such resources as the forest, the ore body, and the fish species that produced a flow of commodities to the economy such as wood, metal, and fish sticks. The environment has been viewed as the medium through which the externalities associated with air, noise, and water pollution have flowed and, sometimes, as the source of amenities. Increasingly, this distinction appears to be artificial as we recognize both the variety of service flows provided by natural resources and the importance of a variety of forms of externalities. This recognition is apparently what Smith had in mind when he suggested the need to "model both natural and environmental resources as assets" (1988, 3) that yield a variety of valuable services. Freeman, Haveman, and Kneese had earlier suggested that we "view the environment as an asset or a kind of nonreproducible capital good that produces a stream of various services for man. Services are tangible (such as flows of water or minerals), or functional (such as the removal, dispersion, storage, and degradation of wastes or residuals), or intangible (such as a scenic view)" (1973, 20). Ecologists are now also adopting this perspective as they refer to "natural capital" and the values of ecosystem services (Prugh 1999; Daily et al. 1997; Daily et al. 2000; Daily et al. 2011).

As this change in perspective is adopted, it will be necessary to take a more expansive view of natural and environmental resources as complex systems with multiple outputs and joint products. The natural resource-environmental complex can be viewed as producing five kinds of service flows to the economy. First, as in the conventional view of resource economics, the resource-environmental system serves as a source of material inputs to the economy such as fossil fuels, wood products, minerals, water, and fish. Second, some components of the resource-environmental system provide life support services for people in the form of a breathable atmosphere, clean water, and a livable climatic regime. Changes in the flows of some of these life support services can be measured in terms of changes in the health status and life expectancies of affected populations. Third, the resource-environmental system provides a wide variety of amenity services, including opportunities for recreation, wildlife observation, the pleasures of scenic views, and perhaps even services that are not related to any direct use of the environment (sometimes called nonuse or existence values). Fourth, this system disperses, transforms, and stores the residuals that are generated as by-products of economic activity. This is usually referred to as the waste receptor service of the environment (Kneese, Ayres, and d'Arge 1970; Freeman, Haveman, and Kneese 1973). Finally, the resource-environmental system serves as a repository of genetic information that helps to determine the stability and resilience of the system in the face of anthropogenic and other shocks. Many of the services provided by natural resource-environmental systems can be characterized as direct services since their benefits accrue directly to people, for example, materials flows and life support services. Other environmental services could be better described as

indirect services in the sense that they support other biological and ecological production processes that yield value to people. Examples include recycling of nutrients, decomposition of organic materials, generation and renewal of soil fertility, pollination of crops and natural vegetation, and biological control of agricultural and other pests.

A forest, such as a unit in the U.S. National Forest system, is an example of a resource-environmental system that provides a wide range of services, from materials such as wood and fiber to amenities like scenic vistas, hiking, and wildlife observation, and from the regulation of stream flow and control of erosion to the absorption of atmospheric carbon dioxide. In addition, since trees are known to emit nonmethane hydrocarbons, at least in some circumstances forests may contribute to the impairment of the life support services (Chameides et al. 1988). In the list of service flows there are examples of joint products—that is, pairs of services that can be increased or decreased together. However, often an increase in the flow of one type of service must be accompanied by a decrease in the flow of some other service, all things being equal. In other words, this system is characterized by scarcity and tradeoffs and requires a multipurpose approach to its management (Bowes and Krutilla 1989).

The economic value of a resource-environmental system as an asset is the sum of the discounted present values of the flows of all of the services. Since many of these service flows are not bought or sold in markets and therefore do not have market prices, the economic value of a natural asset may be quite different from its market value. For example, an acre of wetland might trade in the market for land on the basis of its value for commercial or residential development; but this value could be quite different from the value of its services as wildlife habitat and as means of controlling floods and recharging groundwater aquifers. It is important to emphasize, particularly to noneconomists, that in such a case the true economic value of this wetland includes both the marketed value as well as the nonmarketed ecosystem services.

The benefit of any public policy that increases the flow of one type of service is the increase in the present value of that service. However, the policy may have costs in the form of decreases in the flows of other services. Similarly, what is termed as damage due to pollution, or some other human intervention, is the reduction in the value of the flow of services it causes. All of these changes in resource flows, whether benefits, costs, or damages, have their counterparts in changes in the value of the resource-environmental system as an asset. Some attention must therefore be devoted to the theories of asset pricing, and the role of time and discounting in calculating changes in environmental and resource values. These topics will be taken up in Chapter 6.

Some of the service flows of resource-environmental systems are linked directly or indirectly to markets, and hence are responsive to market forces. Many service flows however, are not properly regulated by markets because of externalities, their public goods characteristics of nonexcludability and nondepletability, and other factors. As is well known, this means that a decentralized market system is

unlikely to lead to the optimal pattern of service flows. Hence, there is a potential role for public policy in the management of resource-environmental systems and a need for information on the values of the service flows.

The Economic Concept of Value

The term "value" can have several different meanings. For example, economists and ecologists often use the term in two different ways in discussions of environmental services and ecosystems. One common use of the term is to mean "that which is desirable or worthy of esteem for its own sake; thing or quality having intrinsic worth" (*Webster's New World Dictionary*). In contrast, economists use the term in a sense more akin to a different definition, "a fair or proper equivalent in money, commodities, etc." (*Webster's* again), where "equivalent in money" represents the sum of money that would have an equivalent effect on the welfare or utilities of individuals.

These two different uses of the word correspond to a distinction made by philosophers between *intrinsic* value and *instrumental* value. According to philosophers, something has *intrinsic* value "if it is valuable *in* and *for* itself—if its value is not derived from its utility, but is independent of any use or function it may have in relation to something or someone else. …an intrinsically valuable entity is said to be an 'end-in-itself,' not just a 'means' to another's ends" [emphasis in original] (Callicott 1989, 131). In contrast, something has instrumental value if it is valued as a means to some other end or purpose. In this view, the value of something lies in its contribution to some other goal (Costanza and Folke 1997, 49).

Some people have argued that nature has intrinsic value for various reasons, including its "harmony" or its natural balance. However, from the perspective of the "new ecology" which emphasizes disturbance and change in ecosystems (for example, Botkin 1990), this explanation of why nature has intrinsic value is problematic. A conservation biologist might argue that the part of nature consisting of the variety of organisms and their interactions, and especially their genetic diversity, has intrinsic value. However, this view does not endow any particular manifestation of nature with more or less intrinsic value than some alternative manifestation. Nature's value is preserved as long as diversity in the broad sense is preserved. Although the concept of intrinsic value as applied to the environment is attractive in many respects, it does not provide a basis for dealing with the kinds of environmental management questions that were identified in the first section of this chapter. In contrast, the concept of instrumental value, and in particular the economic form of instrumental value, is well suited to helping answer these questions.

In order to assess the instrumental value of nature, it is necessary to specify a goal and to identify the contributions that specific components of nature make toward the furtherance of that goal. Economics is the study of how societies organize themselves to provide for the sustenance and well-being of their members. Thus in economics, the goal is increased human well-being. The economic theory of

value is based on the ability of things to satisfy human needs and wants, or to increase the well-being or utility of individuals. Sometimes this view is referred to as the preference-based account of welfare or well-being. See for example Adler and Posner (2006, 29, 33–35). Under this view of welfare, the economic value of something is a measure of its contribution to human well-being. The economic value of resource-environmental systems, then, resides in the contributions that the variety of ecosystem functions and services make to human well-being.

The economic concept of value employed here has its foundation in neoclassical welfare economics. The basic premises of welfare economics are that the purpose of economic activity is to increase the well-being of the individuals who make up the society, and that each individual is the best judge of how well off he or she is in a given situation. Each individual's welfare depends not only on that individual's consumption of private goods and of goods and services produced by the government, but also on the quantities and qualities each receives of nonmarket goods and service flows from the resource-environmental system—for example, health, visual amenities, and opportunities for outdoor recreation. It follows that the basis for deriving measures of the economic value of changes in resource-environmental systems is their effects on human welfare.

The anthropocentric focus of economic valuation does not preclude a concern for the survival and well-being of other species. Individuals can value the survival of other species not only because of the uses people make of them (for food and recreation, for example), but also because of an altruistic or ethical concern. Indeed, numerous studies strongly suggest that people do significantly value the well-being of other species and the preservation of ecosystems to their own end. The latter can be the source of existence or nonuse values, a form of economic value discussed in Chapter 4.

If society wishes to make the most (in terms of individuals' well-being) of its endowment of all resources, it should compare the values of what its members receive from any environmental change or use of a resource (that is, the benefits) with the values of what its members give up by taking resources and factor inputs from other uses (that is, the costs). A society that is concerned with the economic well-being of its citizens should make changes in environmental and resource allocations only if what is gained by the change is worth more in terms of individuals' welfare than what is given up by diverting resources and inputs from other uses.

The standard economic theory for measuring changes in individuals' well-being was developed for the purpose of interpreting changes in the prices and quantities of goods purchased in markets. This theory has been extended in the past 40 years or so to public goods and other nonmarket services such as environmental quality and health. The theory is based on the assumption that people have well-defined preferences among alternative bundles of goods, where bundles consist of various quantities of both market and nonmarket goods. The theory also assumes that people know their preferences, and that these preferences have the property of substitutability among the market and nonmarket goods

making up the bundles. By substitutability, economists mean that if the quantity of one element in an individual's bundle is reduced, it is possible to increase the quantity of some other element so as to leave the individual no worse off because of the change. In other words, the increase in the quantity of the second element substitutes for the decrease in the first element. The property of substitutability is at the core of the economist's concept of value because substitutability establishes tradeoff ratios between pairs of goods that matter to people.

Given the central role of the substitutability property in the definition and measurement of economic values, it is important to consider the evidence supporting the assumption of substitutability. This assumption is the basis of most of the models of individual choice that are used to analyze and predict a wide variety of economic behavior both in and outside of markets. These models include those of consumer demand and response to changes in prices, savings, and of the supply of labor. They also include models of a variety of individuals' behaviors related to environmental and health considerations, including participation in outdoor recreation activities, choices among jobs with varying degrees of risk of fatal accident, and choices of where to live and work when houses and urban centers offer different packages of amenities and environmental pollution. The successful development and application of these models would not be possible if substitutability was not a common feature of individuals' preferences. However, some researchers have found evidence of lexicographic preferences—preferences where substitutability is not evident. For a review and discussion of some of this evidence, see works by Common, Reid, and Blamey (1997), and Veisten, Navrud, and Valen (2006).

The tradeoffs that people make as they choose less of one good and substitute more of some other good reveal something about the values people place on those goods. If the monetary value of one of the goods is known, the revealed values can also be expressed in monetary units. The money price of a market good is just a special case of a tradeoff ratio because the money given up to purchase one unit of one element of the bundle is a proxy for the quantities of one or more of the other elements in the bundle that had to be reduced in order to make the purchase. However, even when money prices are not available, the tradeoff ratios can be interpreted as expressions of economic values. In fact, there is a growing literature exploring such tradeoff ratios. For example, Viscusi et al. (1991) asked respondents about their willingness to trade off the risk of contracting chronic bronchitis against the risk of death in an automobile accident.

Value measures based on substitutability can be expressed in terms of either willingness to pay (WTP), or willingness to accept compensation (WTA). WTP and WTA measures can be defined in terms of any good that the individual is willing to substitute for the good being valued. In the following discussion, money is used as the numeraire in which tradeoff ratios are expressed, but WTP and WTA could be measured in terms of any other good that mattered to the individual. The choice of a numeraire for measuring WTP or WTA is irrelevant in terms of its effect on how any one individual ranks alternative outcomes. However, as Brekke

(1997) has shown, the choice of a numeraire can affect the rankings of outcomes based on aggregation of welfare measures across individuals.

WTP is the maximum sum of money the individual would be willing to pay rather than do without an increase in some good such as an environmental amenity. This sum is the amount of money that would make the individual indifferent between paying for and having the improvement and forgoing the improvement while keeping the money to spend on other things. WTA is the minimum sum of money the individual would require to voluntarily forgo an improvement that otherwise would be experienced—it is the amount that would make a person indifferent between having the improvement and forgoing the improvement while getting extra money. Both value measures are based on the assumption of substitutability in preferences, but they adopt different reference points for levels of well-being. WTP takes as its reference point the absence of the improvement, while WTA takes the presence of the improvement as the base level of welfare or utility. In principle, WTP and WTA need not be exactly equal. WTP is constrained by the individual's income; but there is no upper limit on what that person could require as compensation for forgoing the improvement. Differences between WTP and WTA measures and the question of which measure is appropriate under various circumstances are discussed in more detail in Chapter 3.

Finally, we offer some words of caution based on our working in interdisciplinary research groups and serving on multidisciplinary advisory committees and panels. While semantic issues alone do contribute sizably to the misunderstandings that often arise when noneconomists are first exposed to an economist's notions of valuation, it is important to recognize that researchers from the academic disciplines that must be relied upon in valuing environmental and resource system changes do not always share the economists' commitment to the anthropocentric and preference-based perspectives on value. Nor do they all accept the assumptions of well-formed and stable preference orderings, substitutability, and rational choice that underlie our methods of economic valuation. The same thing is likely to be true of politically responsible decision-makers. Economists must be sensitive to this fact and be willing to engage in broad-ranging discussions of alternative value concepts, approaches to making choices about environmental policy, and alternative valuation methods. Economists must argue for care and rigor in applying the basic tenets of economic value in contexts in which it is important, such as benefit-cost analysis, so as to be sure that apples and oranges are not being mixed. However, there may be policy settings where other concepts may aid decision making (U.S. Environmental Protection Agency 2009). Economists who are not sensitive to these facts run the risk of being ignored in the policymaking process and undermining the value of economic analysis in the process.

Economic Values in Public Policy

There may be potential for substantial gains in economic welfare through better resource management and the judicious use of the principles of resource valuation in some cases, such as those involving the issues described at the beginning of this

chapter. If the objective of management is to maximize the net economic values associated with the use of environmental and natural resources, then benefit-cost analysis becomes, in effect, a set of rules for optimum management and a set of definitions and procedures for measuring benefits and costs. Once the objective of maximum net economic value or economic efficiency has been accepted, policy becomes an almost mechanical (but not necessarily easy) process of working out estimates of marginal benefit and marginal cost curves and seeking their point of intersection.

Most current resource and environmental policy, however, is not based solely, or even primarily, on the efficiency criterion. One reason, of course, is that at the time that many of the basic policy objectives were established, it was not within the capability of analysts to provide the kind of information about values that would be required to implement the efficiency objective. But, it is also true that decision-makers may have other objectives besides economic efficiency. For example, decision-makers may be concerned with equity considerations, intergenerational effects, the sustainability of resource systems, or social risk aversion. Thus, it is not particularly useful to advocate benefit-cost analysis as a routine or simple decision rule. Rather, as Arrow et al. (1996) argued, it should be considered as a framework and a set of procedures to help organize available information. Viewed in this light, benefit-cost analysis does not dictate choices, nor does it replace the ultimate authority and responsibility of decision-makers and open public input processes. It is simply a tool for organizing and expressing certain kinds of information on the range of alternative courses of action. It is in the context of this framework for arraying information that the usefulness of value estimates must be assessed.

Some people may be distrustful of economists' efforts to extend economic measurements to such things as human health and safety, ecology, and aesthetics, and to reduce as many variables as possible to commensurate monetary measures. Some skepticism about the economist's penchant for monetary measurement is no doubt healthy, but it should not be overdone. It is sometimes argued that there are some things like human health and safety or the preservation of endangered species that cannot be valued in terms of dollars or some other numeraire. However, the real world often creates situations where tradeoffs between such things as reducing risks of death and some other things of value cannot be avoided. Where individuals can make choices for themselves about these tradeoffs, their values can be inferred from these choices; and where government policies affecting health and safety are involved, these policy choices imply values. The questions really are how the problem of making choices about such tradeoffs is to be approached and what information can be gathered to help in the problem of choice.

Consider a hypothetical and highly simplified case of an air pollutant. Assume that the following information is known with certainty. At present levels of emissions, the pollutant causes excess mortality of 10 deaths per year in the population at risk. Reducing emissions by 30 percent would cost $5 million and would reduce the excess mortality to 5 deaths per year. Reducing emissions by 60

Table 1.1 Hypothetical example of trade-offs

Level of control (percentage)	*Costs of control (millions of dollars)*	*Excess mortality (deaths per year)*
0	0	10
30	5	5
60	15	3

percent would reduce excess mortality to 3 deaths per year but would cost $15 million. This information can be displayed as in Table 1.1.

The problem is clearly one of tradeoffs between lives and the value of resources used up in the process of controlling emissions. If the monetary value of saving lives was known, the right-hand column of the table could be converted to dollar measures of benefits, and the appropriate benefit-cost rules could be applied to determine the optimum level of emissions control. But, in the absence of some agreed-upon basis for making lifesaving and control costs commensurable in dollar terms, no simple decision rule can be applied to determine the correct choice.

Choices of this sort are made in the political realm by decision-makers such as the administrator of the Environmental Protection Agency (EPA). Whatever the choice, there is an implicit value of lifesaving that is consistent with that choice and can be said to have been revealed by that choice. In this example, if the decision-maker chooses the 30 percent control level, the value of lifesaving is revealed to be at least $1 million per death avoided. The 30 percent control level "buys" 5 lives saved at a cost of $5 million. The choice further reveals that the value of lifesaving to the decision-maker is less than $5 million, since the decision-maker declined the opportunity to "purchase" the additional 2 lives saved that the additional $10 million of control costs would make possible. If the 60 percent control level had been chosen, this would have revealed a value of lifesaving of at least $5 million.

In this example, with only three data points the implicit value can only be determined within some range. If control costs and mortality as functions of the level of control were continuous relationships and known with certainty, the choice of a control level would imply a precise value of life. If it is assumed that the control level was established so as to equate marginal benefits and marginal costs, and marginal costs are known, the marginal benefit or value can be inferred.

A number of studies have shed light on the implied value of life saving by estimating the costs per death avoided for various regulatory policies. See, for example, Cropper et al. (1992), Van Houtven and Cropper (1996), Hamilton and Viscusi (1999), and Tengs et al. (1995).

In the previous example choice revealed value, rather than value determining the choice. Either way, the problem of valuation cannot be avoided, but it can be hidden. However, in a democratic society, the more open decision-makers are about the problems of making choices and the values involved, and the more information they have about the implications of their choices, the better their

choices are likely to be. Estimates of values in monetary terms are one such source of information.

Most resource and environmental management problems have structures similar to the one just discussed. For example, in Table 1.1 the first column could be the rate of harvest in a forest, the second column the net economic value of the harvest, and the third column the probability of survival of an endangered species requiring old-growth forest habitat. For an example, see Montgomery, Brown, and Adams (1994). Reducing harvest levels increases the probability of survival of the species, but at a cost. The rate of harvest actually chosen implies something about the value to the decision-makers of increasing the probability of survival. Of course, the second column need not be in dollars. The tradeoff could be between having more timber wolves and fewer deer. The fact that there is no monetary measure in that example does not make it at its root any less an economic problem. Whatever choice is made about a population level for wolves implies something about the relative values placed on wolves and deer by the person making the decision. This example is another manifestation of the fundamental economic fact of scarcity—that is, that more of one thing means less of something else that people value.

Because policy choices about resources and environmental quality are made in a political context and are likely to involve comparisons and tradeoffs among variables for which there is no agreement about commensurate values, monetary benefit and cost data will not always be the determining factors in decision making. Benefit and cost estimates, however, are an important form of information. Their usefulness lies in the fact that they use easily understood and accepted rules to reduce complex clusters of effects and phenomena to single-valued commensurate magnitudes (that is, to dollars). The value of the benefit-cost framework lies in its ability to organize and simplify certain forms of information into commensurate measures (Arrow et al. 1996).

Classifications of Values

In this section, we describe some ways of classifying the types of environmental and resource service flows for which value measures might be desired. Of course, any classification system contains a certain element of arbitrariness. The usefulness of any particular classification depends on how well it illuminates important similarities and differences among types of service flows. Just which similarities and differences are important depends upon the particular questions being examined.

One basis for classification is the type of resource or environmental media. Environmental effects are often classified according to whether they stem from changes in air quality, water quality, land quality, and so forth. The current legal and administrative division of responsibilities for environmental management and pollution control is consistent with this basis for classification. However, it is becoming less and less relevant as cross-media effects are becoming better

understood. For example, controlling emissions of nitrogen oxides from coal-burning power plants might be part of a cost-effective strategy for improving water quality in estuaries because of the impact of nitrate deposition on nutrient levels in these waters. In addition, controls on land use are a key part of the strategy for reducing nonpoint-source water pollution.

A second type of classification is based on the economic channel through which human well-being is affected. Environmental and resource service flows can be classified according to whether they convey their effects through the market system in the form of changes in incomes to producers and changes in the availability of, and prices for, marketed goods and services to consumers, or through changes in the availability of goods and services not normally purchased through markets. Goods and services not normally purchased could be, for example, health, environmental amenities such as visibility, and opportunities for outdoor recreation. The subject matter of this book is the methods and techniques for measuring the values of these latter nonmarket services. However, many of the policies for managing environmental and resource systems will affect the flows of both market and nonmarket goods and services—so policy assessments will often need to make use of market as well as nonmarket valuation methods.

A third way of classifying environmental and resource service flows is according to whether they impinge directly on humans, indirectly on humans through their impact on other living organisms, or indirectly through inanimate systems. Direct impacts on humans include the morbidity and mortality effects of associated air and water pollutants, hazardous wastes, pesticide residues and the like, and the nonhealth effects of pollutants manifesting themselves as odors, reduced visibility, and reduced visual attractiveness of outdoor settings.

Impacts on humans involving biological mechanisms and other organisms include those on the economic productivity of both managed and natural ecosystems, such as agricultural croplands, commercial forests, and commercial fisheries. Market valuation methods would be used to value these effects. There are also impacts on nonmarket direct service flows to people such as recreational uses of ecosystems for hunting, fishing, and nature observation; and there are impacts on indirect or intermediate ecosystem services, such as pollination, decomposition, biological pest control, and nutrient recycling.

Impacts acting through nonliving systems include: damages to materials and structures and increases in cleaning and repair costs at commercial activities, which would be measured by market valuation techniques; damages to materials and structures and increases in cleaning and repair costs for households, which would be measured by nonmarket valuation techniques; and impacts on weather and climate which would be measured by either market or nonmarket valuation techniques, depending on the nature of the activity affected.

Finally, we can distinguish between those services that an individual values because they make use of them in some way (use values), and those which are valued independent of any kind of observable use. These have been called nonuse

values, or more recently passive use values. Questions of defining use and use vs. nonuse values will be taken up in Chapter 4.

Dealing with Uncertainties in Policymaking

One problem in carrying out an analysis on environmental benefits or costs is that the values for some physical, technical, or economic parameters of the model may not be known with certainty. The state of the art in measurement is not sufficiently far advanced to produce exact measures of value for many kinds of environmental and resource changes. This leads to the question: must policymakers wait for further research to produce exact measures before they can use value and benefit information to guide decision making? If not, how should they interpret the ranges of values that current research has produced?

To counsel waiting for exact measures is equivalent to saying that in many cases value measures should never be used. The state of the art cannot be expected to advance to the point of producing exact values for all kinds of environmental change. This is because of the inherent uncertainty and imprecision in measurement techniques based on statistical inference, and because of the fact that the true values held by individuals will vary with their circumstances (age, income, and so forth) and with the description of the specific changes being valued. So how are policymakers to proceed in the face of continued and inherent uncertainty about values?

The simplest approach is to base the calculations of benefits and costs on the expected values of the uncertain parameters and to base decisions on these expected values. However, decision-makers will often want to know more about the magnitude of the uncertainties in the estimates of benefits and costs. They could be provided with the upper and lower bounds of the ranges of values along with the expected values. Clearly, if the benefits of a policy calculated with the upper end of the range are less than the lower end of the range of estimated costs, the policy is unlikely to be justifiable on economic grounds; and if the benefits calculated with the lower end of the range exceed the upper end of the range of costs, the economic case for the policy is quite strong.

This simple-minded approach is a step in the right direction, but it can be criticized because it does not make use of all of the relevant information contained in the estimates making up the range of values. Formally, the range reflects only the information contained in the two estimates yielding the highest and lowest values. It ignores information on the quality of these two estimates, and it ignores all of the information contained in the other estimates that yielded values within the range. There is a way to make use of the results of all of the available estimates and to incorporate judgments about the quality of each of these estimates. This formal approach is based on viewing probabilities as statements about the degree of confidence held about the occurrence of some possible event. The approach involves assigning probabilities to all of the values produced by the available estimates, where a higher probability reflects a greater degree of confidence in

that estimate. For example, the assignment of a probability of one to a particular estimate means we can be certain that this study has produced the correct value. Once the probabilities have been assigned, various statistical manipulations can be performed. For example, the expected value of the parameter in question (the mean of its probability distribution) can be calculated and used for benefit-cost calculations. The variance of the distribution can be used to determine confidence intervals on the value to be used, thus preserving for policymakers information on the uncertainty about values; or, when there are multiple uncertainties, Monte Carlo methods can be employed to draw from the assumed distributions to generate a probability distribution of outcomes.

Ex Post and Ex Ante Analysis of Values

The decision-maker who is trying to allocate scarce resources and is faced with a number of competing goals needs ex ante analyses of the effects of alternative policy actions to guide decision making. Ex ante analysis involves the prediction of the physical and economic consequences of policies on the basis of a model of the physical and economic processes involved. It involves visualizing two alternative states of the world, one with the policy in question and one without, and then comparing these alternative futures in terms of some established criterion such as net economic efficiency. Ex post analysis of a policy involves measuring the actual consequences of the policy by comparing the observed state with a hypothetical alternative—the state of the world without the policy. Ex post analysis, in effect, treats the policy as a controlled laboratory experiment except that the control is hypothetical rather than real. Natural resource damage assessment is an example of ex post evaluation, in that the damaged state is observed and must be compared with a hypothetical or counterfactual alternative in which there was no pollution event, but other relevant factors are assumed to have remained unchanged.

Ex post and ex ante analyses are not competitive alternatives. Rather, they should be viewed as complementary techniques for improving our knowledge. An ex post analysis of a policy can be viewed as a check on the validity of the ex ante analysis. The ex ante analysis is a prediction of what will happen; the ex post analysis is a check of what actually did happen.

It is particularly important that the economic analysis of environmental and resource policies includes ex post analysis. Our knowledge of the physical and economic systems on which present ex ante analyses are based is extremely limited. It is necessary not only to develop more comprehensive models of the physical, biological, and economic aspects of the system, but also to devote more effort to verifying these models through ex post comparisons of the predictions with observed results.

Although ex ante analyses of environmental, health, and safety regulations have become quite common in the United States (Hahn 2000), ex post analyses of environmental and resource policies are quite rare. An early study of the realized

benefits and costs of U.S. Army Corps of Engineers' water resource development projects was done by Haveman (1972). A notable and controversial study was U.S. EPA's retrospective analysis of the benefits and costs of the Clean Air Act (U.S. Environmental Protection Agency 1997).[2]

It must be emphasized that the ex post verification of the analytical models used in resource valuation is not simply a comparison of actual results with predictions. Ex ante models are based upon some view of the future with projections of economic magnitudes such as population levels, real income, and price levels. Care must be taken in ex post analysis to sort out the effects of unforeseen developments, such as war or uncontrolled inflation, on the variables in question. For example, if the failure of income levels to rise on the projected path results in a shortfall of recreation benefits at a particular site, this is not a failure of the analytical model so much as a reflection on our inability to perceive the future. The real benefit of ex post analysis is in making the most of the opportunity to improve on the analytical models used.

Preview

The major task of this book is to review and summarize the basic theory of economic welfare measurement, and to present resource valuation and benefit measurement techniques that are consistent with this underlying theory. The next chapter provides an overview of valuation and welfare measurement methods and discusses the relationship between the economic methods of valuation and the physical and biological relationships that define the resource and environmental systems being valued.

Chapters 3 and 4 constitute the theoretical core of the book. Chapter 3 lays out the basic premises and value judgments that underlie the economic concept of benefits and presents the basic theory of the measurement of economic welfare changes. Chapter 4 introduces the basic methods and models for deriving welfare and value measures from the revealed choices of individuals, from observed changes in market prices, and from individuals' responses to hypothetical questions. These so-called stated preference methods, including willingness-to-pay surveys and bidding games, direct referendum questions, and questions about how individuals would rank alternative bundles of environmental and private goods are described in more detail in Chapter 12.

Chapter 5 extends the theory of value and welfare change to a situation of risk where people are uncertain about what the actual state of the world will be. Public policies toward the management of environmental resources can affect either the probabilities of alternative outcomes or the magnitudes of environmental services in alternative states of the world. Thus, this chapter describes the application of the basic theory of welfare change to evaluating environmentally induced changes

2 For some of the controversy generated by this report, see Crandall (1997), Lutter and Belzer (2000), and Brown et al. (2001).

in risks. Chapter 6 takes up the question of valuation across time and the role of discounting in welfare measurement.

The remaining chapters describe the application of the various models and methods for welfare measurement to specific situations such as the values of reducing the risk of premature mortality and improving human health (Chapter 7), the values of environmental changes affecting producers' costs and productivity (Chapter 8), the valuation of resources that support recreation activities (Chapter 9), applications of the hedonic price model to housing prices and wage rates (Chapters 10 and 11), and a more detailed treatment of stated preference methods (Chapter 12). Chapter 13 provides an overview of some recent developments in the literature, including benefits transfer methods, combining revealed and stated preference data, some of the implications of the relatively new field of behavioral economics, and the valuation of ecosystem services. Chapter 14 offers some conclusions and identifies areas where additional research is needed.

Mathematical Notation

In this book, we use the following conventions regarding mathematical notation, with exceptions noted where they occur:

- Vectors are represented by boldface uppercase letters; lowercase letters with or without subscripts represent values for individual variables in these vectors, for example, $\boldsymbol{X} = x_1, \ldots, x_i, \ldots, x_N$.
- The subscript letters i, j, k, m, and n and subscripted numbers index elements of vectors.
- The meanings of other subscripted letters are specified when they are first used.
- Superscripted letters are used to index such things as utility functions and production functions to specific individuals and firms. For example, $u^i(\boldsymbol{X})$ gives individual i's utility as a function of that individual's consumption of goods $x_1, \ldots, x_j, \ldots, x_J$.
- Primes and superscripted numerals represent specific values for variables. For example, M^0 represents the initial value of the variable M. Similarly, $\Delta x = x'' - x'$ means the change in x from x' to x''.
- Uppercase letters represent variables expressed as quantities of money. For example, M represents income, and CV is the compensating variation measure of welfare change.

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Chapter 2

Measuring Values, Benefits, and Costs

An Overview

In the first section of this chapter, a simple general equilibrium model of an economy with nonmarket environmental and resource service flows is presented to show how the values of these flows emerge as shadow prices from the solution to a welfare maximization problem. The major types of methods for estimating values and benefits are then presented in the second section. The third section discusses some theoretical frameworks used in revealed preference models. The fourth section presents a simple model to illustrate the relationships between physical and biological changes in environmental and resource systems and the changes in well-being and values realized by the people affected by these changes. These models help to make clear the kinds of data required to carry out resource evaluations, and to show the roles of economists and physical and biological scientists in the evaluation process. The fifth section describes some issues about the noneconomic foundations of resource valuation. The chapter concludes with a discussion of the nature of cost in welfare theory.

Resource Values as Shadow Prices

Economic values can only be defined in terms of some underlying criterion that identifies what is to be considered good. As discussed in Chapter 1, in neoclassical welfare economics good is defined in terms of the well-being of individuals. Here, it is assumed that an individual's well-being can be represented by an ordinal utility function. Of course, this assumption is only a first step in defining what is good because it does not deal with interpersonal comparisons. Specifically, it does not answer the question of whether it is good when one individual's utility increases while another individual's utility decreases.

The concept of *Pareto optimality* is the route economists often choose for dealing with this set of problems. An allocation of resources, goods, and services in an economy is Pareto optimal if there is no feasible reallocation that can increase any one person's utility without decreasing someone else's utility. Of course, there are an infinite number of Pareto optimum allocations for an economy, each with a different distribution of utilities across individuals. In order to rank the allocations it is necessary to have a social welfare function that aggregates the utilities of the

individuals, perhaps by assigning social welfare weights. If such a social welfare function exists, Pareto optimality is a necessary, but not sufficient, condition for maximizing that function. Hence, despite its limitations, Pareto optimality has usefulness in evaluating economic outcomes.

Pareto optimality is the solution to a constrained maximization problem in which some of the constraints are the exogenously determined environmental and resource service flows. The shadow prices on these constraints are the economic values of these service flows. In the general equilibrium model used here, it is clear that the values assigned to the environmental and resource services flows are not fixed parameters, but are determined by their roles in enhancing individuals' well-being, and arise from their scarcity or limited availability. This model is a variation of the general equilibrium externalities model presented in Baumol and Oates (1988).

In this model the subscript j indexes a vector of $\mathcal{J}$ commodities ($j = 1, \ldots, \mathcal{J}$), i indexes the $\mathcal{N}$ individuals in the economy, and k indexes the K firms. Each of the commodities is divisible with well-defined and enforceable property rights. The variable names for these commodities reflect their role in economic activity. $\boldsymbol{X}$ represents the vector of commodities used as consumer goods by individuals. Specifically, let x_{ji} = the amount of commodity j consumed by individual i, ($j = 1, \ldots, \mathcal{J}$) ($i = 1, \ldots, \mathcal{N}$). $\boldsymbol{Y}$ represents the vector of commodities being produced by firms, where negative values indicate commodities being used as resource inputs to production. Specifically, y_{jk} = the amount of commodity j produced (used) by firm k, ($j = 1, \ldots, \mathcal{J}$) ($k = 1, \ldots, K$). There is also an endowment of commodities represented by $\boldsymbol{S}$: s_j = the total of endowment of commodity j available to the economy ($j = 1, \ldots, \mathcal{J}$).

For each commodity, the initial endowment plus the quantities produced by firms must just equal the sum of the quantities used as inputs by firms and consumed by individuals. This requirement is expressed in a set of production-consumption constraints:

$$s_j + \sum_{k=1}^{K} y_{jk} - \sum_{i=1}^{\mathcal{N}} x_{ji} = 0 \qquad j = 1, \ldots, \mathcal{J}. \tag{2.1}$$

In addition, let there be a resource service flow, r. The endowment of r is determined exogenously and is both nonrival and nonexcludable; hence, it is a public good. For simplicity, assume that r does not vary across space. This means that r takes the same value for all individuals. Finally, let d be some environmental quality parameter, say the concentration of a pollutant at a specific point in space. The level of d will depend on the discharges of pollutants by firms and will determine the quantities of pollution experienced by each individual. These relationships will be specified below.

Individuals' preferences can be represented by utility functions:

$$u^i = u^i(\boldsymbol{X}_i, r, c_i) \quad i = 1, \ldots, \mathcal{N} \tag{2.2}$$

Let $\partial u^i / \partial x_{ji} \geq 0$, $\partial u^i / \partial r \geq 0$, and $\partial u^i / \partial c_i < 0$, where c_i is the concentration of the pollutant to which the ith individual is exposed and $\boldsymbol{X}_i K(x_{1i}, \ldots, X_{\mathcal{J}i})$.

The production side of the economy can be represented by a set of production functions for multiproduct firms:

$$f^k(y_{1k}, \ldots, y_{jk}, \ldots, y_{Jk}, r, d_k) = 0 \quad k = 1, \ldots, K, \tag{2.3}$$

where y_{jk} is the production of good j by firm k, r is the level of the nonmarket resource service flow, and d_k is the firm's contribution to the environmental quality parameter d as represented by, for example, emissions of a pollutant. Let $\partial f^k/\partial y_{jk} \geq 0$, $\partial f^k/\partial r \leq 0$, and $\partial f^k/\partial d_k \leq 0$, which implies that both r and d_k can be viewed as inputs into the production process.

The variable d is some function of the emissions of all firms. For simplicity, let $d = \sum_{k=1}^{K} d_k$. Also, assume that $c_i = \alpha_i \times d$, where α_i relates each individual's exposure to the pollutant to the aggregate level of pollution, d. For a perfectly mixed unavoidable pollutant, $\alpha_i = 1$ for all i. It would also be possible to represent averting behavior by making α_i a choice variable, perhaps a function of some x_{ji}.

The Pareto optimum conditions for this economy can be found by determining the conditions that maximize each individual's utility, subject to the production function and resource constraints and the constraint that all other individuals' utilities are held constant at some level u^{i*} representing the status quo. This procedure makes it clear that any Pareto optimum allocation is only as good as our judgment about the associated distribution of utilities across individuals. For individual one, the problem is to choose the values for $\boldsymbol{X}_{j1}$ and d_1 so that

$$\begin{aligned} \max L_1 = u^1(\cdot) &+ \sum_{i=2}^{N} \lambda_i \left[u^i(\cdot) - u^{i*}\right] - \sum_{k=1}^{K} \mu_k f^k(\cdot) \\ &+ \sum_{j=1}^{J} \rho_j \left(s_j + \sum_{k=1}^{K} y_{jk} - \sum_{i=1}^{N} x_{ji}\right) + \gamma\left(r^* - r\right) \end{aligned} \tag{2.4}$$

where r^* is the exogenously determined quantity of r. Assuming an interior solution and letting $\lambda_1 = 1$, the first-order conditions for a Pareto optimum are:

$$\lambda_i \left(\frac{\partial u^i}{\partial x_{ji}}\right) - \rho_j = 0 \Rightarrow \lambda_i \left(\frac{\partial u^i}{\partial x_{ji}}\right) = \rho_j \qquad i = 1, \ldots, N; j = 1, \ldots, J \tag{2.5}$$

$$-\mu_k \left(\frac{\partial f^k}{\partial y_{jk}}\right) + \rho_j = 0 \Rightarrow \mu_k \left(\frac{\partial f^k}{\partial y_{jk}}\right) = \rho_j \qquad k = 1, \ldots, K; j = 1, \ldots, J \tag{2.6}$$

$$\sum_{i=1}^{N} \lambda_i \left(\frac{\partial u^i}{\partial r}\right) - \sum_{k=1}^{K} \mu_k \left(\frac{\partial f^k}{\partial r}\right) - \gamma = 0 \tag{2.7}$$

$$\sum_{i=1}^{N} \lambda_i \left(\alpha^i \frac{\partial u^i}{\partial d}\right) - \sum_{k=1}^{K} \mu_k \left(\frac{\partial f^k}{\partial d_k}\right) = 0 \tag{2.8}$$

where $\lambda_i > 0$, $\mu_k > 0$, $\rho_j > 0$, and $\gamma > 0$.

Conditions (2.5) and (2.6) generate the marginal rates of substitution and marginal rates of transformation for the marketed goods. They can be interpreted as saying either that each individual's marginal valuation of each good must equal the marginal cost to firms of its production, or that each individual's marginal rate of substitution between any pair of goods must equal each firm's marginal rate of transformation between that pair of goods.

In condition (2.7), the Lagrangian term γ is the shadow value for r^* (that is, it gives the increase in the objective function for a marginal increase in the constrained resource service r). Condition (2.7) also shows that the shadow price depends on the utility and production functions. Specifically, the shadow price is equal to the sum of the marginal values attached to r by all individuals and producers in the constrained solution.

The first term in (2.8) is the value to individuals of reducing firm k's emissions by one unit. Condition (2.8) says, in effect, that Pareto optimality requires that the aggregate value of reducing the emissions of firm k must be just equal to the marginal cost of that reduction (the second term in the equation). The second term in (2.8) can also be interpreted as the marginal value to the kth firm of being able to emit one more unit to the environment—that is, it is the marginal value of the waste receptor services of the environment.

Conditions (2.7) and (2.8) give the marginal values for changes in r and d_k. They also imply the existence of aggregate marginal willingness-to-pay curves that give the marginal willingness to pay for the service flows as functions of the quantities of the flows being supplied. As is shown in Chapter 3, willingness to pay can be taken to be equal to the area under such a marginal willingness-to-pay curve. Value estimation, then, involves determining directly or indirectly the shapes of these marginal willingness-to-pay curves for environmental services.

If the services of the environment could be purchased in a perfectly functioning market, there would be observable demand functions for them, making estimation of the marginal willingness-to-pay curves a standard econometric problem. Also, then, environmental and resource management would not be an important public policy matter. However, environmental and resource service flows typically have characteristics such as nonexcludability and nonrivalry in consumption, which make it difficult or impossible for markets for these services to function well. Often, individuals are not free to vary independently the level of the services that they consume. The public good character of environmental services then leads to market failure; and without a market, there are no price and quantity data from which the demand relationships can be estimated.

To sum up, the economic values of r and of reducing d emerge as part of the solution to the welfare maximization problem. These values are context dependent in the sense that changes in preferences (equation 2.2), the production technology (equation 2.3), or the resource endowments and constraints will affect these values. This point is worth emphasizing: just as the values of private goods vary across individuals, so too will the values of nonmarket goods. The value of an improvement in water quality or the reduced risk of illness may be quite different

across individuals due to differences in preferences and/or source constraints. There will rarely be a single value associated with a nonmarket good.

This context dependence also means that if the economy is not at a Pareto optimum allocation as defined by equations (2.5) through (2.8), for example because of an additional constraint, then the shadow prices or values attached to r and d will be different.

Methods for Measuring Values

One principle distinction among methods for valuing changes in environmental goods is based on the source of the data (Mitchell and Carson 1989, 74–87). The data can come either from observations of people acting in real-world settings where people must live with the consequences of their choices or from people's responses to hypothetical questions of the form "what would you do if … ?" or "how much would you be willing to pay for … ?" It is common in the literature to refer to these as *revealed preference* and *stated preference* methods, respectively.

Revealed preference methods are based on actual behavior reflecting utility maximization subject to constraints. One type of revealed preference method is based on observed choices in a "take-it-or-leave-it" setting. For example, a survey might collect information from a family about whether it visited a particular environmental attraction on the previous weekend. Essentially, the family faced a take-it-or-leave-it decision with respect to that visit—either they took the visit and enjoyed the site, or they stayed home and did something else. If the family chose to take the trip, this information "reveals" that the value of the trip exceeded the costs that the family undertook to experience the attraction. In this case, the reported behavior reveals only whether the value of the offered good to the individual was greater or less than the offering price (the cost of admission and travel). Because of the information limitations provided by such data, it is typically necessary for analysts to make assumptions about preferences so that models can be estimated with the data. For this reason (and others discussed below), values from revealed preference methods are subject to limitations.

In many instances, the environmental service does not have a direct offering price, but sometimes its quantity does affect the choices people make about other things such as quantities of market goods. In these cases, the value of the environmental service can be inferred through the application of some model of the relationship between market goods and the environmental service. Most such models are based on the assumption of some kind of substitute or complementary relationship between the environmental service and marketed goods and services. Examples of these models include the household production model (which includes models of household spending on cleaning and on repair of materials damaged by air pollution), the travel cost demand model for visits to a recreation site, and the hedonic property value and hedonic wage models. Revealed preference methods involve a kind of detective work in which clues about the values individuals place on environmental services are pieced together from the evidence that people leave

behind as they respond to prices and other economic signals. The basic properties of these models are discussed in Chapter 4.

The principal difference between revealed preference and stated preference methods is that the latter draw their data from people's responses to hypothetical questions rather than from observations of real-world choices. The earliest stated preference techniques involved asking people directly about the values they place on environmental services by creating, in effect, a hypothetical market. As the responses are contingent upon the specific conditions laid out in the hypothetical market, this form of stated preference methods is broadly referred to as *contingent valuation*. A variety of elicitation formats are possible. For example, people could be directly asked what value they place on a specified change in environmental services. The responses, if truthful, are direct expressions of value, and would be interpreted as measures of willingness to pay. Because the format allows the respondent to provide any possible measure of value, the term *open-ended contingent valuation method* is conventionally used to refer to approaches based on this form of questioning.

While a variety of contingent valuation elicitation formats have been considered, the most popular today is the *dichotomous choice referendum format*, which asks for a yes or no answer to the question: "Given a cost to you of $\$X$, would you vote in favor of a referendum that would achieve the following changes … ?" Responses to such questions reveal only an upper bound (for a no) or a lower bound (for a yes) on the relevant welfare measure. Discrete choice methods applied to a large sample of individual responses can be used to estimate willingness-to-pay functions or indirect utility functions from data on responses and on the characteristics of the individuals in the sample.

In a second type of stated preference question, known as *contingent behavior* questions, individuals are asked how they would change the level of some activity in response to a change in an environmental amenity. If the activity can be interpreted in the context of some behavioral model, such as an averting behavior model or a recreation demand model, the appropriate revealed preference valuation method can be used to obtain a measure of willingness to pay, as if the reported behavioral intentions were actual behaviors. McConnell (1986), for example, applied a recreation demand model to questions of the form "how often would you visit these beaches if they were free of PCBs?" in order to estimate the damages (resource value lost) from the pollution of the waters of New Bedford Harbor, Massachusetts, with polychlorinated biphenyls.

In a third form of stated preference question, respondents are given a set of hypothetical alternatives, each depicting a different situation with respect to the available environmental amenities and other characteristics, and are asked to rate or rank the alternatives in order of preference, or to simply pick the most preferred alternative. Several names have been applied to variations of this approach including *attribute-based stated choice* and *choice experiments*. The rankings or choices can then be analyzed to determine, in effect, the marginal rate of substitution between any other characteristic and the level of the environmental amenity. If

one of the other characteristics is a monetary price, then it is possible to compute the respondent's willingness to pay for the good on the basis of the ranking of alternatives. For a more complete discussion, see Holmes and Adamowicz (2003).

Some issues and problems in stated preference methods are specific to the particular form of the question being asked. For example, when people are asked how much they would be willing to pay for something, they might say "zero" because they reject the idea of having to pay for something they consider to be rightfully theirs. Other problems are generic to all methods based on hypothetical questions, for example, problems in scenario specification, sampling, and item nonresponse. The major questions regarding all stated preference methods concern the validity and reliability of the data; that is, whether the hypothetical nature of the questions asked inevitably leads to some kind of bias or results in so much "noise" that the data are not useful for drawing inferences. Further discussion of these questions is left to Chapter 12.

The Methodology of Revealed Preference Models

Because the revealed preference methods for measuring values use data on observed behavior, some theoretical framework must be developed to model this behavior, and to relate the behavior to the desired monetary measures of value and welfare change. A key element in the theoretical framework is the model of the optimizing behavior of an economic agent (individual or firm) that relates the agent's choices to the relevant prices and constraints, including the level of environmental or resource quality q. If a behavioral relationship between observable choice variables and q can be specified and estimated, this relationship can be used to calculate the marginal rate of substitution between q and some observed-choice variable with a money price tag, thereby revealing the marginal value of changes in q.

Welfare measurement in the case where changes in q affect individuals involves three steps. The first is to derive the expression for willingness to pay as a function of the environmental variable, usually either from the indirect utility function or the expenditure function. This expression gives the compensating change in income that holds utility constant for the change in the environmental parameter. The second step is to develop a model of individual utility maximizing behavior that relates the individual's choices to the relevant prices and constraints, including the level of environmental quality. The first-order conditions for optimization can then be derived. These first-order conditions involve equating measures of marginal value to price, or equating some marginal rate of substitution or marginal rate of transformation to some price ratio. The third step is to examine the model to see whether the first-order conditions include a relationship between the desired marginal value for the environmental change and some observable variable. If they do, then the observable variable can be taken as a measure of the marginal change in welfare. Several types of models using this methodology are presented in Chapter 4.

Drawing inferences about the marginal values of environmental changes can be challenging. However, policymakers often need even more information than this. Often, they will need estimates of values for nonmarginal changes. Deriving these estimates is considerably more difficult since what are needed are not the marginal values but the marginal value *functions* (see for example Kuminoff, Smith, and Timmins 2013 and Phaneuf, Carbone, and Herriges 2009). More will be said about how these might be obtained in subsequent chapters.

The measures of value and welfare change derived from optimization models often produce results that are quite different from those of the naive models employed in the early literature on value and benefit measurement. The early models were often based on what has come to be known as the *damage function approach*. The damage function approach involves estimating some physical relationship between a measure of environmental quality (or its converse, pollution) and some physical measure of damage or loss (such as number of workdays lost to sickness in the case of health, or percentage of crop lost in the case of effects of air pollution on agricultural productivity). Then some unit price is applied to the physical measure to convert it to monetary terms. In some studies, for example, lost wages and medical costs were used to determine the value of avoiding one day of illness induced by air pollution. Similarly, the market price of a crop was often used to determine the value of lost productivity. The benefit of a pollution control program would be estimated as the reduction in the damages calculated according to this damage function approach.

The damage function approach can be considered naive for at least two reasons. First, this approach implies a model of the world in which behavioral and market responses to changes in q are implicitly ruled out. Farmers, in fact, can adjust to changes in air pollution by changing cultivation practices, shifting to more resistant cultivars of the same crop, or even changing to entirely different crops that are less sensitive to pollution. Furthermore, the prices of agricultural crops may change because of changes in crop supplies. It may be that the changes in prices are of greater significance to human welfare than the changes in physical yields of crops. Similarly, people can choose defensive or mitigating activities in response to air pollution that affects health. These behavioral changes result in welfare consequences that have a monetary dimension, which should be taken into account in calculating values.

In addition to behavioral changes, a second problem with some of the early approaches is that they sometimes used the costs of treating an externality as the value of avoiding the externality. This is still a common practice in some health studies where lost wages and/or costs associated with medical care are used to estimate the value of avoiding the morbidity or mortality. This correspondence is true only in some cases (Bockstael and McConnell 2007, ch. 8) and will be discussed in greater length in Chapter 7.

A Model of Environmental and Resource Values

Although measuring values involves the use of economic theory and technique, value measures must be based on other types of knowledge. For example, estimates of the value of a salt marsh in sustaining a marine fishery must be based on knowledge of the biological and ecological links between the marsh and the exploited fish species. Estimates of the health benefits from air pollution control must be based on scientific knowledge of the relationship between pollutant concentrations and human health; and estimates of the recreational fishing benefits stemming from water pollution control require knowledge of the relationships among pollutant levels, biological productivity, and anglers' activities (Freeman 1995). Lack of knowledge of these relationships may be, in some instances, a major barrier to empirical measurement of values. This section lays out a very simple model for examining the relationship between the economic concept of value and the physical and biological dimensions of the resource system being valued. The model helps to make clear that economic valuation requires some knowledge of the underlying physical and biological relationships that determine the quantity and quality of environmental and resource service flows.

The economic values of the service flows from a resource-environmental system can be viewed as the product of three sets of functional relationships. The first relates some measure of environmental or resource quality to the human interventions that affect it. Let q represent a qualitative or quantitative measure of some environmental or resource attribute. Examples include the biomass of some species of fish of commercial or recreational interest, the stock of standing timber in a forest, or the concentration of some pollutant in the atmosphere. Two kinds of human intervention need to be specified. One involves the unregulated activities of the market economy (for example, the commercial exploitation of a fishery or the discharge of pollutants into the air), and these will be left implicit in the relationships presented here. The other kind of intervention is government actions taken to prevent or ameliorate the adverse impacts of unregulated market activities, or to protect or enhance the value of market and nonmarket services provided by the environment. Let $\boldsymbol{G}$ represent the set of government interventions. For example, if q represents the population of waterfowl, $\boldsymbol{G}$ could be the stock of protected habitat and breeding grounds. Alternatively, $\boldsymbol{G}$ could represent a set of regulations designed to attain a stated ambient air quality standard, or it could be a management plan for a national forest. Let us represent the relationship between q and $\boldsymbol{G}$ as

$$q = q(\boldsymbol{G}). \tag{2.9}$$

As discussed below, this relationship could be quite complex in its spatial and temporal dimensions.

Where the government regulates private activities that influence q, the effect of a change in $\boldsymbol{G}$ on q can depend in complex ways on the responses of private

decision-makers to the public regulation. The most obvious example of this dependence is the question of compliance with pollution control regulations. For any given $\boldsymbol{G}$, q increases as the degree of compliance with the regulation increases. Some public regulations by their very nature only indirectly link to the relevant environmental quality measure. An example is the automotive emissions standards set under the Clean Air Act, which regulate emissions of certain pollutants in grams per mile traveled by the automobile. In this case, the effect of a change in $\boldsymbol{G}$ on q depends also on how automobile use is affected. To take account of these complexities, it might be more appropriate to write q as a function of both $\boldsymbol{G}$ and some measure of private responses to the government regulation, $\boldsymbol{R}(\boldsymbol{G})$:

$$q = q[\boldsymbol{G}, \boldsymbol{R}(\boldsymbol{G})]. \tag{2.9′}$$

The second set of functional relationships involves the human uses of the environment or resource and their dependence on q. Let $\boldsymbol{X}$ represent the levels of some set of activities involving use of the environment or resource. For example, $\boldsymbol{X}$ could be days of recreational activity on some water body, tons of fish caught from some commercially exploited fishery, and, where human health depends on the level of environmental quality, some measure of health status. Typically, the level of $\boldsymbol{X}$ will depend not only on q, but also on the inputs of labor, capital, and other materials and resources including time, and these will also depend on q. For example, if $\boldsymbol{X}$ is agricultural output and q is air pollution, farmers might adjust to changes in pollution by changing inputs of water, fertilizer, or labor. Let $\boldsymbol{Y}$ represent these other inputs into the production of environmental services or activities based on the resource. The second functional relationship can be written as

$$\boldsymbol{X} = \boldsymbol{X}\,[q, \boldsymbol{Y}(q)]. \tag{2.10}$$

This expression ignores possible feedbacks from $\boldsymbol{X}$ to q such as when agricultural output is associated with increases in water pollution from pesticide or fertilizer runoff.

The third set of functional relationships gives the economic value of the uses of the environment. Let V represent the money value of the flow of services or activities based on the environment or resource. The relationship

$$V = V(\boldsymbol{X}) \tag{2.11}$$

embodies whatever value judgments society has adopted for economic welfare purposes. Here it is assumed that the value function is a simple aggregation of individuals' values; but $V(\boldsymbol{X})$ could also incorporate social welfare weights that reflect some social equity goals. Alternatively, $V(\boldsymbol{X})$ could incorporate concepts of environmental ethics or social norms. Also, this expression could incorporate nonuse values as in $V = V(\boldsymbol{X}, q)$.

By substitution of (2.9′) and (2.10) into (2.11), we have

$$V = V(\boldsymbol{X}\{q[\boldsymbol{G}, \boldsymbol{R}(\boldsymbol{G})], \boldsymbol{Y}[\boldsymbol{G}, \boldsymbol{R}(\boldsymbol{G})]\}). \tag{2.12}$$

The marginal value of the change in $\boldsymbol{G}$ can be found by taking the total derivative of equation (2.12) to reflect the private adjustments of $\boldsymbol{R}$ and $\boldsymbol{Y}$ to the public intervention. The benefits of a nonmarginal change in policy intervention that increases $\boldsymbol{Y}$ are given by

$$\begin{aligned} B \equiv \Delta V = V\left(\boldsymbol{X}\left\{g\left[\boldsymbol{G}^1, \boldsymbol{R}\left(\boldsymbol{G}^1\right)\right], \boldsymbol{Y}\left[\boldsymbol{G}^1, \boldsymbol{R}\left(\boldsymbol{G}^1\right)\right]\right\}\right) \\ -V\left(\boldsymbol{X}\left\{g\left[\boldsymbol{G}^0, \boldsymbol{R}\left(\boldsymbol{G}^0\right)\right], \boldsymbol{Y}\left[\boldsymbol{G}^0, \boldsymbol{R}\left(\boldsymbol{G}^0\right)\right]\right\}\right) \end{aligned} \tag{2.13}$$

where $\boldsymbol{G}^0$ and $\boldsymbol{G}^1$ are the pre- and post-policy levels of intervention and $\boldsymbol{R}$ and $\boldsymbol{Y}$ are optimally adjusted to the change in $\boldsymbol{G}$.

The set of relationships represented by equation (2.9′) is largely noneconomic in nature because it involves a variety of physical and biological processes. However, there is a social science or behavioral component to the private responses to $\boldsymbol{G}$. The relationship reflected in equation (2.11) is wholly within the realm of economics because it involves the theory of economic welfare and the use of economic data. The set of relationships reflected in equation (2.10) represents the interface between the natural science and social science disciplines. Some aspects of these relationships, for example how recreation use varies with changes in water quality, are primarily behavioral or social. Other aspects are almost wholly physical or biological, as in the effects of air pollution on human health and mortality. However, even here, to the extent that people can "defend themselves" against the adverse effects of air pollution (say, by purchasing home air purifiers), or mitigate the symptoms of illness induced by air pollution, behavioral relationships are embedded in equation (2.10). The effect of an air pollutant on a particular type of vegetation is also a biological question; but if farmers alter crop patterns as a way of adapting to changes in air pollution, then the behavioral and biological aspects of the relationship must be considered together.

Figure 2.1 illustrates the three sets of relationships for the case of the benefits associated with an improvement in ambient water quality. Varieties of substances are discharged into water bodies. Reductions in the discharges can affect physical, chemical, and biological indicators of water quality, such as dissolved oxygen, temperature, algae levels, and fish populations (stage 1). Changes in the indicators can be predicted with water quality models. The resulting water quality, as measured by the indicators, can in turn affect human uses of the water body (stage 2). These could include either withdrawal uses (for example, for industrial or municipal water supply or irrigation) or instream uses (for example, for fishery production or recreation). The major difficulty at stage 2 arises from the fact that only rarely is the level of use a simple function of a single water quality indicator like dissolved oxygen. Rather, some uses (as in the case of commercial fisheries and recreation) depend in complex ways on the whole range of physical, chemical, and biological water quality indicators. The feedback loops from stage 2 to stage 1 reflect possible impacts of changes in human uses on measures of water quality,

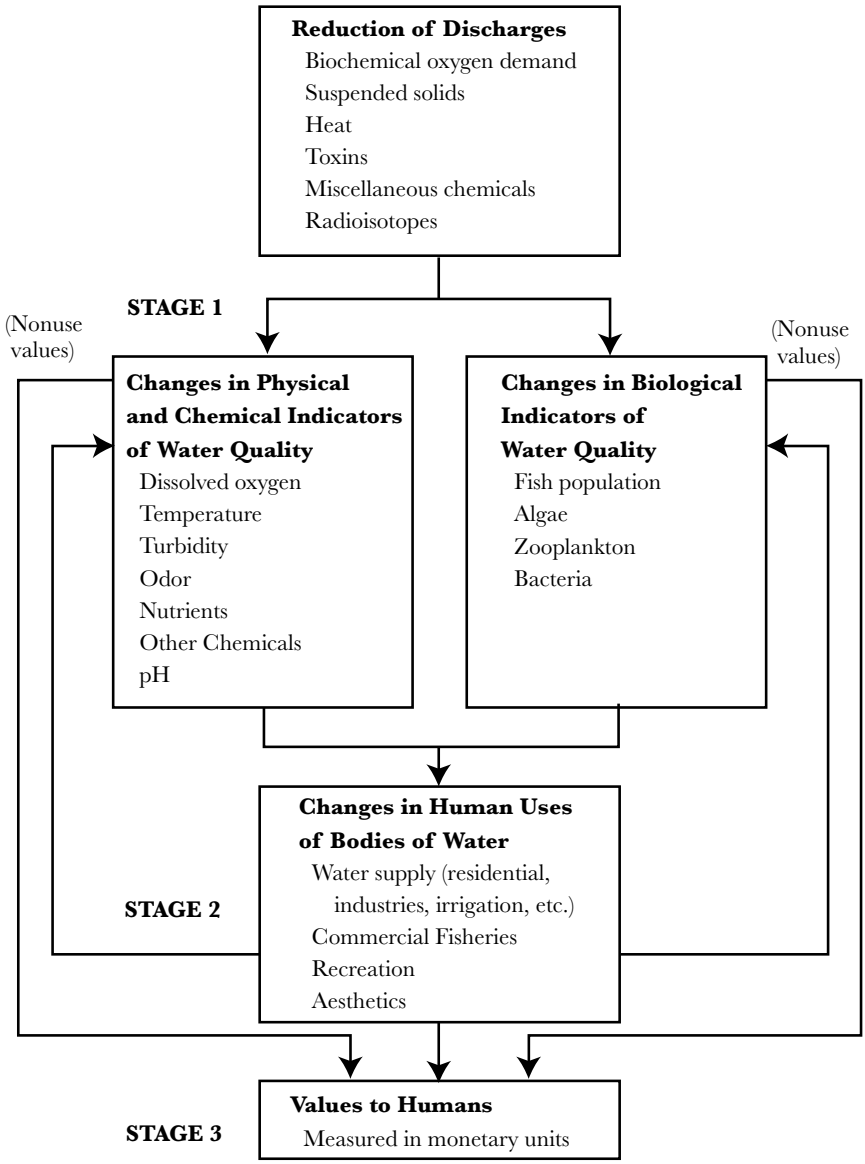

Figure 2.1 The production of benefits from improved ambient water quality

for example reductions in fish stocks due to overfishing. Figure 2.1 also shows nonuse values that are independent of stage 2.

In our illustration, estimating water quality benefits involves determining the monetary values that people place on such things as improved recreational opportunities, increases in fish production, and the availability of particular species

of fish (stage 3). Regarding the analysis of this stage, there is a well-developed theory of economic value. As discussed earlier, there are a number of approaches for estimating these values under different circumstances.

The Noneconomic Foundations of Resource Valuation

In a book that is about determining the economic values attached to the services affected by environmental change, the discussion cannot proceed far without acknowledging the importance of the relationship between environmental and resource quality and the uses of the environment. For example, the value of a recreation user-day at a lake is affected by fish populations and species distribution, algae levels, the number and type of bacteria present, temperature, smell, turbidity, and concentration of toxic substances. Further complicating matters, an increase in the magnitude of one characteristic may affect one use favorably while affecting an alternative use in a negative way. Higher water temperatures, for example, may make for better swimming while adversely affecting trout and salmon populations. Industrial discharges of acids may adversely affect recreation and fisheries while improving the value of water for industrial uses because of retarded algae growth.

The difficulties in tracing out the effects of the discharge of a pollutant on the many parameters of environmental quality and, in turn, their effects on human uses of the environment substantially limit our ability to do careful benefit-cost analyses of environmental quality and improvements. This has not always been fully appreciated by many advocates of greater use of benefit-cost analysis in this field.

In 1968 Allen Kneese wrote,

> I believe that our limited ability to evaluate the recreational losses associated with poor quality water, or conversely, the benefits of water improvement, is an extremely important barrier to rational water quality management … The first [complexity] is the relationship between the level of various water quality parameters and the recreational attractiveness of the water resource. This relationship can be viewed as being composed of two linkages: a natural one and a human one. I think these are both about equally ill-understood. It is my impression … that the biological sciences are almost never able to tell us specifically what difference a change in measured parameters of water quality will make in those biological characteristics of the water that contribute to its recreational value … Perhaps the undeveloped state of forecasting is a result of the fact that biologists have seldom been confronted with the types of questions we would now like them to answer … There is also a human linkage that is ill-understood. What quality characteristics of water do human beings find attractive for recreation? This is still largely an area of ignorance.
>
> (Kneese 1968, 180–181)

Although substantial progress has been made in the past 45 years or so since Kneese wrote this, there are still substantial gaps in our understanding of some of these linkages. What is true of water quality and recreation is also true of the other uses we make of water bodies; and it is also true of the other dimensions of environmental quality and the uses we make of the environment.

Describing Resource and Environmental Quality

An analysis of the value of a resource, or of the benefits of an environmental or resource policy change, must begin with a description of the resource flow or some measure of environmental quality. This description requires choices about what attributes or characteristics of the resource-environmental service are important. Suppose the question is, "What will be the benefits of achieving the auto emissions standards mandated by federal law?" In order to answer this question it is necessary to determine what things that matter to people are adversely affected by automotive emissions, and then to trace out the links between emissions, and those things that are valued. It is now understood that automobile emissions and the subsequent products of their photochemical reactions adversely affect human health, visibility, and agricultural productivity, among other things. Therefore, in order to estimate values, it is necessary to determine what specific measures of air quality are linked to these effects and how these measures of air quality are affected by the mandated standards. The measures used to characterize effects on visibility may be quite different from those relevant to effects on human health.

Two kinds of problems arise in this stage of the analysis. The first concerns the choice of parameters for describing the resource or environmental quality. The second involves determining the functional relationship between the policy instrument and the resource service flow or environmental quality measure.

Consider the case of water quality. A single effluent discharge can contain many substances that affect water quality—oxygen-demanding organic wastes (biochemical oxygen demand), suspended solids, waste heat, and toxic chemicals are all examples. When these substances enter the waterway, they affect—in sometimes simple and sometimes complex ways—such measurable components of water quality as dissolved oxygen, temperature, and concentrations of chemicals. A nondegradable chemical substance, for example, will simply be diluted, and its concentration in the water body will be a calculable fraction of its concentration in the effluent stream. In contrast, organic wastes affect water quality parameters in a more complicated way. As they are degraded by bacteria, they reduce dissolved oxygen levels to an extent and at a rate dependent on water temperature, wind, rates of river flow, and other physical and biological characteristics of the receiving water.

Some of the physical measures of water quality, such as turbidity and smell, affect human uses of the water directly. In addition, these and other physical parameters affect the stream ecology in complex and not always well-understood

ways. The populations and species distributions of fish, algae, zooplankton, and bacteria may also be affected, and not necessarily in the same direction, by changes in the physical and chemical parameters of water quality.

Even providing a descriptive characterization of this first stage is a formidable task. Water quality cannot be represented by a single number on some scale, but rather is an *n*-dimensional vector of the relevant parameters. Which subsets of these parameters are most important in influencing the uses of a water body (commercial fishing, boating, or swimming, for example) is still a major question for research.

Developing predictive models for these parameters is also a major research priority. The most commonly used water quality models relate dissolved oxygen to the biochemical oxygen demand of discharges of organic wastes. Dissolved oxygen levels, however, are only crudely related to the suitability of a water body for fishery production or recreational use.

In the case of air quality, the choice of parameters is somewhat easier, but not without pitfalls. It is only in the past 25 years or so that attention has turned from measurement of sulfur dioxide to its transformation products, sulfate particles. The latter measure of air quality is now known to be a more important air pollutant indicator than sulfur dioxide because of its effects on human health and ecology (U.S. Environmental Protection Agency 1997 and references therein).

Temporal and Spatial Aggregation

Typically, measures of environmental quality vary over time and space. The dissolved oxygen level at any point in a river rises and falls with changes in streamflow, discharge rates, water temperature and the like, and is different at different locations. Air pollution readings vary with the time of day, the day of the week, over the year, and from one place to another. One problem in empirical research on effects of pollution is how to define a variable or set of variables in a way that adequately reflects the temporal and spatial variations in environmental quality while still being manageable.

To put the problem in a concrete setting, consider an attempt to estimate the relationship between the level of an air pollutant and some health effect, say the occurrence of an asthma attack. The air pollution level at any particular point in an urban area is an instantaneous variable that fluctuates over time. The true exposure of an individual located at that point is measured by a trace of the time path of that instantaneous variable over the relevant time period. However, the individual may also move from one point to another in the urban space. The published data on air pollution levels that are used to generate exposure variables for empirical research involve various approaches to summarizing the instantaneous time paths at the locations of air-pollutant measurement devices. One common measure is the annual mean, either arithmetic or geometric. Averages are also struck over shorter time periods—a 24-hour average for particulates, and 8-hour and 1-hour averages for other pollutants, for example. Also, readings taken at one or two points in the

space must be used to represent the space as a whole. Inevitably, summarizing involves loss of information. One or two of these temporal and spatial summaries cannot completely represent the true exposure of any individual. Research on health effects is hampered by our inability to characterize accurately the exposure of individuals.

The Welfare Economics of Costs

So far, this chapter has focused attention on valuing the benefits of environmental changes. It is time now to turn attention to the costs of achieving these changes. It is a commonly held view both within and outside the economics profession that the costs of environmental regulations are relatively easy to measure, at least in comparison with the task of measuring environmental benefits. This optimistic view is consistent with a naive theory of cost, which takes the following form. Firms respond to pollution control regulations by purchasing and installing waste treatment equipment and control systems that, in effect, are bolted on to the existing factory. The purchase and installation costs of this equipment, plus the added operating and maintenance costs it would entail are readily identified in the firm's accounts. These expenditures are often taken, at least as a first approximation, to be the social costs of complying with the regulation.

The naive theory fails to recognize the fundamental symmetry between benefits and costs as changes in the utilities of individuals; and it also neglects several important realities concerning the ways in which government regulations can affect people's welfare. The symmetry of benefits and costs stems from the fact that ultimately all costs take the form of utility losses to individuals in their dual roles as receivers of income and consumers of market and nonmarket goods and services. These losses have their monetary counterparts in compensating measures of welfare change—that is, willingness to pay to avoid the cost and willingness to accept compensation for bearing the cost. Because of this fundamental symmetry, proper measurement of costs involves the same kinds of problems as, and is likely to be as difficult as, the measurement of the benefits of environmental improvement. Once the symmetry is acknowledged, the whole economist's tool kit of revealed preference and stated preference methods of measuring welfare changes becomes available for the cost analyst.

The naive view about cost estimation is also implicitly based on the presumption that only firms cause pollution, and that therefore only firms must incur costs in the process of moving to meet environmental quality objectives, but this is not the case. Regulations, for example, may be placed on household activities, and these regulations might not require any identifiable expenditure of money. Suppose that commuters are required to form car pools as part of a program to meet air quality standards for ozone. Household expenditures on commuting may in fact be lower as a consequence of the regulation. The cost of such regulation takes the form of increased commuting time and loss of convenience. These may be difficult to quantify and value. Consider also a regulation that decreases the quality or stream

of services from a consumer good. In the 1970s for example, new cars complying with tailpipe emissions standards had lower performance and fuel efficiency than earlier models. The true cost of complying with these standards consisted of both the higher cost of purchasing and operating the car and the decreased value of the services from the car.

When an environmental regulation affects household activities or the availability of nonmarket goods and services, there may be opportunities for employing revealed preference methods to draw inferences about the negative values or costs of these changes. For example, if the quality of a good is reduced as a consequence of a regulation, hedonic price models might be used to estimate the marginal implicit price of the relevant attribute. In the example of imposed car-pooling, an estimate of the shadow price of time could be used to derive the costs of the increased commuting time. Also, stated preference methods such as contingent valuation or choice experiments might be used where a comprehensive set of regulations affects a wide range of activities.

Another factor neglected in the naive theory of costs is that market mechanisms are likely both to shift the burden of firms' expenditures and to change the magnitude of the burden so that the true costs are not accurately measured by summing the expenditures by firms. At least for nonmarginal changes, firms will raise prices and will experience decreases in quantity demanded. Price increases cause losses of consumer surplus, which are part of the social costs; and there may be losses in producers' surpluses and factor incomes as well. Thus, it is not correct simply to equate pollution control expenditures and social costs (Portney 1981).

As an extreme example, consider the case of a tax on the carbon content of fuels as a means of reducing the emissions of carbon dioxide (CO_2). If there were no economically feasible technologies for controlling CO_2 emissions from combustion, then the tax would work entirely through raising the prices of carbon-based fuels and reducing demands for them. Consumers would bear at least part of the costs of the revenues raised by the tax, and the rest would come from decreases in resource rents. However, since these tax revenues represent a transfer to the government, they are not part of the social cost of reducing CO_2 emissions. The social cost of controlling CO_2 by taxation comes entirely in the form of the equivalent of the deadweight losses or welfare triangles associated with the tax.

Where the direct impact of a regulation is on firms, and where its effects are transmitted to individuals through changes in prices and incomes, the process of cost estimation must call on two types of models. The first is a model of the firm's production technology and costs. The second is a market model that can be used to calculate the changes in prices and incomes of the affected individuals. This model, in some circumstances, could be a partial equilibrium model. For broad social regulations, a general equilibrium model of the economy may be required (Hazilla and Kopp 1990).

In modeling the behavior of the firm, a number of questions have to be considered. For example, can the technology be modeled as additively separable so that the total cost is the sum of the costs of producing the marketed outputs and

the cost of treating the waste? Or must the technology and costs be modeled as fully joint? The assumption of additive separability may be appropriate if, in fact, firms would respond to a regulation by purchasing add-on control or treatment equipment. However, other kinds of technological responses, such as input substitution and recycling, may have to be modeled in a joint production framework.

Once a model of a firm's cost has been obtained, the next step is to embed that model in a model of the market economy so that the changes in all relevant quantities, prices, and incomes can be predicted. In the simplest case, where the regulation involves a marginal change in costs and where the economy is perfectly competitive, costs can be measured by the predicted change in expenditures on factor inputs. This is because the invariant factor prices are equal to the values of the marginal products of those inputs in other uses; and, in turn, those values of marginal products measure the opportunity losses to consumers associated with the marginal reallocation of inputs.

With nonmarginal changes, it can be expected that there will also be changes in product prices and perhaps factor prices. If a regulation affects only one industry, a partial equilibrium model may be appropriate. Rather than have costs be a function of q, among other things, a parameter reflecting the stringency of the regulation or the degree of pollution control required could be included as a shifter of the cost or production function.

Implicit in this approach are the assumptions that the regulation does not cause any shifts in either the output demand functions or the factor supply functions, either immediately or over time. However, if either of these assumptions are not valid, then a general equilibrium framework would be required. For example, if the regulation affected both the x and y industries, the regulation-induced change in the price of y could shift the demand function for x. Also, if y were an intermediate product and an input in the x industry, the increase in its price or the upward shift in its supply function would have a secondary impact on the cost of producing x. Hazilla and Kopp (1990) have shown that estimating true social costs in a general equilibrium framework can lead to quite different results in comparison with the pollution control expenditure approach utilized by the Environmental Protection Agency (EPA).

More recent work has called attention to another, perhaps more important, general equilibrium effect. If an environmental regulation leads to an increase in the prices of goods and services, then the result is a fall in real wages; and if labor supply elasticities are positive, a reduction in the quantity of labor supplied occurs. Since the labor market is already distorted because of the presence of income and payroll (social security) taxes, the marginal social value of labor exceeds its marginal social cost by a substantial amount. Even a small decrease in the quantity of labor supplied can have a large net welfare cost. This cost is in addition to the direct cost of the environmental regulation. The impact of regulatory costs on the labor market is known as the tax interaction effect. For a clear explanation of this effect and a discussion of its significance for the economic analysis of environmental and other regulatory policies, see Parry and Oates (2000).

Summary

Economists seek measures of values that are based on the preferences of individuals. When value measures are derived using models of behavior, these models should be internally consistent and be based on accepted theories of preferences, choice, and economic interactions. Equally important is the need for a sound understanding of the underlying biological and physical processes by which environmental and resource service flows are generated. However, if empirical observations of individuals' choices are taken without benefit of an underlying theoretical model, researchers may be led to make faulty or erroneous interpretations of the data. An interesting example is the early studies of the land value / air pollution relationship. Researchers discovered that land values and air pollution levels were inversely related in urban areas, other things being equal. They then assumed that changes in welfare associated with reduced pollution would be accurately measured by the associated increases in land values as predicted by the regression equation relating land values at a point in time to air pollution. Subsequent research based on theoretical models of urban land markets has shown that this assumption is not true in general. The relationships among air pollution, land values, and measures of welfare change are discussed in Chapter 10.

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Chapter 3

Welfare Measures

Definitions and Concepts

The theory of the measurement of welfare change has been discussed by others, both at the most rigorous levels of abstraction, and in pragmatic, practical terms of application. See for examples, Johansson (1987), Just, Hueth, and Schmitz (1982, 2004), and Bockstael and McConnell (2007). The earliest work focused on the welfare effects of changes in the prices people pay for the (private) goods they consume, but the literature has expanded broadly into valuing changes in the quantity and quality of both private and public goods. The current chapter provides a systematic development of the definition and measurement of the welfare effects stemming from changes in prices and the quantities and/or qualities of nonmarket environmental and resource service flows.

Changes in environmental quality can affect individuals' welfares through a number of channels: changes in the prices paid for goods bought in markets; changes in the quantities or qualities of nonmarketed goods (for example, public goods such as air quality); changes in the prices received for factors of production; and changes in the risks individuals face. The first two of these channels are the focus of this chapter. After a brief review of the theory of individual preferences and demand, the principles of welfare measurement for price changes are reviewed. These principles are relevant because some forms of environmental change affect people only indirectly through price effects, and because these principles provide a solid foundation for the treatment of quantity and quality changes that follow. Chapter 8 covers the welfare effects of changes in factor prices. The extension of these principles to the valuation of changes in risk—the fourth channel—raises some interesting questions, which will be left to Chapter 5.

The principles and measures developed in this chapter apply equally to decreases and increases in individuals' welfare. It is a basic principle of welfare economics that all costs ultimately take the form of reductions in the utility of individuals. This principle applies equally to the costs of public policies (for example, investment in resource development and the regulation of private activities), and to the costs of private uses of the environment (for example, harvesting from a common property resource and using the waste receptor services of the environment). Hence, the welfare measures developed here provide a foundation for the analysis of both the benefits and the costs of environmental change.

In this chapter, three sets of questions are considered in some detail. The first set concerns how to define an acceptable monetary measure of changes in economic welfare for an individual. The answer to this question hinges partially on what the measure would be used for—that is, a welfare measure should answer the questions posed by policymakers. However, policymakers can ask different kinds of questions. For example, suppose that policymakers wish to evaluate proposed policy changes in terms of an aggregate social welfare function that places different weights on individuals' changes in utility depending on their positions in the income distribution (Bergson 1966). In that case, the welfare measure that answers the policymakers' question must be a money metric of utility changes. Alternatively, if policymakers wish to select policies on the basis of the potential Pareto improvement criterion, they will want measures of required compensation and willingness to make compensating payments. The concluding section of this chapter returns to the question of choosing from among the alternative measures described here.

The second set of questions concerns how changes in welfare would be measured, both in theory and in practice. Theory suggests several alternative ways of calculating either exact or approximate welfare measures using data on observed behavior of individuals—for example, their demand functions for market goods. These alternatives will be described and evaluated, especially from the practical perspective of implementation.

The third set of questions concerns how any measure of welfare changes for individuals might be used to make judgments about social policies affecting many individuals. For example, is it possible to speak of a measure of aggregate welfare for the society as a whole? If so, what significance can be attached to changes in such a measure? Measures of welfare change for an individual can be defined and analyzed without reference to the notions of efficiency and equity. In this sense, the concept is objective—that is, one can define and measure a monetary equivalent of an individual's welfare change without being committed to any particular set of value judgments concerning aggregation across individuals, or the role of such welfare measures in social choice. It is in answering the third set of questions that value judgments about the relative deservingness of individuals, the meaning of efficiency, and the objectives of public policy come into play. Some of these issues are discussed in the Aggregation and Social Welfare section of this chapter.

The following section begins with a review of some of the basic terminology and theory involving individual preferences and demand.[1] Next, the standard case where utility depends only on the consumption of market goods is considered. In this simple context, the theory of measuring the welfare value of changes in

1 Throughout most of this text, the presentation is based upon neoclassical theory and the assumption of a rational consumer. However, in recent years the assumption of a rational consumer has been drawn into question by research into behavioral economics (e.g., Camerer, Loewenstein and Rabin 2004; Mullainathan and Thaler 2001; Shogren and Taylor 2008). Discussion of the implications of this line of research for welfare economics is left to Chapter 13.

the prices of these goods is examined, along with the relationships among the Marshallian and Hicksian surplus measures of welfare change for continuous goods (that is, goods for which the main consumer choice problem is to choose how many units to purchase and consume). There are two reasons for choosing this order of presentation. First, it parallels the historical evolution of the theory of welfare change. Second, it makes for an easier exposition of the basic principles. This section concludes with a review of methods for obtaining exact measures of, and approximations to, the desired Hicksian surpluses. The third section examines the case of the welfare effects of changes in the quantities of continuous goods. While potentially applicable to private goods, these welfare measures will most often be relevant for public goods or nonmarket goods since the quantity available for consumption is not a matter of choice for the individual. The next section considers welfare measures for discrete goods: goods for which the key choice is not how many units to consume, but rather which goods, from two or more alternatives, are consumed. In the final section, a review is provided of some of the issues involved in aggregating measures of individual welfare change for public policy decision making and in selecting the appropriate welfare measure.

Individual Preferences and Demand

Before introducing the various possible welfare measures, it will be useful to review briefly the basic theory of individual preferences and the demand for goods as it relates to welfare theory. For alternative treatments of this and related topics, the reader may wish to consult other texts, such as Just, Hueth, and Schmitz (1982, 2004), Boadway and Bruce (1984), Varian (1992), and Johansson (1987). This theory starts with the premise that individuals are their own best judges of their welfares and that inferences about welfare can be drawn for each individual by observing that individual's choices among alternative bundles of goods and services. If an individual prefers bundle A to bundle B, then bundle A must convey a higher level of welfare.

What things are to be included in the bundles (such as A and B) among which individuals are assumed to have preferences? There is little controversy over the inclusion of all the goods and services that can be bought or sold in markets—consumer goods, the services of household assets such as a house or a car, and consumer durables. Since time can be used in leisure activities, or sold at some wage rate in the labor market, individuals must also have preferences among alternative uses of time, such as reading, outdoor recreation, and working at some wage rate. Since government and the environment both provide a variety of services that enhance the welfares of individuals, these services should also be included in the bundles among which people have preferences. Environmental services include those provided by cleaner air, cleaner water, and scenic amenities. Just as importantly, these environmental services are not limited to direct uses of the environment, such as breathing clean air or observing unspoiled vistas—they can also include services related to the mere presence of environmental goods,

such as the knowledge that pristine wilderness areas exist, or that there is a viable breeding population of a particular endangered species.

If we assume that individuals can rank the alternative bundles according to their preferences, what properties will the resulting ordering of bundles display? For our purposes, two properties are important. The first is nonsatiation, or the "more-is-better" property. This means that a bundle with a larger quantity of an element will be preferred to a bundle with a smaller quantity of that element, other things being equal. Formally, if $\boldsymbol{X}'$ consists of $\left(x_1', \ldots, x_j', \ldots, x_J'\right)$ and $\boldsymbol{X}\mathbf{K}$ consists of $\left(x_1', \ldots, x_j'', \ldots, x_J'\right)$ and $x_j' > x_j\mathbf{K}$ then this individual will prefer $\boldsymbol{X}'$ to $\boldsymbol{X}\mathbf{K}$.

The second property is *substitutability* among the components of bundles. This means that if the quantity of one element of a bundle, say x_j, is decreased, it is possible to increase the quantity of another element, say x_k, sufficiently to make the individual indifferent between the two bundles. More formally, suppose that $\boldsymbol{X}'$ consists of $\left(x_1', \ldots, x_j', \ldots, x_k', \ldots, x_J'\right)$; and $\boldsymbol{X}\mathbf{K}$ consists of $\left(x_1', \ldots, x_j'', \ldots, x_k', \ldots, x_J'\right)$ with $x_j\mathbf{K} < x_j'$. Substitutability means that there is another bundle $\boldsymbol{X}^*$ consisting of $\left(x_1', \ldots, x_j'', \ldots, x_k^*, \ldots, x_J'\right)$ with $x_k^* > x_{kJ}'$, such that the individual is indifferent as to $\boldsymbol{X}'$ and $\boldsymbol{X}^*$. In other words, $\boldsymbol{X}'$ and $\boldsymbol{X}^*$ lie on the same indifference surface.[2]

The property of substitutability is at the core of the economist's concept of value. This is because substitutability establishes tradeoff ratios between pairs of goods that matter to people. In this formulation, the tradeoff ratio is $\left(x_k^* - x_k'\right) / \left(x_j' - x_j''\right)$ or $|\Delta x_k / \Delta x_j|$. In the limit for infinitesimally small changes, this reduces to $|dx_k / dx_j|$, which is the definition of the marginal rate of substitution between x_j and x_k, or the slope of the two-dimensional indifference curve between these two elements. The money price of a market good is just a special case of a tradeoff ratio, because the money given up to purchase one unit of one element of the bundle is a proxy for the quantities of one or more of the other elements in the bundle that had to be reduced in order to make the purchase.

If the preference ordering has the properties described here, it can be represented by an ordinal preference function, or utility function, that assigns a number to each bundle as a function of the quantities of each element of the bundle. Specifically,

$$u = u(\boldsymbol{X}, \boldsymbol{Q}, \boldsymbol{T}), \tag{3.1}$$

where $\boldsymbol{X}$ is a vector of the quantities of market goods, $\boldsymbol{Q}$ is a vector of public goods and environmental and resource services whose quantities or qualities are fixed for the individual, and $\boldsymbol{T}$ is a vector of the times spent in various activities that yield utility to the individual. This utility function is assumed to be increasing in

2 Two other important properties are transitivity and quasi-concavity. If there are three bundles $\boldsymbol{X}'$, $\boldsymbol{X}\mathbf{K}$, and $\boldsymbol{X}^*$, and the individual prefers $\boldsymbol{X}'$ over $\boldsymbol{X}\mathbf{K}$ and $\boldsymbol{X}\mathbf{K}$ over $\boldsymbol{X}^*$, then transitivity is satisfied if the individual prefers $\boldsymbol{X}'$ over $\boldsymbol{X}^*$. For more on the axiomatic description of these properties of preference ordering, see Boadway and Bruce (1984) or Varian (1992).

all of its arguments, and unique up to a monotonic transformation. For purposes of mathematical modeling and analysis, it is convenient also to assume that this function is continuous, convex, and twice differentiable. This preference function is not the same thing as the cardinal utility function of the classical utilitarians. Since there is no unit of measurement for this ordinal utility, it is not possible to add or otherwise compare the utilities of different individuals.

To simplify the exposition and notation, let us now consider an individual whose utility is a function only of private goods that can be bought and sold in markets. Assume that tastes and preferences (that is, the utility function) are given and do not change. The individual faces a set of given prices for these goods and is assumed to choose the quantities of the goods so as to maximize his utility, given the constraints of prices and a fixed money income M. The maximization problem can be expressed as

maximize $u = u(\boldsymbol{X})$,

subject to

$$\sum_{j=1}^{J} p_j x_j = M \tag{3.2}$$

where $\boldsymbol{X} = \left(x_1, \ldots, x_j, \ldots x_J\right)$ is the vector of quantities. The solution to this problem leads to a set of ordinary, or Marshallian, demand functions

$$x_j = x_j(\boldsymbol{P}, M), \tag{3.3}$$

where $\boldsymbol{P} = \left(p_1, \ldots, p_j, \ldots p_J\right)$ is the vector of prices.

Substituting the expressions for x_j as functions of $\boldsymbol{P}$ and M into the direct utility function gives the indirect utility function—that is, utility as a function of prices and income, assuming optimal choices of goods:

$$v = v(\boldsymbol{P}, M). \tag{3.4}$$

According to Roy's identity, the demand functions can also be expressed in terms of derivatives of the indirect utility function,

$$x_j\left(\boldsymbol{P}, M\right) = -\frac{\partial v / \partial p_j}{\partial v / \partial M} \cdot \tag{3.5}$$

The expenditure function represents a useful perspective on the problem of individual choice. The expenditure function is derived by formulating the dual of the utility maximization problem. The individual is assumed to minimize total expenditure,

$$e = \sum_{j=1}^{J} p_j x_j \ , \tag{3.6}$$

subject to a constraint on the level of utility attained,

$$u\left(\boldsymbol{X}\right) = u^0 \ , \tag{3.7}$$

where u^0 is the maximum utility attained with the solution to the primal problem. Just as the solution to the utility maximization problem yields a set of ordinary demand curves conditional on prices and money income, the solution of the expenditure minimization problem yields a set of functions giving optimal quantities for given prices and utility. These are Hicks-compensated demand functions that show the quantities consumed at various prices, assuming that income is adjusted (compensated), so that utility is held constant at u^0. Substituting these demand functions into the expression for total expenditure yields the expenditure function. This expression gives the minimum dollar expenditure necessary to achieve a specified utility level given market prices. In functional notation:

$$e = e(\boldsymbol{P}, u^0), \tag{3.8}$$

where e is the dollar expenditure and u^0 is the specified utility level. The compensated demand functions can also be found by differentiating the expenditure function with respect to each of the prices:

$$\frac{\partial e}{\partial p_j} = h_j = h_j\left(\boldsymbol{P}, u^0\right), \tag{3.9}$$

where h_j is the compensated demand for x_j.

Now consider the set of ordinary demand functions derived from the utility maximization problem. In order to determine the functional form and parameters of these demand functions, it is necessary to know the underlying utility function, and this may not be directly observable. Suppose instead that we observe an individual's behavior and estimate the demand functions that describe the individual's responses to changes in prices and income. These functions are based on the same information as the underlying preferences. This is assured, provided the demand functions satisfy the so-called *integrability conditions*. These conditions require that the Slutsky matrix of substitution terms,

$$\frac{\partial h\left[\boldsymbol{P}, v\left(\boldsymbol{P}, M\right)\right]}{\partial p_k} = \frac{\partial x_j\left(\boldsymbol{P}, M\right)}{\partial p_k} + \frac{\partial x_j\left(\boldsymbol{P}, M\right)}{\partial M} x_k\left(\boldsymbol{P}, M\right), \tag{3.10}$$

is symmetric and negative semi-definite (Hurwicz and Uzawa 1971; Silberberg 1978; Varian 1992). If these conditions are satisfied, the system of demand functions can be integrated to yield the expenditure function, which in turn can be used to derive the indirect and direct utility functions. If the integrability conditions are not satisfied, the implication is that the observed demand functions are not consistent with the maximization of a well-behaved utility function. As explained below, if the integrability conditions are satisfied, it may be possible to utilize empirically derived descriptions of demand behavior to obtain a complete description of the underlying preferences, as well as exact measures of welfare change for a wide range of postulated changes in economic circumstances.

Welfare Measures for Continuous Goods: Price Changes

An Overview

In order to introduce the alternative welfare measures, consider first the simplest case of only two goods and the welfare gain associated with a nonmarginal decrease in the price of one of these goods. Two types of measures of this welfare change have been identified in the literature. The first is the change in ordinary consumer's surplus, a concept with an origin that can be traced back through Alfred Marshall to Dupuit. Mishan (1960) and Currie, Murphy, and Schmitz (1971) provided useful discussions of the history and evolution of the concept of consumer's surplus. As Marshall explained it,

> [The individual] derives from a purchase a surplus of satisfaction. The excess of the price that he would be willing to pay rather than go without the thing, over that which he actually does pay, is the economic measure of this surplus of satisfaction. It may be called consumer's surplus.
>
> (Marshall 1920, 124)

Ordinary consumer's surplus is measured by the area under a Marshallian ordinary demand curve, but above the horizontal price line. As we will see, the

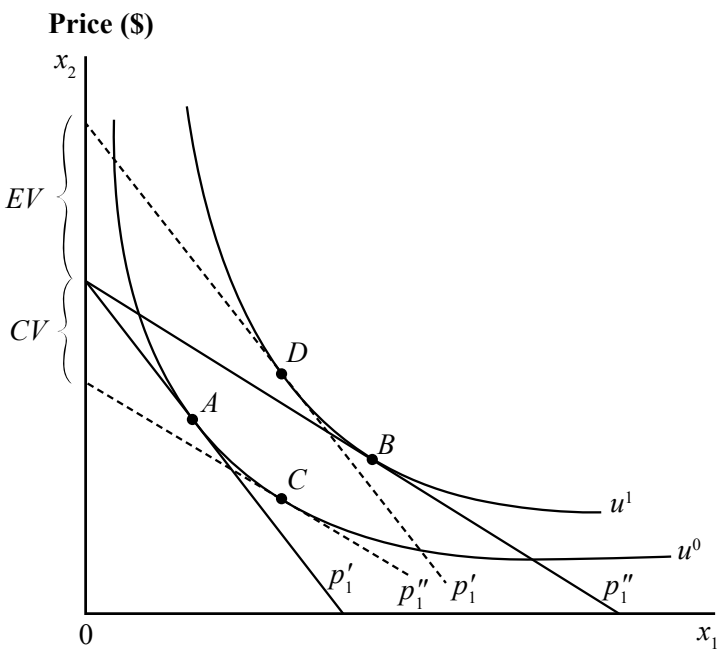

Figure 3.1 Two measures of the welfare gain from a price decrease

consumer surplus measure cannot be defined in terms of the underlying utility function.

The other measures of welfare change are theoretical refinements of the ordinary consumer's surplus (Hicks 1943), and each can be defined in terms of the underlying individual preference mapping. Figure 3.1 is used to illustrate these concepts in the context of two goods (x_1 and x_2), where x_2 is the numeraire good (i.e., $p_2 = 1$). The figure shows two indifference curves for the individual. Assume that an environmental improvement reduces the cost of producing x_1, so that its price drops from p_1' to p_1''. In response to the price reduction, the individual shifts from the consumption bundle marked *A* at utility level u^0 to consumption bundle *B* at utility level u^1. What is the welfare benefit of the price reduction to this individual? Two additional measures of the welfare change can be defined in terms of the numeraire good x_2:

1 Compensating Variation (*CV*). This measure asks what compensating payment (that is, an offsetting change in income) is necessary to make the individual indifferent between the original situation (*A* in Figure 3.1) and the new price set. Given the new price set with consumption point *B*, the individual's income could be reduced by the amount of *CV* and that person would still be as well off at point *C* as at point *A* with the original price set and money income. The measure *CV* is often interpreted as the maximum amount that the individual would be willing to pay for the opportunity to consume at the new price set. However, for a price increase, *CV* measures what must be paid to the individual to make that person indifferent to the price change. For price decreases, the *CV* cannot be greater than the individual's income; but for a price increase, the *CV* could exceed income.
2 Equivalent Variation (*EV*). This measure asks what change in income (given the original prices) would lead to the same utility change as the change in the price of x_1. As shown in Figure 3.1, given the original prices, the individual could reach utility level u^1 at point *D* with an income increase equal to *EV*. *EV* is the income change equivalent to the welfare gain due to the price change. The *EV* measure has also been described as the minimum lump sum payment the individual would have to receive to induce that person to voluntarily forgo the opportunity to purchase at the new price set. For a price increase, *EV* is the maximum amount the individual would be willing to pay to avoid the change in prices.

Note that both the *EV* and *CV* measures allow the individual to adjust the quantities consumed of both goods in response to both changes in relative prices and income levels. Hicks also described two additional measures where the levels of the goods could not be changed. He referred to them as compensating and equivalent surplus. The compensating and equivalent surplus measures for price changes do not answer very useful questions since they both arbitrarily restrict the individual to consuming a specific quantity of the good whose price has changed.

Hence, the original form suggested by Hicks will not be considered further, but we will return to measures of welfare change associated with quantity changes (with no associated price changes) later in this chapter; and note that, following Hicks, such welfare measures are often referred to as compensating and equivalent surplus. The next subsection is devoted to a comparison and evaluation of the compensating and equivalent variations and their relationship to the ordinary consumer surplus.

In the many-good case, x_2 is a composite good that can be treated as an index of the consumption levels of all other goods except x_1. The aggregation of all other goods into a composite good for graphical representation is valid so long as the prices of all of the goods are assumed to move in the same proportion—that is, there are no changes in the relative prices of components of the composite good bundle. This assumption can be maintained, since we are analyzing only the consequences of the change in the price of x_1.

A Closer Look at the Welfare Measures

This section begins with a presentation of the basic welfare measure for a marginal change in one price. Then more rigorous derivations are provided for the consumer surplus, compensating variation, and equivalent variation measures of welfare change for the case of changes in price. For more detailed treatment of these topics see Silberberg (1972), Just, Hueth, and Schmitz (1982, Appendix B), Varian (1992), and Johansson (1987). For a marginal change in, say, p_1, the basic welfare measure is the change in expenditure necessary to hold utility constant. Using equation (3.9) from the previous section, we have

$$w_{p_1} = \frac{\partial e\left(\boldsymbol{P}, u^0\right)}{\partial p_1} = h_1\left(\boldsymbol{P}, u^0\right), \tag{3.11}$$

where w_{p_1}, is the marginal welfare measure. This result also follows from the indirect utility function and Roy's identity:

$$w_{p_1} = x_1 = -\frac{\partial v/\partial p_1}{\partial v/\partial M} \tag{3.12}$$

or

$$\frac{dM}{dp_1} = x_1 . \tag{3.12′}$$

In equation (3.12), the marginal utility of the price change is converted to monetary units by dividing by the marginal utility of income. Equation (3.12′) says that the change in income required to hold utility constant is equal to the change in price multiplied by the quantity of the good being purchased.

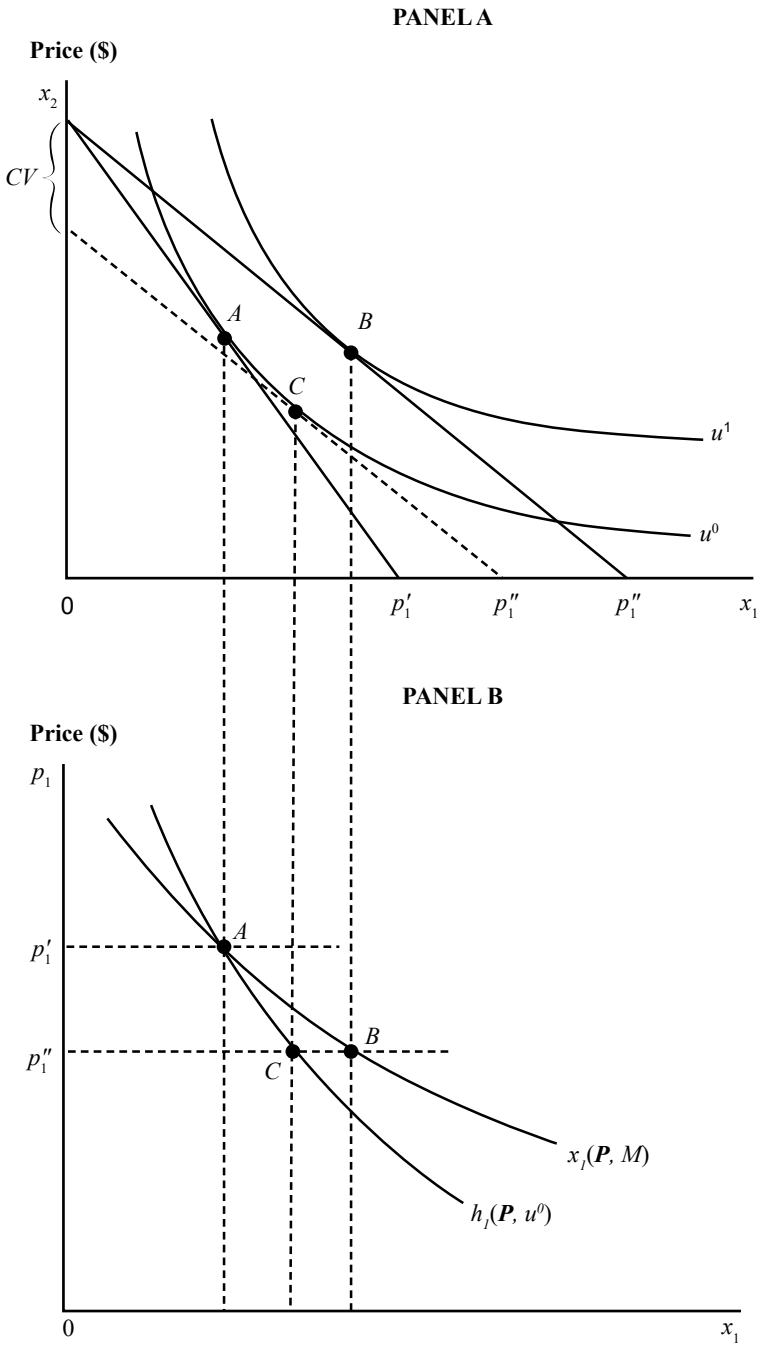

Figure 3.2 The compensating variation and Hicks-compensated demand

Marshallian Consumer Surplus

In Figure 3.2, panel A shows one individual's preference mapping in the simple two-good case. Suppose that the price of good x_1 falls from p_1' to p_1''. The individual responds by moving from the original equilibrium at point A to point B on the new budget line. In panel B of Figure 3.2, these equilibrium positions are plotted in the price and quantity plane. Points A and B are on the ordinary demand curve, holding the price of good x_2 and money income constant. Since the Marshallian surplus associated with the consumption of a good at a given price is the area under the demand curve, the change in surplus for a change in the good's price is the geometric area $p_1'ABp_1''$ in panel B of Figure 3.2. In mathematical form,

$$S = \int_{p_1''}^{p_1'} x_1(\boldsymbol{P}, M)\, dp_1 \,, \tag{3.13}$$

where S is the change in surplus.

The condition under which S can be interpreted as an indicator of utility change can be seen by employing Roy's identity:

$$x_1(\boldsymbol{P}, M) = -\frac{\partial v(\boldsymbol{P}, M)/\partial p_1}{\partial v(\boldsymbol{P}, M)/\partial M} \,, \tag{3.14}$$

and substituting this into equation (3.13) to obtain

$$S = -\int_{p_1''}^{p_1'} \left[\frac{\partial v(\boldsymbol{P}, M)/\partial p_1}{\partial v(\boldsymbol{P}, M)/\partial M} \right] dp_1 \,. \tag{3.15}$$

If the marginal utility of income is constant over the range of the price change, this can be written as

$$S = \frac{v(p_1'', p_2, M) - v(p_1', p_2, M)}{\partial v/\partial M} \,. \tag{3.16}$$

This expression shows that the Marshallian surplus can be interpreted as the utility change converted to monetary units by a weighting factor—the marginal utility of income. If the marginal utility of income is constant, then S can be said to be proportional to the change in utility for any price change. However, as any one price changes, the constancy of the marginal utility of income is a restrictive condition. The marginal utility of income cannot simultaneously be invariant with respect to income and to changes in all of the prices (Samuelson 1942; Johansson 1987, ch. 4).

Alternatively, as Eugene Silberberg explained (1978, 350–361), the integral of equation (3.15) can be viewed as the sum of a series of small steps from an initial price and income vector of (p_1', p_2', M) to (p_1'', p_2', M), following a path on which p_2 and M are held constant. However, there are other paths over which (3.15) can

be integrated involving changes away from the initial values for p_2 and/or M as long as the terminal point is $\left(p_1'', p_2', M\right)$. The other paths, in general, will not lead to the same solution value for the integral. In other words, the integral in general will not be path independent.

A similar problem arises when the Marshallian surplus measure is generalized to simultaneous changes in all prices. In this case, S is defined as a line integral. This integral will be independent of the path of integration (that is, the order in which prices and/or incomes are assumed to change) only if the income elasticities of demand for all goods are equal. The income elasticities of all goods can be equal to each other only if they are all equal to one, in other words, if preferences are homothetic. Finally, if the prices of only a subset of all goods change, a unique S exists if the marginal utility of income is constant with respect to only those prices that are changed. See Just, Hueth, and Schmitz (1982) for more details.

Compensating Variation

Suppose now that as the price of good x_1 is decreased, income is taken away from the individual so that he remains at the initial utility level and indifference curve u^0. Given the price change and the compensating income change, the individual would be in equilibrium at point C in panel A of Figure 3.2. Point C is also plotted in panel B of Figure 3.2. Points A and C are on the Hicks-compensated demand curve, a demand curve that reflects only the substitution effect of the change in relative prices. The device of compensating withdrawals of money income has eliminated the income effect of the price change. Since x_1 is a normal good by assumption—that is, it has an income elasticity greater than zero—the Hicks-compensated demand curve is less price-elastic than the ordinary demand curve. The difference between the Hicks-compensated and the ordinary demand functions is one of the main considerations in the comparison of EV, CV, and consumer surplus measures of welfare change.

Panel A of Figure 3.2 shows the compensating variation measure of the welfare change associated with the price decrease—that is, the reduction in income needed to hold the individual on the original indifference curve. In terms of the indirect utility function, CV is the solution to

$$v\left(\boldsymbol{P}', M\right) = v\left(\boldsymbol{P}'', M - CV\right) = u^0. \tag{3.17}$$

The CV can also be defined in terms of the expenditure function. It is the difference between the expenditures required to sustain utility level u^0, at the two price sets:

$$\begin{aligned} CV &= e\left(p_1', p_2, u^0\right) - e\left(p_1'', p_2, u^0\right) \\ &= M - e\left(p_1'', p_2, u^0\right) > 0. \end{aligned} \tag{3.18}$$

Because CV is defined as the difference between two levels of expenditure, it can also be written as the integral of the marginal welfare measure (equation 3.11) over the relevant range. Specifically,

$$CV = \int_{p_1''}^{p_1'} \frac{\partial e\left(\boldsymbol{P}, u^0\right)}{\partial p_1} dp_1 = \int_{p_1''}^{p_1'} h_1\left(\boldsymbol{P}, u^0\right) dp_1 \,. \tag{3.19}$$

Since spending M at the new price set yields a higher level of utility, we can also write

$$M = e\left(p_1'', p_2, u^1\right), \tag{3.20}$$

and by substitution

$$CV = e\left(p_1'', p_2, u^1\right) - e\left(p_1'', p_2, u^0\right) > 0 \,. \tag{3.21}$$

In other words, although the CV is defined in terms of u^0, it also measures the amount of money required to raise utility from u^0 to u^1 at the new set of prices.

The CV is equal to the area to the left of the Hicks-compensated demand curve between the two prices—that is, the area $p_1'ACp_1''$. The partial derivative of the expenditure function with respect to p_1 gives the change in expenditure (income) necessary to keep the individual on u^0 for small changes in p_1. As shown above, this derivative gives the Hicks-compensated demand curve—that is, it gives the optimal quantity for x_1, holding utility constant. For finite changes, the integral of this derivative is the area to the left of the Hicks-compensated demand curve—that is, the CV. In other words,

$$CV = \int_{p_1''}^{p_1'} h_1\left(\boldsymbol{P}, u^0\right) dp_1 \,. \tag{3.22}$$

Unlike the Marshallian measure of surplus given by equation (3.13), this measure does not rely on any assumption about the constancy of the marginal utility of income. This is because this measure integrates along a constant utility indifference curve at u^0. In the many-good case, when several prices change, the CV of the price changes taken together is the integral of the set of compensated demand functions evaluated by taking each price change successively. The order in which the price changes are evaluated is irrelevant. This follows from the symmetry of the cross price substitution terms—that is, $\partial x_j/\partial p_k = \partial x_k/\partial p_j$.

Equivalent Variation

The equivalent variation can also be derived through the expenditure function. Panel A of Figure 3.3 shows the same preference mapping and price change for an individual. With a price decrease, the EV is defined as the additional expenditure (income) necessary to reach utility level u^1, given the initial set of prices. In terms of the indirect utility function, EV is the solution to

$$v\left(\boldsymbol{P}', M+EV\right) = v\left(\boldsymbol{P}'', M\right) = u^1 \,. \tag{3.23}$$

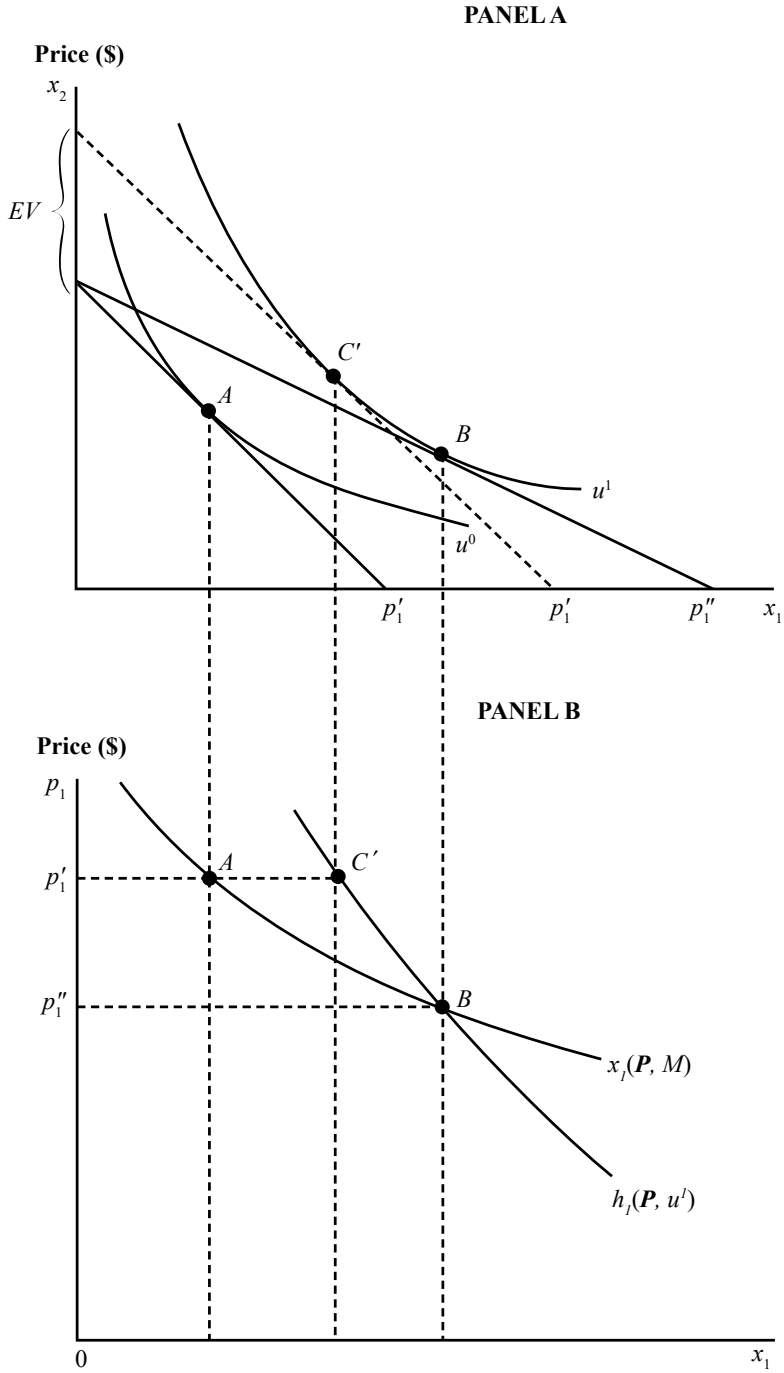

Figure 3.3 The equivalent variation and the Hicks-compensated demand

In Figure 3.3, the EV is the additional expenditure necessary to sustain point C' over point A at the initial prices, or

$$\begin{aligned} EV &= e\left(p_1', p_2, u^1\right) - e\left(p_1', p_2, u^0\right) \\ &= e\left(p_1', p_2, u^1\right) - M > 0. \end{aligned} \tag{3.24}$$

Since the money expenditure levels are the same at point A and point B—that is, $e\left(p_1', p_2, u^0\right) = e\left(p_1'', p_2, u^1\right)$—this can also be written as

$$EV = e\left(p_1', p_2, u^1\right) - e\left(p_1'', p_2, u^1\right). \tag{3.25}$$

In other words, although the EV is defined in terms of the monetary equivalent of a change from u^0 to u^1, it can also be measured by the change in expenditure associated with price changes given utility level u^1.

The EV can also be written as the integral of the marginal value measure (equation 3.11):

$$EV = \int_{p_1''}^{p_1'} \frac{\partial e\left(\boldsymbol{P}, u^1\right)}{\partial p_1} dp_1 . \tag{3.26}$$

The price derivative of the expenditure function (this time holding utility constant at u^1) generates another Hicks-compensated demand curve through point B in panel B of Figure 3.3. The area to the left of this Hicks-compensated demand curve between the two prices (area $p_1'C'Bp_1''$) is the equivalent variation welfare measure. In other words,

$$EV = \int_{p_1''}^{p_1'} h_1\left(\boldsymbol{P}, u^1\right) dp_1 . \tag{3.27}$$

As in the case of the CV, this measure does not require any assumption about the constancy of the marginal utility of income; and the measure for multiple price changes is path independent.

All of this discussion has been in terms of the welfare gain due to a price decrease. The derivation of the welfare cost of a price increase can be worked out in a symmetrical fashion. In general, for any price change, the CV welfare measure is the area to the left of the Hicks-compensated demand curve that passes through the initial position. The EV measure of the welfare change is the area to the left of the Hicks-compensated demand curve that passes through the final position.

A Comparison of the Three Measures

Although the Marshallian consumer surplus has some intuitive appeal as a welfare indicator, it does not measure either of the theoretical definitions of welfare change developed here. In general, it is not a measure of gain or loss that can be employed in a potential compensation test. The Marshallian surplus does lie between the CV and the EV, however, this opens the question of whether it can be a useful approximation to either of these other measures, a question that is taken up below in the subsection Consumer's Surplus Without Apology.

In contrast, the *CV* and the *EV* do represent welfare relevant measures. The *EV* is the monetary equivalent of a price change. It can be interpreted as an index of utility in the sense that it imputes the same monetary value to all changes from an initial position that result in the same final utility level. This is an ordinal utility index (Morey 1984). For example, suppose a change from initial position *A* to position *B* has an *EV* of \$10, while a change from *A* to *C* has an *EV* of \$20. It cannot be inferred that the second change conveys twice as much extra utility as the first change. This is because it evaluates all changes from an initial position at the same set of prices. The *CV* cannot be interpreted as an index of utility—rather, it measures the offsetting income change necessary to "prevent" a utility change. As Silberberg put it, "the [*EV*] imputes a dollar evaluation to a change in utility levels for a particular path of price changes, while the [*CV*] derives dollar values necessary to hold utility constant when prices change" (1972, 948).

The two measures *EV* and *CV* will be the same if the income elasticity of demand for good x_1 is zero. In this case, the ordinary and Hicks-compensated demand curves are identical. With positive income elasticity, the *EV* exceeds the *CV* for price decreases, but the *CV* exceeds the *EV* when price increases are considered. The difference between points *C* and *B* in Figure 3.2, and between points *A* and *C′* in Figure 3.3, is one of income level. If the income elasticity of demand for x_1 were zero, the income differences would have no effect on the purchase of x_1. The *CV* and the *EV* would be exactly equal, and they both could be measured by the area under the ordinary demand curve. The higher the income elasticity of demand for x_1, the larger the difference between the *EV* and the *CV*, and the larger the difference between either of the measures and the ordinary consumer surplus.

There is symmetry between the *CV* and the *EV* measures that can be seen by comparing Figures 3.2 and 3.3, and by comparing equation (3.21) with equation (3.24), and by comparing equation (3.18) with equation (3.25). For simplicity, let I represent the initial price set (with p_1') and let II represent the second price set (with p_1''). The *CV* for moving from I to II with u^0 as the reference utility level is exactly equal to the *EV* of moving from II to I with u^1 as the reference utility level. The *CV* is a welfare measure for the move from *A* to *B* via point *C*; the *EV* starts at point *B* and measures the reduction in income necessary to get to point *A*, and therefore u^0 via point *C′*. Similarly, the *EV* for the move from I to II is just equal to the *CV* starting at II and u^1, and moving to I.

This symmetry relates to the interpretation of *CV* and *EV* as measures of willingness to pay (WTP) and willingness to accept (WTA) compensation. The *CV* is sometimes described as the maximum willingness to pay for the right to purchase the good at the new price level (i.e., the lump sum payment that the individual would be willing to make that would just exhaust the potential for welfare gain from the new price). This description is accurate only for a price decrease. For a price increase, the *CV* defines the minimum payment to the individual sufficient to prevent a utility decrease; in other words, it defines a WTA measure. Similarly, the *EV* defines a WTA measure for a price decrease—that is, the sum of money the individual would require to voluntarily forgo a proposed price decrease. However,

Table 3.1 The implied rights and obligations associated with alternative price sets

Welfare measure	*Price increase*	*Price decrease*
EV – Implied property right in the change	WTP to avoid	WTA to forgo
CV – Implied property right in the status quo	WTA to accept	WTP to obtain

for a proposed price increase, the *EV* is a WTP measure—that is, the maximum sum of money that could be taken away from the individual—yielding a loss of utility equivalent to that caused by the price change. Whatever the direction of the price change, the *CV* takes the initial utility as the reference point.

These two measures can also be interpreted in terms of the implied rights and obligations associated with alternative price sets. The *CV* carries an implicit presumption that the individual has no right to make purchases at a new set of lower prices, but does have a right to the original price set in the case of price increases. In contrast, the *EV* contains the presumption that the individual has a right to (an obligation to accept) the new lower (higher) price set, and must be compensated (make a payment) if the new price set is not to be attained. Based on this interpretation of the two measures, some economists have argued that the choice between them is basically an ethical one—that is, one that depends on a value judgment as to which underlying distribution of property rights is more equitable (Krutilla 1967; Mishan 1976). All of this can be summarized as in Table 3.1.

For two alternative price changes, the welfare measures should be the same if both changes place the individual on the same higher indifference curve. However, if the two price changes place the individual on different indifference curves, the welfare measure should correctly indicate the preference ranking of the two alternatives. The *EV* measure always provides a consistent ranking in this sense, but the *CV* measure does not.

Figure 3.4 illustrates why this is the case. It shows an individual in equilibrium at point *A*, given prices and money income. Suppose that one policy proposal would increase the price of x_1 and decrease the price of x_2 simultaneously. The individual would achieve a new equilibrium at point *B*. The *CV* measure of the welfare change is shown as CV_{AB}. The second policy alternative would decrease the price of x_1 while increasing the price of x_2. This would lead to a new consumer's equilibrium at point *C*. Point *C* has been drawn on the same indifference curve as point *B*. Therefore, the measure of welfare change should be the same for the two policy alternatives. However, as can be seen by inspection, the *CV* for the second policy, CV_{AC}, is larger. The *CV* measure would indicate a preference for the second policy while the individual is in fact indifferent between the two policies. The *EV* gives the same welfare measure for the two policy alternatives. This is because the *EV* measure bases its comparison on a point on the indifference curve passing through the new equilibrium, but with the old prices. If two policies are on the same new indifference curve, the *EV* measure picks the same point for measuring the welfare effects for both policies.

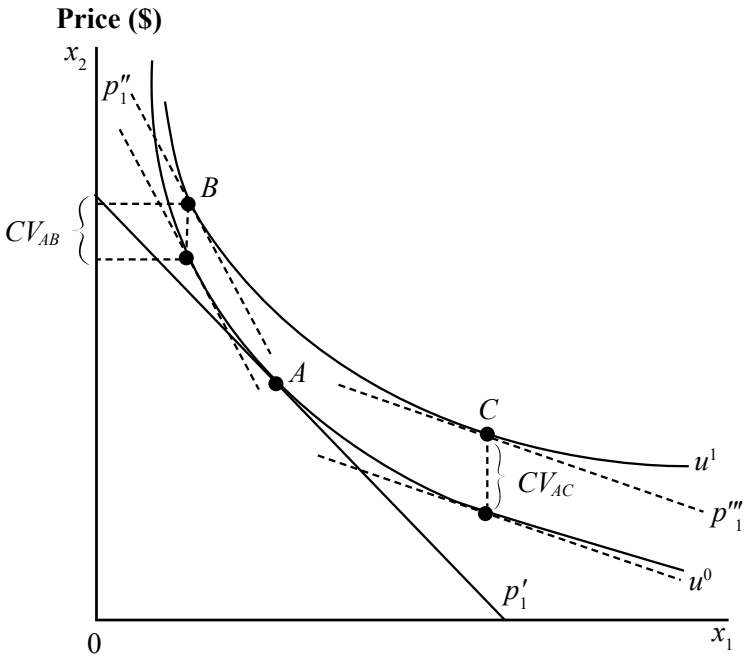

Figure 3.4 The compensating variation incorrectly ranks two alternative policies

If the question being asked by policymakers is, "does the proposed change pass the Kaldor potential compensation test?" then *CV* is the measure to use. The Kaldor potential compensation test is one form of potential Pareto improvement test that asks whether it is possible for the winners to fully compensate all of the losers from the proposed policy change and still leave someone better off. For each person, the *CV* gives the compensating income change required to maintain that person at his or her initial utility level. If the sum of what could be collected from all gainers exceeds the sum of the required compensations for losers, the proposal passes this form of the potential Pareto improvement test. The fact that the *CV* cannot rank consistently two or more policy changes is no obstacle to its use in this manner. This is because the potential Pareto criterion itself provides no basis for ranking two or more proposed policy changes. If two proposed changes both pass the Kaldor potential compensation test, the potential Pareto improvement criterion provides no basis for choosing between them.

On the other hand, if the question being asked by policymakers is, "does the policy pass the Hicks version of the potential compensation test?" then *EV* is the appropriate measure. The Hicksian test asks whether it is possible for the losers to bribe the gainers to obtain their consent to forgo the proposed policy change. The potential gainers would accept a bribe only if it were large enough to raise their utility by the same amount as the proposed policy would have. The offered

bribe would have to be as large as each individual's *EV* measure of welfare gain; and the maximum bribe that would be offered by the potential losers would be their *EV* measure of loss. Thus if the sum of the *EV* of all gainers exceeded the sum of the *EVs* of all losers, the proposal would pass the Hicks form of the potential compensation test. Also, since the Hicks form of the compensation test is based on the *EV* measure, it will consistently rank two or more policy changes, provided that society is indifferent as to the distribution of gains and losses across individuals.

Measurement

Simply put, the problem posed for applied welfare economics is that the desired welfare measures, the *CV* or the *EV*, are based on the unobservable Hicks-compensated demand functions, while the one measure based on the observed Marshallian demand functions is flawed as a welfare indicator. The typical practice had been to use the Marshallian surplus anyway, and to offer such justifications as "income effects are likely to be small"; "with only one price change, path dependence is not an issue"; and "it is the only measure we have and it is better than nothing." Then Robert Willig (1976), in a widely cited article, provided a

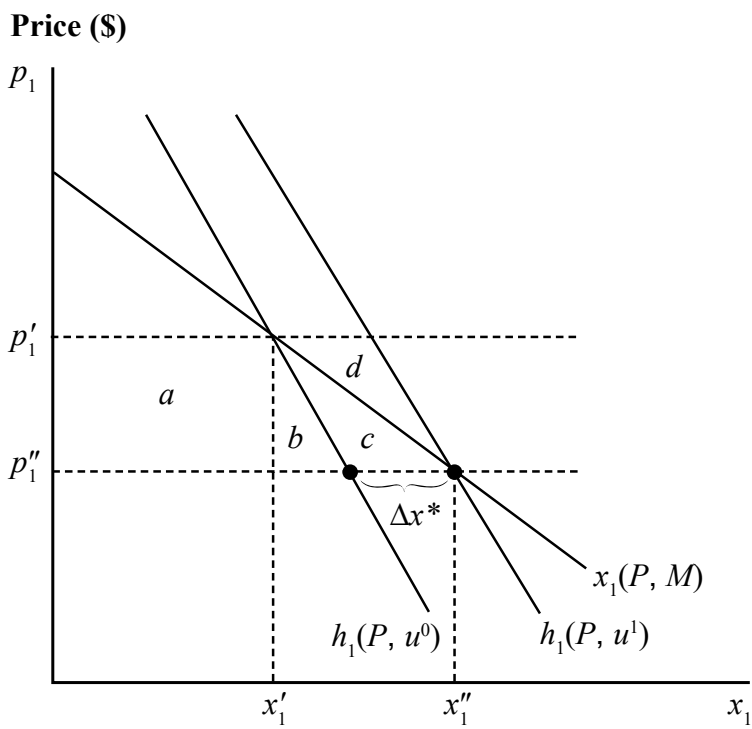

Figure 3.5 Deriving the Willig bounds for *S* as an approximation to *CV*

justification for using the Marshallian surplus by examining the magnitude of the differences between S and CV or EV under different conditions. Willig argued, "In most applications the error of approximation will be very small. In fact the error will often be overshadowed by the errors involved in estimating the demand curve" (1976, 589). Following his work, several authors have developed methods for direct calculation of the CV and EV from information contained in the ordinary demand function, either through a Taylor's series approximation (McKenzie and Pearce 1982; McKenzie 1983), or as exact measures through integration to obtain the indirect utility function and the expenditure function (see for example, Hausman 1981). The second subsection describes Hausman's contribution.

Consumer's Surplus without Apology

Willig (1976) has offered rigorous derivations of expressions relating CV, S, and EV. These expressions provide a way of calculating the magnitude of the differences among the three measures for given prices, quantities, and income. The differences among the three measures depend on the income elasticity of demand for the good in question and consumer surplus as a percentage of income. The differences among the measures appear to be small and almost trivial for most realistic cases. The differences are probably smaller than the errors in the estimation of the parameters of demand functions by econometric methods.

Willig's bounds for the approximation errors are based on the fact that the differences between S and CV or EV arise from an income effect on the quantity demanded; and the size of that effect depends on the change in real income brought about by the price change and on the income elasticity of demand for the good. This can be shown in a nonrigorous way for the case of one price change with the help of Figure 3.5. Although this exposition applies to the case of only one price change, the Willig expressions can be generalized to accommodate multiple price changes (Willig 1979), provided that a specific path of integration is chosen. In Figure 3.5, the ordinary and compensated demand curves are assumed to be linear. Let S represent the area $a + b + c$. So:

$$CV = a + b = S - c, \tag{3.28}$$

and

$$EV = a + b + c + d = S + d. \tag{3.29}$$

The errors in using S to approximate CV and EV are equal to the areas c and d respectively. For a price change from p_1' to p_1'', the factors influencing the size of the approximation error can be seen by examining the determinants of the area c:

$$CV - S = -c = -\tfrac{1}{2}\Delta p \cdot \Delta x^*, \tag{3.30}$$

where Δx^* is the income effect on the quantity demanded of x, which is associated with reducing income sufficiently to hold utility at u^0. Let ΔM^* represent this

income change. By definition, ΔM^* is CV. The definition of income elasticity of demand is

$$E_M = \frac{\Delta x}{\Delta M} \cdot \frac{M}{x} \cdot \tag{3.31}$$

Solving this expression for Δx^* gives

$$\Delta x^* = E_M \cdot x \cdot \frac{\Delta M^*}{M} = E_M \cdot x \cdot \frac{CV}{M} \cdot \tag{3.32}$$

Substituting this into equation (3.30), we obtain

$$CV - S = -\frac{\Delta p \cdot x \cdot E_M \cdot CV}{2M} \cdot \tag{3.33}$$

In general, for small changes in p, $\Delta p \cdot x \approx S$. This is strictly true for the linear demand curve when x is evaluated at the midpoint between x' and x''. Finally, dividing both sides by CV to express the error in percentage terms gives

$$\frac{CV - S}{CV} \simeq -\frac{E_M}{2} \cdot \frac{S}{M} \cdot \tag{3.34}$$

This is similar to the Willig expression for the approximation error. The principal difference is that it expresses the error as a percentage of CV, while Willig's term makes the error a percentage of S. It says that the error is proportional to the income elasticity of demand and consumer surplus as a percentage of income. A similar line of reasoning can be used to derive the relationship between EV and S.

Willig's analysis is more rigorous than this in that it takes into account the possibility that for finite changes in price and quantity, the income elasticity of demand may vary over the range of the price change. Willig derived rules of thumb for calculating the maximum error in using S as an approximation for EV or CV. The rules of thumb are applicable if the following conditions are met:

$$\begin{aligned} \frac{S}{M} \cdot \frac{\underline{E}_M}{2} &\leq 0.05 \\ \frac{S}{M} \cdot \frac{\overline{E}_M}{2} &\leq 0.05 \end{aligned} \tag{3.35}$$

and

$$\frac{S}{M} \leq 0.9 \,, \tag{3.36}$$

where $\underline{E}_M$ and $\overline{E}_M$ are the smallest and largest values, respectively, of the income elasticity of demand for the good in the region under consideration.

Given these conditions, the rule of thumb for CV is

$$\frac{S}{M} \cdot \frac{\underline{E}_M}{2} \leq \left| \frac{CV - S}{S} \right| \leq \frac{S}{M} \cdot \frac{\overline{E}_M}{2} \,, \tag{3.37}$$

and the rule of thumb for EV is

$$\frac{S}{M}\cdot\frac{\underline{E}_M}{2}\leq\left|\frac{S-EV}{S}\right|\leq\frac{S}{M}\cdot\frac{\overline{E}_M}{2}\ . \tag{3.38}$$

The first thing to note is the conditions under which these rules of thumb are valid. Consider equation (3.36) first. The change in consumer surplus as a percentage of income depends on the size of the price change, the price elasticity of demand, and expenditure on this good as a percentage of total income. The smaller the price change and the smaller the proportion of income spent on the good, the smaller S/M becomes. It can readily be shown that

$$\frac{S}{M}\leq\left|\frac{\Delta p}{p}\right|\cdot\frac{px}{M}\ . \tag{3.39}$$

From a given initial situation, S is largest when the demand curve is perfectly inelastic. Then $S=|x\cdot\Delta p|$ and (3.39) holds as an equality. With more elastic demand, $S<|x\cdot\Delta p|$ and the condition follows. For example, it shows that for a good absorbing 50 percent of total income and for a 100 percent price change, S/M cannot exceed 0.5, while for a 10 percent price change for a good absorbing 10 percent of income, S/M will be less than 0.1. Thus, condition (3.36) is likely to be satisfied except for very large price increases for goods with low price elasticities that also absorb a large proportion of the total budget.

As for the first condition, the smaller consumer surplus is as a percentage of income, and the smaller the income elasticity of demand is, the more likely it is that (3.35) be satisfied. For example, if consumer surplus is 5 percent of income, the income elasticity of demand can be as high as 2.0 and still satisfy (3.35). If S/M just barely satisfies condition (3.36), the income elasticity cannot exceed 0.11 to satisfy (3.35).

Assuming that conditions (3.35) and (3.36) hold, then let us turn to the rules of thumb. First, according to (3.35), the maximum error involved in using S as an approximation for either CV or EV is 5 percent. Second, the smaller the change in income elasticity over the range being considered, the more precise (3.37) and (3.38) are as statements of the error involved in using S rather than CV or EV. If the income elasticity of demand does not change over the range being considered, the left-hand and right-hand sides of (3.37) and (3.38) are equal to each other and the errors are zero, as discussed above. Finally, as the income elasticity of demand for the good decreases, the differences among ordinary consumer surplus, CV, and EV decrease, disappearing as E_M goes to zero.

Willig's analysis has been interpreted as providing a justification for using consumer surplus as an approximation of the CV or the EV. However, there are two reasons why one should be cautious about adopting the Willig approach to welfare measurement. The first has to do with limitations on the applicability of the Willig conditions to some kinds of problems of welfare measurement, including some of specific interest to environmental and resource economists.

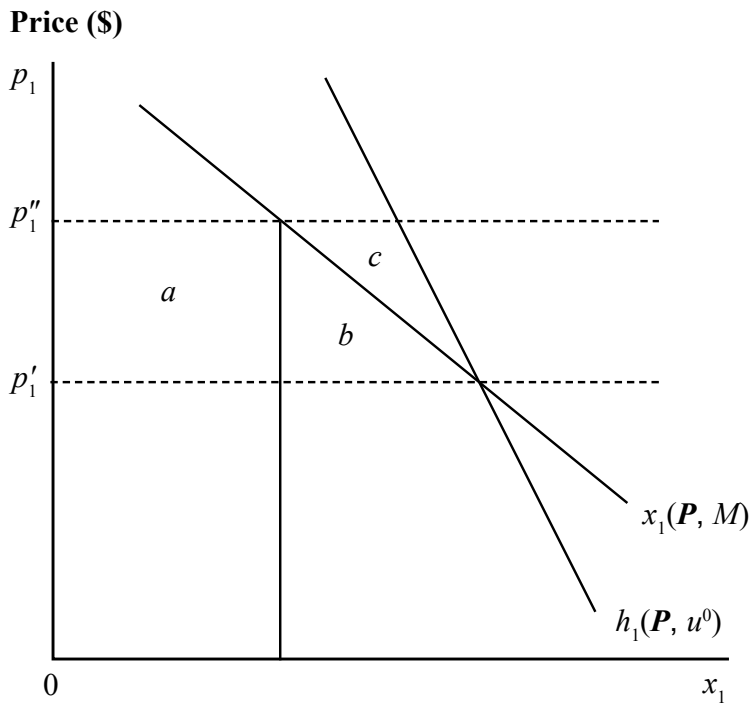

Figure 3.6 The Willig approximation and the error in estimating dead-weight losses

The second arises because of the recent development of new methods for obtaining exact measures of *CV* and *EV* from the same information that is required to use the Willig approximation.

The Willig conditions for valid approximation were developed for changes in *S* resulting from changes in the price of some market good. However, many environmental and resource policy issues require information on the total value of some environmental service as a measure of what would be lost if the resource were destroyed or diverted to some other use. For example, the economic cost of damming a river that provides whitewater canoeing and trout fishing would be measured by the total areas under the Hicks-compensated demand curves for these activities. This is equivalent to measuring the change in consumer surplus for a price increase from the present price to the vertical intercept of the Hicks-compensated demand curve. Bockstael and McConnell (1980) pointed out that for the linear demand function, the income elasticity of demand goes to infinity as the price approaches the vertical intercept; and thus, the approximation error cannot be calculated. In a comment on Bockstael and McConnell (1980), Hanemann (1980) showed that if the parameters of the Marshallian demand function were known, it was unnecessary to compute the Willig approximation error, since the *CV* could be calculated directly. In

this, Hanemann (1980) apparently anticipated the analysis of Hausman (1981), discussed below.

For some questions, the variable of interest to policymakers is not *CV* but some fraction of *CV*—for example, the dead weight loss associated with a tax on a commodity. Suppose an excise tax raises the price of a good from p' to p'', as shown in Figure 3.6. The consumer's loss as measured by *CV* is the area $a + b + c$, but only $b + c$ is an efficiency loss, since a is a revenue transfer to the government. If the ordinary demand curve is used to approximate the consumer loss, the area c is the error. If the Willig conditions are satisfied, c is an acceptably small percentage of *S* and *CV*; but it can be an unacceptably large percentage of the true dead weight loss.

The second reason for being cautious about using the Willig approximation is that better methods of welfare measurement now exist. If the demand functions being used to calculate *S* reflect utility maximizing behavior on the part of individuals, they should satisfy the integrability conditions. If this is the case, it is possible to calculate *CV* and *EV* directly without approximation. On the other hand, if the demand functions do not satisfy the integrability conditions, then it is inappropriate to use the Willig approximations, since their derivation was also based on the assumption of utility-maximizing behavior.

Exact Welfare Measurement

Hausman (1981) presented a procedure for exact welfare measurement based on the recovery of the parameters of the utility function from data on consumers' demand. His procedure, which was developed for the case of only one price change, involves four steps. The first involves combining the ordinary demand function and Roy's identity to obtain a partial differential equation:

$$x_1(\boldsymbol{P}, M) = -\frac{\partial v(\boldsymbol{P}, M)/\partial p_1}{\partial v(\boldsymbol{P}, M)/\partial M} \tag{3.40}$$

If the utility function is separable so that the demand function contains only its own price argument, and if the demand function is linear, this becomes:

$$a - (b \cdot p_1) + (c \cdot M) = -\frac{\partial v(\boldsymbol{P}, M)/\partial p_1}{\partial v(\boldsymbol{P}, M)/\partial M}, \tag{3.41}$$

where the parameters a, b, and c are estimated econometrically, and where p_1 and M are deflated by an appropriate index of the other prices. Changes in p_1 and M that involve moving along an indifference curve must satisfy

$$\left[\frac{\partial v(\cdot)}{\partial p_1(t)}\frac{dp_1(t)}{dt}\right] + \left[\frac{\partial v(\cdot)}{\partial M(t)}\frac{dM(t)}{dt}\right] = 0, \tag{3.42}$$

where t defines a path of price changes. Rearranging this expression, substituting into (3.41), and using the implicit function theorem gives

$$\frac{dM(p_1)}{dp_1} = a - (b \cdot p_1) + (c \cdot M), \tag{3.43}$$

the solution of which is

$$M(p_1) = k \cdot \exp(cp_1) - \frac{1}{c}\left[a - (b \cdot p_1) - \frac{b}{c}\right], \tag{3.44}$$

where k is the constant of integration, which depends on the initial level of utility. If units are arbitrarily chosen so that k is the initial utility level, the quasi-indirect utility function and quasi-expenditure function follow directly:

$$u = k^0 \cdot \exp(-cp_1)\left\{\left(M + \frac{1}{c}\right)\left[a - (b \cdot p_1) - \frac{b}{c}\right]\right\} \tag{3.45}$$

and

$$e = k^0 \cdot \exp(cp_1) - \frac{1}{c}\left[a - (b \cdot p_1) - \frac{b}{c}\right]. \tag{3.46}$$

These expressions are termed "quasi" functions because they do not contain information about the effects of the prices of other goods on utility or expenditure. Hausman's method depends on the ability to solve the differential equation that is obtained from Roy's identity. Hausman has shown a method of solution for the case when only one price changes, and has discussed in general terms the solution in the case of multiple price changes.

Conclusions

Selection of a welfare measure has long involved questions both of appropriateness and of practicality. The Marshallian surplus measure was frequently chosen on the grounds of practicality, even though it was recognized that the measure was inappropriate in that it did not answer any specific well-formed welfare question. Willig's development of the bounds for the errors of approximation in using S gave encouragement to this practice. However, quickly on its heels have come new approaches to exact welfare measurement that offer the opportunity to calculate the more appropriate CV and EV measures directly.

One question related to practicality remains, however—do we know enough about the functional form of the utility function to implement the exact measurement methods? Assuming a functional form for the system of demand functions for purposes of estimation is equivalent to assuming the functional form of the underlying utility function. One approach is to assume a specific functional form for the utility function or indirect utility function, and to derive the demand functions for estimation. If this is the approach taken, then plugging the estimated parameters back into the utility function to calculate welfare changes is straightforward, provided the parameter estimates of the demand function satisfy

the integrability conditions. Since researchers have been reluctant to specify the functional form of the utility function, one alternative has been to specify so-called *flexible forms* for the indirect or direct utility function (for example, Deaton and Muellbauer 1980). Again, if the integrability conditions are satisfied, deriving "exact" welfare measures from the "approximate" flexible functional form of the utility function is straightforward. The alternative is to seek guidance from the data by selecting the functional form for the demand functions based on goodness-of-fit and consistency with the restrictions imposed by theory.

Welfare Measures for Continuous Goods: Quantity Changes

Many environmental policy proposals involve changes in either the quantities or the qualities of nonmarket environmental goods and services, rather than changes in the price of a marketable good. From the individual's point of view, the most important characteristic of some environmental goods is that they are available only in fixed, unalterable quantities. These quantities act as constraints on each individual's choice of a consumption bundle. The analysis of this class of problems is often referred to as the theory of choice and welfare under quantity constraints (Johansson 1987). The imposition of quantity constraints raises some new issues in the theory of choice and welfare measurement. The analysis of these problems has evolved out of the theory of rationing as initially developed by Tobin and Houthakker (1950/1), and Neary and Roberts (1980).

This section provides a brief description of the model of individual preferences and choice under imposed quantity constraints. The corresponding measures of welfare impacts for changes in the quantities of imposed goods are then derived. These measures are essentially similar to the compensating and equivalent surplus measures for price changes presented in Hicks (1943), but the change being considered is one of a quantity or quality change, rather than price. As mentioned earlier, Hicks referred to these measures as compensating or equivalent "surplus," and this terminology convention is continued in this chapter. The section closes with a brief discussion of the value of changes in q when q is a bad.

The Basic Model

Consider an individual whose utility function has the following form:

$$u(\boldsymbol{X}, \boldsymbol{Q}), \tag{3.47}$$

where $\boldsymbol{X} = (x_1, \ldots, x_J)$ is the vector of private goods quantities, and $\boldsymbol{Q} = (q_1, \ldots, q_K)$ is a vector of environmental and resource service flows (unpriced public goods) that is exogenous to the individual. It is possible that there is a positive price for at least some of the elements in $\boldsymbol{Q}$; but to keep the exposition simple, all prices for elements of $\boldsymbol{Q}$ are assumed to be zero. Let $\boldsymbol{P} = (p_1, \ldots, p_J)$ be the vector of prices for $\boldsymbol{X}$. The individual maximizes utility subject to a budget constraint

$$\boldsymbol{P} \cdot \boldsymbol{X} = M \,, \tag{3.48}$$

where M is money income. This yields a set of conditional demand functions for the marketed goods:

$$x_j = x_j\left(\boldsymbol{P}, M, \boldsymbol{Q}\right). \tag{3.49}$$

In general, $\boldsymbol{Q}$ will be an argument in these conditional demand functions, along with prices and income. The term "conditional" refers to the fact that these functions are conditioned upon the imposed $\boldsymbol{Q}$.

Inserting the conditional demand functions into the utility function gives the conditional indirect utility function

$$v = v\left(\boldsymbol{P}, M, \boldsymbol{Q}\right). \tag{3.50}$$

Inverting the conditional indirect utility function for M yields a conditional expenditure function that gives the minimum expenditure on market goods required to produce utility level u, given $\boldsymbol{P}$ and $\boldsymbol{Q}$. This is

$$e = M = e\left(\boldsymbol{P}, \boldsymbol{Q}, u\right). \tag{3.51}$$

For simplicity, in what follows $\boldsymbol{Q}$ is assumed to consist of only one element, q. In order to make graphic presentations of some of the key points, it is assumed that $\boldsymbol{X}$ is the numeraire, represented as x with a price of 1. Finally, it is assumed that at the given prices and income, the individual would choose more of q if given the option (i.e., q is a "good").

To begin with, the marginal value of a small increase in q is the reduction in income that is just sufficient to maintain utility at its original level. If w_q is the marginal value or marginal willingness to pay for a change in q, it is given by the derivative of the restricted expenditure function with respect to q or

$$w_q = -\frac{\partial e}{\partial q}. \tag{3.52}$$

The right-hand side of this expression is also equal (in absolute value) to the slope of the indifference curve through the point at which the welfare change is being evaluated. There are several ways to present compensating surplus (CS) and equivalent surplus (ES) for changes in quantity-constrained goods.

The first way is based on the conditional indirect utility function. The CS and ES measures are defined implicitly as the solutions to the following expressions: CS is the solution to

$$v\left(\boldsymbol{P}, M, q^0\right) = v\left(\boldsymbol{P}, M - CS, q^1\right), \tag{3.53}$$

and ES is the solution to

$$v\left(\boldsymbol{P}, M + ES, q^0\right) = v\left(\boldsymbol{P}, M, q^1\right). \tag{3.54}$$

These two measures can also be defined in terms of the conditional expenditure function. For a change in q, CS is

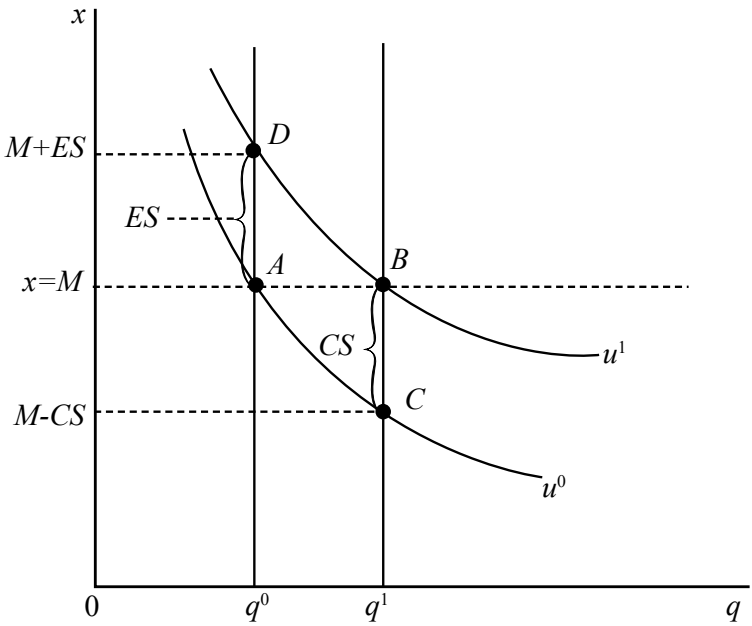

Figure 3.7 Compensating and equivalent surpluses for a change in q

$$CS = e\left(\boldsymbol{P}, q^0, u^0\right) - e\left(\boldsymbol{P}, q^1, u^0\right) = M - e\left(\boldsymbol{P}, q^1, u^0\right). \tag{3.55}$$

The *ES* measure given by the conditional expenditure function is

$$ES = e\left(\boldsymbol{P}, q^0, u^1\right) - e\left(\boldsymbol{P}, q^1, u^1\right) = e\left(\boldsymbol{P}, q^0, u^1\right) - M. \tag{3.56}$$

ES and *CS* are shown graphically in Figure 3.7. The increase in q enables the individual to reach point *B* with utility equal to u^1. The *CS* is the distance *B-C*. Alternatively, if income increased by the *ES* value while holding q constant, the individual could achieve u^1 at point *D*. Thus, *ES* is the distance *A-D*.

A second way to derive the *ES* and *CS* measures is also based on the conditional expenditure function. The value of a nonmarginal change in q is the integral of this function taken over the relevant range, or

$$W_q = -\int_{q^0}^{q^1} \frac{\partial e\left(\boldsymbol{P}, q, u^t\right)}{\partial q} dq. \tag{3.57}$$

This is either a *CS* or an *ES* measure, depending on whether $t = 0$ or $t = 1$.

Before leaving this section, note that there are two ways in which more q could be a bad, rather than a good, for an individual. The first way is when q has a price greater than zero and the individual would prefer to have less than the quantity being imposed given that price. The welfare measures *ES* and *CS* are still defined in the same way, but now they are negative for increases in q and positive for decreases in q.

The second way in which q can be a bad is the more fundamental one—it is when the marginal utility of q is negative. Even at a zero price, the individual would prefer to receive a smaller quantity. In both cases, the welfare measures ES and CS are defined in the same way, and again they are negative for increases in q and positive for decreases in q. In addition, all of the discussion of exact welfare measurement techniques and approximations carries over with appropriate changes to the case of q as a bad.

Welfare Measures for Discrete Goods

In the first part of this chapter the models described for changes in price exploited the marginal equalities revealed when individuals optimize over choice variables that are continuously variable. This is not always a realistic way to model the individual choice problem. Some problems are better viewed as involving the choice of one option from a range of discrete alternatives. For example, the choice might be whether or not to take a once-in-a-lifetime cruise around the world, or whether to travel to work by private auto, bus, or on foot. The solutions of discrete choice problems of this sort are essentially corner solutions. Consequently, there are no tangencies from which a marginal rate of substitution can be inferred. Discrete choice models have been developed both to predict individuals' behaviors in these choice contexts and to draw inferences about welfare change on the basis of observed choices.

In this section, a simple discrete choice model is presented, and measures of welfare change and value are derived from the model. Welfare measures for both price changes and quantity changes are considered. Subsequent chapters present detailed discussions of applications and estimation approaches. A wide range of environmental problems and decision making can be represented in a discrete choice setting, including: voting yes or no on a referendum question; accepting or rejecting a hypothetical offer for an environmental commodity; the choice of which of several alternative houses to live in based in part on environmental quality in their vicinity; and the choices of whether or not to undertake a specific recreation activity or to visit a specific recreation site. For expositions of the specification, estimation, and interpretation of discrete choice models generally, see Ben-Akiva and Lerman (1985) and Train (2009). Hanemann (1999) gave a more advanced exposition in the context of valuing environmental changes. See also Johansson, Kriström, and Mäler (1989) and Hanemann (1989).

Consider an individual's decision regarding which one of several alternative goods to purchase. The individual can choose one good from a set of $\mathcal{J}$ alternatives ($j = 1, \ldots, \mathcal{J}$), where each good has a vector of environmental quality attributes $\boldsymbol{Q}_j$ associated with it. The price for good j is p_j . The individual gets utility from the discrete good chosen and the consumption of a numeraire good.

With this construction, a conditional utility function associated with each alternative can be written as:[3]

$$u_j = u_j(M, p_j, \boldsymbol{Q}_j), j = 1,...,\mathcal{J} \ . \tag{3.58}$$

The individual will choose to consume the alternative that yields the highest utility; that is, the chosen alternative "j" will satisfy

$$u_j(M, p_j, \boldsymbol{Q}_j) > u_k(M, p_k, \boldsymbol{Q}_k), \quad j,k = 1,...,\mathcal{J} \ . \tag{3.59}$$

It is straightforward at this point to implicitly define the compensating and equivalent variation associated with a price change for one or more of the alternatives. Specifically, the compensating variation associated with a decrease in all prices of the discrete alternatives can be written implicitly as:

$$\underset{j}{\text{Max}}\ u_j\left(M, p_j^0, \boldsymbol{Q}_j\right) = \underset{j}{\text{Max}}\ u_j\left(M - CV, p_j^1, \boldsymbol{Q}_j\right), \tag{3.60}$$

where superscript "0" indicates the original price, and superscript "1" indicates the new, lower set of prices. The expression makes clear that the option chosen after compensation is paid could also differ from either the original alternative or the choice without compensation. Likewise, equivalent variation can be written as:

$$\underset{j}{\text{Max}}\ u_j\left(M + EV, p_j^0, \boldsymbol{Q}_j\right) = \underset{j}{\text{Max}}\ u_j\left(M, p_j^1, \boldsymbol{Q}_j\right) \ , \tag{3.61}$$

where the base level of utility is the utility associated with the new price vector rather than the original.

It is also straightforward to construct the compensating and equivalent surplus measures associated with a change in the vector of quality attributes associated with each alternative:

$$\begin{aligned} &\underset{j}{\text{Max}}\ u_j\left(M, p_j, \boldsymbol{Q}_j^0\right) = \underset{j}{\text{Max}}\ u_j\left(M - CS, p_j, \boldsymbol{Q}_j^1\right), \\ &\underset{j}{\text{Max}}\ u_j\left(M + ES, p_j, \boldsymbol{Q}_j^0\right) = \underset{j}{\text{Max}}\ u_j\left(M, p_j, \boldsymbol{Q}_j^1\right). \end{aligned} \tag{3.62}$$

A common representation of the utility function is additive. By also recognizing that the budget constraint implies that the amount of the numeraire that can be consumed when alternative "j" is chosen is $M - p_j$, the conditional utility function can be written as:

$$u_j = \beta(M - p_j) + \tilde{u}_j(\boldsymbol{Q}_j), \quad j = 1,...,\mathcal{J} \ , \tag{3.63}$$

where β can be interpreted as the marginal utility of income, and $\tilde{u}_j(\boldsymbol{Q}_j)$ is a function representing the utility associated with the quality aspects of the alternative. With this specification, the compensating and equivalent surpluses for

3 Note that this is a conditional *indirect* utility function. We depart from our standard notation used throughout the rest of the book and use $u(\cdot)$ to denote an indirect utility function in this case for consistency with the established literature in this area.

a quality change are identical (a direct result of the constant marginal utility of income, β) and can be written as:

$$CS = ES = \frac{1}{\beta}\{\underset{j}{\text{Max}}\, u_j(M, p_j, \boldsymbol{Q}_j^1) - \underset{j}{\text{Max}}\, u_j(M, p_j, \boldsymbol{Q}_j^0)\}, \quad j = 1,\ldots,\mathcal{J}. \tag{3.64}$$

Expression (3.64) is intuitively appealing, as it says that the compensating and equivalent variation associated with a quality change is simply the difference in utility from the most desirable alternatives before and after the change, divided by the marginal utility of income. The marginal utility of income acts to monetize the utility difference.

Thus far, the discrete choice behavioral model and associated welfare measures have been presented in a deterministic form, just as the behavioral model underlying the continuous demand functions and their associated welfare measures were presented earlier in this chapter. Typically, however, analysts employing the discrete choice model recognize that there are individual characteristics and/or omitted variables that are not observable to the researcher, but are known to the individual making the decision. To incorporate this idea, an additive error can be added to the observable component

$$u_j = v_j(M, p_j, \boldsymbol{Q}_j) + \varepsilon_j,\ j = 1,\ldots,\mathcal{J}, \tag{3.65}$$

where ε_j is a random, unobservable component of utility. As before, utility maximizers will choose the alternative that yields the highest utility, but from the perspective of the analysis, the utility is now random. This "random utility maximization" model, or RUM model (Thurstone 1927; Marschak 1960; McFadden 1974, 1978, 1981), implies the probability that the individual chooses to purchase alternative "k" can be expressed as the probability that the utility associated with k is greater than the utilities associated with all the other alternatives:

$$\Pr(k) = \Pr\begin{bmatrix} v_k(M - p_k, \boldsymbol{Q}_k) \\ + \varepsilon_k > v_j(M - p_j, \boldsymbol{Q}_j) + \varepsilon_j \end{bmatrix}, \forall j \neq k. \tag{3.66}$$

McFadden (1974) demonstrated that if the error terms are independently and identically distributed (i.i.d.) with a Type I Extreme Value distribution, a logistic distribution results and this probability can be written simply, as follows:

$$\Pr(k) = \frac{e^{v_k(M-p_k, \boldsymbol{Q}_k)}}{e^{v_k(M-p_k, \boldsymbol{Q}_k)} + \sum_{j \neq k}^{\mathcal{J}} e^{v_j(M-p_j, \boldsymbol{Q}_j)}} = \left(1 + \sum_{j \neq k}^{\mathcal{J}} e^{-\Delta v_{jk}}\right)^{-1}$$

$$\text{where} \quad \Delta v_{jk} = v_j\left(M - p_j, \boldsymbol{Q}_j\right) - v_k\left(M - p_k, \boldsymbol{Q}_k\right). \tag{3.67}$$

The logit model of choice implies certain restrictions on individuals' choices and preferences. The most notable is that choices must have the property of the

Independence of Irrelevant Alternatives (IIA). This issue, and a host of additional topics related to interpretation and estimation of RUMs, will be discussed in later chapters.

The introduction of an error term complicates welfare computation since only the probability of choosing a particular alternative under a price or quality change can be considered. A general expression for the compensating variation associated with a price change is:

$$\underset{j}{\text{Max}}\left[v_j(M, p_j^0, \boldsymbol{Q}_j) + \varepsilon_j\right] = \underset{j}{\text{Max}}\left[v_j(M - CV, p_j^1, \boldsymbol{Q}_j) + \varepsilon_j\right], \tag{3.68}$$

where $CV = CV\left(M, \boldsymbol{P}^0, \boldsymbol{P}^1, \boldsymbol{Q}_j, \varepsilon\right)$ and $\varepsilon = \left(\varepsilon_1, \ldots, \varepsilon_{\mathcal{J}}\right)$ denotes the full vector of error terms. A corresponding equivalent expression for EV can be written. As the notation indicates, this welfare measure will itself be a random variable and its expected value can be computed (Small and Rosen 1981; Hanemann 1984). Using the linear functional form identified in (3.63), compensating and equivalent variations are equal to each other. If in addition, the error terms are Type I Extreme Value, then the mean CV and EV terms take a particularly simple form, with:

$$\overline{CV} = \overline{EV} = \frac{1}{\beta}\left\{\ln\left[\sum_{j=1}^{\mathcal{J}} e^{v_j^1}\right] - \ln\left[\sum_{j=1}^{\mathcal{J}} e^{v_j^0}\right]\right\}, \tag{3.69}$$

where $v_j^t = v_j\left(M, p_j^t, \boldsymbol{Q}_j\right)$ for $t = 0, 1$. Similar calculations can be used to obtain the value of adding or deleting a site with a specified set of characteristics from the individual's choice set. For the addition of site $\mathcal{J} + 1$, the expression is

$$\overline{CV} = \overline{EV} = \frac{1}{\beta}\left\{\ln\left[\sum_{j=1}^{\mathcal{J}+1} e^{v_j}\right] - \ln\left[\sum_{j=1}^{\mathcal{J}} e^{v_j}\right]\right\}, \tag{3.70}$$

and for deleting site $\mathcal{J}$, the expression is

$$\overline{CV} = \overline{EV} = \frac{1}{\beta}\left\{\ln\left[\sum_{j=1}^{\mathcal{J}-1} e^{v_j^1}\right] - \ln\left[\sum_{j=1}^{\mathcal{J}} e^{v_j^0}\right]\right\}. \tag{3.71}$$

These measures are examples of compensating and equivalent variation approaches to defining a welfare measure using a random utility framework. Hanemann described two such approaches and examined the relationships among them (Hanemann 1999, 43–48).

When CV and EV Diverge: Willingness to Pay versus Willingness to Accept Compensation

The results from Willig discussed earlier imply that measures of compensating variation (or surplus) should in theory generally be very close to their associated

equivalent variation (or surplus) measures. Since these measures have willingness to pay for and willingness to accept compensation interpretations, another way to say the same thing is that WTP to acquire a good or price change should typically approximately equal WTA to do without the change. However, there is a substantial body of evidence from stated preference studies, laboratory experiments, and field experiments that suggests that differences between WTP and WTA for the same good can be quite large (Horowitz and McConnell 2002; Sayman and Onculer 2005). Efforts at explaining these differences have taken several paths.

One argument is that these divergences do not represent actual divergences in preferences but reflect experience with the good and the trading environment in which the values are elicited. List (2003, 2004) studied the divergence in an actual marketplace and found that the disparity is highly correlated with experience in the market: those who have extensive experience in buying and selling the good (sports memorabilia at trade shows) exhibit no meaningful disparity. Focusing on the experimental environment in which these values are elicited, Plott and Zeiler (2005) argued that when a full suite of experimental controls is employed, the divergence between WTP and WTA disappears. They presented findings from three experiments to support their argument and concluded that the differences between WTP and WTA reported in the literature relate to misconceptions that subjects have about the task they faced in the experiment, rather than representing a reflection of true value disparity. Fudenberg, Levine, and Maniadis (2012) undertook a similar set of experiments (though their focus was on "anchoring" effects) and found evidence for the existence of the disparity, albeit of smaller size than many previous studies. Other authors have suggested and studied explanations that relate to the value elicitation environment (Hoehn and Randall 1987; Kolstad and Guzman 1999; Guzman and Kolstad 2007).

A second path involves examining the theory of preferences and value more closely to see whether theory predicts the large disparities between true WTA and WTP. One example of this is in the work of Hanemann (1991, 1999). He has shown that the price flexibility of income can be expressed as the ratio of two other terms:

$$E_q = \frac{E_M}{\sigma_q}, \tag{3.72}$$

where σ_q is the aggregate Allen–Uzawa elasticity of substitution between q and the composite commodity X and E_M is the income elasticity of demand for q. If the elasticity of substitution (a measure of the curvature of the indifference curve between q and private goods) is low, σ_q can be close to zero. This can lead to a high value for E_q and a large difference between *CS* and *ES*. However, Hanemann's analysis does not explain the persistent differences between the two measures in experiments with simulated markets involving commonplace goods such as lottery tickets, coffee mugs, and pens (see Knetsch and Sinden 1984; Kahneman, Knetsch, and Thaler 1990).

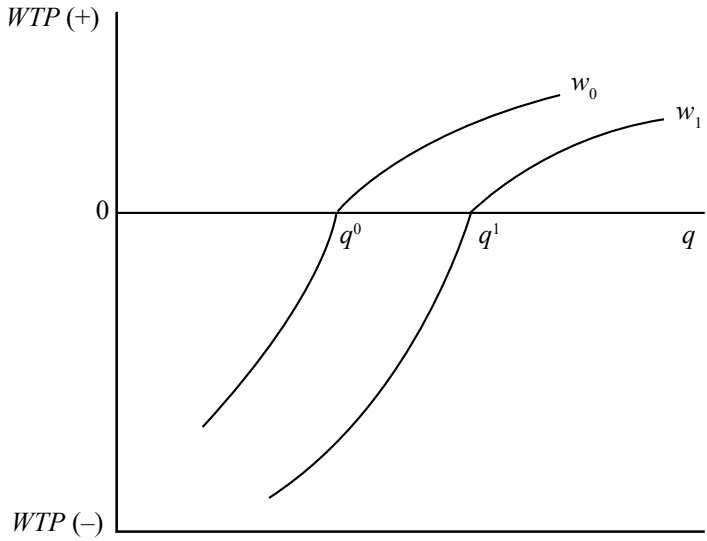

Figure 3.8 The value function and the endowment effect

Zhao and Kling (2001, 2004) have also offered a possible explanation that is largely consistent with the standard paradigm. They considered consumers who make decisions about whether to buy or sell goods whose value is uncertain to them when they have the opportunity to delay the decision and gather more information in the meantime. They demonstrated that there are conditions under which this will lead to lower WTP values and higher WTA values than theory would predict in the absence of this potential for learning. The dynamic welfare measures they derived will be further discussed in Chapter 5.

A final approach, and the one that seems to have gained the most traction, has been to move further from standard economic theory. Thaler (1980) proposed that the reconciliation of theory with observation can be brought about by postulating an "endowment effect" on individuals' valuation functions and a kink in this function at the status quo point. He suggested that this is a reasonable extension and generalization of the prospect theory of Kahneman and Tversky (1979) to choices not involving uncertainty. The idea of the endowment effect and the differential valuation of gains and losses can be shown with the aid of Figure 3.8. The horizontal axis shows the quantity of an environmental good q. The vertical axis shows the compensating welfare measure for changes in q. This measure is positive (WTP) for increases and negative (WTA) for decreases from some status quo point. Suppose that the status quo is q_0. The associated valuation function w_0 shows the monetary payment (compensation) that holds utility constant for a given increase (decrease) in q from q_0. This function is kinked at the status quo point of q_0, showing that the marginal valuation of increases in q is substantially

lower than the marginal valuation of losses from q_0. A change in the endowment of q from q_0 to q_1 shifts the valuation function. In addition, as Figure 3.8 shows, the willingness to pay for an increase from q_0 to q_1 is substantially less than the required compensation for the decrease from q_1 to q_0.

In conclusion, although the observed large differences between WTP and WTA can be explained by replacing the standard utility model with one that incorporates an endowment effect, it is not clear that this is always necessary. These differences can also be explained by the absence of close substitutes in the case of unique and perhaps irreplaceable resources and as the rational response to uncertainty and the high cost of information about preferences.

Aggregation and Social Welfare

Assume now that we have obtained measures of the welfare changes, either plus or minus, for all individuals. How can we use that information to make choices about public policy alternatives? To put the question in its most profound sense, what is the appropriate relationship between the welfare of individuals and the social welfare? What follows is a brief review of alternative social welfare criteria. Since the main concern of this book is with measurement, the question of social welfare criteria—that is, how to use the measures—is off the main track. For a more extensive discussion of the problem, see Mishan (1960), especially section III, and Boadway and Bruce (1984).

In the literature on welfare economics there are basically four ways to approach this question. The first approach to the question is the so-called Pareto criterion. Only policy changes that make at least one person better off (that is, an individual experiences a positive welfare change) and make no individual worse off (that is, no individual experiences a negative welfare change) pass this criterion. This criterion deliberately rules out any attempts to add up, or otherwise make commensurable, the welfare measures of different individuals. Since virtually all actual public policy proposals impose net costs on at least some individuals, most policy actions by the state could not be accepted under this criterion. This would be particularly true in the environmental area, where environmental management costs are often channeled through the production sector while benefits accrue to households in the form of increased levels of environmental services. It is unlikely that this would result in a pattern of incidence of benefits and costs in which no one would lose. The restrictive features of the Pareto criterion have stimulated an ongoing search for a welfare criterion that would justify the state doing certain things that at least some people feel it should be able to do.

The second approach to the question was proposed in slightly different forms by Kaldor (1939) and Hicks (1939)—these are the two different forms of a potential compensation test discussed earlier. Let us review these tests in the present context of aggregation and social welfare.

As noted earlier, the Kaldor version of the test asks whether those who gain by the policy can fully compensate for the welfare losses of those who lose by the

policy. The Kaldor version of the test would be satisfied if the sum of all individual *CV* and *CS* measures of welfare changes were greater than zero. The criterion is essentially one of potential Pareto improvement, since if the compensation were actually paid no one would lose from the policy.

The Hicks version of the potential compensation test asks whether those who lose from the policy could compensate the gainers for a decision not to proceed with the policy. If the answer is yes, the policy should be rejected according to the Hicks criterion. If the policy was rejected and compensation was actually paid, those who would have gained from the policy would be just as well off as if the policy had been adopted, and those who would have lost are at least as well off as they would have been with the policy. The Hicks version of the test takes acceptance of the project as its reference point. In effect, it is a decision to forgo the project that creates the gains and losses that are relevant to the Hicksian version of the potential Pareto improvement criterion.

Should compensation actually be paid in either the Kaldor or Hicks cases? If one thinks the answer should be yes, then the compensation test is transformed into a variation of the Pareto criterion in which the state serves to enforce the taxes and transfer payments that are necessary to ensure that no one actually experiences a welfare loss, assuming that such taxes and transfers would be costless. If one thinks that the answer should be no, this is equivalent to, in effect, assuming that all individual welfare changes are commensurate and can be summed together into an aggregate measure of welfare change. This is the efficiency criterion of the new welfare economics. According to the efficiency criterion, the objective of social policy is to maximize the aggregate value of all of the goods and services people receive, including environmental and resource services. One justification for the Hicks–Kaldor potential compensation test is that a large number of efficient projects will spread benefits sufficiently wide so that everyone is a net gainer from the set of projects taken as a whole, even though some might be losers on individual projects. See Polinsky (1972) for an interesting development of this line of reasoning.

Alternatively, one might believe that whether compensation should be paid depends upon who has to pay and who gets the benefits. This requires consideration of the equity (fairness) in the distribution of income as an element in the evaluation of social policy. The third approach to the question of social welfare criteria, proposed by Little (1957), makes explicit the concern for equality. He proposed a twofold test. First, does the policy pass the Kaldor test? Second, does the resulting change improve the distribution of income? The Little criterion legitimizes a concern with the distributional effects of changes in resource allocation, but it does not resolve the question of what constitutes an improvement.

The fourth approach to the question involves an attempt to make specific social judgments regarding equity, and to introduce equity considerations systematically into the evaluation of social policy. The most common proposal calls for the establishment of a social welfare function that gives different weights to individual welfare changes according to the relative deservingness of the different individuals (Eckstein 1961; Haveman and Weisbrod 1975). Of course, the main problem with

the social welfare weight approach is the determination of the weighting function (Freeman 1971).

Nevertheless, willingness to make explicit value judgments about equity makes it possible to consider a wider range of policy choices. For example, if one opts for the Pareto criterion or the potential compensation version of the Hicks–Kaldor test, one rules out the possibility of accepting a project that has a sum of individual welfare changes that is less than zero, but would substantially improve the distribution of income. An example of such a policy would be one that imposes a welfare loss of $1,000 on a millionaire while bringing benefits of $99 to each of ten impoverished orphans. A welfare-weighting function could approve negative sum policies like this, provided that the weights given to the beneficiaries were sufficiently greater than the welfare weights of the losers. In addition, neither of these criteria would reject a project that imposes costs on no one, but distributes benefits only to the richest in our society. Some might make the value judgment that this, in itself, is undesirable. A social welfare function that included some measure of inequality of the aggregate distribution as an argument might reject inequality-creating projects like this, and it would also be likely to accept negative sum projects that reduced inequality.

The potential compensation test criterion is perhaps the most controversial feature of standard welfare economics. On the one hand, it has been criticized as being incompatible with the Pareto criterion since it allows for a ranking of projects that are Pareto noncomparable. On the other hand, many economists argue that lump sum transfers or other means of transferring wealth are a more appropriate way for addressing equity concerns. Thus, one should adopt projects that pass the potential compensation test *and* also take steps to efficiently address distributional concerns. In any case, these concerns have not deterred governments from using it for some kinds of policy choices, and economists from advocating greater use of it in a wider range of environmental and resource policy questions.

Summary

This chapter has provided a derivation and explanation of the compensating and equivalent measures of individual welfare change for changes in prices and quantities for both discrete and continuous goods. The compensating and equivalent measures answer different kinds of policy-relevant questions because they make different implicit assumptions about the relevant status quo. It is interesting to examine some hypothetical examples.

Suppose that the question is whether to locate a landfill in a particular neighborhood. The neighbors are likely to oppose this proposal, and suppose that it is accepted that the neighbors have a right to an undisturbed neighborhood. Then the relevant measure of the harm for locating the landfill in their neighborhood would be the sum of their compensating measures of loss (*CV* and *CS*). The appropriate measure of the gain to those who would use the landfill would be their willingness to pay to locate it in this neighborhood—also a compensating

Table 3.2 Implied property rights and associated welfare measures

Implicit "rights"	*Policy question*	*Gainers*	*Losers*
To the present polluter	Require cleanup?	Neighbors, compensating measure (WTP)	Polluter, compensating measure (WTA)
To the potential polluter	Allow pollution?	Polluter, equivalent measure (WTA)	Neighbors, equivalent measure (WTP)
To the neighbors	Require cleanup?	Neighbors, equivalent measure (WTA)	Polluter, equivalent measure (WTP)
To the neighbors	Allow pollution?	Polluter, compensating measure (WTP)	Neighbors, compensating measure (WTA)

measure. Alternatively, if it is argued that the larger society has a right to locate the landfill anywhere, then what is relevant is the neighbors' willingness to pay to keep it out of their neighborhood. This is an equivalent measure of the potential loss (*EV* and *ES*). For the users of the landfill, the value of locating the landfill in this neighborhood is what its users would require to compensate them for locating it in a less desirable place—an equivalent measure of benefit.

Suppose, instead, that the offending facility is a polluting factory that has been in the neighborhood for a long time. If the neighbors are deemed to have a right to a clean neighborhood, then the appropriate reference point for welfare measurement is their utility levels after the factory has stopped polluting. This implies an equivalent measure of welfare change (*EV* and *ES*). Specifically, this is a measure of the compensation that the neighbors would require to forgo having the pollution stopped, and a measure of the factory owners' willingness to pay to continue to pollute. Alternatively, if the factory has a right to pollute, compensating measures of the gain from stopping the pollution are appropriate (*CV* and *CS*).

In each case, the appropriate welfare measure can be found by examining the nature of the social transaction that is implied by the policy decision at hand, and by the implicit rights to the services of the environment presumed to be held by the various parties to the transaction. The results for the examples discussed here can be summarized in Table 3.2.

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Chapter 4

Welfare Measures

Theoretical Basis for Empirical Assessment

For market goods, welfare effects due to changes in prices have been defined in terms of the area under the appropriate Hicks-compensated demand curve. For nonmarket goods, welfare effects due to changes in quantities have been defined in terms of the area under the marginal willingness-to-pay curve for the good or service. The marginal willingness-to-pay curves exist for public goods and nonmarketed goods such as the services of the environment; but they cannot be estimated from direct observations of transactions in these goods. Given the absence of markets for public goods and environmental goods, how can information on demand and benefits be obtained?

As described in Chapter 2, there are basically two approaches to obtaining demand and value information for changes in the quantities of nonmarket goods. They are the revealed preference methods that involve the estimation of value from observations of behavior in the markets for related goods and the stated preference methods for deriving values from responses to hypothetical questions. This chapter explores some of the possible relationships between demands for private goods and demands for environmental services in an effort to determine under what circumstances the demands for environmental services can be inferred from information on market transactions for a related private good. Let q denote some measure of environmental or resource quality. The task is to estimate in monetary terms the changes in individuals' welfares associated with changes in q.

The basic thesis of this chapter is that the degree to which inferences about the benefits of increases in q can be drawn from market observations and the appropriate techniques to be used in drawing these inferences, both depend upon the way in which q enters individual utility functions. Broadly speaking, there are three ways that q can affect an individual's utility: (a) q can produce utility indirectly as a factor input in the production of a marketed good that yields utility; (b) q can be an input in the household production of utility-yielding commodities; or (c) q can produce utility directly by being an argument in an individual's utility function. In the third case, there are a variety of ways in which q can interact with one or more market goods in the individual's preference structure. For example, there may be a substitution or complementary relationship between q and some private good. If the nature of the household production process or the forms of

interaction between q and private goods can be specified, it may be possible to infer the value of q to the individual from observations of choices of the related market goods.

The next section briefly explores the case where q is a factor of production for a market good. The bulk of the chapter, however, is devoted to examining models of the ways in which q can affect utility more directly. The exploitation of possible relationships between environmental goods and private goods leads to several empirical techniques for estimating environmental and resource values. These techniques have the following characteristics: (a) they are consistent with the basic theory of demand and consumer preferences; (b) they provide a means for estimating the indirect utility function, the expenditure function, or the compensated demand function for the environmental service; and (c) they are practical in the sense of imposing realistic data and computational requirements. The chapter concludes by examining those sources of value that are potentially missed by relying on revealed preference data alone, broadly classified as "nonuse" or "passive use" values.

Environmental Quality as a Factor Input

When q is a factor of production, changes in q lead to changes in production costs, which in turn affect the price and quantity of output or the returns to other factor inputs, or both. The benefits of changes in q can be inferred from these changes in observable market data. There are several examples where q can be interpreted as a factor input. The quality of river water diverted for irrigation affects the agricultural productivity of irrigated land. The quality of intake water may influence the costs of treating domestic water supplies and the costs of production in industrial operations that utilize water for processing purposes. Agricultural productivity is impaired by some forms of air pollution, and to the extent that air pollution causes materials damages, it can affect the costs of production for a wide variety of goods and services.

Assume that good x is produced with a production function,

$$x = x(k, l, q), \tag{4.1}$$

where k and l are capital and labor, respectively, and where the marginal product of q is positive. With given factor prices, and assuming cost-minimizing behavior, there is a cost function:

$$C = C(p_k, p_w, q, x). \tag{4.2}$$

Since q affects the production and supply of a marketed good, the benefits of changes in q can be defined and measured in terms of changes in market variables related to the x industry. A change in q will cause shifts in both cost curves and factor demand curves. The consequences of these shifts depend on conditions in factor and product markets. Changes in q can produce benefits through two channels. The first is through changes in the price of x to consumers. The second

is through changes in the incomes and profits received by owners of factor inputs used in x production.

To illustrate the first channel, assume that x is produced in a competitive industry under conditions of constant cost—that is, factor supplies to this industry are infinitely elastic. Assume that the change in q affects the cost curves of a significant proportion of producers in the market. As a result, the supply curve shifts downward, causing a fall in the price and an increase in total quantity. The benefit of the price reduction accrues to consumers and can be measured by the methods described in Chapter 3.

To illustrate the second channel—changes in the incomes received by factors of production—consider only one producer who is a price taker in all markets. If the change in q affects only this producer, output price will not be changed. Since the change in q affects the marginal costs of production, the firm's marginal cost and supply curves are shifted down. In this case, the benefit is equal to the increase in quasi-rents to the firm. This benefit will accrue to the owner of a fixed factor—land, for example—or to the residual income claimant as profit. In either case, benefits can be measured by changes in profits and fixed factor incomes. However, if the producers affected by changes in q face less than perfectly elastic factor supply curves, at least some of the benefits will be passed on to factors through changes in factor prices and incomes. The factors' shares of benefits can be approximated by the areas to the left of factor supply curves.

The effects of these two channels are combined in Figure 4.1. When the supply curve of the industry is shifted down to S'', the price decreases to p''. The benefit to consumers of x is approximated by the change in consumers' surpluses, the area $p'BCp''$. Part of this benefit, $p'BFp''$, is at the expense of a reduction in producer and factor surpluses, so the net gain from the lower price is BCF. The lower supply curve results in factor surpluses and quasi-rents equal to $p''CE$. The net increase to producers and factors is $AFCE$, so total benefits are equal to $ABCE$.

Implementation of these measures requires knowledge of the effects of changes in q on the cost of production, the supply conditions for output, the demand curve for good x, and factor supplies. There are two special cases where the estimation of benefits is relatively straightforward.

The first is the case where q is a perfect substitute for other inputs in the production of a good. An increase in q leads to a reduction in factor input costs. If the substitution relationship is known, the decrease in per unit production costs is readily calculated. For example, if water quality improvement results in a decrease in chlorination requirements for drinking water supplies, the decrease in chlorination costs per unit of output can be readily calculated. Where the change in total cost does not affect marginal cost and output, the cost saving is a true measure of the benefit of the change in q. If the change in q affects marginal cost, the benefits should include the effect of the lower cost on output and price. However, if the percentage reduction in marginal costs is small or the marginal cost curve is inelastic, or both, the corresponding increase in output would be relatively small. Thus, the decrease in total cost could still be used to provide a

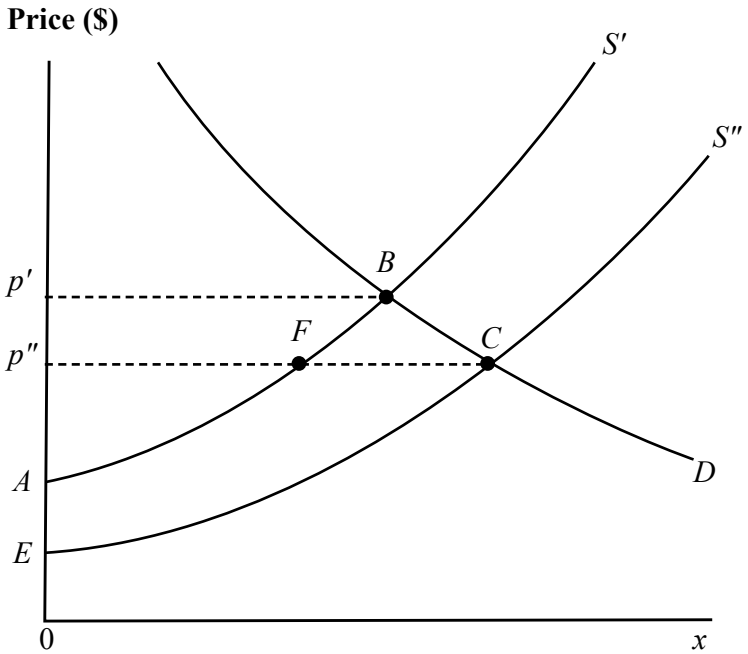

Figure 4.1 The welfare measure when q affects the production of x

rough approximation of true benefits. This approach, sometimes referred to as the "damage function" approach, has been the basis of a number of estimates of the materials, household cleaning, and agricultural crop-loss benefits of air pollution control, and of the benefits to municipalities, industries, and households of reduced contamination of intake water supplies.

The second case that makes the estimation of benefits relatively straightforward is where knowledge of cost, demand, and market structure suggests that the benefits of a change in q will accrue to producers. Then benefits may be estimated from observed or predicted changes in the net income of factor inputs. If the production unit in question is small relative to the market for the final product and for variable factors, it can be assumed that product and variable factor prices will remain fixed after the change in q. The increased productivity then accrues to the fixed factors of production in the form of profit or quasi-rent.

More generally, however, estimates of the value of q require knowledge of the cost and demand functions. In some studies, it has been possible to use econometric methods to estimate a cost function that includes an environmental quality variable (for example, Mjelde et al. 1984; Garcia et al. 1986; Neeliah and Shankar 2010). Other studies have used various simulation approaches to model the behavior of producers and their responses to changes in an environmental variable. Models and techniques for valuing the effects of q on production are discussed in more detail in Chapter 8.

An Individual's Demand for Environmental Quality

In order to analyze those cases where q affects individuals directly, the basic model of individual preference and demand, with environmental quality included as an argument in the utility function, must first be reviewed. The implications of different forms of utility functions for estimation of the demand for q can then be examined.

Consider a single individual who has a utility function

$$u = u(\boldsymbol{X}, q), \tag{4.3}$$

where $\boldsymbol{X} = \left(x_1, \ldots, x_J\right)$ is a vector of private goods quantities. In entering environmental quality as an argument in the utility function, it is assumed here that the individual perceives the effects of changes in environmental quality. For example, if high ozone levels cause respiratory irritation, the individual is assumed to be aware of the irritation, so that he feels "better" when it is reduced. He need not know the cause of the irritation or the actual levels of air pollution. If the individual is not aware of the effects of changes in q, the revealed preferences methods of benefit estimation cannot be applied. For example, individuals may not perceive the effects of long-term exposure to air pollutants on their probability of chronic illness or death. If that is the case, changes in q will not affect their behavior and observations of market behavior will yield no information about the value of reducing risks to health.

Assume that the individual maximizes utility subject to a budget constraint

$$\sum_j p_j \cdot x_j \leq M, \tag{4.4}$$

where M is money income. The individual takes q as given and does not have to pay a price for this freely provided quantity. The solution to this problem yields a set of ordinary demand functions

$$x_j = x_j(\boldsymbol{P}, M, q), \tag{4.5}$$

where $\boldsymbol{P} = \left(p_1, \ldots, p_J\right)$ is the vector of private good prices. Note that in general q could be an argument in all private good demand functions.

The dual to the utility maximization problem can be stated as follows: minimize expenditure ($\sum_{j=1}^{J} p_j x_j$) subject to the constraint that utility equals or exceeds some stated level, say u^0. The solution to this problem gives the expenditure function

$$e(\boldsymbol{P}, q, u^0) = M. \tag{4.6}$$

The expenditure function has a number of useful properties for applied welfare analysis. First, as shown in Chapter 3, the derivative of the expenditure function with respect to any price gives the Hicks-compensated demand function for that good—that is,

$$\frac{\partial e}{\partial p_j} = h_j\left(\boldsymbol{P}, q, u^0\right). \tag{4.7}$$

Similarly, the derivative of equation (4.6) with respect to q (with the appropriate change of sign) gives the Hicks-compensated inverse demand function, or marginal willingness to pay for changes, in q. Let w_q be the marginal willingness to pay or marginal demand price for q, then

$$w_q = -\frac{\partial e\left(\boldsymbol{P}, q, u^0\right)}{\partial q}. \tag{4.8}$$

An alternative expression for the marginal willingness to pay can be obtained by setting the total differential of the indirect utility function equal to zero and solving for the compensating change in income associated with the change in q. Specifically,

$$v = v(\boldsymbol{P}, M, q), \tag{4.9}$$

$$dv = \frac{\partial v}{\partial M} dM + \frac{\partial v}{\partial q} dq = 0, \tag{4.10}$$

and

$$\frac{dM}{dq} = -\frac{\partial v / \partial q}{\partial v / \partial M}, \tag{4.11}$$

where $d\boldsymbol{P}$ is zero by assumption.

If the value of the derivative of the right-hand side of equation (4.8) can be inferred from observed data, then we have a point estimate of the marginal willingness to pay for q. If this derivative can be estimated as a function of q, then we have the marginal willingness to pay function for q. Let W_q represent the benefit to the individual of a nonmarginal increase in the supply of q. W_q is the integral of the marginal willingness-to-pay function, or

$$\begin{aligned} W_q &= -\int_{q'}^{q''} e_q\left(\boldsymbol{P}, q, u\right) dq \\ &= e\left(\boldsymbol{P}, q', u\right) - e\left(\boldsymbol{P}, q'', u\right). \end{aligned} \tag{4.12}$$

This is either a compensating surplus (*CS*) or an equivalent surplus (*ES*) measure of welfare change, depending on the level of utility at which equation (4.12) is evaluated. The question to be discussed in the next section is whether there are any circumstances in which information about equations (4.8) or (4.12) can be derived from observations of market prices and quantities for private goods.

The Structure of Preferences and Measures of Value

The main purpose of this section is to describe the available techniques for revealing these welfare measures, or approximations of them, using observable data on related behavior and individual choice. The strategy will be to explore credible a priori assumptions that support restrictions on the form of the utility function and/or demand functions for market goods or household produced goods that, in turn, aid in revealing the individual's preferences for environmental quality.

Different types of restrictions have different implications for the measurability of the demand for environmental quality. Developing a careful taxonomy of the methods for teasing out the welfare effects of interest helps reveal the broader and more general basis for the welfare economics of environmental valuation.

There are several ways additional structure can be given to the general model of preferences and choice discussed above. Each of these alternatives involves some kind of assumption about the structure of preferences and/or the constraints on individual choice. Each of the assumptions implies some kind of connection between observable demands for market goods and the values of environmental services and public goods; and each assumption provides a basis for inferring the marginal willingness to pay for q from observations of the relationships between q and the demands for market goods. It is also important to examine what is required to obtain values of *nonmarginal* changes in q, since these are what are required for most real-world policy questions. For each of the alternatives, the specific restrictions are identified, along with a discussion as to how inferences about value can be drawn from observations of individual choices.

The relationships between q and other goods that have been found to be of use involve, broadly speaking, either substitution or complementarity relationships between q and other goods. Exactly how these relationships work out methodologically, however, depends on a number of other considerations. In what follows, consideration is first given to those cases in which the environmental good is a substitute for a marketed good that enters the utility function. A fundamentally equivalent construct is one in which the environmental good is an input into a household production function and has marketed-good substitutes in the production process. The latter is perhaps the more general and more useful way of conceptualizing the problem.

Attention is then given to a second category of models, namely those in which the environmental good is in some way complementary to another good. The complementarity is often most usefully conceived such that the environmental good is a quality characteristic of the related good. There are two often-used derivatives of this construct. In one, the related good is itself a nonmarket good produced by the household using a household production process. The second is one in which the related good is marketed, but units of the good are heterogeneous and quality-differentiated. Because the good is marketed, the prices of units with higher levels of quality embodied in them are bid up.

Several of the alternatives involve making some kind of assumption about the separability of the utility function. Thus, it will be useful to first review the concept of separability and the implications of various forms of separability for observable demands for market goods. For a review of concepts of separability, see Goldman and Uzawa (1964), Katzner (1970), Blackorby, Primont, and Russell (1978), or Deaton and Muellbauer (1980).

Separability refers to the possible effect of partitioning the goods entering into the utility function into subsets, and the relationships among these subsets. Suppose that there are three types of market goods, so that

$$u = u(\boldsymbol{X}, \boldsymbol{Y}, \boldsymbol{Z}), \tag{4.13}$$

where $\boldsymbol{X} = (x_1, \ldots, x_J)$, $\boldsymbol{Y} = (y_1, \ldots, y_K)$, and $\boldsymbol{Z} = (z_1, \ldots, z_S)$. A utility function is (*directly*) *weakly separable* if the marginal rate of substitution (MRS) between any pair of goods within the same subset is independent of the quantities of goods in any other subset. The following utility function is weakly separable:

$$u = u[u^1(\boldsymbol{X}), u^2(\boldsymbol{Y}), u^3(\boldsymbol{Z})]. \tag{4.14}$$

This means that the demand functions for goods in one subset can be written as depending only on the prices of goods in that subset and the expenditure share of that subset.

A utility function is strongly *separable* if the MRS between two goods in different subsets is independent of the quantity of any good in any other subset. Specifically, with strong separability the MRS_{x_j, y_k} is independent of z_s. Any additive utility function is strongly separable; and any strongly separable utility function is additive in some monotone transform (Deaton and Muellbauer 1980). The following utility function is strongly separable:

$$u = u[u^1(\boldsymbol{X}) + u^2(\boldsymbol{Y}) + u^3(\boldsymbol{Z})]. \tag{4.15}$$

If the utility function is not separable, there are no restrictions on the terms that are arguments in marginal rates of substitution, or in the demand functions for individual goods. The MRS between any pair of goods depends on the quantities of all goods; and the demand function for any good depends upon the prices of all goods as well as income.

In the rest of this section, three major forms of relationships between q and market goods are examined. These major forms are: (a) some form of substitution relationship between q and one or more market goods; (b) some form of complementary relationship; and (c) the case of differentiated goods where the amount of q embodied in, or attached to, a market good is one of its differentiating characteristics. Since some of these relationships can be modeled either implicitly or explicitly as involving some form of household production, a brief overview of the household production framework for modeling individual preferences and choice is first provided.

The Household Production Framework

The household production function model provides a framework for examining interactions between demands for market goods and the availability of a public good such as environmental quality. This framework is based on the assumption that there is a set of technical relationships among goods used by households in the implicit production of utility-yielding final services. In the household production function literature, the utility-yielding final services are often termed "commodities," while market goods are simply "goods." The terminology used in this chapter is more descriptive of the relationship between

goods being bought in the market and the final service flows that yield utility to individuals.

In the household production framework, utility is a function of the level of final service flows:

$$u = u(\boldsymbol{Z}). \tag{4.16}$$

$\boldsymbol{Z} = (z_1, \ldots, z_S)$ is produced according to a technology common to all households and assumed to be known; and

$$z_s = z_s(\boldsymbol{X}, \boldsymbol{Q}), \text{ for } s = 1, \ldots, S, \tag{4.17}$$

where $\boldsymbol{X} = (x_1, \ldots, x_J)$ represents a vector of market goods available at prices $\boldsymbol{P}_x = (p_{x1}, \ldots, p_{xJ})$, and $\boldsymbol{Q}$ represents a vector of environmental quality attributes. The cost of household production depends on the technology, input prices, and exogenously determined $\boldsymbol{Q}$. That is, $C(\boldsymbol{Z}) = C(\boldsymbol{Z}, \boldsymbol{P}_x, \boldsymbol{Q})$.

Formally, the individual choice problem is to maximize equation (4.16) subject to the constraints provided by equation (4.17), the given level of $\boldsymbol{Q}$, and the budget constraint $\sum_{j=1}^{J} p_{xj} x_j \leq M$. This problem can be solved by a two-step procedure in which the first step is to combine market goods and $\boldsymbol{Q}$ so as to minimize the costs of producing the z_s. This determines the marginal costs of final services $\boldsymbol{C}_z$. These are essentially implicit prices, but unlike market prices, they are not parametric to the individual unless marginal costs are constant. The second step is to maximize (4.16) subject to the budget constraint $C(\boldsymbol{Z}) \leq M$. The observable manifestation of the solution to this problem is a set of derived market goods demands:

$$\boldsymbol{X} = \boldsymbol{X}(\boldsymbol{P}_x, M, \boldsymbol{Q}). \tag{4.18}$$

These demand functions will reflect both the role of $\boldsymbol{Q}$ in the household production technology equation (4.17) and the preferences over the final service flows. As will be seen, some of the existing models for interpreting demands for market goods (for example, the averting behavior/defensive expenditures model and the weak complementarity model) can be interpreted as simpler versions of a household production function model. In fact, as Smith (1991) has argued, the household production framework may be a useful way of thinking about whether a specific market good is likely to be a substitute or a complement for environmental quality. He used the household production framework as an organizing principle to examine a number of models for nonmarket valuation and to interpret several empirical applications of these models. As various forms of complementarity and substitutability relationships are considered, it will be shown how they can also be interpreted as applications of the household production function model.

Substitutes

Mäler (1974, 116–118) has shown that the marginal willingness to pay for q can be expressed in terms of the price of any private good and the marginal utilities of that good and q. The expression is

$$w_q = -\frac{\partial e(\boldsymbol{P}, q, u^0)}{\partial q} = -p_i \left(\frac{\partial u / \partial q}{\partial u / \partial x_i} \right) = p_i \cdot MRS_{qx_i}. \tag{4.19}$$

This would be a useful practical result if it were possible to derive simple expressions for the marginal rates of substitution. Of course, if q could be purchased at a given price the marginal rate of substitution between q and x_i could be inferred from the price ratio. However, when the level of q is determined exogenously, the marginal rate of substitution can only be determined through knowledge of the utility function or household production function.

Consider the case where some assumption about the separability of the utility function is used to isolate q and a good, z, from the other determinants of utility (see Mäler 1974, 178–183). Suppose the utility function is weakly separable and is of the following form:

$$u = u\left\{ u^1(\boldsymbol{X}), u^2(\boldsymbol{Y}), \left[c \cdot z^{-\alpha} + (1-c) q^{-\alpha} \right]^{\frac{-1}{\alpha}} \right\}. \tag{4.20}$$

Given the separability assumption, the marginal rate of substitution between z and q is independent of the quantities of $\boldsymbol{X}$ and $\boldsymbol{Y}$. In this case, the marginal rate of substitution is

$$MRS_{zq} = \frac{c}{1-c} \left(\frac{q}{z} \right)^{\alpha - 1} = \frac{c}{1-c} \left(\frac{q}{z} \right)^{1/\sigma}, \tag{4.21}$$

where σ is the elasticity of substitution, which is constant. From (4.19) we have

$$w_q = -p_z \cdot \left(\frac{c}{1-c} \right) \left(\frac{q}{z} \right)^{1/\sigma}. \tag{4.22}$$

Since, in general, the marginal rate of substitution depends on the ratio q/z, we need to know both the elasticity of substitution, σ, and c in order to compute w_q. This requires knowledge of the utility function.

Perfect Substitutes

There is one special case where the expression for w_q reduces to a usable term. If z and q are perfect substitutes in consumption, the elasticity of substitution between them is infinite, and the expression for the demand price for q reduces to $p_z \cdot s$, where s is the substitution ratio between z and q [$s = c/(1 - c)$]. If perfect substitutability can be assumed, s (or c) should be computable from known or observable technical consumption data or from the household production function.

The perfect substitutability assumption lies behind the simplest application of the defensive expenditures technique for estimating the benefits of pollution

control. Defensive expenditures are made either to prevent or to counteract the adverse effect of pollution. They are also referred to as averting expenditures (for example see Courant and Porter 1981). In effect, a defensive expenditure is spending on a good that is a substitute for higher q. An increase in q is assumed to lead to a decrease in spending on the substitute.

As equation (4.19) shows, the marginal change in the spending on z is the correct measure of the marginal willingness to pay for the change in q; and for nonmarginal changes in q, the benefit is $p_z \cdot \Delta q$. However, this will not necessarily be the same as the observed change in spending on q. The intuition behind this statement is straightforward. The benefit of a nonmarginal change in q is the reduction in the spending on z that is required to keep the individual on the original indifference curve. In general, the individual will not actually reduce spending on z by this amount. There is an income effect on z as well as a substitution effect. The increase in q means that the same level of utility can be maintained with a smaller expenditure on z, and consequentially, the individual will reallocate expenditure among all goods, including z, so as to maximize the increase in total utility. This will result in increases in the expenditures on all goods with positive income elasticities of demand. Hence, the observed decrease in spending on z will be less than necessary to hold utility constant, and the reduction in defensive spending will be an underestimate of the benefits of higher q. In fact, as we will see below, actual spending on z could increase.

A General Model of Substitution

Since perfect substitutes represent a special case, it will be useful to explore a more general model in which q and a market good are less than perfect substitutes, and in which the substitution relationship arises because q and the market good contribute to utility through the same mechanism. Following the analysis of Courant and Porter (1981), this case is modeled in the household production framework. Early analyses dealing with this type of model are presented in Shibata and Winrich (1983), Harford (1984), and Harrington and Portney (1987) and more recent examples include Neidell (2009) and Zivin, Neidell, and Schlenker (2011). Suppose that clean air and soap are substitutes in the production of cleanliness. Let

$$u = u(\boldsymbol{X}, z), \tag{4.23}$$

where z is cleanliness, and $\boldsymbol{X}$ is a vector of market goods with prices $\boldsymbol{P}_x = \left(p_{x1}, \ldots, p_{xJ}\right)$ Suppose that z is produced by households by combining a market good, y, and air quality according to the production function

$$z = z(y, q), \tag{4.24}$$

with positive partial derivatives for both arguments. Further, assume that y contributes to utility only through its contribution to cleanliness. This is equivalent to assuming that the utility function is weakly separable, with y and q in one group.

For any given q, the individual chooses $\boldsymbol{X}$ and y so as to maximize

$$u = u[\boldsymbol{X}, z(y, q)], \tag{4.25}$$

subject to the budget constraint

$$M - \sum_{j=1}^{J} p_{xj} x_j - p_y y \geq 0, \tag{4.26}$$

the household production technology and the exogenous q. The first-order conditions are

$$\begin{aligned} &\frac{\partial u}{\partial \boldsymbol{X}} - \lambda \boldsymbol{P}_x = 0 \\ &\frac{\partial u}{\partial z} \cdot \frac{\partial z}{\partial y} - \lambda \cdot p_y = 0. \end{aligned} \tag{4.27}$$

Substituting the expressions for the demands for $\boldsymbol{X}$ and y into the utility function yields the indirect utility function

$$v = v(\boldsymbol{P}_x, p_y, M, q). \tag{4.28}$$

Setting the total differential of this expression equal to zero, assuming prices do not change, and rearranging terms results in the following expression for the marginal value of increasing q:

$$\mathrm{w_q} = -\frac{dM}{dq} = \frac{(\partial v/\partial q)}{\lambda} = \frac{\left(\frac{\partial u}{\partial z} \cdot \frac{\partial z}{\partial q}\right)}{\lambda}, \tag{4.29}$$

where λ is the marginal utility of income, and the term in brackets is an expansion of $\partial v/\partial q$ making use of the chain rule. Substituting the first-order condition (4.27) gives the marginal willingness to pay for q as a function of the price of the private good and the marginal productivities of y and q:

$$w_q = p_y \cdot \frac{\partial z / \partial q}{\partial z / \partial y}. \tag{4.30}$$

This is the reduction in spending on y, holding z constant—that is, moving along a given isoquant in the household production function. This marginal value can be calculated if the household production function is known.

If y and q jointly produce two goods (or help to avoid two bads), this result does not hold. Suppose $u = u\left[\boldsymbol{X}, z_1(y,q), z_2(y,q)\right]$. Using the procedure described above, we obtain

$$w_q = p_y \left[\frac{(\partial u / \partial z_1)\cdot(\partial z_1 / \partial q) + (\partial u / \partial z_2)\cdot(\partial z_2 / \partial q)}{(\partial u / \partial z_1)\cdot(\partial z_1 / \partial y) + (\partial u / \partial z_2)\cdot(\partial z_2 / \partial y)}\right]. \tag{4.31}$$

The unobservable utility terms do not cancel out. The relationship between the actual change in spending on y and marginal willingness to pay can be obtained

by totally differentiating the household production function with respect to q and using the first-order conditions for the optimum choice of y to obtain

$$\frac{dz}{dq} = \frac{\partial z}{\partial y} \cdot \frac{\partial y^*}{\partial q} + \frac{\partial z}{\partial q}, \tag{4.32}$$

where $y^*\left(\boldsymbol{P}_x, p_y, M, q\right)$ is the derived demand function. Rearranging and employing the second line of equation (4.27) yields

$$p_y \cdot \frac{\partial z / \partial q}{\partial z / \partial y} = w_q = \frac{\partial u / \partial z}{\lambda} \cdot \frac{dz}{dq} - p_y \cdot \frac{\partial y^*}{\partial q}. \tag{4.33}$$

If spending on y decreases ($\partial y^* / \partial q < 0$), then the marginal willingness to pay for q is the sum of the observed reduction in spending and the willingness to pay for the induced increase in the final service flow z. The actual savings in spending on y is an underestimate of the marginal value in this case; and if the lower implicit cost of household production of z leads to a sufficiently large increase in the consumption of z, then it is possible that spending on y could increase. Thus, observed changes in spending on substitute goods are not reliable indicators of marginal willingness to pay for changes in q.

Similar results are obtained for the case where z requires several private goods as well as q for its production. The marginal value of a change in q can be measured by the reduction in spending on any of the market goods for movements along the original z isoquant. Observed changes in spending on market goods are not reliable measures of w_q.

An alternative formulation is to let z represent a bad, with y and q mitigating the adverse effects of z. In other words, $u_z < 0$, $z_q < 0$, and $z_y < 0$, where subscripts indicate partial derivatives. Similar conclusions can be reached about willingness to pay and its relationship to changes in mitigating expenditures. This alternative model is typically referred to as an *averting behavior model* or a *mitigating behavior model*.

Bartik (1988) has shown how lower and upper bounds on the welfare gains of nonmarginal changes in q can be calculated from knowledge of only the household production function. The lower bound on the *CS* measure of benefits of an increase in q is the reduction in the expenditures on y necessary to reach the initial level of z, other things being equal. More formally, the lower bound is given by

$$\text{Lower bound} = p_y\left(y' - y''\right). \tag{4.34}$$

This can be shown to be a lower bound on *CS* by defining a restricted expenditure function, $e^*(q, z', u^0)$, where z' is the initial level of z, and price terms have been omitted for simplicity. The welfare gain associated with an increase in q from q' to q'' holding z at z', is

$$CS^* = e^*\left(q', z', u^0\right) - e^*\left(q'', z', u^0\right). \tag{4.35}$$

The first term on the right-hand side is $M = \sum_{j=1}^{J} p_{xj} x_j' + p_y y'$. The increase in q means that less must be spent on y to produce z', and since $u(\boldsymbol{X}', z') = u^0$, the second term is $\sum_{j=1}^{J} p_{xj} x_j' + p_y y''$. So

$$CS^* = \sum_{j=1}^{J} p_{xj} x_j' + p_y y' - \sum_{j=1}^{J} p_{xj} x_j' - p_y y'' = p_y \left(y' - y'' \right). \tag{4.36}$$

This has to be less than the true CS, since relaxing the constraint holding z at z' would allow the individual to increase utility, unless the marginal utility of z were already zero. In a similar fashion, Bartik showed that an upper bound on the equivalent surplus measure of welfare gain is the decrease in spending on y that is possible while holding z constant at its new equilibrium level; and since, in general, $ES > CS$, the constrained changes in expenditure bound both of the true measures of welfare change.

We can also consider the case where q contributes to utility directly in addition to its contribution to z. Taking the total differential of the indirect utility function, solving for the compensating change in income, and substituting in the first-order condition for the choice of y gives us

$$w_q = -\frac{dM}{dq} = \frac{\partial u / \partial q}{\lambda} + \left[p_y \cdot \frac{\partial z / \partial q}{\partial z / \partial y} \right]. \tag{4.37}$$

Comparing this expression with equation (4.29), we see that the marginal willingness to pay for q now includes an unobservable marginal utility term for the direct effect of q on utility. Similarly, following the same procedures used to derive equation (4.30), we have

$$w_q = \left[\frac{\partial u / \partial z}{\lambda} \cdot \frac{dz}{dq} \right] + \frac{\partial u / \partial q}{\lambda} - \left(p_y \cdot \frac{\partial y^*}{\partial q} \right). \tag{4.38}$$

Again, if spending on y decreases, it is an underestimate of marginal willingness to pay because it neglects both the utility value of the induced increase in z and the direct utility value of the increase in q.

Perfect Complements

Suppose that the single environmental service q is a perfect complement to a market good, say x_1, where perfect complementarity means that x_1 and q must be consumed in fixed proportions, for example $x_1/q = a$. This means that to utilize effectively the services of one unit of q, the individual must purchase $1/a$ units of x_1. In the context of the household production model, this could imply a production function for z_1 of the following form:

$$z_1 = \min\left(\frac{x_1}{a}, q \right). \tag{4.39}$$

If this is the case, there are conditions under which the benefits of changes in q can be estimated from knowledge of the demand function for the complementary market good. As long as p_1 is less than some critical value, say p_1^*, the quantity demanded of x_1 will be independent of its price and determined solely by the availability of q. If q is increased by one unit, the individual will purchase $1/a$ additional units of x_1 and experience a higher level of utility. Thus, the marginal willingness to pay for q will be positive. However, if p_1 is greater than p_1^*, the individual will purchase fewer units of x_1 than are required to fully utilize the available q. Thus, at p_1 greater than p_1^*, the marginal utility and marginal willingness to pay for q are zero.

Mäler (1974, 180–183) has shown that if the demand functions for x_1 and for other market goods are known, it is possible to compute the expenditure function and the demand price for q when p_1 is less than p_1^*. The exact expression for the demand price for q depends upon the specification of the true demand curves. There is no simple generalization of the technique. Also, as Mäler appeared to realize, it is difficult to imagine examples of perfect complementarity between an environmental service and a market good. But as we will now see, there is a less restrictive form of complementarity that also makes possible the calculation of the marginal willingness to pay for q and that appears to have real-world applications.

Weak Complementarity

Suppose that q enhances the enjoyment the individual derives from consuming x_1 and that an increase in q increases the quantity demanded of x_1, other things being equal. One reasonable interpretation of these assumptions is that q is an exogenously determined characteristic of x_1. Examples of environmental services that might fit these assumptions include water quality as a characteristic associated with visits to a lake, and the number of fish caught as a characteristic of fishing trips to a stream. The model can also be applied to estimate the value of a network where the purchase of a market good, a cell phone for example, is required to connect to the network (Hahn, Tetlock, and Burnett 2000).

In this section, it is shown that when the "enjoyment" of q requires the purchase of a market good, or when q can be treated as a characteristic of a market good, it is possible to identify a measure of the value of a change in q that is based on the demand for the market good. The measure is described, along with the conditions under which it is a valid welfare measure. The section closes with a discussion of some issues concerning implementation of the measure.

Suppose that x_1 and q are Hicksian complements—that is, the compensated demand function

$$h_1 = h_1(\boldsymbol{P}, q, u^0) \tag{4.40}$$

is characterized by $\partial h_1/\partial q > 0$. This implies that the Marshallian demand for x_1,

$$x_1 = x_1(\boldsymbol{P}, M, q), \tag{4.41}$$

has $\partial x_1/\partial q > 0$.

Suppose that the complete system of demand equations for this individual is known (or has been estimated econometrically), and that the system satisfies the Slutsky conditions for integrability. In order to derive a valid welfare measure from this system of demand functions, one must be able to integrate this system to solve for the underlying utility function and expenditure function. However, Mäler (1974, 183–189) has shown that, in general, it is not possible to solve completely for the utility and expenditure functions with the information given. Mathematically, the result of the integration contains unknown terms that are themselves functions of q and the constants of integration. It is necessary to impose additional conditions on the problem in order to solve for the unknown terms and determine the constants of integration. The additional conditions involve what Mäler called weak complementarity.

The Conditions

Weak complementarity requires that the marginal utility or marginal demand price of q be zero when the quantity demanded of the complementary private good x_1 is zero. Mathematically, weak complementarity involves two conditions. The first is that x_1 be nonessential in the sense that the compensating variation associated with the elimination of the good is finite—that is

$$\int_0^\infty h_1\left[p_1, q, u^0\right]dp_1 < \infty ,$$

where the terms for all other prices have been omitted for simplicity.

A sufficient condition for this to be the case is that there must exist a finite price $p_1^*(q)$ such that the compensated demand for the good is zero. In other words,

$$h_1\left[p_1^*(q), q, u^0\right] = 0. \tag{4.42}$$

This price is often called the *choke* price. The choke price will usually be an increasing function of q. On a graph, the compensated demand curve has a vertical intercept and there is some level of expenditure on other goods that will sustain u^0 even when x_1 is zero. For ease of exposition, the sufficient condition (a finite choke price) is used in the following discussion rather than the more general, but somewhat tedious, necessary condition.

The second condition is that for values of p_1 at or above $p_1^*(q)$, the derivative of the expenditure function

$$e = e\left(p_1, q, u^0\right) \tag{4.43}$$

must be zero. In other words,

$$\frac{\partial e\left(p_1, q, u^0\right)}{\partial q} = 0 \ \forall \ p_1 \geq p_1^*(q). \tag{4.44}$$

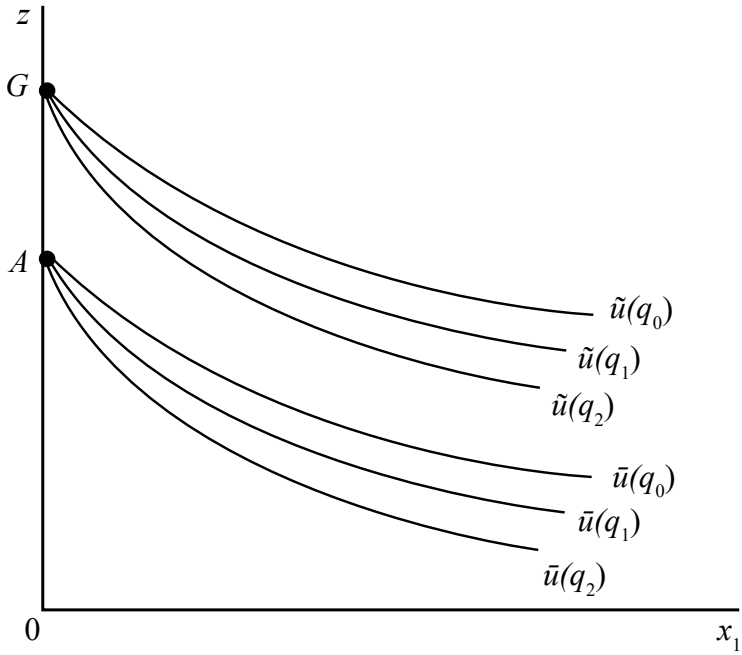

Figure 4.2 Fanned indifference curves

This condition means that at or above the choke price, the marginal utility or marginal willingness to pay for q is zero. Changes in q have no welfare significance unless the price of x_1 is low enough that its compensated demand would be positive. Weak complementarity can also be defined in terms of the underlying utility function, requiring that

$$\left.\frac{\partial u(\boldsymbol{X},q)}{\partial q}\right|_{x_1=0} = 0 \,. \tag{4.45}$$

Weak complementarity establishes an initial position for the individual that can be used to determine the constants of integration. For proof and a demonstration with a numerical example, see Mäler (1974, 183–189). Bradford and Hildebrandt (1977) also obtained results similar to Mäler. Smith and Banzhaf (2004) provided a graphical exposition of the weak complementarity assumption, replicated here as Figure 4.2. The graph provides an indifference map illustrating the tradeoffs between a market commodity (x_1) and a numeraire good (z) for varying levels of a public good (q). The lower group of indifference curves, $\bar{u}(q)$, depict the same level of utility given different levels of the public good, with $q_0 < q_1 < q_2$. So long as x_1 is consumed, the same level of utility can be reached with less of x_1 when $q = q_2$ than when $q = q_0$. Intuitively, the higher level of q enhances (i.e., complements) the experience associated with consuming x_1, so that, other things

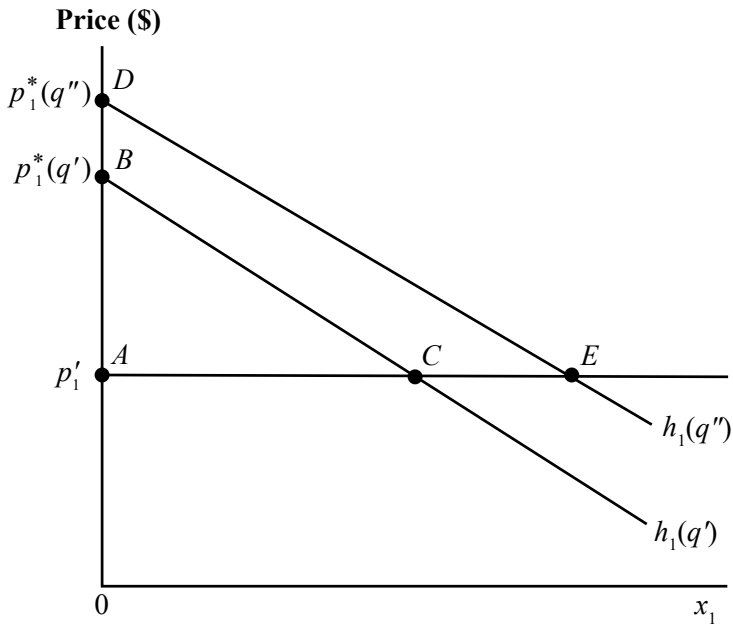

Figure 4.3 The welfare measure for an increase in q when q and x_1 are weak complements

equal, less of x_1 is needed to reach the same level of utility. However, when x_1 is not consumed, the level of q is irrelevant, causing all of the indifference curves $\bar{u}(q)$ to meet at point A. This creates what Smith and Banzhaf referred to as a "fanning" effect on the indifference curves. The second set of indifference curves, $\tilde{u}(q)$, represent a higher level of utility, but retain the fanning pattern in terms of the indifference relationship.

The Welfare Measure

Given the conditions for weak complementarity, the compensating surplus for a change in q can be measured by the area between the two compensated demand curves for x_1. The intuition for this measure can be illustrated graphically. Assume that the compensated demand curve for x_1 has been found for environmental quality level q'. In Figure 4.3, this demand curve is labeled $h_1(q')$. Assume that the price of x_1 is given at p_1' and does not change throughout the analysis. The compensating surplus associated with the use of x_1 is the area ABC under the demand curve. Now assume that quality is improved to q''. The increase in the quality associated with the use of x_1 is assumed to increase the demand for x_1, thus shifting the demand curve outward to $h_1(q'')$. The calculation of the benefit associated with this change is straightforward and can be divided into three steps.

1 Given the initial demand curve $h_1(q')$, postulate a hypothetical increase in price from p_1' to $p_1^*(q')$, the choke price. In order to leave the individual no worse off, that person must be compensated by the area *ABC*.
2 Now postulate the improvement in quality and the shift in the demand curve to $h_1(q'')$. Given the weak complementarity assumption, utility is unaffected since the consumption of x_1 is zero. Therefore, there is no need for compensation, either positive or negative.
3 Now, postulate a return to the old price of p_1'. The individual is made better off by the area *ADE*. In order to restore the individual to the original welfare position, he/she must be taxed by this amount. The net effect of these changes is a gain to the individual (in the absence of the hypothetical compensating payments) of the area *BCED* (= *ADE* – *ABC*). This is the benefit of the change in q.

The role of the two conditions defining weak complementarity can now be made clear. If there were no choke price, there would be no finite compensating variations (*CV*s) in steps 1 and 3 above. In addition, if the derivative of the expenditure function were not zero at the choke price, there would be a welfare change (presumably positive) in step 2 associated with the increase in q, even though the quantity demanded of x_1 were zero. In this case, the area *BCED* would be an underestimate of the benefits of increasing q. If the consumption of x_1 is interpreted as use of an environmental resource (for example, trips to a recreation site), then whatever welfare change is associated with step 2 must be a nonuse value. Nonuse values are discussed in the last section of this chapter.

This result can also be established more rigorously. Recall from Chapter 3 that the compensating welfare measure for a change in q is

$$W_q = e\left(p_1', q', u^0\right) - e\left(p_1', q'', u^0\right). \tag{4.46}$$

The area between the two compensated demand curves is $\Delta = CV_{q''} - CV_{q'}$ where these terms are the *CV*s associated with the ability to purchase x_1 at different levels of q. Specifically,

$$CV_{q'} = e\left[p_1^*\left(q'\right), q', u^0\right] - e\left[p_1', q', u^0\right] \tag{4.47}$$

and

$$CV_{q''} = e\left[p_1^*\left(q''\right), q'', u^0\right] - e\left[p_1', q'', u^0\right]. \tag{4.48}$$

Using these expressions to calculate the area between the compensated demand curves gives

$$\begin{aligned}\Delta = {} & e\left[p_1^*\left(q''\right), q'', u^0\right] - e\left[p_1(q'), q'', u^0\right] \\ & - e\left[p_1^*\left(q'\right), q', u^0\right] + e\left[p_1(q'), q', u^0\right].\end{aligned} \tag{4.49}$$

If the second condition for weak complementarity is satisfied, the first and third terms in (4.49) sum to zero, and we have

$$\Delta = e\left[p_1, q', u^0\right] - e\left[p_1, q'', u^0\right] = W_q. \tag{4.50}$$

Weak Complementarity with More Than One Good

So far, it has been assumed that q is complementary to only one market good. It is possible for q to be complementary to several goods at the same time. For example, if q is water quality in a lake, it could be complementary to several market activities such as fishing, boating, and swimming. As shown in research by Bockstael and Kling (1988), the analysis of weak complementarity can be carried over to the multigood case in a straightforward manner.

Suppose there are two market goods that are complementary to q, with Hicksian demand functions as follows:

$$h_1 = h_1\left(p_1, p_2, q, u^0\right) \tag{4.51}$$

and

$$h_2 = h_2\left(p_1, p_2, q, u^0\right). \tag{4.52}$$

Using the expenditure function, the CS for a change in q is

$$CS = e\left(p_1, p_2, q', u^0\right) - e\left(p_2, p_1, q'', u^0\right). \tag{4.53}$$

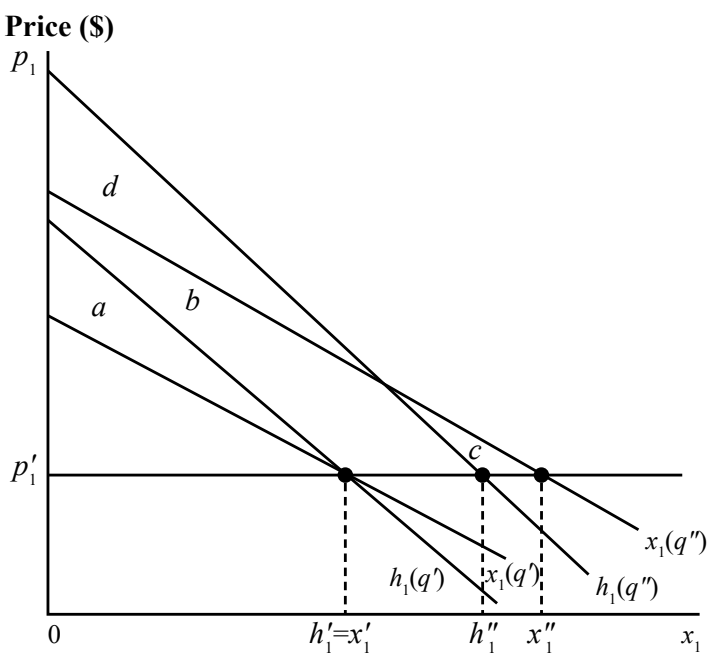

Figure 4.4 Approximating benefits using Marshallian demand curves with weak complementarity

If the conditions for weak complementarity are satisfied, *CS* can be estimated from areas between the compensated demand curves for the two market goods in the following manner. First, calculate the area between the compensated demands for x_1 holding p_2 at the observed price level. Then calculate the area between the compensated demands for x_2 evaluated at the choke price $p_1^*(q'')$ for x_1. Mathematically, this involves calculating the following expression:

$$\begin{aligned}\Delta = &\int_{p_1'}^{p_1^*(q'')} h_1\left[p_1,\ p_2',\ q'',\ u^0\right]dp_1 - \int_{p_1'}^{p_1^*(q')} h_1\left[p_1,\ p_2',\ q',\ u^0\right]dp_1 \\ &+\int_{p_2'}^{p_2^*(q'')} h_2\left[p_1^*(q''),\ p_2,\ q'',\ u^0\right]dp_2 \\ &-\int_{p_2'}^{p_2^*(q')} h_2\left[p_1^*(q'),\ p_2,\ q',\ u^0\right]dp_2.\end{aligned} \tag{4.54}$$

The first and third integrals give the value to the individual of consuming x_1 and x_2 when q is equal to q'' evaluated over a consistent path of integration. In other words, starting from (p_1', p_2'), p_1 is increased to $p_1^*(q'')$ and then p_2 is increased to $p_2^*(q'')$. Similarly, the second and fourth integrals give the value to the individual of consuming x_1 and x_2 when q is equal to q'. The result of this calculation is independent of the order in which the prices are changed.

When each of the integrals in equation (4.54) is expressed as a difference in expenditures, we have the following:

$$\begin{aligned}\Delta = &e\left[p_1^*(q''),\ p_2',\ q'',\ u^0\right] - e\left[p_1',\ p_2',\ q'',\ u^0\right] - e\left[p_1^*(q'),\ p_2',\ q',\ u^0\right] \\ &+e\left[p_1',\ p_2',\ q',\ u^0\right] + e\left[p_1^*(q''),\ p_2^*(q''), q'',\ u^0\right] \\ &-e\left[p_1^*(q''),\ p_2',\ q'',\ u^0\right] - e\left[p_1^*(q'),\ p_2^*(q'),\ q',\ u^0\right] \\ &+e\left[p_1^*(q'),\ p_2',\ q',\ u^0\right].\end{aligned} \tag{4.55}$$

When like terms are canceled out, and if weak complementarity is satisfied, this reduces to $\Delta = CS$ as defined in equation (4.53).

Welfare Measures with Ordinary Demand Functions

The primary empirical requirement for utilizing weak complementarity is to have an estimate of the demand function for the private good as a function of prices, income, and q. Since the compensated demand curves are not directly observable, it is useful to consider whether Willig's (1976) bounds on consumer surplus as an approximation to *CV* and *EV* apply in this situation. Unfortunately, since these bounds were derived for differences in areas under ordinary and compensated demand curves, they do not directly apply to areas between shifting ordinary and compensated demand curves.

This can best be shown graphically, as in Figure 4.4 (Bockstael, McConnell, and Strand 1991). The compensated demand curves for the two levels of q are denoted as $h_1(q')$ and $h_1(q'')$. The ordinary demand curves are denoted by $x_1(q')$ and $x_1(q'')$, and the compensating surplus for a change in q is measured by the area $b + d$. Observe that at the market price of p_1' the increase in q causes the ordinary demand curve to shift out farther to the right than the compensated demand curve. This is because with the ordinary demand curve, there is no compensating reduction in income to hold utility constant. Taking areas between the ordinary demand curves would yield a consumer surplus measure of $a + b + c$. The percentage error arising when the consumer surplus measure is used to approximate the compensating surplus is

$$\%\text{error} = \frac{a+c-d}{b+d}\,. \tag{4.56}$$

As can be seen by inspection, this error could be positive, negative, or by coincidence, equal to zero. Thus, the practice of using ordinary demand curves to estimate welfare changes with weak complementarity can lead to errors of unknown sign and magnitude.

It is possible to use the exact welfare measurement methods described in Chapter 3 to recover weakly complementary preferences from market demands—see for example Larson (1991) and Bockstael and McConnell (1993). However, to do so on the basis of market demand relationships *alone* requires invoking additional assumptions regarding preferences. The fundamental problem is that, in integrating back from Marshallian demand functions to the underlying indirect utility function, one needs to know how the constant of integration changes with changes in quality. The Willig (1978) conditions provide a means for filling in this missing piece of information. As Palmquist (2005) noted, the Willig conditions can be expressed in several ways. In the case of a single market good (x) and a single public good (q), with a corresponding indirect utility function $V(p, M, q)$, one form of the Willig conditions is that

$$\frac{\partial\left(\dfrac{V_q}{V_p}\right)}{\partial M} = 0\,. \tag{4.57}$$

Perhaps the more intuitive version of the Willig condition notes that it requires that the marginal willingness to pay for the public good per unit of the weakly complementary private good be independent of income. In other words,

$$\frac{\partial}{\partial M}\left[\frac{\left(V_q/V_M\right)}{x(p, M, q)}\right] = 0\,. \tag{4.58}$$

Smith and Banzhaf (2004) also demonstrated that the Willig condition restricts how the indifference fans in Figure 4.2 spread with changes in utility level. Given *both* weak complementarity and the Willig condition, the Marshallian surplus

measure approximates the corresponding Hicksian welfare measures in a fashion analogous to the Willig conditions for a price change.

Weak Complementarity and Household Production

As Bockstael and McConnell (1983) have shown, if the individual choice problem is modeled in the household production function framework, there are conditions on the production technology and preferences that yield results equivalent to the weak complementarity model. They showed that if these conditions are satisfied, the welfare value of a change in q can be calculated from knowledge of the market good's compensated demand function even without knowledge of the household production technology. The demand function for the market good is a derived demand. If the conditions established by Bockstael and McConnell are satisfied, the welfare value of the change in q can be calculated by the area between the two compensated demand curves for the market good at the two levels of q.

The individual's choice problem is to maximize

$$u(\boldsymbol{Z}, q) \tag{4.59}$$

subject to

$$\boldsymbol{Z} = \boldsymbol{Z}(\boldsymbol{X}, q) \tag{4.60}$$

and

$$\sum_{j=1}^{J} p_j x_j = M \,. \tag{4.61}$$

The expenditure function is

$$e(\boldsymbol{P}, q, u) = \min\left\{ \sum_{j=1}^{J} p_j x_j \,\middle|\, u^0 = u\left[\boldsymbol{Z}(\boldsymbol{X}, q), q\right] \right\}. \tag{4.62}$$

Suppose that households produce z_1 by combining q and x_1, and that q is used *only* in the household production of z_1. As in the standard model, the compensated demand for x_1 is the derivative of the expenditure function with respect to p_1. In this case, it is a derived demand, so that it reflects features of both preferences and the household production technology.

An increase in q will lower the marginal cost of the household production of z_1 and therefore increase the quantity demanded of x_1. If the change in q is to increase the compensated demand for x_1, then the household production technology and preferences together must result in an increase in the demand for z_1 sufficiently large that more x_1 is required. In other words, restrictions on the household production technology alone are not sufficient to establish the complementarity or substitutability of the derived demand for x_1 with respect to q. A comparative statics analysis of the household maximization problem shows that with

$$\frac{\partial z_1}{\partial x_1} \cdot \frac{\partial x_1}{\partial q} > 0 \tag{4.63}$$

it is possible for an increase in q to reduce both the ordinary and the compensated derived demands for x_1. See, for example, Courant and Porter (1981, 325–326).

The area between the two derived compensated demand curves for x_1 is given by

$$e\left(p_1^*,\ q'',\ u^0\right)-e\left(p_1',\ q'',\ u^0\right)-e\left(p_1^*,\ q',\ u^0\right)+e\left(p_1',\ q',\ u^0\right). \tag{4.64}$$

If the first and third terms of this expression sum to zero, then this gives the welfare value of the increase in q. The sufficient conditions for these terms to sum to zero are: (a) that x_1 be essential in the production of z_1 so that if $x_1 = 0$, $\partial z_1/\partial q = 0$; and (b) that q not affect utility independent of its contribution to the production of z_1, in other words, that $\partial u/\partial q = 0$ when $x_1 = 0$.

These two conditions are sufficient to make the expenditure function independent of the level of q. Thus, they play a role similar to the second condition of the weak complementarity model. The household production framework requires restrictions on both the household production technology and preferences for final services; but there is no restriction on the Hicksian demand analogous to the existence of a choke price for the market good.

Hedonic Prices and the Value of q

The techniques described so far have been developed for the case where the level of the resource service or environmental quality is fixed and is the same for all individuals. Although this represents the textbook version of the public goods problem, it is not descriptive of all possible cases involving public goods or environmental quality. In some circumstances, the level of q can be considered to be a qualitative characteristic of a differentiated market good. In these cases, individuals have some freedom to choose their effective consumption of the public good or environmental quality through their selection of a private goods consumption bundle. Another way to look at it is that there is a kind of complementarity between the public good and the market good, in that as the quantity of the public good embodied in the market good increases, the demand for the market good increases.

In effect, the market for the differentiated private good functions also as a market for the public good or environmental quality. For example, people can choose the level of consumption of local public goods through their choice of a jurisdiction to reside in; thus, the housing market functions also as a market for the purchase of local public goods.

Where these choices are possible, information on public good demand is embedded in the prices and consumption levels for private goods. For example, if air quality varies across space in an urban area, individuals may choose their exposure to air pollution through their decisions about residential location. Residential housing prices may include premiums for locations in clean areas and discounts for locations in dirty areas. If they do, it may be possible to estimate the demand for public goods such as clean air from the price differentials revealed in private markets. These price

differentials are implicit prices for different levels of the public good. A job can also be considered as a differentiated good. Different jobs have different bundles of characteristics and different wage rates. Wage differentials can be interpreted as the implicit prices of job characteristics. These wage and price differentials and the implicit prices they reflect are the subject matter of hedonic price theory.

The hedonic price technique is a method for estimating the implicit prices of the characteristics that differentiate closely related products in a product class. Colwell and Dilmore (1999) cited examples of the technique being applied to prices of farmland as early as 1922. The earliest modern example appears to be Griliches' (1961) application to the prices of automobiles. Rosen (1974) developed the formal theory of hedonic prices in the context of competitive markets. For an early development of the use of hedonic prices for estimating the demand for environmental quality characteristics, see Freeman (1974).

In principle, if the product class contains enough products with different combinations of characteristics, it should be possible to estimate an implicit price relationship that gives the price of any model as a function of the quantities of its various characteristics. This relationship is called the hedonic price function. The partial derivative of the hedonic price function with respect to any characteristic gives its marginal implicit price—that is, the additional expenditure required to purchase a unit of the product with a marginally larger quantity of that characteristic.

More formally, let Y represent a product class. Any model of Y can be completely described by a vector of its characteristics. Let $\boldsymbol{Q} = (q_1, \ldots, q_K)$ represent the vector of characteristics of Y. Then any configuration of Y, say y_i, can be described by its characteristics—that is $y_i = y_i(q_{i1}, \ldots, q_{iK}) = y_i(\boldsymbol{Q}_i)$, where q_{ik} is the quantity of the kth characteristic provided by configuration i of good Y. The hedonic price function for Y gives the price of any configuration, as a function of its characteristics. Specifically, for y_i,

$$p_y = p_y(q_{i1}, \ldots, q_{iK}). \tag{4.65}$$

If $p_y(\cdot)$ can be estimated from observations of the prices and characteristics of different models, the price of any model can be calculated from knowledge of its characteristics.

Before turning to the interpretation of the hedonic price function, it will be useful to describe briefly how the hedonic price function is generated in a competitive market for a differentiated product. Following the analysis of Rosen (1974), assume that each individual purchases only one unit of Y in the relevant time period. An individual's utility depends upon that person's consumption of the numeraire, z, and the vector of characteristics provided by the unit of Y purchased:

$$u = u(z, \boldsymbol{Q}) \tag{4.66}$$

or, substituting out the numeraire good using the budget constraint $p_{yi} + z = M$

$$u = u\left[M - p(\boldsymbol{Q}), \boldsymbol{Q}\right]. \tag{4.67}$$

In order to maximize equation (4.67), the individual must choose levels of each characteristic (q_k) to satisfy

$$\frac{(\partial u / \partial q_k)}{(\partial u / \partial z)} = \frac{\partial p_y}{\partial q_k}. \tag{4.68}$$

That is, the marginal willingness to pay for q_k must just equal the marginal cost of purchasing more of q_k, with other things being equal.

While $p_y(\boldsymbol{Q})$ represents the price that an individual *must* pay for configuration $\boldsymbol{Q}$, Rosen (1974) distinguished this from what the individual would be willing to pay for the configuration. Specifically, he defined the bid function $B\left(M, \boldsymbol{Q}, u^*\right)$ as implicitly solving

$$u\left[M - B\left(M, \boldsymbol{Q}, u^*\right), \boldsymbol{Q}\right] = u^*, \tag{4.69}$$

where u^* is the solution to the constrained utility maximization problem. Since $M = z + p_y(\boldsymbol{Q})$, the bid function can also be viewed as providing the indifference curve mapping, capturing the tradeoffs among the individual attributes of Y and the numeraire good z for a given level of utility u^*.

In order to focus attention on a single attribute, q_k, let

$$B_k(q_k) = B\left(M, q_k, \boldsymbol{Q}^*_{-k}, u^*\right) \tag{4.70}$$

denote the bid function for attribute k, fixing all of the individual attributes *except* q_k at their optimal levels $\boldsymbol{Q}^*_{-k}$. Because of differences in preferences and/or incomes, individuals can have different bid functions. Two such functions for individuals a and b are shown in Figure 4.5. They both show diminishing marginal willingness to pay for q_k or a diminishing marginal rate of substitution between q_k and z. Given the hedonic price function, these two individuals choose q_k^a and q_k^b, respectively.

Turning to the supply side of the market, firms' costs of production depend upon the levels of the characteristics of the models they produce. Assume that firms are heterogeneous so that their cost functions are different. Inverting a firm's profit function yields an offer curve for the characteristic of the form

$$C_k = C_k\left(q_k, \boldsymbol{Q}^*_{-k}, \pi^*\right), \tag{4.71}$$

where π^* is the maximum attainable profit. Offer curves and the optimal quantities of q_k supplied for firms α and β are shown in Figure 4.6 for a given hedonic price function.

For all firms and individuals to be in equilibrium, all of the bid and offer curves for characteristics for each participant in the market must be tangent to the hedonic price function. Thus, the hedonic price function is a double envelope of the two families of bid curves and offer curves (Rosen 1974). As a double envelope, the hedonic price function depends on the determinants of both the supply side

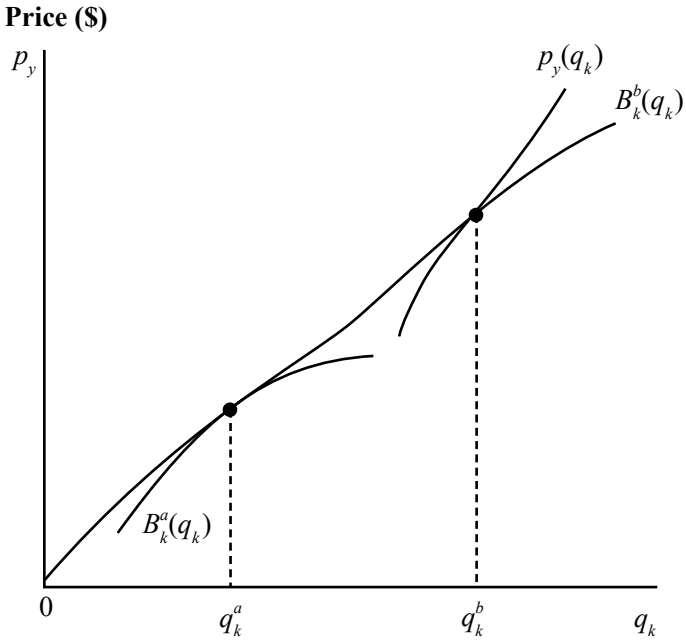

Figure 4.5 Bid curves of buyers in a hedonic market

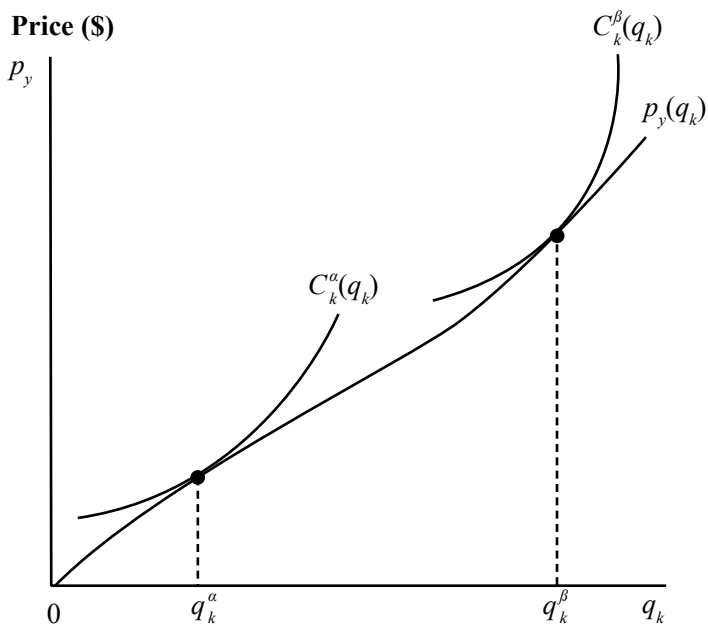

Figure 4.6 Offer curves of sellers in a hedonic market

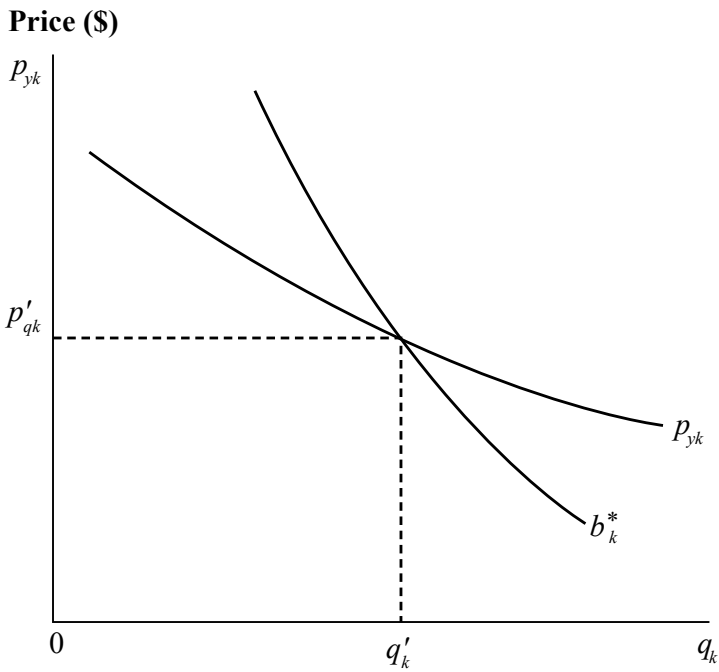

Figure 4.7 The marginal implicit price and inverse demand curves for *q*

and the demand side of the characteristics markets. It is a locus of equilibria between bids for, and offers of, characteristics.

The marginal implicit price of a characteristic can be found by differentiating the hedonic price function with respect to that characteristic—that is,

$$\frac{\partial p_y}{\partial q_k} = p_{yk}(q_1, \ldots, q_K). \tag{4.72}$$

This gives the increase in expenditure on Y that is required to obtain a model with one more unit of q_k, other things being equal. If equation (4.65) is linear in the characteristics, then the implicit prices are constants for individuals. However, if equation (4.65) is nonlinear, then the implicit price of an additional unit of a characteristic depends on the quantity of the characteristic being purchased.

Equation (4.65) need not be linear. Linearity will occur only if consumers can "arbitrage" attributes by untying and repackaging bundles of attributes (Rosen 1974, 37–38). For example, if individuals are indifferent between owning two two-door cars and one four-door car, other things being equal, they can create equivalents of four-door cars by repackaging smaller units. If both sizes exist on the market, the larger size must sell at twice the price of the smaller one, and the hedonic price of a car will be a linear function of the number of doors. The example suggests that nonlinearity will be a common feature of hedonic price functions.

Another way of looking at the market equilibrium for a characteristic helps to make clearer the welfare implications of the hedonic price model. First, individual a's choice problem can be solved to obtain that person's uncompensated inverse demand function for q_k

$$b_k^{*a} = b_k^{*a}\left(q_k, \boldsymbol{Q}_{-k}^{*}, M - p_y\right), \tag{4.73}$$

where a indexes the individual. This is shown in Figure 4.7 along with the marginal implicit price function for q_k, $p_{yk}\left(\boldsymbol{Q}\right)$. For each individual, the quantity of q_k purchased is known by observation and its implicit price is known from equation (4.72). This point $\left(q_k', p_{yk}'\right)$ can be interpreted as a utility-maximizing equilibrium for this individual resulting from the intersection of the individual's inverse demand curve and the locus of opportunities to purchase q_k, as defined by the marginal implicit price function. Thus, from the first-order conditions of equation (4.68) p_{yk}' can be taken as a measure of the individual's equilibrium marginal willingness to pay for q_k.

Individuals can be viewed as moving out along p_{yk}' as long as their willingness to pay, as reflected in their inverse demand curves, exceeds the marginal implicit price. Thus, p_{yk}' is a locus of individuals' equilibrium marginal willingness to pay.

It is also possible to specify a compensated marginal willingness-to-pay function for q_k,

$$b_k^{a} = b_k^{a}\left(q_k, \boldsymbol{Q}_{-k}^{*}, u^*\right). \tag{4.74}$$

This function gives individual a's marginal willingness to pay for q_k, holding utility constant. The value of a nonmarginal change in q_k can be easily calculated if the compensated marginal willingness-to-pay function equation (4.74) is known. This value is given by the integral of equation (4.74) over the range of the change in q_k.

This leads us naturally to the question of whether individuals' marginal willingness-to-pay and/or their inverse demand functions can be identified from observations of marginal implicit prices and quantities. The answer depends on the circumstances of the case. If the implicit price function is linear in q_k, then it is not possible to identify a demand curve for q_k. The price observation is the same for all individuals. However, p_{yk}' can be interpreted as the marginal willingness to pay or marginal benefit for small changes in q_k for each individual.

If the hedonic price function is nonlinear, different individuals selecting different bundles of characteristics will have different marginal implicit prices for q_k. There is one situation where the inverse demand function can be immediately identified—when all individuals have identical incomes and utility functions. When this occurs, the marginal implicit price function is itself the inverse demand function. Recall that the marginal implicit price curve is a locus of equilibrium points on individuals' inverse demand curves. With identical incomes and preferences, all individuals have the same inverse demand curve. Since all the equilibrium points fall on the same inverse demand curve, they fully identify the curve.

In the case where differences in incomes, preferences, or other variables result in individuals having different inverse demand functions, Rosen (1974)

argued that implicit price and quantity data from a single market could be used to estimate this inverse demand function, provided that the standard identification problem of econometrics could be solved. It is now clear that this analysis is incorrect. The problem is that the data from a single hedonic market are insufficient to identify how the same individuals would respond to different implicit prices and incomes.

There are at least two ways in which estimates of inverse demand functions for q_k can be obtained from hedonic analysis. The first is to increase the quantity of information obtained from marginal implicit prices by estimating hedonic price functions for several separate markets, and then pooling the cross-sectional data on the assumption that the underlying structure of demand is the same in all markets (Freeman 1974; Brown and Rosen 1982; Palmquist 1984). The second approach is to impose additional structure on the problem by invoking a priori assumptions about the form of the underlying utility function. The nature of the identification problem in hedonic price models, and alternative approaches to dealing with it, are major topics of Chapter 10.

Nonuse Values

Up to this point in the chapter, the values derived for environmental amenities have been obtained by looking for telltale "footprints" in the marketplace associated with how individuals *use* these amenities to enhance their welfare— producing utility-yielding commodities, or using them as substitutes for, or complements to, market goods (such as recreation trips). This section considers the possibility that people value natural resources and environmental characteristics independently of any present or future use they might make of those resources. For example, people may gain utility from their knowledge that the Grand Canyon is preserved even though they expect never to visit the canyon. People may value the survival of whales, eagles, and other endangered species even though they never expect to see one of them. Nonuse values raise two key questions that will be discussed further in this chapter. First, is there a meaningful way to define use so that values that arise from use can be distinguished from those that are independent of use? Second, what methods are available to measure these "nonuse" values and/or test for their existence?

In the economics literature, natural resource values that are independent of people's present use of the resource have been variously termed "existence," "intrinsic," "nonuse," and "passive use" values. This latter term was coined by the D.C. Circuit Court of Appeals in its ruling in *Ohio v. U.S. Department of Interior* (880 F.2d 432, 1989), which legitimized the inclusion of these values in natural resource damage cases brought by the federal government. These values may arise from a variety of motives, including a desire to bequeath certain environmental resources to one's heirs or future generations, a sense of stewardship or responsibility for preserving certain features of natural resources, and a desire to preserve options for future use.

Evidence suggests that nonuse values can be large in the aggregate, at least in some circumstances. For example, a research team working for the state of Alaska

estimated that the lower-bound median lost nonuse value for U.S. residents due to the *Exxon Valdez* oil spill was about $30 per household in 1990 dollars or about $2.8 billion for the U.S. as a whole (Carson et al. 2003). If nonuse values are large, ignoring them in natural resource policymaking could lead to serious errors and resource misallocations.

The goal of this section is to examine the role of nonuse values in assessing the *total value* (i.e., the combined use and nonuse value) of a natural resource or environmental amenity. The section begins with a review of some of the earlier literature in which various forms of nonuse values are discussed. A utility theoretic framework is then developed to define nonuse value—though in doing so it becomes clear that the distinction between use and nonuse values is an artifact of the specific functional forms chosen by the analyst. The section concludes with a discussion of the empirical techniques for the measurement of whatever we choose to call nonuse values. The possibility of deriving measures of nonuse values from observed behavior of individuals is considered, but this does not appear to be a fruitful approach. This leaves stated preference methods (alone or in combination with revealed preference methods) as the basis for both measuring nonuse values and for testing for their existence. Some discussion is provided regarding the issues involved in applying stated preference valuation techniques to the measurement of nonuse values, though this is left in large part to Chapter 12.

Background: Motivation and Definitions

John Krutilla (1967) introduced the concept of existence or nonuse values into the mainstream economics literature. In his classic article "Conservation Reconsidered," he argued that individuals do not have to be active consumers of a resource, whose willingness to pay can be captured by a price-discriminating monopoly owner, in order to derive value from the continuing existence of unique, irreplaceable environmental resources. Krutilla wrote that "when the existence of a grand scenic wonder or a unique and fragile ecosystem is involved, its preservation and continued availability are a significant part of the real income of many individuals" (Krutilla 1967, 779). In an accompanying footnote he added that "these [individuals] would be the spiritual descendants of John Muir, the present members of the Sierra Club, the Wilderness Society, the National Wildlife Federation, the Audubon Society and others to whom the loss of a species or the disfigurement of a scenic area causes acute distress and a sense of genuine relative impoverishment" (Krutilla 1967, 779n7).

Krutilla went on to suggest at least two reasons why people might hold values unrelated to their current use of a resource. These reasons were related to bequeathing natural resources to one's heirs and preserving options for future use. In a footnote he suggested that the "phenomenon discussed may have an exclusive sentimental basis, but if we consider the bequest motivation in economic behavior … it may be explained by an interest in preserving an option for one's heirs to view or use the object in question" (Krutilla 1967, 781n11). He also wrote, "an

option demand may exist not only among persons currently and prospectively active in the market for the object of the demand, but among others who place a value on the mere existence of biological and/or geomorphological variety and its widespread distribution" (Krutilla 1967, 781). However, literature that is more recent has recognized that the value of preserving options arises out of uncertainty about future demand and/or supply and has different implications for environmental valuation. For a brief discussion of this issue and references to the literature, see Chapter 5 of this book.

Later, Krutilla and Fisher wrote:

> In the case of existence value, we conceived of individuals valuing an environment regardless of the fact that they feel certain they will never demand *in situ* the services it provides … however, if we acknowledge that a bequest motivation operates in individual utility-maximizing behavior … the existence value may be simply the value of preserving a peculiarly remarkable environment for benefit of heirs.
>
> (Krutilla and Fisher 1975, 124)

Whereas Krutilla and Fisher offered a bequest motivation as one of several possible explanations for a pure existence value, McConnell (1983) took a different point of view:

> In most cases, resources are valued for their use. Existence value occurs only insofar as bequest or altruistic notions prevail. We want resources there because they are valued by others of our own generation or by our heirs. Thus, use value is the ultimate goal of preferences that yield existence demand, though the existence and use may be experienced by different individuals.
>
> (McConnell 1983, 258)

Thus, in McConnell's view, an altruistic attitude toward other people's use of a resource is the primary source of existence value. Although the source of existence value is related to someone's use, it is independent of any use made of the resource by the person holding the existence value.

Some have argued that people can have what are essentially existence values out of an ethical or altruistic concern for the status of nonhuman species or proper rules of human conduct (Kopp 1992). While philosophers of ethics are not in agreement as to the validity and proper form of such concerns—for example, see one of the collections on environmental ethics such as Van De Veer and Pierce (1994), DesJardins (1999), or Jamieson (2000)—it is possible that some people hold such values and are willing to commit resources on the basis of those values. Such values could be entirely independent of any use of the environment.

There has also been a lot of discussion about the various forms that nonuse values might take. A typical approach in the literature is to define use value as the economic value associated with the *in situ* use of a resource—for example,

through visiting a recreation site or observing a natural wildlife population. Total value is viewed as an individual's willingness to pay to preserve or maintain a resource in its present state. If total value exceeds use value, the difference is a nonuse value, or as some have called it, an "existence value," an "intrinsic value," or a "preservation value." Several authors have chosen one of these terms to represent the total difference and then identified various possible components of this total, often based on assumed motivations for holding these values. For example, Fisher and Raucher (1984) used the term "intrinsic value" to refer to the aggregate, and stated that the total intrinsic value is the sum of option value, aesthetic value, existence value, and bequest value. Sutherland and Walsh (1985) used "preservation value" to refer to the aggregate and stated that it is the sum of option, existence, and bequest values.

The attention given to questions of classification and motivation is perhaps misdirected. Motivations do not play an important role in the empirical analysis of the demands for market goods. There is little talk of "prestige value" and "speed value" in the literature on the demand for automobiles. So why should motivations be important in the case of nonuse values? Arguments about motivations seem to be offered primarily to persuade the reader of the plausibility of the hypothesis that nonuse values are positive. However, the real test of this hypothesis will come from the data. Rather than further debating definitions and possible motivations, it would be more useful to proceed with a test of the hypothesis that nonuse values (defined in a way that makes testing of the hypothesis feasible) are positive. If the evidence supports this hypothesis, then further research might be devoted to testing hypotheses about the determinants of the size of nonuse values in different cases. At that point, investigation of motivations might be useful in formulating hypotheses for testing. Then the choice of terms and explanatory variables, however, would be governed by what are empirically meaningful distinctions.

One of the major issues in the literature on nonuse values is how to define the use that lies behind use value. The most common approach is to identify some market good or service that is a complement to the resource, as regards consumption, and to define and measure use in terms of the purchased quantity of this complementary good. If the resource is a park, the complementary good is the purchase of travel services to the park, and use of the resource is measured by the number of trips purchased. However, this approach to defining use is clearly a simplification of a more complex reality. The physical proximity that one normally thinks of as being an essential part of use can occur independently of the purchase of a complementary good such as travel. For example, people who live within the natural range of an endangered species such as the California condor or wood stork may be able to view one of these birds and experience the utility and value associated with the sightings as an incidental part of their daily routine. In that case, there is no connection between that kind of use and any market good.

Some have argued that use does not require the physical proximity of the user and the resource. In their view, use can be defined to include the purchase of a

complementary market good that embodies some visual or literary representation of the resource. Randall and Stoll (1983), for example, argued that there can be off-site uses, which they label as "vicarious consumption." Examples of vicarious consumption could include watching a nature video, observing a live webcam of an eagle's nest, or watching a YouTube post of a forest fire. This is essentially what Boyle and Bishop (1987) chose to call indirect use value.

Defining use in this way creates some problems for the measurement of value. One problem with such vicarious or indirect forms of use is that the observable market transaction, such as the purchase of a nature magazine, often entails the simultaneous use of many environmental resources, so that allocation of the market transaction to specific resources is not possible. Furthermore, where vicarious uses involve information conveyed by photographs, films, and the like, the public-good dimension of information seems likely to virtually destroy any meaningful relationship between observed market behavior and underlying values. In addition, vicarious use has the odd feature that it can occur even though the resource itself no longer exists (as in the viewing of films and photographs).

This discussion of the possible definitions of, and motivations for, various types of use and nonuse values highlights the difficulty that authors have had in making clear distinctions among the concepts. However, definitions can be considered in part a matter of taste. A set of definitions can be considered useful if it furthers research objectives and leads to useful answers to meaningful questions, and if the definitions are based on operationally meaningful distinctions. If use values are limited by definition to those associated with *in situ* use, as measured by the purchase of a complementary good, then the definition has the virtue of distinguishing between cases where use of a resource generates observable market data and cases where no meaningful data can be obtained by observing market transactions. However, this definition leaves out vicarious or indirect uses as well as what one might call incidental uses. This argues for dropping the distinction between use and nonuse values and instead distinguishing between values that can be estimated with revealed preference data versus those that cannot. In the next section, a theoretical framework is provided that distinguishes between those cases where changes in the level of q lead to changes in the behavior of individuals and those where they do not. This framework is then used to examine whether total value can be usefully decomposed into component parts.

A Theoretical Framework

Assume that an individual has a preference ordering over a vector of market goods, $\boldsymbol{X}$, and some nonmarketed resource, q. The individual has no control over the level of q, but takes it as given. Here, q is taken to be a scalar measure of some characteristic of the environment, such as the population of some species or the value of some parameter of water quality. In the abstract, q can represent a measure of either a quantity or a quality. The choice of a unit for measuring q

has important implications for measurement in practice, but that question is not addressed here. The assumption that the environmental resource can be described by a single attribute is clearly a simplification. A more realistic model would allow for simultaneous changes in two or more quantitative and/or qualitative characteristics of the resource.

The notion that the total value of the resource q is not solely tied to the purchase of a complementary market good can be captured by assuming that the individual's direct utility function takes the form

$$U = T\left[u(\boldsymbol{X},q),q\right], \tag{4.75}$$

where $T(u,q)$ is increasing in u and q, and $u(\boldsymbol{X},q)$ is increasing and quasi-concave in $\boldsymbol{X}$ and q. In this case, the nonmarketed resource q enters utility in two distinct places, first by interacting with the marketed commodities, $\boldsymbol{X}$, and second on its own. The key feature of this functional form is that the marginal rates of substitution among the marketed goods, which dictate their observed consumption levels, are independent of this second role of q. Thus, information on $\boldsymbol{X}$ provides no information on this portion of preferences.

The functional form in (4.75) has been used by a number of authors to decompose the total value associated with a change in q into a series of component elements (e.g., Hanemann 1988, Freeman 2003, and Herriges, Kling and Phaneuf 2004, among others). The indirect utility function implied by (4.75) is given by

$$\begin{aligned} V(\boldsymbol{P},q,M) &= \max_{\boldsymbol{X}}\left\{T\left[u(\boldsymbol{X},q),q\right] \middle| \sum_j p_j x_j = M\right\} \\ &= T\left[v(\boldsymbol{P},q,M),q\right] \end{aligned} \tag{4.76}$$

where

$$v(\boldsymbol{P},q,M) = \max_{\boldsymbol{X}}\left\{u(\boldsymbol{X},q) \middle| \sum_j p_j x_j = M\right\} \tag{4.77}$$

captures that portion of utility that derives from the interaction of $\boldsymbol{X}$ and q in the marketplace. The total compensating variation CS^T for a change in the resource from q^0 to q^1 can then be implicitly defined by

$$T\left[v\left(\boldsymbol{P},q^0,M\right),q^0\right] = T\left[v\left(\boldsymbol{P},q^1,M-CS^T\right),q^1\right]. \tag{4.78}$$

Hanemann (1988) suggested identifying that portion of CS^T *revealed* by market transactions, as CS^R. This is defined implicitly by

$$T\left[v\left(\boldsymbol{P},q^0,M\right),q^0\right] = T\left[v\left(\boldsymbol{P},q^1,M-CS^R\right),q^0\right]. \tag{4.79}$$

The residual component of value (i.e., $CS^E \equiv CS^T - CS^R$), which is called *existence value* here, is then implicitly defined by

$$\begin{aligned} T\left[v\left(\boldsymbol{P},q^0,M\right),q^0\right] &= T\left[v\left(\boldsymbol{P},q^1,M-CS^R\right),q^0\right] \\ &= T\left[v\left(\boldsymbol{P},q^1,M-CS^R-CS^E\right),q^1\right]. \end{aligned} \tag{4.80}$$

The first line captures the impact that the change in q has on the portion of individual preferences revealed through the market (CS^R). The second line then indicates the *additional* compensation required to make the individual whole. However, as Mäler, Gren, and Folke (1994) and others have noted, this decomposition of total value is arbitrary. One could, for example, reverse the order of changes in (4.80), and decompose total value using $CS^T = \widetilde{CS}^E + \widetilde{CS}^R$, where the components of this decomposition are implicitly defined by

$$\begin{aligned} T\left[v\left(\boldsymbol{P},q^0,M\right),q^0\right] &= T\left[v\left(\boldsymbol{P},q^0,M-\widetilde{CS}^E\right),q^1\right] \\ &= T\left[v\left(\boldsymbol{P},q^1,M-\widetilde{CS}^R-\widetilde{CS}^E\right),q^1\right]. \end{aligned} \tag{4.81}$$

The problem is analogous to the one that arises when multiple price changes are sequenced in calculating compensating variations. By path independence, the sum, CS^T, is independent of the sequence of evaluation. However, as in the case of multiple price changes, the individual components of the sum are not.

Herriges, Kling, and Phaneuf (2004) further decomposed CS^R when q is tied to a specific market commodity or subset of market commodities, as is the case, for example, when q represents an environmental attribute associated with a recreational site. Let x_1 denote the number of trips to the associated site. CS^R can then be decomposed into two pieces, direct use value, and indirect use value as follows:

$$CS^R = CS^U + CS^{IU} \tag{4.82}$$

where $CS^U \equiv CS^R - CS^{IU}$ and CS^{IU} is implicitly defined by

$$v\left[p_1^*\left(q^0\right),\boldsymbol{P}_{-1},q^0,M\right] = v\left[p_1^*\left(q^1\right),\boldsymbol{P}_{-1},q^1,M-CS^{IU}\right], \tag{4.83}$$

with $p_1^*\left(q^s\right)$ denoting the choke price for x_1 when $q = q^s$ (s = 0, 1) and $\boldsymbol{P}_{-1} = \left(p_2,\ldots,p_J\right)$ denoting the vector of all the remaining market prices. Herriges, Kling, and Phaneuf (2004) suggested referring to CS^{IU} as a measure of "indirect use" value, as it corresponds to welfare changes stemming from changes in q when the corresponding market commodities are not in use, whereas CS^U corresponds to direct use value. The main distinction between CS^{IU} and CS^E is that the former affects market behavior, and therefore may be revealable from observations on market transactions, whereas the latter leaves no such footprint.

Unfortunately, while the above decompositions of total value have some intuitive appeal, they are of little guidance to the practitioner, particularly one faced with only revealed preference data. With only revealed preference data, the best one can hope to identify is the revealable portion of value CS^R. Yet, even in this instance, a fundamental problem remains—as noted above, division of total value between CS^E and CS^R is not unique. Indeed, Herriges, Kling, and Phaneuf (2004) provided an empirical example in which two different specifications for the partial indirect utility function $v(\boldsymbol{P}, q, M)$ that yield *identical* Marshallian demand equations result in estimates of CS^R that differ by more than a factor of three.

One tempting approach to this problem would be to take a "conservative" approach acknowledging the possibility of existence values, but to report CS^R as a conservative estimate of total value. Unfortunately, CS^R is not necessarily a lower bound on the overall compensating variation. Indeed, Herriges, Kling, and Phaneuf (2004) provided several counter-examples to show that there is no unique decomposition and that specific results are analogous to the path dependence problem in integrating demand functions over multiple price changes.

These findings are troubling. In simple terms, they imply that if an individual's preferences do not exhibit weak complementarity, then it is not even possible to estimate a lower or upper bound on that individual's value of a change in environmental quality based on market-based information. Intuitively, the lack of weak complementarity means that not all of the information related to use value of a quality change is contained in the demand for the market good. If the lack of weak complementarity arises due to the fact that there is more than one private good that has joint weak complementarity with q, then the Bockstael and Kling model described above can be used to estimate the total value. If this option is not possible, then this leaves analysts with two real alternatives in terms of capturing the total value associated with a change in q.

The first, as Bockstael and McConnell (2007, 66) suggested, is to take the position that, while weak complementarity is not a testable hypothesis (at least on the basis of revealed preference data alone), it "is often a good assumption to maintain" and that "cases where weak complementarity would be a misguided assumption—for example, a travel cost study of the demand for access to the Arctic National Wildlife Refuge—will often be obvious." Assuming weak complementarity eliminates the ambiguity in terms of the resource's value, since, in this case, $CS^E = CS^{IU} = 0$, so that $CS^T = CS^R = CS^U$ (i.e., all the value associated with the resource q is tied to the *use* of the weakly complementary market commodity x_1).

Second, the analyst can seek to eliminate the ambiguity in terms of individual preferences by augmenting available revealed preference data using information from stated preference sources. As Ebert (1998) argued, the fundamental problem when assessing the welfare impact of changes in the level of a nonmarketed commodity is that the ordinary demand equations for the marketed commodities do not contain all of the information needed to recover preferences. In particular, any effort to integrate back from observed demands to underlying preferences must make assumptions about how the constant of integration changes with changes in

the public good, q. Weak complementarity provides one such set of assumptions. Ebert argued, however, that stated preference techniques can be used to obtain the missing information in the form of inverse demand equations for the public goods. Eom and Larson (2006) provided an illustration of this approach. The downside, of course, is that the technique relies on the assumption that stated preference elicitation methods provide an accurate representation of individual preferences—an assumption that (in the case of nonuse values at least) is once again untestable using revealed preference data alone. This topic will be explored in more depth in Chapter 12.

The Household Production Framework and Use Values

The theory of household production provides an alternative framework for thinking about how resources affect utility and welfare and what it means to assume that x_1 is a measure of the use of a resource. Recall that in the household production framework, the demand functions for market goods are derived demands. Assume that q is an input in the production of only one final service flow, z_1. If x_1 is also an input in the production of z_1, then (as shown earlier in this chapter) q will be an argument in the ordinary and compensated demand functions for x_1. If x_1 and q are the only inputs in z_1, then the value of q in producing z_1 will be reflected in the derived demand for x_1 (Bockstael and McConnell 1983). If x_1 and q are both essential inputs, then the marginal product of each is zero if the other input is absent. A positive level of input for x_1 is required to utilize q in household production. This is equivalent to saying that the use of q, and hence its use value, is reflected in the level of x_1.

Another advantage of thinking about resource values in the context of the household production framework is the light that it sheds on the nature of what have been called indirect, or vicarious, use values. It appears that there are two categories of indirect or vicarious values—those that derive from household production and those that derive from market production. If the vicarious use involves the viewing of home-produced films and photos, then the theory suggests that the place to look for measures of indirect use values is in the influence of q on the demands for other inputs in the production of films and photos. If a market good is an essential input to this indirect use, then in principle the use value of changes in q can be measured by areas between the demand curves for the input in home production.

On the other hand, if indirect, or vicarious, use is the viewing of commercially produced films and television programs or the reading of books and magazines, the source of value for q is in its influence on either the production function or the quality-differentiated demand functions for these marketed goods. The place to look for measures of this type of value is either in changes in costs and prices or in the hedonic price functions for these goods. Thus, the role of q is analytically no different from the role of, say, air quality in the production of agricultural crops. There is a well-developed methodology for measuring values in this case, as

shown in Chapter 8. However, it seems doubtful that a reduction in the availability of a specific resource would increase the cost of producing the editorial content of films, books, and magazines, so the empirical significance of this approach to measuring values of changes is in question.

It seems reasonable to assume that the demand for a specific magazine or television program will depend in part on the quality and the information content of the material presented. However, the link between changes in q and changes in the demand for information about a specific resource is complex and may not meet the requirements of the standard models for estimating welfare values from changes in demand. For example, a major pollution event that damages a unique resource system may increase the demand for information about that resource; and producers are likely to respond with more articles and programs about that resource. This would be an increase in vicarious use, but it could hardly be called a benefit resulting from damage to the resource.

Altruism and the Relevance of Nonuse Values

As noted in the first part of this section, one possible reason that nonusers might value the preservation of a resource is an altruistic feeling toward those who do use it. There is literature on altruism and how the presence of altruistic feelings toward others affects the Pareto optimum or efficient allocations of resources. One conclusion in this literature is that if those feelings take a form known as nonpaternalistic altruism, they have no effect on the Pareto optimum allocation—that is, they are irrelevant for benefit-cost analysis (see Lazo, McClelland, and Schulze 1997, and references cited therein). If nonusers' willingness to pay for the preservation or improvement of a resource is due to nonpaternalistic altruism, this means that the nonuse values should not be counted in a benefit-cost analysis (Milgrom 1993).

Nonpaternalistic altruism refers to a case where one individual cares about the general level of well-being of others but does not have any preferences regarding the composition of consumption bundles of others. If individual A had such preferences, they could be represented by

$$u^A = u^A\left[\boldsymbol{X}^A, u^B\left(\boldsymbol{X}^B, q^B\right)\right]. \tag{4.84}$$

An increase in q would result in an increase in u^B; and therefore, A would also be better off. A's willingness to pay for this improvement could be defined in the usual fashion; but it would not be proper to add this to B's willingness to pay in an economic assessment of the policy, at least if the increase in q was not costless. This is because someone has to bear the cost of the increase in q. If B bears the cost, this decreases B's utility, and therefore A's willingness to pay for the increase. It can be shown that as long as altruism takes the form shown in equation (4.84) the terms representing altruism cancel out of the conditions for Pareto optimality. However, if A's altruism stems from a concern for the level of q that B experiences, then A's willingness to pay for B's improvement in

q is relevant for economic assessments (Lazo, McClelland, and Schulze 1997; McConnell 1998).

In summary, if the sole source of nonuse values is nonpaternalistic altruism, these values are irrelevant for policy purposes. However, if altruism takes the paternalistic form, then the resulting nonuse values are relevant for policy analysis; and, nonuse values can arise for other reasons besides, or in addition to, altruism.

Toward Measurement

If one accepts the argument that for practical purposes nonuse values should be defined as those values not revealed by market behavior, then by definition other strategies for valuation must be sought. Stated preference methods are an obvious approach, but, before leaving behavioral observations altogether, it is useful to ask whether there is any other behavior, outside of market behavior, where information that might lead to measurement of nonuse values could appear. Since it is concluded that these are unlikely to be fruitful avenues of research, the remainder of the subsection focuses on how stated preference methods might be used as part of a research strategy for measuring nonuse values as a separate component of total value. The subsection concludes with a discussion of other issues that must be resolved in any effort to measure and/or isolate nonuse values.

Because preservation of a resource for those who do not make *in situ* use of it has the properties of nonexcludability and nondepletability, one would expect markets to fail to provide these preservation services, or at least to provide them in suboptimal quantities. Unless there is a market for preservation of a resource or for enhancing its quality for nonusers, there will be no market transactions to reflect the preservation values of individuals, and only stated preference methods of estimating nonuse values will be feasible.

Environmental organizations are observed undertaking a variety of activities to protect and preserve natural environments, and people support these activities through voluntary contributions of time and money. The question is whether this revealed behavior provides an adequate basis for measuring willingness to pay for preservation. For several reasons, it seems unlikely that economic data on either the activities of these organizations or individuals' contributions to them can be relied upon as measures of the value of preservation for policymaking purposes.

The activities of environmental organizations can be placed in one of two categories: (a) direct provision of preservation through acquisition, and (b) advocacy in an effort to influence public-sector provision of preservation. Organizations such as The Nature Conservancy accept private donations and use the funds to purchase lands with special ecological, geological, or scenic characteristics for the purpose of protection and preservation. Individuals' donations and dues paid to such organizations are manifestations of willingness to pay for preservation. However, if "free-rider" behavior were significant, these donations would be only a lower bound of true aggregate willingness to pay. Furthermore, at least in some instances, the lands acquired by such organizations are accessible to individual

use; therefore, individual donations could reflect a combination of use as well as nonuse values.

Many environmental organizations devote a substantial portion of their budgets to advocacy activities on behalf of environmental preservation in general and for policy actions to protect specific natural resources. Again, individuals' membership dues and donations in support of these activities reflect individuals' willingness to pay for preservation. However, because of free-rider behavior, aggregate donations are likely to represent less than total willingness to pay. In addition, the organizations undertaking these activities frequently have multiple purposes—they provide services to members such as magazines and other publications, and field trips. This means that only the portion of dues and donations supporting the incremental cost of advocacy to the organization is relevant for estimating preservation values.

In the policymaking arena, another factor weakens the relationship between individual donations to support advocacy and preservation values. This factor is the uncertainty concerning the outcome of the policy process and the contribution of advocacy activities to the desired outcome. A rational organization with limited resources would estimate the probabilities of successful advocacy as a function of the resource commitment for each specific issue, and would allocate resources so as to maximize the expected value of the outcome. The observed allocation of advocacy resources across specific issues would reflect the interaction of the probabilities of a successful advocacy, the marginal productivity of advocacy in increasing the probability of success, and the value of success to the organization (Freeman 1969). It would probably prove difficult, if not impossible, to model this complex policy process so as to identify individuals' willingness to pay on the basis of observed contributions to advocacy on specific issues.

It is reasonable to conclude that individuals' donations to environmental organizations involved in acquisition and advocacy reflect willingness to pay for preservation and nonuse values. These activities provide evidence in support of the hypothesis of significant nonuse values. However, for the reasons outlined here, observed acquisition and advocacy expenditures are likely to be an underestimate of the values held by people who are not *in situ* users of the resource.

The stated preference valuation methods described in Chapter 12 are likely to offer the only feasible approaches to estimating nonuse values. Of course, what matters for policy purposes is total value regardless of how it is divided between use and nonuse value. For individuals who are nonusers (or who do not alter observable behavior in response to changes in q), stated preference questions provide estimates of total value, which consists entirely of nonuse value. For individuals who are users and have separable preferences such as those represented by equation (4.75), stated preference methods still yield estimates of total value; and, revealed preference methods yield underestimates of total value.

However, for research purposes there is some interest in learning more about the magnitudes of nonuse values and factors determining their size. Broadly speaking, there are two approaches to estimating nonuse values. The first is to use a stated

preference question to obtain an estimate of total value and then to deduct from this a separately obtained estimate of use value. The latter could be obtained, for example, by the travel cost method for estimating the demand for visits to a recreation site. As should be apparent, one problem with this approach to imputing nonuse values is that the imputation is based on the difference between two other values, both of which are measured with some unknown error. Therefore, without some understanding of the error properties of the other measures, one cannot know whether the imputed value is simply the result of measurement error or is a true nonuse value. Moreover, as demonstrated by Herriges, Kling, and Phaneuf (2004), the division of total value between use and existence value is itself an arbitrary allocation.

The second approach to estimating nonuse values is to ask people explicitly about their nonuse values, either by questioning people who are known to be nonusers or by asking people to assume that their own use is zero. Any value revealed by a nonuser is, by definition, a nonuse value; but if present users are asked to assume that they are not using the resource, their responses to stated preference questions may not be valid indicators of nonuse value. This is because the situation they are being asked to see themselves in is unfamiliar, and they might misunderstand it or reject it outright.

Two additional issues concerning research study design deserve at least brief mention. The first issue concerns the definition and description of the resource to be valued. Resources have both quantitative and qualitative dimensions that can be affected by policy decisions or damaged by pollution events. Yet, in the theoretical discussion above, it was assumed that a resource was measured in a single dimension; therefore, respondents must be given a clear description of the changes in all relevant dimensions of resource quantity and quality. The determination of what is relevant must come in part from the judgment of experienced researchers in this field and in part from research specifically designed to determine what characteristics of the resource are important to people.

The second issue concerns the relevant population for sampling when nonuse values are involved. If the resource to be valued is in California, for example, should the sample include East Coast residents, or should it be limited only to westerners or to California residents? Casting the sampling net too wide wastes scarce research resources, but important values may be missed if the geographic scope of the sample is too narrow. Even small per capita values can loom large when aggregated over a large population. Again, experience and the results of research designed specifically to shed light on this set of issues can help to guide research study design. It is unclear what more can be said about this issue until we know a lot more about what characteristics of resources are likely to give rise to significant nonuse values. Some resources such as the Amazonian rain forest and African elephants may have worldwide significance, implying that the relevant population is the world population. In addition, one cannot rule out the possibility that where there are regionally significant resources, important nonuse values are held only by people within that region. This is an important research question.

Summary

This chapter has provided and reviewed the conceptual basis for empirical techniques that attempt to take advantage of market information. Where q is an input into the production of marketed goods and services, the techniques focus on the effects of changes in q on output and factor markets, not on the utility of q per se. It was also shown that there is a valid theoretical basis for examining changes in factor incomes such as land rents, costs savings in production, and changes in consumer surplus associated with the private good outputs.

When q is a consumption good and enters directly into the utility function, the problems are more severe. One of the more promising techniques is based on the concept of weak complementarity. Benefits are measured in terms of shifts in the demand curve for the private complementary good. The approach may be applicable to estimating recreation benefits due to the water quality changes.

Where the public good is either implicitly or explicitly an input, along with one or more market goods, in the household production of a final service flow, benefit estimation is straightforward if the substitution relationship or marginal rate of technical substitution is known, and the problem has a sufficiently simple structure so that unobservable utility terms cancel out. Defensive expenditures and measures of additional costs (for example, for household cleaning or for medical care and drugs in the case of health) are examples of estimates that are based on approximations of this approach.

Another promising case is where q varies across space, as in air pollution, or as a characteristic embodied in some private good. Then individuals can choose different quantities of q by varying residential locations or by choosing different private good models. Finally, the hedonic price approach can be used to measure the implicit price of q; and, under some circumstances, the demand curve for q can be identified.

In most of the models described in the first part of this chapter, it makes intuitive sense to speak of the individual as using the environmental or resource service in question. For example, in the models of substitution and averting behavior, the individual can be interpreted as using clean air to produce a clean household or good health. Thus, the values estimated with the models described here are sometimes referred to as use values. It is also possible for individuals to value environmental services independently of any use they might make of those services. The concept of so-called nonuse values is the topic of the last section of this chapter. It was shown there that, when nonuse values are defined and modeled in a plausible way, the result is that transactions in market goods reveal nothing about these nonuse values. Therefore, the revealed preference methods described in this chapter can shed no light on the possible magnitude of these values; and instead, we must rely on stated preference methods, which are the topic of Chapter 12.

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Chapter 5

Valuing Changes in Risk

Up to this point, the discussion in this book has been based, at least implicitly, on the assumption of perfect certainty. For example, individuals have been assumed to know what prices, income, and environmental quality will prevail over the relevant planning horizon. This clearly cannot be literally true; and for some questions, the essence of the matter is uncertainty about some event or condition, such as the probability of an accident or a natural disaster (say, a flood) occurring or the probability of contracting a serious illness following exposure to a chemical. The topic of this chapter is how to define and measure changes in economic welfare when uncertainty is an important characteristic of the economic world in which people are living and making choices.

The uncertainty faced by individuals can take many forms. For example, people could be uncertain about future prices of consumer goods, future income, or the returns on financial assets. However, since there is already an extensive literature on the welfare effects of price and income variability, those problems are not treated here. See, for example, Just, Hueth, and Schmitz (2004) and Rothschild and Stiglitz (1976). The uncertainty that is important in this chapter is that which individuals who are users or potential users of some environmental service flow or resource face. Individuals might be uncertain as to whether a specific environmental service flow or resource will be available for use in the future; they might be uncertain as to whether they will actually want to use some resource in the future; or they could be uncertain as to whether their exposures to environmental hazards and pollutants will actually result in illness or death.

These uncertainties raise two kinds of questions for policymakers. First, what form of welfare criterion is appropriate under conditions of uncertainty, or more specifically, what modifications to the Hicks–Kaldor potential compensation test are required? Second, when public policies change the uncertainties facing people, how are the resulting welfare changes to be measured? These two questions are connected, as we shall see, in that the choice of a welfare criterion has important implications for how welfare changes are to be measured.

Most of the analysis in this chapter is based on the now standard model of individual preferences under uncertainty, which in turn is based on the hypothesis that individuals maximize their expected utility. The theory is very attractive as a

normative prescription for behavior and choice. However, there is now substantial evidence that individuals' choices frequently violate expected utility theory. For examples of this evidence, and discussions of its implications for the theory of preferences under uncertainty, see Kahneman and Tversky (1979), Arrow (1982), Thaler (1987), Machina (1990), and Kahneman (2011). Despite this evidence on the descriptive inadequacies of expected utility theory, its simplicity and elegance still make it attractive for expositional purposes.

In the next section, the standard theory is developed and used to define welfare measures in several contexts. Several alternative welfare indicators that can be derived from this model of preferences are discussed. In the second section, alternative aggregate welfare criteria and their implications for the way individuals' welfares change are described. In the third section, dynamic welfare measures that reflect the ability of individuals to gather information, learn, and plan their purchase decisions accordingly are introduced. In the fourth section, some models for the measurement of individuals' willingness to pay for risk-reduction and risk-prevention policies using revealed preference methods are presented. The implications of some forms of nonexpected utility preferences for welfare measurement are also examined. In the fifth section, the concept of option value is introduced and related to the welfare measures developed in the chapter.

Individual Uncertainty and Welfare

The term individual uncertainty refers to situations in which an individual is uncertain as to which of two or more alternative states of nature will be realized and is not indifferent as to which state actually occurs. In this discussion, individuals are assumed to assign probabilities to these alternative states of nature and these probabilities are assumed to be correct in the sense of summing to one and incorporating all available information. Thus, no distinction is made between risk (where probabilities are known) and uncertainty (where probabilities are unknowable). For a discussion of this often-cited distinction, which is often attributed to Frank Knight, see LeRoy and Singell (1987). This discussion also abstracts from questions of risk perception, how individuals make assessments of probabilities, and how these assessments are revised in light of new information. For introductions to some of the issues concerning risk perceptions and probability assessments, and for references to the relevant literature in psychology and economics, see DellaVigna (2009), Arrow (1982), Viscusi and Zeckhauser (2006), and Machina (1989).

Risk and Environmental Policy

Consider some environmental risk such as a chemical spill or pollution event, or some natural hazard such as a flood or an earthquake. Other examples of such risks include tornadoes and hurricanes, accidents at nuclear reactors, explosions in chemical plants, releases of toxic materials from hazardous waste

storage sites, and air pollution episodes associated with unusual meteorological conditions. Risks such as these can be described in terms of two characteristics: the range of possible adverse consequences, and the probability distribution across consequences. In this analysis, adverse consequences are measured in units that reflect the consequences for people (for example, the number of buildings damaged and degree of destruction, or the number of days of illness) rather than in terms of measures such as height of flood stage, Richter scale reading, or maximum atmospheric concentration of a pollutant. To keep the exposition simple, consider only one possible adverse event and two states of nature: the event occurs with a given set of consequences, or the event does not occur.

Suppose there is some public policy action that has the effect of reducing the probability of the adverse event, reducing the severity of its consequences, or both. Examples of policy measures that reduce the magnitude of an adverse event include regulations requiring earthquake-resistant construction techniques and the building of public shelters for protection against tornadoes and hurricanes. These are often referred to as *risk-reduction* measures. Examples of policy measures that reduce the probability of an adverse event include regulations for nuclear reactor safety, standards for durable containment techniques at hazardous waste storage sites, and the construction of dams to control streamflow to reduce the probability of flooding. These are often referred to as *risk-prevention* measures.

The distinction between risk reduction and risk prevention may sometimes be arbitrary. For example, the benefits of wearing a seatbelt could be modeled in terms of the belt's effect on the severity of injury associated with accidents of unchanged probability, or in terms of its effects on the probabilities of experiencing injuries of various severities per mile driven. The choice of a modeling strategy may depend upon such pragmatic considerations as availability of data.

Many risk-reduction and risk-prevention measures are public goods in that they have the characteristics of nonrivalry and nonexcludability. The public good character of these measures means that a private market system will fail to provide them in efficient quantities. Thus, there is a case for government intervention to improve the efficiency of resource allocation. In order to determine whether risk-reduction and risk-prevention measures result in improvements in welfare, it is necessary to define and measure the benefits and costs of changes in risks.

Individual Preferences and Expected Utility

Turning to the characterization of individual preferences in situations involving risk, assume that an individual has a well-behaved preference ordering over bundles of goods $\boldsymbol{X}$, and that there is some adverse environmental event A over which that person has no control. The variable A measures the severity of the adverse event. Let $A = A^*$ represent the occurrence of the adverse event, and $A = 0$ represent the absence of the adverse event. A takes the value A^* with probability π and 0 with the probability $1 - \pi$. For any given state of nature, this preference ordering can

be represented from an ex post perspective by $u(\boldsymbol{X},A)$ with $u(\boldsymbol{X},0) > u(\boldsymbol{X},A^*)$. Assume $u_A < 0$ and $u_{\boldsymbol{X}} > 0$, where subscripts indicate partial derivatives. To avoid unnecessary complexity, income M and prices $\boldsymbol{P}$ are assumed to be certain. Finally, assume that $u(\cdot)$ is the same in both states, but that preferences over $\boldsymbol{X}$ are state-dependent. The consumption bundle actually chosen can depend on the state of nature—that is, on whether $A = 0$ or A^*. This means that the marginal utility of $\boldsymbol{X}$ can vary across states of nature. The model can be easily generalized to many states and to make $u(\cdot)$ depend on the state of nature (for examples see, Graham 1981 and Freeman 1984b, 1985, 1989).

Given utility maximization, there is an ex post indirect utility function $v = v(\boldsymbol{P}, M, A)$ that shows the maximum attainable utility given $\boldsymbol{P}$, M, and A, where $\boldsymbol{P}$ is the vector of prices and M is income. It has the properties

$$\begin{aligned} &v_M > 0, \quad v_{MM} < 0, \\ &v_A < 0, \quad v_{AA} < 0. \end{aligned} \tag{5.1}$$

From here on the $\boldsymbol{P}$ term is suppressed, since prices are assumed to be unchanged through the analysis.

Assume that individuals know the magnitude of the adverse event A^*, the probability of its occurrence, and that there are no opportunities for individual protective activities—that is, no ways of spending money to reduce either π or A^*. Ehrlich and Becker (1972) referred to the latter activities as self-protection (for reducing π) and self-insurance (for reducing A^*) and analyzed individual behavior in the case of purely monetary losses. To the extent that self-protection and self-insurance activities are possible in the case of risks of nonmonetary losses, they may provide a basis for empirical estimation of the values of risk changes. This possibility is investigated in the section on revealed preference methods in this chapter.

Let D represent the monetary value of the damages caused by the event, given that the adverse event has occurred. D is the maximum sum of money the individual would give up to experience $A = 0$ rather than A^*, and is the solution to

$$v(M, A^*) = v(M - D, 0). \tag{5.2}$$

Thus, D is a form of compensating surplus (CS) measure of welfare change where the reference level of utility is the utility realized if A^* occurs.

People often have to make choices before the state of nature is revealed, in other words, ex ante choices. Let us assume that individuals make these choices so as to maximize their expected utility, where expected utility is defined as

$$E[u] \equiv \pi \cdot u(X, A^*) + (1 - \pi) \cdot u(X, 0). \tag{5.3}$$

The expected utility expression provides a basis for an alternative measure of the value of avoiding A^*, namely, the WTP ex ante. This state-independent payment is sometimes referred to as option price, or OP. Option price is defined as the maximum payment the individual would make to change from the status quo risk to a situation in which A^* would not occur. It is also a form of compensating

surplus, but one where the reference point is defined in terms of expected utility. *OP* is the solution to

$$\pi \cdot v\left(M, A^*\right) + (1-\pi) \cdot v\left(M, 0\right) = v(M - OP, 0)\,. \tag{5.4}$$

OP and $\pi \cdot D$ will not, in general, be equal. This is because they are measuring two different things, or more precisely, measuring the monetary equivalents of two different forms of utility change: the ex post change in $v(\cdot)$ and the ex ante change in $E[v]$. The relationship between these two measures is the basis of the extensive literature on option value. A review of this literature is contained in the fifth section of this chapter. Also, these two measures are not the only possible ways of expressing a compensating measure of the welfare value of preventing the adverse event. This can best be seen with the aid of Graham's (1981) WTP locus.

The Willingness-to-Pay Locus

Given the assumption that individuals maximize expected utility, their behavior can be described as the solution to the following problem:

$$\text{Max}: E[u] \equiv \pi \cdot u\left(X, A^*\right) + (1-\pi) \cdot u\left(X, 0\right), \tag{5.5}$$

subject to the usual budget constraint that expenditure equal income. The solution to this problem is denoted as E^1.

Now consider a policy that would reduce the magnitude of the adverse event from A^* to 0. If the individual were required to pay *D* to avoid the adverse effects given that the event occurs, and would pay nothing otherwise, the realized utility in each of the two states would be unchanged. Thus, one set of payments that would not change expected utility is (*D*, 0). Alternatively, if the individual were to pay *OP* before the uncertainty were resolved, by definition expected utility would also be unchanged. Thus, another pair of payments that leaves expected utility unchanged is (*OP*, *OP*). However, these are only two of an infinite number of possible pairs of payments that would leave the individual indifferent, in expected utility terms, between the status quo and a situation in which the consequences of the adverse event were avoided with certainty. The set of these payments is denoted as $\left(t^*, t^0\right)$:

t^* = payment given the state in which the adverse event occurs, and

t^0 = payment given the state in which it does not occur.

These payments satisfy the following condition:

$$E^1 = \pi \cdot v\left(M - t^*, A^*\right) + \left(1-\pi\right) \cdot v\left(M - t^0, 0\right). \tag{5.6}$$

This equation defines Graham's WTP locus, which is shown in Figure 5.1.

Similarly, for a risk-prevention policy the WTP locus is given by the solution to

$$E^1 = E^2 = \pi' \cdot v\left(M - t^*, A^*\right) + \left(1-\pi'\right) \cdot v\left(M - t^0, 0\right), \tag{5.7}$$

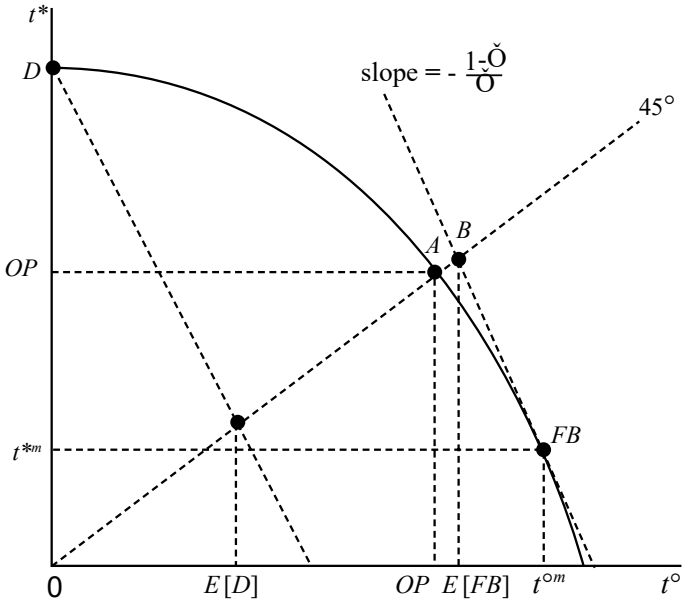

Figure 5.1 The willingness-to-pay locus

where π' is the probability of the adverse event with the risk-prevention policy in place. All of the subsequent analysis pertains to risk-reduction policies, but with suitable modification it can also be applied to the benefit-cost analysis of policies that change probabilities.

By setting the total differential of equation (5.6) equal to zero, we obtain an expression for the slope of the WTP locus:

$$\frac{dt^*}{dt^0} = \frac{(1-\pi)v_M^0}{\pi v_M^*}, \tag{5.8}$$

where v_M^* is the marginal utility of income evaluated at $M - t^*$ given the event has occurred, and v_M^0 is evaluated at $M - t^0$ given no adverse event. If the individual is risk-averse as assumed above ($v_{MM} < 0$), then the WTP locus will be concave to the origin—increasing t^* (for example) raises v_M^* relative to v_M^0. Risk neutrality, with its constant marginal utility of income, would result in a linear WTP locus.

Three points on the WTP locus are of particular interest as possible welfare measures. The first is the vertical intercept in Figure 5.1. This point involves a payment of zero if the event does not occur and there are no realized damages, and a payment of $t^* = D$ if the event does occur. This sum represents the maximum payment the individual would make to experience the consequence 0 rather than A^* given that the event occurs. This sum is also equal to the damages (D) from the event were it to occur. There is a line from this intercept with a slope of $-(1-\pi)/\pi$ representing the locus of all pairs of monetary values having the same expected value as D. Similar loci can be constructed for any t^*. These loci satisfy

$$E[t] = \pi \cdot t^* + (1-\pi)t^0 . \tag{5.9}$$

Differentiating, this gives

$$\frac{dt^*}{dt^0} = -\frac{(1-\pi)}{\pi} . \tag{5.10}$$

This line is one of a family of iso-expected payment lines. The point where the iso-expected payment line through point *D* intersects the 45° line can be used to identify the expected value of damages (*E*[*D*]). This is the ex post expected *CS* measure of the welfare change of preventing A^*.

A second point of interest in Figure 5.1 is *A*, which shows an alternative state-independent payment scheme in which $t^* = t^0 = OP$. In this example, the state-independent payment is greater than the expected value of damages, but this will not always be the case. The difference between *OP* and *E*[*D*] can be either positive or negative, and in some circumstances it can be quite large (see, for example, Graham 1981 and Freeman 1984b, 1985, 1989).

Finally, there is an iso-expected payment line tangent to the WTP locus at point *FB* in Figure 5.1. This point represents that state-dependent payment scheme, (t^{*m}, t^{0m}), which maximizes the expected value of the individual's payments (*E*[*FB*]). The tangency of the WTP locus and the iso-expected payment line at point *FB* implies that the marginal utilities of income are equal in the two states. Since the slopes of the WTP locus and iso-payment lines are equal, making use of equation (5.7) yields

$$-\frac{(1-\pi)}{\pi} = -\frac{(1-\pi)v_M^*}{\pi \cdot v_M^0} , \tag{5.11}$$

and $v_M^* / v_M^0 = 1$. This equality is the condition for the efficient distribution of risk and for the optimum purchase of contingent claims at actuarially fair prices.[1] In this sense, point *FB* represents an optimum contingent payment scheme. In Graham's (1981) terminology, this is the fair bet point.

As illustrated in Figure 5.1, these three alternative payment schemes can give quite different summary measures of maximum willingness to pay (WTP) for the policy. This is because they differ with respect to the implied opportunities for insurance—that is, opportunities to alter the payments in different states of nature to reflect attitudes toward risk. In the example of Figure 5.1, the fair bet point allows the individual to make a larger payment in the state of nature in which no adverse event occurs and in which, therefore, the individual is otherwise better off.

In order to see the relationship between state-dependent payments of this sort and insurance, suppose that actuarially fair insurance is available to the individual. Specifically, suppose that the individual could make a payment *I* before the uncertainty was resolved to purchase an insurance policy that would

1 A contingent claim is a contract specifying in advance a set of payments or receipts, or both, in which the amounts depend on the state of nature. An insurance policy is a contingent claim.

pay R if the adverse event occurred. This means that the individual can exchange income across states of nature according to $I/(R - I) = k$, the price of insurance. If the price of insurance is actuarially fair, then $k = \pi/(1 - \pi)$. In Figure 5.1, the insurance price line has the same slope as the iso-expected payment lines. The condition for the optimal purchase of insurance is, in this case, the equality of the marginal utilities of income in the two states of nature. This is because the individual chooses I (and therefore R) so as to maximize

$$E[u] = \pi \cdot v(M + R - I, A^*) + (1 - \pi) v(M - I, 0), \tag{5.12}$$

subject to the constraint that $I/(R - I) = k$. Substituting the constraint into the objective function and differentiating gives the first-order condition

$$\frac{\pi}{(1-\pi)} \frac{v_M^*}{v_M^0} = k. \tag{5.13}$$

If the insurance is fair, then $k = \pi(1 - \pi)$ and $v_M^* = v_M^0$.

Suppose now that the individual is required to make the payment represented by point D if the adverse event occurs and pays nothing otherwise. Paying D is equivalent to experiencing damages equal to D in this state of nature. Suppose also that the individual can purchase insurance at fair prices. The iso-expected payment line through point D shows that the individual could purchase an insurance contract that would require the individual to pay $E[D]$ if the adverse event did not occur and would reimburse him so that his net payment would be $E[D]$ if the adverse event did occur. Since this point lies inside the WTP locus, it results in a higher expected utility than could be realized in the absence of the policy that would reduce A^* to zero. This means that if the individual can also purchase insurance so as to adjust his state-dependent payments, he would be willing to pay more than D for the policy that reduces A^* to zero. By failing to take into account the individual's opportunities for diversifying risk through insurance, the expected damage measure of welfare change underestimates the true willingness to pay of the individual.

Now suppose that there is some institutional barrier to imposing a payment scheme that varies across states of nature, but that the individual can still purchase contingent claims at fair prices. The maximum state-independent payment that could be extracted from the individual ex ante is actually represented by point B in Figure 5.1. Since point B lies outside of the WTP locus, the individual would be left worse off with this payment scheme in the absence of fair insurance. But with fair insurance, the individual can move down to the right and reach the fair bet point through the optimal purchase of insurance. Thus, the availability of fair insurance makes the maximum expected payment of the fair bet point feasible even when varying payments across states of nature are not possible.

The existence of the WTP locus and points such as B (in Figure 5.1) that can be reached by purchasing contingent claims raises an important question. If welfare change is to be measured by the maximum payment that holds expected utility constant, which pair of state-dependent payments is the best welfare measure? We

are not yet ready to answer this question, because how welfare is to be measured depends in part on the form of the social welfare criterion. The Potential Pareto Improvement (PPI) form of social welfare criterion requires a comparison of the welfare changes of those who gain and those who lose by the project. When losers face uncertainty about costs, the potential for state-dependent payments and compensation to redistribute risks between gainers and losers must be taken into account, along with the ability of each group to insure itself through the purchase of contingent claims. A full consideration of these issues must be left to the next section. However, first we must show how uncertainty about costs can be represented in Graham's state-dependent payment model.

The Required Compensation Locus

The costs of government risk-management activities come ultimately in the form of reductions in the utilities of individuals. This is true whether the activity involves direct government spending or regulations imposing requirements on the private sector. Furthermore, these costs, as correctly measured, may be quite different from the direct expenditures of the government or regulated firms. For an example, see Hazilla and Kopp (1990). The measurement of these costs from an ex ante perspective entails first determining the reductions in expected utility for all of those who would bear the costs and then finding the minimum compensation required to restore each person to her original level of expected utility in the absence of the government activity.

Any individual who would bear part of the cost of a project has a reference level of expected utility in the absence of the project, say

$$E^2 = \pi \cdot v\left(M, A^*\right) + \left(1-\pi\right) v\left(M, 0\right). \tag{5.14}$$

The project would impose costs on the individual that could vary across states of nature. These costs could take several forms: a reduction in money income through taxation, the direct disutility of restrictions on nonmarket activities, or changes in product and factor prices.

If these costs vary across states, then compensation must be calculated so as to restore each cost-bearer's expected utility to the status quo level. Let c^* and c^0 represent all of the nonmonetary dimensions of costs and let M_c^* and M_c^0 represent money income inclusive of project-induced income changes. So that the problem can be analyzed graphically, suppose the probabilities affecting costs and benefits depend on the same random process in nature, so that π and $1 - \pi$ represent the relevant probabilities. This assumption will be relaxed later.

Given these assumptions, the project would result in a level of expected utility in the absence of compensation of

$$E^{2'} = \pi \cdot v\left(M_c^*, A^*, c^*\right) + \left(1-\pi\right) v\left(M_c^0, 0, c^0\right), \tag{5.15}$$

where M_c^* and M_c^0 represent net income and c^* and c^0 represent the direct negative effects of the policy on utility in each state. There is an infinite number

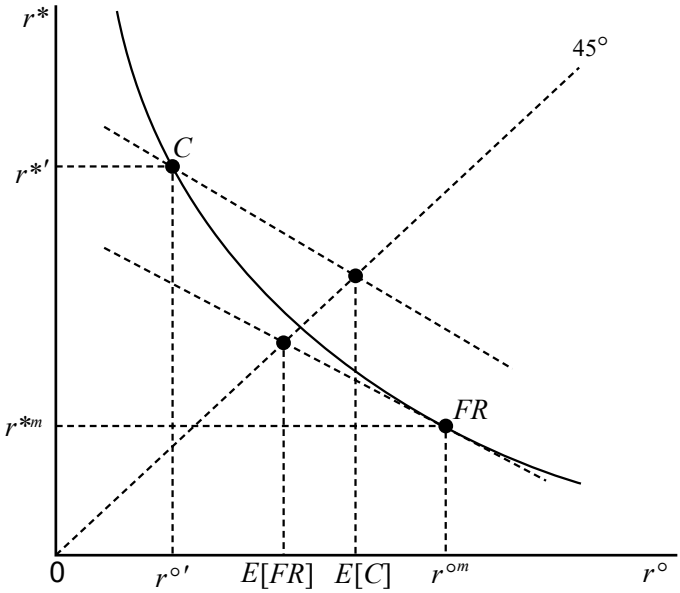

Figure 5.2 The required compensation locus

of pairs of state-dependent compensation payments, r^* and r^0, that can be made to restore expected utility to E^2. These pairs are solutions to

$$E^2 = \pi \cdot v\left(M_c^* + r^*, A^*, c^*\right) + (1-\pi) v\left(M_c^0 + r^0, 0, c^0\right). \tag{5.16}$$

The solution to this expression gives the required compensation (RC) locus for an individual who is harmed by the policy, shown in Figure 5.2. This is the locus of all possible compensating payments that leave those who bear the costs no worse off in terms of expected utility than if the policy had not been undertaken. The slope of the RC locus is found by setting the total differential of equation (5.16) equal to zero:

$$\frac{dr^*}{dr^0} = -\frac{(1-\pi)}{\pi} \frac{v_M^0}{v_M^*}. \tag{5.17}$$

Diminishing marginal utility of income assures that the locus is convex to the origin. The probabilities of the two states also define a family of iso-expected payment lines. They can be used to find the set of compensations that optimally distributes the risk to the cost-bearer across states—that is, that combination of payments that equates the marginal utilities of income in the two states. This is shown as $\left(r^{*m}, r^{0m}\right)$ at point *FR* in Figure 5.2. The intersection of the iso-expected payment line through this point with the 45° line gives the expected value of the fair compensation point, *E* [*FR*].

It is also possible to define an expected value of the required compensation from an ex post perspective. Given the occurrence of the event, there is an $r^{*\prime}$ that satisfies

$$v\left(M+r^{*'},A^{*},c^{*}\right)=v\left(M,A^{*}\right). \tag{5.18}$$

Similarly, $r^{0'}$ satisfies

$$v\left(M+r^{0'},0,c^{0}\right)=v\left(M,0\right). \tag{5.19}$$

These solutions define one point on the *RC* locus, labeled as *C* in Figure 5.2. Expected cost is

$$E[C]=\pi\cdot r^{*'}+(1-\pi)r^{0'}. \tag{5.20}$$

Finally, we can define a state-independent compensation analogous to *OP* (say, *OR*). Since it also holds expected utility constant, it is the solution to

$$E^{0}=\pi\cdot v\left(M+OR,A^{*},c^{*}\right)+(1-\pi)v\left(M+OR,0,c^{0}\right). \tag{5.21}$$

But, as will become clear in the next section, neither the expected value of cost nor *OR* will in general be useful as welfare measures for benefit-cost analysis.

Aggregation and the Welfare Criterion[2]

A major issue in the literature on benefit-cost analysis under uncertainty, at least since Weisbrod's classic article (1964), has been finding the appropriate measure of the welfare change for an individual who benefits from a project. The early option-value literature focused on comparing the expected value of consumer surplus with option price. Graham (1981) redirected the discussion by showing that there are an infinite number of alternative state-dependent payment vectors, and that the choice of a vector depends at least in part on the availability of contingent claims and individuals' opportunities to redistribute risk. Graham added a third candidate to the list of potential welfare measures, the expected value of the fair bet point; and he showed under what conditions the third candidate would be preferable to expected surplus and option price.

The literature since Graham has continued to focus on the question of the availability of contingent claims and whether risks are individual or collective. For examples of works dealing with this issue, see Mendelsohn and Strang (1984), Graham (1984), Cory and Saliba (1987), Colby and Cory (1989), and Smith (1990). For the most part this literature has ignored the cost side of the problem or treated it in a simplified fashion. In particular, most of this literature has neglected explicit treatment of uncertainty in costs, the possibility for changing the nature of risk through state-dependent compensation schemes, and the implications of the latter possibility for the form of the PPI criterion. An exception is the work of Meier and Randall (1991).

Also, for the most part this literature has dealt with only the simplest form of uncertainty (i.e., that where only two alternative states of nature are possible). Exceptions include Graham's 1981 and 1992 articles. The latter paper presented

2 This section is adapted, with permission, from Freeman (1991b).

a much more formal, abstract treatment of some of the problems considered here. The simple two-state model and its graphic treatment can be maintained only if both cost and benefit uncertainty result from the same random process and there are only two possible outcomes. If there are $\mathcal{N}$ possible benefit outcomes and $\mathcal{J}$ possible cost outcomes, the total number of alternative states of nature is $\mathcal{N} \times \mathcal{J}$.

In this section, alternative forms of the PPI criterion under uncertainty are discussed. A simple model with one gainer and one loser (so that the problems of aggregation and collective risk as opposed to individual risk are not important) is presented to show that none of the three candidates for welfare measurement is a reliable screen for potential Pareto improvements. This is because all three candidates ignore the opportunity for redistribution of risk between the gainer and the loser through state-dependent payments and compensation. The analysis is extended to a many-person economy. In those cases where the collective nature of risk precludes contingent claims markets, the importance of opportunities for redistributing risk between gainers and losers through state-dependent payment and compensation remains. Neither state-independent payments and compensation nor expected values of fair bet points are reliable indicators of potentially Pareto improving projects.

Potential Pareto Improvements under Uncertainty

Recall that the foundation of the PPI criterion is that a project passes only if there is some way to make redistributive payments so that no one is made worse off by the project and some people are made better off. It is generally agreed that in the context of risk and uncertainty "better off" and "worse off" are defined in terms of expected utility. However, there are several ways of formulating potential compensation tests in terms of expected utility. As shown above, there is an infinite number of potential payments and compensations for each individual that will leave expected utility unchanged. However, not all of these payment and compensation vectors are feasible in the aggregate, in the sense that whatever state of nature occurs, the potential payment equals or exceeds the required compensation. In other words, not all of the payment and compensation vectors pass an ex post balanced budget test. Here the PPI criterion is developed in a general form that is consistent with the balanced budget test. Then in the next subsection, the implications of this criterion for project selection and for welfare measurement in the simple two-state model are examined. The generalization to many states is straightforward. Finally, the aggregation of individual welfare measures across many gainers and losers is discussed.

A proposed policy can be deemed an improvement in PPI terms only if there is at least one set of state-dependent payments that will finance (in the sense of balancing the budget in all states of nature) a set of state-dependent compensations without making anyone worse off and leaving at least one person better off in expected utility terms. More formally,

let $i = 1, \ldots, \mathcal{N}$ index alternative outcomes of the random process affecting benefits;

let $j = 1, \ldots, \mathcal{J}$ index alternative outcomes of the random process affecting costs; and

if $\boldsymbol{T}$ and $\boldsymbol{R}$ represent points on the $\mathcal{N}$- and $\mathcal{J}$-dimensional WTP and RC loci respectively, a project passes the PPI criterion if (and only if) there is at least one $\boldsymbol{T}$ and $\boldsymbol{R}$ such that $t_i \geq r_j$ for all i, j with the strict inequality holding for at least one pair.

Graphically, a project represents a PPI only if the WTP locus in the $(\mathcal{N} \times \mathcal{J})$-dimensional space intersects the RC locus. Notice that since this expression of the PPI criterion compares t_i and r_j independently of the probabilities associated with alternative outcomes, expected values are irrelevant in screening for PPI projects. An equivalent procedure is to aggregate the positive and negative payments into a net WTP locus. If this locus has a positive segment in the $(\mathcal{N} \times \mathcal{J})$-dimensional space, then the project passes the PPI test.

This version of the PPI criterion can pass projects that fail more restrictive forms of the criterion. This is because the criterion takes into account the possible benefits of redistributing risks between gainers and losers through state-dependent payments and compensation (Cook and Graham 1977). With this form of the criterion, project evaluation requires consideration of both the real effects of the project and project financing, because it is the latter that determines the benefits, if any, of redistributing risks.

Consider a project that would affect only two people. One person would gain, at least in one state of nature, and the other would bear the costs. In this simplification, the problem of how to aggregate benefits and costs within the beneficiary and cost-bearer groups is avoided. Assume that it is not possible to buy contingent claims on income across the two states of nature that are relevant. Should the project be built? The answer depends on which version of the PPI compensation test one adopts.

One form of compensation test was offered by Bishop (1986), who called it an Ex Ante Compensation Test. For the two-person economy considered here, this test asks whether the option price of the gainer exceeds the required compensation of the loser in every state of nature. Ready (1988) also discussed this criterion, calling it a "weak" form of potential Pareto improvement. The implications of this form of compensation test can be seen in Figure 5.3. Point A on the WTP locus shows the option price of the gainer. The ex ante compensation, or weak PPI criterion, will pass only those projects whose costs lie below and to the left of point A. Points B and C represent projects with uncertain costs. Projects B and C would not pass the test because if the adverse event occurred, the person bearing the cost would require compensation that exceeded the amount collected in the form of option price.

Ready (1988) also proposed an alternative, which he called the "strong" potential Pareto criterion. A project passes the strong form of this criterion if

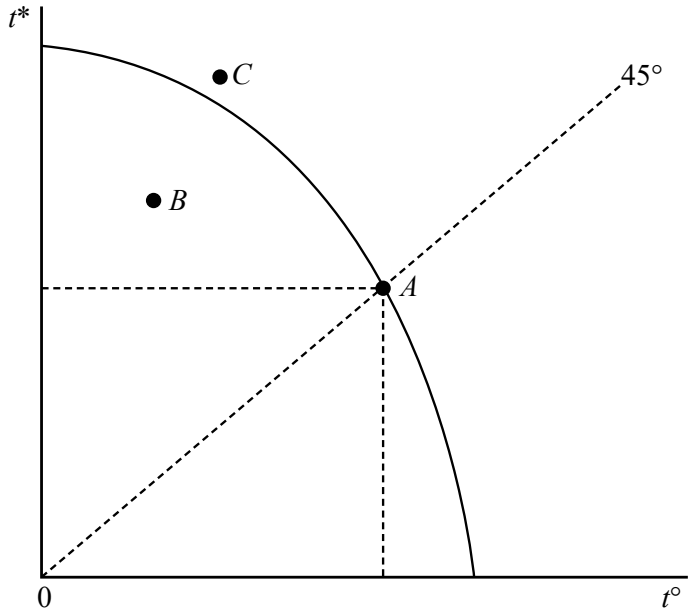

Figure 5.3 The willingness-to-pay locus and project costs

there exists some set of state-dependent payments that exceed the statedependent project costs in every state. The strong form of PPI criterion would pass a project with uncertain costs represented by point *B* in Figure 5.3, but it would reject a project whose costs were represented by point *C*. The strong form of the PPI criterion justifies the use of the expected value of the fair bet point as a benefit measure. If $E[FB]$ exceeds $E[FR]$, then the project passes.

Note that although both weak and strong forms of the criterion allow for cost uncertainty, they both treat costs as a point in the payment and compensation space. Both also ignore opportunities to redistribute risk among cost bearers or between gainers and losers through state-dependent payments and compensation.

Recall that according to the general form of the criterion stated above, a proposed policy is an improvement only if there is at least one set of balanced-budget, state-dependent payments that will finance a set of state-dependent compensations without making anyone worse off and leaving at least one person better off. In terms of Figure 5.3, the general form of PPI criterion would pass a project with costs represented by point *C* if the required compensation locus through point *C* intersected the WTP locus. As noted above, this form of the criterion takes account of both the real effects of the project and the redistribution of risks through project financing.

Ready (1993) has raised an interesting issue regarding the benefits of redistributing risk. The analysis here counted the benefits of redistributing risk whether or not the compensation is actually paid. Ready argued that they should

be counted only if compensation is actually paid, since they result from creating a more efficient distribution of risk, and this is qualitatively different from project evaluation under certainty where project finance and compensation only involve the redistribution of wealth. He showed that it would be possible for a project that passes this form of PPI test because of its redistribution of risk benefits to make all parties worse off if the compensation were not in fact paid.

Applying the Criterion in a Two-Person World

Again consider the simplest case of a project that would affect only two people and where it is not possible to buy contingent claims on income across the two states of nature. To provide a specific example, suppose that the project is to divert water from a river through an existing generator to produce hydroelectric power for a beneficiary. The cost would be borne by a farmer who would otherwise use the water for irrigation. In a rainy year, the benefit would be large because of the large flow through the generator. The cost to the farmer would be small because the rainfall would substitute for the irrigation water. Similarly, in a dry year the benefit would be small and the cost to the farmer large.

Should the project be built? Since the benefits and the costs are subject to the same random process, the WTP and RC loci can be combined in a single diagram, which is shown in Figure 5.4. Points *B* and *C* show the state-dependent

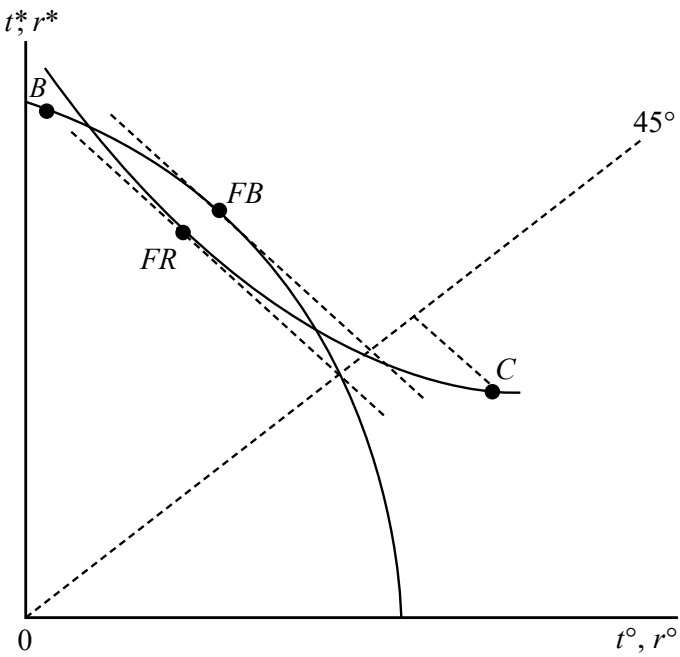

Figure 5.4 A potential Pareto improving project with expected costs exceeding expected benefits

benefits and costs, and the WTP and RC loci pass, respectively, through these two points. Another way of stating the PPI criterion is that it asks whether there is a set of state-dependent transfers from the beneficiary to the cost-bearer such that neither party is made worse off and at least one of the parties gains. Any state-dependent transfer is represented by one point in the WTP-RC space. Any such point on or below the WTP locus leaves the beneficiary no worse off; if the point is on or above the RC locus, the costbearer is no worse off. Thus, any set of transfers that lies in the lens-shaped area between the two loci will successfully finance this project, and therefore, the project passes the PPI test.

There are several important points to note about this conclusion. First, neither the expected value of cost nor the expected value of benefit is relevant. In the example of Figure 5.4, the expected value of cost substantially exceeds the expected value of benefit. But the project still passes the PPI test because there are feasible payment schemes that redistribute the risks associated with the states of nature in a way that can make both parties better off.

The irrelevance of expected benefit and expected cost holds even if benefits and costs are certain, as long as individuals are risk-averse. For example, suppose that points *C* and *B* in Figure 5.4 are both on the 45° line, with *C* above *B*. It is still possible that the aggregate WTP and RC loci through these points could intersect over some range. This demonstrates the importance of considering the manner in which the payment and compensation scheme redistributes risk in project evaluation. For example, suppose that both individuals faced substantial uncertainty about their incomes. The beneficiary would prefer a payment scheme with the larger payment in the state of nature in which income was high. The cost-bearer would prefer to receive more compensation in the low-income state. Project financing provides both individuals with a means of hedging the income uncertainty. The risk of the project also has to be evaluated in the context of both individuals' total risk portfolios.

The second point about the conclusion above is that option prices are not reliable indicators of PPI projects. Again, the example in Figure 5.4 shows that it is possible for the state-independent compensation to exceed the state-independent willingness to pay and yet have the project pass the PPI test because of the ability of state-dependent payments and compensation to redistribute risk in a more favorable manner.

The third point is that neither the expected value of the fair bet point nor the expected value of fair compensation is relevant to project evaluation. Although the potential Pareto improving project in Figure 5.4 has an expected value of the fair bet point exceeding the expected value of compensation, this condition is not sufficient to assure that the project is potentially Pareto improving. A case in which the expected value of the fair bet point exceeds the expected value of fair compensation is shown in Figure 5.5. Yet, because the WTP locus lies below the RC locus everywhere, there is no state-dependent transfer scheme that would make this project potentially Pareto improving. Although on average (that is, in an expected value sense) WTP exceeds RC, there are some states of nature in

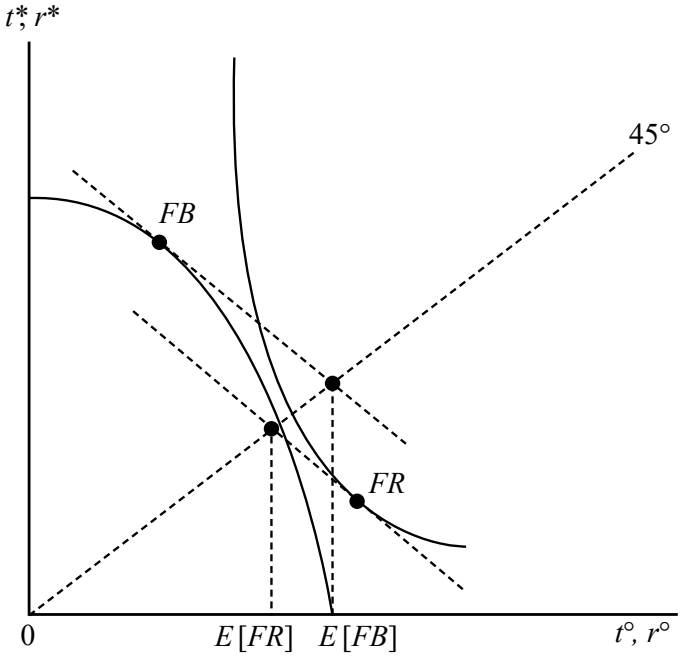

Figure 5.5 A nonpotential Pareto improving project

which the funds for the required compensation could not be collected from the beneficiary.

These results can be summarized in the form of three statements about necessary and sufficient conditions for identifying PPI projects.

1 The expected value of benefits being greater than the expected value of costs is neither necessary nor sufficient for a PPI project.
2 The option price of beneficiaries being greater than the option price of cost-bearers is sufficient but not necessary for a PPI project.
3 The expected value of the fair bet point exceeding the expected value of the fair compensation point is necessary but not sufficient to identify a PPI project.

Aggregation in the Many-Person Economy

In the many-person economy the principal issue to be addressed is how the WTP and RC loci of individuals should be aggregated for the purposes of social welfare analysis. The answer depends on the nature of the risks faced by gainers and losers—that is, whether risks are collective or individual in nature. Collective risk refers to the case in which, if the event affecting benefits and costs occurs, all of the potentially affected individuals experience the event. Individual risk refers to a case in which the risks facing different individuals are independent.

Consider the case of individual risk first. With individual risks, the analysis of aggregate benefits and costs is much simplified. Let one individual who would be a beneficiary have a probability of π^b of experiencing the event. Then if N is the number of potential beneficiaries, $\pi^b \cdot N$ individuals will experience the event with virtual certainty, with sufficiently large N. Similarly, $\pi^c \cdot K$ of the K bearers of the costs will experience the cost state associated with the event with virtual certainty, with sufficiently large K. Although individuals are uncertain as to their own outcomes, aggregate payments and compensation can be calculated with virtual certainty.

Assume for the moment that all people are identical. If those cost-bearers who experience the event are compensated by $r^{*\prime}$ and the rest of the cost-bearers are compensated by $r^{0\prime}$, then the aggregate compensation is

$$\pi^c \cdot K \cdot r^{*\prime} + \left(1-\pi^c\right)K \cdot r^{0\prime} = K\left[\pi^c \cdot r^{*\prime} + \left(1-\pi^c\right)r^{0\prime}\right]. \tag{5.22}$$

The aggregate compensation is minimized by finding the expected fair compensation for an individual and multiplying by K. The same procedure is used to find the maximum aggregate willingness to pay. If

$$N\left[\pi^b \cdot t^{*M} + \left(1-\pi^b\right)t^{0m}\right] > K\left[\pi^c \cdot r^{*\prime} + \left(1-\pi^c\right)r^{0\prime}\right], \tag{5.23}$$

then the project passes the PPI test. If individuals are different, all individuals' fair bets and compensations must be found and summed. In general, if each gainer makes the state-dependent payments represented by her fair bet point, the aggregate payment is the sum of the expected values of the fair bet points. Results are similar for the losers. Also, there is a virtual certainty that the gainers can compensate the losers and still be better off.

As long as a social welfare judgment is made not to require actual payment and compensation, this is all that needs to be said on the matter. However, if a social welfare judgment is made to require payment and compensation, then one must address the question of the feasibility of the required state-dependent payment and compensation schemes. If it should turn out that state-dependent payment and compensation is not feasible, then individual option prices are relevant, unless contingent claims could be purchased at actuarially fair prices. In the case of individual risks, it is more likely that such contingent claims will in fact be available.

In the case of collective risk, the full aggregate WTP and RC loci are required for welfare analysis. The procedure for aggregating individuals' WTP loci is explained in Graham (1981, 718–719). For each possible marginal rate of substitution (MRS) between contingent payments in different states as measured by dt^*/dt^0, each individual's payment vector is determined and all of these payment vectors are summed to obtain

$$T^{*m} = \sum_i^N t_i^{*m} \text{ and } T^{*0} = \sum_i^N t_i^{*0}, \tag{5.24}$$

where i indexes the N beneficiaries, and the superscript m indicates that all individuals' payments are at their fair bet points for a given common MRS. This procedure gives one point on the aggregate WTP locus for each MRS. Each point also represents an efficient distribution of risk within the group of

beneficiaries since all individuals have the same MRS for contingent payments. A similar procedure of aggregating individuals' required compensation at different marginal rates of substitution will yield the aggregate RC locus. Each point on the RC locus corresponds to an efficient distribution of cost risk among cost-bearers.

A project passes the potential Pareto improvement test if there exist payment and compensation points, $T^* = R^*$ and $T^0 = R^0$, that lie both on or above the aggregate RC locus and on or below the aggregate WTP locus. The graphic analysis is similar to that in the two-person world shown in Figures 5.4 and 5.5. Aggregate option price is sufficient but not necessary as a screen for PPI projects; and the aggregate fair bet points are necessary but not sufficient as indicators of PPI projects.

There are two exceptions to these conclusions. First, if contingent claims could be purchased at actuarially fair prices, the expected values of the fair bet and fair compensation points do provide a sufficient test for potential Pareto improvement. This is because individuals' actual payments and compensations can be made at the expected values of the fair bet and fair compensation points. Individuals can then adjust their own positions through contingent claims purchases to reach their individual WTP or RC locus at their fair bet point. However, the nature of collective risks is likely to preclude a market for contingent claims at actuarially fair prices.

Second, if a social welfare judgment is made to require compensation, and if state-dependent payments and compensations are not feasible, then as is the case in the two-person world, the welfare criterion must be based on option price measures. If individuals are identical, the appropriate test is based on the aggregate OP^B and OR^C because they are equal to the sums of the individual OPs and ORs. But if individuals are different, then the sums of the individuals' option price measures will be less than the aggregate measures. This is because the former do not provide for the efficient distribution of risk across individuals within each group. The appropriate welfare test becomes

$$\sum_{i}^{N} OP_i > \sum_{j}^{J} OR_j \ . \tag{5.25}$$

In this section, some of the consequences of extending the model of benefit-cost analysis for uncertain projects to reflect the symmetry of benefits and costs as changes in the expected utilities of affected individuals have been studied. One of the implications of this analysis is that there is much work still to be done on the development of measures of benefits and costs in order to implement the appropriate welfare criteria. Specifically, since presently available methods of benefit and cost estimation under uncertainty generally give only expected cost and benefit and option price measures, it is necessary to develop new methods for estimating the WTP and RC loci.

Uncertainty and Welfare in a Dynamic Setting

The derivation of option price and Graham's WTP locus assumes that individuals are faced with some type of uncertainty, but that they do not have the opportunity

to learn and reduce that level of uncertainty over time and then decide whether to purchase the good. Likewise, these models do not consider the possibility that individuals might have the opportunity to reverse their decisions if, after receiving more information, they decide that the good was not worth the price paid. However, learning and the ability to delay or reverse a decision once made, are part of many real world decisions with respect to both private goods and public goods. If through the acquisition of information, an individual can learn more about a good and reduce or eliminate the uncertainty associated with making a decision, then that individual might choose to delay the decision. The value of delaying a decision until additional information becomes available is the motivation for the concept of quasi-option value. Quasi-option value is a term coined by Arrow and Fisher (1974) to describe the welfare gain associated with delaying a decision when there is uncertainty about the payoffs of alternative choices, and when at least one of the choices involves an irreversible commitment of resources (or more generally, positive costs associated with reversal).

Much of the early literature concluded that consideration of quasi-option value would lead to relatively less irreversible development and relatively more preservation of natural environments (Arrow and Fisher 1974; Conrad 1980; Miller and Lad 1984; Fisher and Hanemann 1987; Mäler and Fisher 2005). In these models quasi-option value stems only from the value of the information gained by delaying an irreversible decision (such as developing a historically pristine natural area). But it is not difficult to imagine situations where the relevant information to guide future decisions can be gained only by undertaking at least a little development now. For example, suppose there is uncertainty about the magnitude of a mineral deposit underlying a wilderness area. Perhaps the only way that the uncertainty about the magnitude of the benefits of development relative to preservation can be resolved is through exploratory drilling—that is, through a little bit of development. In such cases there can be positive quasi-option value to development, or equivalently, a negative quasi-option value to preservation (Miller and Lad 1984; Freeman 1984a; Kolstad 1996). Thus, depending on the sources of learning and degrees of irreversibility, it can make sense to delay or hurry a project (Pindyck 2000, 2002; Balikcioglu, Fackler, and Pindyck 2011); it can also make sense to adjust the size of the project in response to the magnitude of the uncertainty (Zhao 2001, 2003).

While the quasi-option value literature suggests that there are times when the presence of uncertainty can mean that the timing or size of a project should be altered for efficiency decisions under uncertainty, Zhao and Kling (2004, 2009) demonstrated that these same factors can alter the magnitude of the willingness to pay or sell for a good at any given point of time. Rather than considering the effect of uncertainty and learning opportunities on the optimal timing of a decision, they considered the maximum price an individual would be willing to pay *now* when they know that by purchasing today they forgo the opportunity to learn and reduce the risk of making a "bad" purchase decision. An individual faced with uncertainty about the value of a good might still be willing to make a purchase if

the price is low enough to compensate for the loss of the opportunity to acquire more information and make the decision later. This intuition forms the basis for the concept of dynamic willingness-to-pay measures.

As an example, consider an individual who is considering purchasing an environmentally-friendly vehicle (perhaps a dual-fueled or electric car). Suppose that the individual is willing to pay a premium to be recognized as a green-conscious consumer, but only if the fuel and maintenance costs of the vehicle are not too high. Given the newness of the vehicle, the lifetime fuel and maintenance costs may be highly uncertain. This consumer faces a dynamic decision problem. She can decide to purchase the vehicle in the current period and thereby enjoy the benefits of being a green consumer now and for all future periods, or she can delay the decision and wait until more information is available on the variable costs of maintaining and running the vehicle. A delay would mean that the consumer does not get the benefits of being a green consumer in the current period. In forming her maximum willingness to pay for the green vehicle, the consumer will consider both the risk of making a decision she will later regret with the lost benefits of not obtaining the good for consumption this period. A low enough price in the current period will make the tradeoff acceptable.

Consider another example: that of greenhouse gas emissions and climate change. Much of the political debate about the need for addressing global warming relates to the uncertainties associated with the extent of climate change that is likely to occur under a range of human behaviors, and what the associated damages from those changes are likely to be. This uncertainty has led some to argue that society should delay investments in climate change mitigation until this uncertainty is resolved, or at least reduced, thereby lowering the risk of incurring costs now that later turn out to have been unnecessary. Implicit in this argument is that by waiting to obtain more information, the costs from investing in greenhouse gas reducing technologies can be avoided without any increased damages from climate change. Alternatively, a number of authors have argued that delay could ultimately lead to *higher* damages in the future. Invoking the concept of the "precautionary principle," they argue it may make sense to proceed more quickly to address climate concerns than if no uncertainty were present (an argument more akin to the early quasi-option value examples). Thus, depending on the source and magnitude of uncertainty one considers, climate change is a case in which quasi-option values may lead to either a delay, or a speeding up, of the decision to implement a policy of project. Either way, the maximum willingness to pay to reduce greenhouse gas emissions in the current period may be affected by the possibility of obtaining additional information about future damages.

To study the conditions under which dynamic welfare values will differ from their static counterparts (i.e., option prices), consider a simple model based on Zhao and Kling (2009) with three time periods (today and two future periods). The individual's per period utility function is $v(x, M, \theta)$, where x is a public good, M is a composite good (equivalent to income in this situation), and θ is a parameter that affects the value of an improvement in the public good from an initial level, x_0,

to an improved level, x_1. In the climate change case, θ might represent the degree of warming or variability in storm events and x might represent a flood prevention structure (which would prevent flood damages associated with higher rainfall and more severe storm events associated with climate change). Suppose that the project takes one period to build and that while the value of θ is uncertain today (period 1), its value will be known with complete certainty at the beginning of period 2 (imagine that the definitive climate change study will be completed and released precisely at the beginning of the period). For simplicity, assume that there are two possible outcomes for $\theta_i = (\theta_L, \theta_H)$. If θ_H is realized, then the flood prevention project would have been highly valuable, but if θ_L is realized then there is little or no value to the project. In this situation, how much would the consumer be willing to pay for the flood prevention project if she must commit today to an annualized payment in all future periods once the project is finished?

To answer this question, it will be instructive to consider two cases, one in which the consumer cannot take advantage of the opportunity to delay and learn (the no-learning case), and the other in which it is possible to delay and wait until the uncertainty is resolved before making a decision (the learning case). In the no-learning case, the most an individual would be willing to pay is the amount that equates the expected value of purchasing the good in the first period and having it available for consumption in the second and all future periods with the utility of not having the good in those periods, and paying nothing. This results in a per period payment (starting in period 2) of p_{nl}, where p_{nl} is defined implicitly by

$$\begin{aligned}&\tfrac{1}{r}\left[\pi v(x_0, M, \theta_H) + (1-\pi) v(x_0, M, \theta_L)\right] = \\ &\tfrac{1}{r}\left[\pi v(x_1, M - p_{nl}, \theta_H) + (1-\pi) v(x_1, M - p_{nl}, \theta_L)\right],\end{aligned} \tag{5.26}$$

and both sides are divided by r, the discount rate, to reflect the fact that this stream of utility occurs in perpetuity. The reader will quickly recognize that the WTP value, p_{nl}, is simply the option price. Thus, when delay and/or learning is not possible, there is an equivalence between the option price and the dynamic WTP.

To consider how the WTP might change when delay and learning is possible, it is useful to realize that by waiting, the individual can eliminate the downside risk of making the "wrong" decision (paying a positive price for x_1 in the event that θ_L is realized, or not having x_1 when θ_H is realized). For simplicity, assume that if $\theta_i = \theta_L$ there is no value from the project and therefore the ex post WTP would be zero. Likewise, assume that if $\theta_i = \theta_H$ there is a range of positive prices that would generate positive surplus.

The expected value of purchasing the good in the current period is the sum of the expected values of the current period (without the good) plus the expected value of the utility of receiving the good in the second and all future periods:

$$\begin{aligned}v_0(p) = {} & \pi v(x_0, M, \theta_H) + (1-\pi) v(x_0, M, \theta_L) + \\ & \tfrac{1}{r}\left[\pi v(x_1, M - p, \theta_H) + (1-\pi) v(x_1, M - p, \theta_L)\right].\end{aligned} \tag{5.27}$$

If instead of purchasing the good in the first period, the individual delays and waits to observe the realized value of θ_i, he will be able to purchase the increase in x only if its value is positive (i.e., if $\theta_i = \theta_H$). Essentially, he eliminates the second term in brackets in (5.27), which is the lower utility associated with paying a positive price x_1 when $\theta_i = \theta_L$, and is instead guaranteed that he will only purchase x_1 when it generates positive surplus. But, this means that he will not be able to enjoy x_1 in the second period if its value is high. The expected utility of delaying until the uncertainty is resolved can be written as

$$\begin{aligned} v_1(p) &= \left(1+\frac{1}{1+r}\right)\left[\pi v(x_0, M, \theta_H) + (1-\pi)v(x_0, M, \theta_L)\right] + \\ &\quad \left(\frac{1}{r(1+r)}\right)\left[\pi v(x_1, M-p, \theta_H) + (1-\pi)v(x_0, M, \theta_L)\right] \\ &= \left(1+\frac{1}{r}\right)\left[\pi v(x_0, M, \theta_H) + (1-\pi)v(x_0, M, \theta_L)\right] + \\ &\quad \left(\frac{1}{r(1+r)}\right)W(p) \end{aligned} \tag{5.28}$$

where $W(p) = \pi\left[v\left(x_1, M-p, \theta_H\right) - v\left(x_0, M, \theta_H\right)\right] > 0$ is the value associated with being able to wait and obtain the new information. The first line of the equation represents the expected utility from the first and second period when the project has not been built. The second line represents the expected utility from building the project only when it makes sense to build it (i.e., when $\theta_i = \theta_H$), and not building it when it has no value (i.e., when $\theta_i = \theta_L$). The second term in brackets incorporates the value of waiting $W(p) = \pi U(x_1, M-p, \theta_H) + (1-\pi)U(x_0, M, \theta_L)$, and it is enjoyed in perpetuity, which is why the term is divided by r in equation (5.28).

To determine the maximum WTP of an individual who has the option of waiting, solve for the price that equates the right-hand side of equations (5.27) and (5.28) which, after some rearranging, yields the expression

$$\begin{aligned} &\tfrac{1}{r}\left[\pi v(x_0, M, \theta_H) + (1-\pi)v(x_0, M, \theta_L)\right] = \\ &\tfrac{1}{r}\left[\pi v(x_1, M-p_l, \theta_H) + (1-\pi)v(x_1, M-p_l, \theta_L)\right] - \left(\frac{1}{r(1+r)}\right)W(p_l). \end{aligned} \tag{5.29}$$

The value of p_1 (the price when learning is an option) represents the WTP for the improvement in x that an individual would be willing to commit to in the current period when they know that by not doing so they can wait until the uncertainty is resolved and only purchase the good if it is valuable. The expression for p_1 as written in (5.29) has the same left-hand side expression as does the expression for p_{nl} in (5.26) making comparison straightforward. In forming the WTP when learning is not available (p_{nl}), the individual will set the expected utility of not purchasing the good (left-hand side of both expressions) with the expected utility of purchasing it as expressed in the right-hand side of (5.26). But, in forming the

dynamic WTP, the individual will equate the expected utility of not purchasing the good with the expected utility of waiting and deciding whether to purchase the good in the second period. The two components of the right-hand side of (5.29) represent this value. Since $W(p)$ is positive, the right-hand side of (5.29) is lower than the right-hand side of (5.26). This, in turn, implies that the solution for p_{nl} in (5.26) must be greater than p_1 in (5.29); that is, $p_{nl} > p_1$. By committing to purchase the good now, a consumer gives up the option to learn and delay until better information is available. To be induced to do so, the consumer will need to be "compensated" by paying a lower price today than otherwise. It is important to emphasize that the lower price refers to the price the consumer would be willing to commit to today, but could be either higher or lower in the second period since it will be conditional on the realized value of θ.

For simplicity, the expressions derived in (5.26) to (5.29) were restricted to state-independent payments, but as Zhao and Kling (2009) showed, this can be generalized to a set of state-dependent payments so that a full locus of payments can be derived just as in Graham's locus. They demonstrated that in general, the dynamic WTP locus will lie below the static (no-learning) locus of Graham, except for sharing a common value at the "certainty" point.

There are a number of implications of understanding that welfare measures may be dynamic and conditional on expectations about being able to delay a decision until more information is available. These relate to both the use of welfare measures in benefit-cost analysis and the interpretation of those measures, particularly those generated by stated preference methods. The appropriate welfare measure for use in deciding whether a project should be built when learning and delay are possible is the dynamic WTP measure. However, if the project is not built it will be necessary to reconsider the questions when additional information has become available and has reduced the uncertainty. Just because a project does not pass a benefit-cost analysis with the current information set does not mean that it will not necessarily pass such a test in the future. In essence, benefit-cost analysis needs to become a dynamic process in this situation, where the decision-maker may need to revisit the decision once increased information becomes available over time. On a related point, it is important to recognize that dynamic WTP measures incorporate considerations about delay and uncertainty, and it would be a form of double counting if quasi-option values were calculated in addition to using dynamic WTPs in evaluating the efficiency of a project.

It is also important for analysts to understand whether the value they have elicited from a welfare measurement study contains considerations of delay and learning so that it can be interpreted and used appropriately. For example, if an analyst designs a stated preference survey to elicit respondents' WTP for a dam project that could be delayed until additional information is available concerning environmental effects of the project and the analyst has carefully communicated this information to respondents, then a comparison between the costs and estimated benefits should yield the correct efficiency decision. However, if respondents answer the question implicitly assuming that the dam can be built in future years once more

information is available, when in fact the opportunity to purchase the land for the dam is possible only in the near term, then the analyst might end up comparing a welfare measure that is based incorrectly on the assumption that delay and learning is possible. Such a measure would be an underestimate of the true WTP.

Zhao and Kling (2001) also argued that one source of the often-observed disparity between WTP and WTA measures of value can be explained by the dynamic nature of these values. Specifically, suppose consumers perceive that the loss of an environmental good (e.g., a pristine wilderness area or large tract of open space) is difficult to reverse, but that a decision to allow development is always possible in the future. This may lead to large values of WTA because, in this case, consumers would need an extra large payment to be compensated for the loss of the chance to learn more about the value of the pristine wilderness and allow development. More discussion of the disparity between WTP and WTA and possible explanations for it appears in Chapter 3.

Revealed Preference Methods for Measuring Values

Stated preference methods will often be the method of choice for estimating ex ante WTP or WTA values under uncertainty, but there are some cases in which revealed preference methods may prove useful (e.g., Desvousges, Smith, and Fisher 1987; Cameron and Englin 1997; Cameron 2005; Nguyen et al. 2007). Specifically, when individuals have opportunities to adjust to risky positions through transactions in related private goods markets, it may be possible to use one of the revealed preference methods described in Chapter 4 to infer individuals' values for risk changes. In the situation where learning and delay are possible, the values that are estimated will correspond to the dynamic measures described in the previous section. Otherwise, the values will correspond to conventional option prices.

In this section, models of averting behavior and hedonic prices that can be used to obtain measures of value based on observable behavior are described. The models developed in this section are initially based on the hypothesis of expected utility maximization. Then the models are extended to take account of other assumptions about individuals' preferences. Since most observable transactions involve ex ante state-independent payments, the models described here are useful only in estimating option price measures of value.

Expressions for Option Price and Marginal Willingness to Pay

Suppose public policies are under consideration that will reduce either the magnitude of an adverse event or its probability. For simplicity, assume that learning and delay are not possible so that the policymaker is interested in the option price for a policy that reduces the magnitude of an adverse event—that is, the set of state-independent payments that results in the same level of expected utility as would have occurred in the original uncertain situation. Denote this payment as OP^A to indicate that it refers to a risk-reduction policy. Making use of

the indirect utility function, OP^A for a policy that reduces A from A^* to zero is the solution to

$$\begin{aligned}&\pi\cdot v\left(M,A^*\right)+\left(1-\pi\right)v\left(M,0\right)\\&=\pi\cdot v\left(M-OP^A,A^*\right)+\ 1-\pi\cdot v\left(M-OP^A,0\right)\\&=v\left(M-OP^A,0\right).\end{aligned}\tag{5.30}$$

Similarly, the value of reducing A from A^* to $A' > 0$ is the solution to

$$\begin{aligned}&\pi\cdot v\left(M,A^*\right)+\left(1-\pi\right)v\left(M,0\right)\\&\quad=\pi\cdot v\left(M-OP^{A'},A'\right)+\left(1-\pi\right)v\left(M-OP^{A'},0\right).\end{aligned}\tag{5.31}$$

For some purposes, it will be more useful to deal with marginal values. The option price for a marginal reduction from A^* is the marginal change in income that holds expected utility constant. This can be found by taking the total differential of the expression for expected utility in its indirect utility function form, setting it equal to zero, letting $dw = 0$, letting $dA = 0$ for $A = 0$, and rearranging terms. The individual's marginal WTP ex ante is given by

$$w^{A*}=\frac{dM}{dA^*}=-\frac{\pi\cdot v_A^*}{\left[\pi\cdot v_M^*+\left(1-\pi\right)v_M^0\right]},\tag{5.32}$$

where * and ^{0} indicate the values of A (i.e., A^* versus $A^0=0$) at which the partial derivatives are evaluated. Recalling that $v_A < 0$, this expression is positive for reductions in A^* ($dA^* < 0$), indicating a positive willingness to pay to reduce the magnitude of the uncertain event.

Equation (5.32) shows a variant of a standard result from welfare theory. The marginal willingness to pay for a change in A^* is equal to the marginal disutility of A^* converted to a money measure by using the marginal utility of income. But since this is an ex ante willingness to pay, the marginal utility terms are the expected values of the relevant marginal utilities—that is, the weighted averages of the marginal utility terms in the two states of nature where the weights are the probabilities of the two states.

The option price for reducing the probability of the event to zero OP_π^0 is found by solving the following expression for

$$\begin{aligned}&\pi\cdot v\left(M,A^*\right)+\left(1-\pi\right)v\left(M,0\right)\\&\quad=0\cdot v\left(M-OP_\pi^0,A^*\right)+\ 1\cdot v\left(M-OP_\pi^0,0\right)\\&\quad=v\left(M-OP_\pi^0,0\right).\end{aligned}\tag{5.33}$$

Comparison of this expression with equation (5.30) shows that the option price for reducing the probability of an event to zero is equal to the option price for reducing its severity or magnitude to zero. Thus, in this case there is no difference

between the value of risk reduction and the value of risk prevention, since in both cases the policy being valued eliminates the risk entirely.

The option price for a reduction in the probability of the event from π to π' is found by solving the following expression for OP'_{π}:

$$\begin{aligned}&\pi \cdot v\left(M, A^*\right) + (1-\pi) v(M, 0) \\ &\quad = \pi' \cdot v\left(M - OP'_{\pi}, A^*\right) + \left(1 \cdot \pi'\right) v\left(M - OP'_{\pi}, 0\right).\end{aligned} \tag{5.34}$$

The marginal value for a change in π can be derived by taking the total differential of equation (5.34), setting it equal to zero, and holding dA^* at zero. The result is a standard result in the analysis of the value of risk prevention (Jones-Lee 1974; Cook and Graham 1977; Machina 1983; Smith and Desvousges 1988):

$$w^{\pi} = \frac{dM}{d\pi} = \frac{v(M, 0) - v\left(M, A^*\right)}{\left[\pi \cdot v_M^* + (1-\pi) v_M^0\right]}, \tag{5.35}$$

where w^{π} is the WTP ex ante for a change in the probability of A^*. This expression is positive, indicating a positive willingness to pay for reductions in π ($d\pi < 0$).

A comparison of equations (5.35) and (5.32) is instructive. Whereas equation (5.32) is based on the marginal disutility, as given by the term v_A^*, equation (5.35) is based on the nonmarginal difference in utility between the two states of nature, converted to monetary units by a weighted average of the marginal utilities of income in the two states of nature.

Averting Behavior

Observations of an individual's averting behavior might provide a basis for inferring the values of changes in risk. Suppose that, ex ante, an individual can select a level of private spending R, that will reduce the magnitude of A^* given that the event occurs according to the relationship $A^* = A(R, G)$, where G is the level of government protective spending. An example would be purchasing and wearing a seat belt to reduce the severity of injury given the occurrence of an accident. Assume that this function has the following properties:

$$A(0,0) = A^* \quad \text{and} \quad A_R^* < 0 \quad \text{and} \quad A_G^* < 0.$$

The individual chooses R, given G so as to maximize expected utility:

$$E[u] = \pi \cdot v\left[M - R, A(R, G)\right] + (1-\pi) v[M - R, 0]. \tag{5.36}$$

The first-order conditions include

$$\frac{1}{A_R^*} = \frac{\pi \cdot v_A}{\pi \cdot v_{M^*} + (1-\pi) v_{M^\circ}}, \tag{5.37}$$

where v_{M^*} is the marginal utility of income evaluated at the level of A associated with the given level of G. The term $\frac{1}{A_R^*}$ is the reciprocal of the marginal productivity of

expenditure on risk reduction, or equivalently, the marginal private cost of reducing A^*. The right-hand side is the marginal value of reducing A^* that was derived above.

The relevant value for policymakers is the value to the individual of an increase in public spending on risk reduction, dG. To find the option price measure, take the total differential of equation (5.36), set it equal to zero, and substitute the first-order condition for the choice of private protective spending into this expression. After some simplification, we have

$$\frac{dM}{dG} = -\frac{A_G^*}{A_R^*} = \frac{\partial R}{\partial G}. \tag{5.38}$$

This means that the individual's marginal willingness to pay for a small increase in government spending is the ratio of the marginal productivities of private spending and public spending in reducing A^*, or the marginal rate of technical substitution between R and G in reducing A^*. This measure can be calculated if the technical relationship $A(R, G)$ is known. The relationship is observable in principle. The welfare change is also given by the marginal rate of substitution between private and public spending, holding expected utility constant. This is not the same thing as the observed change in private spending. For example, if G increases, the individual will reduce R but will also attain a higher level of expected utility. If R enters the utility function directly, then the welfare measure will include unobservable marginal utility terms, and the measure derived here will be an underestimate (overestimate) if R provides positive (negative) utility.

Similar results can be obtained for the case where individual ex ante spending has the effect of reducing the probability of the adverse event. Now let the production function relating private and public expenditures to the probability of the adverse event be

$$\pi = \pi(R,G), \tag{5.39}$$

where $\pi(0, 0) = \pi^*$ and $\pi_R < 0$, $\pi_G < 0$. The individual chooses R so as to maximize expected utility given by

$$E[u] = \pi(R,G)v(M-R,A^*) + [1-\pi(R,G)]v(M-R,0). \tag{5.40}$$

The first-order conditions include

$$\frac{1}{\pi_R} = -\frac{v(M-R,0) - v(M-R,A^*)}{\pi(R,G)v_M^* + [1-\pi(R,G)]v_M^0}. \tag{5.41}$$

Again, the left-hand side is the reciprocal of the marginal productivity of private expenditure on reducing the probability, or, equivalently, the marginal private cost of reducing π. The right-hand side is the marginal value of reducing π as given by equation (5.35).

The marginal value of an increase in public spending to reduce the probability of the event is found by totally differentiating equation (5.40), setting the result equal to zero, and substituting the first-order condition where appropriate. The result is

$$\frac{dM}{dG} = -\frac{\pi_G}{\pi_R} = \frac{\partial R}{\partial G}. \tag{5.42}$$

This result is similar to the case of public spending to reduce A^*. The individual's marginal willingness to pay for public spending is equal to the ratio of the marginal productivities of private and public spending to reduce π, or to the marginal rate of technical substitution between R and G, holding A^* constant. Again, this result is analogous to those derived in the existing literature on protective spending in the absence of uncertainty.

Unfortunately, these results do not carry over to the case where the averting activity jointly produces reductions in π and A^*. Repeating the steps described above but making both A^* and π functions of R and G leads to the following expression (Shogren and Crocker 1991):

$$\frac{dM}{dG} = -\frac{\pi \cdot v_A^* \cdot A_G + \pi_G \cdot \left(v^* - v^0\right)}{\pi \cdot v_A^* \cdot A_R + \pi_R \cdot \left(v^* - v^0\right)}. \tag{5.43}$$

The unobservable utility terms do not cancel out of this expression, so marginal willingness to pay cannot be inferred from information on the averting technology. The inability to use the averting behavior model in this case is due to the jointness of the implicit production technology. It is not a consequence of introducing risk into the analysis.

Hedonic Prices

If either the probability or the magnitude of a risk (or both) is a characteristic of heterogeneous goods such as housing, hedonic price estimation can be used to obtain the relevant ex ante marginal values for risk changes. Suppose that the magnitude of the adverse event varies across the space used for residential housing. For example, the dose of a toxic chemical from an accidental release would depend on the distance from the source of the release. If people are aware of this spatial variation, then they should be willing to pay more for houses in those areas with lower-magnitude risks. Competition for these more attractive houses would result in a systematic inverse relationship between the price of housing, P_h and A_i^*, where i indexes the spatial location of the house. For simplicity, suppose that the magnitude of the event is the only relevant characteristic of housing. Then the price of a house at location i can be found from the hedonic price function $P_h\left(A_i^*\right)$. Given income, the probability of the event, and the magnitude of the event, the individual chooses a location so as to maximize expected utility:

$$E[u] = \pi \cdot v\left[M - P_h\left(A_i^*\right), A_i^*\right] + (1-\pi)v\left[M - P_h\left(A_i^*\right), 0\right]. \tag{5.44}$$

The first-order condition is

$$-\frac{\partial P_h}{\partial A_i^*} = \frac{\pi \cdot v_{A_i}^*}{\pi \cdot v_M^* + (1-\pi)v_M^0}. \tag{5.45}$$

Because the righthand side of this condition is the ex ante marginal value of a reduction in A_i^* (see equation (5.32) again), this condition says that expected utility maximization calls for setting the marginal value of risk reduction equal to its marginal implicit price, the slope of the hedonic price function. Thus, if individuals and the housing market are in equilibrium, the estimated marginal implicit price of risk reduction for each individual reveals each individual's marginal ex ante valuation for risk reduction. However, since a house is a longlived asset, and P_h is an asset price, equation (5.45) yields a compensating wealth measure of the lifetime welfare change associated with a permanent change in π.

If the relevant housing characteristic that varies across space is the probability of the adverse event, and housing prices reflect differences in π_i, the results are similar. The expression for expected utility is

$$E[u] = \pi_i \cdot v\left(M - P_h(\pi_i), A^*\right) + (1-\pi_i)v\left(M - P_h(\pi_i), 0\right) \tag{5.46}$$

and the first-order condition is

$$\frac{\partial P_h}{\partial \pi_i} = -\frac{v\left[M - P_h(\pi_i), 0\right] - v\left[M - P_h(\pi_i), A^*\right]}{\pi_i \cdot v_M^* + (1-\pi_i)v_M^0}. \tag{5.47}$$

Again, the right-hand side is the ex ante marginal value of the probability change in equation (5.35). Thus, the observed implicit price of probability reduction also reveals the individual's marginal ex ante value of risk reduction (Smith 1985). If both the probability and the magnitude of the event vary independently across space, housing prices will be a function of both characteristics. Both equations (5.45) and (5.47) must be satisfied in equilibrium. Hedonic price functions that do not include both characteristics as explanatory variables will be misspecified.

Welfare Change with Non-Expected Utility Preferences[3]

The models described above for revealed preference benefit measurement are based on expected utility as a representation of individuals' preferences under uncertainty. As mentioned above, there is quite a bit of empirical evidence against expected utility maximization as a description of behavior. An important question, therefore, is whether models of value of the sort described here can be modified for use with nonexpected utility preferences.

Revising the models in this way is straightforward and involves no additional complications, at least in certain circumstances. This follows from the key features of revealed preference methods for estimating individuals' values from data on behavior. Generally speaking, the models involve first deriving the expression for welfare change, finding the first-order conditions for optimization, and substituting them into the expression for welfare change. Given the assumptions of the models described here, the substitution allows for the canceling out of any observable

3 This section is adapted, with permission, from Freeman (1991a).

utility terms. It turns out that at least for many nonexpected utility representations of preferences, the same result occurs, so that the derived observable welfare measures are independent of the particular form of preferences. This is a straightforward consequence of the envelope theorem. In what follows, this is demonstrated first for the general case and then for two specific forms of nonexpected utility preferences.

Let I be some general index of preferences where the preferences depend on income, prices (implicitly), the probabilities of different states of nature and the magnitudes of the adverse event in different states. Thus,

$$I = f\left(M, A, \pi\right). \tag{5.48}$$

This function is assumed to be convex and twice differentiable. This expression could be nonlinear in the probabilities or incorporate regret and rejoice terms or other deviations from the standard expected utility function, or both. Expected utility preferences also fit this general formulation.

Consider the averting behavior model where $A^* = A(R, G)$. The first-order condition for the optimum R is

$$\frac{\partial I}{\partial R} = -f_{M^*} + f_{A^*} \cdot A_R^* = 0 \tag{5.49}$$

or

$$f_M = f_{A^*} \cdot A_R^*. \tag{5.50}$$

To find the marginal welfare measure for a policy that reduces A^*, totally differentiate equation (5.48), rearrange terms, and substitute in the first-order condition to obtain

$$dI = f_{M^*} \cdot dM + \left(f_{A^*} \cdot A_R^* - f_M\right) dR + f_{A^*} \cdot A_G^* \cdot dG = 0 \tag{5.51}$$

$$\frac{dM}{dG} = -\frac{f_{A^*} \cdot A_G^*}{f_{M^*}} = -\frac{A_G^*}{A_R^*}. \tag{5.52}$$

Thus, the marginal willingness to pay for publicly supplied risk reduction is equal to the marginal rate of technical substitution between public and private risk reduction.

Suppose that in the hedonic model it is probabilities that vary across space. Then the general index of preferences would be

$$I = f\left[M - P_h\left(\pi_i\right), A, \pi_i\right]. \tag{5.53}$$

The first-order condition for the selection of the risk characteristic of housing is

$$\frac{\partial I}{\partial \pi_i} = -f_M \cdot \frac{\partial P_H}{\partial \pi_i} + f_{\pi_i} = 0 \tag{5.54}$$

or

$$f_{\pi_i} = f_M \cdot \frac{\partial P_h}{\partial \pi_i}. \tag{5.55}$$

Totally differentiating equation (5.53) to obtain the welfare measure for the change in π and substituting the first-order condition gives

$$dI = f_M \cdot dM + \left(f_{\pi_i} - f_M \cdot \frac{\partial P_h}{\partial \pi_i} \right) d\pi_i + f_{\pi_i} \cdot dG = 0 \tag{5.56}$$

$$\frac{dM}{dG} = -\frac{f_{\pi_i}}{f_M} = \frac{\partial P_h}{\partial \pi_i}. \tag{5.57}$$

The marginal willingness to pay for publicly supplied risk prevention is equal to the observable marginal implicit price of the risk characteristic of housing.

To illustrate this general result, consider the prospect theory model of Kahneman and Tversky (1979). In the two-state model based on the indirect utility function, the index of preferences takes the following form:

$$I = g(\pi) \cdot v(M, A^*) + g(1-\pi) \cdot v(M, 0), \tag{5.58}$$

where

$$g(0) = 0, \;\; g(1) = 1 \tag{5.59}$$

and

$$g(\pi) + g(1-\pi) < 1 \text{ for } 0 < \pi < 1. \tag{5.60}$$

Assume that the magnitude of the adverse event depends on both the level of expenditure on a private averting activity R and public expenditure G. The value of reducing A^* is

$$\frac{dM^*}{dA^*} = \frac{g(\pi) v_A^*}{g(\pi) v_M^* + g(1-\pi) v_M^0}, \tag{5.61}$$

which is not directly observable because of the utility and probability weighting terms, but it can be inferred. Given the level of G, the individual's optimal level of the private averting activity is given by

$$\frac{\partial I}{\partial R} = -g(\pi) v_M^* + g(\pi) v_A^* A_R^* - g(1-\pi) v_M^0 = 0. \tag{5.62}$$

Thus,

$$\frac{1}{A_R^*} = \frac{g(\pi)}{g(\pi) v_{M^*} + g(1-\pi) v_{M^\circ}} \cdot v_{A^*}. \tag{5.63}$$

The marginal value to the individual of a change in G is found by totally differentiating equation (5.51), setting it equal to zero, and solving for

$$\frac{dM}{dG} = \frac{dR}{dG} - \frac{g(\pi)}{g(\pi) v_M^* + g(1-\pi) \cdot v_M^0} \cdot v_A^* \cdot A_R^* \frac{dR}{dG} - \frac{g(\pi)}{g(\pi) \cdot v_M^* + g(1-\pi) \cdot v_M^0} \cdot v_A^* \cdot A_G. \tag{5.64}$$

After substituting in equation (5.63), this becomes

$$\frac{dM}{dG} = -\frac{A_G^*}{A_R^*}. \tag{5.65}$$

Similarly, if it is π that can be reduced by private and public expenditure, the value of a reduction in π is

$$\frac{dM}{d\pi} = \frac{g(1-\pi)\cdot v(M-R,0) - g(\pi)\cdot v(M-R,A^*)}{g(\pi)\cdot v_M^* + g(1-\pi)\cdot v_M^0}. \tag{5.66}$$

The first-order condition for private averting expenditure is

$$-\frac{1}{\pi_R} = \frac{v(M-R,0) - v(M-R,A^*)}{g(\pi)\cdot v_M^* + g(1-\pi)\cdot v_M^0}. \tag{5.67}$$

After substitution, the value of the public risk-prevention expenditure is

$$\frac{dM}{dG} = -\frac{\pi_G}{\pi_R}. \tag{5.68}$$

Similar results can be derived for other forms of preferences, for example the regret theory of Loomes and Sugden (1982). See Jindapon and Shaw (2008) for a derivation of a complete WTP locus in a model based on rank-dependent expected utility (Quiggin 1982).

Extending these results to the case of weak complementarity is straightforward. Suppose that the consumption of some market good x_i increases the probability of occurrence of some adverse event for the purchaser. Suppose further that there is some public policy action represented by G that can reduce the risk associated with consuming x_i for all consumers. So

$$\pi = \pi(x_i, G), \tag{5.69}$$

with

$$\partial\pi/\partial x_i > 0$$

and

$$\partial\pi/\partial G < 0.$$

This policy will increase the general preference index for all consumers of x_i and will cause the demand curve for x_i to shift outward. Formally, the general preference index can be written as

$$I = f(M, A, \pi, p_i), \tag{5.70}$$

where p_i is the price of the complementary good. The general form of the expenditure function is

$$e = g(p_i, A, \pi, I^*). \tag{5.71}$$

A sufficient condition for weak complementarity requires that there be a choke price for x_i, and that at the choke price

$$\frac{\partial e}{\partial \pi} = 0 . \tag{5.72}$$

As shown in Chapter 4, when the conditions for weak complementarity are satisfied, the monetary equivalent of the increase in well-being is the area between the compensated demand curves for the good before and after the public policy change. Similarly, if the market good affects the severity of the adverse event, weak complementarity requires that $\partial e/\partial A^*$ or $\partial I/\partial A^*$ be zero when the good is not purchased. Since these results are independent of any particular specification of the preference function (other than the conditions of weak complementarity), they will hold for expected utility and nonexpected utility.

The three broad classes of models that have been developed to measure the benefits of environmental change from revealed preferences under certainty can be easily generalized to apply to valuing changes in risk. This generalization of the models does not require that individual preferences take the expected utility form. The principal requirement is that individuals be maximizing some objective function. Then by the envelope theorem, welfare measures that contain unobservable preference terms can be reduced to functions of observable relationships by substitution of the first-order conditions for preference maximization. Thus, if the conditions for utilizing these models are satisfied, there is no particular need to be concerned with how people make their choices under uncertainty.

Option Price, Option Value, and Expected Damages

There is a long history of using ex post expected damage measures in benefit-cost analysis. Perhaps the earliest example of this approach to measuring the value of risk changes is the U.S. Army Corps of Engineers' method for estimating the benefits of flood control projects. For many years, the Corps calculated the reduction in the costs of replacement, repair, and cleaning up after a flood for each possible flood stage, multiplied the monetary damages by the probability associated with that level of flooding, and summed across all possible flood stages (Eckstein 1961). Of course, we would now add to the repair and cleaning costs some monetary measure of the loss of utility associated with the flooding events.

Expected damage measures have the virtue of being relatively easy to calculate from experience with risky events. For example, for most river basins in the United States there is a large body of data on hydrology, land use, and the distribution of structures that can be used to calculate expected flood damages under alternative proposed flood control projects. But even if these losses (including lost utility) were accurately measured, as shown in earlier sections, they will not in general correctly measure welfare change from the ex ante perspective.

Since at least in some circumstances expected damage measures may be relatively easy to obtain, an important question is whether they can be taken as

useful approximations of the other forms of ex ante values when policymakers would prefer to have the latter. Are the differences between the two types of measure likely to be small or large? Is the expected damage measure likely to be larger or smaller than the option price measure? What factors determine the sign of the difference and its magnitude?

The derivation of an expected damage measure for risk reduction is straightforward. The first task is to establish a measure of the value of avoiding an event given that an event will occur with certainty. The willingness to pay to avoid the consequences of the event given its occurrence is a *CS* measure of damage, *D*, and is found as the solution to

$$v\left(M, A^*\right) = v(M - D, 0). \tag{5.73}$$

The expected damage measure of the value of avoiding the event that has probability π of occurring is the mathematical expectation of the willingness to pay given by equation (5.73); that is, $\pi \cdot D$. The marginal value of reducing A^* given that the event has occurred can be found by totally differentiating the left-hand side of (5.73) setting it equal to zero, and solving for

$$\frac{dM}{dA^*} = -\frac{v_A^*}{v_M^*}. \tag{5.74}$$

Let this be denoted by d^A to indicate that it is a marginal value and that it refers to changes in *A*. Equation (5.74) is a standard result showing that, ex post, the marginal value is the marginal utility of A^* converted to a money measure by using the marginal utility of income. Taking the expectation yields the ex post value of a marginal reduction in A^*:

$$E\left[d^A\right] = -\pi \frac{v_{A^*}}{v_{M^*}}. \tag{5.75}$$

The percentage difference between the option price and expected damage measures is defined according to the following expression, using equations (5.32) and (5.75):

$$\%\text{ difference} \equiv 100 \cdot \frac{w^* - E\left[d^A\right]}{w^*} = 100 \cdot \left\{(1-\pi) - \left[(1-\pi)\ \frac{v_M^0}{v_M^*}\right]\right\}. \tag{5.76}$$

This expression shows that if the marginal utility of income is independent of the event, so that $v_M^0 = v_M^*$, the difference in the measures is zero. This would be the case, for example, if the individual were risk-neutral or could purchase actuarially fair insurance.

If the adverse event reduces the marginal utility of income so that v_M^* is less than v_M^0, then the expected damage measure will exceed the option price measure and the difference will be negative. On the other hand, if the adverse event increases the marginal utility of income, the option price measure will be larger than the expected damage measure. In both cases the magnitude of the difference will be large when the difference in the marginal utilities of income is large and when the

probability of the adverse event is low. To summarize, the sign of the difference between the option price and expected damage measures depends on the way in which the marginal utility of income varies with A; and the magnitude of the difference depends on how much it varies and on the probability of the event.

Freeman (1989) computed these differences for several explicit forms of von Neuman–Morgenstern cardinal utility functions with different degrees of risk aversion. Examples included

$$v = (M - A^*)^b \text{ with } 0 < b < 1 \tag{5.77}$$

$$v = -e^{-b(M - A^*)} \text{ with } b > 0 \tag{5.78}$$

$$v = -e^{-bM(1 - A^*)} \tag{5.79}$$

$$v = \ln[(1 - A^*)M] \ . \tag{5.80}$$

For equations (5.77) and (5.78), the price flexibility of income for A^* is zero, meaning that the marginal value of a change in A is independent of M. Risks of this sort are essentially equivalent to financial risks. An arbitrary choice of units for measuring A^* can be made such that A^* is equal to the ex post D. For some of these functional forms, the marginal utility of income is higher given the adverse event. One way of interpreting this last characteristic is that income and the absence of A^* are substitutes in consumption such that if A^* does not occur, the marginal utility of the consumption that M allows is diminished. This might be the case if the event increased the marginal utilities of ameliorating activities such as cleaning up and repairing damages after a flood. Option price values for reducing A^* will be greater than expected damages, and more so for larger losses occurring with lower probabilities.

The findings for utility functions with this characteristic can be summarized by the following statements:

1 Since the differences are positive as predicted by equation (5.76), empirical valuation measures based on changes in expected damages will understate the option prices.
2 The difference is an increasing function of CS, since a larger CS means a larger difference between v_M^0 and v_M^*.
3 The differences are for the most part trivially small for small losses (equivalent to up to 1 percent of income). Only for high degrees of risk aversion does the difference approach 10 percent for low probability losses.
4 For relatively large losses (equivalent to, say, 10 percent of income), the differences range between 5 percent and 20 percent, depending on the probability and for moderate degrees of risk aversion. For relatively high degrees of risk aversion, the differences range above 60 percent.
5 For catastrophic loss (equivalent to 50 percent of income), the differences are large for all but the least risk-averse form of utility function and most likely event. For the more risk-averse forms of utility function, this means

that option prices for risk reduction can exceed expected damages by factors of 2.5 to more than 100.

This last conclusion might be a possible explanation for a phenomenon that some observers have noted—that is, that the public seems unwilling to accept some very low-probability, high-consequence risks that experts have judged to be acceptable. If the experts' judgments are based on ex post valuation measures and if people are relatively risk-averse and their preferences take this form, the experts could be vastly underestimating the true potential welfare costs that these risks would impose on people.

For other forms of utility functions, v^0_M is greater than v_{M^*}. Examples include

$$v = (1 - A^*)^c M^b \text{ with } 0 < b, c < 1 , \tag{5.81}$$

and

$$v = (1 - A^*) \ln M . \tag{5.82}$$

One way of interpreting this characteristic is that income and the absence of A^* are complements in consumption such that if the event does not occur, the marginal utility of the consumption that M allows is high. But if the event does occur, the capacity for consumption to generate utility is diminished. For example, the event "broken leg" decreases the marginal utility of expenditures for hiking and skiing trips. For these functions, expected damages exceed option prices.

Calculations with these functions show that differences are negative as predicted and are less than 10 percent except for low probabilities of very large losses (50 percent of income). For most functional forms, the magnitude of the difference is larger for the most risk-averse utility functions. But for one functional form the difference is largest when the degree of risk aversion is zero; and for one functional form, even the sign of the difference depends on the degree of risk aversion.

As these results demonstrate, option price and expected damage measures of the value of risk reduction are likely to be different. Expected damage measures are very unreliable proxies for option price measures. The sign of the difference depends on specific features of the functions chosen to represent preferences. The degree of risk aversion is not a reliable predictor of the size or even the sign of the differences. Values for risk reduction based on expected damages are not likely to be useful, and could be seriously misleading as guides for risk management decisions.

Next, we turn to the value of risk prevention. The expected damage measure of the value of reducing the probability of a loss is the reduction in the expected value of the compensating surplus associated with the loss. As explained above, the expected loss, $E[D]$, is $\pi \cdot D$. By differentiation, we have

$$dE[D] = d(\pi \cdot D) = D \cdot d\pi . \tag{5.83}$$

In order to examine the relationship between the option price and expected damage measures of the value of a probability change, the right-hand sides

of equations (5.35) and (5.83) must be compared, however, they are not easily comparable. The option price measure starts with the difference between two utility levels and converts it to a money measure by using the weighted average of the marginal utilities of income in the two states. The expected damage measure is already in monetary units; specifically, it is the difference in money expenditure necessary to achieve the same level of utility under two different sets of conditions.

Again, Freeman (1989) assumed specific functional forms and parameters and calculated differences. The results are difficult to summarize except to say that differences can be positive or negative, large or small, and can be large even in the absence of risk aversion.

In estimating the values of reducing the probability of adverse events, expected damage measures are unreliable indicators of the desired option prices. Expected damage measures can be either underestimates or overestimates, and the errors involved in using them as proxies for option prices can be large. However, they cannot be predicted without detailed knowledge of the specific characteristics of individuals' preferences.

Is Option Value a Value?

The concept of option value was introduced by Weisbrod in a much-cited paper nearly 50 years ago (Weisbrod 1964). This concept has had an interesting history since its birth, both in terms of its development as a theoretical construct and because of its relationship to, and role in, policy discussions concerning environmental resources. Weisbrod argued that an individual who was unsure of whether he would visit a site such as a national park would be willing to pay a sum over and above his expected consumer surplus to guarantee that the site would be available should he wish to visit it. Weisbrod called this extra sum the option value of the site. Option value was seen to arise when an individual was uncertain as to whether he would demand a good in some future period and was faced with uncertainty about the availability of that good. If option price (*OP*) is defined as the maximum sum the individual would be willing to pay to preserve the option to visit the site before his own demand uncertainty is resolved, then the excess of option price over expected consumer surplus can be called option value (*OV*). It was thought that option value should be measured, if it were possible, and added to expected consumer surplus in order to obtain the full measure of the value of providing an environmental service.

Weisbrod apparently viewed the existence of positive option value as being intuitively obvious. Indeed, there is no formal mathematical or logical proof of the existence of option value in Weisbrod's paper. But as the subsequent literature has shown, there are a number of subtleties and complications to the concept, as well as traps for the unwary investigator. For example, Cicchetti and Freeman thought they had proved that option value was positive for risk-averse individuals (Cicchetti and Freeman 1971), but other modelers demonstrated that option value could be negative in this case (see Schmalensee 1972; Anderson 1981; Bishop

1982). Schmalensee showed that even for a risk-averse individual, option value could be greater than, equal to, or less than zero depending upon the particular circumstances. This is essentially a matter of the relationship between the expected damage (or surplus) measure and the option price measure discussed in a preceding section of this chapter. As Bohm (1975) pointed out in a comment on Schmalensee, what matters is the relationship between the marginal utilities of income in the different states of nature.

The matter seemed to rest there until Hartman and Plummer (1987) and Freeman (1984b) provided specific characterizations of the nature of demand uncertainty and examined the implications of different types of demand uncertainty for the relationship between the marginal utilities of income for different states of nature. For example, Hartman and Plummer showed that if an individual was uncertain about future income and the demand for the good in question was a positive function of income, option value is unambiguously negative for risk-averse individuals. Freeman (1984b) showed that in this case risk lovers would have positive option values, and that one plausible form of state-dependent preferences assured positive option values for risk-averse individuals.

At about the same time, Bishop (1982) suggested that useful insights might be obtained by considering the simpler case in which an individual is certain of demand but faces uncertain supply of the good. He showed that option value is greater than zero for a project that eliminates the uncertainty of supply. Thus, option value appeared to be resurrected as a form of benefit associated with a guaranteed future supply of goods, such as national parks. However, Freeman showed that Bishop's conclusion did not hold for all possible forms of reduction in uncertainty of supply. What Freeman called supply-side option value could also be either positive or negative (Freeman 1985). See also Wilman (1987) for more on this topic.

Providing a rationale for preservation of wilderness and scenic beauties seems to have been the motivation for the original investigation of option value. Milton Friedman (1962) in *Capitalism and Freedom* had argued that since there were no externalities associated with uses of national parks, all of the relevant economic values could be captured by the owners of parks through admission fees. Thus, the allocation of land to national parks should be subjected to a market test. Weisbrod's 1964 paper was clearly, at least in part, a response to this argument; and Krutilla (1967) included option value in his list of reasons why markets might fail to achieve allocative efficiency for unique environmental resources. Option value was presented as an economic value over and above the expected use values; and it could not be captured through admission fees. The preservation of the option to visit the park was a form of public good that might justify the preservation of a natural area even when expected use values were less than opportunity costs.

Schmalensee's (1972) theoretical proof that the sign of option value is ambiguous seems to have reduced interest in option value as a policy-relevant concept, at least temporarily. But, in 1983 the Environmental Protection Agency (EPA) listed option value as one form of intrinsic benefit that could be included

in the benefit-cost analyses of proposed regulations required under the terms of Executive Order 12291 (U.S. Environmental Protection Agency 1983). Measurement of option value was also part of the work to be done in several EPA-sponsored research projects during the 1980s and early 1990s. In light of the state of the theory, this continuing effort to measure option value and to use it in policymaking is an interesting comment on the power of the idea.

The idea of option value may have been most useful through stimulating the more rigorous analysis of the theory of welfare measurement under uncertainty. Because of the theoretical contributions of authors such as Schmalensee (1972), Graham (1981), Bishop (1982), and Smith (1987a, 1987b), we can now see that what has been called an option value is really just the algebraic difference between the expected values of two different points on a WTP locus. Specifically, it is the algebraic difference between the expected value of the consumer surplus and the state-independent willingness to pay (option price). Since these two points represent alternative ways of measuring the same welfare change, the difference between their expected values cannot be a separate component of value. Furthermore, option value cannot be measured separately—it can only be calculated if we have enough information on preferences to calculate both option price and expected surplus. Finally, as has been shown in this chapter, neither of these points on the WTP locus has any particular claim as a superior welfare measure. Perhaps in recognition of this, option value is not mentioned in EPA's most recent set of guidelines for economic assessment (U.S. Environmental Protection Agency 2000).

Summary

In this chapter, the various ways in which welfare changes can be defined and measured for changes in risks have been described. Graham's WTP locus, option prices, and dynamic WTP measures were all defined and related to each other. The relationship between option price and option value was discussed, as was the fact that option value does not constitute a unique component of value. These measures were related to the PPI criterion when it is employed in benefit-cost analyses of policies dealing with risk. In addition, dynamic welfare measures were described and related to their static counterparts.

The methods presented here are based on an important maintained assumption, which is that individuals are capable of assigning probabilities to alternative outcomes that are consistent with probability theory and contain all relevant information. However, there is a substantial body of evidence that people find it hard to think in probabilistic terms and often have serious misperceptions about the magnitude of important environmental risks. For references to some of this evidence and discussions of its implications for economic analysis, see Machina (1990) and Arrow (1982). Other important references include Slovic, Fischhoff, and Lichtenstein (1979, 1980, 1982). As noted above, people often make choices that are not consistent with expected utility theory.

The observation that our models of behavior may not be very good descriptions of how people actually behave toward risk raises some interesting and difficult questions concerning the evaluation of public policies dealing with risks. From a theoretical perspective it is appropriate to use the preferences and probability assessment of individuals who are facing the risks to form welfare measures for use in a cost-benefit analysis. However, when an individual's perceptions of the risk is significantly out of line with an objective values of that risk, a case can be made that society would be better off if the more accurate representation were used.

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Chapter 6

Aggregation of Values across Time

For purposes of economic planning, management, and policymaking, it may often be useful to view environmental and natural resources as assets that yield flows of services over time. The theoretical framework necessary for defining and measuring the values of these service flows to the individuals who receive them has now been developed. However, this theory of value has been developed without explicit consideration of the temporal dimension of these service flows. Specifically, the theory has been applied to the values of services only at a single point in time or over some relatively short period of time (a day, week, or year) during which it is reasonable to treat other things such as income, prices, and the consumption of other goods and services as fixed. The theory also has defined values only at that moment in time at which the service flow is received; and thus, we are left with the question of how to aggregate values that are realized at different moments in time and that might vary from one moment to another.

This chapter deals explicitly with the temporal dimension of value and welfare theory. In the first section, the standard theory of individual preferences and choice in an intertemporal setting is reviewed. In the absence of taxes and other capital market imperfections, utility maximizing individuals borrow or lend so as to equate their marginal rate of substitution between present and future consumption with the market rate of interest. Thus, the interest rate can be taken as a revealed preference indicator of individuals' intertemporal marginal rates of substitution.

In the second section, an individual's marginal rate of substitution is used to convert a marginal welfare measure for one moment in time into its equivalent at any other point in time. Measures for several different points in time can be made commensurate by converting them all to one period—for example, the present. Then these measures can be added to obtain an intertemporal aggregate welfare measure.

After reviewing the theory of intertemporal choice and using this theory to develop measures of intertemporal welfare change, the third section addresses the question of choosing an interest rate in a world with many different market interest rates and other intertemporal prices. The multitude of market interest rates arises because of inflation, taxation, and risk, among other things. Consequently,

there is no single number that can be identified as the "correct" interest rate for intertemporal welfare measurement in all circumstances; but it is possible to suggest a reasonable range that will be appropriate in many situations.

One consequence of the multiplicity of interest rates is that environmental policies that divert resources from other investments with high rates of return may have intertemporal opportunity costs at the margin that are different from individuals' intertemporal marginal rates of substitution. One approach to calculating these opportunity costs and making them commensurate with the measure of welfare gains is developed in the fourth section.

The theory of intertemporal welfare measurement presented here has been developed to examine the welfare implications of changes that affect one individual at different points in time. However, there are environmental issues that have long horizons where policy decisions made now will affect future generations. Global climate change is an area in which this concern has become a major point of discussion. The question of interpersonal welfare comparisons across time and the choice of discount rate when multiple generations are affected is taken up the final section of this chapter.

Individual Preferences and Intertemporal Choice

The theory of preferences used up to this point models only single-period choices and makes single-period utility a function of the single-period consumption of marketed goods and services and environmental quality. With time introduced explicitly, the single-period utility in time period t is given by

$$u_t = u_t\left(X_t, q_t\right), \tag{6.1}$$

where X_t measures the expenditure on consumption in t (X_t is a numeraire good with a unit price), and q_t is the level of the environmental or resource service flow at t. This model has obvious limitations in that it cannot be used to analyze intertemporal choices and such things as saving and lending.

The alternative is a model of lifetime utility. This general representation of preferences would have an individual at any point in time experiencing a level of "lifetime" utility from the goods and services and environmental quality that person anticipates over his or her remaining lifetime:

$$u^* = u^*\left(X_1, \ldots, X_t, \ldots, X_T, q_1, \ldots, q_t, \ldots, q_T\right), \tag{6.2}$$

where T is the number of years of remaining life. For simplicity, this formulation implies that the individual gains no satisfaction from making a bequest to others at the end of his or her life; but it is a straightforward matter to add a bequest motivation if that is an important part of the problem being analyzed (see, for example, Cropper and Sussman 1988). This analysis abstracts from all considerations of uncertainty concerning the future.

The existence of single-period preferences as described by equation (6.1) implies that the lifetime utility function (6.2) is separable. Brekke (1997, 93, 108–113)

argued that this is a dubious assumption since the utility of X_t and q_t could plausibly depend on the past or future levels of either variable for a variety of reasons. If Brekke's argument is accepted, this means that the expressions for the marginal values of q_t derived later in this chapter would have to include as arguments all past and future levels of X_t and q_t. This would vastly complicate the empirical task of estimation, so as a pragmatic simplification lifetime utility is written as

$$u^* = u^*\left[u_1\left(X_1, q_1\right), \ldots, u_t\left(X_t, q_t\right), \ldots, u_T\left(X_T, q_T\right)\right]. \tag{6.3}$$

It is common to assume that this expression is additively separable and that the single-period utility function is the same for all periods. Thus, we can write

$$u^* = \sum_{t=1}^{T} D_t \cdot u\left(X_t, q_t\right), \tag{6.4}$$

where $u(\cdot)$ is invariant in t, $\partial u^*/\partial X_t$ is positive and decreasing in X_t, and where D_t is meant to capture the individual's time preference.[1] For example, if an individual intrinsically preferred present consumption over future consumption, D_t would decrease with t and the marginal lifetime utility of X_t would be a decreasing function of t.

A useful way to characterize time preference is to let $D_t = 1/(1 + d)^t$ where d can be interpreted as a subjective rate of time preference analogous to an interest rate. This form of constant d and geometrically decreasing D_t over time is referred to as exponential discounting and has been shown to be necessary for intertemporal consistency of individual choice (Strotz 1956; Page 1977). Given exponential discounting, the relationship between d and time preference is as follows:

- $d > 0$ represents positive time preference with the marginal utility of X_t decreasing with t;
- $d = 0$ represents neutral time preference; and
- $d < 0$ represents negative time preference with the marginal utility of X_t increasing with t.

Now assume that the individual knows with certainty that he or she will receive a stream of income payments, M_t, and a stream of environmental services, q_t, and that the individual can borrow or lend in a perfect capital market at an interest rate of r% per period. Assume that wealth is measured as of the beginning of the first period and that income and expenditures are made at the end of each period. Then, if the individual does not wish to leave a bequest and assuming that all accounts must be settled in year T, the individual's lifetime budget constraint is

$$\sum_{t=1}^{T} X_t \cdot (1+r)^{-t} = W^* = W_0 + \sum_{t=1}^{T} M_t \cdot (1+r)^{-t}, \tag{6.5}$$

1 See Trostel and Taylor (2001) for an interesting model of time preference that makes $u(\cdot)$ a function of age.

where W^* is lifetime wealth and W_0 is initial wealth. Knowledge of the initial wealth, the streams of income and environmental services, and the interest rate is sufficient to determine the lifetime pattern of consumption of X that maximizes lifetime utility. In each period, if X_t is greater than (less than) M_t, the individual must borrow (lend) the difference.

The first-order conditions for maximizing equation (6.4) subject to the wealth constraint (6.5) take the form

$$\frac{\partial u^* / \partial X_t}{\partial u^* / \partial X_{t+1}} = 1 + r \,, \tag{6.6}$$

or, since $\partial u^* / \partial X_t = D_t \cdot \partial u / \partial X_t = (1+d)^{-t} \, \partial u / \partial X_t$,

$$(1+d)\frac{\partial u / \partial X_t}{\partial u / \partial X_{t+1}} = 1 + r \quad \text{for all } t = 1, \ldots, t, \ldots, T-1 \,. \tag{6.7}$$

The individual's intertemporal marginal rate of substitution must be equal to one plus the interest rate. Thus, the individual's intertemporal marginal rate of substitution can be inferred by observing the interest rate that governs these intertemporal tradeoffs. The first-order conditions imply that if both r and d equal 0, the individual will equate the single-period marginal utility of consumption across all periods, thus choosing a constant stream of consumption over time. If $r = 0$ and $d > 0$, the single-period marginal utility of consumption in t must be less than that in $t + 1$, implying that the individual chooses to consume relatively more in period t. An increase in r, other things being equal, requires adjustments to increase the marginal utility of consumption in earlier periods, thus implying a deferral of consumption to future time periods.

Measures of Welfare Change

Next, the development of measures of intertemporal welfare change are considered. First, consider a marginal increase in q_t. The individual would be willing to accept a reduced quantity of the numeraire in that period to hold utility constant. This compensating surplus measure of welfare change is the marginal rate of substitution between q_t and X_t, or

$$w_{qt}^{t} = \frac{\partial u / \partial q_t}{\partial u / \partial X_t} \,. \tag{6.8}$$

From equation (6.6) or (6.7), in intertemporal equilibrium the individual would be indifferent between paying w_{qt}^{t} in period t and paying

$$w_{qt}^{0} = \frac{w_{qt}^{t}}{(1+r)^t} \tag{6.9}$$

now. If the whole stream of future $\boldsymbol{Q}$ increases, the marginal willingness to pay now is the sum of the willingness to pay for each of the components of the increase, or

$$w_{\boldsymbol{Q}}^{0} = \sum_{t=1}^{T} \frac{w_{qt}^{t}}{(1+r)^{t}}. \tag{6.10}$$

If the changes in $\boldsymbol{Q}$ are nonmarginal, aggregation over time is not so simple. As in Chapter 3, we can define single-period compensating surplus (*CS*) and equivalent surplus (*ES*) welfare measures for changes in any q_t. These are changes in single-period income based on single-period utility. In an analogous fashion, we can use equation (6.2) or (6.3) combined with equation (6.5) to define compensating and equivalent changes in initial wealth that are based on lifetime utility. What is of interest is the relationship between these two alternative perspectives on the measurement of welfare change.

To keep it simple, consider a policy that would result in a change in q during only one period. Specifically, suppose that the policy results in $q_t'' > q_t'$. What is the welfare gain to the individual of this change? The single-period compensating surplus measure is the solution to

$$u\left(X_t - CS_t, q_t''\right) = u\left(X_t, q_t'\right). \tag{6.11}$$

The present value of the single-period measure of gain is $(1+r)^{-t} CS_t$.

In order to derive the compensating and equivalent wealth measures of lifetime welfare change, the first step is to use the solution to the lifetime utility maximization problem to obtain the lifetime indirect utility function:

$$v^{*}\left(W^{*}, q_1, \ldots, q_t, \ldots, q_T\right). \tag{6.12}$$

Then from the perspective of lifetime utility, the individual is willing to make a payment at $t = 0$ that will equate the lifetime utilities with and without the change in q_t. Call this the compensating wealth, or *CW*. It is the solution to

$$v^{*}\left(W^{*} - CW, q_1, \ldots, q_t'', \ldots, q_T\right) = v^{*}\left(W^{*}, q_1, \ldots, q_t', \ldots, q_T\right). \tag{6.13}$$

Following the analysis by Blackorby, Donaldson, and Moloney (1984), it can be shown that the present value welfare measures based on equations (6.11) and (6.13) are different except under certain special conditions. Specifically:

$$\left|CW\right| \geq \left|(1+r)^{-t} CS_t\right|. \tag{6.14}$$

This result can be easily explained. If the individual pays *CS* in period *t*, then according to equation (6.11) single-period utility is restored to its initial level, and so is lifetime utility. However, if the single-period marginal utility of consumption in *t* is affected by the change in q_t, then the first-order conditions for an optimum allocation of consumption over time will not be satisfied. The individual will wish to reallocate consumption across time through some pattern of additional borrowing and lending. Such reallocation of consumption will increase utility. This means that the individual could have made a larger single-period payment to keep lifetime utility at its original level. This establishes the inequality of equation (6.14).

The equality will hold only if the marginal utility of consumption is invariant to the combined changes in q_t and its offsetting CS_t. For example, suppose that

the increase in q_t causes a decrease in the marginal utility of consumption in that period. The payment of CS_t will increase the marginal utility of consumption in period t. These two forces might exactly offset each other, leaving the marginal utility of consumption and the optimal intertemporal pattern of consumption unchanged. However, if the single-period utility function is separable in q_t, such that the marginal utility of consumption is unchanged by the change in q_t, or if the increase in q_t leads to an increase in the marginal utility of consumption, then the net effect of the change in q_t and the offsetting compensation payment will be to increase the marginal utility of consumption in period t, and to cause a reallocation of consumption across time.

Similar inequalities can be derived for equivalent surplus and equivalent wealth (EW) measures of welfare gain and for both types of measures for welfare losses associated with a decrease in q_t. For example, suppose that q_t decreased. If the individual instead had to pay ES_t as the equivalent measure of welfare loss, and if this resulted in an increase in the marginal utility of consumption in t, then the individual could restore part of the loss of lifetime utility by reallocating spending toward period t, and ES_t would be an overestimate of the lifetime welfare loss. These results can be summarized as shown in Table 6.1. All of these expressions can be easily generalized to the case where there are changes in $\boldsymbol{Q}$ in many periods.

These results have somewhat disturbing implications for applied welfare analysis. The typical practice in evaluating environmental policies that affect welfare over many years is to estimate single-period welfare measures and to compute the present value of these single-period changes using some interest rate. But as shown here, if compensating surplus measures of single-period welfare change are used, this procedure will underestimate the true lifetime welfare benefit of the stream of improvements in $\boldsymbol{Q}$ and overstate the loss of decreases in $\boldsymbol{Q}$. If the present value of the single-period CS_t is greater than zero, then the project unambiguously passes a Kaldor potential compensation test; but projects with negative aggregate CS_t could still be efficient. Conversely, if equivalent surplus single-period measures are used, the conventional practice leads to upward-biased estimates of total welfare gains and downward-biased estimates of lifetime losses. Thus, if the sum of the single-period ES_t is less than zero, the project unambiguously fails a Hicks potential compensation test; but projects with positive aggregate present values of single-period ES_t might decrease aggregate welfare (Blackorby, Donaldson, and Moloney 1984).

Table 6.1 Derived inequalities

For gains (q_t increases)	*For losses (q_t decreases)*
$CW \geq (1+r)^{-t} CS_t$	$\lvert CW \rvert \leq \lvert (1+r)^{-t} CS_t \rvert$
$EW \leq (1+r)^{-t} ES_t$	$\lvert EW \rvert \geq \lvert (1+r)^{-t} ES_t \rvert$

Of course, the important question is, how big is this bias? Keen (1990) has shed some light on this question. He showed that the difference between the single-period and the lifetime welfare measures was a term that he called the intertemporal compensating variation, or the maximum amount the individual would be willing to pay to be able to reallocate expenditure and utility across periods. Keen then derived expressions for this term and showed that in most instances the term would be of only second-order importance. In general, the error will be significant only if high values for the elasticity of intertemporal substitution combine with large values of CS_t over time.

Although most of the methods for measuring welfare change produce only single-period measures, there are cases in which it is possible to obtain direct unbiased estimates of *CW* or *EW*. If changes in $\boldsymbol{Q}$ affect the market price of an asset, the change in its price might provide a basis for estimating *CW* or *EW*. For example, the hedonic price method is based on differences in the prices of assets such as houses. Since what is observed are differences in the market prices for the housing asset, price differentials between low- and high-quality houses reflect the compensating wealth change associated with the difference in the expected stream of environmental quality associated with the house. It is common practice in applied welfare analysis to convert the compensating wealth measure to an annual equivalent by using some interest rate or discount factor. Provided the interest rate properly represents the individual's opportunities for intertemporal substitution, this annual equivalent can be used as an unbiased indicator of lifetime welfare change.

To summarize, if an individual can reallocate his stream of consumption over time through transactions at some interest rate r, that person will equate his intertemporal marginal rate of substitution with a discount factor equal to $1 + r$. This means that r can be interpreted as an intertemporal price or measure of intertemporal value, at least for marginal changes. For nonmarginal changes in $\boldsymbol{Q}$, the correct measure of intertemporal welfare change is the compensating or equivalent change in initial wealth. But many of the available methods for measuring welfare change yield estimates of single-period values rather than lifetime values; and the present value of these single-period welfare measures is, at best, only an approximation (an upper or lower bound) on the lifetime welfare measure. Given the limited availability of data, the approximation may be all that can be obtained.

An important example of computing intertemporal welfare measures is the construction of estimates of the social cost of carbon for use in regulatory analysis for the U.S. government. An interagency working group was formed in 2009 and issued a report in 2010 (Interagency Working Group 2010; Greenstone, Kopits, and Wolverton 2013) that describes the approach the group used to generate estimates of the economic damages generated by additional carbon emissions from 2010 to 2050. To generate this path of emission damages, the authors drew on three integrated assessment models (IAMs) and other data sources to develop a baseline trajectory of emissions, population, world GDP, and consumption levels.

Then, they postulated additional carbon emissions in a given year (say 2020) and used the models to predict changes in the temperature and consumption levels for each year after the postulated change, computed the marginal damages from those changes, and discounted those changes back to the base year. This was done for each year under a range of socioeconomic assumptions. Their estimates of the social cost of carbon rise over time with a "central" social cost of carbon value beginning at $21 (in 2007 dollars) per ton of CO_2.

Which Interest Rate is the Right Intertemporal Price?

How can we estimate an individual's rate of time preference? Standard economic theory predicts that in a simple economy with no taxes and with perfect capital markets in which all individuals can borrow or lend at the market rate of interest, an individual will arrange her time pattern of consumption by borrowing and lending so that at each point in time her rate of time preference is just equal to the observed market rate of interest (Lind 1982; Varian 1990). In such a simple economy, this is the interest rate that should be used for the intertemporal aggregation of individuals' single-period welfare measures. There are, however, in fact taxes (certainly), inflation (at least sometimes), and various market imperfections; and we observe a multiplicity of market interest rates. The purpose of this section is to discuss briefly the choice of an interest rate for welfare evaluation when factors such as inflation and taxation are present and to suggest some approaches to choosing an intertemporal price when faced with many rates of interest and tax rates.

Consider first the problem of the effects of taxes on income. Taxes can drive wedges between market prices and individuals' marginal rates of substitution. Suppose that an individual's interest income is taxed at the rate of t%. Then money lent at r% will return only $r \cdot (1 - t)$% net of tax. It is this after-tax rate of return to which the optimizing lender will equate his intertemporal marginal rate of substitution. Similarly, if interest expense is treated as deductible in calculating tax liabilities, then the net cost of borrowing at a market rate of r is $r \cdot (1 - t)$%. Again, it is this after-tax net borrowing cost that will govern the borrowing decisions of individuals. Discounting for welfare evaluation should be done at this after-tax interest rate.

A second problem is inflation. If there is price inflation, then the distinction between real and nominal interest rates becomes important. The individual would still arrange her time pattern of consumption so as to equate her rate of time preference with the expected real rate of interest. If inflation of i% is perfectly anticipated, then the observed market interest rate m and the real rate of interest r are related as follows:

$$\frac{(1+m)}{(1+i)} = 1 + r \quad \text{or} \quad m = r + i + r \cdot i \,. \tag{6.15}$$

Because for reasonable rates of inflation and real interest rates the third term in equation (6.15) will be small and can be ignored, the real rate is (approximately) the market interest rate less the rate of inflation. If the estimates of individuals' single-period CS_t are based on the expectation of actual prices in that period, then discounting should be done at the market rate of interest, which will also reflect the anticipated rate of inflation, i. However, the conventional practice in benefit measurement is to abstract from inflation by basing future period estimates of CS_t on the present price level. In that case, discounting should be done at the real rate of interest.

What is a reasonable value for this real rate of interest? One range of estimates comes from Lind (1982, 84). After reviewing historical evidence on rates of return to a variety of investments, Lind concluded that rates of time preference could be as low as 0 percent, if historical real rates of return on U.S. Treasury bills were taken as the benchmark. Over this same period of time, the real after-tax return on a broad portfolio of common stocks was about 4.6 percent (Lind 1982, 83). This could be taken as an upper bound on the appropriate discount rate and would be the correct discount rate if the payments or receipts being discounted had the same risk characteristics as the market portfolio of common stocks.

Given the current openness of financial markets and the ease of moving funds between nations, world average interest rates may be more relevant than rates from any one nation's financial market (Lind 1990). Barro and Martin (1990) presented estimates of an index of real expected and real actual short-term interest rates for nine major industrialized nations, including the United States. For more than two-thirds of the period studied, average pretax real rates of interest were below 2 percent, and they never exceeded 6 percent. At least in the United States, real rates have been in the lower half of this range during most of the first decade of this century. These results are broadly consistent with Lind's conclusions (Lind 1982). They also demonstrate the variability of real rates of interest over time. In a recent assessment of this issue, Boardman et al. (2011) examined returns on Moody's AAA rated corporate bonds and after adjusting for inflation concluded that 4.5 percent is a reasonable estimate of the marginal real rate of return on investment.

Rather than look at market rates of interest, an alternative approach is to estimate the implicit interest rate that must be guiding individuals' observed choices over alternative temporal patterns of benefits and costs. For example, suppose an individual who has a choice between receiving $10 today and $15 in one year chooses the smaller immediate payment. This implies that the individual discounts future consumption at a rate of at least 50 percent. Empirical studies on choices of this sort often reveal unusually high implicit interest rates—more than 20 percent and up to and above 100 percent.

Frederick, Loewenstein, and O'Donoghue (2002, 389) summarized and discussed over forty empirical studies and estimates of the rate of time preference that had been generated using either revealed preference studies (inferring the rate of time preference from actual economic choices) or stated preference experiments

(where inference is based on stated choices). They concluded that this comparison "reveals spectacular disagreement among dozens of studies that all purport to be measuring time preference." They suggested that the disparity is likely due to the fact that each study is measuring both time preference and a combination of other factors such as inflation, habit formation, and uncertainty about the future returns. They also noted, however, that the estimates generally suggest discount rates that are relatively large (many studies generate estimates in excess of 10 to 15 percent). The authors concluded that these "large" estimates may also be due to the confounding factors creating disparate rates in the first place. One striking finding in the review by Frederick, Loewenstein, and O'Donoghue is that the studies that focus on short-term intertemporal tradeoffs generally find much higher implied discount rates than those that focus on longer-term tradeoffs (see Chabris, Laibson, and Schuldt 2008 for additional evidence). In fact, many experiments in which people are offered either hypothetical or real choices between payments of different sizes and dates consistently reveal implicit discount rates that are both high for short periods of time and declining as the interval lengthens (Loewenstein and Thaler 1989; Ainslie 1991; Cropper, Aydede, and Portney 1994). This evidence implies nongeometric discounting, also referred to as hyperbolic discounting. In hyperbolic discounting, each element in a stream of payments or receipts is discounted at a rate that decreases as the period increases. For discussions of the implications of nongeometric or time-inconsistent discounting for individual behavior and the evaluation of public policies, see Harvey (1994), Laibson (1997), O'Donoghue and Rabin (1999), and Cropper and Laibson (1999).

Behavioral economists have advanced a number of possible explanations for this apparent use of hyperbolic discounting relating to limited self-control and immediate gratification (Robinson and Hammitt 2011). Robinson and Hammitt noted these findings pose challenges to the use of standard exponential discounting in benefit-cost analysis. They concluded, however, that since most of the evidence of hyperbolic discounting stems from decisions and behaviors that are short term in nature and most programs being evaluated by benefit-cost analysis are intended to be longer term, it is appropriate to use more traditional market rates. Robinson and Hammitt's recommendation seems appropriate at this juncture; however, as behavioral economics advances, it may be appropriate to reconsider these questions.

Even if we have estimates of market rates of interest or implicit discount rates from revealed or stated behavior, another problem arises in the calculation of welfare measures because of the fact that, for a variety of reasons, people face different interest rates. For example, individuals' real after-tax rates of interest will differ, since individuals face different marginal tax rates and may have different portfolios of investments. Differences in effective interest rates can also arise from market imperfections, transactions costs, and differences in the tax treatment of interest paid and interest received. How should we deal with differences in effective interest rates across individuals? In principle, each individual's single-period welfare changes should be discounted at the interest rate that the person uses to make intertemporal allocation decisions. However, in practice, we typically have single-

period welfare measures that are aggregated across individuals who may face quite different rates of interest. Assume for the moment that no individual borrows and lends at the same time and that at any point in time each individual's transactions are entered into at a single interest rate that may vary across individuals. Consider an individual who at period t could borrow for one period at r_{bt} or could lend at another rate r_{st}, where both are real after-tax rates of interest. The individual, in effect, faces a kinked budget line. Whichever direction the individual chooses to move in, the interest rate governing that form of transaction will be the one with which that person equates his or her intertemporal marginal rate of substitution. That interest rate is the one that should be used to discount that individual's welfare measure over that interval of time. So if the individual is a net lender between period t and period $t + 1$, and there is a change in q_{t+1}, the after-tax lending rate should be used to discount CS_{t+1} back to t. If the individual neither borrows nor lends, there is no transaction from which to infer a marginal rate of substitution.

Now consider aggregation across individuals who face different effective interest rates at any point in time because of, for example, differences in marginal income tax rates, or differences in borrowing and lending behavior. Specifically, assume that the ith individual faces an effective real interest rate r_i that is constant over time. Assume then that we wish to calculate the sum of the compensating wealth payments that would leave each individual in his initial lifetime utility position. For individual i, the lifetime CW_i for a change in $\boldsymbol{Q}$ is approximated by the present value of the stream of single-period compensating surpluses:

$$CW_i \simeq \sum_{t=1}^{T} \frac{CS_{it}}{(1+r_i)^t} . \tag{6.16}$$

The aggregate change for the N affected individuals is

$$\overline{CW} \simeq \sum_{i=1}^{N} \sum_{t=1}^{T} \frac{CS_{it}}{(1+r_i)^t} . \tag{6.17}$$

There is a simple way of calculating a weighted average discount factor that can be used with equation (6.17). First, each individual's share of each period's aggregate compensating surplus is calculated

$$s_{it} = \frac{CS_{it}}{\sum_{j=1}^{N} CS_{jt}} , \tag{6.18}$$

and then this expression is used to substitute for CS_{it} in equation (6.17) to obtain

$$\overline{CW} \simeq \sum_{i=1}^{N} \sum_{t=1}^{T} s_{it} \frac{\sum_{j=1}^{N} CS_{jt}}{(1+r_i)^t} . \tag{6.19}$$

The preceding analysis has been based on the assumption that each individual enters into transactions at only one interest rate in any one period. However, introspection and anecdotal evidence suggest that this assumption is often violated

in practice. Many people simultaneously save by contributing to pension funds and borrow to buy new houses and consumer durables. Yet, they may earn after-tax rates of return on savings instruments that are only a third or a quarter of their effective borrowing costs on outstanding credit card balances. This behavior raises two questions. First, how can this behavior be reconciled with the assumption of economic rationality? Second, what does this behavior imply for inferring intertemporal marginal rates of substitution and making intertemporal welfare measurements?

Lind (1990) suggested that developments in behavioral economics can help to explain these practices. One key paper in this literature (Shefrin and Thaler 1988) hypothesized that people place different types of income, expenditures, and assets into different mental accounts and that they make different intertemporal allocation decisions for different sets of accounts. If it should prove possible to identify unique marginal rates of substitution or implicit discount rates for each mental account, and if it is possible to discover to which mental accounts people assign changes in environmental quality, then it may prove possible to develop a consistent discounting framework for intertemporal welfare measurement.

In summary, the existence of multiple interest rates in the marketplace does not preclude the consistent intertemporal aggregation of individuals' welfare measures. In principle, each individual's set of welfare measures is aggregated across time, using that individual's intertemporal marginal rate of substitution as revealed by the interest rate at which that person transacts. In practice, a weighted average discount factor reproduces this calculation. The case of individuals who simultaneously transact at different interest rates is more problematic. However, the concept of mental accounts may help to explain this behavior and may also provide a basis for calculating the relevant intertemporal marginal rate of substitution.

Capital Costs of Environmental Policies

A major issue in the economic evaluation of public investment projects has been how to take account of the fact that financing the public investment through some combination of taxes and borrowing is likely to displace some private investment having a higher rate of return than the effective interest rate governing individuals' intertemporal substitutions. The divergence between the rate of return on private investment and the social discount rate can be attributed at least in part to the taxes on corporate and personal income. In this section, the "shadow-cost-of-capital" approach to measuring the cost of public investment projects is reviewed and the implications of using the shadow cost of capital for the economic evaluation of public policies that require private investments that may displace other private investments are discussed.

Consider a public investment project with an initial capital cost of K_0. Assume that no further operating or maintenance costs are associated with the project. Let the project yield benefits of B per year in perpetuity. As before, let r represent the

effective after-tax rate of interest governing all individuals' borrowing and lending decisions. Since r will reflect individuals' willingness to trade off present for future consumption, it should be used as the intertemporal price for welfare evaluation, given the individualist welfare perspective adopted in this book. Let s represent the real marginal rate of return on investments in the private sector; but because of taxes on capital income at both the corporate and personal level, $s > r$.

Using r, we can calculate the present value of the perpetual stream of benefits as a lower bound to the aggregate compensating wealth payment that is our desired lifetime welfare indicator. However, if we find $B/r > K_0$, this does not necessarily signal that the project will improve welfare, because the relevant welfare comparison is not between the present value of the stream of benefits and the capital cost of the project. Rather, the relevant comparison is between the present value of the benefits of the project and the opportunity cost of the project as reflected in the stream of consumption forgone. In other words, the capital cost must be converted to its consumption equivalent for comparison with the benefit stream.

If K_0 were invested at the marginal rate of return in the private sector, it would produce a perpetual stream of $s \cdot K_0$ per year of future consumption. This is what is lost by diverting K_0 of capital resources from private investment to public investment, and this is the opportunity cost of the public investment. The present value of the stream of future consumption forgone is

$$\frac{s \cdot K_0}{r}. \tag{6.20}$$

Thus, in this simple example the shadow price or true social opportunity cost of one dollar of capital diverted from private investment is s/r, which is greater than one. In practice, the shadow price of capital will depend on a variety of factors reflecting the extent to which private investment is displaced, on net. If no private investment is displaced, the shadow price is one. More realistic formulations of the shadow price expression would take into account the facts that public investment will not necessarily displace private investment on a dollar-for-dollar basis, that public and private investments have less than infinite lifetimes, and that some portion of the future returns from displaced private investment could be reinvested for future consumption rather than being consumed immediately. For a review of some of these issues and some estimates of the magnitude of the shadow price, see Lind (1982). Earlier important references include Bradford (1975) and Marglin (1963).

To summarize, the shadow price of capital is the present value (discounted at the individual rate of time preference, $\dot{r}$) of the stream of future consumption forgone from one dollar of public investment. The proper procedure for project evaluation is to discount benefits (and operating and maintenance costs, if any) at the social rate of discount. The present value of the benefit stream would then be compared with the social cost of the capital investment. The latter would be calculated by multiplying the dollar cost of the investment by the appropriately calculated shadow price of capital.

Now suppose there is a pollution control regulation that, instead of requiring public investment, requires private firms to invest capital in pollution control. As in the case of public investment, the relevant opportunity cost is the present value of displaced private consumption. The private investment can affect consumption through a variety of channels. A simple model will illustrate the basic principles. For more detailed analyses, see Kolb and Scheraga (1990) and Lesser and Zerbe (1994).

Suppose also that this regulation will yield environmental improvements with an aggregate value of B per year in perpetuity. Once again, the present value of the benefit stream from the regulation is B/r. Assume that it is estimated that firms can comply with the regulation by installing equipment with an aggregate capital cost of K_0, that there are operating, maintenance, and repair costs of R per year, and that the equipment will last forever. Suppose further that firms face infinitely elastic factor supply curves for all factor inputs and have constant returns-to-scale production functions. This assures that all of the costs of complying with the regulation will be passed on to consumers in the form of higher prices.

Because under these assumptions the aggregate supply of capital is infinitely elastic, the regulation does not result in any displacement of investment in the economy. The cost of the regulation comes in the form of higher prices to consumers. Prices must be raised sufficiently to amortize the investment of K_0 and to cover the operating costs of R per year. Ignoring the effects of higher prices on demand, the required price increase must generate additional revenues of $s \cdot K_0 + R$. This is the annual cost of the regulation to consumers. To calculate the present value of this stream of costs, the effective consumers' interest rate, r, should be used. The present value of the social cost to consumers is approximated by $(s/r) \cdot K_0 + R/r$. In effect, the capital cost is first annualized using the pretax marginal rate of return on private investment (s) and then converted back to a present value using the consumption rate of interest (r).

In order to take account of the effects of higher prices on demand and output, it is necessary to solve for that K_0 which is required to comply with the regulation after taking account of the effect of higher prices on output and the need for capital. For example, if some firms are expected to exit the industry, then fewer firms would have to make investments. The aggregate capital requirement would be smaller than if output and the number of firms was assumed unchanged. Also, the conceptually correct measure of cost is the CV of the consumers who must pay the higher prices.

Discounting and Aggregation Across Generations

We have seen that the aggregation of an individual's single-period welfare measures over time can be done using a discount factor that represents the individual's marginal rate of intertemporal substitution or rate of time preference. For rational individuals this rate will be equal to the relevant after-tax real rate of interest. The ethical justification for discounting benefits and costs that accrue within one

generation's lifetime lies in the observation that this discount rate is a reflection of the individuals' preferences for their own present consumption relative to future consumption.

However, discounting future benefits or costs over the lifetimes of two or more generations is controversial. Applying any discount rate to a future sum can make it look very small in presentvalue terms when the interval is 100 years or more. For example, at 2 percent, the present value of $1 in 100 years is about 14 cents; if the $1 comes in 200 years, its present value is only 2 cents at 2 percent.

The principal argument for discounting for economic welfare analysis also provides some insight into why one might choose not to make discounting calculations across generations. Recall that what is being discounted is individuals' willingness to pay or required compensation, either for a change (compensating measure) or to avoid a change (equivalent measure). The ethical justification for making these welfare calculations is their use in applying either the Kaldor or Hicks form of compensation test to determine whether it is possible to compensate the losers so that no one is made worse off. When the gainers and losers are part of the same generation, actual compensation is feasible, and in any event, both groups can participate in the decision. If a proposed policy would impose costs on a future generation, it is hard to imagine mechanisms for assuring that the compensation would be paid if it were thought desirable or necessary to obtain the consent of future generations—the future generation has no voice in present decisions. Similar problems arise in the case of a policy that would yield benefits to the future but impose costs on the present. The real problem with discounting across generations is not, then, that it results in very small numbers for the present values of future effects; rather, it is that it is hard to imagine a meaningful role for measures of compensation in the context of intergenerational resource reallocations.

So, what is to be done? One place to start is with the recognition that, just as in the case of interpersonal welfare comparisons within a generation, a social welfare function is required in order to make comparisons across generations. One approach to defining a social welfare function has its origins in work by Frank Ramsey nearly 90 years ago (Ramsey 1928). For a more recent restatement and discussion, see Arrow et al. (1996). In this formulation, the social rate of discount, r_s, emerges as the sum of two components. The first is the social rate of time preference, ρ, which reflects the society's judgment regarding the relative intrinsic deservingness of different generations. The most ethically appealing judgment is that ρ is zero meaning that all generations are treated as equally deserving. The second component reflects the expected relative economic position of the present and future generations. It is the product of the elasticity of the marginal utility of consumption, θ, and the expected rate of growth of future per capita consumption, g.

Combining these terms gives

$$r_s = \rho + \theta \cdot g \,. \tag{6.21}$$

The justification for the second term is that if per capita consumption is higher in the future, an extra dollar of consumption in the future will have a lower marginal utility to that generation and therefore should be discounted accordingly. Following this approach, with $\rho = 0$, a growth rate of consumption of 1 percent and an elasticity of 1.5 would yield a social discount rate of 1.5 percent. If growth in consumption were expected to be negative, equation (6.21) would produce a negative discount rate to give increments of consumption to future generations a greater social value than equal increments to the present generation.

Weitzman (1998, 2001) put forth a separate, but important, point concerning the appropriate discount rate to use. He argued for using very low discount rates when evaluating the present value of benefits and or costs that accrue in what he calls the "distant future" (76 to 300 years) and the "far distant future" (more than 300 years). His argument is based solely on the notion that future discount rates are uncertain, largely because of uncertainty about future technological progress. Weitzman defined the "certainty-equivalent" of an uncertain discount factor as its expected value and demonstrated that over a long period this expected value converges to the discount factor evaluated at the lowest discount rate possible. Thus, he provided a coherent justification for the existence of a hyperbolic discount rate schedule.

Newell and Pizer (2003, 53–54) provided a simple numerical example to illustrate what is involved. Consider an investment that is expected to yield benefits of \$1000 in 200 years. Let it be equally likely that the discount rate r will be 1 percent or 7 percent at that time. Thus, the expected value of r is 4 percent. If this rate is used to discount the future benefit, its present value is 34 cents ($\$1000e^{-0.04 \times 200}$). However, the expected value of the discounted present values at 1 percent and 7 percent is 200 times larger ($0.5(\$1000e^{-0.01 \times 200}) + 0.05(\$1000e^{-0.07 \times 200}) = \$68$). Weitzman (2001, 270) also demonstrated the empirical magnitude of adopting this approach. The appropriate marginal discount rate varies from about 4 percent for the immediate future (out to 5 years), down to about 1 percent for the distant future (75 to 300 years, and 0 percent after 300 years). Portney and Weyant (1999) provided additional discussion of discounting and intergenerational equity. In a recent Policy Forum published in *Science*, a number of economists called for the adoption of a declining discount rate schedule based on Weitzman's logic (Arrow et al. 2013).

The appropriate discount rate to use when costs and benefits with intergenerational stakes are at issue remains challenging and consensus among economists has not been reached. An example of this disagreement—as well as a warning concerning the importance of the choice of discount rate in intergenerational settings—is the controversy surrounding the speed at which nations should address climate change. Using a discount rate of 1.4 percent, Stern (2009) concluded that significant resources should be immediately invested to reduce global warming, calling for about 1 percent of world GDP to be used for this purpose. In contrast, Nordhaus (2008), using an average rate of 4 percent in his model, concluded that only modest investments are called for. While there are other differences between their models that account for differences in findings, the choice of discount rate is critically important.

In recognition of the issues raised by discounting costs and benefits that affect future generations, the U.S. Office of Management and Budget (2003) recommends that analysts examine the sensitivity of their findings using lower discounts than they recommend for short-term projects. Specifically, they recommend discount rates in the range of 1 to 3 percent. In constructing the social cost of carbon estimates discussed earlier, the authors seem to have followed this strategy as they adopted three discount rates for their computations including 2.5, 3, and 5 percent per year (Greenstone, Kopits, and Wolverton 2013).

Conclusions

Taking all of these considerations into account, what interest rate is the right intertemporal price? The U.S. Office of Management and Budget recommends that agencies compute the present value of benefits and costs using both a 3 percent and a 7 percent annual discount rate for the following reasons:

> The 7% rate is an estimate of the average before-tax rate of return to private capital in the U.S. economy, based on historical data. It is a broad measure that reflects the returns to real estate and small business capital as well as corporate capital. It approximates the opportunity cost of capital, and it is the appropriate discount rate whenever the main effect of a regulation is to displace or alter the use of capital in the private sector.
>
> The 3% discount rate is based on a recognition that the effects of regulation do not always fall exclusively or primarily on the allocation of capital. When regulation primarily and directly affects private consumption, a lower discount rate is appropriate. The alternative most often used is sometimes called the "social rate of time preference." This term simply means the rate at which "society" discounts future consumption flows to their present value. If one assumes the rate that the average saver uses to discount future consumption is a measure of the social rate of time preference, the real rate of return on long-term government debt may provide a fair approximation. Over the last 30 years, this rate has averaged around 3% in real annual terms on a pre-tax basis.
>
> (U.S. Office of Management and Budget 2003, 11)

Debates concerning the appropriate rate of discount to apply in benefit-cost analysis will continue, particularly when long time horizons are involved. Practitioners will want to carefully review the recommendations and best practices provided in the literature and by agency directives such as the U.S. EPA's *Guidelines for Preparing Economic Analyses* (2010).

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Chapter 7

Valuing Longevity and Health

The support of human life is one of the basic services provided by the environment. Changes in the life support capacity of the environment brought about by, for example, pollution of the air or water can lead to increases in the incidence of disease, impairment of daily activities, and perhaps reduction of life expectancy. Human alteration of the environment can affect health through a number of channels, including:

- organic compounds, which may contaminate aquifers used as sources of drinking water;
- poorly treated sewage or septic tank leachate, which may spread disease-causing bacteria and viruses into drinking water supplies and among shellfish bound for human consumption;
- elevated levels of particulate matter air pollution, which can increase the risk of premature mortality;
- air emissions from manufacturing facilities, auto body repair and painting shops, and the like, which may include carcinogens; and
- climate change induced by greenhouse gas emissions that can cause increased periods of heat and associated heat-related illness and premature mortality.

The purpose of this chapter is to describe and evaluate the currently available methods and techniques for estimating the monetary values of changes in human health that are associated with environmental changes.[1] Two links must be established in estimating these values. The first is the link between the environmental change and the change in health status; and the second is the link between the change in health status and its monetary equivalent, willingness to pay or willingness to accept compensation. There are two alternative strategies for using revealed preference methods to value environmental changes that affect human health. The first strategy is to develop a comprehensive model

1 The authors are indebted to Maureen Cropper for her contributions to this chapter in the first edition of this book, especially the portions dealing with morbidity benefits and the life-cycle models for valuing changes in mortality risks.

of individual behavior and choice in which environmental quality is one of the determining variables. Such models can provide a basis for measuring willingness to pay directly as a function of the environmental change. The second strategy is to deal with the two links separately. Economic values of changes in health status or health risk would be derived first. They would then be combined with independently derived predictions of health changes or risk changes as a function of environmental change. An example of this approach would be to use measures of the value of risk reduction derived from studies of wage rates and occupational risk, combined with epidemiological studies of the relationship between air pollution and mortality rates, to estimate the economic value of an air quality improvement that reduced the risk of premature mortality.

In the first section of this chapter, models and measurement techniques for valuing changes in mortality as measured by the probability of dying are described. In the second section, revealed preference models based on averting and mitigating behavior and household production are reviewed as a means to valuing reduced risk of morbidity and illness. In the third section, some broader issues concerning valuation of changes in health, including measures of health status such as "quality-adjusted life years" (QALYs) and the valuation of changes in the health of children are considered.

Valuing Reduced Mortality Risks

Because some forms of pollution may increase mortality, economists have had to confront the question of the economic value of reducing the risk of premature death. This area of nonmarket valuation has been prone to serious misunderstanding, and consequently, unnecessary controversy, particularly when the values are discussed in the popular press. The root of the problem is that the value of risk reduction is often reported in conjunction with the number of lives that a policy can be expected to save over the long run. This in turn gives the appearance that economists are placing value (prices!) directly on human lives, as if buying and selling human life were acceptable. However, this interpretation of valuing risk reduction is wrong; it is of utmost importance to recognize that what is being valued is the change in the risk of premature death, a fundamentally probabilistic and ex ante concept. Economists are not valuing the life of a specific individual or group of individuals, an ex post concept. When seen from the view of valuing a change in risk, it is straightforward to see that individuals in their day-to-day actions, and governments in their decisions about social policy, do in fact make tradeoffs between changes in the risk of premature death and other goods that have monetary values. These tradeoffs make it possible to infer the implicit prices that people (or their governments) are attaching to changes in the probabilities of their deaths. For example, when a family chooses to save money by driving to Disneyworld, they have implicitly decided to accept a higher risk of accidental death relative to flying. This observed risk–money tradeoff tells us

about their willingness to accept a small increase in risk in exchange for money (i.e., other goods and services).

The subject of this section is the economic theory of value as it is applied to the decisions made by individuals and governments concerning the tradeoff between mortality risk and money. To begin, a brief description of the economic basis for assigning monetary values to certain kinds of life-saving activity is described, as are some of the important economic and ethical issues that have been identified in the literature. More detailed discussions of models and methods for measuring willingness to pay for reduced mortality risk are then presented. A review of some of the issues that arise in using willingness-to-pay measures in the evaluation of environmental policy concludes this section.

Willingness to Pay

In keeping with the assumption that individuals' preferences provide a valid basis for making judgments concerning changes in their economic welfare, reductions in the probability of death due to accident or illness should be valued according to what an individual is willing to pay to achieve the reductions or is willing to accept in compensation to forgo the reductions. Schelling (1968) appears to have been the first to propose applying willingness-to-pay concepts in this area. Other early contributors include Mishan (1971) and Jones-Lee (1974, 1976). The use of willingness-to-pay concepts presupposes that individuals treat longevity more or less like any other good. This seems to be a reasonable assumption, at least for small changes in mortality risk. Individuals in a variety of situations act as if their preference functions include life expectancy or the probability of survival as arguments. In their daily lives, they make a variety of choices that involve trading off changes in the risk of death for other economic goods whose values can be measured in monetary terms. As in the trip to Disneyland scenario described above, many examples of people accepting higher risk to save money can be observed in transportation choices. For example, some people travel to work in cars rather than by bus or by walking because of the increased convenience and lower travel time of cars, even though these people increase their risk of dying prematurely. Also, some people accept jobs with known higher risks of accidental death because the jobs pay higher wages (see Chapter 11). In such cases, people must perceive themselves to have been made better off by the alternatives they have chosen; otherwise, they would have chosen some other alternative. When what is being given up (or gained) can be measured in dollars, the individual's willingness to pay (or compensation required) to accept higher risk is revealed by these choices. These choices are the basis of measures of the economic value of reductions in the risk of death.

As the ordinary, everyday nature of these examples should make clear, the economic question being dealt with here is not about how much an individual would be willing to pay to avoid his or her certain death or how much compensation that individual would require to accept that death. Most people would be willing

to pay their total wealth to avoid certain death; and there is probably no finite sum of money that could compensate an individual for the sure loss of life. Rather, the economic question is about how much the individual would be willing to pay to achieve a small reduction in the probability of death during a given period or how much compensation that individual would require to accept a small increase in that probability. This is an appropriate question to investigate because most environmental regulatory programs—even those aimed at fairly serious toxins—result in relatively small changes in individuals' mortality risks. Readers interested in critiques of the economic approach to valuation are directed to Broome (1978).

For this kind of situation, the theory of individual choice under uncertainty described in Chapter 5 provides a useful analytical framework. Individuals do not know which of several alternative states of the world will exist at some specified date in the future; but they often must make choices affecting their future utility before the future is revealed. Individuals are assumed to assign probabilities to alternative states of the world and to make their choices so as to maximize their expected utility. One aspect of uncertainty about the future is the date of one's death, or to put it differently, whether one will survive or succumb to some hazard during some time interval. Individuals can affect the probabilities of death during present and future periods by the choices they make. The value of a reduction in risk to an individual is the amount of money that person would be willing to pay to achieve it, other things being equal. As discussed thus far, the willingness-to-pay approach focuses on the individualistic dimensions of human behavior—that is, an individual's willingness to pay to increase his or her own life expectancy. However, there is nothing in the logic of the willingness-to-pay approach to prevent consideration of the effects of kinship and friendship—that is, an individual's altruistic willingness to pay to reduce the probability of death of close relatives and friends. Again, be clear that the question is not the willingness to pay to prevent an imminent or highly probable death of another person (that is, the ransom for the kidnaped child or the search for the lost hiker or boater) but the willingness to pay for a small reduction in the probability of death for the group of which the friend or relative is a part. This would be a form of paternalistic altruism.

A Shorthand Measure: the Value of a Statistical Life

When evaluating policies that reduce the risk of death, economists and policy analysts found it convenient to aggregate the risk reductions experienced by many different individuals into a single measure, referred to as the value of a *statistical life* (VSL) or the value of a *statistical death avoided*. A statistical life can be thought of as the sum of enough risk reductions so that just 1.0 premature death is avoided; and the value of a statistical life is the willingness to pay to achieve that amount of risk reduction. For example, suppose that there were a group of 10,000 people, each of whom has a probability of .0004 of dying during the next year. Suppose a pollution control policy would reduce that probability to .0003, a change of .0001 (1 in 10,000). Furthermore, suppose that each individual in that group expresses

a willingness to pay $500 for this policy. Since the policy would affect all of the people equally, it is a form of collective good for the group. The total willingness to pay of the group is $5 million. If the policy is adopted, there will be on average one less death during the year. Thus, the total willingness to pay for the policy resulting in one less death is $5 million. This is the value of statistical life.

The confusion and misunderstanding generated by the terminology "value of a statistical life" (or its even shorter abbreviation "value of life") has led a number of authors to call for changing the nomenclature associated with the concept. Cameron (2010), in her paper entitled "Euthanizing the Value of a Statistic Life" made a compelling case for permanently discarding this term. She noted that nonspecialists all too easily interpret the term to be a monetary value placed ex post on a life (even specialists can occasionally fall into this trap). Further, she argued that routinely aggregating the risk reductions achieved by a policy to the level of one statistical life is often confusing and unnecessary. Cameron suggested that the profession adopt the expression "willingness to swap (WTS) alternative goods and services for a microrisk reduction in the chance of sudden death (or other types or risks to life and health)" (Cameron 2010, 162–163).

The idea of changing terminology has also gained traction within the U.S. EPA. The Agency's preferred alternative is "value of mortality risk" (VMR) which it suggests be reported in "units using standard metric prefixes to indicate the size of the risk change and the associated time scale, for example, $/μr/person/yr (dollars per micro [10^{-6}] risk per person per year)" (U.S. EPA 2010a, 16). In its review of the EPA white paper suggesting this change, the Science Advisory Board strongly endorsed the value of changing the title, but preferred "value of risk reduction" (U.S. EPA 2010c). While the jury may still be out on the best replacement wording, there is strong consensus that a change is needed. However, since the VSL terminology is so deeply ingrained in the literature, it is not likely that a movement away from it can occur quickly. In this book, the VSL term will be used when referring to past research in order to avoid confusion, but when possible, its use will be avoided.

Human Capital as a Measure of Welfare?

One early concept of the economic value of an individual life is what that individual produces in the way of marketed goods and services in their lifetime and that this productivity is accurately measured by earnings from labor. This argument suggested that earnings before taxes reflect the government's, and therefore society's, interest in each individual's total productivity. With the death of the individual, that output is lost. This approach has a long tradition; in fact, Landefeld and Seskin (1982) trace the idea back almost 300 years. It has been the basis of some widely cited early estimates of the benefits of air pollution control (for example, Lave and Seskin 1970, 1977).

However, the human capital approach is fundamentally at odds with the individualistic perspective of welfare economics and the theory of value. In effect,

by asking what the individual is worth to society, the human capital approach ignores the individual's own well-being, preferences, and willingness to pay. Furthermore, it defines the social worth of the individual in a very narrow way; that is, as the individual's market productivity, and thereby ignores the value of that person's health and well-being to loved ones. Although the human capital approach is inappropriate for valuing reductions in the risk of death, it may make economic sense as the starting point for determining compensation for dependents in wrongful death settlements since these are ex post situations.

Although the human capital approach is flawed in principle as an economic measure of welfare change, one could ask if it might be a reasonable approximation to the value of statistical life based on willingness to pay. Unfortunately, both theoretical reasoning and empirical evidence (both presented below) suggest that human capital measures are a poor proxy for the desired willingness-to-pay measure of value for small changes in the risk of death. While it is likely that an individual's income and the consumption it allows are positively related to the utility that person derives from his or her own life, the human capital method does not reflect the probabilistic nature of death and death avoidance and individuals' differing attitudes toward risk. By definition, an individual with no financial wealth could pay no more than the present value of his expected earnings stream to avoid certain death. However, his statistical value of life based on willingness to pay for small probability changes could be several times his discounted earnings stream.

Modeling Individual Choice and Willingness to Pay

In this section, some of the results generated by economic models of individual choice under uncertainty are described. The purposes of analyzing the models here include deriving hypotheses about the determinants of an individual's willingness to pay for a reduction in the risk of mortality, and examining the relationships between these predicted willingness-to-pay measures and other economic variables such as the individual's earnings and purchases of life insurance.

These models are based on the assumption that, as discussed earlier, individuals make choices that affect their risk of death so as to maximize the mathematical expectation of utility. This means that a key assumption of the models is that individuals know the relevant probabilities of dying and know how they are changed by the choices they make. Since individuals must choose alternatives before the uncertainty is resolved, expected utility is an ex ante concept. These models are used to derive the willingness to pay (WTP) for a reduction in the probability of death, defined as the maximum sum of money that can be taken from the individual ex ante without leading to a reduction in that person's expected utility. This is a compensating surplus measure of welfare change. Alternatively, these models could be used to derive the willingness to accept compensation for an increase in the probability of dying, defined as the sum of money that just compensates for the greater risk by increasing consumption sufficiently to equalize the expected utilities of the two alternatives.

Two aspects of the choice problem are of interest in this analysis: uncertainty and time. The uncertainty concerns the unknown timing of the individual's death and the effects of choices of occupation and consumption activity on the probabilities of surviving to any given date. The time aspect concerns the effects of choices made at any given time on the probabilities and utilities associated with future periods. This is of particular relevance for environmental problems, since many of the most important environmental policy issues are characterized by a substantial interval of time between exposure and the perceived effects on health. Most models of intertemporal choice have focused on identifying the optimal consumption stream given an income stream and opportunities for borrowing and lending. The concern here is with choices made in the present that affect the probabilities of survival at future dates. The problem is to identify the marginal willingness to pay now for increases in the probabilities of survival during some future period.

Static Models

To begin, a simple one-period choice model is developed. The marginal willingness to pay for increases in survival probability is identified. Then the model is extended to the intertemporal case to see whether estimates of the value of statistical life based on intra-temporal models can be applied to the intertemporal case.

Assume that an individual derives utility from the consumption of a composite good, z, with a price normalized to one. The initial endowment of z and the probability of surviving to enjoy its consumption, π, are both given to the individual. Let z^0 and π^0 represent this initial endowment. Arbitrarily normalizing the utility function so that the utility of death is zero, expected utility is

$$E[u] = \pi^0 \cdot u\left(z^0\right). \tag{7.1}$$

For an expression for the individual's marginal willingness to pay in units of z (MWTP) for a reduction in π^0 take the total differential of equation (7.1) and set it equal to zero:

$$MWTP = \frac{u\left(z^0\right)}{\pi^0 \cdot \partial u / \partial z^0}. \tag{7.2}$$

As equation (7.2) shows, one of the fundamental results of models of this sort is that if individuals are not free to adjust their survival probabilities in the market, their marginal willingness to pay for enhanced survival will depend on their initial survival situation. Other things being equal, the higher is the risk of death (lower π^0), the higher will be the marginal willingness to pay to reduce that risk. Another result is that diminishing marginal utility of z implies a higher marginal willingness to pay for changes in π with higher initial endowments of z, other things being equal. These propositions can be verified by taking the derivatives of equation (7.2) with respect to π and z.

This model, while providing useful insights, does not suggest a method for estimating WTP other than by stated preference. To use revealed preference

methods we must link WTP to voluntary risk-taking behavior—that is, π must be made endogenous. Now suppose that the individual has the opportunity to rearrange his consumption and survival position through exchange, for example by giving up some z in order to improve his chances of surviving to enjoy the remainder. Let p_π represent the price at which consumption can be exchanged for enhanced survival probability. There is assumed to be no opportunity for insurance. Comprehensive models encompassing bequest motivation (utility derived from unconsumed z remaining at death) and insurance behavior can be developed. For examples, see Jones-Lee (1976), Conley, (1976), Thaler and Rosen (1976), and Cropper and Sussman (1988).

The individual chooses z and π so as to maximize expected utility, subject to the budget constraint

$$\text{Max}: E[u] = \pi \cdot u(z) + \lambda\left\{z^0 - z + p_\pi \cdot \left(\pi^0 - \pi\right)\right\}. \tag{7.3}$$

The first-order conditions for a maximum of expected utility can be combined to obtain

$$\frac{u(z)}{\pi \cdot \partial u / \partial z} = p_\pi . \tag{7.4}$$

This expression requires that the individual equate her marginal willingness to pay for enhanced survival (the left-hand side of the expression) with the given price of enhanced survival. This is similar to the expression for marginal willingness to pay for reductions in risk derived in Chapter 5, since here utility and the marginal utility of consumption are zero in the event of death.

The utility-maximizing individual may choose either to forgo consumption in order to enhance her survival probability or to take on increased risk (lower π) in order to enhance consumption opportunities. The actual choices depend on the initial endowments of z^0 and π^0, the price of π, and the individual's preferences. Whatever the final outcome, equation (7.4) allows us to infer the individual's marginal willingness to pay for enhanced survival, since this value will be equated to the observable price of changes in π.

If the risk in question is an environmental risk, it may not be feasible to have a market for reductions in π. However, if there are other risks, what is required is that the individual be able to affect the level of one of these risks and view equal size reductions in any of these risks as equally valuable, or to put it differently, be indifferent as to source of the risk of death. Suppose now that there are three sources of risk of death, one exogenous environmental risk, one job-related risk, and one related to the level of consumption of a private good x_i, an element in the vector of market goods $\boldsymbol{X}$. Assume that in addition to affecting the risk of death, this good conveys utility directly. The corresponding conditional probabilities of death are denoted ρ_e, ρ_j, and $\rho_x(x_i)$. Assuming that these causes of death are independent, the probability of surviving the current period is the product of the probabilities that the individual does not die from each of the three causes (Sussman 1984); that is,

$$\pi = (1-\rho_e)(1-\rho_j)[1-\rho_x(x_i)]. \tag{7.5}$$

In general, good x_i could either increase or decrease the risk of death. Some goods such as skydiving and cigarettes increase risk, while others such as smoke detectors decrease risk. Our interest is in risk-reducing goods, so that $\partial\rho_x/\partial x_i < 0$. Also, suppose that all types of jobs are alike in every respect except for the risk of accidental death, and that riskier jobs have higher wage rates—in other words, the individual receives an annual wage, $M(\rho_j)$, where $\partial M/\partial\rho_j > 0$.

If I is exogenous income, then total income, M^*, is $I + M(\rho_j)$. Expected utility is given by

$$E(u) = (1-\rho_e)(1-\rho_j)[1-\rho_x(x)]\cdot u[M^*(\rho_j) - p_x \cdot x_i, X]. \tag{7.6}$$

By total differentiation of equation (7.6), willingness to pay for a marginal change in exogenous risk of death, $dI/d\rho_e$, is given by

$$\left(w_{\rho_e} = (1-\rho_j)(1-\rho_x)\frac{u(\cdot)}{\pi\dfrac{\partial u}{\partial M^*}} \right). \tag{7.7}$$

This is the value of the utility lost if the individual dies $u(\cdot)$, converted to dollars by dividing by the expected marginal utility of income ($\pi \cdot \partial u/\partial M^*$), and multiplied by the probability that the individual does not die due to other causes $[(1-\rho_j)(1-\rho_x)]$.

The major question of interest is whether MWTP can be estimated by observing risk-taking behavior in consumption or in the labor market. To answer this, assume that the individual chooses x_i and a job with its associated risk ρ_j, so as to maximize equation (7.6) and derive the first-order conditions for choice of job risk and consumption of x_i. They are

$$\frac{\partial M}{\partial \rho_j} = (1-\rho_e)(1-\rho_x)\frac{u(\cdot)}{\pi\dfrac{\partial u}{\partial M^*}} \tag{7.8}$$

and

$$-\frac{p_x}{\partial\rho_x/\partial x_i} = (1-\rho_e)(1-\rho_j)\frac{u(\cdot)}{\pi\left[\dfrac{\partial u}{\partial M^*} - \dfrac{\partial u/\partial x_i}{p_x}\right]}. \tag{7.9}$$

Equation (7.8) implies that the individual equates the marginal wage income forgone by moving to a safer job with the marginal benefit of a reduction in job risk. The latter is almost identical to the value of an exogenous risk change—equation (7.7)—except that the probability of not dying due to other causes is now $(1-\rho_j)(1-\rho_x)$ instead of $(1-\rho_e)(1-\rho_x)$. If $(1-\rho_e) \approx (1-\rho_j)$, then the marginal price of risk reduction can be used as an estimate of the willingness to pay

for a change in exogenous risk. This marginal price, which is an implicit price, can be estimated with a hedonic wage model. That method is described in Chapter 11.

The first-order condition for the choice of x_i differs from the expression for marginal willingness to pay in part because it includes a term for the marginal utility of a dollar spent on x_i. This term is present because of the assumption that x_i conveys utility directly as well as through its effect on risk. Equation (7.9) can be used as an approximation for marginal willingness to pay only if $\partial u/\partial x_i = 0$. Then, if $(1-\rho_e) \approx (1-\rho_x)$, the left-hand side of (7.9), which is the marginal cost of reducing risk through purchasing x_i, can be used to approximate MWTP for an exogenous risk change.

In order to use data on "safety" goods to estimate the value of risk reduction, one must establish that the good's contribution to safety is known. What governs each individual's purchase decision is that person's perception of the risk-reduction capability of the good. In the absence of adequate information, individuals' perceptions might vary substantially and be difficult to observe. This would make the use of safety good purchases for estimating values of risk reduction problematic.

The revealed preference model of consumer choice is based on the assumption that the safety good is divisible; however, this assumption is often invalid. Some of the goods used in actual studies—for example, smoke detectors—are indivisible; that is, their purchase involves a 0–1 decision. The good is purchased if its marginal benefit is equal to or greater than its marginal cost. The equality of willingness to pay and price occurs only for the marginal purchaser of the good. In order to estimate an average value of MWTP, we must have data on the cost of the safety good and on its effect in reducing risk of death for a cross-section of individuals. If marginal cost and marginal risk reduction vary across individuals in the sample, we can estimate the average value of MWTP by using a discrete choice model as described in Chapter 3.

Perhaps the most important conclusion to be drawn from this review of static models of individual choice and willingness to pay is that each person is likely to attach a different value to a small reduction in the probability of dying because of differences in underlying preferences, degree of risk aversion, wealth, the current level of risk exposure, age, the number of dependents, and, perhaps, the quality of life-years expected to be gained from the reduced risk. This conclusion must be kept in mind when interpreting and using the results of empirical estimates of willingness to pay that are based on averages of groups of perhaps quite heterogeneous people.

A second conclusion is that in the case of multiple risks of death where the individual can "purchase" reductions in some component of risk, the observed price or marginal cost of reducing that component of risk can be taken as a close approximation of the individual's willingness to pay for reductions in other components of risk, provided that the safety good or safer job does not also convey utility directly and that the individual values equal reductions in all components of risk equally.

Introducing Time

The static models discussed above are relevant to questions such as transportation and occupational safety in which the individual's actions today affect the probabilities of dying today. However, many of the important environmental and occupational health questions involve actions taken today whose effects on the probability of dying are realized only at some time, perhaps 10 to 20 years, in the future. How much would an individual be willing to pay now to control current pollution when the effects of improved health might be realized only at some future time? The next step in the analysis is to develop a simple multiperiod model that allows us to investigate how willingness to pay now for a reduction in the probability of death is influenced by the time period to which the probability applies. To focus on the intertemporal aspect of the problem, assume only one cause of death and one consumption activity.

As an extension of the model of intertemporal choice developed in Chapter 6, assume that individuals maximize their expected lifetime utility u^*, where u^* is an additively separable function of the consumption stream:

$$E\left[u^*\right] = \sum_{t=1}^{T} \pi_t \cdot D^{t-1} \cdot u\left(z_t\right), \tag{7.10}$$

where $D = 1/(1 + d)$,
$d \equiv$ subjective rate of discount or own time preference, and
$\pi_t \equiv$ probability of surviving from period one through period t. That is, the probability of living for at least t years from now (year one), with

$$\pi_t = \prod_{s=1}^{t} \left(1 - \rho_s\right). \tag{7.11}$$

The term ρ_s is the conditional probability of dying during year s; that is, the probability of dying in year s given that the individual has survived to the beginning of year s. Similarly, π_s is the probability of surviving the year given being alive at the beginning of the year. For now, assume that the stream of consumption z_t is given exogenously and cannot be altered by borrowing or lending.

The form of the intertemporal utility function deserves a brief comment. As explained in Chapter 6, the assumption that individuals maximize discounted utilities does not necessarily impose any restrictions on the nature of individuals' preferences regarding present versus future consumption. The subjective rate of discount d could be positive, negative, or zero. A variety of types of preferences and behavior can be encompassed in this model, depending on the value of d. In equilibrium, borrowing and lending behavior depends on market interest rates, present and future income levels, and the rate at which the marginal utility of income is diminishing, as well as time preference.

From equation (7.10) the marginal rate of substitution (MRS) between present consumption and present mortality reduction is

$$MRS_{\pi_1 z_1} = \frac{u(z_1) + \sum_{t=2}^{T} \frac{\pi_t}{\pi_1} D^{t-1} \cdot u(z_t)}{\pi_1 \cdot \partial u / \partial z_1}. \tag{7.12}$$

The second term in the numerator shows the dependence of willingness to pay to reduce present risk on life expectancy and the associated stream of expected future utilities. Higher future consumption (z_t) and longer life expectancy (T and π_t) both increase the willingness to pay now for a higher current survival probability.

Suppose now that the marginal rate of substitution between present mortality reduction and present consumption is known, for example, from a study of the demand for occupational safety. Can this marginal rate of substitution be used to estimate the demand for reductions in future mortality that might be obtained through an environmental or occupational health program? The marginal rate of substitution between a mortality reduction in future period t' and present consumption is

$$MRS_{\pi_{t'} z_1} = \frac{\sum_{t=t'}^{T} \frac{\pi_t}{\pi_{t'}} D^{t-1} \cdot u(z_t)}{\pi_1 \cdot \partial u / \partial z_1}. \tag{7.13}$$

The two marginal rates of substitution might differ for any of three reasons. First, equation (7.12) could be greater than (7.13) because it includes a term for first-period utility, $u(z_1)$. Also, the more distant that period t' is from the present, the smaller the stream of future periods being summed in (7.13). However, equation (7.13) could be greater than (7.12) if $\pi_{t'}$ is sufficiently smaller than π_1. Thus, marginal rates of substitution estimated from one type of probability choice problem—for example, the single-period problem—cannot, in general, be used as predictors of the marginal rates of substitution for other types of probability choice problems with different intertemporal dimensions.

In this model, the individual takes the intertemporal pattern of consumption opportunities as given. There is no borrowing or lending. Cropper and Sussman (1990) have extended this model by incorporating borrowing and lending at a riskless interest rate. This allows individuals to adjust their consumption streams over time. Measures of willingness to pay will be different when this opportunity is available to people. To see this, we turn now to a full life-cycle model of intertemporal choice.

A Life-Cycle Model of Willingness to Pay

Several authors, including Usher (1973), Conley (1976), Cropper and Sussman (1990), and Rosen (1994), have used a life-cycle consumption-saving model with uncertain lifetime to analyze an individual's willingness to pay at age j for a change in the conditional probability of dying at age t, $t \geq j$. In this section, the approach of Cropper and Sussman is followed. This model can be used to examine:

- the relationship between WTP for a change in current probability of death and age;
- the relationship between the present value of expected lifetime earnings (the human capital measure) and WTP;
- the relationship between WTP and the latency of the risk, that is, the interval between the exposure to the risk (a carcinogen, for example) and its manifestation (death due to cancer); and
- the relationship between the willingness to pay in advance to reduce a given risk and the willingness to pay at the time the risk is experienced (a discounting question).

Using this approach shows that: (a) WTP will generally decrease with age; (b) under plausible circumstances, the present value of lifetime earnings will be less than WTP; (c) latency reduces WTP; and (d) the willingness to pay at year one for a reduction in risk in year t' is equal to the willingness to pay in year t' for that probability of change discounted back to year one by a discount factor, which in general will be different from the market interest rate.

In the life-cycle model, an individual of any given age has a probability distribution over the date of his or her death. Let j denote the individual's current age; and let $\rho_{j,t}$ be the probability that the individual dies at the end of the year in which he attains age t; that is, the person lives exactly $t-j$ more years. Since the $\{\rho_{j,t}\}$ constitutes a probability distribution, it must be true that

$$\rho_{j,t} \geq 0, t = j, j+1, \ldots, T, \text{ and that } \sum_{t=j}^{T} \rho_{j,t} = 1, \tag{7.14}$$

where T is the maximum attainable age. The probability that the individual survives to his tth birthday, given that he is alive at age j, is $\pi_{j,t}$, which also is the probability that he dies at $t+1$ or later. Formally,

$$\pi_{j,t} = \sum_{s=t+1}^{T} \rho_{j,s} . \tag{7.15}$$

Let δ_t, be the probability of dying at age t conditional on being alive at the beginning of that year. Thus, the conditional probability of surviving that year is $1-\delta_t$. This term can also be derived from the survival probabilities

$$1-\delta_t = \frac{\pi_{j,t+1}}{\pi_{j,t}}. \tag{7.16}$$

Expected lifetime utility at age j is the sum of the utility of living exactly $t-j$ more years, times the probability of doing so. As in the preceding section, assume that utility is additively separable, and assume that there is no bequest motive. Then we can write expected lifetime utility at age j as

$$V_j = E\left[u^*\right] = \sum_{t=j}^{T} \pi_{j,t} \cdot D^{t-j} \cdot u\left(z_t\right). \tag{7.17}$$

The utility function for each period, $u(z_t)$, is assumed to be increasing in z_t, strictly concave, and bounded from below.

Two points about this formulation of the problem should be emphasized. First, the model is based on the assumption that the utility of living depends only on consumption and not on length of life per se. The concavity of the utility function implies that it is always desirable to spread a given amount of consumption over a longer time span. Thus, lifetime utility is an increasing function of life expectancy. However, this is only because of the effect on consumption, not because of the value of being alive per se. This point is returned to later. Second, this model treats survival probabilities as exogenous to the individual. To keep things simple, no attempt is made to introduce opportunities for the individual to alter risk levels into this model—see Conley (1976) and Viscusi and Moore (1989) for examples of intertemporal models incorporating this additional element of choice.

The individual has to choose a time pattern of consumption, given initial wealth W_j, annual earnings M_t, $t = j, \ldots, T$, and capital market opportunities, so as to maximize expected lifetime utility as given by equation (7.17). Arthur (1981) and Shepard and Zeckhauser (1982, 1984) assume that the individual can save by purchasing actuarially fair annuities and borrowing via life-insured loans. If actuarially fair annuities are available, an individual who invests \$1 at the beginning of his jth year will receive $\$(1 + A_j)$ at the end of the year with probability $1 - \delta_j$ and nothing with probability δ_j. For the annuity to be fair (that is, to have an expected payout of $1 + r$ where r is the riskless rate of interest) there must be an annuity rate of interest a_j that satisfies

$$(1 + a_j)(1 - \delta_j) = 1 + r. \tag{7.18}$$

If the individual borrows, she must cover the possibility that she might die before repaying the loan. Agreeing to pay $1 + a_j$ if she survives is equivalent to paying $1+r$ on survival, plus purchasing a life insurance policy in this amount at actuarially fair rates. Thus, we can call a_j the actuarial rate of interest.

The individual's budget constraint can be expressed as the requirement that the present value of expected consumption equal initial wealth plus the present value of lifetime earnings, where discounting is done at the riskless rate, r:

$$\sum_{t=j}^{T} \pi_{j,t} \cdot (1+r)^{j-t} \cdot z_t = W_j + \sum_{t=j}^{T} \pi_{j,t} \cdot (1+r)^{j-t} \cdot M_t \,. \tag{7.19}$$

Alternatively, making use of equation (7.18) and the fact that

$$\pi_{\mathrm{j,t}} = \prod_{s=j}^{t-1} (1-\delta_s)\,, \tag{7.20}$$

the budget constraint can be expressed in terms of the actuarial rate of interest:

$$\sum_{t=j}^{T} \left[\prod_{s=j}^{t-1} (1+a_s)^{-1} \right] \cdot z_t = W_j + \sum_{t=j}^{T} \left[\prod_{s=j}^{t-1} (1+a_s)^{-1} \right] \cdot M_t \,. \tag{7.21}$$

The pattern of consumption over the life cycle is chosen so as to maximize equation (7.17) subject to (7.19) or (7.21). Formally, it is the solution to the Lagrangian problem:

$$\begin{aligned}\underset{X_t}{\text{Max}}\, V_\lambda &= \sum_{t=j}^{T} \pi_{j,t} \cdot (1+d)^{j-t} \cdot u(z_t) \\ &\quad + \lambda \left[W_j + \sum_{t=j}^{T} \pi_{j,t} \cdot (1+r)^{j-t} \cdot (M_t - z_t) \right].\end{aligned} \tag{7.22}$$

Now consider how a government health and safety regulation that reduces conditional probabilities of death affects lifetime utility. A government regulation can alter the probability that a person dies in any year only if that person is alive at the beginning of the year. Consider a regulation that reduces $\delta_{t'}$, the conditional probability of dying at age t'; that is, the probability that the individual dies between his t'th and $(t'+1)$st birthdays. Note that when the conditional probability of death is altered at age t', it affects the probabilities of surviving to ages $t'+1$ and beyond (for example, $\pi_{j,t'+k}$) since, by repeated use of the definition of δ_t,

$$\pi_{j,t'} = (1-\delta_j)(1-\delta_{j+1})\cdots(1-\delta_{t'-1}). \tag{7.23}$$

Formally, let $w_{j,t'}$ be the individual's marginal willingness to pay at age j for a change in $\delta_{t'}$. It is measured by the wealth that must be taken away from that person at age j to keep her expected utility constant, given the reduced risk of death, or

$$w_{j,t'} \equiv \frac{dW_j}{d\delta_{t'}} = -\frac{dV_j / d\delta_{t'}}{dV_j / dW_j}. \tag{7.24}$$

The envelope theorem implies that

$$\frac{dV_j / d\delta_{t'}}{dV_j / dW_j} = \frac{\partial V_\lambda / \partial \delta_{t'}}{\partial V_\lambda / \partial W_j}. \tag{7.25}$$

Thus, using equation (7.17), we can express marginal willingness to pay as

$$w_{j,t'} = (1-\delta_{t'})^{-1} \sum_{t=t'+1}^{T} \pi_{j,t} \cdot \left[\begin{array}{l} (1+d)^{j-t} \cdot \dfrac{u(z_t)}{\lambda} + \\ (1+r)^{j-t} \cdot (M_t - z_t) \end{array} \right]. \tag{7.26}$$

Willingness to pay at age j for a reduction in the conditional probability of death at age t' equals the gain in expected utility from year t' onward, converted to dollars by dividing by the marginal utility of income in year j, λ, and discounted at the individual's own rate of time preference. Added to this is the effect of the change in $\delta_{t'}$ on the budget constraint. A reduction in $\delta_{t'}$ makes the individual wealthier by increasing the present value of her expected lifetime earnings from age $t'+1$ onward. However, an increase in survival probabilities also decreases the

consumption that the person can afford in each of the years $t' + 1$ through T from a given earnings stream. Thus, her willingness to pay is reduced by the change in the present value of the consumption stream.

It should be noted that $w_{j,t'}$ is the rate at which the individual is willing to trade wealth for a unit change in risk. To compute the dollar value of a small change in risk, equation (7.26) must be multiplied by the magnitude of the risk change. For example, if $w_{j,t'} = \$2 \times 10^6$ but the change in risk is only 10^{-6}, then willingness to pay for the risk change is \$2.

There are a number of points to be made about willingness to pay for reductions in the risk of death on the basis of equation (7.26). First, for a policy that affects the conditional probabilities for dying over a number of years, the total marginal willingness to pay is the sum of the willingness to pay for the changes in each of the $\delta_{t'}$.

Second, a key assumption of models of this type is that expected lifetime utility depends only on expected lifetime consumption, as Linnerooth (1979) has noted. Therefore, what is being calculated is the willingness to pay for the opportunity to continue consumption. As Bergstrom (1982) has pointed out, if the intertemporal objective function is derived from preferences among alternative lotteries, it should include a term that values survival per se. If this term is an increasing function of the $\{\pi_{j,t}\}$, any willingness-to-pay measure derived from equation (7.17) must be regarded as a lower bound to true willingness to pay. This condition was first noted in a static context by Conley (1976) and Cook (1978).

Third, even granting the assumption that utility depends only on consumption, as long as the individual's average utility of consumption exceeds his marginal utility, willingness to pay exceeds human capital, and human capital must be interpreted as a lower bound to willingness to pay. By use of the first-order conditions for utility maximization, the term in brackets in equation (7.26) can be written as

$$(1+r)^{j-t}\left(\frac{u(z_t)}{\partial u / \partial z_t} - z_t + M_t\right). \tag{7.27}$$

This implies that if $u(z_t)/[\partial u/\partial z_t] - z_t > 0$ for all t, then each year's contribution to $w_{j,t'}$ as given by equation (7.27) exceeds that year's contribution to the present value of lifetime earnings, $(1 + r)^{j-t} \cdot M_t$, and willingness to pay must exceed the present discounted value of lifetime earnings. As noted by Conley (1976) and Cook (1978), the condition that $u(z_t)/[\partial u/\partial z_t] - z_t > 0$ implies that the average utility of consumption exceeds its marginal utility, a condition that holds for all increasing, concave utility functions, provided consumption exceeds a subsistence level.

Blomquist (1981), expanding on Linnerooth's analysis, reviewed the results of nine empirical estimates of willingness to pay and showed that the implied value of statistical life was typically much larger than the expected lifetime earnings of the members of the sample population used in each study. His analysis lends strong support to the assertion that values of life based on lifetime earnings are a poor proxy for the willingness to pay for reduced risk of mortality.

Fourth, equation (7.26) implies that as t' (the age at which risk of death changes) increases, $w_{j,t'}$ must decline, at least as long as the individual is above subsistence. This implies that the value of reducing a person's current probability of dying must always be greater than the value of reducing exposure at age j to a carcinogen with a latency period of $t' - j$ years. In the latter case, fewer expected life-years are saved. Furthermore, the longer the latency period, the smaller willingness to pay is, other things being held constant. This means that willingness-to-pay measures based on behavior toward contemporaneous risk (for example, from studies of wages and occupational accident mortality) will not be good proxies for willingness to pay to reduce latent risks.

Cropper and Sussman (1990) have shown that there is an alternative approach to dealing with the latency problem. Suppose we have obtained a measure of $w_{t',t'}$ for a group at age t by examining, for example, the group members' tradeoffs between wages and contemporaneous job risks. Cropper and Sussman have shown that $w_{j,t}'(j < t')$ can be calculated from the following expression:

$$w_{j,t'} = \prod_{s=j}^{t'} (1 + a_s)^{-1} \cdot w_{t',t'} . \tag{7.28}$$

That is, the willingness to pay in advance for a reduction in risk in t- is the contemporaneous willingness to pay discounted back by a factor derived from the actuarial interest rates over the interval. Recall that these actuarial interest rates are a combination of the riskless interest rate and conditional probabilities of death. As such, they vary over time for the individual and across individuals, depending on age and other factors.

The fifth point to be made is that it is interesting to see how $w_{j,j}$ varies over the life cycle. Shepard and Zeckhauser (1982, 1984) have used an expression like equation (7.27) to examine this question. If consumption were constant for all t, as would be the case if the riskless rate of interest were equal to the subjective rate of time preference, $w_{j,j}$ would decline monotonically with age. Younger persons would always have a higher WTP to reduce current risk of death than older persons because there would be more years of consumption and utility in the summation. If, however, consumption increases over some portion of the life cycle, $w_{j,j}$ may also increase with age up to some point, and then decline. If, for example, the individual cannot be a net borrower but can lend at the riskless rate of interest, her consumption is likely to be constrained by income at the beginning of her life. This will cause the present value of the utility of consumption and, hence, $w_{j,j}$ to increase up to some point, and then to decline.

Toward Measurement

The most commonly used revealed preference method for estimating the value of reduced risk of death is the hedonic wage-risk tradeoff approach that was introduced above and is described in more detail in Chapter 11. A number of meta-analyses have been undertaken that summarize estimates of the VSL

generated by wage-risk tradeoff studies from the literature. In an early study, Viscusi (1993) examined estimates of the VSL for U.S. populations. More recent meta-analyses that provide estimates for the U.S. include Miller (2000), Mrozek and Taylor (2002), Viscusi (2003), Kochi, Hubbell, and Kramer (2006), and the U.S. EPA (2010c). Cropper, Hammitt, and Robinson (2011, 321) summarized the central tendency of the estimates reported by the first four of these studies, indicating that these range from a low of $2–$3.3 million up to $11.1 million (in 2009 USD).

Values for risk reduction might also be revealed by choices regarding purchases of goods that reduce mortality risks. For example, smoke detectors and seat belts are goods whose primary purpose is to produce safety; that is, to reduce the risk of death for those who purchase them. Data on the purchase and utilization of these goods have been used to estimate the values of reducing risk of death. Some results are reviewed in Viscusi (1993). Safety could also be one of the characteristics of a differentiated product like an automobile. Different automobile models have measurable differences in accident rates (probability) and the severity of injury. If these differences are systematically related to the prices of different models of automobiles, then the hedonic price model described in Chapters 4 and 10 can be applied to estimate individuals' willingness to pay for reductions in the risk of accident or death. For an example of this approach based on U.S. data, see Atkinson and Halvorsen (1990).

Stated preference methods have also been increasingly used to estimate the value of risk reduction and VSLs. These methods are attractive in that they permit researchers to investigate the relationships between WTP and such variables as age at risk, income, health status, cause of death, the level of baseline risk, whether the risk is voluntarily encountered or not, and the size of the risk reduction. Stated preference studies also allow the analyst to provide the context to respondents that best fits the policy situation in which the risk change will occur. For example, if the value of risk reduction related to lowering mortality from a regulation that will lower the average exposure to a toxic pollutant is sought, a stated preference survey can target the affected population (e.g., elderly, urban residents, etc.) and can describe the salient aspects of the proposed risk change (e.g., latency, size of the risk reduction, etc.).

Cropper, Hammitt, and Robinson (2011) identified three recently published meta analyses of stated preference VSL estimates (Kochi, Hubble, and Kramer 2006; Dekker et al. 2011; Lindhjem, Navrad, and Braathen 2010). Their estimates of the central tendency of VSL range from $2.7–$8.5 million (also in 2009 USD). Note that this range of estimates is below the range reported from the revealed preferences studies summarized above. The fact that the estimates from stated preference methods generally lay below those from revealed preference methods remains one of the puzzles of this literature (Cropper, Hammitt, and Robinson 2011).

Using the Value of Risk Reduction and VSL in Policy Evaluation

The benefit of a policy that reduces the risk of premature death for a specified group of people is the sum of the individual WTPs for the reduction in risk of all of the members of the group. These WTPs could vary across the group for a variety of reasons including differences in their age, income, health status, cause of death, and the level and type of baseline risk. However, historically the practice of environmental and safety policy has been to ignore these differences and to use a VSL based on the WTP of the average individual in the group that provided the data for the WTP estimate. For example, if the source of data were a hedonic wage regression, the sample mean values for all of the independent variables would be used to compute the WTP of the sample mean individual. This in turn would provide the basis for computing the VSL for the sample.

In the United States, this practice has been sanctioned for all federal agencies by the Office of Management and Budget (U.S. Office Management and Budget 2000) and the Environmental Protection Agency (EPA) (U.S. Environmental Protection Agency 2010b). EPA uses a VSL of $7.4 million in 2006 dollars. This figure is based on an analysis of 5 stated preference and 21 wage-risk studies in which a Weibull distribution was fitted to the VSLs and the mean of the Weibull distribution was calculated. As acknowledged in the *Guidelines for Preparing Economic Analyses* (U.S. EPA 2010b, Appendix B) these values are in need of updating (all of the studies from which the data are drawn were performed prior to 1991, indicating that they are based on relatively old data and methods).

In the past decade, the practice of using a single value expressed in terms of VSL for all regulations that alter risk has come under increasing criticism for failing to reflect adequately the variety of factors that have been seen to influence individuals' WTP for risk reduction (Sunstein 2004). EPA is working to update their guidance and considering adjustments related to the demographics of the affected populations (age and health status) and the characteristics of the risk. In the latter category, they note that eight different risk dimensions have been considered and shown to have effects on the valuation of risk reduction including whether the risk is involuntary, man-made, controllable, or continuous, among others (U.S. EPA 2010a, 2010b). The fact that EPA and other federal agencies continue to rely on a single value for VSL in the presence of evidence suggesting that these factors can significantly affect the magnitude of the value is problematic (Kniesner, Viscusi, and Ziliak 2010; National Research Council 2008). Recent studies continue to support the importance of differential values across many subpopulations (examples include work by Evans and Smith 2010, Hammitt and Haninger 2010, and Cameron and DeShazo 2013).

Discounting Statistical Lives?

The practice of discounting statistical lives is common in the economic appraisal and evaluation of environmental and safety public health initiatives. The practice allows policymakers to compare policies that affect probabilities of premature death with different time patterns. However, the practice is also controversial from ethical perspectives. Discounting is justified on the basis of two factors. The first is the observation that people tend to prefer present consumption over future consumption (time preference). The second factor is the productivity of capital investments meaning that $1 of resources invested today rather than consumed will make it possible to consume more than $1 at some future time.

The practice of discounting lives saved might be attractive to policymakers because it gives the appearance of making possible the comparison of policies that affect probabilities of premature death with different time patterns and different costs, but without engaging in the controversial practice of assigning monetary values to statistical lives saved. However, at a fundamental level, it is not possible to separate the thing being discounted (lives) from its economic value, especially in the realm of policymaking and choices among alternative policies with different costs. Thus, if economic values should be discounted, then discounting should also apply to statistical lives saved. Discounting of statistical lives saved is a controversial practice. Here, the objection to the practice offered by moral philosophers will be outlined first. Then, two complementary arguments in favor of discounting from economics will be offered.

Speaking explicitly of discounting lives saved, the philosopher Douglas MacLean (1990) has argued that mere differences in the timing of events can have no moral significance. Similar people should not be treated differently solely on account of differences in the timing of their deaths. For example, the premature death of any 20-year-old person today can be considered as no worse than the premature death of a similar 20-year-old 10 years from now. Similarly, if it is better to save 100 lives today than to save 99 lives today, it must also be better save 100 lives 10 years from now than to save 99 lives today.

This is just a special case of a more general point that Broome (a philosopher and economist) attributes to the 19th century philosopher/economist Henry Sidgwick. Broome said:

> [*F*]*rom a universal point of view* the time at which a man lives cannot affect the value of his happiness. A universal point of view must be impartial about time, and impartiality about time means that no time can count differently from any other. In overall good, judged from a universal point of view, good at one time cannot count differently from good at another. Nor can the good of a person born at one time count differently from the good of a person born at another.
>
> (Broome 1992, 92, italics in original)

Thus, whether considering lives or well-being, the amount realized must be given the same moral weight independent of the time of its realization. This is equivalent to saying that in moral terms the discount rate is zero.

Looking at deaths alone, the ethical perspective has validity. However, it abstracts from two important considerations. The first consideration is individual's preferences concerning the timing of the benefits and costs that they experience and that contribute to their well-being. The second is the opportunity cost of resources committed to life-saving policies. Each consideration will be discussed in turn.

As shown in Chapter 6, if individuals wish to maximize their lifetime utility, they must equate their marginal rate of substitution between present and future consumption with the market rate of interest. This is true whether they have a positive, zero, or negative time preference. Furthermore, as long as market goods and nonmarket goods such as health and risk reduction are substitutes in preferences, the marginal utilities of future nonmarket goods should also be discounted. It follows that if policy choices should be based on (or at least reflect) individuals' preferences regarding the benefits and costs they receive, then the benefits and costs of policies including policies that affect the risk of premature death should be discounted.

For example, consider the question of choosing between policy A that prevents x immediate deaths in a group now (e.g., reduces the risks of fatal accidents) and policy B that reduces the same group's exposure to a carcinogen thereby preventing x deaths in, say, 30 years. The group would undoubtedly prefer policy A because of the larger number of life years saved. This would be reflected in their higher willingness to pay now for policy A than for policy B. One way for policymakers to reflect this preference would be to discount the avoided deaths of policy B.

Turning to the second consideration, discounting concerns choices among policies that involve commitments of resources that have opportunity costs; and these opportunity costs can include forgoing other opportunities to reduce mortality at other times. The practice of discounting is a way of bringing these opportunity costs into the decision process. Consider a policy involving costs now that avoids x deaths per year in perpetuity. The undiscounted sum of lives saved is infinite. Without discounting, this would justify an infinite commitment of resources. Surely, however, there are limits on the cost that the present generation is obliged to incur to save future lives. The discounted present value of this stream of lives saved converges on x/r, and the present generation can (must) then decide what cost it is willing to incur to save a finite sum of discounted lives.

What about policy A that prevents x deaths in a group of 40-year-old people now vs. policy B that prevents y ($> x$) deaths in a group of otherwise similar 40-year-old people in 30 years? The ethical perspective claims that policy B is preferred because of the larger number of deaths avoided. However, if the resources used in this policy were instead invested to earn r% for 1 year and then used in a policy similar to B (call it policy C), then y' ($> y$) deaths would be avoided. Thus, C should be preferred to B, and so forth for policies D, E, etc. involving further

postponement of the lifesaving policy. None of the policies would be undertaken since an additional postponement can always increase the undiscounted number of deaths avoided. Discounting avoids this counterintuitive result.

These two lines of argument (time preference and opportunity cost) come together in the following way. In order to identify policies that are Potential Pareto Improvements (PPIs), both the benefits and costs of life-saving policies must be discounted at a rate that reflects the interaction of time preference and opportunity cost—the market interest rate in simple models.

Valuing Reduced Morbidity

Morbidity is a general term that refers to cases of disease or being in less than good health. Morbidity can be classified in a variety of ways, among them duration of the condition (chronic or acute), degree of impairment of activity, or type of symptom. An episode of acute morbidity would last only a matter of days and would have a well-defined beginning and end. Chronic morbidity refers to cases of a longer-term illness of indefinite duration. The degree of impairment could be defined in terms of, for example, "restricted activity days" on which a person is able to undertake some, but not all, normal activities; "bed disability days" on which a person is confined to bed, either at home or in an institution, for all or most of a day; or "work loss days" on which a person is unable to engage in ordinary gainful employment. However, these measures of morbidity reflect responses to ill health rather than the health condition itself. Whether a given clinical manifestation of ill health results in any restriction on activity, bed disability, or work loss depends upon a number of socioeconomic variables, such as employment and labor-force status, non-labor sources of income, the presence of other income-earners in the household, and so forth. Morbidity can also be measured by "symptom days," that is, by the occurrence of specific symptoms such as an asthma attack, a headache, a cough, throat irritation, or diarrhea.

The choice of symptoms for defining and measuring morbidity has implications for the economic valuation of health effects. The economic perspective on health focuses attention on effects that people are aware of and wish to avoid (that is, effects that would reduce their utility). Yet some biomedical clinical research focuses on effects of questionable significance to individuals and measures health effects that are difficult to relate to individual perceptions and behavior. The question of how to define morbidity is tied to a legal and policy issue that vexes the U.S. Environmental Protection Agency: What constitutes an adverse health effect? The Clean Air Act of 1970 calls for setting air quality standards so as to protect individuals from *adverse* health effects. The question of whether or not an effect is adverse arises (for example) when clinical studies reveal that exposure to an air pollutant under controlled conditions leads to detectable changes in organ structure or function without necessarily causing pain, impeding people's activities, or reducing life expectancy. Are these changes adverse? From an economic perspective based on the willingness-to-pay definition of value, the

answer depends on whether the changes matter to the individual and whether the individual reveals or expresses a willingness to pay to avoid the effect, or requires compensation to experience it.

Measures of morbidity must also take into account the fact that, unlike mortality, morbidity is not a discrete event but a process involving time. Cases observed during a period of time may fall into one of four categories: (a) onset of morbidity occurs before the period and morbidity terminates by either recovery or death during the period; (b) onset of morbidity occurs before the period and morbidity terminates after the period; (c) onset of morbidity occurs during the period and morbidity terminates during the period; and (d) onset of morbidity occurs during the period and morbidity terminates after the period. The prevalence rate encompasses all four categories. It is defined as the total number of cases in the period as a percentage of the average number of persons in the population during the period. The incidence rate covers only the third and fourth categories. It is defined as the number of new cases during the period as a percentage of the average number of persons in the population. Incidence rate data would be more appropriate for investigating causal relationships between changes in exposure and changes in health status. Because willingness to pay to reduce morbidity is likely to depend on the length as well as the number of cases, the prevalence rate and measures incorporating data on duration would be more appropriate for analyzing the social costs of morbidity.

Broadly speaking, monetized estimates of reduced morbidity take one of two forms, those based on individual preferences (willingness to pay or required compensation), and those based on the resource and opportunity costs associated with illness. The latter form is typically referred to as a cost-of-illness or sometimes a damage cost measure. These are examples of the damage function method described in Chapter 2. They seek to identify the real costs of illness in the form of lost productivity and output and the increase in resources devoted to medical care. Costs per case of illness or per day are multiplied by the number of cases or days sick to determine an aggregate value. However, as shown formally in this section, the damage cost approach to valuation will yield an incorrect measure of welfare change. Thus, while costs of illness avoided may be relevant for some policy decisions, they are not a reasonable substitute for WTP values.

To see this in a simple way, consider the case of an individual who experiences one less day of asthma attacks because of an improvement in air quality. The benefit to that person might include avoiding the lost wages associated with being unable to work one day and the reduction in costs for medicine and treatment; but in addition, the individual avoids the discomfort associated with the attack itself. The first two components are captured by the cost-of-illness approach. Only a comprehensive willingness-to-pay measure would capture the discomfort component as well.

Now consider an individual who experiences no asthma attacks at present levels of air pollution because he spends money operating an air purifier and stays at home indoors on high-pollution days in order to prevent the attacks that

would be associated with exposure to the outdoor air. If air quality is improved, this individual benefits from being able to reduce the monetary expenditures and the lost wages and opportunities for leisure activities that are associated with these defensive activities. This individual will benefit from reduced air pollution even though there is no observed reduction in the actual incidence of asthma attacks related to air pollution or in the associated cost of illness.

As these two examples show, air pollution that affects human health can reduce people's well-being through four channels: the medical expenses associated with treating disease induced by air pollution (including the opportunity cost of time spent in obtaining such treatment); the lost wages resulting from the inability to work; the defensive or averting expenditures and activities associated with attempts to prevent disease induced by air pollution (including the opportunity cost of time); and the disutility associated with the symptoms and lost opportunities for leisure activities caused by the illness. Improving environmental quality can yield benefits to individuals by reducing some or all of these adverse effects. Portions of the first three of these effects have readily identifiable monetary counterparts, but effects of the fourth kind may not. A truly comprehensive benefit measure should be capable of capturing all of these relevant effects. Measures based solely on decreases in medical costs or lost wages are not comprehensive because they omit major categories of beneficial effects.

Although individual willingness to pay is the correct starting point for analyzing health-related values, there is one important respect in which society's valuation of changes in health might diverge from that of the affected individual. Society has developed several mechanisms for shifting some of the costs of illness away from the individual who is ill and onto society at large. These mechanisms include medical insurance, which spreads the costs of treatment among all policyholders, and sick leave policies, which shift at least part of the cost of lost work days onto the employer and ultimately onto the consumers of the employer's products. An individual's expressed willingness to pay to avoid illness would not reflect those components of the costs of her illness that are borne by or shifted to others. However, the value to society of avoiding her illness includes these components. Empirical measures of the value of reducing illness must take account of these mechanisms for shifting costs. This will be discussed in more detail below.

In this section, a general model of individual choice is developed that captures the principal ways by which individuals can affect their health status. This model is used to derive measures of the value of decreases in pollution that affect health and of improvements in health per se. Several extensions of the basic model are described. The problems that arise when this approach to valuation is applied to reductions in the incidence of chronic disease are considered.

A Basic Model of Health Production and Choice

Most of the formal models used for deriving the value of reduced morbidity use some variant of the health production function first developed by Grossman

(1972). Cropper (1981) introduced a pollution variable into the health production function. Harrington and Portney (1987) extended the model to examine explicitly the relationships among willingness to pay for a reduction in pollution, reductions in the cost of illness, and changes in defensive expenditures. The model developed here is an expanded version of the Harrington and Portney model. Dickie (2003) presented an excellent exposition of these models.

The health production function relates exogenous variables (including environmental variables such as air pollution) and choice variables (such as preventive medicine and treatment costs) to some measure of health status. It is assumed that individuals know their health production function, choose the output level optimally, and choose inputs so as to minimize the cost of production of any level of health. Of course, these are strong assumptions.

As originally formulated by Grossman (1972), the health production function was dynamic, allowing for "investments" in "health capital" that yielded benefits in the form of reduced illness over several time periods. The simple model presented here will abstract from this intertemporal dimension of the problem. Intertemporal models are briefly discussed in a later section.

Let health in any time period be measured by the number of days sick, represented by s. This is a simplification in that it makes no distinction between one episode of illness of two days' duration and two separate illnesses of one day each, and in that differences in the types of symptoms and the severity of illness are ignored. Some of the implications of richer specifications of the health variable are discussed in a later section.

Assume that one of the determinants of health status is the level of exposure to, or dose of, some environmental contaminant. Dose is represented by the scalar variable d, which depends on the concentration of pollution, c, and the amount of an averting activity, a, undertaken to avoid or reduce exposure to pollution. Examples of averting activities include filtering tap water before drinking and staying indoors on days of high air pollutant levels. If the contaminant is an air pollutant, c could be interpreted as the number of days during which some measure of air pollution exceeds a stated standard, the mean value of the pollutant averaged over the relevant time period, or the highest value recorded for the pollutant during that period. Since a change in pollutant emissions is likely to change all of these measures in a predictable way, the choice of a measure for d should be based on whatever is the best predictor of changes in health status. Similar questions arise for other forms of environmental contamination, such as chemicals in drinking water and pesticide residues in food.

Assume also that there is a set of mitigating activities and treatments that can be undertaken to reduce the health impact of any given exposure to pollution, represented by b. Examples of mitigating activities include taking antihistamines and visiting a doctor to obtain relief from a sinus headache. In this model, assume that the level of mitigating activities can be chosen by the individual so as to maximize utility.

The health production function for an individual can be written as

$$s = s(d, b) \tag{7.29}$$

$$d = d(c, a) \tag{7.30}$$

and by substitution

$$s = s(c, a, b) \tag{7.31}$$

with

$$\partial s/\partial c > 0 \tag{7.32}$$

$$\partial s/\partial a, \partial s/\partial b < 0\,. \tag{7.33}$$

The health production function can be estimated from cross-section data on illness, pollution, and averting and mitigating activities. It would also be necessary to control for other determinants of health status such as physical and socioeconomic characteristics of individuals for example, age, sex, use of tobacco, income, and education.

The individual derives utility from the consumption of a numeraire good, z, normalized with a price of one, and leisure, f. Illness causes disutility; thus,

$$u = u(z, f, s)\,, \tag{7.34}$$

with

$$\partial u/\partial z, \partial u/\partial f > 0 \tag{7.35}$$

$$\partial u/\partial s < 0\ . \tag{7.36}$$

The individual chooses z, f, a, and b so as to maximize utility subject to the budget constraint of

$$I + p_w(T - f - s) = z + p_a a + p_b b\ , \tag{7.37}$$

where

I = nonlabor income,
p_w = the wage rate,
T = total time available,
p_a = the price of averting activities, and
p_b = the price of mitigating activities.

Where there are time costs associated with the averting or mitigating activities, they should be incorporated into the full income budget constraint. For an example, see Gerking and Stanley (1986).

The first-order conditions for a maximum include

$$\partial u/\partial z = \lambda \tag{7.38}$$

$$\partial u/\partial f = \lambda \cdot p_w \tag{7.39}$$

and

$$\lambda \cdot \frac{p_b}{\partial s / \partial b} = \frac{\partial u}{\partial s} - \lambda \cdot p_w = \lambda \cdot \frac{p_a}{\partial s / \partial a}, \tag{7.40}$$

where λ is the Lagrangian multiplier and can be interpreted as the marginal utility of income.

The Marginal Value of Reduced Pollution

This model of choice can be used to derive an observable measure of the individual's marginal willingness to pay to reduce pollution. An individual's willingness to pay for a small reduction in ambient pollution is the largest amount of money that can be taken away from that person without reducing her utility. If pollution enters the utility function directly, for example, because of aesthetic disamenities associated with pollution, then there are additional benefits that are not associated with health. This point is discussed further in the section on complex models.

In the health production model, in which pollution affects utility only through health, willingness to pay is the reduction in the cost of achieving the optimal level of health made possible by the decrease in pollution. For example, if a reduction in ozone levels from 0.16 to 0.11 parts per million (ppm) reduces the number of days of respiratory symptoms from 6 to 4, and if an expenditure of $20 on averting activities or on medicine has the same effect, then (all else being equal) the individual's maximum willingness to pay is $20 for the ozone reduction.

Formally, marginal willingness to pay for a reduction in pollution (w_c) is given by the reduction in sick time associated with the reduction in pollution times the marginal cost of reducing sick time. The latter is given by the cost of an additional mitigating input divided by the reduction in sick time that input produces, or, alternatively, by the cost of averting behavior divided by the reduction in sick time that averting behavior produces. To see this, first totally differentiate the indirect utility function, $v(I, p_w, p_a, p_b, c)$, and solve for willingness-to-pay (w_c) defined as dI / dc to obtain:

$$\frac{dI}{dc} = -\frac{\partial v / \partial c}{\partial v / \partial I} = -\frac{\partial v / \partial c}{\lambda}. \tag{7.41}$$

Next note that the effect of c on utility consists of two components: the direct loss of utility due to illness and the opportunity cost of time spent sick valued at the wage rate, or

$$\frac{\partial v}{\partial c} = \frac{\partial u}{\partial s} \cdot \frac{\partial s}{\partial c} - \lambda \cdot p_w \cdot \frac{\partial s}{\partial c} = \left(\frac{\partial u}{\partial s} - \lambda \cdot p_w \right) \frac{\partial s}{\partial c}. \tag{7.42}$$

Then, substitute the first-order condition equation (7.40) for the term in parentheses in equation (7.42):

$$\frac{\partial v}{\partial c} = \left(\lambda \cdot \frac{p_b}{\partial s / \partial b}\right)\frac{\partial s}{\partial c}. \tag{7.43}$$

Finally, substitute equation (7.43) into equation (7.41) to obtain

$$w_c = -p_a \frac{\partial s / \partial c}{\partial s / \partial a} = p_a \frac{\partial a}{\partial c}. \tag{7.44}$$

A similar procedure leads to

$$w_c = -p_b \frac{\partial s / \partial c}{\partial s / \partial b} = p_b \frac{\partial b}{\partial c}. \tag{7.45}$$

The right-hand terms in equation (7.44) and equation (7.45) follow from application of the implicit function rule.

Most of these results can also be derived from the expenditure function, as follows:

$$e \equiv \min\left[z + p_a \cdot a + p_b \cdot b + p_w \cdot s\right] + \mu\left[u^0 - u(z, f, s)\right], \tag{7.46}$$

where $s = s(c, a, b)$. The first-order conditions include

$$\frac{p_a}{\partial s / \partial a} = \mu \cdot \frac{\partial u}{\partial s} - p_w. \tag{7.47}$$

So,

$$w_c = -\frac{\partial e}{\partial c} = \left(\mu \cdot \frac{\partial u}{\partial s} - p_w\right)\frac{\partial s}{\partial c} = p_a \cdot \frac{\partial s / \partial c}{\partial s / \partial a}, \tag{7.48}$$

which is the same as equation (7.44). Equation (7.45) can be found by similar manipulations.

There are several things to note about these expressions for marginal willingness to pay. First, the ratios

$$\frac{\partial \mathrm{s} / \partial \mathrm{c}}{\partial \mathrm{s} / \partial \mathrm{b}} \text{ and } \frac{\partial \mathrm{s} / \partial \mathrm{c}}{\partial \mathrm{s} / \partial \mathrm{a}} \tag{7.49}$$

can be interpreted as marginal rates of technical substitution (MRTS) between pollution and the other variable in producing a constant level of sickness. Marginal willingness to pay can be expressed in terms of any of the MRTSs between pollution and another input in the production of health, since to minimize the cost of producing health, the values of marginal products of all inputs must be equal at the margin. Second, all of the measures are functions of observable variables that can be calculated given knowledge of the health production function. Third, as shown by the right-hand terms, marginal willingness to pay can be calculated from the reductions in expenditures on either mitigating or averting behavior that are required to attain the original health status, holding all else constant. As

noted in Chapter 4, this will not be, in general, equal to the observed reduction in mitigating or averting behavior associated with the reduction in pollution.

In order to compute equations (7.44), (7.45), or (7.48), it is necessary to estimate a production function for the health outcome of interest and evaluate the numerator and denominator of the equation at current levels of all inputs. In practice, this has proven to be a difficult task. However, for one effort to implement this basic model, see Gerking and Stanley (1986). In addition to having data on the relevant health outcome and on ambient pollution levels, one must identify averting and mitigating behaviors and measure their costs. In practice the most effective method of reducing exposure, given ambient pollution levels, is to spend time indoors. Although the amount of time spent indoors could be measured, determining its cost would be difficult. Devices that reduce indoor pollution concentrations (air conditioners, air filters) have costs that can be measured; but they produce other services, such as reducing indoor temperature, so that it is inappropriate to allocate all of these costs to pollution avoidance. The implications of this possibility are discussed in the section on complex models.

Because of the difficulties of implementing these measures, it is useful to consider an alternative expression that shows the relationship between the observable cost of illness and marginal willingness to pay. The first step in deriving this expression is to obtain the demand functions for a and b: $a^*(I, p_w, p_a, p_b, c)$ and $b^*(I, p_w, p_a, p_b, c)$. These functions give the optimal quantities of a and b as functions of income, prices, and pollution. The second step is to take the total derivative of the health production function:

$$\frac{ds}{dc} = \frac{\partial s}{\partial a} \cdot \frac{\partial a^*}{\partial c} + \frac{\partial s}{\partial b} \cdot \frac{\partial b^*}{\partial c} + \frac{\partial s}{\partial c} \cdot \tag{7.50}$$

This gives the impact of a change in pollution on illness after taking account of the optimal adjustments of a and b to the pollution change. This expression can be rearranged as follows:

$$\frac{\partial s}{\partial c} = \frac{ds}{dc} - \frac{\partial s}{\partial a} \cdot \frac{\partial a^*}{\partial c} - \frac{\partial s}{\partial b} \cdot \frac{\partial b^*}{\partial c}, \tag{7.51}$$

and multiplied by the first-order conditions of equation (7.40)

$$-\frac{p_a}{\partial s / \partial a} = p_w - \frac{\partial u / \partial s}{\lambda} \tag{7.52}$$

to obtain

$$\begin{aligned} -p_a \frac{\partial s / \partial c}{\partial s / \partial a} = & \left(p_w - \frac{\partial u / \partial s}{\lambda} \right) \cdot \frac{ds}{dc} - \left(p_w - \frac{\partial u / \partial s}{\lambda} \right) \cdot \frac{\partial s}{\partial a} \cdot \frac{\partial a^*}{\partial c} \\ & - \left(p_w - \frac{\partial u / \partial s}{\lambda} \right) \cdot \frac{\partial s}{\partial b} \cdot \frac{\partial b^*}{\partial c} \end{aligned} \tag{7.53}$$

or, after rearranging:

$$w_c = p_w \cdot \frac{ds}{dc} + p_b \cdot \frac{\partial b^*}{\partial c} + p_a \cdot \frac{\partial a^*}{\partial c} - \frac{\partial u / \partial s}{\lambda} \cdot \frac{ds}{dc} \cdot \tag{7.54}$$

This expression says that MWTP is the sum of the observable reductions in the cost of illness and averting activities and the monetary equivalent of the disutility of illness. The change in the cost of illness consists of the economic value of reductions in sick time and mitigating expenditures. The term $p_w\left(ds/dc\right)$ includes both actual lost wages and lost leisure time valued at the wage rate.

To compute ds/dc we do not need to estimate a health production function, but can instead estimate a dose-response function—a reduced-form relationship between illness and ambient pollution controlling for other variables that affect health status. In the health production framework a dose-response function is obtained by substituting the demand functions for b and a into the health production function. Full implementation of equation (7.54) as a measure of value therefore requires estimation of these demand functions.

As a practical matter, the first three terms in equation (7.54) can be approximated after the fact by using the observed changes in illness and averting and mitigating expenditures. In this way, equation (7.54) can be used to derive a lower bound to individual WTP. Since the last term in the equation is negative, $\partial u/\partial s < 0$, the first three terms—the value of lost time plus the change in averting and mitigating expenditures—give a lower bound to WTP. In the health literature, the term "cost of illness" typically refers only to the social cost of lost earnings plus the medical expenditures associated with illness. This term therefore ignores two components of the social cost of illness—the social value of averting expenditures and the cost of lost leisure time that result from illness.

The Marginal Value of Reduced Illness

The marginal willingness to pay for an exogenous reduction in illness falls out of the above model as a special case. Suppose that averting behavior is not possible and that mitigation (b) reduces realized s from its exogenous level s^* according to

$$s = f\left(s^*, b\right) = s^* - s\left(b\right). \tag{7.55}$$

The analogs of equations (7.41)–(7.44) are

$$\frac{dI}{ds^*} = -\frac{\partial v / \partial s^*}{\partial v / \partial I} = -\frac{\partial v / \partial s^*}{\lambda} \tag{7.41′}$$

$$\frac{\partial v}{\partial s^*} = \frac{\partial u}{\partial s^*} - \lambda p_w \tag{7.42′}$$

$$\frac{\partial v}{\partial s^*} = \lambda \cdot \frac{p_b}{\partial s / \partial b} \tag{7.43′}$$

$$w_{s*} = -p_b \cdot \frac{\partial b}{\partial s^*} \cdot \tag{7.44'}$$

Similarly, the analog to equation (7.53) is

$$w_{s*} = p_w + p_b \frac{\partial b^*}{\partial s^*} - \frac{\partial u / \partial s^*}{\lambda}, \tag{7.54'}$$

or, marginal willingness to pay is the sum of the cost of illness (lost wages and mitigation costs) and the monetary equivalent of the lost utility (pain and suffering).

Valuing Nonmarginal Changes in Pollution

In the preceding subsection, a measure of value for a small change in health status is defined. This value is likely to be a function of health status itself. Specifically, for reasons analogous to the standard assumption of diminishing marginal utility, the marginal willingness to pay for further decreases in morbidity is likely to decrease as health status increases. In principle, this functional dependence of marginal willingness to pay on health status should be taken into account whenever large (that is, nonmarginal) changes in morbidity are being valued. The proper way to do this is straightforward in principle, but may be hard to implement in practice.

If the function relating marginal willingness to pay to health status is known, the value of the nonmarginal change is simply the integral of this function over the relevant range. Typically, however, what empirical methods produce is a point estimate of marginal willingness to pay. If it can be assumed that marginal willingness to pay is (approximately) constant over the relevant range, the value of the nonmarginal change can be calculated by multiplying the marginal willingness to pay times the change in health status. The larger the change in morbidity and the more rapidly marginal willingness to pay decreases with increases in health status, the larger the error in using this simple approach. Alternatively, one could compute a lower bound to MWTP. As shown in Chapter 4, the change in averting and mitigating expenditures, holding illness constant, constitutes a lower bound to willingness to pay for pollution improvement.

Extensions of the Health Production Model

The basic model of choice and value described in the preceding section involves a considerable amount of simplification in order to obtain some insights about the relationship between observed behavior and economic value. In this section, the model is extended in several directions toward greater realism to see to what extent the conclusions about values are preserved.

Multiple Symptoms

The model can be generalized to the case of many symptoms and various forms of averting and mitigating behavior. Consider the special case in which each mitigating activity enters the separate health production function for one symptom, $s_i(c, a, b)$. Following procedures similar to those of the basic model, it can be shown that

$$w_c = \sum p_i \cdot \frac{\partial s_i / \partial c}{\partial s_i / \partial b_i}, \tag{7.56}$$

where p_i is the price of b_i. Estimation of w_c in this case involves estimating production functions for all symptoms. Similarly, it can be shown that

$$w_c = p_a \cdot \frac{\partial s_i / \partial c}{\partial s_i / \partial a} \tag{7.57}$$

for any symptom.

Unfortunately, simple expressions such as these cannot be obtained when mitigating activities affect more than one symptom. Suppose that there are two symptoms, s_1 and s_2, neither of which restricts activities, and two mitigating activities, b_1 and b_2, with health production functions:

$$s_1 = s_1\left(c, a, b_1, b_2\right) \tag{7.58}$$

and

$$s_2 = s_2\left(c, a, b_1, b_2\right). \tag{7.59}$$

For example, b_1 could be a multi-symptom cold remedy. The first-order conditions are:

$$\frac{\partial u / \partial s_1}{\lambda} \cdot \frac{\partial s_1}{\partial b_1} + \frac{\partial u / \partial s_2}{\lambda} \cdot \frac{\partial s_2}{\partial b_1} - p_1 = 0 \tag{7.60}$$

$$\frac{\partial u / \partial s_1}{\lambda} \cdot \frac{\partial s_1}{\partial b_2} + \frac{\partial u / \partial s_2}{\lambda} \cdot \frac{\partial s_2}{\partial b_2} - p_2 = 0 \tag{7.61}$$

$$\frac{\partial u / \partial s_1}{\lambda} \cdot \frac{\partial s_1}{\partial a} + \frac{\partial u / \partial s_2}{\lambda} \cdot \frac{\partial s_2}{\partial a} - p_a = 0\,. \tag{7.62}$$

As Dickie and Gerking (1991) showed, as long as the derivatives of the health production function are known and the number of mitigating and averting activities (three here) equals or exceeds the number of symptoms (two), this system of equations can be used to solve for the unobserved utility terms, for example,

$$\frac{\partial u / \partial s_1}{\lambda}. \tag{7.63}$$

The solution can be used to substitute into the following expression to obtain a measure of willingness to pay:

$$w_c = -\frac{\partial v / \partial c}{\lambda} = -\left[\frac{\partial u / \partial s_1}{\lambda} \cdot \frac{\partial s_1}{\partial c} + \frac{\partial u / \partial s_2}{\lambda} \cdot \frac{\partial s_2}{\partial c}\right]. \tag{7.64}$$

However, if the number of symptoms exceeds the number of mitigating and averting activities, then the expression for MWTP will necessarily include some unobservable marginal utility terms.

Shifting the Cost of Illness

The measures derived above give each individual's willingness to pay to avoid pollution-induced illness. However, where social mechanisms have been developed for spreading some of the costs of illness, the social value of reducing pollution may exceed the aggregate of the individuals' willingness to pay. For example, where employers grant a certain number of days of paid sick leave, the individual does not incur any financial cost for the loss of a workday. Thus, estimates of WTP from either revealed or stated preference will be conditioned on part of these costs being shifted and will not include, in general, the lost wages component of the first term on the right of equation (7.54). Nevertheless, society loses that person's output for that day.

The burden of this lost output is borne initially by the employer, but ultimately shows up as some combination of higher prices for outputs and lower equilibrium wages. A nonmarginal reduction in pollution that resulted in fewer lost workdays would reduce the costs of paid sick leave plans. In principle, this would result in changes in general equilibrium wages and prices.

Similarly, medical insurance has the effect of shifting the marginal cost of treatment away from the sick individual. A 100 percent coverage plan reduces the price of medical treatment to zero. Thus, marginal willingness-to-pay measures based on the health production function would have to focus on changes in averting expenditures. Again, changes in the cost of treatment covered by insurance should be added to the aggregate of individuals' willingness to pay to obtain the social benefit of reduced pollution-induced illness.

As Harrington and Portney (1987) pointed out, the introduction of sick leave and medical insurance plans changes incentives by changing the prices of these activities and thus could alter individuals' choices of mitigating and averting activities and days of illness. If the effects of these plans on prices have been taken into account in estimating the models of individual behavior, then the measures of social values described here will be correct.

Health Capital and the Health Production Function

The basic model outlined above ignores any dynamic aspects of health and illness. In that model, averting or mitigating behavior and pollution in any one period of time affect only illness in that time period. It would be more in the spirit of Grossman's (1972) original treatment of the health production function to formulate the choice problem as one of optimizing investment in health capital over time. Health capital can be thought of as a measure of health status that depreciates over time unless it is maintained or augmented with some kind of investment. To formalize this, let h_t be the stock of health capital at t. Then h_t can be augmented by investment of time, y_t, in activities such as exercising and by the purchase of "health" goods, x_t. Health capital is also subject to depreciation at the rate of g percent per year. The health stock changes over time according to

$$\frac{dh_t}{dt} = I_t\left(x_t, y_t\right) - g \cdot h_t \,. \tag{7.65}$$

The rate of depreciation itself can be made a function of age and other characteristics.

One approach to deriving willingness to pay is to assume that health capital is an argument in the symptom production functions. Then, measures of willingness to pay similar to those of the basic model can be derived. Cropper (1977, 1981) used a different approach. She made the rate of depreciation a function of pollution, among other things. She also assumed that illness was not a direct argument in the utility function. She derived a willingness-to-pay measure that involves two terms. The first term is the value of the reduction in sick time. The second is the change in the rate of investment in health capital caused by the change in pollution. The sign of this term is ambiguous, since reducing pollution both lowers the price of an investment in health capital and increases the desired quantity of health capital. This second term is analogous to the expressions involving averting and mitigating activities. Once again, the conclusion is that true willingness to pay is different from changes in the cost of illness because of the induced changes in other activities and expenditures that affect health.

Complex Models

The models outlined so far have been simple in the sense that pollution affects only sick days and averting behavior serves only to reduce exposure to pollution. However, if either pollution or averting activities also affect utility directly, then the model becomes more complex in the sense that it is no longer possible to derive expressions for marginal willingness to pay that are functions only of observable data. Rather, the expressions for MWTP become contaminated with unobservable marginal utility and/or marginal disutility terms.

Suppose that in addition to causing illness, pollution causes disutility directly—for example, by impairing the view from one's house. Then the expression for willingness to pay in this modified version of the basic model is

$$w_c = \frac{\partial v / \partial c}{\lambda} = -\frac{(\partial u / \partial s - \lambda \cdot p_w)\partial s / \partial c + \partial u / \partial c}{\lambda}. \tag{7.66}$$

The first-order condition can be used to eliminate $(\partial u/\partial s)/\lambda$; but the unobservable $(\partial u/\partial c)/\lambda$ remains. Observations on averting and mitigating behavior are not sufficient to measure MWTP. However, if pollution varies across housing sites, then the hedonic housing price model provides a way around this problem (see Chapter 10). Similarly, suppose a generates utility in addition to reducing exposure. For example, a could be air conditioning or water filters that reduce unpleasant tastes in tap water. Then the first-order condition for a becomes

$$\left(\frac{\partial u}{\partial s} - \lambda p_w\right) = \frac{\left(\lambda p_a - \frac{\partial u}{\partial a}\right)}{\partial s / \partial a}. \tag{7.67}$$

When this is substituted into the indirect utility function, the unobservable $(\partial u/\partial a)/\lambda$ remains.

Chronic Morbidity

Now we turn to some of the problems that arise when these models of choice and value are applied to the case of chronic morbidity. One problem is how to quantify and measure chronic morbidity. Chronic morbidity is more like a state of being than like an illness where incidence and duration are the primary characteristics of concern. It may be more useful to model chronic morbidity in terms of state-dependent preferences. For example,

$$u = u(\boldsymbol{X}, s), \tag{7.68}$$

with $s = 0$ for health and $s = 1$ for chronic disease.

A second problem concerns the choice between an ex ante and an ex post perspective. We have already discussed some of these issues in the context of mortality risks. Suppose that air pollution increases the risk of chronic lung disease. The ex ante perspective asks for a healthy individual's willingness to pay to reduce the risk of incurring chronic lung disease at some time in the future. The ex post perspective would be applicable to individuals already suffering from chronic lung disease who would be asked their willingness to pay to be restored to a healthy state. As discussed in Chapter 5, the ex ante and ex post perspectives might give quite different values to the same chronic health effect, even leaving aside the likelihood that an individual who actually experiences a chronic health effect might learn something about it that would alter his willingness to pay. Which perspective is appropriate would depend on the nature of the policy question being asked. If a potential cure for the chronic condition is available, the ex post perspective yields the correct answer to the valuation question. If the policy in question is one that would reduce exposure to, say, air pollution leading to reduced risk of chronic disease, then the ex ante perspective is appropriate.

Toward Measurement

In order to implement measures based on the health production function, such as equations (7.44) and (7.54), we would require the following data for a cross section of individuals over some time period:

1 Frequency, duration, and severity of pollution-related symptoms;
2 Ambient pollution levels to which the individual is exposed;
3 Actions the individual takes to avoid or mitigate the effects of air pollution;
4 Costs of avoidance and mitigating activities;
5 Other variables affecting health outcomes (age, general health status, presence of chronic conditions, and so forth).

These data would be used to estimate health production and input demand functions, which, in turn, would be used to calculate the expression for marginal willingness to pay.

An alternative approach to the revealed preference method based on the health production function is to employ stated preference methods. For example, people could be asked directly what they would be willing to pay to reduce pollution. If willingness to pay for morbidity benefits varies with health status, age, and income, pollution control policies may have very different benefits depending on the characteristics of the target population. To value the morbidity benefits of different policies we must therefore know the distribution of key variables in the population affected by the policy, and how WTP (or the cost of illness) varies with these characteristics.

In principle, each of the valuation techniques presented above is capable of describing how the value of reduced morbidity varies with the characteristics of the respondent. The averting behavior models suggest that WTP depends on any variables that affect the marginal product of pollution, mitigating activities, or avoidance activities. In practice these variables would include health status (whether or not the respondent has a chronic respiratory condition), age, and perhaps education. WTP will also vary with factors such as earned income or education that affect the cost of averting activities. The effect of these variables on WTP could be calculated from the health production function or from equations describing the unit cost of averting activities. When WTP is estimated by a stated preference method, data are typically gathered on variables that would enter the health production function or affect the level of averting (mitigating) behavior undertaken. WTP responses can then be regressed on these variables.

Significant progress has been made in the last decade in developing and implementing stated preference approaches for valuing morbidity, either alone or in conjunction with increased mortality. Cameron and DeShazo (2013) reported on an extensive and thorough study where reduced risk of both morbidity and mortality are valued. The authors considered four possible future health states ranging from "current health" to "sickness," "recovered/remissions years," and

"lost life years." Using a stated choice experiment (a form of stated preference elicitation that will be covered in Chapter 12), they elicited values associated with reducing the risk of an undesirable future health state. In addition to providing value estimates for the type of good most appropriate for many environmental and health assessment (the ex ante reduced risk of illness/death), their careful approach allows estimates of WTP that vary by age, risk beliefs, health status and other attributes. In another study that considers both mortality and morbidity reductions simultaneously, Adamowicz et al. (2011) developed a choice experiment that was implemented across a sample of Canadians. They too developed estimates of WTP that vary across sub-populations including age, gender, health status, among others and, importantly, they estimated separate values for reducing mortality and morbidity from microbial disease versus cancer risk.

A range of studies valuing reduced morbidity using stated preference methods have also appeared recently. Using a nationwide survey with a sample of over 3,500 people, Viscusi, Huber, and Bell (2012) estimated values for reducing gastrointestinal illness from drinking water. Their results confirm that these values can vary considerably across the population. Specifically, households who perceive themselves to be at higher risk, who consume large amounts of tap water, and/or who identify themselves as environmentalists, report the highest values. These findings substantiate the need for benefit-cost analysis to carefully consider the population that will be affected by a policy or regulation. Other examples of using stated preference methods to value reduced morbidity include values for foodborne illness (Teisl and Roe 2010), chronic pain (Chuck et al. 2009), skin cancer (Bateman and Brouwer 2006), and asthma (Blomquist, Dickie, and O'Conor 2011).

What are the prospects for measuring WTP using the averting behavior approach? It is usually difficult to measure the cost of activities that reduce exposure to pollution or that prevent symptoms from occurring altogether. The mere presence of an air conditioner in a home or in a car is an imperfect measure of reduced exposure to air pollution, and the cost of averting behavior is inherently difficult to measure. For example, to determine the cost of spending leisure time indoors rather than outdoors, we cannot rely on observed prices but must question individuals directly. Also measuring the cost of averting behavior is complicated by the fact that many avoidance activities produce joint products.

Two recent papers overcame these challenges using carefully collected data and survey design. Richardson, Champ, and Loomis (2012) used revealed preference information to construct a health production function for exposure to smoke from forest fires in Los Angeles County. The use of a home air cleaner is found to be an effective averting action that reduces expected symptom days. Combining the effectiveness of air cleaners with their cost yields an estimate of about $84/day to avoid a day of symptoms resulting from exposure to wildfire smoke. In contrast, Mansfield, Johnson, and Van Houtven (2006) combined revealed preference diary data on outdoor activity and a stated preference conjoint analysis of parents' WTP to avoid restrictions on their children's outdoor activities to provide an estimate of

the cost of averting behavior for parents who keep their children indoors to reduce exposure to high ozone level.

Special Topics

Subtle Health Effects

In addition to the mortality and morbidity effects discussed in this chapter, there is now substantial evidence that chemical contaminants can produce subtle physical and mental changes in people. For example, relatively low levels of lead in blood have been implicated in decrements to IQ in children, in low birth weight, and in lower body growth rates and, presumably, height at maturity (U.S. Environmental Protection Agency 1986). To the extent that subtle effects such as low birth weight are associated with other adverse health effects and illness, they pose no special methodological problems from an economic perspective. The models and techniques discussed in this chapter can be used to estimate the monetary values of avoiding these adverse health effects. However, problems arise in estimating monetary values for avoiding more subtle effects such as decrements in IQ and reduced growth rates. Some studies have estimated the costs of special education and lost productivity of individuals with impaired intellectual development due to elevated levels of lead in blood (for example, U.S. EPA 1997). However, from a conceptual perspective, these cost-of-illness and lost-wage measures are only partial and incomplete measures of the total willingness to pay to avoid adverse health effects. Surely, there is a loss of utility or well-being associated with intellectual impairment; but basing estimates of the value of this loss on an individual's willingness to pay is problematic.

Valuing Health Effects on Children

The economic valuation of policies that protect or improve the health of children raises interesting ethical, as well as empirical questions, especially what normative perspective to adopt in valuation. For a discussion of some of these issues, see U.S. EPA (1999). The standard theory of welfare economics is based on the assumptions that each individual is the best judge of how well off he/she is in a given situation, and that individuals have well-defined preferences over alternative states and choose rationally among alternatives subject to the usual constraints. These assumptions define what might be called the consumer sovereignty normative perspective of welfare economics. When we turn to the welfare economics of children, there appear to be three alternative ethical or normative perspectives. The first is a natural extension of consumer sovereignty to children or "children's sovereignty." The second perspective can be termed "parental sovereignty," and would use the parents' values for changes in the health of their children. According to the third or "adult as child" perspective, values would be based on what the adult would have chosen for him/herself in childhood. None of these ethical perspectives is entirely satisfactory. Each is discussed in turn.

Children's Sovereignty

This perspective is consistent with the individualistic basis for welfare economics, however, it is not ethically attractive. Children are immature and lack the cognitive ability to make choices about health and safety. They may not have well-defined preferences over the full range of alternatives necessary to make reasoned choices, and they do not control the financial resources that are required to make tradeoffs between money and health or safety.

Parental Sovereignty

How is this perspective justified on ethical grounds? One possibility is to assert guardianship or stewardship. Another possibility is parental altruism of some form. Some authors simply presume altruism without much discussion. Others note that parents do not always seem to be the best judges of what is good for their children and sometimes engage in activities such as smoking and drinking that actually harm their children.

Parental sovereignty also has some ethically unattractive implications at a more fundamental level. The economic analysis of fertility choice emphasizes the utility that children convey to parents and the potential economic benefits they bring through providing labor for household production, and in the long term, economic security for their parents. If the marginal utility of a child (or its marginal productivity) is decreasing in the number of children, then the value to the parent of reducing the risk of death to the child or preventing disease depends on the number of children in the family and, perhaps, on its birth order.

Child as Adult

This is the perspective most consistent with the basic welfare economics principles, but one that is very difficult to implement. We cannot observe the relevant choices, so stated preference methods would have to be used, and they would impose difficult cognitive tasks on respondents.

In conclusion, the choice of an ethical or normative perspective involves some difficult questions with no easy answers. However, each perspective has implications for how we go about measuring values for children. Perhaps it is more accurate to say that each of the methods discussed in this chapter maps back to one of these perspectives. The household production function and averting behavior methods are consistent with Parental Sovereignty, since it is the choices of parents that we observe. When we attempt to use benefits transfer where the values come from adults and are adjusted on some basis, this implies the Child as Adult perspective. Finally, cost of illness measures may still have a role as lower bound estimates of value from either of the perspectives.

Quality-Adjusted Life Years

The models described above take as measures of the outcome of policy the change in the probability of dying and the reduction in days spent sick. However, as noted above, these outcome measures are simplifications of a more complex reality. Another outcome measure that has been used extensively in the health economics literature is the quality-adjusted life year (QALY). The QALY is a measure of the performance of medical treatments and interventions. It captures in a single metric two important dimensions of medical outcomes: the degree of improvement in health, and the time interval over which the improvement occurs, including any increase in the duration of life itself. Most of what is said here about QALYs applies also, with appropriate modification, to disability-adjusted life years, health-adjusted life years, and related concepts. Duration is measured in years of life; and quality is indexed by a number between 0, representing death, and 1, representing perfect health. Therefore, a treatment that is expected to increase the duration of life by 1 year of perfect health is said to produce 1 QALY. A treatment that improves health status from an index number of 0.25 to 0.75 for two years also produces 1 QALY, and a treatment that extends a life by 5 years at a quality level of 0.4 produces 2 QALYs. In the evaluation of alternative health policies or treatment programs, the numbers of QALYs produced for each patient or recipient of treatment are simply added up to obtain an aggregate measure of program effectiveness. As a summary measure with simple intuitive appeal, can the QALY be used in benefit-cost analysis to describe the outcomes of environmental policies? There are no simple answers to this question.

Many of the earliest papers on this topic relied on the judgments of medical professionals to provide the quality weights for different health states (Torrance 1986). However, the preferred approach has been to obtain the weights by some form of questioning of a sample of individuals representative of the population of interest. There are three principal ways of asking questions in order to elicit values or weights. They all start with a description of a health state including symptoms, degree or level of pain, degree of impairment of activity or function, and so forth (Gold et al. 1996; Fabian 1994).

In the simplest form of questioning, respondents are simply asked to assign a weight or numerical value between one and zero that reflects the utility they assign to the health state relative to states of perfect health and of death, respectively. Often, respondents are provided with a visual aid such as a horizontal line with a scale between zero and one marked on it.

The second approach is known as the time tradeoff approach. Respondents are asked to choose between two options: living in a given state of less than perfect health for a fixed period of time (T), for example, 5 years, and living a shorter period of time (N) in perfect health. The number of years of perfect health is varied until the individual expresses indifference between the two options. The value or weight attached to the impaired health state is simply N/T.

The third approach to questioning is the standard gamble question derived from the method first outlined by von Neumann and Morgenstern (1944) for eliciting cardinal utilities. Respondents are asked to choose between two options, where option A is living with the impaired health state with certainty for the rest of one's normal life span, and option B is a gamble in which one outcome is living for the same period of time in perfect health (with a probability of p) and the other outcome is immediate death (with a probability of $1 - p$). The probability is then varied until the individual expresses indifference between the gamble and the given health state with certainty. This means that the expected utilities of the two choices are equal. The quality weight for the given health state is simply the probability (p) that makes the individual indifferent between the two choices.

Advocates of the use of QALYs in policy evaluation cite as one advantage of the concept that QALYs are based on individuals' preferences. To the extent that QALYs are derived from the responses of representative individuals, there is something to this claim. However, as several authors have shown, given the way that QALYs are used in policy evaluation, they are consistent with utility theory and the economic theory of individual preferences only if individuals' utility functions and preference structures satisfy some quite restrictive conditions. Some examples will illustrate the nature of some of these restrictions (this discussion is based on Freeman, Hammitt, and De Civita 2002). For further discussion and illustration, readers should consult Broome (1993), Johansson (1995), Fabian (1994), Garber et al. (1996), Johnson and Lievense (2000), Hammitt (2002), Pliskin, Shepard, and Weinstein (1980), and Bleichrodt, Wakker, and Johannesson (1997). The first condition is risk neutrality over longevity, which means that an individual is indifferent to patterns of mortality risks that have the same life expectancy. For example, an individual must be indifferent between living 25 more years for certain, and a gamble offering a 50 percent chance of living 50 more years and a 50 percent chance of dying immediately.

The second condition is constant proportional tradeoff (of longevity for health), which implies that the fraction of remaining longevity an individual would trade to improve his health from one state to another (for the rest of his life) does not depend on his longevity. For example, if he is willing to give up 10 of 50 remaining years to improve his health from fair to excellent, he would also be willing to give up one of 5 remaining years for the same health improvement. Alternatively, if future QALYs are discounted, as is recommended practice (Gold et al. 1996), then it is the fraction of discounted longevity an individual is willing to give up that must remain constant (Johannesson, Pliskin, and Weinstein 1994). Johnson and Lievense (2000) cited evidence suggesting the assumption of constant proportional tradeoff is unduly restrictive.

An additional condition is that an individual's preferences for health and longevity are utility independent of his wealth and future income, which means that his preferences for risks that affect health or longevity do not depend on income. This assumption implies that the effect of income on utility is positively related to the quality weight for each health state. There is little empirical information available

on this point, although the notion that the marginal utility of income is smaller in impaired health than in full health is consistent with one study (Sloan et al. 1998).

Because QALYs impose restrictive assumptions on preferences, the ranking of health interventions using QALYs may differ systematically from the ranking using WTP. For example, given these restrictions, the QALY value of reducing mortality risk within the current year to different people should be proportional to life expectancy. This implies that the value of reducing risk to a 20-year-old is about three times larger than the value of reducing risk to a 65-year-old. In contrast, individual WTP to reduce mortality risk does not fall as sharply with decreasing life expectancy, and may even increase as life expectancy declines over some range of ages. Using WTP, it is not necessarily more valuable to reduce mortality risk to a younger person, and in any case, the differential value assigned is likely to be smaller than under the QALY approach. WTP to reduce mortality risk does not fall in proportion to life expectancy because the opportunity cost of spending on risk reduction also falls with decreasing life expectancy, as the individual has less to save for (Hammitt 2000).

In summary, QALYs are not, in general, an appropriate measure for use in benefit-cost analysis and welfare assessment. Where WTP estimates of health values are available, they are a superior reflection of individuals' preferences. When WTP estimates are not available, QALYs may provide useful information but their limitations must be carefully considered.

Summary

The economic framework for valuing reductions in illness and risk of death has been described in this chapter. The aggregate benefit of a reduction in the risk of death and incidence of illness is the sum of what each of the affected people is willing to pay to reduce his or her own risks of death and illness.

For an individual to have a positive WTP for a reduction in risk of death or illness implies that the individual can perceive and is aware of changes in his health status. It does not require that the individual know that the reduction is attributable to a specific environmental change. If the value of reduced risk or symptom days is known, policy analysts can calculate benefits if they can predict the magnitude of the reduction caused by pollution control. Assumptions about individuals' knowledge do play important roles in the empirical estimation of WTP in some circumstances, however. For example, if WTP for risk reductions is to be inferred from wage-risk premia, then it must be assumed that individuals know the relative risk levels of different jobs. Also, if WTP for reduced illness is to be inferred from an individual's averting or mitigating behavior in response to changes in pollution, then it must be assumed that the individual knows the relationship between pollution and her illness experience.

In valuing risks to life, two forms of revealed preference methods have been widely used. One is based on compensating wage differentials received by workers in risky occupations; the other is based on the cost of goods that increase safety,

such as seat belts and smoke detectors. Safety goods are typically 0–1 activities whose benefits exceed their costs for most persons. Thus, unless the data permit estimation of the parameters of a discrete choice model, basing risk valuations on these activities is likely to understate the value of a risk reduction. On the other hand, if the safety good also conveys utility directly, then the value of risk reduction alone will be overestimated. The compensating wage approach may also understate the value of a risk reduction because persons who are willing to be paid to accept increased risk (such as structural ironworkers) may have lower values for risk reduction than the average person.

A serious shortcoming of studies of job-risk premia and use of safety equipment is that they value only voluntarily assumed risks of accidental death (for example, risk of death on the job, or risk of dying in an auto accident). By contrast, environmental risks are largely involuntary, and may lead to a painful illness (such as cancer) before death occurs. These considerations suggest that individuals might be willing to pay more to reduce environmental risks than to reduce risk of death in an auto accident. On the other hand, to the extent that environmental risks may not manifest themselves until after a long latency period, fewer years of life will be lost than are risked from death in an auto accident during the current year. This suggests that willingness to pay estimates for a reduction in current risk of death may overstate willingness to pay to reduce environmental risks.

One advantage of stated preference methods is that the risks being valued can be tailored to circumstances relevant to environmental health policy and the most relevant populations can be sampled. For example, people can be asked their willingness to pay to reduce exposure to a pollutant that increases their risk of dying of cancer in 20 years. However, as the risk being valued becomes smaller in magnitude and more distant in time, individuals may have trouble understanding what it is they are valuing.

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Chapter 8

Environmental Quality as a Factor Input

In addition to providing services directly to individuals as consumers, environmental and resource systems can affect the costs and output levels in the production sector of an economy. Costs and output levels can be affected, for example, by changes in the flow of minerals and petroleum from the ground, the negative effects of air pollution on the flow of food and fiber from agriculture, changes in precipitation and/or temperature associated with the accumulation of greenhouse gases, and the impact of pollution on the costs of manufactured goods through requirements for more frequent cleaning, repair, and replacement of materials. The effects of these changes will be transmitted to individuals through the price system in the form of changes in the costs and prices of final goods and services and changes in factor prices and incomes. All of these examples involve a common economic mechanism. Improvements in the resource base or environmental quality lower costs and prices, and increase the quantities of marketed goods, thus leading to increases in consumers', and perhaps producers', surpluses. Similarly, increases in pollution can cause economic harm to producers and consumers by decreasing their surpluses.

Most of the early studies of the effects of air and water pollution on producers were based on the damage function approach (Freeman 1982, chs. 5, 9). This approach involves (a) estimating a dose/damage function that relates some measure of pollution to a physical measure of damage; (b) applying this function to estimates of the inventory of materials exposed or at risk; and (c) multiplying the result by some unit value.

For example, in the case of damages to materials and structures, the physical damage might be corrosion, soiling, or loss of thickness of paint. This measure would have to be translated into an estimate of the increase in the frequency of some repair or replacement activity. The unit cost of this activity would provide the basis for estimating a monetary damage. The problem with this approach is that paint thickness and corrosion rates are not the economically relevant impacts. Rather, what is required is some understanding of the responses of producers in terms of increased frequency of repair, degree of degradation of performance, and so forth, all of which translate into increases in the cost of production. Similarly, in the case of agriculture the damage function approach focuses on

reduction in harvestable yield and multiplies this by a market price. However, this approach ignores adaptive behavior on the part of farmers and impacts on consumers resulting from possible changes in market price and changes in the cost of production of a crop.

Properly specified economic models of the effects of pollution on producers make use of cost functions or production functions to link the physical effects of changes in environmental quality to changes in market prices and quantities, and ultimately to changes in consumers' and producers' surpluses. Either directly, in the case of the production function approach, or implicitly, in the case of the cost function approach, these models incorporate the whole range of possible producers' responses to changes in pollution levels (through, for example, material substitution, increased protection activities, and changes in maintenance and repair schedules).

In the first section of this chapter, the theory is laid out in more detail for the case of single-product firms in a competitive industry. In the second section, the analysis is developed for the more likely case of multiproduct firms. In the third section, the method through which welfare effects on factor owners and consumers are passed through vertically linked markets for inputs and intermediate products when there are several intermediate stages of production are examined. In the fourth section, the effects that market distortions, such as monopoly power, have on welfare measures are considered. In the fifth section, simple models describing how the methods presented in this chapter can be used to value changes in the productivity of natural resource systems such as commercially exploited forests and fisheries are presented. For other treatments of many of these topics, see Bockstael and McConnell (2007), Just, Hueth, and Schmitz (2004), and McConnell and Bockstael (2005). The basic theory for estimation of productivity benefits is outlined in Chapter 4.

Basic Theory[1]

The purpose of this section is to derive welfare measures for changes in some parameter, q, which enters directly into the production functions of single-product firms. This parameter can be interpreted as a measure of environmental quality or the quality of a resource input into production, or alternatively, it could be a measure of innovation or technological change. Whatever the interpretation, an increase in this variable is assumed to increase the output attainable with any given set of inputs. Following on the earlier contributions of Anderson (1976), Schmalensee (1976), and Just and Hueth (1979), Just, Hueth, and Schmitz (2004) provided a rigorous analysis of how to measure changes in welfare due to price distortions in factor and product markets. These models provide a basis for analyzing the effects of productivity-induced changes in product and factor prices. As we will see, in the case of single-product firms the results that they derive for

1 This section is adapted, with permission, from Freeman and Harrington (1990).

price distortions carry over in a straightforward manner to the case of parametric shifts of cost and supply functions caused by environmental changes.

In this section, two alternative measures of welfare change for both marginal and nonmarginal changes in q for the case of a single-product firm are considered. One measure is the value of the marginal product of q. The other measure is based on the aggregate cost function for the industry and can be interpreted as the area between the old and new supply curves. The analysis in this section is essentially short-run, focusing on changes in quasi-rents to firms and on consumers' surpluses. In the long run, quasi-rents are competed away, except for those accruing to specialized factors owned by the firms. These rents can be viewed as increases in the prices of such factors.

Consider a competitive industry with N firms producing a single good, x. The ith firm is assumed to have a production function $x_i = x_i(\boldsymbol{V}_i, k_i, q)$, $(i = 1, \ldots, N)$, where $\boldsymbol{V}_i \equiv (v_{i1}, \ldots, v_{iJ})$ is a vector of variable factor inputs to firm i, k_i is a fixed factor, and q measures environmental or resource quality. For example, q could be a measure of air or water quality or natural soil fertility. Alternatively, q could be some measure of technical efficiency such as the rate at which an electrical generating station converts thermal energy to electricity. In this case, the model developed here would measure the benefits of investments in increasing technical efficiency. To simplify the notation, the k_i terms will be suppressed in all that follows. Both $\boldsymbol{V}_i$ and q have positive marginal products, but q is given exogenously to the industry.

Assume that the industry faces perfectly elastic supplies for all factor inputs at prices f_j, or in vector notation, $\boldsymbol{F}$. If factor supplies are less than perfectly elastic, the changes in rents to factor owners as factor prices change must also be included in the social welfare measure. Also, as Just, Hueth, and Schmitz (2004, ch. 9 and its appendix) show in their analysis of multimarket changes, general equilibrium supply and demand curves must be used, as discussed below.

Let aggregate industry output be denoted by $y = \sum_{i=1}^{N} x_i$. The industry faces an inverse demand function for its product, $p = p(y)$, where p is the market price, and income and all other prices are assumed constant. Define an aggregate production function

$$y = y(v_{11}, \ldots, v_{NJ}, q) \tag{8.1}$$

to be the sum of the N firm production functions.

The Production Function Approach

Now assume that demand functions are compensated so that consumer welfare changes can be measured by the appropriate areas. Then the social welfare W associated with producing y is the area under the demand curve for y, less the cost of the inputs:

$$W\left(v_{11}, \ldots, v_{N\bar{J}}, q\right) = \int_0^y p(u)\,du - \sum_{i=1}^{N} \boldsymbol{F} \cdot \boldsymbol{V}_i \ . \tag{8.2}$$

The first-order conditions for choosing the v_{ij} so as to maximize social welfare are

$$\frac{\partial W}{\partial v_{ij}} = \frac{\partial y}{\partial v_{ij}} \cdot p(y) - f_j = 0 \quad \text{for all } i, j \ . \tag{8.3}$$

Because each firm is a price taker, this welfare optimum is also the competitive equilibrium. These first-order conditions define input demand functions $v_{ij}^*\left(f_j, q\right)$, and, in turn, an output function

$$y^*(q) = y\left[v_{11}^*(f_1, q), \ldots, v_{N\bar{J}}^*\left(f_{\bar{J}}, q\right), q\right] \tag{8.4}$$

and a social welfare function

$$W(q) = y\left[v_{11}^*(f_1, q), \ldots, v_{N\bar{J}}^*\left(f_{\bar{J}}, q\right), q\right]. \tag{8.5}$$

The asterisks indicate optimally chosen quantities. In what follows, the f_j will not appear as function arguments, except where factor prices are assumed to be variable.

Using the envelope theorem, we have

$$\frac{\partial W}{\partial q} = p\left(y^*\right) \frac{\partial y^*\left[v^*(q), q\right]}{\partial q}. \tag{8.6}$$

The net welfare gain from an increase in q is the value of the marginal product of q in the production function. Note that $\partial y^* / \partial q$ is not the observed increase in y^*, but rather, it is the increase in y that would occur holding all other inputs constant. Knowledge of the production function is required to implement this measure.

The Cost Function Approach

The variable cost functions for firms can be added to obtain an aggregate variable cost function for the industry:

$$C = C\left[y^*(q), q\right] \ . \tag{8.7}$$

Assuming profit maximization with price equal to marginal cost, this also defines a market supply curve. The total contribution to social welfare made by the production and consumption of this good is the sum of compensating surpluses and quasi-rents to firms:

$$W = \int_0^y p(u)\,du - C\left(y^*, q\right). \tag{8.8}$$

This, of course, is maximized when output is set where price equals marginal cost,

$$p(y) = \frac{\partial C\left(y^*, q\right)}{\partial y} \ , \tag{8.9}$$

and, as before, the optimal output is a function of the parameter q.

Again applying the envelope theorem, we have the following expression for welfare change associated with marginal changes in q:

$$\frac{\partial W}{\partial q} = -\frac{\partial C\left(y^*, q\right)}{\partial q} \ . \tag{8.10}$$

This marginal value can be calculated if the cost function is known. The duality of the cost function and the production function ensures that the two measures of marginal welfare change given by equations (8.6) and (8.10) are equivalent.

The change in total welfare is positive if increasing q reduces costs. However, this fact does not necessarily imply that both producers' surplus and compensating surplus must increase. Compensating surplus (CS) is given by the area under the compensated demand curve less the actual expenditure, or

$$CS(q) = \int_0^{y^*} p(u)\,du - \left[p\left(y^*\right) \cdot y^*(q)\right]. \tag{8.11}$$

The marginal effect of a change in q is the derivative of this expression, or

$$\frac{\partial CS}{\partial q} = -\frac{\partial p}{\partial y^*} \cdot \frac{\partial y^*}{\partial q} \cdot y^* \ . \tag{8.12}$$

Of the three terms in equation (8.12), $\partial p / \partial y^* \leq 0$ by the law of demand, and $y^* > 0$. Thus, CS will decrease if, and only if, equilibrium output y^* decreases with an increase in q. This possibility cannot be ruled out.[2] It is possible for the marginal cost curve to increase over the relevant range, leading to an increase in price, even though total cost is reduced. Therefore, it is possible for consumers to be made worse off by an increase in q.

The marginal change in producers' quasi-rent, R, is the marginal welfare change given by equation (8.10) less the change in consumers' surplus given by equation (8.12); that is,

$$\frac{\partial R}{\partial q} = -\frac{\partial C}{\partial q} + \frac{\partial p}{\partial y^*} \cdot \frac{\partial y^*}{\partial q} \cdot y^* \ . \tag{8.13}$$

---

2 To determine the sign of $\partial y^* / \partial q$, differentiate the first-order condition (8.9):

$$\frac{\partial p}{\partial y^*} \cdot \frac{\partial y^*}{\partial q} = \left(\frac{\partial^2 C}{\partial y^{*2}} \cdot \frac{\partial y^*}{\partial q}\right) + \frac{\partial^2 C}{\partial y^* \partial q} \ ,$$

and rearrange to obtain

$$\frac{\partial y^*}{\partial q} = \frac{\left(\partial^2 C / \partial y^* \partial q\right)}{\left[\left(\partial p / \partial y^*\right) - \left(\partial^2 C / \partial y^{*2}\right)\right]}.$$

Since the denominator of this expression is always negative, $y^*(q)$ is decreasing whenever $\partial^2 C / \partial y^* \partial q$ is positive.

The first term is positive, while the second term is usually negative. Thus, quasi-rent could decrease only if the second term were larger in absolute value. The second term can also be expressed in terms of the price elasticity of demand:

$$\varepsilon = \frac{\partial y^*}{\partial p} \cdot \frac{p}{y^*} < 0 \,. \tag{8.14}$$

When this expression is substituted into equation (8.13), we see that the magnitude of the second term varies inversely with the elasticity of demand. For sufficiently inelastic demands, the fall in price brought about by the increase in quantity could actually harm producers by reducing their quasi-rents.

Nonmarginal Changes in q

For nonmarginal changes in q, say a change from q^0 to q^1, the aggregate benefit can be found by integrating either equation (8.6) or equation (8.10):

$$\begin{aligned} W_q &= \int_{q^0}^{q^1} p\left(y^*\right) \cdot \frac{\partial y\left[V^*(q), q\right]}{\partial q} dq \\ &= -\int_{q^0}^{q^1} \left\{ \frac{\partial C\left[y^*(q), q\right]}{\partial q} \right\} dq. \end{aligned} \tag{8.15}$$

As equation (8.15) indicates, both of these terms require integration along a path. To implement them, not only are we required to know either the production function or the cost function, but we also must know how equilibrium output changes with q, or how the levels of inputs change with q, or both.

However, W_q may be calculated directly from the demand curve and the cost function if the initial and final output levels y^{*0} and y^{*1} are known. This calculation is the change in the area bounded by the demand curve and cost curve:

$$W_q = \int_0^{y^{*1}} p(y)dy - C\left(y^{*1}, q^1\right) - \int_0^{y^{*0}} p(y)dy - C\left(y^{*0}, q^0\right). \tag{8.16}$$

This is shown in Figure 8.1, where $p_y(y)$ is the inverse demand curve and $MC_y\left(q^0\right)$ and $MC_y\left(q^1\right)$ indicate the industry marginal cost curves as a function of y^* and q. Graphically, equation (8.16) is equivalent to

$$\begin{aligned} W_q &= (a+b+c+d+e) - (c+e) - (a+b+c) + (b+c) \\ &= (b+d). \end{aligned} \tag{8.17}$$

The welfare change can be represented diagrammatically by the area between the old and new cost curves bounded by the demand curve. The division of the gain between producers and consumers is also shown graphically in Figure 8.2. Consumers gain the areas $u + v + w$, and producers lose area w but gain $s + t + u$.

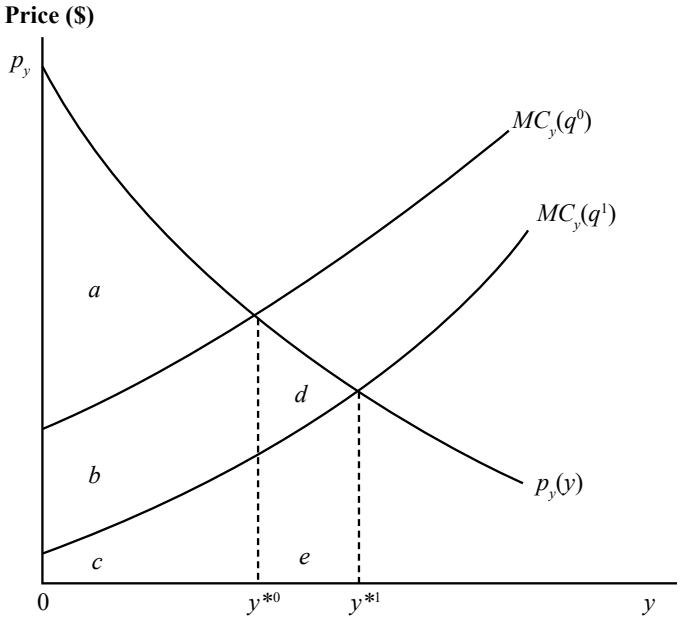

Figure 8.1 The welfare change for single-product firms

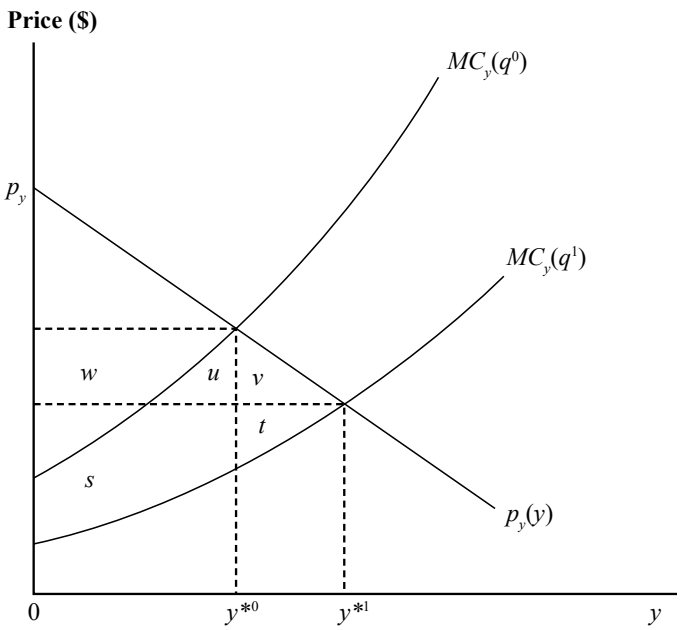

Figure 8.2 The division of the welfare gain between consumers and producers

The net social gain is $s + t + u + v$. If the cost and the demand functions are known, these areas can be calculated by taking the appropriate integrals.

Conclusions

In this section, two alternative measures are identified for the welfare change associated with a change in a nonmarket input such as q. One is based on shifts in the cost or supply curves (the cost function approach); the other is based on the direct impact of the change in q on the production function (the production function approach). Of course, because of duality, they are both based on essentially the same information and must give the same results. As a practical matter, the choice of approach depends on the availability of data.

In some cases, it may be possible to estimate the production function from data on physical inputs and outputs and environmental quality. For example, the availability of experimentally derived crop loss functions for ozone (see Heck et al. 1983) makes it feasible to use the production function approach to measure the benefits to agriculture of reducing ozone air pollution. Using time series data, Schlenker and Roberts (2009) estimated the relationship between corn, soybean, and cotton yields and temperature to assess the degree to which global warming scenarios will impact agriculture. Similarly, Auffhammer, Ramanathan, and Vincent (2006) estimated the effect that atmospheric black carbon (so-called brown clouds) and greenhouse gases have on rice harvests through their effects on solar radiation levels. Alternatively, one could estimate the cost function for an industry (Mathtech 1982) or the rent or profit function (Mjelde et al. 1984).

The production function and cost function methods are based on observations of the optimizing behavior of producers. An alternative to observing this behavior is to simulate it through formal optimization models. One example of this approach is the model used by Adams, Hamilton, and McCarl (1984) and Adams and McCarl (1985) to estimate the effects on the agricultural sector of controlling ozone air pollution. They combined a mathematical programming model of representative farms in the five Corn Belt states with an experimentally derived damage function to simulate producers' responses to changes in ozone levels. In another study of the economics of controlling ozone air pollution and agriculture, Kopp et al. (1985) constructed supply functions from region-specific farm budget and farm practice data. This model lacks the formal optimization characteristics of the Adams and McCarl model, but it allows for more detailed treatment of differences in farm economics across regions.

Multiproduct Firms

In the case of multiproduct firms with joint production technologies, things are not so simple. For marginal changes, the production function and cost function approaches can still be applied with appropriate modification. However, for nonmarginal changes, it is not correct simply to add up areas between observed shifts

of marginal cost or supply curves. When production is characterized by jointness, the model used for measurement must take account of the interconnectedness of the marginal cost or supply functions for the various outputs. The gain in welfare for an increase in q is measured by the sum of the changes in consumers' surpluses in the markets for all of the affected products plus the aggregate change in quasi-rents to the affected firms. However, in measuring the changes in quasi-rents, it is necessary to take account of how changes in the output of one good affect the marginal cost curves of the other goods being jointly produced.

For notational simplicity, assume there are only two products. The other notation is unchanged. The production function for the ith firm is given by

$$x_i\left(x_{1i}, x_{2i}, v_{i1}, \ldots, v_{iJ}; q\right) \geq 0. \tag{8.18}$$

That is, $x_i \geq 0$ is feasible production and $x_i = 0$ is efficient production. Also, assume $\partial x_i / \partial q \geq 0$, which assures that increasing q will increase the welfare associated with the production of x_{1i} and x_{2i}.

Both the aggregate production function and cost function approaches still provide a straightforward basis for measuring the marginal welfare change. As before, assume that each firm is a price taker in all of its product and factor markets and that the industry faces infinitely elastic factor supply curves. In a fashion similar to the analysis leading to equations (8.6) and (8.10) define the aggregate production function as the sum of the N firm production functions:

$$y\left(y_1, y_2, \boldsymbol{V}, \mathrm{q}\right) = 0 \ , \tag{8.19}$$

where $y_1 = \sum_{i=1}^{N} x_{1i}$, $y_2 = \sum_{i=1}^{N} x_{2i}$, and $\boldsymbol{V} = \left(v_{11}, \ldots, v_{NJ}\right)$ is the vector of all inputs for all firms. As before, the social welfare function is

$$W\left(\boldsymbol{V}, q\right) = \int_0^{y_1} p_1\left(u_1\right) du_1 + \int_0^{y_2} p_2\left(u_2\right) du_2 - \sum_{i=1}^{N} \boldsymbol{F} \cdot \boldsymbol{V}_i \ , \tag{8.20}$$

subject to $y\left(y_1, y_2, \boldsymbol{V}, q\right) = 0$.

The first-order conditions for a maximum of social welfare are

$$W_1 = p_1\left(y_1\right) - \lambda \partial y / \partial y_1 = 0 \, , \tag{8.21}$$

$$W_2 = p_2\left(y_2\right) - \lambda \partial y / \partial y_2 = 0 \, , \tag{8.22}$$

$$W_{vj} = -f_j - \lambda \partial y / \partial v_j = 0 \, , \tag{8.23}$$

where λ is the Lagrangian multiplier on the production constraint. These conditions define optimal $y_1^*(q)$, $y_2^*(q)$, and $\boldsymbol{V}^*(q)$ in the usual way. By the envelope theorem,

$$\partial W / \partial q = -\lambda \cdot \partial y / \partial q. \tag{8.24}$$

If we now differentiate the production function with respect to q, we have

$$\left(\frac{\partial y}{\partial y_1}\cdot\frac{\partial y_1^*}{\partial q}\right)+\left(\frac{\partial y}{\partial y_2}\cdot\frac{\partial y_2^*}{\partial q}\right)+\frac{\partial y}{\partial q}+\left(\sum_j\frac{\partial y}{\partial v_j}\cdot\frac{\partial v_j^*}{\partial q}\right)=0\,. \tag{8.25}$$

Combining this with equation (8.24) yields

$$\begin{aligned}\frac{\partial W}{\partial q}&=-\lambda\cdot\left(\frac{\partial y}{\partial q}\right)\\&=\left(\lambda\cdot\frac{\partial y}{\partial y_1}\cdot\frac{\partial y_1^*}{\partial q}\right)+\left(\lambda\cdot\frac{\partial y}{\partial y_2}\cdot\frac{\partial y_2^*}{\partial q}\right)\\&\quad+\frac{\partial y}{\partial q}+\left(\sum_j\lambda\cdot\frac{\partial y}{\partial v_j}\cdot\frac{\partial v_j^*}{\partial q}\right)=0.\end{aligned} \tag{8.26}$$

Substituting terms from the first-order conditions yields

$$\frac{\partial W}{\partial q}=\left[p_1\left(y_1^*\right)\cdot\frac{\partial y_1^*}{\partial q}\right]+\left[p_2\left(y_2^*\right)\cdot\frac{\partial y_2^*}{\partial q}\right]-\sum_j f_j\frac{\partial v_j^*}{\partial q}\,. \tag{8.27}$$

Alternatively, in terms of the aggregate cost function, $C\left(y_1^*, y_2^*, q\right)$, the benefits are

$$\frac{\partial W}{\partial q}=-\frac{\partial C\left(y_1^*, y_2^*, q\right)}{\partial q}\,. \tag{8.28}$$

In the case of nonmarginal changes in q, it is tempting to say that all that needs to be done is to add the areas between the new and old supply curves for each of the products of the multiproduct firms, but that would be wrong. Because of the interdependencies of the marginal cost curves under joint production, adding up these areas will not give a correct measure of the change in quasi-rents. The benefit of a nonmarginal change is the increase in the social values of the outputs, net of any changes in the joint cost of production. Again, for a change from q^0 to q^1 we have

$$W_q=\int_{y_1^0}^{y_1^1}p_1(y_1^*)dy_1+\int_{y_2^0}^{y_2^1}p_2(y_2^*)dy_2+\left[C\left(y_1^0, y_2^0, q^0\right)-C\left(y_1^1, y_2^1, q^1\right)\right]. \tag{8.29}$$

The change in joint costs, ř C, can be decomposed into three steps:

$$\begin{aligned}\Delta C&=\left[C\left(y_1^0, y_2^0, q^0\right)-C\left(y_1^0, y_2^0, q^1\right)\right]\\&\quad+\left[C\left(y_1^0, y_2^0, q^1\right)-C\left(y_1^1, y_2^0, q^1\right)\right]\\&\quad+\left[C\left(y_1^1, y_2^0, q^1\right)-C\left(y_1^1, y_2^1, q^1\right)\right].\end{aligned} \tag{8.30}$$

Substituting this into equation (8.29) gives

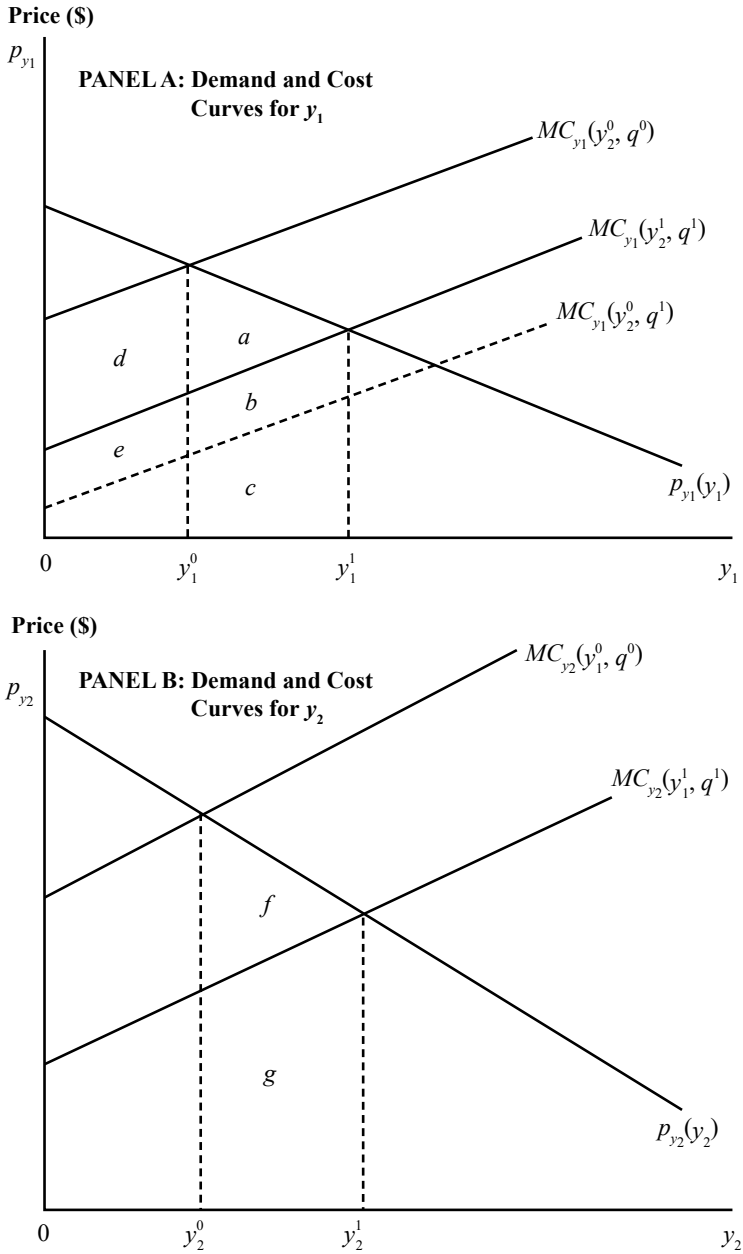

Figure 8.3 The welfare measure for multiproduct firms when outputs are substitutes in production

$$W_q = \int_{y_1^0}^{y_1^1} p_1(y_1^*)dy_1 + \int_{y_2^0}^{y_2^1} p_2(y_2^*)dy_2 - \int_{q^0}^{q^1}\left[\frac{\partial C\left(y_1^0, y_2^0, q\right)}{\partial q}\right]dq$$
$$-\int_{y_1^0}^{y_1^1}\left[\frac{\partial C\left(y_1, y_2^0, q^1\right)}{\partial q}\right]dy_1 - \int_{y_2^0}^{y_2^1}\left[\frac{\partial C\left(y_1^1, y_2, q^1\right)}{\partial q}\right]dy_2. \tag{8.31}$$

Note that in each of the markets one of the two marginal cost curves used in defining the welfare measure presented here is not actually observed. Implementation of equation (8.31) requires full knowledge of the joint cost function. Basing measures on comparisons of observed marginal cost curves before and after the change will result in error, the sign and magnitude of which will depend on the specific characteristics of the joint technology.

A graphical interpretation of equation (8.31) may prove to be helpful. First consider the case where y_1 and y_2 are substitutes in production in the sense that $\partial^2 C/\partial y_1 \partial y_2 = \partial^2 C/\partial y_2 \partial y_1 > 0$. This case is shown in Figure 8.3. The solid cost curves represent the pre- and post-change observed marginal cost curves. The dashed cost curve in panel A corresponds to the fourth integral in (8.31), where q has changed holding y_2 at y_2^0.

The geometric areas in panels A and B corresponding to the five integrals in equation (8.29) are given by the following:

$$\begin{aligned} W_q &= (a + b + c) + (f + g) + (d + e) - c - g \\ &= a + b + d + e + f. \end{aligned} \tag{8.32}$$

The welfare change is measured by the area between the two cost curves for y_1 holding y_2 at y_2^0 plus the observed welfare triangle f in the market for y_2 (panel B). An empirical measure based on areas between observed cost curves would yield an underestimate of the true welfare change.

In contrast, Figure 8.4 shows the case where y_1 and y_2 are complements in production—that is $\partial^2 C/\partial y_1 \partial y_2 = \partial^2 C/\partial y_2 \partial y_1 < 0$ geometric equivalent to equation (8.31) is

$$\begin{aligned} W_q &= (a + b + c) + (f + g) + d - (b + c + e) - g \\ &= a + d - e + f. \end{aligned} \tag{8.33}$$

Again, measuring the first term requires knowledge of the unobserved cost curve for y_1 after the change in q but holding y_2 at y_2^0.

It is of interest to know the magnitude of the error that would result from failure to recognize the existence of joint costs. To get an approximation of the error, form a second-order Taylor polynomial expanded around $\left(y_1^0, y_2^0, q^0\right)$. The approximate change in cost is

$$\Delta C = C\left(y_1^1\ y_2^1, q^1\right) - C\left(y_1^0\ y_2^0, q^0\right) = \Delta C^1 + \Delta C^2 + \Delta C^J, \tag{8.34}$$

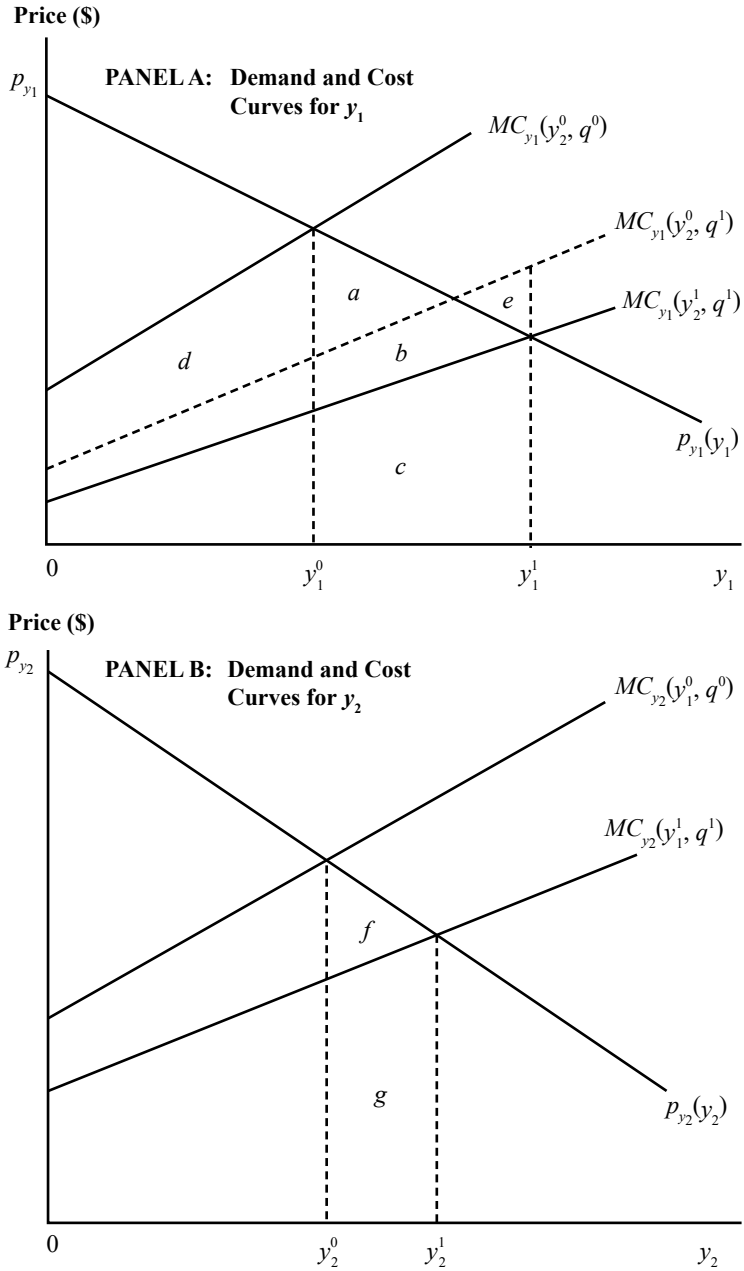

Figure 8.4 The welfare measure for multiproduct firms when outputs are complements in production

where

$$\Delta C^1 = \left[\left(y_1^1 - y_1^0\right)\cdot\frac{\partial C}{\partial y_1}\right] + \left[\frac{1}{2}\left(y_1^1 - y_1^0\right)^2\cdot\frac{\partial^2 C}{\partial y_1^2}\right] + \left[\left(y_1^1 - y_1^0\right)\left(q^1 - q^0\right)\frac{\partial^2 C}{\partial y_1 \partial q}\right] \tag{8.35}$$

$$\Delta C^2 = \left[\left(y_2^1 - y_2^0\right)\cdot\frac{\partial C}{\partial y_2}\right] + \left[\frac{1}{2}\left(y_2^1 - y_2^0\right)^2\cdot\frac{\partial^2 C}{\partial y_2}\right] + \left[\left(y_2^1 - y_2^0\right)\left(q^1 - q^0\right)\frac{\partial^2 C}{\partial y_2 \partial q}\right] \tag{8.36}$$

$$\Delta C^J = \left(y_1^1 - y_1^0\right)\left(y_2^1 - y_2^0\right)\frac{\partial^2 C}{\partial y_1 \partial y_2}\cdot \tag{8.37}$$

The third term, ř C^J, is the contribution to the change in cost by the jointness in y_1 and y_2.

There is another way of looking at the difficulties involved here. Recall that one component of the welfare change is the change in quasi-rents to firms. With joint production, it is generally not possible to measure the quasi-rent of a multiproduct firm from data from just one market. However, Just, Hueth, and Schmitz (2004) have shown that in the case of price changes, there are circumstances in which it is possible to measure the change in quasi-rents in either a single-factor market or a single-product market. Their analysis can be extended to the case of parametric shifts in the production technology. What is required is that there is either a necessary input or a necessary output. A necessary output is one for which there is some positive minimum price at which the firm will choose to stop producing not only that output but all other products as well—in other words, production will shut down completely. All of the quasi-rent to a firm can be attributed to the necessary output and can be measured by integrating above the supply curve for that output from the shutdown price to the current market price. Alternatively, the quasi-rent can be measured by the area under the demand curve for any necessary factor input. An input is deemed to be necessary if there is some price for that input at which its derived demand falls to zero, and if all of the firm's outputs fall to zero when that input is set at zero.

The measurement of changes in quasi-rents can be shown graphically for the case of two necessary outputs, y_1 and y_2, and one necessary input, v_1. Suppose that, as shown in Figure 8.5, an increase in q shifts the supply functions for both outputs and the demand function for the necessary input to the right. The consumer surplus component of the welfare change is the areas $w_1 + v_1 + w_2 + v_2$ in panels B and C. The increase in the quasi-rents to all firms can be measured alternatively by $u_1 + t_1 - w_1$ in panel B, or $u_2 + t_2 - w_2$ in panel C, or b in panel A.

To see this, first suppose that y_1 is the necessary output. At q^0 if p_1 is at p_1^0, other things being equal, firms shut down and quasi-rents are zero. If p_1 increases to p_1^1,

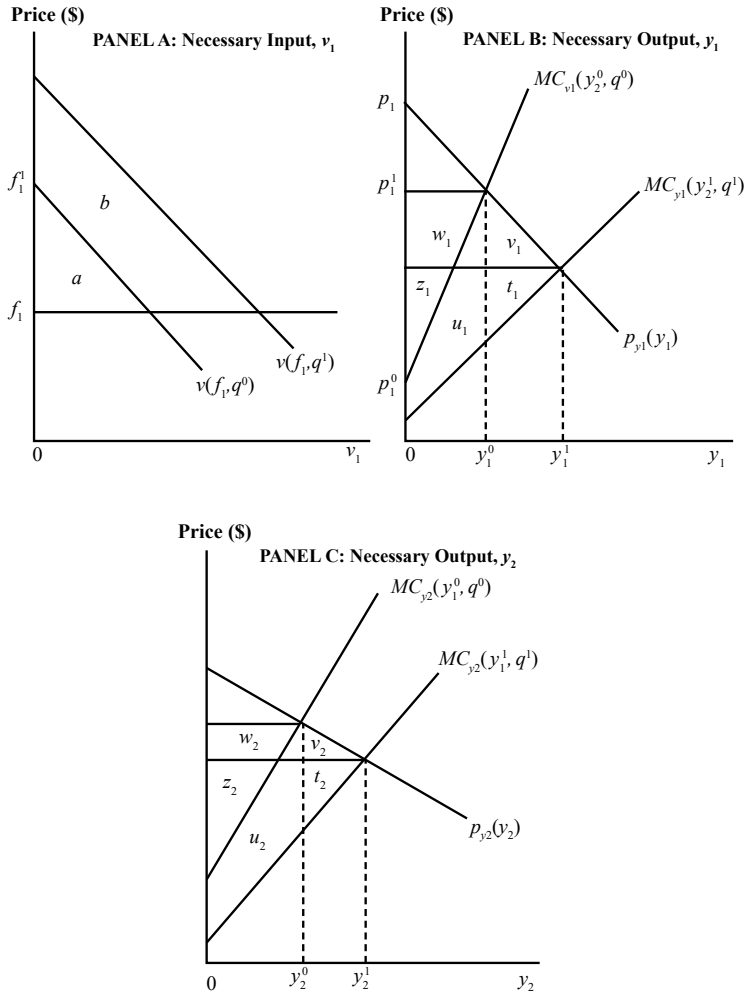

Figure 8.5 The welfare gain for multiproduct firms with necessary outputs

firms produce both goods in positive quantities; but all of the quasi-rents are attributable to the increase in p_1 and are measured by the area $w_1 + z_1$. Comparing quasi-rents at q^1 and q^0 gives $\Delta R = R^1 - R^0 = u_1 + t_1 + z_1 - w_1 - z_1 = u_1 + t_1 - w_1$. For the necessary input v_1, if its price is f_1^1, firms shut down and quasi-rents are zero. If f_1 falls, firms produce both goods. All of the quasi-rents are attributable to the increased use of v_1 and are measured by the area a.

The three alternative measures of the total welfare gain are:

y_1 is necessary:

$$\begin{aligned} W_{1q} &= w_1 + v_1 + t_1 + w_2 + v_2 + u_1 - w_1 \\ &= u_1 + v_1 + t_1 + w_2 + v_2 \end{aligned} \tag{8.38}$$

y_2 is necessary:

$$W_{2q} = w_1 + v_1 + w_2 + v_2 + u_2 + t_2 - w_2 = w_1 + v_1 + u_2 + v_2 + t_2 \tag{8.39}$$

v_1 is necessary:

$$W_{vq} = W_1 + v_1 + w_2 + v_2 + b. \tag{8.40}$$

Where empirical measures of producers' benefits have been reported in the literature, production has typically been modeled either implicitly or explicitly as being nonjoint, even in those cases where multiproduct firms are typical. For example, Adams, Hamilton, and McCarl (1984), Adams and McCarl (1985), Kopp et al. (1985), and Adams, Crocker, and Katz (1984) have all provided estimates of the benefits to agriculture of controlling ozone air pollution, using variations of the cost function approach. The first two studies used supply functions derived from models of farm behavior rather than from econometric estimates. Independence of the marginal cost or supply curves by crop was a characteristic of these models. Adams, Crocker, and Katz (1984) used econometric estimates of supply functions for four crops; but the estimating equations imposed nonjointness on the production technology even though many farms produced at least two of the four crops modeled (corn and soybeans). Other researchers, such as Mjelde et al. (1984), Garcia et al. (1986), and Mathtech (1982), aggregated across products to obtain industry profit or cost functions. However, consistent aggregation of this sort is possible only if marginal cost functions are independent, a condition that is not satisfied when there is joint production. The analysis presented here suggests that in the future it will be important to confront the question of joint production directly. If in fact the industries being studied are characterized by joint production, the empirical models of production must reflect this; and the measures of welfare change must take this into account in the way just described.

Vertically Linked Markets

The next case to consider is that of vertically linked markets in which the output of one set of firms is purchased as an input by another set of firms. Assume that q affects costs and prices in one industry that is part of a set of vertically linked industries. The model used to derive welfare measures in this case must take account of the fact that every price change affects both buyers and sellers, but in different directions; that is, for a price increase, the buyer loses while the seller gains, and vice versa for price decreases. For simplicity, in what follows assume that all firms are single-product firms. Some firms are producers that purchase primary factor inputs and sell an intermediate product y_w. Other firms buy the intermediate product and factor inputs and sell y_r at retail to consumers. Assume that all factor supply curves but one are infinitely elastic; the exception is the supply curve for labor used in the wholesale industry, v_w. Assume also that the compensated supply curve for v_w is upward sloping and that y_w and y_r are produced and sold in perfectly competitive markets. Figure 8.6 shows the case.

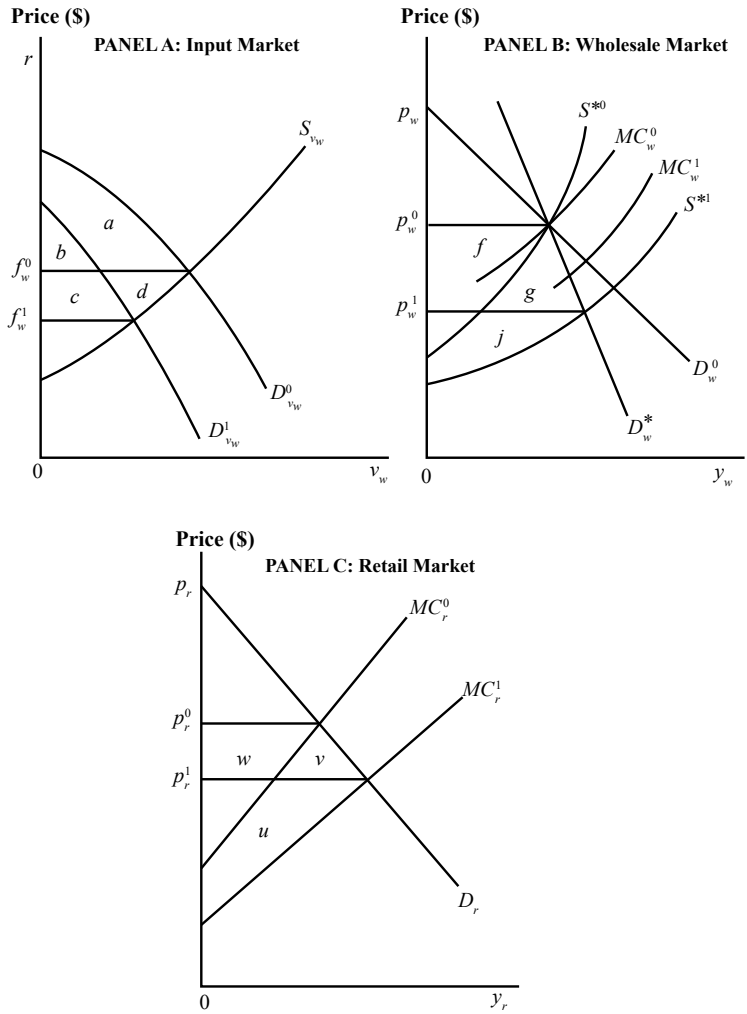

Figure 8.6 Measuring welfare gains in vertically linked markets

Assume now that q affects the producers of y_w. Suppose that at the initial level of q the supply function for y_w is MC_w^0 and the demand curve is D_w^0 as shown in panel B of Figure 8.6. The first effect of an increase in q is to shift MC_w down to MC_w^1. This results in a similar downward shift of MC_r and a decrease in the retail price, as shown in panel C of Figure 8.6. In the wholesale market, the demand curve for y_w is derived on the assumption that the retail price is constant, but when the retail price falls, the derived demand for y_w shifts to the left. The wholesale producers respond to the changes in demand along a path that reflects the adjustments in price in the retail market. The path of this adjustment can

be described by what Just, Hueth, and Schmitz (2004) call a general equilibrium demand curve. This demand curve shows the maximum willingness to pay by the y_r industry given its costs and the retail price at which it must sell its product, and assuming that all other input prices are constant. This is shown as D_w^* in panel B of Figure 8.6.

At the same time, the demand curve for v_w in panel A of Figure 8.6 is shifting for two reasons. First, the decrease in p_w pushes the demand curve for v_w to the left. Second, the change in q also affects the productivity of, and demand for, v_w. The direction of this effect depends on the specific characteristics of the production function and on the elasticity of demand for y_w. As the price of v_w changes, this shifts MC_w in the same direction. There are also two general equilibrium supply functions, S^{*0} and S^{*1}, in panel B of Figure 8.6, one for each level of q. They describe the supply adjustments to changing factor prices, and they show the minimum supply price for y_w for each level of q, taking account of changes in factor prices as the derived demand for v_w shifts. The net result is an equilibrium in this market at the intersection of D_w^* and S^{*1}, as shown in panel B of Figure 8.6.

Three groups of economic agents are affected by the change in q, so there are three components to the measure of welfare gain. The first group, the consumers of y_r, receives an increase in consumer surplus, shown as $w + v$ in panel C of Figure 8.6. The second group, the owners of v_w, experiences a change in their rents because of the change in its price. They may either gain or lose depending upon how the demand for v_w changes; in Figure 8.6, panel A, they lose the area $c + d$. The third group consists of the firms in the wholesale and retail industries that experience changes in their quasi-rents. The change in quasi-rents in the retail industry can be measured directly in the retail market by the area $u - w$. As discussed above, if v_w is a necessary input, then the change in quasi-rents to the wholesale industry can be measured in the factor market by the area $c - a$. Considering all of the changes, and netting out transfers between buyers and sellers resulting from price changes, we have

$$\begin{aligned} W_q &= \Delta PS_v + \Delta R_w + \Delta R_r + \Delta CS_r \\ &= -(c+d) + (c-a) + (u-w) + (w+v) \\ &= -(d+a) + (u+v). \end{aligned} \tag{8.41}$$

Following Just, Hueth, and Schmitz (2004), it can be shown that the welfare gain can also be measured in the directly affected wholesale market, provided that the measurements are based on the general equilibrium supply and demand curves. These curves reflect the adjustments of prices in linked markets that transfer gains and losses among buyers and sellers. Specifically,

$$W_q = \Delta CS_w^* + \Delta R_w^*, \tag{8.42}$$

where $C_w^*(= \Delta R_r + \Delta CS_r)$ and $R_w^*(= \Delta R_w + \Delta PS_v)$ are the areas behind the general equilibrium demand and supply curves respectively. So from equation (8.42),

$$W_q = f + g + j - f = g + j. \tag{8.43}$$

Thus, if general equilibrium demand and supply functions for the directly impacted market can be estimated, it may be more convenient to measure welfare gains using this expression than to attempt to measure separately the changes in quasi-rents and surpluses in all of the remaining vertically linked markets.

Now consider the case of multiproduct firms that are part of a chain of vertically linked markets. As just shown, the welfare gain can be measured in the directly affected market. However, the welfare measure must be modified to take account of the multiproduct characteristic of the producing industry. Specifically, if there is any jointness in production, it is not correct simply to add the areas $g + j$ across all products. Rather, the correct measure is the sum of the consumer surplus changes as measured by areas like $f + g$, and the change in producing firms' quasi-rents as measured by $j - f$, in the market for one essential output.

Monopoly Markets

The welfare measures derived in the preceding sections come from models in which all markets are perfectly competitive. In this section the effects on welfare measurement when there is monopoly power in the output market is considered.

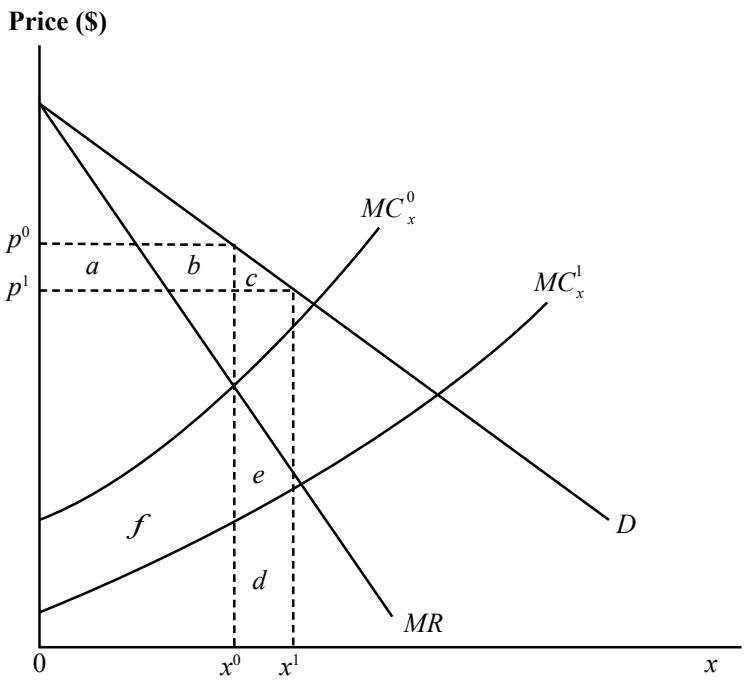

Figure 8.7 The welfare change in a monopoly market

The correct welfare measure in the case of monopoly can be derived graphically in a straightforward manner, but measurement is another question. Figure 8.7 shows the demand, marginal revenue, and marginal cost curves for a monopolist. If an increase in q shifts the marginal cost curve outward, the welfare measure is the sum of the increases in consumer surplus and monopoly quasi-rents:

$$W_q = \Delta CS + \Delta R, \tag{8.44}$$

where

$$\Delta CS = a + b + c \tag{8.45}$$

$$\Delta R = \Delta TR - \Delta C = d + e - (d - f) = e + f. \tag{8.46}$$

Thus,

$$W_q = a + b + c + e + f. \tag{8.47}$$

As in the case of perfect competition, consumers always gain because of the price reduction, but in this case the monopolist also always gains from an increase in q. The monopolist's profit is

$$I = p(x) \cdot x - C(x,q) \tag{8.48}$$

$$\frac{\partial I}{\partial q} = \left(p + x\frac{dp}{dx} - \frac{\partial C}{\partial x} \right)\frac{dx}{dq} - \frac{\partial C}{\partial q}. \tag{8.49}$$

The first-order condition for profit maximization requires that the term in parentheses be equal to zero. So,

$$\frac{\partial I}{\partial q} = -\frac{\partial C}{\partial q} > 0. \tag{8.50}$$

The problem posed for measurement is that the marginal cost curve of the monopolist cannot be observed from market data on optimally chosen prices and quantities. One possible approach to deriving marginal cost functions is to construct models of the firm based on engineering or technological data (see Russell and Vaughan 1976 for an example).

Valuing Changes in the Productivity of Natural Resource Systems

The productivity of commercially exploited natural biological systems can depend on such things as the flow of nutrients into the system, the population of a predator species, climatic variables such as precipitation and temperature, or the level of a pollutant. Any of these factors might be subject to human manipulation, so it would be useful to have an economic framework for evaluating the welfare consequences of policies to change these things. Since the outputs of such resource systems are traded in markets, the framework presented in this chapter is

applicable. The welfare consequences come in the form of changes in producers' and consumers' surpluses. However, there are some special features of natural resource systems that sometimes need to be taken into account.

This section provides a brief description of how an environmental quality variable can be introduced into the standard economic models of two types of natural resource systems—the commercial forest and the commercial fishery. In the case of the forest, an investment in environmental quality may alter the optimum time of harvest, so the intertemporal features of the forest optimization model must be examined. Moreover, an environmental quality variable might affect the economic value of a nonmarket output such as recreation. Thus, the interaction between the nonmarket and market outputs must be analyzed in a multiple-use framework.

In the case of the fishery, there are two commonly used bio-economic models in the literature, the Schaefer–Gordon growth model, and the Beverton–Holt stock-recruitment model. Environmental parameters can be easily introduced into both models, but the economic implications of a change in environmental quality are more transparent in the case of the Schaefer–Gordon model, as will be shown. In addition, the economic value of an environmental quality change will depend upon the institutions for ownership or management of the fishery. This is demonstrated by a comparison of the economic value of an environmental change under the alternatives of optimum management and open-access exploitation.

Commercial Forests and the Role of Time

The models described in the preceding sections of this chapter were timeless in the sense that environmental changes and changes in prices and quantities were contemporaneous. There was no need to model explicitly any intertemporal effects. Where the linkages between environmental changes and market changes are not contemporaneous, time must be built into the model explicitly. An example is the case of the effects of changes in air quality on commercial forest productivity.

One of the things that makes the case of the effects of environmental change on commercial forests interesting from an economic perspective is that a change affecting a stand of young trees today will not have an effect on marketed outputs for perhaps 40 years, when the trees are harvested and sold. If there were an active market in forestland, any increase in the growth rate of young trees now would have an immediate impact on the market price of land where there are standing, growing trees. Even so, any consumer surplus benefit from lower prices of forest products would not be realized until the increased harvests actually took place. In what follows, the product price effects on consumers are ignored and only rents accruing to forests and owners are considered.

Suppose that the trees on a plot of land grow in net value over time according to

$$G_t = G(t,q), \tag{8.51}$$

where G_t is the stumpage value—that is, the market price of the harvested volume at age t, net of harvest costs and transportation to the mill. This is a very general formulation that allows for growth in harvestable volume as well as for changes in the price per unit of volume because of changes in quality. We also assume that replanting and management costs are zero.

Assuming no economically relevant alternative uses for a unit of land, the landowner's economic problem is to choose a sequence of harvesting dates to maximize the present value (V) of the stream of net receipts at each harvest—that is,

$$\begin{aligned} \underset{t}{\text{Max}} : V &= \sum_{h=1}^{\infty} e^{-h \cdot r \cdot t} G(t, q) \\ &= \frac{G(t, q) e^{-r \cdot t}}{1 - e^{-r \cdot t}} \end{aligned} \tag{8.52}$$

where t is the age of the stand at the time of harvest, r is the interest rate, and h indexes the generation of the stand. For simplicity, assume that there are no costs for planting, thinning, and other management activities during the rotation. For a more complete treatment of the forestry optimization problem, see Samuelson (1976), Hyde (1980), and Bowes and Krutilla (1989).

The stand should be harvested at the age that satisfies

$$\frac{\partial V}{\partial t} = \frac{\dfrac{\partial G}{\partial t} - \left[r \cdot G(\cdot)\right] - \dfrac{r \cdot G(\cdot) e^{-r \cdot t}}{1 - e^{-r \cdot t}}}{1 - e^{-r \cdot t}} = 0 , \tag{8.53}$$

or

$$\frac{\partial G}{\partial t} = \left[r \cdot G(\cdot)\right] + \frac{r \cdot G(\cdot) e^{-r \cdot t}}{1 - e^{-r \cdot t}} . \tag{8.54}$$

The stand should be allowed to grow as long as the marginal gain in value through growth $\partial G / \partial t$ exceeds the interest forgone by not realizing G through harvest ($r \cdot G$) plus the opportunity cost of postponing the stream of returns from future rotations. Harvest should occur when the marginal gain from waiting just equals the marginal opportunity cost.

Two questions of interest are the effects of changes in q on V and on the optimum rotation length. From equation (8.52):

$$\frac{\partial V}{\partial q} = \frac{\partial G}{\partial q} \cdot \frac{e^{-r \cdot t}}{1 - e^{-r \cdot t}} . \tag{8.55}$$

The increase in q increases G at the time of the next harvest, and the second term in this expression gives the present value of the stream of these increases over the infinite future. So, if the growth function were known, including the effect of q, then calculating the effect of changes in the steady-state level of q on the stream of rents would be straightforward. However, q often varies over time, because, for

example, of trends in pollution associated with economic growth: therefore, such calculations would require more detailed knowledge of the growth function than is currently available. Answering the second question appears to be more difficult. If t^* is the solution to equation (8.54), it appears to be impossible to determine unambiguously the sign of $\partial t^* / \partial q$.

Environmental Quality and Multiple-Use Management of Forests

Suppose that in addition to the periodic harvest of marketable products, the forest unit provides a flow of nonmarket services such as recreation that depends on the age of the forest and on q. Models for measuring the value of recreational resources are discussed in Chapter 9.

Let the value of the nonmarket service be given by $R(t,q)$. Over one harvest cycle, the present value of the nonmarket service flow is

$$V_n = \int_0^{t^*} R(t,q)dt. \tag{8.56}$$

The objective function becomes:

$$\underset{t}{\text{Max}} : V = \sum_{h=1}^{\infty} e^{-h \cdot r \cdot t} \left[G(t, q) + \int_0^t R(t, q)dt \right]. \tag{8.57}$$

As Hartman (1976) first showed, including the value from the standing forest in the objective function will either increase the optimum age at harvest (t^*) or make harvesting uneconomic entirely. This is because the standing forest adds a second term, the marginal benefit of delaying the harvest, to the left-hand side of equation (8.54). The optimum harvest (if it exists) occurs when

$$\frac{\partial G}{\partial t} + R(\cdot) = r \cdot G(\cdot) + \frac{r \cdot G(\cdot) e^{-r \cdot t}}{1 - e^{-r \cdot t}}. \tag{8.58}$$

An increase in q can be valuable both because of its increase in the present value of the flow of harvestable product and because of its effect on the value of the flow of services from the standing forest. For further discussion of the Hartman result and multiple-use forest management in general, see Bowes and Krutilla (1989), Swallow and Wear (1993), and Swallow, Talukdar, and Wear (1997).

One benefit of modeling exercises of this sort is to provide guidance for the natural science research that is required to support future economic analyses. As these models suggest, it is important to know how changes in q affect the whole time pattern of the growth of trees and the flow of nonmarket services over the life of the forest.

Environmental Quality and Commercial Fisheries

The economic analysis of fisheries rests on a foundation of a biological model of the growth and mortality of a species. Two alternative biological models have

dominated the literature on the economics of fisheries. The first is the so-called Schaefer–Gordon model, which makes the growth rate of the aggregate biomass of the species at any point in time a function of the current level of biomass. The alternative Beverton–Holt model explicitly describes both the number of fish in each age cohort and their weight or size. The Beverton–Holt model may be more realistic from a biological perspective; but it is also more complex from an economic perspective.

The Schaefer–Gordon Model

In this model the relationship between the growth of the aggregate stock, g, and the aggregate size of the stock, z, takes the following form:

$$g = b \cdot z\left(1 - \frac{z}{k}\right) = b \cdot z - \frac{b}{k} \cdot z^2, \tag{8.59}$$

where b is the intrinsic growth rate and k is the carrying capacity of the environment. As this shows, when the stock has grown to the carrying capacity $(z = k)$, the growth rate is zero. This quadratic growth function results in a logistic time path for the stock in the absence of harvest, with z approaching the carrying capacity asymptotically.

Environmental parameters have been incorporated into the Schaefer–Gordon growth model in several empirical studies of fisheries. Either the intrinsic growth rate or the carrying capacity (or both) can be made a function of an environmental quality parameter, q. For example, in his study of the North American lobster fishery, Bell (1972) included seawater temperature as one variable helping to explain the annual harvest. Lynne, Conroy, and Prochaska (1981) examined the effects of changes in the acreage of marine wetlands on annual harvests of blue crabs in the Florida Gulf Coast fishery. For other examples, see Bell (1989), Swallow (1994), and Barbier and Strand (1998).

The Schaefer–Gordon model can be incorporated into an economic model to determine the optimum levels of harvest and stock. This model can also be used to trace the effects of changes in environmental parameters on harvest, stock, and economic welfare. If an environmental quality parameter can be increased by public policy at some cost, optimum levels of investment in environmental quality can also be determined.

The standard approach is to specify a production function that makes the fishery industry's annual harvest, h, a function of economic inputs and the stock of fish to be caught, z. For simplicity, the economic inputs are aggregated into a measure of effort, e. A unit of effort can be called a "boat" and can be interpreted as an optimal combination of labor and capital. Assume that each unit of effort has a cost of p_e. For any chosen level of h, the production function can be solved for the required level of effort, given the size of the stock. Since the size of the stock at any point and time is the net result of past growth (which depends in part on q) and past harvest, the cost function can be written as

$$C = p_e \cdot e = C(h, z, q). \tag{8.60}$$

Managing the fishery for economic objectives is a dynamic problem, since current harvests and costs depend on past harvest decisions and how they have affected the size of the stock. The economic objective is to maximize the present value of the net economic return from the fishery over time, subject to the biological constraint imposed by the growth function. If $p_h(h)$ is the inverse demand function for fish, the objective function is

$$\underset{h_t}{\text{Max}}: \int_0^{\infty} \left[\int_0^{h} p_h(h_t)dh - C(h_t, z_t, q) \right] e^{-r \cdot t} \cdot dt, \tag{8.61}$$

subject to

$$\frac{dz}{dt} = g(z_t, q) - h_t. \tag{8.62}$$

For any given initial conditions, this problem can be solved for the sequence of harvests that maximizes the objective function. The economic dimensions of the problem are easiest to see if we focus attention on the conditions for the long-run, steady-state equilibrium where growth and harvest are equal and the stock is of constant size. The first-order conditions for this optimum are

$$p_h = \frac{\partial C}{\partial h} + \rho \tag{8.63}$$

and

$$r = \frac{\partial g}{\partial z} - \frac{1}{\rho} \cdot \frac{\partial C}{\partial z}, \tag{8.64}$$

where r is the interest rate and ρ is the shadow value of the stock, reflecting its effect on the rate of growth and the cost of harvest. See, for example, Fisher (1981) or Clark (1976, 1985). Equation (8.63) says that the price of fish must equal the full marginal cost of their harvest, which includes the opportunity cost of decreasing the stock, ρ. Equation (8.64) defines the optimum intertemporal tradeoff. The interest rate is the opportunity cost of forgoing $1 worth of harvest now, and must equal the benefit of forgoing harvest, which has two components. The first component is the contribution the additional stock makes to future growth. The second reflects the contribution that the extra stock makes to lowering harvest costs.

Valuation of Changes in q *in the Schaefer–Gordon Model*

An increase in q shifts the cost function in equation (8.61) and changes the growth function of (8.62). This leads to new solution values for h and z in every period. Assuming optimal management and the satisfaction of (8.63) and (8.64), the welfare value in each period is the increase in the net value term in brackets in equation (8.61)—that is, the area between the marginal cost curves bounded by the demand curve. The stream of changes in that value must be discounted to its present value to obtain the total welfare value of the change in q.

Taking wetland acreage as a measure of q, Ellis and Fisher (1987) drew on earlier work by Lynne, Conroy, and Prochaska (1981) to calculate the net welfare value of increases in wetlands in the blue crab fishery of the Gulf Coast of Florida. They abstracted from the dynamic, intertemporal dimension of the fishery management problem by focusing only on a single-period optimum and by assuming that the cost of harvest was independent of this stock. This is equivalent to asserting that $\rho = 0$, so that an optimum is defined by the price of harvest equaling its marginal cost.

However, the economic value of a change in environmental quality depends not only on the economic and biological parameters of the model, but also on the institutional arrangements for ownership or public management of the resource. The preceding discussion (and the Ellis–Fisher analysis) is based implicitly on the assumption of either private ownership with perfect competition or public regulation to achieve the economic optimum. However, most fisheries resources are characterized by absence of private ownership and more or less open access to the resource, and where there is public regulation, it is seldom designed to achieve an economically efficient outcome. Consequently, the net economic value of the resource will be lower; and the economic value of changes in q will also be affected by the ownership and management arrangements.

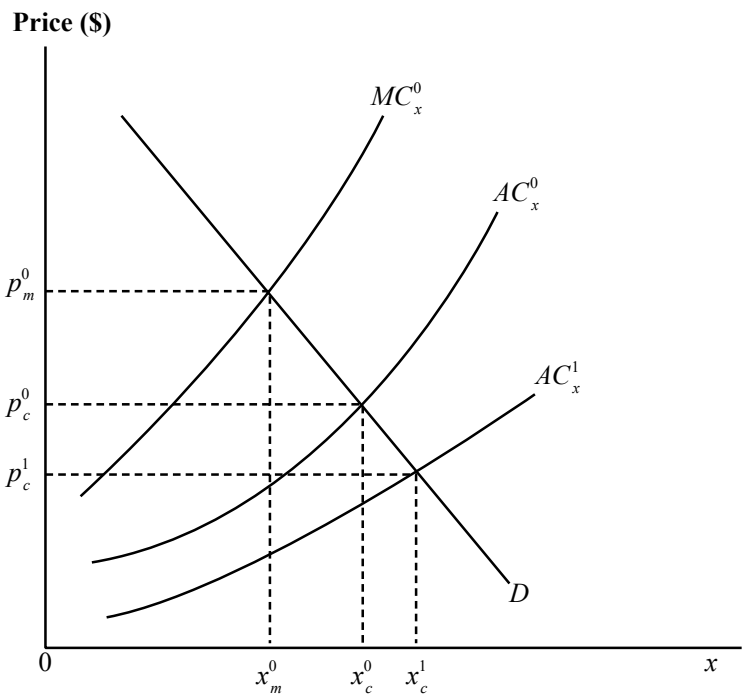

Figure 8.8 Welfare measurement for open-access resources: the case of a fishery

A simple static model can be used to show the effect of open access on the welfare value of a change in the productivity of the fisheries resource such as might result from an improvement in water quality (see also McConnell and Strand 1989). It is well known that under open access without regulation, competition drives rent to zero (Gordon 1954; Scott 1955). Let MC_x^0 in Figure 8.8 be the marginal cost of harvesting fish at the original level of q. If the fishery is privately owned, or if it is optimally managed, the output and price will be given by the intersection of the marginal cost of harvest function and the demand curve at p_m^0 and x_m^0. An improvement in q would shift the marginal cost of harvest function outward to the right, resulting in a higher quantity and a lower equilibrium price. The new marginal cost curve is not shown in Figure 8.8. The welfare measure would be the same as that presented above.

If there is open access to the fishery, the condition for equilibrium is each fisherman earning zero profits. This requires that price be equal to the average cost of harvesting fish. Compared to the efficient price and output under private ownership, more fishermen enter in pursuit of profit. The increase in fish caught decreases the stock, raising costs or decreasing the price of fish, or both. Entry continues until these forces eliminate the incentive for entry. At the initial level of q, the average cost of harvesting fish is AC_x^0, as shown in Figure 8.8. The open-access equilibrium is at p_c^0 and x_c^0.

An improvement in q also shifts the average cost curve outward, resulting in a lower price and a higher quantity; however, since price equals average cost both before and after the change in environmental quality, there is no change in producer surplus. The benefit consists entirely of the increase in consumer surplus associated with the price decrease. The more elastic is the demand curve for fish, the smaller is the welfare gain associated with the environmental improvement. Freeman (1991) presented some illustrative calculations of the magnitude of these effects based on the Ellis–Fisher data. In the limit, if this fishery is small relative to the market and the demand for fish is perfectly elastic, there is no welfare gain under open access. The physical improvement in productivity brought about by the higher water quality is entirely dissipated by the uneconomic competition of fishermen for the potential increase in rents. However, with inelastic demand the welfare gain from an increase in q is slightly higher under open access than under optimal regulation.

Empirical Examples

Problems associated with nutrient over-enrichment in many freshwater and coastal systems have been documented around the world and are increasingly receiving policy attention (Rabotyagov et al. 2014). A myriad of ecosystem services are affected by low oxygen levels (hypoxic conditions) and "dead zones" that result from excess nutrient flowing into these systems. One potentially important welfare consequences of this water quality problem is damage to commercial fisheries and the methods described here are directly relevant.

Several papers have studied the welfare losses associated with eutrophication from excess nutrients flowing into the Pamlico Sound estuary, which is located in North Carolina. Huang, Smith, and Craig (2010) employed a spatial bioeconomic model to relate hypoxic conditions to brown shrimp stock and found that hypoxic conditions accounted for about a 13 percent decline in harvest in the early 2000s. Using the bioeconomic model developed by Huang, Smith, and Craig, Huang et al. (2012) studied the welfare effects of hypoxic conditions considering both supply and demand effects. Because the North Carolina shrimp industry is small in the world market, and therefore demand is elastic, the authors found that there is almost no loss in consumer surplus from hypoxic conditions in the region. While there are losses in producer surplus associated with hypoxia, they are relatively small when supply elasticity is accounted for.

Another fishery where welfare effects from low oxygen levels have been addressed is the blue crab fishery, also in North Carolina. Smith (2007) developed a spatial bioeconomic model to allow for two locations (patches) where hypoxic conditions occur in one but not the other. The model was used to estimate changes in producer and consumer surplus from a 30 percent decrease in nitrogen entering the fishery (which is the primary driver of hypoxic conditions) under both open access and an effort constrained scenario. Smith found that the benefits of addressing the water quality problem depend on the management regime with total benefits ranging from $1 million to $7 million annually.

The Beverton–Holt Model

The Schaefer–Gordon model has been criticized as being too aggregated, and not being biologically realistic for many species. Also, one of its key predictions is not supported by observation for some species. The prediction is that if effort increases, eventually the stock must decrease, leading to a decline in growth and harvest and eventually to the biological and economic collapse of the fishery. Yet for several fisheries, plots of effort and harvest over time show increasing effort associated with almost no change in harvest. Townsend (1986) showed this to be the case for the North American lobster fishery originally analyzed by Bell (1972), and McClelland (1991) has also shown this to be the case for several of the Florida Gulf Coast fisheries analyzed by Bell (1989). This observed pattern of harvest and effort is entirely consistent with the Beverton–Holt biological model of the fishery.

The Beverton–Holt model combines a model of the growth of individual members of a species and the number of individuals in a single expression. The first element of the model is the recruitment of k individuals of the same age into a cohort at time $t = 0$. In the basic model, k is considered to be exogenous; but clearly, it could be a function of environmental factors. Consider, for example, the oyster. If recruitment is defined as the attachment of larva to solid surfaces where shell development begins, the limiting factor in this stage of development may be the quantity of suitable surface available for attachment, not the number of eggs released and fertilized in the spawning stage.

The second element of the model is an expression for the population of the cohort at any point in time. Over the lifetime of the species, the total population changes because of natural mortality, m, and the harvest of fish, f. The latter is sometimes referred to as fishing mortality. Both of these forms of mortality can vary over time. Thus, the population of the cohort at time t is given by

$$n_t = k \cdot e^{-(f_t + m_t)^t} . \tag{8.65}$$

The third element of the model is an expression giving the weight of a typical member of the cohort at any point in time. Each fish grows over time, so its weight is given by

$$w_t = w_\infty \left[1 - e^{-\alpha(t)}\right]^\beta . \tag{8.66}$$

The total biomass of the cohort at time t is the product of equations (8.65) and (8.66). The last element of the model shows the impact of fishing effort on the cohort. The total weight of fish harvested at any point in time is

$$h_t = w_t \cdot n_t \cdot f_t , \tag{8.67}$$

where f_t depends, at least in part, on the level of effort.

Cost and revenue functions can be specified and the optimum time path of the exploitation of a cohort can be derived. For examples see Clark (1976, 1985). However, since the number of recruits in one cohort is largely independent of the fishing mortality experienced by earlier cohorts, the intertemporal properties of the Beverton–Holt model are not nearly as interesting as those of the Schaefer–Gordon model.

What is of interest for our purposes is the number of places at which environmental quality parameters can enter into the biological model. For example, environmental variables can affect the number of fish available for harvest at a point in time, their individual weights, or both. Recruitment and natural mortality may both depend upon environmental quality. The individual growth rate, α, and the upper limit on size are also likely to depend on environmental quality. An interesting and potentially fruitful area for future research is the role of environmental variables in the Beverton–Holt model and the development of expressions for the economic value of these changes.

Summary

When an environmental quality variable affects the production costs of firms, the welfare value of the change is measured by changes in the surpluses of producers and consumers. Measuring these values requires a model of the market for the output. In the simplest case of single-product firms in a competitive industry, developing such a model and deriving estimates is straightforward, given knowledge of how q affects the production or cost functions, or both. In this chapter, the simple model has been extended to deal with several kinds of complications, including the cases

of multiproduct firms, and vertically linked markets. Monopoly power models for optimum management of biological resources—such as forests and fisheries—have also been adapted for the purposes of welfare measurement by considering how environmental quality parameters can be incorporated into these models.

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Chapter 9

Recreation Demand

Many natural resource systems such as lakes, rivers and streams, estuaries, and forests are used extensively for various kinds of recreation activities, including fishing, hunting, boating, hiking, and camping. As places to conduct such activities, natural resource systems provide valuable services. From an economic perspective, these services have two important features. First, the economic value of these services depends upon the characteristics of the natural resource system. The characteristics determining value can be affected by air and water pollution and by resource management decisions about such things as the rates of harvest of timber and fish, the extraction of minerals and petroleum, and the allocation of water flows between diversionary uses and various in-stream uses. Knowledge of the values of these services may be important for a variety of resource management decisions.

The second important feature is that access to the resource for recreation is usually not allocated through markets. Rather, access is typically open to all comers at a zero or nominal entrance fee that bears no relationship to the cost of providing access. Moreover, there is little or no variation in these access prices over time or across sites to provide data for econometric estimation of demand functions.

At first blush, the lack of direct access fees would seem to suggest little hope for recovering the underlying demand for recreation and the implied value of the services recreational sites provide. Fortunately, Hotelling (1947), in an unpublished letter to the U.S. Department of the Interior, suggested otherwise, noting that each individual visit to a recreation site involves an implicit transaction in which the cost of traveling to the site is incurred in return for access to the site. These travel costs include both explicit costs (e.g., in the form of gasoline, tolls, etc.) and the implicit opportunity costs of time (both traveling to the site and time on-site). Different individuals will face different travel costs to any site, and one individual will face different travel costs for the different sites he or she might visit. The responses of people to this variation in the implicit prices of visits are the basis for estimating the demand for recreation and the values of recreation sites and changes in site quality.

Travel cost models of recreation demand (or simply "recreation demand" models) have evolved considerably in the 65 years since Hotelling's original insight.

Early models relied on zonal data, relating total trips per capita in a region to an average measure of travel cost. As micro-level data became available, analysts became more concerned with the microeconomic foundations of recreation demand models (e.g., the role of substitute sites and the opportunity cost of time), as well as with econometric issues (e.g., the count nature of trip demand and the prevalence of corner solutions). Indeed, as Phaneuf and Smith (2005, 673) note, "[t]oday, economic analyses of recreation choices are among the most advanced examples of the microeconometric model of consumer behavior in economics." The resulting recreation demand models have come to play a central role in nonmarket valuation, particularly in terms of informing regulators in setting environmental policy, ex post cost benefit analysis, and in natural resource damage assessment cases.

In the first section of this chapter, a generic recreation demand model is developed to illustrate the key features of this valuation technique. The second section then describes the nature of the data typically available to analysts (i.e., seasonal trip counts to one or more recreational sites). The features and limitations of such data, in many ways, drive the various recreation demand models that have emerged in the literature. The third section describes single-site models. While such models are relatively rare these days, the models provide a useful starting point, highlighting the importance of controlling for substitute sites and the challenges associated with modeling count data with frequent corner solutions. The fourth section then provides the more general approaches for characterizing systems of recreation demand equations. The fifth section highlights some of the prominent issues in modeling recreation demand, and the sixth section provides summary and conclusions. Space constraints, as always, make it difficult to go into detail in some areas of recreation demand modeling, especially in terms of the myriad of econometric issues. Haab and McConnell (2002, chs. 6–8) provided an excellent treatment of econometric issues, while Phaneuf and Smith (2005) provided additional perspective on the evolution and state of the art in recreation demand modeling.

The Generic Recreation Demand Model

A wide range of modeling frameworks has emerged over the past 40 years seeking to characterize recreation demand, and the implied demand for the associated environmental amenities. In some instances, the starting point is the specification of Marshallian trip demand functions, while others begin with a representation of individual preferences for both trips and site amenities in the form of a direct or indirect utility function. Historically, approaches beginning with a demand function specification focused almost exclusively on the quantity aspect of the consumer decision (how many trips to take over a given time period), while those beginning with a utility function focused on the discrete site selection aspect (which recreation site provides the best combination of price and site characteristics). See Freeman (2003) for an exposition of this dichotomy.

However, over time, researchers have developed models that combine information from both margins (which site to visit and how often to take trips) to infer the value of recreation sites and their attributes from behavior. Thus, the dichotomy between quantity and site selection model approaches is less useful than it once was—at the heart of all of the models is the fundamental notion that individuals choose where and how often to recreate based on the attributes of the available sites and the cost of accessing the site. To the extent that individuals travel great distances to visit a site, incurring costs in the form of both time and money, they reveal information regarding the value they place in the amenities of the chosen site. A simple recreation demand model, adapted from McConnell (1985), helps to illustrate the basic elements of, and issues associated with, recreation demand.

A Simple Conceptual Model

Assume, for now, that there is only one site available and that all visits have the same duration. While relatively few empirical studies currently employ single-site models, this presentation provides an important foundation for more realistic multiple-site approaches. Modeling of the choice of the length of a visit to a site and of the choice of which sites to visit when there are alternatives will be taken up in later sections of this chapter. Furthermore, assume that the individual's utility depends on the total time spent at the site, the quality of the site, and the quantity of a numeraire. With the duration of a visit fixed for simplicity, the time on site can be represented by the number of visits. The individual solves the following utility maximization problem:

$$\max_{z,x} u(z,x,q)\,, \tag{9.1}$$

subject to the twin constraints of monetary and time budgets:

$$M + w \cdot t_w = z + p_x \cdot x \tag{9.2}$$

and

$$t^* = t_w + (t_1 + t_2)x\ , \tag{9.3}$$

where
z = the quantity of the numeraire whose price is one,
x = number of visits to the recreation site,
q = environmental quality at the site,
M = exogenous income,
w = wage rate,
p_x = monetary cost of a trip,
t^* = total discretionary time,
t_w = hours worked,
t_1 = round-trip travel time, and
t_2 = time spent on site.

Assume that x and q are complements in the utility function. As discussed in Chapter 4, this means that the number of visits will be an increasing function of the site's environmental quality. The time constraint reflects the fact that both travel to the site and time spent on the site take time away from other activities. Thus, there is an opportunity cost to the time spent in the recreation activity. Assume also that the individual is free to choose the amount of time spent at work and that work does not convey utility (or disutility) directly. Thus, the opportunity cost of time is the wage rate. The nature of the time constraint and the opportunity cost of time are discussed in greater detail in a later section. Finally, assume that the monetary cost of a trip to the site has two components: the admission fee, f, which could be zero, and the monetary cost of travel. This cost is $p_d \cdot d$, where p_d is the per-mile cost of travel and d is the round trip distance to the site.

Substituting the time constraint (9.3) into the monetary budget constraint (9.2) yields

$$M + w \cdot t^* = z + c \cdot x \,, \tag{9.4}$$

where c is the full price of a visit given by

$$\begin{aligned} c &= p_x + w \cdot (t_1 + t_2) \\ &= f + p_d \cdot d + w \cdot (t_1 + t_2). \end{aligned} \tag{9.5}$$

As equation (9.5) makes clear, the full price of a visit consists of four components: the admission fee, the monetary cost of travel to the site, the time cost of travel to the site, and the cost of time spent at the site. On the assumption that individuals are free to choose the number of hours worked at a given wage rate, the two time costs are valued at the wage rate. In a more realistic model with income and payroll taxes, time would be valued at the after-tax wage rate.

Maximizing equation (9.1) subject to the constraint of equation (9.4) will yield the individual's demand function for visits:

$$x = x(c, M, q) \,. \tag{9.6}$$

If all individuals spend the same amount of time at the site and have the same wage, then this component of the price of a visit is the same for all individuals. Given these assumptions, the data on rates of visitation, travel costs, and variation in entry fees (if any) can be used to estimate the coefficient on c in a travel cost-visitation function.

There are a number of key assumptions underlying this basic version of the travel cost model:

- First, it is assumed that the wage rate is the relevant opportunity cost of time. In a provocative article, Alan Randall (1994, 88, 90) argues that for several reasons "travel cost is inherently unobservable." One of the more compelling elements of his argument concerns the difficulties in defining and measuring the opportunity cost of time spent in travel, which is

characterized by Randall as "an empirical mystery." The fundamental issue of valuing time is returned to later in this chapter.

- Second, it is assumed that all visits entail the same amount of time spent on the site. This assumption plays two important roles in the simple form of the model: it makes it possible to measure site usage by the scalar x, the number of visits; and it makes the full price of a visit, c, a parameter to the individual. If the individual chooses the amount of time of each visit, then c is an endogenous variable. Modeling the choice of time spent on site is discussed in the section on time below.
- Third, it is assumed that there is no utility or disutility derived from the time spent traveling to the site. If part of the trip involves the pleasures of driving through scenic countryside, then travel cost is overestimated by equation (9.5). Conversely, screaming children in the back seat of the car can raise the full travel cost beyond c. The way various uses of time affect utility and the shadow value of time is discussed in the section on time below.
- Fourth, it is assumed that each trip to the site is for the sole purpose of visiting the site. If the purpose of the trip is to visit two or more sites or to visit a relative en route, then at least part of the travel cost would be a joint cost that cannot be uniquely allocated among different purposes. If an alternative destination can be identified as the primary purpose of the trip, then the relevant cost of the visit to the recreation site is the incremental cost of adding the site visit to the trip given the trip to the primary destination. Parsons (2003) discussed this and other approaches to dealing with multiple destination trips.
- Fifth, it is assumed that there are no alternative recreation sites available to these individuals. If there are other sites available, then it is likely that the number of visits that an individual makes to the site in question will depend not only on its implicit price but also on the implicit prices of any substitute sites in the region. Omitting the price of a substitute site will bias the estimates of both the intercept term and the travel cost parameter (Caulkins, Bishop, and Bouwes 1985; Kling 1989; McKean and Revier 1990). The sign of the bias on the own-price coefficient depends on the correlation between it and the omitted substitute price variables, which in turn depends upon the spatial distribution of the population relative to the available sites.
- Sixth, it is assumed that the individual's choice of where to live (which is one determinant of the cost of a trip to a recreation site) is independent of preferences for recreation visits. If people choose residential locations so as to be near preferred recreation sites, then the price of a visit is endogenous. Parsons (1991) has suggested an instrumental variables approach that may avoid the bias that such choices would otherwise impart to the estimation of trip demand functions.

Beyond the assumptions underlying the generic single-site model in equation (9.6), an important practical limitation of the single-site model is that it provides little

basis for identifying the marginal impact of changing site attributes (i.e., changes in q). Most recreation demand databases are cross-sectional in nature, providing no variation in measured site attributes. To the extent that q represents *perceived* site characteristics (such as expected fish catch rates or a subjective measure of overall water quality), variation in q can be found in cross-sections. However, relying upon these subjective measures creates two further complications. First, the subjective measures themselves must now be elicited from individuals in the sample. Second, valuing changes in *physical* site attributes requires understanding how changes in the physical attributes translate into changes in *perceived* site attributes.

Analysis of the demand for a single site would be appropriate if the researcher is interested in valuing the availability of that single site as long as information on the price and quality of substitute sites is appropriately included in the specification. However, many policy-relevant questions involve changes in the value of a set of sites due to changes in the number and availability of sites, or to changes in the qualities of these sites. In such cases, the interactions and the substitution effects among sites must be modeled explicitly. This calls for some form of multisite model.

Multisite models are estimated as systems of demand equations. For example, for each site j one might specify a demand equation of the following form:

$$x_j = x_j\left(c_j, \boldsymbol{C}_{-j}, M, q_j, \boldsymbol{Q}_{-j}\right), \quad j = 1, \ldots, \mathcal{J}, \tag{9.7}$$

where x_j is the number of visits to the site, c_j is the full price of a visit to j, $\boldsymbol{C}_{-j} = \left(c_1, \ldots, c_{j-1}, c_{j+1}, \ldots, c_{\mathcal{J}}\right)$ denotes the set of substitute prices for visits to other sites, q_j denotes the quality attribute for site j, and $\boldsymbol{Q}_{-j}$ denotes the vector of attributes for the other sites. As the number of sites and demand equations increases, multisite models can become cumbersome and difficult to estimate. However, over the past 25 years, a series of multisite models have been developed that, together with advances in computational capabilities, make it possible to characterize recreation demand when there are well over 50 alternatives in the individual's choice set (e.g., von Haefen, Phaneuf, and Parsons 2004; Murdock 2006; Herriges, Phaneuf, and Tobias 2008). These multisite models, and their limitations, are described in detail below.

Data Challenges and Competing Perspectives

As with many areas of applied economics, the evolution of recreation demand modeling reflects, in part, changes in the nature of the data available to analysts, as well as improvements in modeling and computing power. Early efforts to model recreation demand (e.g., Clawson 1959; Clawson and Knetsch 1966) employed aggregate or market-level data, obscuring the role that individual attributes and preference heterogeneity might play in the demand for recreation. The usual practice these days is to rely upon individual or household-level information on recreation activity aggregated over the course of a season. For example, a survey might be used to elicit the total numbers of trips taken by a sample of individuals to

each of a series of sites during the course of a year, along with socio-demographic characteristics of each survey respondent. While this provides a rich data set for use in modeling recreation demand, it also presents a number of challenges for the analyst seeking to reflect all of the unique features of the data into a theoretically consistent and empirically manageable framework. Some of the more challenging features of the data include:

- *Frequent corner solutions*: In most applications, a large portion of the population chooses to not visit any or all of the alternatives in the choice set. This creates difficulties in data gathering, in that larger sample sizes will be needed in order to find site users. It also creates challenges in terms of the underlying economic and econometric specifications, requiring that the modeling framework allow for frequent zeros to avoid biased welfare estimates.
- *Count data*: The trip data obtained in recreation demand surveys takes the form of nonnegative integers. Traditional continuous demand models, solving the optimization problem in equations (9.1) through (9.3) using standard first-order conditions, will at best only approximate what is essentially the solution to a discrete choice problem.
- *Seasonal trip aggregates*: As noted above, recreation surveys typically elicit counts of the numbers of trips to a series of sites over the course of a season. As such, the data preclude modeling the roles that temporally varying factors (such as weather) play in influencing recreation choices, since it is not known when individual trips are taken. It also precludes modeling any dynamics associated with recreational choices, such as habit formation or variety seeking, since the sequence in which sites are visited is not known. While these problems can be avoided by gathering diary data (e.g., day-by-day or week-by-week records of trip activity), such databases are rare (Provencher and Bishop 1997; Moeltner and Englin 2004), because they are costly to administer and they typically suffer from sample attrition over time.
- *Limited information on site characteristics*: One advantage in recreation demand modeling is that the data exhibit substantial variation in the price variable (i.e., the travel cost), since each individual faces a different set of travel costs due to their differing proximities to the sites being modeled. Unfortunately, this is typically offset by limited information regarding the attributes of the individual sites. Most studies have available only a few site attributes (e.g., fish catch rates or some measure of water quality). As a result, recreation demand models can suffer from omitted variables bias—a problem that has received increased attention in recent years (e.g., Murdock 2006).

While many of the difficulties above can be individually (or in some cases jointly) addressed using econometric or statistical techniques, the challenge is to do so in a manner that still allows the analyst to impute the welfare implications of changing site availability or attributes. The next two sections review the single- and multiple-site models that have emerged in the recreation demand literature over the past

40 years. Each model focuses on a subset of the above issues, abstracting in each case from the remaining concerns in order to keep the model tractable. In many ways, it is like the parable of the blind men and the elephant, each encountering a different part of the elephant and perceiving it to be a different object (the ear suggesting the elephant is a fan, the leg suggesting that it is a pillar, etc.). Each approach to modeling recreation demand deals with aspects of the recreation demand process, but may not show the whole picture.

Single-Site Models

As noted above, most empirical applications of recreation demand include multiple sites. Further, single-site recreation demand specifications along the lines of (9.6) are rarely estimated as most recreational opportunities face competition from substitute sites. It is more common to estimate demand for a site using some version of (9.7), for example

$$x_1 = f_1\left(c_1, \boldsymbol{C}_{-1}, M, q_1, \boldsymbol{Q}_{-1}; \beta_1\right) + \varepsilon_1, \tag{9.8}$$

where $f_1(\cdot)$ denotes the specific functional form chosen to represent how the demand for trips to site 1 depends upon the full cost of visiting that site (c_1), the vector of costs of visiting potential substitute sites ($\boldsymbol{C}_{-1}$), income (M), the quality attribute for site 1 (q_1), and the attributes of substitute sites ($\boldsymbol{Q}_{-1}$). β_1 denotes the vector of parameters to be estimated and ε_1 denotes the random error term. The random term's specific role has historically been left vague, capturing a myriad of possible factors, including measurement or specification errors by the analyst and/or optimization errors by the individual consumer. The problem with the practice of simply "tacking on" error terms in this fashion is that how the error term is used in subsequent welfare analysis depends critically on what type of error it reflects. This issue is revisited below.

A second general issue with single-site models is that they are typically plagued by little, if any, variation in each site's attributes (i.e., the q_j). Most recreation demand databases are cross-sectional in nature, providing only a single observation on the attributes of each site in the choice set. For example, with data on the number of trips to lake 1 during the course of a year, q_j might measure the annual mean water quality at lake *j*. Although lake quality will typically vary over the course of the year, it is not possible to directly link this variation to observed demand, since it is not known when individual trips are taken.

Without variation in the site attributes across individuals, their impact on trip demand cannot be identified in a single-site model, leading most analysts to employ a simpler functional representation:

$$x_1 = f_1\left(c_1, \boldsymbol{C}_{-1}, M; \beta_1\right) + \varepsilon_1. \tag{9.9}$$

In the following subsection, two competing single-site models are described: censored regression models and count data models. Unless otherwise indicated, it is assumed that the unit of observation is a randomly selected individual from the

target population. Additional details regarding the estimation and interpretation of these models, as well as the impact of alternative sampling approaches, can be found in Haab and McConnell (2002, ch. 7).

Censored Regression Models

As noted in the data section, a common feature of recreation demand data is the nonnegative nature of trip demand, with a large number of corner solutions (i.e., many individuals choose not to visit the site being modeled). A popular framework used to capture these data characteristics is the Tobit model, a special case in the more general class of censored regression models. The model in equation (9.9) is replaced by

$$x_{i1} = \begin{cases} f_1(c_{i1}, \boldsymbol{C}_{-i1}, M_i; \beta_1) + \varepsilon_{i1} & x_{i1}^* > 0 \\ 0 & x_{i1}^* \leq 0, \end{cases} \tag{9.10}$$

where the subscript i is introduced to denote the observation for a given individual ($i = 1, \ldots, N$) and

$$x_{i1}^* = f_1\left(c_{i1}, \boldsymbol{C}_{-i1}, M_i; \beta_1\right) + \varepsilon_{i1} \tag{9.11}$$

is a latent (unobserved) variable determining participation, sometimes referred to as *potential* demand. If x_{i1}^* is positive, the individual participates in recreation and observed demand equals potential demand (i.e., $x_{i1} = x_{i1}^*$). If, however, potential demand is negative (or zero), the individual chooses not to recreate and observed demand is zero. Thus, observed demand is a censored version of potential demand, censoring potential demand from below at zero.

Estimating the underlying demand relationship in (9.10) requires specifying not only the functional form for $f_1(\cdot)$, but also the underlying distribution of the error term ε_{i1}. Typically it is assumed that error terms are independent and identically distributed (iid) normal random variables (i.e., $\varepsilon_{i1} \sim \text{iid}\, \mathcal{N}\left(0, \sigma^2\right)$). Given this assumption, the parameters (β_1, σ) can be estimated using maximum likelihood techniques or Heckman's (1976) two-step estimation procedure. Interpreting the coefficients of the model becomes more complex, even when $f_1(\cdot)$ is linear. When the error term is normally distributed, expected demand becomes

$$\begin{aligned} E\left(x_{i1}\right) &= \Pr\left(x_{i1}^* > 0\right) E\left(x_{i1}^* \mid x_{i1}^* > 0\right) \\ &= \Phi\left[\frac{f_1\left(c_{i1}, \boldsymbol{C}_{-i1}, M_i; \beta_1\right)}{\sigma}\right] f_1\left(c_{i1}, \boldsymbol{C}_{-i1}, M_i; \beta_1\right) \\ &\quad + \sigma\phi\left[\frac{f_1\left(c_{i1}, \boldsymbol{C}_{-i1}, M_i; \beta_1\right)}{\sigma}\right], \end{aligned} \tag{9.12}$$

where $\Phi(\cdot)$ and $\phi(\cdot)$ denote the standard normal cumulative distribution function (cdf) and probability density function (pdf) respectively.

Welfare analysis using the Tobit model is typically conducted on the basis of consumer surplus measures. One issue alluded to above is how the error term (ε_{i1}) in the model is treated during these calculations. Bockstael and Strand (1987) noted that, if the error term represents measurement error, expected demand along the lines of (9.12) should be used in computing consumer surplus (essentially integrating out the unknown measurement error). If, on the other hand, the error term captures potential omitted variables or specification errors, then they argued that observed demand (x_{i1}) is the best predictor of demand to use in computing consumer surplus. Haab and McConnell (2002, 163) provided a numerical example in which the two approaches can yield welfare estimates that differ by a factor of five, making the interpretation of the error term a potentially important factor in conducting welfare analysis.

Finally, while the Tobit model is a popular form of the censored regression models, it does impose considerable structure on the relationship between the participation decision (i.e., decisions at the *extensive* margin) and the numbers of trips taken once participation is chosen (i.e., decisions at the *intensive* margin). Specifically, the same variables and error terms drive both decisions and the parameters are assumed to be the same. Bockstael et al. (1990) provide a discussion of alternative frameworks that relax these restrictions.

Count Data Models

An obvious limitation of censored regression models is that they do not explicitly reflect the discrete count nature of most trip data. When the counts themselves are typically large (e.g., in the twenties and thirties), a continuous approximation to these discrete count outcomes is likely to perform reasonably well. However, in many recreation demand settings, a large portion of the sample will take few, if any, trips to the site being modeled and a continuous approximation will not perform as well. This has led many analysts (e.g., Hellerstein and Mendelsohn 1993; Smith 1988) to turn to count data regression models, which explicitly limit the dependent variable to nonnegative integer values. A general discussion of count data models can be found in Cameron and Trivedi (1998), with discussion focused on the recreation demand context found in Haab and McConnell (2002, sec. 7.4).

The most basic of the count models is the Poisson regression model, with the conditional probability density function for trips x_{i1} given by

$$\Pr(x_{x1} = k \mid \boldsymbol{Z}_i) = \begin{cases} \dfrac{e^{-\lambda_i}\lambda_i^k}{k!} & k = 0, 1, 2, \ldots \\ 0 & \text{otherwise} \end{cases} \tag{9.13}$$

where $\boldsymbol{Z}_i$ denotes the factors thought to impact the number of trips taken by individual *i*, and

$$\lambda_i = \lambda(\boldsymbol{Z}_i;\beta) = E(x_{i1}|\boldsymbol{Z}_i) = \mathrm{Var}(x_{i1}|\boldsymbol{Z}_i) \tag{9.14}$$

denotes both the conditional mean and conditional variance of the count variable x_{i1}. For example, in the simplest version of the single-site model along the lines of (9.9), one might specify $\boldsymbol{Z}_i = (c_{i1}, \boldsymbol{C}_{-i1}, M_i)$. It is typically also assumed that $\lambda(\boldsymbol{Z}_i;\beta)$ has a linear exponential form. In other words,

$$\lambda(\boldsymbol{Z}_i;\beta) = \exp(\boldsymbol{Z}_i\beta). \tag{9.15}$$

This structure insures that (a) mean trips are positive and (b) the log-likelihood function used in maximum likelihood estimation is globally concave in the parameters β, which in turn eases estimation by insuring a unique optimum.

The simplicity of the Poisson regression model (with a linear exponential mean) carries with it a number of advantages. First, as already noted, unlike the censored regression model, the framework explicitly reflects the count nature of the trip data. Second, while the model is non-linear, interpretation of the parameters is relatively straightforward. Specifically, consider a change in z_{ik}, the *k*th element of $\mathbf{Z}_i$. The marginal effect of such a change on expected trips is given by

$$\frac{\partial E(x_{i1}|\boldsymbol{Z}_i)}{\partial z_{ik}} = \beta_k \exp(\boldsymbol{Z}_i\beta) = \beta_k E(x_{i1}|\boldsymbol{Z}_i). \tag{9.16}$$

Thus, the marginal impact of any factor is proportional to the expected number of trips, with the factor of proportionality being the corresponding parameter. Moreover, the maximum likelihood estimation of the Poisson regression model is mean fitting, with

$$\overline{x}_1 \equiv \tfrac{1}{N}\sum_{i=1}^{N} x_{i1} = \tfrac{1}{N}\sum_{i=1}^{N} \exp\left(\mathbf{Z}_i\hat{\beta}\right), \tag{9.17}$$

where $\hat{\beta}$ denotes the maximum likelihood parameter estimates. As a result, the mean marginal effect for the sample used in estimation reduces to $\hat{\beta}_k\overline{x}_1$.

Third, welfare analysis, typically conducted on the basis of consumer surplus, is straightforward. In particular, as Haab and McConnell (2002, 167) demonstrate, the welfare loss to individual *i* from closure of a site, obtained by integrating under the expected demand function in (9.15), is given by

$$CS_{1i} = \frac{1}{-\beta_{1c}}\exp(\mathbf{Z}_i\beta), \tag{9.18}$$

where β_{1c} denotes the coefficient on travel cost to site 1. Again, employing (9.17) the average welfare loss from the closure of site 1 reduces to $\overline{CS}_1 = -\overline{x}_1 / \hat{\beta}_{1c}$.

The advantages of the Poisson regression model, however, do not come without a cost. In particular, the Poisson specification assumes, as noted in (9.14), that the conditional mean of trips is equal to the conditional variance of trips, a property known as *equidispersion*. Unfortunately, this assumption is frequently rejected in applications, with it often being the case that *overdispersion* holds (i.e., $\mathrm{Var}(x_{i1}|\mathbf{Z}_i) > E(x_{i1}|\mathbf{Z}_i)$). If it remains the case that $E(x_{i1}|\mathbf{Z}_i) = \exp(\mathbf{Z}_i\beta)$, then maximum likelihood estimates from the Poisson model will still be consistent, but

the estimated standard errors will need to be corrected. Alternatively, one can replace the Poisson specification with a more general count data model that does not impose *equidispersion*, such as the Negative Binomial or zero-inflated Poisson models (see, for example, Haab and McConnell 1996 and 2002, sec. 7.4.2).

Multiple-Site Models

While single-site models continue to be used, they are relatively rare, in large part because they are of limited use in policy analysis. In a cross-sectional setting, where one observes trips to a single-site given a fixed set of available site attributes, the impact of those site attributes cannot be isolated. Analysts are typically limited to estimating models of the sort identified in (9.9). While these models can be used to estimate the welfare impact of site closures or changes in access fees, they provide no information on the welfare implications of changing site amenities (e.g., improved water quality). Multisite models, on the other hand, by comparing visitation patterns to a series of available sites with differing site attributes and differing travel costs, provide the basis for identifying the marginal impact of those site attributes on individual well-being. In this section, four competing multisite models are reviewed. Additional details regarding the econometric issues associated with these models can be found in Haab and McConnell (2002, ch. 8).

The Linked Model

The introduction of multiple sites to the recreation demand problem significantly complicates modeling. Even in the simplest of settings in which trips are of fixed length, the analyst faces the task of characterizing the integer-valued numbers of trips to each of $\mathcal{J}$ available sites over some time horizon, subject to both budget and time constraints. The earliest efforts (e.g., Hanemann 1978; Feenberg and Mills 1980; Caulkins, Bishop, and Bouwes 1986) focused on the choice among recreational sites *conditional* on taking a trip, the so-called *site selection* problem. This abstracts from the total numbers of trips taken, the so-called *participation* problem. The linked model, originally introduced by Bockstael, Hanemann, and Kling (1987), provided a way of integrating these two problems into a two-stage model, with the participation model characterizing total trips as a function of indices measuring the overall appeal of the available sites, with the latter estimated as a part of the site selection model.

Site Selection

Site selection models represent an application of discrete choice analysis in which consumers are assumed to select one alternative from a finite choice set. Much of this literature is couched in terms of the hypothesis of random utility maximization (RUM) (McFadden 1974, 1981). Within the RUM construct, an

individual i is assumed to receive utility from choosing alternative j, represented by the conditional utility function

$$u_{ij} = u\left(p_{ij}, \boldsymbol{Z}_{ij}, M_i\right), \quad j = 1, \ldots, \mathcal{J}, \tag{9.19}$$

where p_{ij} denotes the cost of alternative j to individual i, $\boldsymbol{Z}_{ij}$ denotes the attributes of the individual and/or the alternatives that impact the individual's conditional utility, and M_i denotes the individual's income. In the context of site selection, this conditional utility is typically assumed to take the form

$$u_{ij} = u\left(M_i - c_{ij}, \tilde{\boldsymbol{Q}}_j, \tilde{\boldsymbol{S}}_i\right), \tag{9.20}$$

where c_{ij} denotes the travel cost for individual i in visiting site j (so that $M_i - c_{ij}$ denotes the individual's residual income after purchasing a visit to site j), $\tilde{\boldsymbol{Q}}_j$ denotes the attributes of site j, and $\tilde{\boldsymbol{S}}_i$ denotes the attributes of the individual. The RUM hypothesis assumes that individual i simply chooses the alternative that maximizes his/her utility. That is, alternative j is chosen (denoted by $y_{ij} = 1$, as opposed to $y_{ij} = 0$ when it is not chosen) if

$$u\left(M_i - c_{ij}, \tilde{\boldsymbol{Q}}_j, \tilde{\boldsymbol{S}}_i\right) > u\left(M_i - c_{ik}, \tilde{\boldsymbol{Q}}_k, \tilde{\boldsymbol{S}}_i\right) \; \forall \; k \neq j \; . \tag{9.21}$$

It is important to emphasize that there is nothing random in this choice process from the individual's perspective, at least at the time the actual choice is made. They are assumed to have well-defined preferences over the alternatives in the choice set and to be able to identify the alternative yielding the maximum utility.[1] The randomness in the RUM framework is from the perspective of the analyst, who does not know all of the factors impacting the individual's decision or the correct functional relationship among the factors. Instead, the researcher represents the conditional utility function using the form

$$u_{ij} = v\left(M_i - c_{ij}, \boldsymbol{Q}_j, \boldsymbol{S}_i\right) + \varepsilon_{ij}, \tag{9.22}$$

where $\boldsymbol{Q}_j$ denotes the observable site characteristics (versus the full set of characteristics $\tilde{\boldsymbol{Q}}_j$) and $\boldsymbol{S}_i$ denotes the observable set of individual characteristics (versus the full set $\tilde{\boldsymbol{S}}_i$). The error captures the effect of both unobservables, those factors in $\left(\tilde{\boldsymbol{Q}}_j, \tilde{\boldsymbol{S}}_i\right)$ but not in $\left(\boldsymbol{Q}_j, \boldsymbol{S}_i\right)$, and model misspecification, since

$$\varepsilon_{ij} \equiv u\left(M_i - c_{ij}, \tilde{\boldsymbol{Q}}_j, \tilde{\boldsymbol{S}}_i\right) - v\left(M_i - c_{ij}, \boldsymbol{Q}_j, \boldsymbol{S}_i\right) \; . \tag{9.23}$$

With this limited information, the analyst can only specify the probability that an individual will choose a given alternative with

$$\begin{aligned} \Pr\left(y_{ij} = 1 \middle| \text{Trip}\right) &= \Pr\left[v_{ij} + \varepsilon_{ij} > v_{ik} + \varepsilon_{ik} \; \forall k \neq j\right] \\ &= \Pr\left[\Delta v_{ijk} > \Delta \varepsilon_{ijk}\right], \end{aligned} \tag{9.24}$$

1 If the individual has any uncertainty over the alternatives in the choice set, he/she is assumed to be able to integrate that uncertainty into an overall measure of utility (e.g., forming an expected utility to be used in comparing alternatives).

where $v_{ij} \equiv v\left(M_i - c_{ij}, \boldsymbol{Q}_j, \boldsymbol{S}_i\right)$, $\Delta v_{ijk} \equiv v_{ij} - v_{ik}$, and $\Delta\varepsilon_{ijk} \equiv \varepsilon_{ik} - \varepsilon_{ij}$. Note that the probability in (9.24) is conditional on the individual choosing to take a trip. Different discrete choice models result, depending upon the specification of the error terms ε_{ij}. If, for example, it is assumed that the error terms are independently and identically distributed with a Type I Extreme Value distribution, then a logit model results with

$$P_{ij} \equiv \Pr\left(y_{ij} = 1 \middle| \text{Trip}\right) = \frac{\exp\left(v_{ij}\right)}{\sum_{k=1}^{\mathcal{J}} \exp\left(v_{ik}\right)} \quad . \tag{9.25}$$

Alternatively, if $\boldsymbol{\varepsilon}_{i\bullet} \equiv \left(\varepsilon_{i1}, \ldots, \varepsilon_{i\mathcal{J}}\right)$ is drawn from a generalized extreme value (GEV) distribution, then a nested logit model results.

Welfare analysis is typically conducted on the basis of compensating variation. Specifically, consider a change in travel cost, site attributes and/or the set of alternatives available to the individual from $\left(\boldsymbol{C}_i^0, \boldsymbol{Q}^0, \mathcal{J}^0\right)$ to $\left(\boldsymbol{C}_i^1, \boldsymbol{Q}^1, \mathcal{J}^1\right)$ where $\boldsymbol{C}_i^s = \left(c_{i1}^s, \ldots, c_{i\mathcal{J}}^s\right)$ and $\boldsymbol{Q}^s = \left(\boldsymbol{Q}_1^s, \ldots, \boldsymbol{Q}_{\mathcal{J}}^s\right)$ for $s = 0,1$. The compensating variation associated with this change is implicitly defined by

$$\begin{aligned} &\underset{j \in \mathcal{J}^1}{\text{Max}} \left[v\left(M_i - c_{ij}^1 - CV_{|\text{Trip}}, \boldsymbol{Q}_j^1, \boldsymbol{S}_i\right) + \varepsilon_{ij} \right] \\ &= \underset{j \in \mathcal{J}^0}{\text{Max}} \left[v\left(M_i - c_{ij}^0, \boldsymbol{Q}_j^0, \boldsymbol{S}_i\right) + \varepsilon_{ij} \right], \end{aligned} \tag{9.26}$$

where $CV_{|\text{Trip}} = CV_{|\text{Trip}}\left(M_i - c_i^0, M_i - c_i^1, \boldsymbol{Q}^0, \boldsymbol{Q}^1, \mathcal{J}^0, \mathcal{J}^1, \boldsymbol{\varepsilon}_{i\bullet}\right)$. The compensating variation represents the amount that equates the individual's utility before and after the change, allowing for the possibility that the individual may change their preferred alternative. As the notation indicates, the compensating variation is itself a random variable from the perspective of the analyst, since it depends on $\boldsymbol{\varepsilon}_{i\bullet}$. It is important to note that, in the context of the site selection model, the compensating variation $CV_{|\text{Trip}}$ is conditional on taking a trip. The unconditional welfare impact will be potentially smaller, since the individual has the option of simply staying at home.

If, as is typically the case in the literature, it is assumed that the marginal utility of income is constant, then

$$v_{ij}^s \equiv v\left(M_i - c_{ij}^s, \boldsymbol{Q}_j^s, \boldsymbol{S}_i\right) = \beta_M\left(M_i - c_{ij}^s\right) + \widehat{v}\left(\boldsymbol{Q}_j^s, \boldsymbol{S}_i\right) \ , \tag{9.27}$$

where β_M denotes the marginal utility of income and $\widehat{v}\left(\boldsymbol{Q}_j^s, \boldsymbol{S}_i\right)$ denotes the remaining terms' impacting site utility. One can then explicitly solve for $CV_{|\text{Trip}}$ in (9.26), with

$$\begin{aligned} CV_{|\text{Trip}} = \frac{1}{\beta_M} \Bigg\{ &\underset{j \in \mathcal{J}^1}{\text{Max}} \left[v\left(M_i - c_{ij}^1, \boldsymbol{Q}_j^1, \boldsymbol{S}_i\right) + \varepsilon_{ij} \right] - \\ &\underset{j \in \mathcal{J}^0}{\text{Max}} \left[v\left(M_i - c_{ij}^0, \boldsymbol{Q}_j^0, \boldsymbol{S}_i\right) + \varepsilon_{ij} \right] \Bigg\}. \end{aligned} \tag{9.28}$$

The term in curly brackets represents the change in utility, and dividing by the marginal utility of income monetizes this change.

In the case of the logit model, computing the mean value of $CV_{|\text{Trip}}$ becomes particularly straightforward, since

$$E\left\{\underset{j\in\mathcal{J}^s}{\text{Max}}\left[v\left(M_i - c_{ij}^s, \boldsymbol{Q}_j^s, \boldsymbol{S}_i\right) + \varepsilon_{ij}\right]\right\} = \ln\left[\sum_{j\in\mathcal{J}^s}\exp\left(v_{ij}^s\right)\right] + K \quad s = 0,1 \tag{9.29}$$
$$= I\left(\boldsymbol{C}_i^s, M_i, \boldsymbol{Q}^s, \boldsymbol{S}_i\right) + K$$

where K is Euler's constant and $v_{ij}^s \equiv v\left(M_i - c_{ij}^s, \boldsymbol{Q}_j^s, \boldsymbol{S}_i\right)$. A similar expression exists for nested logit models (see, for example, McFadden 1984; Morey 1999). The term $I\left(M_i, \boldsymbol{C}_i^s, \boldsymbol{Q}^s, \boldsymbol{S}_i\right)$ is referred to as the *inclusive value*. Using this result along with (9.28) yields

$$\overline{CV}_{|\text{Trip}} = E\left(CV_{|\text{Trip}}\right) = \frac{1}{\beta_M}\left\{\ln\left[\sum_{j\in\mathcal{J}^1}\exp\left(v_{ij}^1\right)\right] - \ln\left[\sum_{j\in\mathcal{J}^0}\exp\left(v_{ij}^0\right)\right]\right\}. \tag{9.30}$$

The term in curly brackets represents the change in expected utility. An interesting special case arises when considering the welfare impact from a site closure, say site 1, where

$$\overline{CV}_{1|\text{Trip}} = \frac{1}{\beta_M}\left\{\ln\left[\sum_{j=2}^{J}\exp\left(v_{ij}^0\right)\right] - \ln\left[\sum_{j=1}^{J}\exp\left(v_{ij}^0\right)\right]\right\} \tag{9.31}$$
$$= \frac{1}{\beta_M}\ln\left(1 - P_{i1}\right) \approx \frac{-P_{i1}}{\beta_M}.$$

As one would expect, closing a site represents a loss, with the loss increasing as the site represents a more popular alternative.

The simple functional form for the logit model probabilities in (9.25) makes estimation, typically via maximum likelihood, relatively easy. Moreover, welfare analysis is also straightforward using the compensating variation formula in (9.30). However, these simplifications come with a cost. The logit model imposes the well-known Independence of Irrelevant Alternatives (IIA) assumption, which means that the relative choice probabilities for any two alternatives (i.e., P_{ij}/P_{ik}) depend only on the attributes of the two alternatives and not on any of the other alternatives available to the individual. This assumption is often rejected in practice (Kling and Thomson 1996).

Another way to view the restrictiveness of the logit specification is to consider what it says about the correlation among the error terms (i.e., the ε_{ij}). Specifically, it assumes that the error terms are independently and identically distributed. Yet the error terms by construction represent those unobservable factors influencing the choices individuals make. In most applications, particularly those with limited information about the individuals and/or the alternative characteristics, it seems unlikely that the unobservables will be uncorrelated across choice alternatives. For

example, in a model of lake recreation, one unobservable might be whether the individual owns a powerboat or not. This unobservable is likely to be a part of the error term for all the lakes in which power boating is allowed, inducing correlation among the associated error terms.

Over the past 20 years, advances in computational capabilities have enabled analysts to employ a much wider range of error specifications, allowing for complex patterns of correlation among the error terms. These models include mixed logit, multivariate probit and latent class (or finite mixture) models. Train (2009) provides an excellent general discussion of these models, while Herriges and Phaneuf (2002) provide a discussion in the context of recreation demand. While these models complicate both estimation and welfare calculations, the underlying process is similar, with welfare analysis conducted on the basis of Hicksian welfare measures.

Participation

The participation portion of the linked model focuses on the total number of trips, with a generic representation,

$$x_i = g\left(\boldsymbol{L}_i, \boldsymbol{Z}_i, M_i\right) + \eta_i \ , \tag{9.32}$$

where $x_i = \sum_{j=1}^{J} x_{ij}$ denotes the total number of trips taken by individual i, $\boldsymbol{L}_i$ denotes a vector of variables that link the participation equation to the site selection equation, $\boldsymbol{Z}_i$ denotes a set of other variables thought to influence the number of trips taken by an individual, and η_i is a random error term.

In the first linked model by Bockstael, Hanemann, and Kling (1987), BHK hereafter, the linking variable was the inclusive value term $I\left(\boldsymbol{C}_i, M_i, \boldsymbol{Q}, \boldsymbol{S}_i\right)$ implicitly defined in (9.29). Thus, the number of trips taken by an individual depends on the expected utility derived from taking a trip. BHK suggested that the welfare impact of a policy change could then be computed as the expected compensating variation of the change per trip, calculated using the site selection model's equation (9.30), multiplied by the number of trips (either before or after the change).

There have been a number of subsequent variations on the linked model proposed in the literature. Parsons and Kealy (1995) and Feather, Hellerstein, and Tomasi (1995) proposed using as linking variables, not the inclusive value function, but rather "average price" and "average quality" variables, where the weights in computing each average are the site selection probabilities predicted by the site selection model. Thus, (9.32) becomes

$$x_i = g\left(\overline{c}_i, \overline{\boldsymbol{Q}}_i, \boldsymbol{Z}_i, M_i\right) + \eta_i \ , \tag{9.33}$$

where

$$\overline{c}_i = \sum_{j=1}^{J} \hat{P}_{ij|\mathrm{Trip}} c_{ij} \tag{9.34}$$

$$\bar{\boldsymbol{Q}}_i = \sum_{j=1}^{\mathcal{J}} \hat{P}_{ij|\text{Trip}} \boldsymbol{Q}_j \tag{9.35}$$

and $\hat{P}_{ij|\text{Trip}}$ denotes the fitted site selection probabilities.

Finally, Hausman, Leonard, and McFadden (1995) suggested a slight variant on the BHK approach, replacing the inclusive value with its monetized counterpart (i.e., $p = I\left(M_i, \boldsymbol{C}_i, \boldsymbol{Q}, \boldsymbol{S}_i\right) / \beta_M$), arguing that it can formally be viewed as a price index for recreation demand. The authors claimed that the resulting model is utility-theoretic and consistent with a two-stage budgeting process. Unfortunately, this claim has subsequently been shown to not be true in general (e.g., Smith 1997). Instead, as Parsons and Kealy (1995, 360) suggested, the linked model should not be viewed as "derived from a single overall utility maximization problem," but rather as an approximation to the underlying optimization.

The Repeated RUM Model

While the linked model represented a significant step forward in recreation demand analysis, addressing both the site selection and participation decisions, its lack of an underlying unified utility theoretic framework forces the analyst to rely on approximations when conducting welfare analysis. Moreover, by dividing the two interrelated decisions into distinct econometric tasks, efficiency is likely to be sacrificed.

Morey, Rowe, and Watson (1993) proposed an approach based on the simple concept of adding one alternative to the choice set found in the standard site selection model, namely the alternative of staying at home, and repeating this choice over a series of T choice occasions. For example, in the context of annual recreational usage, T might be 52, allowing for the individual to take a trip (or stay at home) during each week of the year. The resulting repeated RUM model has become the workhorse of the recreation demand literature, providing a unified framework for modeling both site selection and participation and an internally consistent basis for welfare analysis.

The starting point in the repeated RUM model is similar to the site selection model described in the previous subsection. In this case, individual i is assumed to receive utility $\tilde{u}_{ijt}$ from choosing alternative j on choice occasion t, where a traditional specification might take the form

$$\tilde{u}_{ijt} = \begin{cases} \beta_M M_i + \delta \boldsymbol{S}_i + \varepsilon_{ijt} & j = 0 \\ \beta_M \left(M_i - c_{ij}\right) + \gamma \boldsymbol{Q}_j + \varepsilon_{ijt} & j = 1, \ldots, \mathcal{J}. \end{cases} \tag{9.36}$$

In this case, site attributes ($\boldsymbol{Q}_j$) impact the utility received from visiting the corresponding recreational site. Individual attributes ($\boldsymbol{S}_i$), such as age, gender, and education, impact the individual's propensity to stay at home by increasing or decreasing $\tilde{u}_{i0t}$ relative to all the other $\tilde{u}_{ijt}$, but do not impact the relative appeal of the recreational sites (i.e., $\tilde{u}_{ijt}$ versus $\tilde{u}_{ikt}$, $j, k = 1, \ldots, \mathcal{J}$). One can generalize

the specification in (9.36) to allow for the latter effects by introducing interaction terms between $\boldsymbol{S}_i$ and $\boldsymbol{Q}_j$ into the trip utilities (i.e., $\tilde{u}_{ijt}, j = 1,\ldots,\mathcal{J}$).

The specification in (9.36) assumes that the marginal utility of income is constant at β_M. Since $\beta_M M_i$ appears in all of the choice utilities, it has no impact on their relative values and, hence, has no impact on the choices made. Consequently, an equivalent representation of consumer utility is

$$u_{ijt} = \begin{cases} \delta \boldsymbol{S}_i + \varepsilon_{ijt} & j = 0 \\ \beta_M c_{ij} + \gamma \boldsymbol{Q}_j + \varepsilon_{ijt} & j = 1,\ldots,\mathcal{J}. \end{cases} \tag{9.37}$$

As in the case of the site selection model, a key step in completing the model specification is the choice of the error distribution. Assuming that the error terms are iid Extreme Value will yield logit probabilities analogous to those in (9.25). The problem is that this assumes the error terms are uncorrelated, a restriction that is unlikely to hold, particularly among the trip alternatives. A more common specification is to assume that the error vector $\boldsymbol{\varepsilon}_{i\bullet t} \equiv \left(\varepsilon_{i0t}, \varepsilon_{i1t}, \ldots, \varepsilon_{i\mathcal{J}t}\right)$ is drawn from a Generalized Extreme Value (GEV) distribution yielding a nested logit model with the trip alternatives belonging to a single nest. Intuitively, this nesting structure implies that the trip alternatives are more similar to each other than to the stay-at-home option. Statistically, the assumption implies that the ε_{ijt} ($j = 1,\ldots,\mathcal{J}$) are correlated with each other, but not with ε_{i0t}. Formally, the resulting choice probabilities take the form

$$P_{ijt} = \Pr\left(y_{ijt} = 1\right) = \Pr\left(y_{ijt} = 1 \middle| y_{i0t} = 0\right)\Pr\left(y_{i0t} = 0\right) = P_{ij|\text{Trip}} P_{\text{Trip}}, \tag{9.38}$$

where

$$P_{ij|\text{Trip}} = \frac{\exp\left(\dfrac{v_{ij}}{\theta}\right)}{\displaystyle\sum_{j=1}^{\mathcal{J}} \exp\left(\frac{v_{ik}}{\theta}\right)} \tag{9.39}$$

denotes the probability of visiting site j conditional on choosing to take a trip (i.e., to not stay at home),

$$P_{\text{Trip}} = \frac{\left[\displaystyle\sum_{j=1}^{\mathcal{J}} \exp\left(\frac{v_{ij}}{\theta}\right)\right]^{\theta}}{\exp\left(v_{i0}\right) + \left[\displaystyle\sum_{j=1}^{\mathcal{J}} \exp\left(\frac{v_{ij}}{\theta}\right)\right]^{\theta}} \tag{9.40}$$

denotes the probability of taking a trip, and

$$v_{ij} = \begin{cases} \delta \boldsymbol{S}_i & j = 0 \\ -\beta_M c_{ij} + \gamma \boldsymbol{Q}_j & j = 1,\ldots,\mathcal{J}. \end{cases} \tag{9.41}$$

The parameter θ in equations (9.39) and (9.40) is known as the *dissimilarity* coefficient and is restricted to lie in the unit interval, specifically $\theta \in (0,1]$ for consistency with the RUM model. The smaller θ becomes, the more *similar* the trip alternatives become (with greater correlation among their error terms). At the other extreme, with $\theta = 1$ the model reduces to a standard logit model.

Notice that the choice probabilities in (9.38) do not vary by choice occasion. This is essentially required by the data, since the analyst does not typically know when individual trips are taken, so that time-dependent factors cannot be accounted for. Also, notice that equation (9.39) corresponds to the site selection model (9.25) discussed above. With the error terms assumed to be iid, the contribution of individual i to the likelihood function becomes

$$L_i = \prod_{j=0}^{\mathcal{J}} P_{ij}^{n_{ij}} , \tag{9.42}$$

where $n_{ij} = \sum_{t=1}^{T} y_{ijt}$ denotes the number of times that individual i chose alternative j across the T choice occasions.

Welfare analysis using the repeated RUM model is straightforward. Consider a policy change from $\left(\boldsymbol{C}_i^0, \boldsymbol{Q}_j^0, \mathcal{J}^0\right)$ to $\left(\boldsymbol{C}_i^1, \boldsymbol{Q}_j^1, \mathcal{J}^1\right)$. For each choice occasion, the compensating variation for the change (CV_{CO}) is calculated just as it was for the site selection model, only now the choice set includes the option of staying at home. Specifically,

$$\begin{aligned} &\underset{j \in \mathcal{J}^1}{\text{Max}} \left[v\left(M_i - c_{ij}^1 - CV_{CO}, \boldsymbol{Q}_j^1, \boldsymbol{S}_i\right) + \varepsilon_{ijt} \right] \\ &= \underset{j \in \mathcal{J}^0}{\text{Max}} \left[v\left(M_i - c_{ij}^0, \boldsymbol{Q}_j^0, \boldsymbol{S}_i\right) + \varepsilon_{ijt} \right]. \end{aligned} \tag{9.43}$$

If the nested logit model is assumed, as in (9.38) through (9.40), then the mean compensating variation per choice occasion has a convenient closed form expression:

$$\overline{CV}_{CO} = E\left(CV_{CO}\right) = \frac{1}{\beta_M} \left(\begin{aligned} &\ln\left\{ \exp\left(v_{i0}^1\right) + \left[\sum_{j \in \mathcal{J}^1} \exp\left(\frac{v_{ij}^1}{\theta} \right) \right]^{\theta} \right\} - \\ &\ln\left\{ \exp\left(v_{i0}^0\right) + \left[\sum_{j \in \mathcal{J}^0} \exp\left(\frac{v_{ij}^0}{\theta} \right) \right]^{\theta} \right\} \end{aligned} \right) \tag{9.44}$$

where v_{ij}^s denotes (9.41) evaluated under scenario s, s = 0,1. The expected compensating variation for the season as a whole is then simply $T \cdot \overline{CV}_{CO}$.

Consider the special case in which the policy scenario corresponds to the closure of site 1. As Haab and McConnell (2002) demonstrated in their equation (8.45), the corresponding expected compensating variation per choice occasion becomes

$$\overline{CV}_{1,CO} = \frac{1}{\beta_M} \ln\left[1 - P_{\text{Trip}} + \left(1 - P_{1|\text{Trip}}\right)^{\theta} P_{\text{Trip}}\right]. \tag{9.45}$$

Comparing this to the expected compensating variation from the site selection model in equation (9.31), one can show that

$$\overline{CV}_{1,CO} = \frac{1}{\beta_M} \ln\left[1 - P_{\text{Trip}} + \exp\left(\beta_M \overline{CV}_{1|\text{Trip}}\right) P_{\text{Trip}}\right]. \tag{9.46}$$

Clearly, $\overline{CV}_{1,CO} \to \overline{CV}_{1|\text{Trip}}$ as $P_{\text{Trip}} \to 1$, but in general $\left|\overline{CV}_{1,CO}\right| < \left|\overline{CV}_{1|\text{Trip}}\right|$ (i.e., the site selection model will overstate the unconditional welfare impact, ignoring the option to substitute to the no-trip option). At the same time, using (9.45), $\overline{CV}_{1,CO} \to 0$ as $\theta \to 0$. The intuition here is that as $\theta \to 0$, the various trip alternatives are becoming more similar (i.e., better substitutes). Losing one of the sites has little impact in this situation since the individual can readily substitute to another similar site.

Since its introduction 20 years ago, a variety of issues have emerged with the basic repeated RUM model. First, the model assumes that there are a fixed number of choice occasions (T). The choice of T is ad hoc, though empirically the resulting welfare calculations have been found not to be particularly sensitive to changes in its value (Parsons, Jakus, and Tomasi 1999; Shaw and Ozog 1999). Typically, T is chosen to accommodate the largest number of trips taken by individuals in the sample or some reasonable upper limit on the numbers of trips.

Second, the basic nested logit model described above assumes that the unobservable factors influencing where the individual chooses to recreate, captured by the error term vector $\varepsilon_{i \bullet t}$, are independent over choice occasions, which seems unlikely. One generalization that relaxes this assumption is the mixed logit framework, which replaces (9.41) with

$$v_{ij}(\zeta_i) = \begin{cases} \delta_i \boldsymbol{S}_i & j = 0 \\ \beta_{Mi} c_{ij} + \gamma_i \boldsymbol{Q}_j & j = 1, \ldots, \mathcal{J}. \end{cases} \tag{9.47}$$

where $\zeta_i = (\beta_{Mi}, \delta_i, \gamma_i)$ denotes the parameters of the model, which are now assumed to be individual specific and drawn from some specified distribution $f(\zeta)$. The choice probabilities, as well as their contribution to the likelihood function, become conditional on the individual's realization of ζ_i. Instead of (9.42), the unconditional likelihood function becomes

$$L_i = \int L_i(\zeta) f(\zeta) d\zeta = \int \left\{ \prod_{j=0}^{\mathcal{J}} \left[P_{ij}(\zeta)\right]^{n_{ij}} \right\} f(\zeta) d\zeta \ . \tag{9.48}$$

The introduction of random parameters induces correlation over the choice occasions. The parameters can be drawn from either a continuous or finite mixing distribution. In the former case, the integration in (9.48) is usually conducted using simulation techniques. In the latter case, also known as the latent class model, E-M algorithms are often used for estimation. E-M algorithms are iterative procedures

that have been found to greatly simplify estimation in this context. See Train (2009) for additional discussion of these models and their estimation.

Finally, in recent years, considerable attention has been paid to concerns about omitted variable bias. In most applications, relatively few attributes of the sites are observed, such as fish catch rates (Morey, Rowe, and Watson 1993), fish toxin levels (Phaneuf, Kling, and Herriges 2000), or dummy variable indicators capturing different levels of water quality (Parsons, Helm, and Bondelid 2003), although there are some notable exceptions (Egan et al. 2009). The risk in this setting is that unobserved site attributes may be correlated with the observed attributes or travel costs (or both), leading to omitted variables bias for the estimated parameters, and biasing any subsequent welfare calculations. One solution to this problem, described in Murdock (2006), is to introduce alternative specific constants into the model, replacing (9.41) with

$$v_{ij} = \begin{cases} \delta_i \boldsymbol{S}_i & j = 0 \\ \alpha_j - \beta_M c_{ij} & j = 1,\ldots,\mathcal{J}, \end{cases} \tag{9.49}$$

where

$$\alpha_j = \gamma \boldsymbol{Q}_j + \xi_j \ , \tag{9.50}$$

and ξ_j denotes the unobserved attributes of site *j*. One can no longer include $\gamma \boldsymbol{Q}_j$ directly in (9.49), since it will be perfectly collinear with the alternative specific constants. Instead, in the first stage of estimation, the parameters $\left(\alpha_1,\ldots,\alpha_{\mathcal{J}},\beta_M,\gamma\right)$ are estimated, typically via maximum likelihood. The γ's are recovered in a second stage by regressing the estimated alternative specific constants on observable site attributes using (9.50). The downside of this approach is that, while it avoids problems stemming from omitted variable bias, it complicates estimation, since there are now $\mathcal{J}$ alternative specific constants to estimate along with β and γ. This can be challenging when there are a large number of alternatives in the choice set. Murdock (2006) provided an iterative approach using a contraction mapping technique, which simplifies estimation when a logit model is being used. Abidoye, Herriges, and Tobias (2012) provided an alternative technique drawing on Bayesian simulation tools.

Multivariate Count Data Models

An alternative to the repeated RUM model is to generalize the single-site count data model to allow for a system of counts. There have been several efforts along these lines in the recreation demand literature, though they remain relatively rare. The paper by Ozuna and Gomez (1994) represents one of the earliest applications. The authors estimated a two-equation system of counts using what they refer to as the SUPREME model. Formally, they specified

$$\begin{aligned} x_{i1} &= x_{i1}^* + \omega_i \\ x_{i2} &= x_{i2}^* + \omega_i \end{aligned} \tag{9.51}$$

$x_{ij}^{*} \sim \text{Poisson}\left(\lambda_{ij}\right)$, where $\lambda_{ij} = \exp\left(\boldsymbol{Z}_{ij}\beta_j\right)$ takes the usual linear exponential form introduced for the single-site count data model, and $\omega_i \sim \text{Poisson}\left(\lambda_{\omega}\right)$, where λ_{ω} is a constant to be estimated. Since the sum of two Poisson distributions also follows a Poisson distribution, the x_{ij} follow Poisson distributions with conditional means equal to $\lambda_{ij} + \lambda_{\omega}$. Moreover, the counts are now correlated through the shared ω_i. The advantage of the SUPREME model is that it allows for correlated counts. On the other hand, the correlation is restricted to be positive, which may not always be the case, and it has generally been limited to two-equation systems, which restricts its usefulness in practice.

Englin, Boxall, and Watson (1998) took another tack. They estimated a system of four count equations in which each count follows a Poisson distribution, with a linear exponential mean, starting with

$$E\left(x_{ij}\right) = \exp\left(\alpha_j + \sum_{k=1}^{J} \beta_{jk} c_{ik} + \gamma_j M_i\right). \tag{9.52}$$

They showed that imposing integrability restrictions onto this demand system requires that there be (a) no cross-price effects (i.e., $\beta_{jk} = 0,\ j \neq k$) and (b) common income effect (i.e., $\gamma_j = \gamma, \forall j$). Furthermore, the intercepts are restricted to satisfy

$$\alpha_j = \alpha_k \left(\frac{\beta_{jj}}{\beta_{kk}}\right). \tag{9.53}$$

While this results in a utility theoretic system of Poisson demand equations, the use of independent Poisson distributions ignores potential correlation among the count variables.

In recent years, more general count systems have emerged, blending these two approaches. Shonkwiler (1999), for example, used the basic structure in Englin, Boxall, and Watson (1998), but introduced mixing distributions analogous to the mixed logit approach to allow for a general pattern of correlation among the four counts in his recreation demand system. Herriges, Phaneuf, and Tobias (2008) developed a related model, though they used Bayesian simulation tools to allow for a much larger system of counts, with 29 recreational sites.

The Kuhn–Tucker Model

A conceptual limitation of the system of counts approach to modeling recreation demand is that it is agnostic regarding the source of variation in the numbers of trips taken by households, even once one conditions on observable individual and site characteristics. Yet, as noted earlier, how one should treat a model's error terms in conducting welfare analysis depends upon the source of the error term itself (e.g., whether it is due to measurement error versus unobserved source of preference heterogeneity). Moreover, welfare analysis using count data models is generally couched in terms of consumer surplus, with the accompanying limitations of this welfare measure. While the repeated RUM framework avoids both of these

problems, it is not without its own limitation. First, the model is based on an artificial construct (i.e., that individuals face a fixed number of choice occasions during which they choose among the available alternatives in the choice set). There is evidence in the literature that welfare estimates are not highly sensitive to the number of choice occasions used, though this research assumes that the number of choice occasions is the same for all individuals. Second, the repeated RUM model, as it has been employed in the literature, imposes weak complementarity, with the utility the individual receives from choosing an alternative (or set of alternatives) during the course of a season depending only on the attributes of the chosen sites.

An alternative modeling approach that avoids all of these issues is the corner solutions, or Kuhn–Tucker (KT), model originally developed by Wales and Woodland (1983) and Hanemann (1978), and first applied to recreation demand by Phaneuf et al. (2000). The KT model takes a direct approach for dealing with the frequent number of corner solutions observed in recreation demand applications by starting with the underlying consumer utility maximization problem subject to standard budget and non-negativity constraints. Corner solutions arise naturally due to the familiar Kuhn–Tucker conditions. The error terms are assumed to stem from unobservable factors altering consumer preferences, imbedded in the individual's direct utility function, and carry through to the implied demand equations. As such, the Kuhn–Tucker strategy provides a unified and internally consistent framework for characterizing the occurrence of corner solutions. Moreover, like the repeated RUM model, welfare analysis can be conducted on the basis of compensating variation.

The starting point in the KT model is the specification of the consumer's utility function. In particular, with $\mathcal{J}$ available recreational sites, it is assumed that individuals solve

$$\underset{\boldsymbol{X},z}{\text{Max}}\ u(z,\boldsymbol{X},\boldsymbol{Q},\varepsilon)\ , \tag{9.54}$$

subject to

$$\boldsymbol{P}'\boldsymbol{X}+z\leq M\ , \tag{9.55}$$

and

$$\boldsymbol{X}\geq\boldsymbol{0}\ \text{and}\ z\geq 0\ , \tag{9.56}$$

where $u(z,\boldsymbol{X},\boldsymbol{Q},\boldsymbol{\varepsilon})$ is assumed to be a quasi-concave, increasing, and differentiable function of $(\boldsymbol{X},\boldsymbol{z})$, $\boldsymbol{X}=(x_1,\ldots,x_{\mathcal{J}})$ is the vector of trips taken to the $\mathcal{J}$ sites in the choice set, z is a numeraire good, $\boldsymbol{Q}=(q_1,\ldots,q_{\mathcal{J}})$ is a vector of site attributes, and $\boldsymbol{P}=(p_1,\ldots,p_{\mathcal{J}})$ is the vector of travel costs to the $\mathcal{J}$ sites.[2] The error term vector $\boldsymbol{\varepsilon}=(\varepsilon_1,\ldots,\varepsilon_{\mathcal{J}})$ captures unobserved factors that influence consumer preferences. These factors are assumed known by the decision-maker, but not the analyst, much

2 For simplicity, each site is characterized here using a single-site attribute, though the KT model more generally would allow for a vector of site attributes.

like in the RUM framework. The first-order necessary and sufficient conditions for utility maximization then become

$$\begin{aligned} &u_j(z,\boldsymbol{X},\boldsymbol{Q},\varepsilon) \equiv \frac{\partial u(z,\boldsymbol{X},\boldsymbol{Q},\varepsilon)}{\partial x_j} \leq \lambda p_j \quad j=1,\ldots,\mathcal{J} \\ &x_j \geq 0 \\ &x_j\left[u_j(z,\boldsymbol{X},\boldsymbol{Q},\varepsilon)-\lambda p_j\right]=0 \end{aligned} \tag{9.57}$$

$$\begin{aligned} &u_z(z,\boldsymbol{X},\boldsymbol{Q},\varepsilon) \equiv \frac{\partial u(z,\boldsymbol{X},\boldsymbol{Q},\varepsilon)}{\partial z} \leq \lambda \quad j=1,\ldots,\mathcal{J} \\ &z \geq 0 \\ &z\left[u_z(z,\boldsymbol{X},\boldsymbol{Q},\varepsilon)-\lambda\right]=0 \end{aligned} \tag{9.58}$$

and

$$\begin{aligned} &\boldsymbol{P}'\boldsymbol{X}+z \leq M \\ &\lambda \geq 0 \\ &\lambda\left[M-\boldsymbol{P}'\boldsymbol{X}-z\right]=0 \end{aligned} \tag{9.59}$$

where λ denotes the marginal utility of income.

Assuming that the numeraire good is essential, a realistic assumption in the context of modeling recreation demand, equation (9.58) implies that $\lambda = u_z(z,\boldsymbol{X},\boldsymbol{Q},\varepsilon)$ and $z = M - \boldsymbol{P}'\boldsymbol{X}$. Substituting these results into (9.57) yields a revised form of the first order conditions:

$$\begin{aligned} &\delta_j(M-\boldsymbol{P}'\boldsymbol{X},\boldsymbol{X},\boldsymbol{Q},\varepsilon) \equiv \frac{u_j(M-\boldsymbol{P}'\boldsymbol{X},\boldsymbol{X},\boldsymbol{Q},\varepsilon)}{u_z(M-\boldsymbol{P}'\boldsymbol{X},\boldsymbol{X},\boldsymbol{Q},\varepsilon)} \leq p_j \quad j=1,\ldots,\mathcal{J} \\ &x_j \geq 0 \\ &x_j\left[\delta_j(M-\boldsymbol{P}'\boldsymbol{X},\boldsymbol{X},\boldsymbol{Q},\varepsilon)-p_j\right]=0. \end{aligned} \tag{9.60}$$

The term $\delta_j(M-\boldsymbol{P}'\boldsymbol{X},\boldsymbol{X},\boldsymbol{Q},\varepsilon)$ denotes the individual's marginal willingness to pay (MWTP) for site *j*. If the individual chooses to visit site *j* ($x_j > 0$), they choose a level of usage that equates their MWTP to the corresponding travel cost. If, however, the associated travel cost exceeds their MWTP at any quantity level, then they choose to not visit the site (i.e., $x_j = 0$).

The final conceptual step in the KT model is in terms of the error terms. Specifically, it is assumed that each ε_j is uniquely related to a corresponding site, with

$$\frac{\partial u_j}{\partial \varepsilon_k}\begin{cases} >0 & k=j \\ =0 & k\neq j \end{cases} \tag{9.61}$$

and $\partial u_z / \partial \varepsilon_j = 0 \; \forall j$. This implies that both the marginal utilities and the marginal WTP for a site are strictly increasing functions of only one error term. In particular, $u_j\left(M - \boldsymbol{P}'\boldsymbol{X}, \boldsymbol{X}, \boldsymbol{Q}, \boldsymbol{\varepsilon}\right) = \tilde{u}_j\left(M - \boldsymbol{P}'\boldsymbol{X}, \boldsymbol{X}, \boldsymbol{Q}, \varepsilon_j\right)$ and $\delta_j\left(M - \boldsymbol{P}'\boldsymbol{X}, \boldsymbol{X}, \boldsymbol{Q}, \boldsymbol{\varepsilon}\right) = \tilde{\delta}_j\left(M - \boldsymbol{P}'\boldsymbol{X}, \boldsymbol{X}, \boldsymbol{Q}, \varepsilon_j\right)$. The benefit of this assumption is that one can now draw implications for the error terms in the model that can be used in estimation. Specifically, let $g_j = g_j\left(M - \boldsymbol{P}'\boldsymbol{X}, \boldsymbol{X}, \boldsymbol{Q}\right)$ be defined implicitly as the level of the error term that would just induce the individual to want to start visiting site *j*; in other words,

$$\tilde{\delta}_j\left(M - \boldsymbol{P}'\boldsymbol{X}, \boldsymbol{X}, \boldsymbol{Q}, g_j\right) = p_j. \tag{9.62}$$

Then the first order conditions in (9.60) imply that

$$\varepsilon_j \begin{cases} = g_j\left(M - \boldsymbol{P}'\boldsymbol{X}, \boldsymbol{X}, \boldsymbol{Q}\right) & x_j > 0 \\ \leq g_j\left(M - \boldsymbol{P}'\boldsymbol{X}, \boldsymbol{X}, \boldsymbol{Q}\right) & x_j = 0. \end{cases} \tag{9.63}$$

Thus, for sites that are visited, the observed consumption levels pin down the associated error terms (i.e., the unobservables), whereas for the sites not visited one can only bound the corresponding error terms. This information can in turn be used to derive the likelihood function required to estimate the parameters of the individual's utility function via maximum likelihood. Note that this is essentially what happens in the Tobit model described above for single-site models. The difference here is the structural underpinnings of the model.

Implementation of the KT model to date has been limited to relatively simple functional forms for the utility function $u\left(z, \boldsymbol{X}, \boldsymbol{Q}, \boldsymbol{\varepsilon}\right)$ and distributional assumptions for the error terms (i.e., $\boldsymbol{\varepsilon}$). The most commonly used utility functions are variations on the linear expenditure system, with

$$u\left(z, \boldsymbol{X}, \boldsymbol{Q}, \boldsymbol{\varepsilon}\right) = \Psi_j \ln\left(x_j \omega_j + \theta_j\right) + \ln\left(z\right) \;, \tag{9.64}$$

where $\Psi_j = \exp\left(\sum_{k=1}^{K} \upsilon_k q_{jk} + \varepsilon_j\right)$ and $\omega_j = \exp\left(\sum_{k=1}^{K} \zeta_k q_{jk}\right)$ potentially depend upon observable site characteristics (q_{jk}). The appeal of the linear expenditure system is that one can explicitly solve for the implied conditional demand equations and the functional form for the boundary variable $g_j\left(M - \boldsymbol{P}'\boldsymbol{X}, \boldsymbol{X}, \boldsymbol{Q}\right)$. The error vector $\boldsymbol{\varepsilon} = \left(\varepsilon_1, \ldots, \varepsilon_{\bar{J}}\right)$ is typically assumed to be either composed of iid extreme value terms or drawn from a GEV distribution (allowing for nesting of similar sites). Even with these assumptions, estimation and welfare analysis using the KT model can be challenging. Initial efforts (Phaneuf, Kling, and Herriges 2000) were limited to relatively few sites. Subsequently, von Haefen, Phaneuf, and Parsons (2004) developed estimation techniques allowing for much larger choice sets (i.e., in excess of 60), and generalizations of the linear expenditure system for use with the KT model (von Haefen 2007). Bhattacharya (2010) developed and implemented a panel version of the model. Even so, the KT model remains relatively rare in recreation demand models, despite the unified framework that it provides for modeling both the intensive and extensive margins of recreation

demand. One of the limitations of the KT model, beyond its computational burden, is that it does not explicitly account for the count nature of recreation demand, but instead treats demand as a continuous variable.

Ongoing Issues

There are a myriad of ongoing issues in the modeling of recreation demand. This section of the chapter highlights some of the more important issues and describes the associated research efforts to date.

Defining the Choice Set

In describing the various recreation demand models above, it was assumed that the analyst knows the choice set being used by the individual. In practice, this is rarely the case. Instead, the researcher must identify the set of activities that represent reasonable substitutes for the site choices being modeled and which it is reasonable to assume that individuals are aware of. The issue of choice set was originally associated with RUM models, but it applies equally to all multiple-site approaches since the researcher must identify the potential set of substitutes (or complements) for inclusion, regardless of whether the approach taken is a Kuhn–Tucker model, a linked approach, or any other modeling strategy. Errors can emerge from the choice set being too large or too small. For example, suppose there is an excellent fishing site that relatively few people are aware of. If the site is included in everyone's choice set, an estimated recreation demand model might imply that people do not value high quality fishing, since few individuals would have visited the site, when the fact of the matter is that they simply did not know that the site was an option. Alternatively, since older residents are more likely to have learned of the site, the analysis might be used to infer that only older people value such fishing opportunities. As this example suggests, the choice set itself is likely to be individual specific and dynamic, evolving over time as the individual visits and learns about surrounding recreational opportunities. Analogous problems have been considered in the context of career and college choice and the marketing literatures. Treating the choice set as exogenous and using the universal choice set (e.g., all of the lakes in Iowa for a study of Iowa recreation), as is often done in practice, is at best an approximation whose realism will depend upon the application at hand.

At the other extreme, excluding alternatives from a choice set can create problems by understating an individual's ability to substitute away from an alternative when conditions deteriorate, consequently overstating the welfare impact of those changes. Exclusions can also limit the analyst's ability to identify important marginal effects. For example, Peters, Adamowicz, and Boxall (1995) allowed survey respondents to identify alternatives that they "considered" choosing, narrowing the choice set used in estimation to this so-called "consideration set." The risk here is that potentially valuable information is lost.

Sites may not be considered precisely because of their specific environmental conditions. Excluding them from the model will hamper the analyst's ability to identify which environmental characters are valued. Parsons, Massey, and Tomasi (1999) explored an alternative approach in which the choice set is restricted to sites that the individual is familiar with. While this avoids some of the concerns regarding the use of "consideration sets," the approach does require additional data-gathering efforts. Haab and Hicks (1997), on the other hand, applied a variation of Manski's (1977) model of choice set generation which treats the choice set itself as endogenously generated.

A practical issue in defining the choice set is that the choice set can become so large as to become unmanageable from an econometric perspective. Several approaches have been suggested to alleviate this problem. Parsons and Hauber (1998) proposed limiting the choice set for each individual to alternatives within a fixed distance of the person's home, arguing that alternatives beyond a reasonable distance are effectively irrelevant, particularly when day trips are being modeled.

An alternative to eliminating sites from the choice set is to aggregate alternatives. Kaoru and Smith (1990) were the first to analyze the effects of aggregation on preference parameter estimation and welfare measurement in the context of recreation demand. Their work suggested that models with only a mild degree of site aggregation (i.e., 35 sites aggregated to 23 or 11 composite sites) performed relatively well in capturing site selection. The results, however, were not as promising in terms of subsequent welfare calculations. For example, the welfare impact from the closure of an aggregate site was understated by more than a factor of 2 using either site aggregation. The welfare gain from site quality improvements fared even worse, being understated by a factor of 5 when 11 composite sites were used.

Parsons and Needelman's (1992) subsequent paper identified two distinct sources of bias stemming from site aggregation, one linked to the number of sites being aggregated (the so-called size effect) and the other tied to the degree of heterogeneity among the sites being combined. Their empirical results suggest that aggregation tends to lead to significant bias in parameter estimates (except for the travel cost coefficient), particularly when large numbers of sites are aggregated. The authors suggest minimizing heterogeneity of sites within aggregates and controlling for the number of sites in the aggregate groups by including a size measure as a factor in the conditional utility of each aggregate site.

A series of subsequent papers have largely confirmed the findings in Parsons and Needelman (1992). For example, Kaoru, Smith, and Liu (1995) found that parameters on the price and size variables are consistent across the different aggregation schemes, but that other parameters are sensitive to aggregation. Feather and Lupi (1998) considered partial aggregation as a compromise between a fully disaggregated model and a traditional aggregate model, with the idea being to leave as disaggregate the most popular and policy relevant sites, aggregating only those sites that are of little interest from a policy perspective or that are only infrequently used. As one might expect, this tends to mitigate any aggregation

bias. Parsons, Plantinga, and Boyle (2000) suggested a similar partial aggregation approach, treating sites of policy interest and their close substitutes as individual sites and aggregating other sites to some degree.

The Value of Time

Both travel to a site and the recreation activity itself take time, and time is scarce. Also, the time spent in any activity could be a separate argument in individuals' utility functions. For example, time at work could yield either positive or negative utility; and almost certainly, time spent at a recreation site has a positive marginal utility, at least up to some point. The opportunity cost of time must be included in a properly specified model of the demand for recreation visits. The appropriate shadow price of time will also depend on the alternative uses to which the time could be put and on the nature of the constraints on individual choice.

It has long been recognized that the time spent in travel to a site should be included as a component of travel cost for purposes of estimating the demand for visits in a travel cost model (for example, Cesario and Knetsch 1970) or the probability of selecting a site in an RUM model. From equation (9.5), travel cost is defined as the sum of the monetary and time costs of travel to the site. If the time component of costs is omitted in the estimation of the demand function for visits, the cost variable is biased downward. The result will be that the estimated parameter on cost or price will be biased upward. As a consequence, the estimated demand curve will be more elastic than the true demand curve, and the benefits of the recreation site (the area under the demand curve) will be underestimated.

The choice of the shadow price of time is critical to the estimation of the elasticity of demand for the site and the calculation of the value of the site. The choice of a high shadow price of time raises the importance of time cost in explaining visits as a function of distance. With a higher shadow price of time, the predicted reduction in the number of visits because of an increase in the entry fee is smaller, and the estimated demand curve is less elastic. Clearly, though, using a shadow price of time that is too high can have the opposite effect on estimates of the elasticity of site demand and value; that is, underestimates of site demand and overestimates of site value.

One of the key assumptions in the simple travel cost model described above is that individuals are free to choose the hours they work at a given wage and that this wage governs the tradeoff between work and leisure. Given this assumption, the individual maximizes utility by allocating time among alternative activities so as to equate the marginal values of time in these activities with the wage rate. Thus, the wage can be taken as an indicator of the shadow value or marginal opportunity cost of time.

One practical difficulty in using the wage rate in recreation demand studies is that surveys often provide data on family income rather than the hourly wage rate. If the wage is inferred by dividing family income by some estimate of hours worked (typically, 2,000 hours per year), measurement error is introduced.

There is also the fact that there is empirical evidence from other types of behavior that is inconsistent with the standard model of the labor–leisure tradeoff at the market wage. Cesario (1976) was the first to point this out in the context of the analysis of recreation demand. He cited evidence from choices of transportation mode by commuters that the revealed opportunity cost of time was perhaps only one-third of the market wage. For more recent evidence from stated preference analysis of transportation mode choice, see Calfee, Winston, and Stempki (2001). Following Cesario, the practice of using a shadow price of time of one-third of the wage became common in recreation demand analysis (Shaw and Feather 1999). Englin and Shonkwiler (1995a) used a latent variable approach in an effort to estimate the unobserved true cost of travel time (including time cost) as a function of observable indicator variables. In their application of the model to data on recreation visits to Lake Champlain, they found that travel time has an estimated value that is about 40 percent of the average wage of the sample.

The evidence of a lower shadow price of time could be explained either by institutional constraints on the choice of hours of work, such as the standard 40-hour workweek and monthly or annual salaries or by a richer theory of choice that recognized that time spent in various activities including recreation, traveling, and at work might convey utility (positive or negative) directly. Each possibility is considered in turn.

Suppose that all (or most) jobs offer a 40-hour workweek at a fixed weekly salary. Then the individual cannot equate values and costs of leisure time at the margin. Several authors have analyzed models in which there is no tradeoff between work and recreation time. Smith, Desvousges, and McGivney (1983) assumed that individuals could allocate a fixed nonrecreational time budget between work and other activities. However, the recreation time budget was fixed and exogenous. Individuals could only choose to allocate time between two or more sites. They found that the relevant price of a visit to a site was the sum of the money travel cost and a proportion of the total time per visit, $t_1 + t_2$ in (9.3). This proportionality factor depended upon the wage rate and the shadow prices attached to the constraints on money and time. Thus, the proportionality factor was not observable, but it could be estimated from the data.

Bockstael, Strand, and Hanemann (1987) presented a model that was flexible in that it could accommodate either of two different cases regarding the individual's labor-market equilibrium. These authors assumed that the duration of a visit was fixed. If the individual could alter hours of work at the margin, the relevant price was the sum of the money travel cost and the total time of the visit $(t_1 + t_2)$ valued at the marginal wage rate. If the individual could not make marginal adjustments in the number of hours worked, the money travel cost and time cost per visit entered the demand function for visits separately. The parameter for the time cost component was a function of parameters of the preference function. Bockstael, Strand, and Hanemann assumed a specific functional form for preferences, derived the alternative demand functions, and estimated the parameters from a

data set that distinguished between those who could and those who could not alter their work time at the margin.

Using a different model, Feather and Shaw (1999) used individuals' responses to survey questions to classify them according to whether they were able to choose freely the number of hours they worked at a given wage, were required to work more hours than they wished at the given wage (were "overemployed"), or would prefer to work more hours at the given wage (were "underemployed"). They were then able to estimate a shadow wage function for their sample and to compute shadow wages for the individuals in each group. As the standard model predicts, the shadow wage for those able to freely choose their hours of work was not significantly different from the mean observed wage, while for those who were overemployed (underemployed), the shadow wage was greater (less) than the observed wage. Since the three groups each represented about one-third of the working sample, this result does not support the practice of using a shadow price of time of one-third the wage for all recreationists.

Another line of research questions whether the wage rate, however measured, is the appropriate shadow price of time even for those people who can freely trade off time and work at the going wage. One of the key assumptions in the standard model, in which the wage rate is made the appropriate shadow price, is that there are no time variables appearing directly or implicitly as arguments in the utility function. Suppose that the time spent at work has either a positive or a negative utility at the margin, then there will be a divergence between the shadow price or scarcity value of time in other activities and the wage rate as a measure of the opportunity cost of time at work. This basic idea can be traced back to Johnson (1966). Cesario (1976) first analyzed its implications in the context of the travel cost model of recreation demand. To illustrate this point, the simple travel cost model described earlier in this chapter is modified to include time at work as an argument in the utility function.

The constrained maximization problem becomes

$$\max_{z,x} u(z,x) + \lambda(M - wt_w - z - p_x x) + \mu\left[t^* - t_w - (t_1 + t_2)x\right], \tag{9.65}$$

where site quality q has been omitted for simplicity. Since μ is the marginal utility of time, μ/λ is the scarcity value or marginal willingness to pay for time. The relevant first-order conditions include

$$\frac{\partial u/\partial x}{\lambda} = p_x + \frac{\mu}{\lambda}(t_1 + t_2) \tag{9.66}$$

and

$$\frac{\partial u/\partial t_w}{\lambda} + w = \frac{\mu}{\lambda}. \tag{9.67}$$

Equation (9.66) says that the marginal willingness to pay for a visit must equal the full cost of a visit. The full cost is the sum of the monetary cost p_x and the time cost of the visit, where the time cost $t_1 + t_2$ should be valued at the scarcity value of time, μ/λ. However, equation (9.67) shows that the wage rate may not be a

good measure of the scarcity value of time. If the marginal utility of the time spent working is negative, then the wage rate is an overestimate of the scarcity value of time because the wage is also compensating for the disutility of work. In other words, the opportunity cost of diverting an hour of time away from work is less than the wage rate.

In contrast to this suggestion, McConnell and Strand (1981) derived an alternative model that allowed for the estimation of the relationship between the scarcity value of time and the wage rate from the sample data. Suppose the demand for visits can be specified as a linear function of travel cost:

$$x = a + b_1\left[p_x + (\mu/\lambda)(t_1 + t_2)\right]. \tag{9.68}$$

First, let s be the ratio of the shadow price of time to the wage rate. Substituting $s \cdot w$ for μ/λ in equation 9.68, one can then estimate

$$x = a + b_1 \cdot p_x + b_2 \cdot w \cdot (t_1 + t_2). \tag{9.69}$$

Since $b_2 w = b_1 s w$, then $s = b_2/b_1$. For McConnell and Strand's sample, s was about 0.6.

In the McConnell and Strand model, although the scarcity value of time depends in part on the wage rate, it depends also on the alternative use of time, specifically on the marginal utility of work. A richer model of time and choice would allow for other utility-yielding uses of time. For example, there could be other leisure activities; or time could be an input into the household production of utility-yielding goods.

In a more recent series of papers, Larson and Shaikh (2001, 2004) and Shaikh and Larson (2003) emphasized the importance of explicitly incorporating both budget and time constraints when specifying the functional form for recreation demand equations. Rather than a single Roy's identity linking Marshallian demands to the underlying indirect utility function, they demonstrate that two Roy's identities apply, which in turn implicitly link money and time prices. As the authors noted, "the most common practice in empirical recreation demand specification, which includes a time price (as a component of full price) without a time budget (money income only) … is inconsistent with two-constraint choice" (Shaikh and Larson 2003, 954). In their accompanying application, Shaikh and Larson used a functional representation for the marginal value of time that nests a number of commonly used specifications. The authors rejected the most frequently used specifications for the value of time (including a fixed fraction of the wage rate) and argued instead for the flexibility of an endogenous marginal value of time approach. Clearly, additional empirical research is needed to corroborate these results, couched in the context of a single-site application, and to suggest the key factors in characterizing the marginal value of time.

Finally, in their closing remarks, Shaikh and Larson (2003) suggested that it may be necessary to allow for multiple time constraints when modeling recreation demand, reflecting the nature of the recreation activity. Palmquist, Phaneuf, and Smith (2010) pursued this line of inquiry, suggesting that the marginal value of

time increases for longer trips, since it is more difficult to "cobble-together" the necessary block of time for such trips. The authors used stated preference survey data to infer the marginal opportunity cost of time and, as expected, found that it increases with duration (from $27.89 for a 2-hour trip to $32.15 for an 8-hour trip). While these changes are relatively small, they highlight the complex nature of the marginal value of time.

Time on Site

As the models described above have shown, time spent on the recreation site is part of the cost of the recreation visit. Therefore, in principle, time on site should be included in the estimated demand function for recreation visits. However, there are some assumptions under which this is not necessary. As McConnell (1985) has shown, if all individuals in a sample choose visits of the same duration and if they all have the same opportunity cost of time, time cost on site becomes part of the constant term in the estimated equation. The assumption of identical duration of visits may be reasonable in some cases, for example, where day-trip activities predominate, or if separate equations are estimated for visits of different duration. However, the assumption of identical opportunity costs of time is probably not reasonable, especially given that most travel cost studies now are based on observations of individuals rather than aggregates of people grouped by distance zone.

A related issue is how to model the choice of the duration of a visit. McConnell (1992) has investigated the implications of incorporating the choice of time on site for the proper specification of the recreation demand function. He specified the following utility function:

$$u = u\left(z, x, t_2\right), \tag{9.70}$$

and assumed that it displays what he called joint weak complementarity between x and t_2. By this he means that if $x = 0$, the marginal utility of on-site time is zero, and if on-site time is zero, the marginal utility of a trip is zero. The individual must choose x, t_2, and z to maximize equation (9.70), subject to the constraint of equation (9.4), or

$$M + w \cdot t^* = z + c \cdot x, \tag{9.71}$$

where $c = f + p_d \cdot d + w \cdot \left(t_1 + t_2\right)$. Since c is not exogenous to the choice problem, it cannot be an argument in a Marshallian demand function. McConnell showed that it is still possible to define a Marshallian demand function based on the exogenous components of trip cost. First, define a new term:

$$c^* = f + p_d \cdot d + c \cdot t_1. \tag{9.72}$$

After maximizing equation (9.70) and solving for the demand functions, the indirect utility function can be obtained:

$$v = v\left(c^*, w, M\right). \tag{9.73}$$

McConnell then showed that Roy's identity can be employed to derive a Marshallian demand function that contains only exogenous variables:

$$x = x\left(c^*, w, M\right). \tag{9.74}$$

This expression can be estimated. For other efforts at modeling the determinants of time on site, see Smith, Desvousges, and McGivney (1983), Kealy and Bishop (1986), and Wilman (1987).

Sampling Issues

A practical problem that researchers frequently encounter in recreation demand studies is that only a small fraction of the population may visit a site of interest. In this setting, a random population survey can be expensive, requiring a relatively large sample in order to yield enough users to accurately estimate the parameters of interest. Screening surveys can be used to identify visitors to a site, who can then be followed up with a more detailed survey regarding their usage, but this approach can also be expensive and time consuming. An alternative procedure is to intercept users at the site in question, guaranteeing information regarding site users. Unfortunately, the process creates two problems: (a) truncation (with only users being surveyed) and (b) endogenous stratification (with avid users more likely to be intercepted on site). Both problems will tend to inflate the numbers of trips obtained from an on-site sample relative to those in the general population.

A number of authors have developed techniques designed to "undo" the effects of on-site sampling. Shaw (1988) developed a simple correction for both truncation and endogenous stratification in the case of a single-site Poisson count data model. Englin and Shonkwiler (1995b) subsequently extended Shaw's correction to the case of the Negative Binomial model. Egan and Herriges (2006) extended this approach to a multivariate count data model setting. The risk with all of these procedures is that they rely heavily on maintained distributional assumptions, particularly in terms of inferring the behavior and preferences of nonusers. This suggests that if an on-site sample is employed, efforts should be made to gather a representative sample of nonusers, which can be used to augment an on-site sample and discern any departures from maintained assumptions.

Choice Dynamics and General Equilibrium Concerns

The models of recreation demand that dominate the literature are largely static in nature, assuming a fixed (and often exogenous) choice set and few, if any, interactions among the trips taken by an individual over time. The need for this simplification is driven to a large extent by the nature of the data available to researchers, consisting primarily of total trip counts during a season. There are, however, obvious dynamic elements that are being abstracted from. As already

noted, the choice set itself is surely dynamic in nature, evolving over time as individuals explore the available recreational opportunities and, in doing so, learn about sites and their attributes. This is a topic that has been explored in other settings, such as in the job and college search literatures, but has yet to be explored in the context of recreation demand. Perhaps the closest application along these lines is the paper by Provencher and Bishop (1997), in which the authors developed a dynamic discrete choice model using diary data. Although in their application the choice set is given, individuals learn about catch rates at individual sites through trips to the site.

A second dynamic element in recreation demand is the potential linkage between current and past trips, a phenomenon known as "state dependence." The notion is that an individual's past trips may influence their inclination to revisit the same sites in the future. "Habit formation," for example, might arise if the individual finds comfort in the familiar. Once they visit a given site, their inclination would be to return to the same site on subsequent choice occasions. Other individuals might exhibit "variety seeking" and would, ceteris paribus, avoid soon visiting the same site again. Moeltner and Englin (2004) provided one of the few applications examining the impact of both state dependence and time varying site attributes on recreation demand, with an application to the choice of ski resorts. Once again, the availability of diary data is key to their analysis. Interestingly, they found that, while habit formation tends to dominate variety seeking as the form of state dependence, time varying attributes (such as weather conditions) are also critical to predicting individual choice. Neither factor is controlled for in traditional repeated RUM models.

Finally, most recreation demand models treat site attributes as exogenous factors to which individuals respond. There is, however, a growing literature recognizing the fact that some site attributes are endogenously determined. One such attribute is site congestion. Congestion of a recreation site occurs when the number of users is so large that it diminishes the utility and therefore the willingness to pay of those users. The degree of crowding at a site can be considered as one of the site attributes that influence the quality of the recreation services each individual experiences and the utility each individual obtains from visiting the site. The presence of congestion at a recreation site has implications for the estimation of recreation demand and the measurement of the value of environmental quality change.

Congestion, like pollution, is a negative externality. It affects all users of a site. The earliest analyses of the problem of congested recreation facilities include Fisher and Krutilla (1972) and Cicchetti and Smith (1973). Freeman and Haveman (1977) developed a simple model of congestion to show the effects of congestion on demand for a site and on the prediction of recreation use. The model illustrates how the value of improving quality at a site is reduced if congestion is present, and how improving quality at one site can lead to additional benefits by reducing congestion at other sites. Other theoretical analyses can be found in McConnell and Duff (1976), McConnell (1980), and Smith (1981). Dorfman (1984) provided an elegant treatment of the optimal congestion and optimal pricing problems.

More recently, Timmins and Murdock (2007) drew on locational sorting models to estimate the disamenity impact of site congestion. Their analysis clearly showed that not only is congestion an important site characteristic, but also the failure to recognize its endogeneity can lead to significant bias in assessing both the marginal impact of congestion and the general equilibrium implications of policy scenarios. Papers by Schuhmann and Schwabe (2004) and Hicks, Horrace, and Schnier (forthcoming) have generalized this model, allowing for both heterogeneity in how individuals perceive the impact of congestion, and nonlinearities in the marginal impact of congestion. Phaneuf, Carbone, and Herriges (2009) developed a similar model using the Kuhn–Tucker framework.

Summary and Conclusions

The recreation demand literature has evolved substantially over the last 40 years. While much of the early literature was segmented, with one branch focused on modeling the total numbers of recreational trips taken (i.e., the participation decision) and the other concerned with where individuals recreated (i.e., the site selection literature), this distinction has largely disappeared. Recreation models, whether based on RUM, counts, or continuous demand specifications, are largely couched as systems of demand, controlling for substitution possibilities among the myriad of alternatives in the individual's choice set and allowing for frequent corner solutions. What distinguishes the models is which aspect of the consumer's optimization problem the model abstracts away from (and, hence, which part of the "elephant" it chooses to focus on). The repeated RUM model assumes a fixed and common number of choice occasions faced by decision-makers, whereas the Kuhn–Tucker model ignores the count nature of the data. Systems of counts lack an underlying utility maximization structure that consistently integrates the random nature of the counts themselves. Moreover, all of these models are constrained by the nature of the data typically available to analysts (i.e., seasonal counts of total trips to each of the sites in the choice set). This limits the ability of the analyst to allow for both temporal effects (e.g., due to weather) and inter-temporal effects (e.g., such as learning, habit formation, and variety seeking). While diary data would allow for more complex models, collecting such data faces significant challenges (both in terms of nonparticipation and attrition); and modeling the dynamics of individual behavior in these settings often requires strong assumptions (in terms of what the individual knows about their choices and when) in order to make the analysis tractable.

Despite the limitations of the various recreation demand models, they have become a mainstay in nonmarket valuation, informing both policymakers and natural resource damage assessment efforts. Their strength comes from the rich variation in travel costs across individuals, as well as their reliance on revealed preference. At the same time, ongoing research is clearly needed, particularly in terms of providing a richer characterization of the value of time and the choice

set being used by the consumer. There is also a need to better understand both the choice of time on site (e.g., day trips versus weeklong vacations) and the impact of multiple-destination trips.

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Chapter 10

Property Value Models

Economists have documented the relationship between the prices of housing units and quantities of environmental amenities since before this relationship was recognized as an application of the theory of hedonic prices (for example, Ridker and Henning 1967). Indeed, examples of the statistical analysis of the linkage between farmland prices and the characteristics of the land can be found as early as 1922. See Colwell and Dilmore (1999) for a review. The past 35 years have seen an explosion of both theoretical and empirical studies of the monetary values of nonmarket amenities and disamenities based on hedonic price theory. It is now well accepted that housing price differentials do reflect differences in the quantities of various characteristics of housing and that these differentials have significance for applied welfare analysis. For example, Smith and Huang (1995) conducted a meta-analysis of hedonic studies of air pollution and housing prices. They reported finding 37 studies and more than 160 separate estimates of the effects of air quality on housing prices.

At the same time, much has also been learned about the limitations of standard hedonic property analysis. As will be discussed later, the bulk of the empirical literature has focused on estimating the hedonic price function itself (that is, the equilibrium relationship between a home's prices and its characteristics). This "first stage" in the hedonic model can be used, under certain conditions, to infer the *marginal* willingness-to-pay (MWTP) for individual housing characteristics, including environmental amenities. While such information is valuable, most environmental policy scenarios envision discrete changes for which marginal analysis is no longer sufficient. Hedonic analysis can be extended to value discrete changes, but such "second stage" hedonic analyses are rare in practice, requiring complex econometrics and/or strong a priori restrictions on preferences (see Palmquist 2005).

In part, responding to this limitation of traditional hedonic price analysis, the "equilibrium sorting" literature has emerged in the past decade as a competing paradigm for characterizing how amenities are capitalized into the value of a property. Rather than focusing on a largely reduced form characterization of the hedonic price function, the sorting literature emphasizes the sorting process itself; that is, how the distribution of both individual characteristics and preferences interact with the distribution of housing characteristics in a market to determine

market prices. Having characterized the sorting process, an analyst can, in theory, examine both the partial and general equilibrium implications of a discrete change in the distribution of property amenities. Smith et al. (2004), for example, estimated the impact of the projected ozone reductions in the Los Angeles air basin stemming from the 1990 Clean Air Act Amendments. The authors found that estimates of these impacts, and their distribution in the population, changed substantively depending upon whether or not individuals are allowed to re-sort (that is, move) and property values are allowed to change in response to the discrete air quality improvements. Such changes are indicative of the long run, general equilibrium impact of an environmental policy. The sorting literature, of course, is not without its limitations, requiring strong assumptions regarding the structure and distribution of preferences.

This chapter provides a summary of the current state of knowledge regarding these two approaches to property value modeling: hedonic price and equilibrium sorting. The first section focuses on hedonic pricing, beginning with a brief review of the evolution of economic thinking about property prices and environmental amenities, followed by a detailed exposition of the hedonic property value model. This presentation includes discussions of: (a) problems in estimating the hedonic price function; (b) approaches to recovering information on preferences and the demands for characteristics from the hedonic price function; and (c) the measurement of welfare change. The second section turns to the more recent equilibrium sorting literature. The distinction is made between what Kuminoff, Smith, and Timmins (2013) refer to as the "pure characteristics" sorting models of Epple and Sieg (1999) (among others) and the random utility sorting models pioneered by Bayer, McMillan, and Reuben (2004).

Hedonic Pricing

Historical Background

The theory of rents holds that the equilibrium price for a parcel of land will be the present value of the stream of rents produced by the land. Economic theory has long recognized that the productivity of land differs across sites. These productivity differentials will yield differential rents to land, and therefore differential land values. Where land is a producer's good, competition and free entry are sufficient to ensure that productivity differentials are fully reflected in the land rent structure. For any property where the land rent is less than the productivity, the activity occupying that land must be earning a profit. Some potential entrant will be willing to bid above the going rent in order to occupy that site and reap the rewards of a superior productivity. It is this competition that bids up land rents and eliminates the profit. Rent differentials will be equal to productivity differentials; and since the price at which a unit of land sells in the market is the present value of the stream of future rents, productivity differentials will also be reflected in land prices.

Some environmental characteristics such as air or water quality may affect the productivity of land as either a producer's good or a consumer's good. Where this is so, the structure of land rents and prices will reflect these environmentally determined productivity differentials. These results from classical rent theory aroused considerable interest among economists about the possibility of using data on land rent or land value for residential properties to measure the benefits to households brought about by improvements in environmental characteristics such as air or water quality. Ridker (1967) was the first economist to attempt to use residential property value data as the basis for estimating the benefits of changes in measures of environmental quality such as air pollution. He reasoned as follows:

> If the land market were to work perfectly, the price of a plot of land would equal the sum of the present discounted streams of benefits and costs derivable from it. If some of its costs rise (for example, if additional maintenance and cleaning costs are required) or if some of its benefits fall (for example, if one cannot see the mountains from the terrace), the property will be discounted in the market to reflect people's evaluation of these changes. Since air pollution is specific to locations and the supply of location is fixed, there is less likelihood that the negative effects of pollution can be significantly shifted on to other markets. We should therefore expect to find the majority of effects reflected in this market, and we can measure them by observing associated changes in property values.
>
> (Ridker 1967, 25)

The last sentence of the passage raises three questions. The first is whether environmental variables such as air pollution do systematically affect land prices. Assuming an affirmative answer to this question, the second is whether knowledge of this relationship is sufficient to predict changes in land prices when, say, air pollution levels change. The third question is whether changes in land prices accurately measure the underlying welfare changes.

Ridker (1967) and Ridker and Henning (1967) provided the first empirical evidence that air pollution affects property values by regressing median census tract property values in an urban area on a measure of sulfate air pollution. They then asserted positive answers to the second and third questions. Specifically, they argued that the coefficient on the air pollution variable in the regression equation could be used to predict the change in the price of any residence, conditioned on a change in its air pollution level. The sum of all such changes, they argued, could be taken as a measure of the benefit of improving air quality in an urban area (Ridker 1967, 136–137; Ridker and Henning 1967, 254).

This work stimulated a now large literature on the proper theoretical interpretation of the observed air pollution–property value relationship. Early contributions included those by Freeman (1971, 1974a, 1974b), and Anderson and Crocker (1972). Subsequent efforts to provide a sound theoretical basis for interpreting the air pollution–property value relationship have taken one of two

paths. The first has been the development of models of the urban land market to determine whether and under what circumstances changes in aggregate land values accurately measure the benefits associated with environmental improvements. Early efforts in this direction included those by Strotz (1968), Lind (1973), Pines and Weiss (1976), Polinsky and Shavell (1976), and Kanemoto (1988). Although some of these models can be given an interpretation in the context of hedonic price theory, they do not lend themselves to empirical application, so they are not covered in this book. For further discussion of this branch of the literature, see Bartik and Smith (1987) and Palmquist (1991).

The second path drew on hedonic price theory, introduced in Chapter 4, to interpret the derivative of the cross-section regression equation with respect to air pollution as a marginal implicit price, and therefore, a marginal value for the air quality improvement (see Freeman 1974b and Rosen 1974). This section describes how hedonic price theory provides a basis for deriving welfare measures for public goods from observed differences in the prices of houses. The primary emphasis is on model specification and interpretation rather than econometric estimation. The goal of the section is to provide an overview of the methods of welfare measurement based on hedonic price theory and to identify the major conceptual issues. For other current treatments of these matters with greater emphasis on econometric and estimation issues, see Palmquist (2005) and Taylor (2003).

The Basic Hedonic Property Value Model

Assume that each individual's utility is a function of that person's consumption of a composite commodity z and a vector of amenities ($\boldsymbol{Q}$) associated with the house that the person occupies. These amenities, of course, include the structural characteristics of the house (such as size, number of rooms, age, and type of construction). They also include characteristics of the neighborhood in which the house is located (such as quality of local schools, accessibility to parks, stores, and work place, and crime rates), as well as location-specific environmental amenities (such as the local air and water quality).

Any large area has in it a wide variety of sizes and types of housing with different structural, neighborhood, and environmental characteristics. An important assumption of the hedonic technique is that the area as a whole can be treated as a single market for housing services. Individuals must have information on all alternatives and must be free to choose a house anywhere in the market. It is as if the area were one huge supermarket offering a wide selection of varieties. Indeed, Rosen's (1974) model assumed that the household can choose from a continuum of housing attributes, an assumption that is, at best, approximated in actual housing market settings.

Since the focus here is on the values of characteristics to buyers of houses, there is no need to model formally the supply side of this market. Instead, assume that the housing market is in equilibrium—that is, that all individuals have made their utility-maximizing residential choices given the prices of alternative housing

locations, and that these prices just clear the market given the existing stock of housing and its characteristics. Under these assumptions, the rental price of the jth residential location can be taken to be a function of the structural, neighborhood, and environmental characteristics of that location. In other words,

$$R_j = R\left(\boldsymbol{Q}_j\right). \tag{10.1}$$

As explained in Chapter 4, this relationship can be linear in a characteristic if repackaging of that characteristic is possible. However, in general this need not be the case. Two living rooms with six-foot ceilings are not equal to one living room with a twelve-foot ceiling. Where repackaging is not possible, equation (10.1) will be nonlinear.

To model the problem more formally, consider an individual who occupies house j. Her utility is given by

$$u = u\left(z, \boldsymbol{Q}_j\right), \tag{10.2}$$

where z is a Hicksian composite good with a price of 1. This assumption makes the demands for characteristics independent of the prices of other goods, a convenient property for empirical work. The individual maximizes $u(\cdot)$ subject to the budget constraint:

$$M - R(\boldsymbol{Q}) - z = 0. \tag{10.3}$$

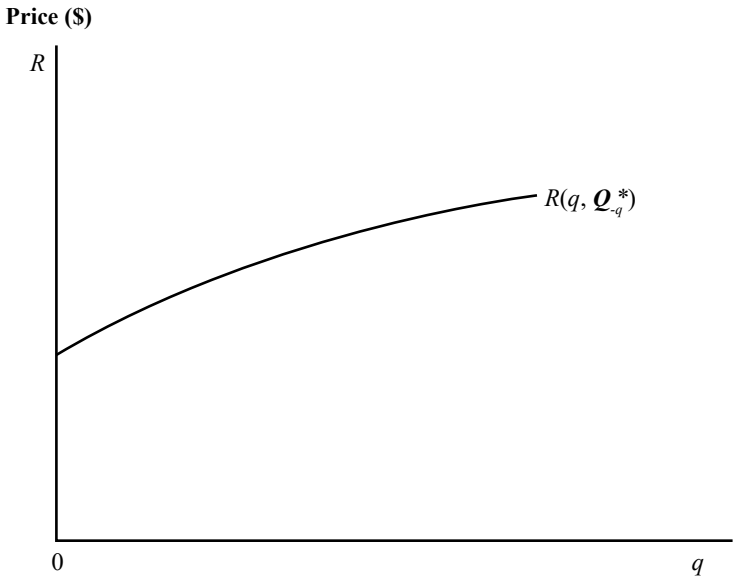
A typical first-order condition for the choice of amenity q (an element in $\boldsymbol{Q}$) is

$$\frac{\partial u/\partial q}{\partial u/\partial z} = \frac{\partial R(\boldsymbol{Q})}{\partial q}. \tag{10.4}$$

Assume now that the hedonic price function $R(\boldsymbol{Q})$ has been estimated for an urban area. Its partial derivative with respect to any of its arguments, for example q, gives the implicit marginal price of that characteristic—that is, the additional amount that must be paid by any household to move to a housing bundle with a higher level of that characteristic, other things being equal. If this function is nonlinear, the marginal implicit price of a characteristic is not constant, but depends on its level and perhaps the levels of other characteristics as well. If the individual is assumed to be a price taker in the housing market, that person can be viewed as facing an array of implicit marginal price schedules for various characteristics. An individual maximizes utility by simultaneously moving along each marginal price schedule until she reaches a point where her marginal willingness to pay for an additional unit of that characteristic just equals the marginal implicit price of that characteristic. If an individual is in equilibrium, the marginal implicit prices associated with the housing bundle actually chosen must be equal to the corresponding marginal willingness to pay for those characteristics.

Panel A in Figure 10.1 shows the partial relationship between q and $R(\boldsymbol{Q}) = R\left(q, \boldsymbol{Q}_{-q}\right)$ as estimated from equation (10.1), where $\boldsymbol{Q}_{-q}$ denotes all of

PANEL A: The Hedonic Price Function



PANEL B: The Marginal Implicit Price Function and Individuals' Willingness to Pay for *q*

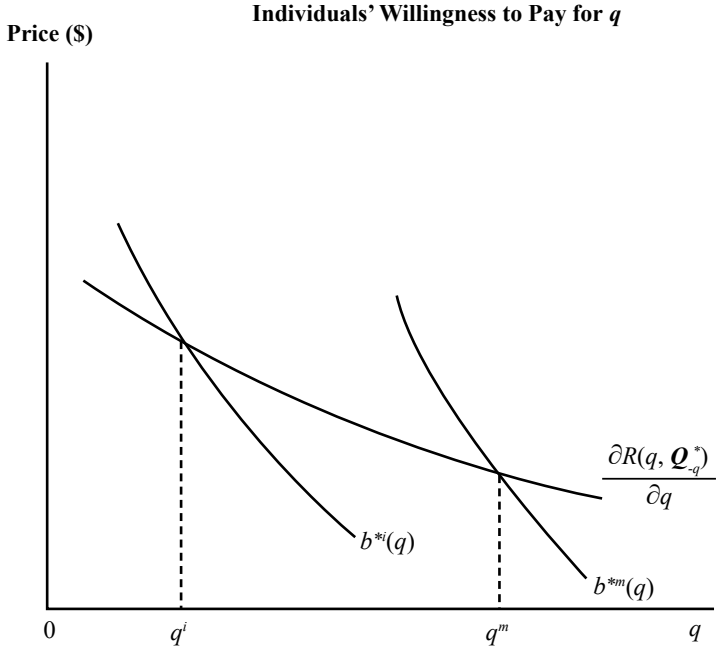


Figure 10.1 Property prices and environmental quality

the attributes except q. In the figure, $\boldsymbol{Q}_{-q}$ is fixed at some level $\boldsymbol{Q}^*_{-q}$. Panel B shows the marginal implicit price of q, $\partial R\left(q,\boldsymbol{Q}^*_{-q}\right)/\partial q$. It also shows the marginal willingness-to-pay curves for two individuals, i and m, who have chosen utility maximizing bundles of housing characteristics, labeled $b^{*i}(q)$ and $b^{*m}(q)$. These curves show each individual's marginal willingness to pay for changes in the characteristic, holding utility constant at the level achieved by maximizing equation (10.2) subject to (10.3). Let this level be u^*. Both individuals in Figure 10.1 have chosen locations where their marginal willingness to pay for q is equated with its marginal implicit price.

The analysis described here results in a measure of the price of, and the marginal willingness to pay for, q, but it does not directly reveal the marginal willingness-to-pay function. The fundamental problem is that, for any one individual i, we observe only a single point along their marginal willingness-to-pay function, namely the intersection between $b^{*i}(q)$ and $\partial R\left(q,\boldsymbol{Q}_{-q}\right)/\partial q$. The second stage of the hedonic technique seeks to combine the quantity and implicit price information, together with restrictions on the structure of preferences, in an effort to identify the marginal willingness-to-pay function for q. The individual's demand price or willingness to pay for q is a function of the level of q. Since there may be substitute and complementary relationships among characteristics, the willingness to pay for q may also depend on the levels (or marginal implicit prices) of other characteristics. It is convenient to assume that the utility function is weakly separable in housing so that prices of other goods can be omitted in the specification of the marginal willingness-to-pay function. Also, it is convenient to assume that each individual purchases only one housing bundle. Purchasing more than one would necessitate that the bundles be identical, or that the hedonic price function be linear in all characteristics. This is because there can be only one marginal implicit price recorded for each individual for each characteristic.

Given these assumptions, for the ith individual we can derive a marginal willingness-to-pay function for q by differentiating the expenditure function, as shown in Chapter 4. The result is

$$b^{*i} = b^{*i}\left(q,\boldsymbol{Q}^*_{-q},u^*\right). \tag{10.5}$$

If equation (10.5) can be estimated, it can be used to estimate the welfare change of an individual associated with changes q, assuming that other things are held equal. Specifically, if the quantities of other characteristics and amenities do not change, the welfare change can be found by integrating b^{*i} over the relevant range of the change in q. However, a change in the quantity of one characteristic can result in changes in the quantities of other characteristics the individual chooses and in changes in the hedonic price function itself. The task of welfare measurement when individuals can fully adjust to the new supply of amenities and characteristics is discussed below in the section Measuring Welfare Changes.

Estimating the Hedonic Price Function

The Dependent Variable

In the discussion so far, the hedonic price has been the annual rental price of the property. What is observed usually is the purchase price of the house, which will be denoted as P_h for the rest of this chapter. P_h can be interpreted as the discounted present value of the stream of expected rental values; but this leads to two complications. First, when housing price differentials are used to estimate welfare changes that are usually expressed as annual flows, care must be taken to convert the house price measures into the appropriate temporal dimension. This topic is discussed in a later section. Second, it might be necessary to take account of expected changes in the characteristics of a house, especially environmental changes when estimating and interpreting the hedonic price function. For example, if air pollution at a given location is expected to improve over time, the present price of the house should be bid up to reflect not only the current conditions but the expected improvement as well.

One question to be asked is whether the dependent variable to be explained should be a pure site or land value or the full price of the house and land together. Since the environmental amenities of interest are location specific but not a part of the structure, the values of the environmental amenities should be reflected in the price of land alone. However, at least in the United States, land is not usually traded separately from the structures placed upon it, so the observed prices reflect the values of both the land and its structural improvements. This causes no problems at the theoretical level, but it does require that the hedonic price equation adequately control for structural characteristics.

Another question concerns the source of data on housing prices. Data on actual market transactions are preferable. For rental housing there is a regular monthly "market transaction," from which fairly accurate data on housing rents could be gathered. However, the majority of residential housing is owner-occupied, and only a small percentage of the total owner-occupied housing stock is exchanged through the market each year. The most preferred source of data is systematically collected information on actual sales prices of individual dwellings, along with relevant characteristics. Fortunately, in many parts of the country these data are collected by multiple-listing services and tax assessing agencies. Increasingly, these data are available electronically, either directly from the primary source (for example, online sites for individual county assessor offices) or from third-party data aggregators (for example, Dataquick). In recent years, virtually all published hedonic property value studies have used micro data on individual transaction prices.

An alternative source of property value data would be professional appraisals of individual properties constructed for taxation or other purposes. The appraisals themselves are typically based on one of two approaches: (a) a sales approach that assesses a home's value based on transaction prices for similar homes in the same area, and (b) a cost approach that assesses the cost of replacing the home. Some

jurisdictions have developed computer-based systems of appraisals, made for tax purposes, which include data not only on appraised values but also on a variety of structural and site characteristics. For citations to studies examining the accuracy of appraisals and tax assessments, see Kiel and Zabel (1999, Table 1).

There are two fundamental problems with the use of appraisal data. First, to the extent that a sales approach is used, the data will mask underlying variability in property values, ignoring idiosyncratic factors influencing a home's value. Indeed, as the sales approach often involves a statistical analysis of recent transactions, the resulting assessments can be viewed as providing fitted values from a first stage hedonic regression. Subsequent analysis of these assessed values will be "successful" (that is, in the sense of fitting the data) only to the extent that the researcher happens to choose a functional form that mimics what was used by the assessor. Second, for assessed values based on a cost approach, the reported home prices reflect only the cost side of an equilibrium price function.

There are, of course, concerns with individual transaction data. For example, it is important to ensure that the transactions reflect so-called "arms-length" transactions, eliminating sales between individuals who are related or those in which the sales price has been altered for reasons unrelated to the home's underlying value. It is also good to keep in mind the underlying assumption in hedonic property analysis—that the housing market is in equilibrium (that is, that all opportunities for possible gains from further trade at the revealed set of prices have been exhausted). This is a heroic assumption, because buyers and sellers often operate with substantial ignorance about the true willingness to pay and willingness to accept offers of other potential buyers and sellers. Sellers typically state an asking price that effectively truncates potential offers at that price. Sellers must choose to accept or reject offers more or less at the time they are received, without knowing when (or even if) a higher offer might come along; and buyers lack information on possible prior bids made by others for a given property (Horowitz 1986). Horowitz developed an alternative model of the bidding and acceptance strategies of buyers and sellers, and estimated both a standard hedonic model and his alternative model with the same data set. He found that the statistical performance of his bidding model was substantially superior to the standard model in predicting sales prices. However, since he used principal components in his estimation procedure rather than actual attributes, it is not possible to analyze the impact of his alternative modeling strategy on marginal implicit prices for environmental attributes.

Explanatory Variables

In choosing the appropriate explanatory variables, the first question to be addressed is the way in which environmental amenities and location characteristics enter the hedonic price function. The typical practice has been to enter a simple scalar measure of an amenity—for example, parts per million (ppm) of an air pollutant, or distance to a park. However, Parsons (1990) showed that this practice

is not consistent with a restriction imposed on the hedonic price function by profit maximizing behavior on the supply side of the housing market. The implication of profit maximization is that the effect of the environmental amenity can only be captured without bias by weighting the amenity by the area of the lot on which the house sits. The restriction is that if an area of land of given q is developed and sold in two or more different lot sizes, the prices of the lots must be such that the return per acre is independent of the sizes of the lots. For example, if two one-acre lots sell for $\$X$ apiece, one two-acre lot must sell for $\$2X$. This means that the premium on lots with higher levels of q must be twice as high for the two-acre lot compared with the one-acre lots. The higher premium on the larger lot is necessary to compensate the landowner for the forgone opportunity to capture two premiums with the smaller lots.

Although the argument is correct in principle, there is some question about its relevance in practice. As Parsons points out, once lots are developed, the cost of changing the size of lots on which houses sit may be too high to force amenity premiums to take the weighted form in the secondhand market for houses. Although Parsons showed that biased estimates of implicit prices for characteristics are possible, few empirical researchers have used his proposed weighted amenity values.

The levels of some environmental amenities are fixed by location, while the levels of others, especially those related to air quality, vary over time with changes in emissions and meteorological conditions. With time-varying amenities, there is the question of how best to represent the level of the amenity in the regression equation. The typical practice in air pollution–property value studies has been to use the annual mean as a summary statistic. However, Murdoch and Thayer (1988) have shown that in the case of visibility, using more information on the probability distribution of visual range improves the statistical performance of the hedonic price function.

A number of other conceptual and practical issues must be resolved in the course of selecting a set of explanatory variables for a hedonic price function. These include:

- Which measures of environmental quality should be used to characterize environmental amenities?
- Is it possible to separate the effects of different amenities on property values when measures of the amenities are correlated?
- What objective data best capture "neighborhood" characteristics?
- Does the spatial scale of the socioeconomic data often used in these studies correspond to peoples' perceptions of these characteristics? (For example, is there sufficient homogeneity within census tracts so that census tract means or medians adequately measure "neighborhood"?)
- Is there sufficiently close correspondence between peoples' perceptions of amenity levels (which presumably govern the choices reflected in property prices) and the objective measures of amenity levels that are available to the researcher?

Since the objective of the hedonic analysis is to determine the effect of one amenity on property values, other things being equal, a key issue is the control for structural, neighborhood, and other environmental variables. The issue is made more difficult by the likelihood of multicollinearity among housing characteristics. This raises the troublesome question of the tradeoff between increasing bias through the omission of variables that are correlated with the variable of concern and increasing the variance or imprecision of coefficient estimates when collinear variables are included. Theory does not provide any hard-and-fast answers to this question. Work by Atkinson and Crocker (1987) and Graves et al. (1988) suggested the value of approaching this question systematically, using Bayesian principles. These authors have also examined the effects of errors in the measurement of other explanatory variables on the estimates of the coefficients on the environmental variables of concern.

Finally, hedonic property analysis has often been employed to assess the impact of NIMBY ("not in my backyard") sites, such as toxic waste facilities (Kohlhase 1991), incinerators (Kiel and McClain 1995), Superfund sites (Kiel 1995), and animal confinement units (Palmquist, Roka, and Vukina 1997; Herriges, Secchi, and Babcock 2005). A difficulty in this setting is that a property may be exposed to multiple NIMBY sites, raising the question as to which site or sites to include in the analysis, and how to quantify their joint and marginal impacts. Most studies focus on the nearest site and assume that its marginal impact diminishes with distance. However, for some externalities, distance alone is a poor proxy for exposure. In the context of animal confinement units, for example, exposure to the odor generated by the various facilities can depend upon the prevailing winds. Cameron (2006) developed a more general representation of the hedonic price function that allows the marginal impact of a single site to depend upon its directional location relative to the property and to identify the direction from which the property value gradient is highest. The approach, however, still requires that the analyst focus on a single NIMBY site.

Functional Form

Functional forms for the hedonic price function that have been proposed or used in the literature include the linear, the quadratic, the log-log, the semi-log, the inverse semi-log, the exponential, and the Box–Cox transformation. The first step in choosing a functional form is to see what theory can tell us. According to theory, a hedonic price function is an equilibrium relationship derived from the interaction of individuals' preferences and suppliers' cost or profit functions. The only obvious general restriction on the form of the hedonic price function is that its first derivative with respect to an environmental characteristic be positive (negative) if the characteristic is a good (bad).

Rosen (1974) and Epple (1987) showed that it is possible to solve for the hedonic price function analytically after making specific assumptions about the form of individual utility functions and the distribution of suppliers' characteristics (Rosen)

or the exogenous supply of housing characteristics (Epple). These analytical solutions are only possible for a very limited set of assumed forms of preferences and supply. For example, Rosen assumed that individuals' utility functions were linear. This is not an attractive assumption, especially if the ultimate objective of the analysis is to measure welfare values for changes in supplies of environmental characteristics.

Early researchers tried alternative functional forms for the hedonic price function (typically the semi-log, inverse semi-log, and log-linear) and selected one on the basis of goodness-of-fit criteria. Goodman (1978) was one of the first to experiment with a flexible functional form. He employed a Box–Cox transformation of the dependent variable:

$$P_h^{[\lambda]} = \frac{P_h^{\lambda} - 1}{\lambda}. \tag{10.6}$$

For $\lambda = 1$, this is a simple linear function. As λ approaches zero, this becomes the semi-log form. Some authors have found estimates of λ that were significantly different from both zero and one, indicating that this more complicated form fits the data better than either the linear or semi-log forms.

Transforming only the dependent variable still produces a very limited range of possibilities. Halvorsen and Pollakowski (1981) proposed estimating what they called a quadratic Box–Cox functional form. It would have the form

$$P_h^{[\lambda]} = \alpha_0 + \sum_{j=1}^{J} \alpha_j \cdot q_j^{[\gamma]} + \tfrac{1}{2} \sum_{j=1}^{J} \sum_{k=1}^{J} \beta_{jk} \cdot q_j^{[\gamma]} \cdot q_k^{[\gamma]}, \tag{10.7}$$

where j and k index the characteristics and λ and γ are estimated from the data.

Cassel and Mendelsohn (1985) pointed out that for welfare analysis, it is not the goodness-of-fit of the hedonic price function that matters; rather, it is the estimate of the marginal implicit price of the environmental attribute. In the regression equation for housing price, the environmental variable is likely to have relatively little influence in determining the estimated magnitude of γ. However, the estimate of γ would have a major impact on the estimated marginal implicit price of the environmental characteristic.

A more general flexible form would be an extension of the quadratic Box–Cox of equation (10.7) to

$$P_h^{[\lambda]} = \alpha_0 + \sum_{j=1}^{J} \alpha_j \cdot q_j^{[\gamma_j]} + \tfrac{1}{2} \sum_{j=1}^{J} \sum_{k=1}^{J} \beta_{jk} \cdot q_j^{[\gamma_j]} \cdot q_k^{[\gamma_k]}. \tag{10.8}$$

This functional form allows for different transformations for each independent variable.[1] This general form may not be estimable when there are a large number of characteristics. A compromise, proposed by Palmquist (1991), would be to set $\gamma_j = \gamma_k$ for all $j,k = 1, \ldots, n-1$, where n indexes the environmental attribute of

1 For dummy variables (for example, indicating whether or not a housing unit is in a particular neighborhood), γ_j cannot be identified, and is simply set equal to one.

interest. This would be responsive to the point raised by Cassel and Mendelsohn. Allowing for a separate transformation of the environmental amenity should give better results.

One question about functional form is whether the form chosen allows the marginal implicit price of the environmental characteristic to depend on the levels of the other attributes of houses. Of the commonly used functional forms, only the log and the Box–Cox transformation make the implicit prices of characteristics depend on the levels of other characteristics—the other forms impose independence. However, this is a question that should be answered by the data, not by assumption.

One of the early efforts to systematically consider the question of functional form was a study by Cropper, Deck, and McConnell (1988). The authors simulated the performance of a housing market using real data on buyer and housing characteristics drawn from the Baltimore, Maryland, area. After specifying the functional form and parameters of individuals' utility functions and the distribution of characteristics that reflect taste differences across individuals, they solved the assignment problem, producing a housing market equilibrium with each house being sold to the individual with the highest willingness to pay for its bundle of characteristics. With knowledge of the utility function parameters, it was then possible to calculate the true marginal implicit price for each individual and for the mean across all individuals. The authors used the equilibrium prices to estimate six alternative functional forms for the hedonic price function. They were then able to compare the mean true marginal bids with the bids calculated from each hedonic price function. They found that when all of the housing characteristics were included in the hedonic price function, the linear and quadratic versions of the Box–Cox transformation provided the most accurate estimates of marginal implicit prices. However, when they experimented with various forms of incorrectly specified hedonic price functions (by omitting variables or using proxy variables) they found that the linear version of the Box–Cox transformation was consistently superior in generating marginal implicit prices.

For over 20 years, this research has provided a rationale for relying on relatively simple functional forms when estimating hedonic price functions. More recently, however, Kuminoff, Parmeter, and Pope (2010) revisited this issue in light of newer econometric techniques, including spatial fixed effects and quasi-experimental methods, designed to control for omitted variables bias. The authors undertook an extensive Monte Carlo analysis evaluating over 540 different hedonic models, with a range of underlying utility functions and assumed functional forms for the price functions, as well differences in the nature of controls used by analyst to deal with omitted variables. Their overall conclusion was that "the more flexible specifications for the price function, such as the quadratic Box–Cox model, outperform the linear, log-linear, and log-log specifications that have dominated the empirical practice for the past two decades" (Kuminoff, Parmeter, and Pope 2010, 159).

The Hedonic Price Function as a Market Equilibrium

Interpreting the marginal implicit prices as measures of households' marginal willingness to pay requires the assumption that each household is in equilibrium with respect to a given vector of housing prices and that the vector of housing prices is the one that just clears the market for a given stock of housing. These conditions assure that the hedonic price function is the price vector that makes all participants in the market in aggregate just willing to hold the existing stock of housing. For these two aspects of equilibrium to be fully achieved, we require first that households have full information on all housing prices and attributes and that their transactions and moving costs be zero; and second, that the price vector adjust instantaneously to changes in either demand or supply. The market for housing can be viewed as a stock-flow model where the flow (change in stock) is a function of prices, but the prices at any point in time are determined only by the stock at that point in time.

This idealized model is clearly not an accurate representation of real-world housing markets. However, in evaluating the strength of this criticism of the hedonic price model, we must focus on several distinct issues. One issue concerns the speed of adjustment of the market to changed conditions of supply and demand. If adjustment is not complete, then observed marginal implicit prices will not accurately measure household marginal willingness to pay. A major question is whether imperfect adjustment will lead to systematic biases in estimates of willingness to pay.

Consider households' imperfect adjustment to changing prices. First, an increase in housing prices need not affect the marginal implicit prices of attributes, in which case no adjustment of the attribute bundles is necessary. Even if marginal implicit prices change, households will not move unless the potential utility gains to returning to full equilibrium exceed the information costs, transactions costs, and moving costs associated with the change. These costs help to define a band within which observed marginal implicit prices can diverge from household marginal willingness to pay for housing attributes. If housing prices change so that the marginal implicit price schedule for an attribute moves consistently in one direction, households will consistently lag in their adjustment to that change; and the marginal willingness to pay will be overstated or understated according to whether the marginal implicit price is rising or falling.

A second issue concerns expectations about future environmental amenity levels. Market prices for long-lived assets such as housing reflect the discounted present value of the stream of expected future services from that asset. A change in expectations about future environmental amenity levels can affect housing prices and marginal implicit prices independently of the present level of these amenities. For example, if there are widespread expectations of an improvement in air quality and the market adjusts reasonably quickly to these expectations, the price differential between presently dirty houses and clean houses should decrease. Correlating these prices with existing levels of air pollution would lead to an underestimate of the marginal implicit price of air quality.

Divergences from full equilibrium of the housing market in many circumstances will only introduce random errors into the estimates of marginal willingness to pay. However, where market forces are moving continuously in one direction or are expected to move in one direction, incomplete market adjustment, or full adjustment to changing expectations, or both, can introduce biases in both directions. We should be much more cautious about utilizing the cross-section hedonic price model in those cities and at those points in time during which market forces and environmental quality levels are changing rapidly (granted that "rapidly" is an imprecise term). However, it is also possible in these circumstances to determine the direction of bias. Thus, estimates of marginal willingness to pay derived from such studies can be labeled as an upper bound or a lower bound on the basis of that analysis.

Another issue concerns the possibility of corner solutions. If there is not a sufficiently wide variety of housing models available, corner solutions are likely. The hedonic price function defines an opportunity locus across attribute space. A household chooses a housing model such that its indifference surface is tangent to the given opportunity locus, provided that a model with that precise set of attributes is available. If the optimum model is not available, the household must pick the nearby housing model that gives the highest utility level; but then the first-order conditions for utility maximization are not satisfied as equalities (Mäler 1977, 361–362).

The hedonic model is based on an assumption that the implicit price function is differentiable and continuous. However, this is an artifact of the statistical and mathematical technique. If this assumption is not satisfied in practice, two sorts of problems can arise. The first problem is that the statistically fitted hedonic price function is a good approximation only when the number of housing units is large and there is more continuous variation in characteristics among units. A small number of distinctly different types of housing units might be better analyzed with one of the discrete choice models described later in this chapter. The second type of problem arises if there are no units available with particular combinations of attributes. If there are substantial gaps in the opportunity locus, some households will not be able to satisfy the first-order conditions as equalities. This could be a problem for certain subsets of the urban population.

Market Segmentation

Straszheim (1974) was the first to raise the question of market segmentation in the context of estimating hedonic price functions for housing within an urban area. He argued that the urban housing market really consisted of a series of separate, compartmentalized markets with different hedonic price functions in each. As evidence in support of the segmentation hypothesis, Straszheim showed that estimating separate hedonic price functions for different geographic areas of the San Francisco Bay area reduced the sum of squared errors for the sample as a whole.

For different hedonic price functions to exist in an urban area, two conditions must be met. The first is that the structure of demand, the structure of supply, or both must be different across segments—either buyers in separate submarkets must have different structures of demands, or the structure of characteristics of the housing stocks must be different. The second condition is that purchasers in one market segment must not participate significantly in other market segments. In other words, there must be some barrier to mobility of buyers among market segments that prevents arbitrage from occurring in response to differences in marginal implicit prices. Such barriers could be due to geography, discrimination, lack of information, or a desire for ethnically homogeneous neighborhoods. Even with buyer immobility, if demand and supply structures are the same they will produce similar structures of hedonic prices. Perfect mobility and information on the part of buyers will also eliminate differences in the implicit prices for any characteristic across market segments.

If market segmentation does exist, the hedonic price function estimated for the urban area as a whole will provide faulty estimates of the implicit prices facing subsets of buyers in different market segments. Thus, estimates of benefits and estimates of demand functions based on faulty price data will also be faulty. If market segmentation does exist, separate hedonic price functions must be estimated for each segment; and benefit and demand functions must be separately estimated for each segment with a different set of implicit prices.

It is not clear how significant the problem of market segmentation is for air pollution–property value studies within single urban areas—although there are enough positive results in the literature to suggest that it is not a problem that can be dismissed out of hand. Some authors have found evidence of different hedonic price functions for submarkets within larger urban areas, suggesting segmentation; however, this could be due to misspecification of the model, as others have not found evidence of segmentation in their data.

The existence of market segmentation does not render the hedonic price technique invalid; but rather, it makes application of the technique more difficult. If the appropriate basis for segmentation can be identified, it is conceptually possible to estimate separate implicit price functions for each submarket. Although these functions would be different across markets, they each would accurately reflect the outcome of the market processes in each submarket. Thus, the functions could be used to estimate equilibrium marginal willingness to pay.

Econometric Concerns

While the emphasis in this chapter has been on model specification and interpretation, a number of econometric developments in recent years are worth noting. Most of them stem from concerns regarding omitted variables. First, given the spatial nature of hedonic analysis, it is likely that the error terms in a hedonic price model are correlated over space, with the correlation being larger for housing units closer to each other, precisely because these units share common

omitted variables. This so-called *spatial autocorrelation* is analogous to serial correlation in a time series setting, where in the latter case the correlation increases for observations closer to each other in time. Ignoring spatial autocorrelation will lead the researcher to overstate the precision with which individual parameters are estimated, potentially resulting in erroneous conclusions regarding statistical significance. A more problematic form of spatial dependence arises when the price of one unit depends upon the sales prices of other houses in the same region (known as spatial autoregression). Ignoring spatial autoregression can result in biased estimates of the price function parameters. Palmquist (2005) provided a helpful overview of the econometric techniques for handling both types of spatial dependence. Specific applications in the hedonic pricing literature include Bockstael (1996), Can (1992), and Bell and Bockstael (2000).

Second, as noted earlier, omitted variables can result in biased parameter estimates to the extent that the omitted factors are correlated with the regressors included in the hedonic price function. One approach to dealing with unobserved housing unit attributes (including locational amenities) is the use of repeat sales data (Palmquist 1982). Consider a housing price function given by

$$P_{hjt} = \alpha_0 + \sum_{k=1}^{K} \beta_k q_{kjt} + \xi_j + \varepsilon_{jt} = \alpha_0 + \sum_{k=1}^{K} \beta_k q_{kjt} + \eta_{jt}, \tag{10.9}$$

where P_{hjt} denotes the sales price of house j in time period t and q_{kjt} denotes the kth attribute for the same unit in time period t. The term ξ_j denotes unobserved attributes of the housing unit, which are subsumed into the error term $\eta_{jt} = \xi_j + \varepsilon_{jt}$. To the extent that ξ_j is correlated with the observable housing unit attributes (that is, the q_{kjt}), least squares regression will yield biased parameter estimates. However, if repeated sales data are available (say in time periods t and t'), then the two sales prices can be differenced, yielding a new regression equation:

$$\Delta P_{hj} \equiv P_{hjt'} - P_{hjt} = \sum_{k=1}^{K} \beta_k \left(q_{kjt'} - q_{kjt}\right) + \left(\varepsilon_{jt'} - \varepsilon_{jt}\right) = \sum_{k=1}^{K} \beta_k \Delta q_{kj} + \Delta\varepsilon_j. \tag{10.10}$$

In this case, by focusing on changes in housing prices over time for the same unit, any unobserved attributes of the house that are constant over time are differenced out.

The use of repeat sales data is not without its drawbacks. First, for any given market and time period, there will be fewer housing units that will have sold more than once, limiting the sample size available for the hedonic regression. Increasing the sample size by considering a larger market or a longer time period risks violating the assumption that a single hedonic price function applies for all the observations. Second, first differencing will also eliminate any observable site attributes that have remained constant over time. Third, if there are unobserved

housing attributes that have changed over time, the potential for omitted variable bias remains, though it is likely to have been attenuated.[2]

In recent years, a variety of *quasi-experimental* techniques have been developed to eliminate or reduce the potential bias from omitted variables (see, for example, Angrist and Pischke 2008 and Imbens and Wooldridge 2009 for general treatments). Two prominent approaches are:

- *Difference-in-differences*: A natural measure of the impact that a locational amenity (for example, air quality) has had on property values is the difference (D_0) in housing prices before and after changes have occurred in that amenity. The risk, of course, is that other things have changed as well. In its simplest form, the difference-in-differences (DID) approach attempts to control for these other factors by also measuring the difference (D_0) in housing prices over the same time period for a control group that did not experience changes to the locational amenity of interest. The difference in these differences (that is, D_1–D_0) is assumed to eliminate the nuisance of other factors and isolate the impact of the locational amenity. Chay and Greenstone (2005) provided an example of the use of DID in measuring the impact of air quality on the housing market, while Kuminoff and Pope (2012) illustrated the risks in its use.
- *Spatial discontinuities*: The regression discontinuity approach provides a means of controlling for unobservables by examining outcomes (property values in the hedonic setting) on either side of an arbitrary threshold. For example, in considering the impact of school quality on the housing market, homes immediately on either side of a school district's boundary will share commonly unobserved locational attributes (such as crime rates, access to shopping, community "feel," etc.), with differences in property values attributable to differences in observable structural characteristics of the home and differences in the associated schools. Greenstone and Gallagher (2008) examined the impact of Superfund-sponsored cleanups on housing prices using a regression discontinuity approach, comparing housing prices in areas immediately above and immediately below the threshold to qualify for Superfund cleanup. Lee and Lemieux (2010) provided a general discussion of the technique.

Estimating Characteristics Demands

The attractiveness of the hedonic price model for applied welfare analysis lies in the potential for using estimates of individuals' marginal implicit prices for a characteristic to recover the uncompensated inverse demand function for q or information on the underlying structure of preferences. Rosen (1974) had argued

2 For a more extensive discussion of the repeat sales model, see Freeman (2003, 388–390).

that the inverse demand and marginal supply price functions could be estimated from the information contained in the hedonic price function in the following manner:

> … compute a set of implicit marginal prices … for each buyer and seller evaluated at the amounts of characteristics … actually bought or sold, as the case may be. Finally, use estimated marginal prices … as endogenous variables in the second-stage simultaneous estimation of [the inverse demand and supply price functions]. Estimation of marginal prices plays the same role here as do direct observations on prices in the standard theory and converts the second-stage estimation into a garden variety identification problem.
>
> (Rosen 1974, 50)

This suggestion has been the source of a large literature for more than three decades. Since the emphasis in this book is on models and basic economic method rather than on econometric issues, only an overview is provided here of the sources of problems in estimating and identifying demand functions for characteristics and alternative approaches to solving them.[3]

The difficulties in estimating and identifying the inverse demand functions from hedonic price data come in two forms. The first arises from the fact that the source of data for the dependent variable in the marginal willingness-to-pay function is not direct observation of the inverse demand prices; rather, it is the calculation of the marginal implicit price $\partial P_h / \partial q$ from the estimated hedonic price function. However, this variable is itself computed as a function of the same characteristics that are explanatory variables in the marginal willingness-to-pay function. Brown and Rosen (1982) and Mendelsohn (1987) showed that at least in some cases this procedure leads to parameter estimates for the marginal willingness-to-pay function that are identical to the estimated coefficients in the hedonic price function. As Brown and Rosen put it,

> Contrary to Rosen's original statement, we claim that marginal attribute prices constructed as above will not necessarily play the same role in estimation that direct observation on prices would play if they were available. Because such constructed prices are created only from observed sample quantities, any new information that they may provide (that is, any information beyond that already provided directly by observed sample quantities) can only come from a priori restrictions placed on the functional form of the price function $P_h(\cdot)$. In the absence of such additional restrictions, second-stage structural

3 Readers interested in more technical discussion, especially from an econometric perspective, should consult Brown and Rosen (1982), Epple (1987), Bartik (1987), and McConnell and Phipps (1987). Bartik and Smith (1987) and Palmquist (1991) also provided useful reviews.

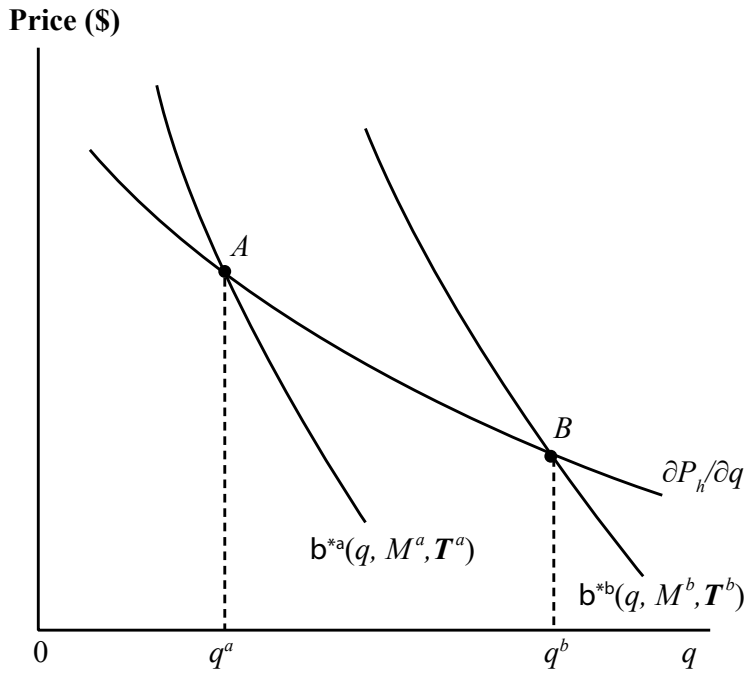


Figure 10.2 The identification problem when income and the quantity of the characteristic are correlated

> estimation of the sort suggested by Rosen may only reproduce the information already provided by the first-stage estimation of the $P_h(\cdot)$ function.
>
> (Brown and Rosen 1982, 176: notation changed by the authors)

In other words, since the second-stage estimation procedure utilizes no additional data beyond that already contained in the hedonic price function, it can do no more than reproduce the coefficients estimated from the hedonic price function.

The nature of the problem can be illustrated with Figure 10.2. This example is due to Bartik (1987, 84–85). Consider two individuals with the same income and uncompensated inverse demand functions of the form

$$b^* = b^*(q, M, \boldsymbol{T}), \tag{10.11}$$

where M is income, and $\boldsymbol{T}$ is a vector of unobserved determinants of tastes. For the first individual, we observe point A on her inverse demand function; but we have no information on the demand price for other levels of q. If the two individuals choose different levels of q, for example, q^a and q^b, it must be because of differences in tastes. This means that the demand-shifter, which is unobserved in the data of Figure 10.2, is correlated with the observed choices of q. As Palmquist put it, "the other marginal prices [on the individual's marginal willingness-to-pay function] are only observed for other individuals with other socioeconomic

characteristics and provide no information on the original consumer's bid for different quantities of the characteristic" (Palmquist 1991, 96). This makes it very difficult to separate out the effects of demand-shifters from the price–quantity relationship itself.

One approach to solving the identification problem is to impose sufficient structure on the problem by assumption to assure that the conditions for identification of the inverse demand function are met. Quigley (1982), who assumed a functional form for preferences that included homotheticity as a property, provided an early example. See Chattopadhyay (1999) for another example of this approach. By specifying the relationship between income and demand, this assumption made it possible to separate the effects of income and quantity change on the marginal willingness to pay for characteristics. Note, however, that the assumptions about functional form are not testable.

Recently, Ekeland, Heckman, and Nesheim (2004) argued that the criticisms of Rosen's two-stage estimation procedure are misleading, based on "arbitrary linearizations that do not use all of the information in the model … Nonlinearities are generic features of equilibrium in hedonic models and a fundamental and economically motivated source of identification" (2004, S60). They went on to examine identification in the context of a normal-linear-quadratic version of a single market hedonic price model, and demonstrated that "[w]ith mild functional form assumptions, the model is completely identified" (2004, S96). While the approaches developed in Ekeland, Heckman, and Nesheim (2004) still rely upon structural assumptions, the line of research would seem to be promising in that the specifications considered are relatively general and authors draw instead on implications of underlying hedonic equilibrium to bolster identification.

An alternative approach to solving the identification problem is to find cases where the marginal implicit prices of characteristics vary independently of the other demand-shift variables. Specifically, this means finding cases where individuals with the same preferences, income, and other traits face different marginal implicit prices. This can only occur if similar individuals must choose in markets with different hedonic price functions, which in turn implies either segmented markets within a city or observations taken from several different housing markets (as, for example, in different cities).

The first step in implementing this approach is to estimate separate hedonic price functions for each housing market, using the same specification. The second step is to compute the marginal implicit price faced by each individual from the hedonic price function in that market. Then the computed marginal implicit prices can be regressed on the observed quantities of the characteristics and the exogenous demand-shifters to obtain the uncompensated bid function. Assuming sufficient independent variation across markets, and assuming that there are no unobserved differences in preferences across individuals, this approach will lead to reliable and properly identified bid functions. For examples of this approach, see Palmquist (1984), Bartik (1987), and Zabel and Kiel (2000).

The second source of difficulty in estimating inverse demands for attributes lies in the fact that both the quantity of the characteristic and its marginal implicit price are endogenous in the hedonic price model. Unlike the standard market model in which an individual faces an exogenously determined price and chooses a quantity, and unlike a quantity-rationed market in which an individual faces an exogenously determined quantity and reveals a marginal willingness to pay, the individual chooses both a point on the hedonic price schedule and its associated quantity. The choice of that point simultaneously determines the marginal willingness to pay and the quantity of the characteristic.

One approach to solving this problem is to find truly exogenous variables to be used as instruments. This appears to be a difficult task, however. For some of the suggested possibilities and their problems, see Mendelsohn (1984, 1985), Bartik (1987), Bartik and Smith (1987), and Palmquist (1991). Another possibility is to assume that there is a characteristic in the marginal implicit price function that is not an argument in the marginal willingness-to-pay function for another characteristic. This makes it possible to use the omitted characteristic as an instrument. Recently many authors have been critical of this approach, since the results are only as good as the assumptions imposed to obtain them—see Mendelsohn (1987), Bartik and Smith (1987), Horowitz (1987), and Palmquist (1984). Unfortunately, such assumptions are not testable. More recently, Ekeland, Heckman, and Nesheim (2004) suggested the use of nonlinear instrumental variables as an alternative approach to addressing the endogeneity problem.

Measuring Welfare Changes

It has been established that in a housing market in equilibrium, utility-maximizing individuals equate their marginal willingness to pay for housing characteristics with the marginal implicit prices of these characteristics, and that in some circumstances it may be possible to estimate inverse demand functions based on this information. The question then is how this information on prices and preferences extracted from the hedonic housing market can be used to calculate measures of welfare change for changes in environmental amenities. The basic concepts of welfare measurement at the level of the individual are straightforward and were introduced in Chapter 4. However, measurement of aggregate welfare changes based on hedonic prices is made difficult by the adjustments that people are likely to make in response to changes in environmental attributes and the possibility that the hedonic price function will change. Also, in principle, it is necessary to consider possible changes in the supply side of the hedonic property market.

In this subsection, the basic welfare measure is defined for marginal changes in a characteristic or environmental amenity, holding other things constant, in particular, individuals' choices of housing bundles. The benefits of a nonmarginal change in an amenity are then considered, assuming that individuals cannot adjust their housing bundles by moving. This measure looks only at benefits to

purchasers of housing bundles. A fully general measure of welfare change is next examined, one that includes possible changes in profits on the supply side of the hedonic market as well as the consequences of individuals' adjustments on the demand side of the market. Since this conceptually correct measure is not implementable in practice, lower or upper bound approximations of the correct measure are considered. Finally, the subsection concludes with the consideration of localized changes in environmental quality that lead to benefits to some people without changing the hedonic price function.

Since a change in an environmental amenity in an urban area is nonexcludable and nondepletable, it is, in effect, a public good. The marginal value of the change, then, is simply the sum of the marginal willingness to pay of each of the N affected individuals evaluated at the existing housing equilibrium. In other words, for the amenity q:

$$w_q = \sum_{i=1}^{N} b^{*i} = \sum_{i=1}^{N} \left(\frac{\partial P_h}{\partial q_i} \right), \tag{10.12}$$

where w_q is the aggregate marginal welfare change and b^{*i} is the ith individual's marginal willingness to pay. Although most proposed environmental policy changes are nonmarginal in magnitude, the ease of calculating equation (10.12) may make it useful for indicating whether some improvement is desirable, by comparing this measure with an estimate of the marginal cost of the improvement.

Welfare Changes without Adjustments

Recall from Chapter 4 that the equilibrium hedonic price function is given by the double envelope of the bid and offer curves for all of the characteristics. A change in the level of q will place at least some individuals out of equilibrium, given the existing set of marginal implicit prices. Their efforts to restore their equilibria will result in changes in the hedonic price function and marginal implicit prices. Also, in principle, changes in the hedonic price function could trigger changes in the supplies of houses with different bundles of characteristics. In the next section, welfare measurement in the context of such changes will be discussed. But for now, assume that all individuals are constrained to stay at their original location, as might be the case with high transactions and moving costs or if a very short run perspective is taken; and also, assume that there is no supply response to the change in q.

Given these assumptions, the welfare value of the change in q from q^0 to q^1 is given by the sum of the areas under each individual's marginal willingness-to-pay curve over the change in q, or

$$W_q = \sum_{i=1}^{N} \int_{q^0}^{q^1} b^{*i}\left(q, \boldsymbol{Q}^*_{-q}, u_i^*\right) dq \,, \tag{10.13}$$

where W_q is the aggregate benefit. Notice that this measure requires knowledge of the marginal willingness-to-pay functions of individuals. If the uncompensated

bid functions from the second stage of the hedonic price estimation are used, the welfare gain from an increase in q will be overestimated.

There is a method for calculating exact welfare measures for nonmarginal changes in a characteristic, holding all other things constant. This method is based on an adaptation by Horowitz (1984) of Hausman's technique for exact welfare measurement for price changes (see Chapter 3). Suppose that the ith individual's uncompensated inverse demand function for q,

$$b^i = b^i\left(q, \boldsymbol{Q}^*_{-q}, M - P_h\right), \tag{10.14}$$

has been identified. Using the indirect utility function, in equilibrium,

$$\frac{\partial v/\partial q}{\partial v/\partial M} = b^i(\cdot) \tag{10.15}$$

and for individual i

$$b^i = \left(\partial P_h/\partial q\right). \tag{10.16}$$

The left-hand side of equation (10.15) is the slope of the indifference curve between the numeraire, M, and q. So, in equilibrium

$$\frac{dM}{dq} = b^i(\cdot) = \left(\partial P_h/\partial q\right)_i. \tag{10.17}$$

This expression can be solved for

$$M = f\left(q, \boldsymbol{Q}^*_{-q}, C\right), \tag{10.18}$$

where C is a constant of integration. The benefit of an increase in q is

$$W_q = f\left(q^0, \boldsymbol{Q}^*_{-q}, C\right) - f\left(q^1, \boldsymbol{Q}^*_{-q}, C\right). \tag{10.19}$$

See Horowitz (1984) and Palmquist (2005) for examples of this approach with various forms of the utility function.

In those cases where neither the uncompensated or compensated inverse demand functions are available, welfare changes could be estimated by making some assumption as to the shape of the marginal willingness-to-pay function through the original equilibrium point. Three alternative assumptions can be used to establish bounds on the true measure. One is to assume that the marginal willingness to pay for each individual is constant—that is, that the marginal willingness-to-pay function for each individual is a horizontal line through the known point. In this case, each individual's benefit for the postulated improvement in the amenity is approximated by the product of the (assumed) constant marginal willingness to pay and the change in the amenity. The aggregate benefit is obtained by summing over all individuals. This assumption leads to an estimate of aggregate benefits that is biased upward.

A second convenient assumption would be that each individual's marginal willingness-to-pay curve decreases linearly from its observed point to the point of the highest attainable level of the amenity. Marginal willingness to pay would be zero at this point. It is not clear whether this approximation would lead to

an overestimation or an underestimation of true benefits. The third assumption would be that all individuals' marginal willingness-to-pay functions are identical. Then, as discussed in Chapter 4, the marginal implicit price curve is identified as the marginal willingness-to-pay curve for the representative individual.

Welfare Changes with Full Adjustment

Bartik and Smith (1987, 1223) presented a welfare measure that takes account of all of the adjustments that individuals make in response to the nonmarginal change in q. At any location, the value of a nonmarginal change can be taken to be the integral of the values of a series of infinitesimal changes in the amenity. The value of each small change is taken to be the willingness to pay of the occupant of that site at that point in the sequence of changes. The measure for all sites together is the sum of the values for each site. It is given by

$$W_q = \sum_{j=1}^{J} \int_{q_j^0}^{q_j^1} \frac{\partial P_h\left(q, \boldsymbol{Q}_{-q}^*\right)}{\partial q} dq \,, \tag{10.20}$$

where $\boldsymbol{Q}_{-q}^*$ is the vector of all other site characteristics that are held constant by assumption, j indexes locations, and where the hedonic price function itself is changing in response to the adjustments that people are making.

In principle this measure allows individuals to relocate in response to changes in the quantity and price of the amenity, since, in effect, it sums individuals' marginal values as the amenity changes at each site. This is important, because a major limitation of some of the measures to be described below is their inability to account for individual relocation decisions. Furthermore, this measure does not require knowledge of either the marginal willingness to pay or the bid function. It relies on the fact that at each point in the sequence of changes, each individual's marginal bid is revealed by the marginal implicit price of the characteristic. However, since the hedonic price function is shifting as a consequence of the change in the amenity level, it is necessary to know how the hedonic price function and the marginal implicit prices at each location change as the levels of the amenities at each location change along the path of integration. As a practical matter, this is a major limitation of the measure.

This limitation has forced researchers to look for practical measures that can be interpreted as approximations or upper or lower bounds on the true welfare change. Following the analysis by Bartik (1988), suppose there are increases in several environmental amenities in an urban area. These increases need not be uniform across the area. Specifically, consider the case where the vector $\boldsymbol{Q}$ increases from $\boldsymbol{Q}^0$ to $\boldsymbol{Q}^1$.

First, assuming that individuals cannot move to new locations and that the hedonic price function does not change, the benefit to individuals is given by an expanded version of equation (10.11):

$$W_q = \sum_{i=1}^{N} \int_{Q_i^0}^{Q_i^1} \boldsymbol{B}^i \left(\boldsymbol{Q}, u_i^*\right) d\boldsymbol{Q}_i \,, \tag{10.21}$$

where each individual's welfare gain is computed from a path-independent line integral over the changes in the individual elements in $\boldsymbol{Q}$, and $\boldsymbol{B}^i(\cdot)$ is the vector of individual marginal willingness-to-pay functions for the characteristics.

Now, at the existing hedonic price function, some people may wish to choose different bundles of characteristics. If they do change, it must be because they perceive themselves to be better off after the adjustment. This welfare gain is in addition to that given by equation (10.21). Thus, (10.21) can be interpreted as a lower bound on the true measure; and it requires knowledge of only the bid or compensated inverse demand functions for the characteristics that change. Furthermore, the effort to adjust to different characteristics bundles is likely to affect the hedonic price function, unless the number of people wishing to do so is quite small relative to the market. It is also possible that the suppliers of housing will respond to changes in the hedonic price function by offering different bundles of housing characteristics. This could have further repercussions on the hedonic price function, and it will increase the profits of housing suppliers.

When all of these repercussions have worked themselves out, the aggregate benefit to individuals can be defined in terms of each individual's total willingness to pay for a housing unit with given characteristics, holding utility constant. Let this total willingness to pay be given by

$$WH_i\left(\boldsymbol{Q}_i^{*j}, u_i^*\right), \tag{10.22}$$

where $\boldsymbol{Q}_i^{*j}$ ($j = 0, 1$) indicates the vectors of environmental and other characteristics actually chosen by the individual in the original and new equilibrium. Each individual's total benefit is the increase in total willingness to pay for the characteristics actually chosen, holding utility constant, minus any increase in actual expenditure on housing. Summing across all individuals, we obtain

$$W_q = \sum_{i=1}^{N}\left[WH_i\left(\boldsymbol{Q}_i^{*1}, u_i^*\right) - WH_i\left(\boldsymbol{Q}_i^{*0}, u_i^*\right)\right] - \sum_{i=1}^{N}\left[P_h^1\left(\boldsymbol{Q}_i^{*1}\right) - P_h^0\left(\boldsymbol{Q}_i^{*0}\right)\right]. \tag{10.23}$$

Turning to the supply side of the market, producers, in aggregate, realize a change in aggregate profits given by the increase in expenditures on housing net of any change in their costs. This is given by

$$\Delta\text{Profit} = \sum_{i=1}^{N}\left[P_h^1\left(\boldsymbol{Q}_i^{*1}\right) - P_h^1\left(\boldsymbol{Q}_i^{*0}\right)\right] - \sum_{i=1}^{N}\left[C_h^1\left(\boldsymbol{Q}_i^{*1}\right) - C_h^1\left(\boldsymbol{Q}_i^{*0}\right)\right], \tag{10.24}$$

where $C_h^j(\cdot)$ is the cost function for producers.

The welfare change for society as a whole is the sum of equations (10.23) and (10.24). Notice that one component of this sum is simply a transfer of revenue from buyers to sellers, so it nets out. The total welfare change is the sum of the increase in total willingness to pay of individuals minus any cost increase on the part of producers of housing. Full implementation of this welfare measure would require enormous amounts of information. However, note that this measure

reduces to equation (10.21) if the hedonic price function does not change and if the change in environmental amenities does not affect the costs of supplying housing for producers.

Even if this set of conditions is not satisfied, equation (10.21) can be interpreted as a lower bound on the true measure of benefits. This can be seen by decomposing the true benefit measure into a three-step sequence of changes and adjustments. Consider first the change in amenity levels without any adjustment on the part of individuals or suppliers. The welfare change associated with this step is given by equation (10.21) plus any reduction in the costs of supplying existing houses at the affected locations. Second, suppose hypothetically that the hedonic price function is shifted to its new equilibrium position but that no adjustments to the new price function by individuals or suppliers are permitted. At this stage, although some individuals and suppliers may gain while others lose, on net all of the price changes sum to zero. At this stage, there is no net change in welfare.

Finally, allow individuals and suppliers to respond to the new hedonic price function. Any adjustments that take place at this stage must represent welfare improvements for those responding. The total welfare change is the sum of equation (10.21), any costs reduction to suppliers, and the benefits of adjusting to the price change. The latter two components are either zero or positive. Thus, equation (10.21) represents a lower bound on the true measure of benefits; and the smaller the adjustment to the changes in the hedonic price function, the smaller is the error involved in using (10.21).

Localized Amenity Changes: A Special Case

If the hedonic price function does not shift, then exact welfare measurement may be a relatively easy task. Palmquist (1992a) identified one situation in which the hedonic price function could be assumed to be constant. That is when the number of sites at which there is a change in the amenity level is small relative to the total urban market. If this is the case, and if individuals can move without cost from one site to another in response to the change in environmental amenity levels, then exact welfare measurement is straightforward. The hedonic price function can be used to predict the changes in the prices of affected properties. Benefits are exactly measured by the increase in the values of the affected properties; and knowledge of the marginal bid functions is not required.

Consider the case of an improvement from q^0 to q^1 at just one site, as shown in Figure 10.3. Assume that moving costs for occupants who choose to relocate are zero (the impact of positive moving costs is discussed below). The change in the amenity level results in an increase in the price of this house from P_h^0 to P_h^1. The owner of the property is made better off by this increase in wealth. Even though the occupant of the property experiences the increase in amenity level, he or she is worse off because of the increase in the cost of occupying the property. The occupant is shifted from point A on the curve B^0 to point B on the curve B^1. However, with costless moving, the occupant can relocate to his or her

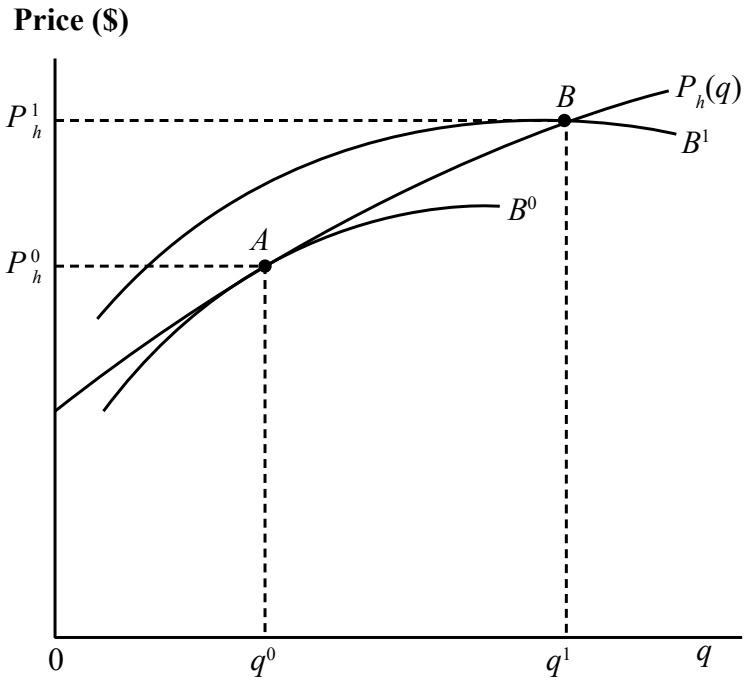


Figure 10.3 The benefit of an amenity improvement at one site when the hedonic price function is unchanged

original equilibrium position. So, the net welfare change is the increase in wealth to the owner. If the owner and occupant are the same person, the result is still the same. This individual might choose to move to a property with an amenity level somewhat greater than q^0 because of a wealth effect. However, the increase in wealth fully captures the benefit of the amenity improvement to this individual.

Now consider the case where the number of affected sites is small so that the hedonic price function does not change, but where moving costs are positive. The renter either loses the moving costs involved in adjusting his/her housing bundle or bears a loss of utility associated with staying at a less preferred location after the amenity change. In either case, the increase in property prices is an upper bound on the total benefit (Palmquist 1992b).

Rents, Taxes, and Property Prices

In the development and exposition of the theoretical model, the discussion has ignored the temporal dimension of housing prices and how welfare measures based on property prices might be converted to the annualized form usually used in welfare evaluation. It is typically the market price of a property that is observed. Inferences about the streams of rents and of benefits are drawn by converting observed present values to annual streams. The institutions of income and real

property taxation affect the way in which the market capitalizes rents (and changes and differentials in rents) into market prices for properties. These effects must be properly understood if the process of retracing these steps to infer rents from property value observations is to be successful. This subsection develops and expands on some ideas first presented by Niskanen and Hanke (1977).

In the simplest case of a stream in perpetuity and with no taxes, the conversion of property value to rent is given by

$$R = P_h \cdot r\,, \tag{10.25}$$

where r is the appropriate discount rate. The discussion proceeds by examining first the effects of the two forms of taxation separately, then their combined effects. It is shown that the effects of these two kinds of taxation on the relationship between observed differences in property values and welfare measures depends of the specific features of the tax system and parameter values such as interest rates and tax rates.

Ad valorem taxation of property can be viewed as a device for capturing some of the rent of land for the government. Since taxation affects the net return to the property owner, it should affect the market value of property as an asset. An individual would purchase a property as an asset only if its market price, P_h, is equal to or less than the discounted present value of the rental stream net of property taxation. Market forces would establish the following relationship between property values and rents:

$$P_h = \frac{R - t \cdot P_h}{r}\,, \tag{10.26}$$

where t is the ad valorem tax rate.

If property values are known, the rental stream they represent can be computed by rearranging equation (10.24):

$$R = (r + t)P_h\,. \tag{10.27}$$

Assume that the property value–amenity relationship, $P_h(\cdot)$, has been estimated. The marginal benefit of a change in q at a site is

$$w_q = \frac{\partial R}{\partial q} = (r + t)\frac{\partial P_h}{\partial q}\,. \tag{10.28}$$

The present value of this stream of benefits is

$$\frac{w_q}{r} = \left(1 + \frac{t}{r}\right)\left(\frac{\partial P_h}{\partial q}\right). \tag{10.29}$$

In other words, when the hedonic price function is defined in terms of property value, ignoring the effect of property taxation on the capitalization of rents can lead to the underestimation of benefits. The term (t/r) is a measure of the percentage error resulting from omitting the tax term in the calculation of benefits. For an interest rate of 10 percent and an effective tax rate of 10–20

mils per dollar of market value (1–2 percent), the error is between 10 and 20 percent.

However, this is not the whole story. The income tax code treats the imputed rental income of homeowners differently than it does the rental income of landlords. The absence of a tax liability for imputed rent further complicates the task of inferring annual rents and benefits from observations of (capitalized) market prices for housing assets. This is because the market will place different values on two assets with the same rental stream if one is subject to income taxation while the other is not.

Assume two perpetual assets indexed as a and b with equal annual returns of R per year. The return to the first asset is taxable at the rate g percent, while the return to the second asset incurs no tax liability. If r represents the market rate of return on assets with taxable returns, the two assets will be priced so as to equalize the after-tax rate of return:

$$P_a = R_a / r \tag{10.30}$$

and

$$P_b = \frac{1}{(1-g)} \frac{R_a}{r} > P_a . \tag{10.31}$$

If P_b is observed, the tax-free rental stream can be computed as

$$R_b = r(1-g)P_b . \tag{10.32}$$

Taking account of this adjustment and using equations (10.26) and (10.27), the marginal annual benefits of amenity changes to homeowners and their present value are

$$w_q = r(1-g)\partial P_b / \partial q \tag{10.33}$$

$$\frac{w_q}{r} = (1-g)\partial P_b / \partial q . \tag{10.34}$$

Ignoring the effects of income taxation leads to an overestimation of benefits. The discount factor $r(1-g)$ is analogous to the municipal bond rate, and it arises for the same reason. However, where the marginal tax rate is itself a function of income, g varies across individuals, and equations (10.33) and (10.34) must be computed separately for each individual.

The tax code confers additional benefits on homeowners by exempting them from taxation on capital gains realized on the sale of a primary residence and by allowing them to deduct property tax payments in calculating taxable income. This latter provision lowers the real cost of the property tax by g percent. Combining these effects (ad valorem taxation, deductibility, and exemption of imputed rental income), we have

$$P_b = \left[R_b - (1-g)tP_b\right] / r(1-g) . \tag{10.35}$$

Solving for R_b gives

$$R_b = (r+t)(1-g)P_b. \tag{10.36}$$

Marginal benefits are calculated by

$$w_q = (r+t)(1-g)\left(\frac{\partial P_b}{\partial q}\right) \tag{10.37}$$

and

$$\frac{w_q}{r} = \left(1+\frac{t}{r}\right)(1-g)\left(\frac{\partial P_b}{\partial q}\right). \tag{10.38}$$

The effects of ignoring taxation in calculating benefits depend on the magnitudes of g and t/r. The higher the marginal income tax rate, the more likely benefits would be overstated if taxes were ignored. For an example, suppose the marginal income tax rate is 30 percent, the opportunity cost of capital is 10 percent, and the property tax rate is 2 percent. Then the terms in parentheses come to 0.84. Ignoring tax effects would lead to an overstatement of benefits by almost 20 percent. However, lower income tax and discount rates can reverse this conclusion. An alternative approach to dealing with taxation and discounting is to base the hedonic equation on measures of user cost (called gross rent by Sonstelie and Portney 1980). This variable captures the full cost of owning (and using) an asset such as a house. User cost would include property taxes and the opportunity cost of capital plus any change in the market price of the asset over the interval, say a year. It would be calculated as follows:

$$u = (r+t+m)P_h, \tag{10.39}$$

where m is the percentage rate of change in market value.

The user cost approach differs from that outlined above in two respects. The first is the inclusion of the change in market value over time. P_h could be changing because of physical depreciation of the house, general price inflation, changes in the price of housing relative to other goods, and changes in the variables determining P_h. Only the latter changes have relevance for benefit estimation, and they would be captured by modified versions of equations (10.37) and (10.38), which are generalized from the assumption of constant streams in perpetuity. However, the depreciation term might be useful in empirical work as an approximation of expected changes in these variables, provided that it were adjusted to net out general price level effects.

The second difference arises in considering the effects of some provisions of the income tax code on user cost. For one thing, the tax exemption for imputed rent does not affect the user cost of holding a house, since user cost is an opportunity cost. For another, the tax deductibility of mortgage interest does affect user cost, but it does not affect the market capitalization of streams of benefits. If user cost is used to compute benefits, the net result of these two effects is to overstate benefits in comparison with equations (10.37) and (10.38).

Discrete Choice Models

The hedonic price model is based on the assumption that each attribute of the housing bundle is a continuous variable and that an individual can choose any point on the continuous and differentiable hedonic price function in the n-dimensional attribute space. As noted above, this is clearly not a completely realistic assumption, and in some respects it may seriously misrepresent the problem of choosing a bundle of housing attributes. For example, the number of bedrooms in a house is not a continuous variable. There may be no one-bedroom houses on one-acre lots, or four-bedroom houses with swimming pool and attached garage on one-quarter-acre lots. Discrete choice models provide an alternative way of looking at housing choice and inferring values for housing attributes.

Some of the discrete choice models that have been used in the literature focus on the individual's bid function for housing bundles. Such models are based on the probability that an individual will be the highest bidder for a specified bundle of housing attributes. These are known as bid rent models or random bidding models. An alternative approach is to focus on the individual's utility function defined on housing attributes. These models investigate the probability that a specified bundle of housing attributes (including the price of the bundle) will be chosen by the individual, drawing on the Random Utility Maximization (RUM) models described in Chapter 3, and used extensively in the recreation demand literature.

Both types of models can be used to derive the marginal bid or marginal willingness-to-pay function for individual attributes from an estimate of the bid function or indirect utility function. Thus, in principle, both types of models allow for the calculation of the benefits of changes in an environmental attribute, at least assuming no relocation and no changes in the hedonic price function. Both types of models start with the assumption of utility maximization.

In the bid rent model developed by Ellickson (1981), the utility maximization problem is solved, subject to the standard budget constraint defined by income, prices of market goods, and the hedonic price function, to obtain the individual's bid function—that is, the bid as a function of the housing attributes and income, holding utility constant. As a practical matter, individuals are then grouped into broad "type" categories (based on, for example, income, household size, and other socio-demographic attributes), assuming that bid functions are homogenous within type. The bid function for individuals of type t for a particular housing unit j can be written as a function of the observable housing attributes ($\boldsymbol{Q}_j$) plus a random error term reflecting unobserved attributes of either the individual or the housing bundle:

$$B_{tj} = B_t\left(\boldsymbol{Q}_j\right) + \varepsilon_{tj}\,. \tag{10.40}$$

For examples of this type of model, see Ellickson (1981), Lerman and Kern (1983), and Gross (1988).

The probability that a household of type t will occupy housing unit j will be determined by whether or not they outbid any other household type; that is,

$$\begin{aligned}\Pr(t \mid j) &= \Pr\left(B_{tj} > B_{t'j}\ \forall t' \neq t\right) \\ &= \Pr\left[B_t\left(\boldsymbol{Q}_j\right) - B_{t'}\left(\boldsymbol{Q}_j\right) + \varepsilon_{tj} - \varepsilon_{t'j}\ \forall t' \neq t\right].\end{aligned} \tag{10.41}$$

If the random error terms are independently and identically distributed with a Type I Extreme Value distribution, this probability can be written in the logit form:

$$P\left[t \mid \boldsymbol{Q}_j\right] = \frac{\exp\left[B_t\left(\boldsymbol{Q}_j\right)\right]}{\sum_{s=1}^{T}\exp\left[B_s\left(\boldsymbol{Q}_j\right)\right]}, \tag{10.42}$$

where T denotes the number of household types. As Lerman and Kern (1983) and Gross (1988) pointed out, estimation of equation (10.42) fixes only the slope of the bid function, not its level. However, information on the bids actually paid can be used in the estimation process to fix these values and to make it possible to calculate the marginal bid functions for individual attributes.

Note that the focus in the bid-rent model is on which type of individual occupies a given housing unit. In contrast, in the RUM model, the emphasis is on modeling which housing unit is chosen by a given individual. Specifically, suppose that the conditional utility that individual i receives from choosing housing unit, j, is given by

$$V_{ij} = V\left(R_j, \boldsymbol{Q}_j, M_i, \boldsymbol{S}_i; \beta\right) + \varepsilon_{ij}, \tag{10.43}$$

where R_j denotes the rental cost of housing unit j, $\boldsymbol{S}_i$ is a vector of socio-demographic characteristics for individual i, β is a vector of parameters for the conditional utility function, and ε_{ij} is a random error term assumed to capture unobservable attributes of the individual/housing unit, known to the individual but unobserved by the analyst. The individual is presumed to choose the housing unit that maximizes their utility, so that

$$\begin{aligned}\Pr(j \mid i) &= \Pr\left(V_{ij} > V_{ij'}\ \forall j' \neq j\right) \\ &= \Pr\left[V\left(R_j, \boldsymbol{Q}_j M_i, \boldsymbol{S}_i; \beta\right) - V\left(R_{j'}, \boldsymbol{Q}_{j'} M_i, \boldsymbol{S}_i; \beta\right) + \varepsilon_{ij} - \varepsilon_{ij'}\ \forall j' \neq j\right].\end{aligned} \tag{10.44}$$

If, as in Cropper et al. (1993), the error terms are assumed to be independent and identically distributed with a Type I extreme value distribution, then the probability that individual i will choose housing unit j can be written in the standard logit form, with

$$\Pr(j \mid i) = \frac{\exp\left[V\left(R_j, \boldsymbol{Q}_j M_i, \boldsymbol{S}_i; \beta\right)\right]}{\sum_{j'=1}^{\mathcal{J}}\exp\left[V\left(R_{j'}, \boldsymbol{Q}_{j'} M_i, \boldsymbol{S}_i; \beta\right)\right]}, \tag{10.45}$$

where $\mathcal{J}$ denotes the total number of housing units in the individual's choice set. Knowledge of the parameters of the indirect utility function makes it possible to

compute welfare measures for changes in any of the housing attributes, including an environmental amenity. One practical issue with the RUM model in this setting is defining the choice set. An individual moving to a large metropolitan region may have literally thousands of housing units to choose from, but may actually only consider a small fraction of these options. In practice, analysts often focus on modeling the choice of a community or neighborhood, assuming that the housing units are largely homogeneous within a given neighborhood.

In an interesting simulation study, Cropper et al. (1993) compared estimates of welfare measures derived from a hedonic price model with those from the random utility model for given known household preferences. After simulating an equilibrium in an urban housing market, they used the resulting hedonic price and individual choice data to estimate both a hedonic price model with its marginal bid functions and a random utility model. They then calculated welfare measures for 25 percent and 100 percent changes in each of ten attributes, including both neighborhood attributes and attributes of individual houses (such as number of bathrooms, lot size, and age). They found that the random utility model provided more accurate estimates of the known welfare measure than the hedonic price model, and this was true even when they assumed that the researcher did not know the true form of individuals' utility functions. They suggest that the reason for this is the difficulty in identifying and obtaining accurate estimates of the marginal bid functions with the hedonic price model when data are generated by only a single market.

Other researchers (see for example, Bartik and Smith 1987, 1224–1225, and Palmquist 1991, 119) have suggested that while one of the strengths of the discrete choice model is its ability to generate welfare measures for nonmarginal changes relatively easily, the model only does so because it forces the researcher to make strong assumptions about the functional form of the utility function or bid function. If similar strong assumptions are made about the functional form of the inverse demand functions for attributes in the hedonic model, these functions can be identified too. Cropper et al. (1993) suggested that even when the functional form of preferences is known, the discrete choice model outperforms the hedonic model as a way of measuring welfare change. For other examples of empirically based efforts to compare welfare measures from standard hedonic models with random utility and random bidding models, see Chattopadhyay (1998, 2000) and Palmquist and Israngkura (1999).

Equilibrium Sorting Models

During the past decade, an alternative paradigm has emerged for modeling the supply and demand for differentiated commodities such as housing. Whereas the hedonic pricing literature focuses attention on the equilibrium outcome of the market, the equilibrium sorting literature seeks to characterize the sorting process itself (that is, how the distributions of individual preferences, production costs, and

locational amenities interact to yield a market equilibrium in which the supplies and demands for housing amenities are equated, and both the distributions of housing types and their prices are determined).[4] With a model of the sorting process in hand, it is conceptually straightforward to evaluate both marginal and nonmarginal policy scenarios by simply simulating the new sorting equilibrium that would emerge. These capabilities, of course, come at a cost, as they invariably rely upon assumptions regarding the structure and/or distribution of preferences.

This section provides an overview of equilibrium sorting models. Much of the discussion draws on a recent and more comprehensive review of this literature by Kuminoff, Smith, and Timmins (2013). Attention here is focused on two particular sorting model frameworks: (a) the *pure characteristics sorting model* developed by Epple and Sieg (1999) and (b) the *random utility sorting model* developed by Bayer, McMillan, and Reuben (2004).[5] These are sometimes referred to in the literature as the *vertical* and *horizontal* sorting models, respectively. At the heart of both models is Tiebout's (1956) notion that households "vote with their feet," choosing among available communities both in terms of the private amenities that a specific house in the community would provide and the accompanying exposure to both positive and negative public goods. Specifically, let $\boldsymbol{G}_k$ denote the vector of public amenities provided by community $k = 1,\ldots, K$ and $\boldsymbol{H}_j$ denote the vector of private amenities provided by housing unit $j \in k$ in community k. In the notation of the previous section, the complete vector of amenities for housing unit j is then given by $\boldsymbol{Q}_j = \left(\boldsymbol{G}_k, \boldsymbol{H}_j\right)$. The individual is assumed to choose a community k, a housing unit $j \in k$ in that community, and a level of expenditures on a numeraire good z so as to maximize her utility subject to a budget constraint; that is,

$$\max_{k,j\in k,z} u\left(z, \boldsymbol{G}_k, \boldsymbol{H}_j; \alpha_i, \boldsymbol{S}_i\right) \qquad \text{s.t.} \quad z + R_{j\in k} = M_i, \tag{10.46}$$

where $R_{j\in k}$ denotes the annual rental expenditure for housing unit j in community k, α_i denotes unobserved individual characteristics influencing the individual's preferences and $\boldsymbol{S}_i$ denotes observable individual characteristics. In general, the individual's income might also depend upon the community chosen, but for ease of notation, this generalization is ignored here. The two modeling frameworks differ in the additional structure imposed on the consumer's optimization problem.

4 To be fair, of course, the hedonic pricing literature incorporates in theory many of these factors as well, as in Ekeland, Heckman, and Nesheim (2004), but has focused on the equilibrium price outcome as a practical matter.

5 Kuminoff, Smith, and Timmins (2013) also consider a third framework, which they refer to as the *Nechyba–Ferreyra model*, developed by Ferreyra (2007) and based on theoretical models of Nechyba (1997, 1999, 2000). This approach, however, has largely been applied to modeling housing and school choices, and the interaction of these choices with the production of education.

The Pure Characteristics Sorting Model

The vertical sorting model begins by assuming that the annual rental expenditures associated with living in housing unit *j* can be decomposed into a housing quantity index $h_j = h(\boldsymbol{H}_j)$, reflecting the private attributes of the unit, and a price index $p_k = p(\boldsymbol{G}_k)$, reflecting the cost of purchasing a home in community *k* (per unit of the housing index); that is, $R_{j\in k} = p_k h_j$. Thus, $h(\boldsymbol{H}_j)$ reflects the *amount* of housing one receives from a specific unit as it varies in physical attributes, such as square footage, number of bathrooms, etc., whereas $p(\boldsymbol{G}_k)$ captures how the price of the housing changes across communities with different levels of public amenities. In addition, it is traditionally assumed that the public good attributes of the community can be summarized by a one-dimensional index of the attribute vector $\boldsymbol{G}_k$ (i.e., $\tilde{g}_k = g(\boldsymbol{G}_k)$). Conditional on choosing community *k*, the individual agent's maximum utility becomes[6]

$$v_k = v(\tilde{g}_k, p_k, \alpha_i, M_i) \equiv \max_{h,z} u(z, \tilde{g}_k, h; \alpha_i) \qquad \text{s.t.} \quad z + p_k h = M_i . \tag{10.47}$$

In this context, α_i is treated as an indicator of the individual's preference for public goods; that is, all else equal, an individual with a high value of α_i will prefer to live in a community with more public goods.

The household is assumed to choose the community that maximizes its utility. Prices adjust to reflect the demand for housing in the community and the available stock of housing and public amenities. An equilibrium emerges when individuals are in their desired community and no longer wish to move. The problem is that, without further structure, it is not clear that such an equilibrium exists, or that it is unique.

The solution employed in the vertical sorting literature is the so-called *single crossing condition*, introduced by Ellickson (1971) and generalized by Epple and Platt (1998). The single crossing condition requires that indifference curves in $(\tilde{g}, p)$ space, implied by $v(\tilde{g}, p, \alpha, M)$ in equation (10.47), cross only once for different individuals. Formally, as Kuminoff, Smith, and Timmins (2013) note, the slope of these indifference curves is given by

$$F(\tilde{g}, p, \alpha, M) = -\frac{\partial v(\tilde{g}, p, \alpha, M)/\partial \tilde{g}}{\partial v(\tilde{g}, p, \alpha, M)/\partial p} . \tag{10.48}$$

The single crossing condition is satisfied if *F* is increasing in $(\alpha | M)$ and is increasing in $(M | \alpha)$. Figure 10.4 illustrates this condition for three individuals with the same preference parameter α, but different income levels. For any one individual, with their given level of income (*M*) and attitude towards public goods (α), utility is increasing as one moves down (due to lower housing prices) and to the right (due to higher levels of public goods). Individual *i* in the figure is

6 For simplicity, the observed individual attributes are dropped, subsumed for now into the preference parameter *i*.

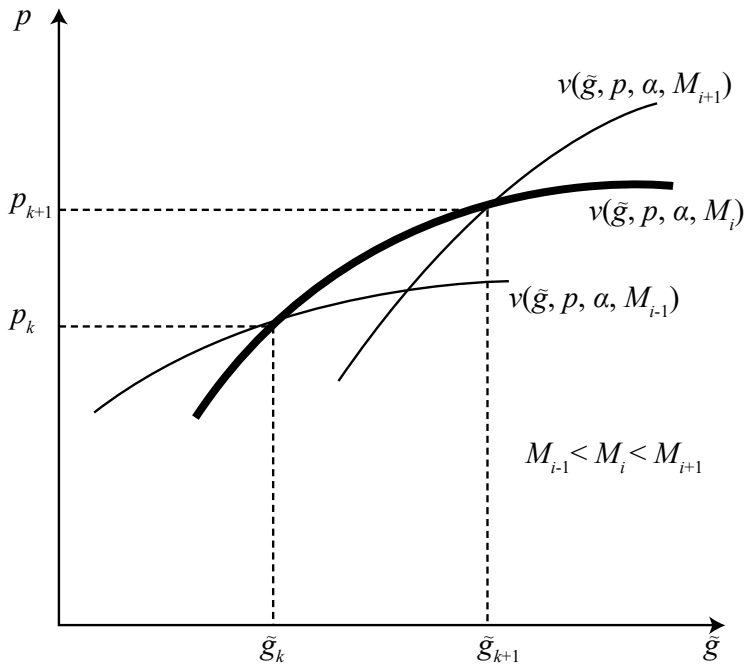


Figure 10.4 The single crossing condition

indifferent between communities k and $k + 1$, whereas individual $i - 1$ would prefer community k and individual $i + 1$ prefers community $k + 1$.

The single crossing condition yields three properties of the sorting equilibrium that are key to estimation: *boundary indifference, increasing bundles*, and *stratification* (Epple and Platt 1998). To define these properties, suppose that the K communities are ordered by the public good index $\tilde{g}_k$, with $\tilde{g}_1 < \tilde{g}_2 < \ldots < \tilde{g}_K$. Then the three properties are:

- Boundary indifference: There exists a "boundary" B_k between any two adjacent communities (k and $k + 1$) defined in (α, M)-space such that
$$B_k = \left\{(\alpha, M) \middle| v(\tilde{g}_k, p_k, \alpha, M) = v(\tilde{g}_{k+1}, p_{k+1}, \alpha, M)\right\}. \tag{10.49}$$
The term "adjacent" in this context refers to the communities' proximity to each other in the ordering from 1 to K, not necessarily their proximity geographically.
- Increasing bundles: The communities' rankings by the public good index match the rankings by price (i.e., $p_1 < p_2 < \cdots < p_K$).
- Stratification: For a given level of income, individuals sort themselves by preference parameter α; and individuals in communities $k - 1$, k, and $k + 1$ with the same income sort such that

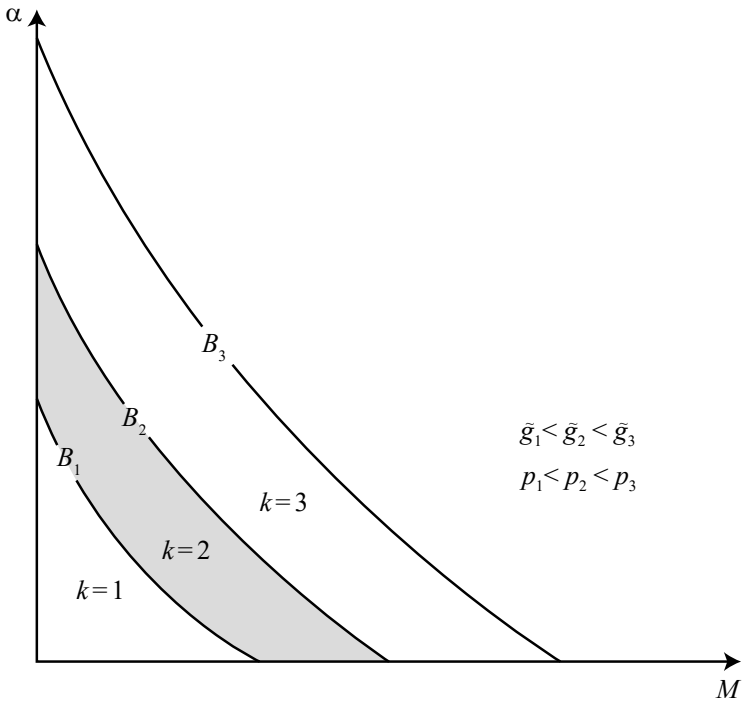


Figure 10.5 Market segmentation

$$\left(\alpha_{k-1}\middle|M\right) < \left(\alpha_k\middle|M\right) < \left(\alpha_{k+1}\middle|M\right). \tag{10.50}$$

Likewise, for a given level of the preference parameters, individuals sort themselves by income. In other words,

$$\left(M_{k-1}\middle|\alpha\right) < \left(M_k\middle|\alpha\right) < \left(M_{k+1}\middle|\alpha\right). \tag{10.51}$$

The implication of these three properties in terms of sorting is illustrated in Figure 10.5. The solid lines indicate the boundaries (B_k) between communities. The shaded region indicates the set of individuals (defined in terms of α and M) choosing to live in community $k = 2$. Note that the make-up of individuals in a given community is not defined solely in terms of either income (M) or solely in terms of preferences (i.e., α), but by their interaction. An individual with a relatively low income will choose to live in community 2 rather than the less expensive community 1 if they place a high enough value in the additional public goods that community 2 offers (that is, they have a high enough α). Conversely, an individual with a relatively high income may choose to live in community 2 rather than community 3 if they care relatively little for the available public goods (that is, they have a low enough α).

In order to close the model, additional structure is still needed. In particular, one needs a functional form for the indirect utility function in (10.47), an assumption

regarding the joint distribution of income and preferences, and a functional form for the public goods index $\tilde{g}_k = g(\boldsymbol{G}_k)$. A commonly used specification for the indirect utility function is a constant elasticity of substitution (CES) form, with

$$v(\tilde{g}, p, \alpha, M) = \left\{ \alpha \cdot \tilde{g}^{\rho} + \left[\exp\left(\frac{M^{1-\upsilon} - 1}{1-\upsilon} \right) \exp\left(\frac{\beta \cdot p^{\eta+1}}{1+\eta} \right) \right]^{\rho} \right\}^{\frac{1}{\rho}}. \tag{10.52}$$

As Kuminoff, Smith, and Timmins (2013) noted, this structure implies that housing demand in the chosen community is given by

$$h = \beta p_k^{\eta} M^{\upsilon}, \tag{10.53}$$

so that η represents the price elasticity of housing demand and υ denotes the income elasticity of housing demand. The parameter ρ determines the substitutability between private and public goods.

The public good index is typically assumed to be a linear function of both the observed and unobserved public good attributes of a community; that is,

$$\tilde{g}_k = \gamma_1 g_{1,k} + \gamma_2 g_{2,k} + \cdots + \gamma_L g_{L,k} + \xi_k, \tag{10.54}$$

where $g_{\ell,k}$ denotes the ℓth observed public good for community k and ξ_k is a composite of all the unobserved attributes of community k. An important characteristic of this index is that it does not vary by individual. This implicitly assumes that all individuals weigh the component elements of the public good index the same. This is likely to be a strong assumption in some settings. For example, if one of the public goods is elementary education and another is public health care facilities, individuals with children may weigh these attributes differently than those who are single or elderly. In other settings, the assumption may be more innocuous, but it represents an important limitation of the vertical sorting model as it has been implemented to date and an area where additional research is needed.

The last element in specifying the model is to choose a joint distribution for the preference parameter and income pair (α, M). These are typically assumed to be from a joint lognormal distribution. Parameters of this distribution are inferred, in part, by comparing the observed income distributions in individual communities with those implied by the stratification boundaries (such as for community 2 in Figure 10.5).

Details of the econometric procedures required to estimate vertical sorting models are not described here, as they have varied somewhat across individual applications. Kuminoff, Smith, and Timmins (2013) provide a general discussion of some of the approaches used. To get a sense of at least part of the process, consider a stylized and simple version of Figure 10.5 depicted in Figure 10.6, in which it is assumed that there are only three communities, $k = 1, 2, 3$. Furthermore, suppose that *both* income (M) and preferences for public goods (α) are observable and known to be independently and uniformly distributed in the population; that is, $M \sim U[0, \overline{M}]$ and $\alpha \sim [0, \overline{\alpha}]$. Finally, suppose that the shaded region in Figure 10.6 depicts those individuals actually living in community 2. According to the

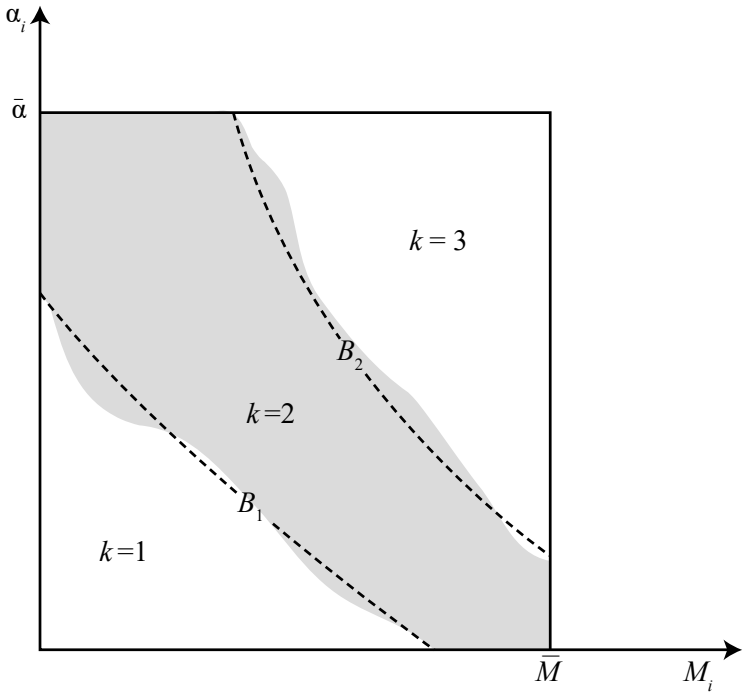


Figure 10.6 Estimating the parameters of a vertical sorting model

vertical sorting model outlined above, the proportion of the population living in community 2 is dictated by the region between the boundaries B_1 and B_2 defined in equation (10.10). Given observed housing prices in the three communities, these boundaries are in turn determined by the preference parameters in (10.13); that is, (β, η, ν, ρ). Estimation of these parameters involves adjusting them so that the resulting estimated boundaries (the dashed lines in Figure 10.6) imply as closely as possible the observed distribution of individuals in the three communities.

With the parameter estimates in hand, the general equilibrium implications of a policy changing the levels of public goods can be evaluated. Changes in the public goods will cause a shift in the community boundaries, which will in turn lead to a mismatch between existing housing supply and the new levels of demand in each community. The new equilibrium is constructed by finding the levels of housing prices that restore supply and demand in each community. Evaluation of the impact of the proposed policy can then include the overall impact of the policy and distribution of its impact on various subpopulations.

Individual applications include Epple and Sieg's (1999) study of school quality and public safety in the Boston metropolitan area, Smith et al.'s (2004) examination of general versus partial equilibrium implications of air quality improvements in the Los Angeles basin, and Walsh's (2007) analysis of open space as an endogenous public amenity.

The Random Utility Sorting Model

The second locational sorting model, initially developed by Bayer, McMillan, and Reuben (2004), builds on the discrete choice literature and McFadden's (1974) RUM model. Indeed, the starting point for this approach is essentially the same as the discrete choice RUM model discussed above at the end of the hedonic pricing model section. Specifically, an individual is assumed to face a discrete set of alternatives in their choice set, defined in terms of communities ($k = 1,\ldots,K$) and housing units (or housing types) available within each community (that is, $j \in k, j = 1,\ldots,\mathcal{J}_k$). The conditional utility that individual i receives from choosing housing unit j in community k is assumed to take the form[7]

$$\begin{aligned} u^i_{j\in k} &= \boldsymbol{\alpha}^i_H \boldsymbol{H}_{j\in k} + \boldsymbol{\alpha}^i_G \boldsymbol{G}_k + \alpha_R R_{j\in k} + \xi_{j\in k} + \varepsilon^i_{j\in k} \\ &= v^i_{j\in k} + \varepsilon^i_{j\in k}\ , \end{aligned} \tag{10.55}$$

where $\boldsymbol{\alpha}^i_H$ and $\boldsymbol{\alpha}^i_G$ are vectors of individual-specific parameters associated with the respective vectors of housing and community characteristics, $\xi_{j\in k}$ represents unobserved attributes of the housing unit j (including potentially unobserved community characteristics), and $\varepsilon^i_{j\in k}$ denotes idiosyncratic unobservable factors influencing $u^i_{j\in k}$. The individual-specific parameters are typically modeled as functions of observable individual attributes (i.e., such as age, gender, etc. denoted by $\boldsymbol{S}_i$ in equation (10.46) above), with

$$\boldsymbol{\alpha}^i_a = \boldsymbol{\alpha}_{a0} + \sum_{\ell=1}^{L} \boldsymbol{\alpha}_{a\ell} s^i_\ell\,, \tag{10.56}$$

where s^i_ℓ denotes the ℓ th attribute in $\boldsymbol{S}_i$.

Substituting (10.56) into (10.55) and collecting terms, we can rewrite (10.55) as

$$u^i_{j\in k} = \alpha_{j\in k} + \tilde{\boldsymbol{\alpha}}^i_H \boldsymbol{H}_{j\in k} + \tilde{\boldsymbol{\alpha}}^i_G \boldsymbol{G}_k + \varepsilon^i_{j\in k}\,, \tag{10.57}$$

where

$$\alpha_{j\in k} = \boldsymbol{\alpha}_{H0} \boldsymbol{H}_{j\in k} + \boldsymbol{\alpha}_{G0} \boldsymbol{G}_k + \alpha_{R0} R_{j\in k} + \xi_{j\in k} \tag{10.58}$$

denotes a housing-type specific constant and

$$\tilde{\boldsymbol{\alpha}}^i_a = \sum_{\ell=1}^{L} \boldsymbol{\alpha}_{a\ell} s^i_\ell\,. \tag{10.59}$$

Individuals are assumed to choose a community (k) and housing unit ($j \in k$) that maximizes their utility.

Before getting into either the equilibrium sorting of households or the econometric issues associated with estimating the discrete choice model in (10.57) and (10.58), there are several of its features that are worth highlighting. First, unlike in the vertical sorting model, individuals are able to have differing preferences

7 One can also make the coefficient on rental costs (that is, αR) a function of individual observable attributes (that is, $\boldsymbol{S}i$), but this complication is not considered here in order to simplify the exposition.

regarding both the observable housing characteristics ($\boldsymbol{H}_{j\in k}$) and the community level public goods ($\boldsymbol{G}_k$). The baseline (or common) impact of each attribute is captured by either $\boldsymbol{\alpha}_{H0}$ or $\boldsymbol{\alpha}_{G0}$, but $\tilde{\boldsymbol{\alpha}}_H^i$ and $\tilde{\boldsymbol{\alpha}}_G^i$ allow for differences in the marginal utility of an attribute depending on socio-demographic characteristics. Thus, families with children can value local public schools and health care facilities differently than those who are single or elderly. Recall that in the vertical sorting model, everyone is assumed to weigh the community's public goods in the same manner (see equation 10.54). Second, the horizontal sorting model can readily handle discrete quality attributes for both communities and housing units. This is in contrast to the standard hedonic pricing model, wherein individuals are assumed to face a continuum of housing units in all quality dimensions. Third, the structure can also allow for friction in the sorting process in the form of moving costs, for example by incorporating a fixed cost associated with alternatives in either different communities or different cities (see, for example, Bayer, Koehane, and Timmins 2009). Finally, notice that the conditional indirect utilities in (10.57) include an alternative specific constant (ASC) $\alpha_{j\in k}$. These ASCs reflect the overall appeal of the alternative, which, as indicated by equation (10.58), depends upon the observed features of the housing unit ($\boldsymbol{H}_{j\in k}$), the public goods available in the community ($\boldsymbol{G}_k$), the cost of the unit ($R_{j\in k}$), and features of the unit not observed by the analyst ($\xi_{j\in k}$). The terms $\tilde{\boldsymbol{\alpha}}_H^i$ and $\tilde{\boldsymbol{\alpha}}_G^i$ capture how the appeal of the unit varies by observable individual attributes. The advantage of this structure is that it controls for a myriad of possible unobservable attributes for the housing unit. Unfortunately, it also means that there are as many ASCs to estimate as there are alternatives in the choice set. This creates a practical problem in terms of estimation, as the choice set can become large in individual applications. This has led to the use of a two-stage estimation procedure outlined below.

There are also econometric issues associated with insuring the consistency and asymptotic normality of the estimators used (see Berry, Linton, and Pakes 2004). This has led researchers to aggregate housing units into types or classes within each community. For example, Klaiber and Phaneuf (2010) organized housing units within communities by size (so that j references small, medium, and large housing types), whereas Tra (2010) organized housing units by ownership status, number of bedrooms, dwelling type, and when the housing unit was built.

Estimation of the model typically proceeds in two steps. If the $\varepsilon_{j\in k}^i$ are assumed to be independently and identically distributed according to a Type I extreme value distribution, then the probability that an individual i chooses housing unit (or housing type) j in community k is given by the usual logit structure, with

$$P_{j\in k}^i = \frac{\exp\left(v_{j\in k}^i\right)}{\sum_{m=1}^{K}\sum_{\ell=1}^{\bar{J}_m}\exp\left(v_{\ell\in m}^i\right)}. \tag{10.60}$$

The alternative specific constants (i.e., the $\alpha_{j\in k}$) and the socio-demographic parameters $\tilde{\boldsymbol{\alpha}}_H^i$ and $\tilde{\boldsymbol{\alpha}}_G^i$ are recovered. One of the challenges at this stage of

estimation is the potentially large number of alternative specific constants to be estimated. Researchers have typically drawn on a number of convenient features of the logit model to simplify estimation (see, for example, Murdock 2006 and Klaiber and Phaneuf 2010).

The second stage in the estimation process involves using the fitted alternative specific constants in order to estimate the parameters in (10.59). The key issue here is that the unobserved housing and community characteristics, represented by $\xi_{j\in k}$, are likely correlated with the observed housing and community characteristics, as well as the housing price itself (i.e., $R_{j\in k}$). In order to resolve this potential for omitted variables bias, the horizontal sorting literature has drawn on instrumental variables procedures developed in the industrial organization literature (for example, Berry, Levinsohn, and Pakes 1995).

Up to this point, the model is essentially no different from the discrete choice housing models discussed in the previous section. The difference arises when one recognizes that the choice probabilities themselves provide a measure of aggregate demand ($d_{j\in k}$) for housing by community and housing type. Specifically,

$$\begin{aligned} d_{j\in k} &= d_{j\in k}\left[\boldsymbol{H}_{j\in k},\boldsymbol{G}_k,\alpha_{j\in k}\left(\boldsymbol{H}_{j\in k},\boldsymbol{G}_k,R_{j\in k}\right)\right] \\ &= \sum_{i=1}^{N} P^i_{j\in k}\left[\boldsymbol{H}_{j\in k},\boldsymbol{G}_k,\alpha_{j\in k}\left(\boldsymbol{H}_{j\in k},\boldsymbol{G}_k,R_{j\in k}\right),\boldsymbol{S}_i\right]. \end{aligned} \tag{10.61}$$

Note that the price of housing ($R_{j\in k}$) has its impact in this specification entirely through the alternative specific constant. If $t_{j\in k}$ denotes the total available housing of type *j* in community *k*, then the equilibrium is characterized by

$$t_{j\in k} = d_{j\in k}\left[\boldsymbol{H}_{j\in k},\boldsymbol{G}_k,\alpha_{j\in k}\left(\boldsymbol{H}_{j\in k},\boldsymbol{G}_k,R_{j\in k}\right)\right]. \tag{10.62}$$

The general equilibrium impact of a change in the community public goods (for example, an improvement in air quality or public education, changing $\boldsymbol{G}_k$ to $\tilde{\boldsymbol{G}}_k$) is found by solving for the housing prices (say $\tilde{R}_{j\in k}$) that re-establish the equilibrium:

$$t_{j\in k} = d_{j\in k}\left[\boldsymbol{H}_{j\in k},\tilde{\boldsymbol{G}}_k,\alpha_{j\in k}\left(\boldsymbol{H}_{j\in k},\tilde{\boldsymbol{G}}_k,\tilde{R}_{j\in k}\right)\right]. \tag{10.63}$$

This involves solving for the new set of alternative specific constants

$$\tilde{\alpha}_{j\in k} = \alpha_{j\in k}\left(\boldsymbol{H}_{j\in k},\tilde{\boldsymbol{G}}_k,\tilde{R}_{j\in k}\right), \tag{10.64}$$

that re-establishes the equilibrium in (10.63) and backs out the implied housing prices. Changes in welfare can then be inferred using standard log-sum formulas for evaluating changes in an RUM model. Changes in the composition of individuals living in a given community can be inferred by decomposing the housing demand in (10.61) by groups of interest (for example, the elderly, households with children, etc.).

There have been a number of applications of the horizontal sorting model to date, including Klaiber and Phaneuf's (2010) analysis valuing open space in the Twin Cities, Tra's (2010) examination of the impact in the Los Angeles basin of

the 1990 Clean Air Act Amendments, and Bayer, Ferreira, and McMillan's (2007) consideration of how school quality impacts housing and community selection. As noted above, there are a number of appealing features of the horizontal sorting approach to valuing quality-differentiated commodities. There are, of course, limitations as well. First, much of this literature has relied on the standard logit structure in modeling choice probabilities. This specification makes the econometrics substantially easier, but requires the rather strong Independence of Irrelevant Alternatives (IIA) assumption (discussed in Chapter 3). In particular, it requires that the idiosyncratic error terms in (10.57) (that is, $\varepsilon^{i}_{j\in k}$) are uncorrelated across the choice options for a given individual. Since these terms capture unobserved individual specific factors influencing housing choice, this can be a strong assumption, depending upon how much information is observed about individual (and housing) characteristics. Relaxing this structure and using a nested logit or mixed logit model would seem worth consideration. Second, defining the choice set may be difficult, much like it is in the context of recreation demand. The housing units considered by individuals may span differing time periods (i.e., they may be looking for a house over a short or long time period) and communities. Banzhaf and Smith (2007) provided an excellent discussion of this issue, examining the impact of a wide range of possible choice sets. Their analysis, however, focused on the use of a single choice set across individuals, whereas the choice set itself may be individual specific. More research is needed into defining the choice set.

Summary

Hedonic price theory provides a coherent basis for explaining housing prices as a function of the levels of characteristics embedded in each house, including the environmental amenities and disamenities determined by the housing unit's location. Measures of value for marginal and nonmarginal changes in local public goods can be derived from a properly specified hedonic price model. Values for marginal changes in amenity levels are found simply by adding up the observed or computed marginal willingness to pay for all affected individuals. However, for nonmarginal amenity changes, when the hedonic price function itself might shift, welfare measurement requires knowledge of the inverse demand function or the income-compensated bid function for the amenity. These, in turn, require a solution to the daunting identification problem. As a result, much of the empirical literature has limited its attention to the so-called first stage of hedonic analysis—that is, estimating the hedonic price equation.

Beyond the difficulty in estimating both stages of the hedonic price model, and hence using it to assess nonmarginal shifts in amenities, there are some additional limitations to the hedonic property value model for use in estimating welfare effects.

First, the hedonic model assumes that consumers of housing can select their most preferred bundle of characteristics from a complete range of levels of all

characteristics. In practice, the available housing units are finite both in numbers and in individual housing characteristics (for example, number of bathrooms and bedrooms, styles, etc.). This weakens the theoretical connection, depicted in equation (10.4), between the marginal cost of a housing attribute and the individual's marginal willingness to pay for the attribute that lies at the heart of using hedonic price analysis for welfare evaluation.

Second, since the property value models are based on the consequences of individuals' choices of residence, they do not capture willingness to pay for improvements in environmental amenities at other points in the urban area—for example, in the work place, shopping areas, or parks and recreational areas, or at second homes (see for example, Smith 2007).

Third, because the property value models are based on observing behavioral responses to differences in amenity levels across houses, they only capture willingness to pay for perceived differences in amenities and their consequences. For example, if there are subtle, long-term health effects associated with reduced environmental quality at some housing sites, but people are unaware of the causal link between these effects and the housing site, their willingness to pay to avoid the effects will not be reflected in housing price differences. Despite these limitations, the traditional hedonic price model provides a valuable tool in assessing or bounding the welfare implications of changing environmental amenities. It should also be noted that, while most applications of the hedonic model have used residential property values, the technique is also applicable to commercial and agricultural properties. For example, Mendelsohn, Nordhaus, and Shaw (1994) provided an analysis of the impact of climate on agricultural land values with a goal of assessing the effects of global warming on agriculture.

The equilibrium sorting literature has emerged in recent years as a means of addressing some of the issues associated with the hedonic pricing model. Both the vertical and horizontal sorting models allow for evaluation of nonmarginal shifts in public goods, including environmental amenities, by modeling the equilibrium process itself. This is particularly important in that many policy scenarios, such as those resulting from the 1990 Clean Air Act Amendments, involve significant changes in the environment that are likely to alter the housing market's equilibrium. Many of the applications to date suggest that ignoring general equilibrium effects of a policy (changing the hedonic pricing function and the resorting of households) can significantly bias the estimated welfare impacts of that policy. The horizontal sorting model also provides resolutions to two other issues in the hedonic price model by allowing for (a) a discrete, rather than continuous, stock of available housing units and (b) frictions in the housing market stemming from moving costs. The gains from the equilibrium sorting models, of course, come at a cost, including the need for additional assumptions regarding the structure of consumer preferences and the choice set available to consumers. At the same time, they represent a promising avenue for future research in efforts to understand how environmental amenities are capitalized into the housing market.

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Chapter 11

Hedonic Wage Models

The hedonic wage model is a formalization of the concept of compensating wage differentials, which can be traced back to Adam Smith. The basic idea is that, other things being equal, workers will prefer jobs with more pleasant working conditions as opposed to those that are less pleasant. The greater supply of workers for pleasant jobs will depress the wage levels of such jobs. In equilibrium, the difference in wages between two jobs with different working conditions will reflect the workers' monetary valuations of these differences.

The basic hedonic wage model has been refined and has been applied empirically to two important questions of particular interest to environmental and resource economists and policymakers. One question concerns the value of reducing the risk of death, injury, or illness. The hedonic wage model has been used to estimate the wage–risk tradeoff as a revealed preference measure of this value. The other question concerns the values of the environmental and social amenities that vary across regions. Wage differences across regions have been used as indicators of the values of region-specific environmental, cultural, and social amenities.

From a worker's perspective, a job can be viewed as a differentiated product; that is, a good with a bundle of characteristics such as working conditions, prestige, training and enhancement of skills, and levels of risk of accidental injury and exposure to toxic substances. If workers are free to move from one urban area to another, then jobs are also differentiated, in part, by the environmental and other characteristics of the urban areas in which the jobs are located. If workers are free to choose from a menu of differentiated jobs, the hedonic price technique can be applied to the data on wages, job characteristics (including their locations), and worker characteristics to estimate the marginal implicit prices of these job characteristics.

Employers, from their perspective, can be viewed as choosing from among a set of workers of different characteristics. This is a distinguishing feature of labor markets. In the typical application of the hedonic theory to differentiated goods, producers are viewed as selling a good embodying a package of characteristics and as being indifferent to the characteristics of the purchaser of the good. In hedonic wage studies, the employer is viewed as selling a package of job characteristics

(including the quality of the work environment); but at the same time the employer is purchasing work effort and cannot be indifferent to the productive characteristics of the firm's employees. Thus, the hedonic wage equation must be interpreted as an equilibrium relationship that reflects not only the interaction of supply and demand for job characteristics, but also the interaction of supply and demand of worker characteristics (see Lucas 1977 and Rosen 1979). This means that both worker and job characteristics must be included as arguments in the estimated hedonic wage equation.

As in the case of hedonic property values, the derivative of the hedonic wage function with respect to any job characteristic can be interpreted as the marginal implicit price of that characteristic. If the worker is maximizing utility, the marginal implicit price can be taken as an estimate of the worker's marginal willingness to pay for the characteristic. It gives the change in income necessary to just compensate for a small change in the characteristic. Since in general the hedonic wage function need not be linear, these marginal values may be different for different workers. Similarly, the derivative of the hedonic wage function with respect to any worker characteristic gives its marginal implicit price and, assuming profit maximization, the marginal value of that characteristic to the employer.

In order to estimate the value of a nonmarginal change in a characteristic, it is necessary to know the compensated inverse demand function for it. As in the case of hedonic property values, the inverse demand function cannot be estimated from data from a single labor market unless additional restrictions are imposed. Some examples of efforts to identify these functions are described in the next section.

The interpretation of the hedonic wage function as revealing marginal implicit prices and marginal values requires that all of the transactions that make up the data be undertaken in the same market. In other words, each buyer (seller) in the market must have had the opportunity to match up with any of the other sellers (buyers) and to choose the most preferred given prices, and so forth. In the terminology of Chapter 10, the market must be in equilibrium and must not be segmented into submarkets with incomplete mobility among segments. When hedonic wage equations are estimated using data from several urban areas, it is necessary to assume that these areas are part of a single market. In practice, labor markets can be segmented on the basis of geography, with moving costs and lack of information on job alternatives imposing barriers between labor markets in different parts of the country. Markets can also be segmented on the basis of education and skill requirements, and between blue-collar and professional–managerial workers. Geographic segmentation can lead to different marginal implicit price schedules in different regions. Segmentation on the basis of occupation or education level can lead to different marginal implicit price functions across occupational categories. One approach to the problem of geographic segmentation is to estimate the hedonic wage function only for occupational groups that are believed, on a priori grounds, to be part of a national labor market. In general, the extent of market segmentation

and its significance for empirical estimation of hedonic wage functions are not known.

The next section of this chapter discusses the application of the hedonic wage model to measure the value of risk reduction, and reviews the evolution of empirical applications of this model. Much of the recent literature calls into question earlier estimates of the marginal value of risk reduction, as well as their applicability to valuing the impact of environmental programs (see, for example, Cropper, Hammitt, and Robinson 2011, Black and Kniesner 2003, and Black, Galdo, and Liu 2003). At the same time, new data sources and econometric procedures hold out hope for resolving a number of these issues. This chapter then goes on in the second section to outline and describe some of the models that have been developed to explain and interpret interregional wage differences as reflections of the values people place on regional amenities. Included in this discussion is the recent introduction of equilibrium sorting models, described in Chapter 10, to the process of modeling the impact of local amenities on both labor and housing markets.

Wage Differences and the Value of Reducing Risks

Chapter 7 introduced the concept of the value of statistical life (VSL) and its role in estimating the value of policies that reduce the risk of death. It was also noted that inferences about the value of reductions in risk and the VSL could be extracted from revealed preferences (through the wage–risk tradeoff, for example). This section discusses the use of the hedonic wage model as one approach to estimating the individuals' willingness to pay for reductions in risk. Most of the applications of the hedonic model to risk valuation have dealt with risks of death due to accidents on the job.

Suppose that each individual (with attributes $\boldsymbol{S}$) chooses a job so as to maximize expected utility from consumption of the numeraire, z, and from the vector of job characteristics, $\boldsymbol{Q}$. In addition to $\boldsymbol{Q}$, each job is characterized by its risk of accidental death, δ. Individuals face a hedonic wage function that is the locus of points at which firms' marginal wage offers (as functions of job and worker characteristics) equal workers' marginal acceptance wages (see Chapter 4 for a description of equilibrium in hedonic markets). This function is

$$p_w = p_w(\delta, \boldsymbol{Q}, \boldsymbol{S}) \tag{11.1}$$

where p_w is the weekly or monthly wage and where hours of work per period could be one of the characteristics in $\boldsymbol{Q}$. Individual characteristics enter the hedonic wage function because they may impact both an individual's preferences and their productivity as a worker. The individual chooses a job to maximize expected utility subject to the wage constraint—that is

$$\max E[u] = \pi \cdot u(z, \boldsymbol{Q}, \boldsymbol{S}) + \lambda\left[p_w(\delta, \boldsymbol{Q}, \boldsymbol{S}) - z\right], \tag{11.2}$$

where π is the probability of surviving the period and being able to consume z. In wage–risk studies, it is risk of death, δ, rather than survival probability, π, which is observed. The relationship between the two is given by $\pi = (1-\delta)(1-\varphi)$, where φ is the probability of dying from non-work related causes. Since φ is usually small for the working-age population, π is approximately equal to $(1-\delta)$.

The first-order conditions governing the choices of z, job risk (δ), and job characteristics ($\boldsymbol{Q}$) are

$$\pi \cdot \left(\frac{\partial u}{\partial z}\right) = \lambda, \tag{11.3}$$

$$\frac{u(\cdot)}{\lambda} = \frac{\partial p_w}{\partial \delta}, \tag{11.4}$$

and

$$\frac{\pi \partial u / \partial Q}{\lambda} = -\frac{\partial p_w}{\partial Q}, \tag{11.5}$$

for all job characteristics, $\boldsymbol{Q}$.

From equation (11.3), λ is the expected marginal utility of consumption, which, by assumption, is positive. According to equation (11.4), the marginal willingness to pay for an increase in the probability of surviving the job risk must equal its marginal implicit price. Equation (11.4) also implies that wages must be lower for jobs that are safer; that is, the marginal implicit price of an increase in π is a decrease in the wage rate. Equation (11.5) requires that the marginal willingness to pay for each job characteristic equal its marginal implicit price.

If workers know the relationship between market wages and job attributes and risks, then each worker selects the collection of job attributes and risks that equates the marginal benefit of each attribute to its marginal cost. In the case of risk of death, the marginal cost of working in a less risky job is the lower wage received, $\partial p_w / \partial \delta$, and this must be equated with the marginal willingness to pay for lower job risk. In other words, the risk premium associated with a higher-risk job must be equal to the individual's marginal willingness to accept compensation for risk.

Estimating Marginal Values for Risk

Data on wages, job attributes, and worker attributes are used to estimate the hedonic wage function—an equilibrium relationship between wages on the one hand, and job characteristics and variables affecting worker productivity on the other. If the hedonic wage function can be estimated satisfactorily, the risk premium for a marginal change in risk can be calculated by evaluating the partial derivative of the function at a given risk level and set of individual and job attributes. However, if the wage–risk tradeoff locus is nonlinear, this marginal willingness to pay will

vary with the baseline risk of each worker. For nonmarginal changes in risk, the value of the change to the individual cannot be calculated from the wage–risk tradeoff curve alone because of the convexity of the individual's indifference curve. For example, see Figure 4.5.

The task of estimating the hedonic price function is, of course, not without its challenges. Indeed, a number of recent reviews call into question a large portion of the existing literature. Cropper, Hammitt, and Robinson (2011, 331), for example, argue that "many of the older studies suffer from both data and econometric problems" and "it is time to replace the older set of studies with newer results." Fortunately, improvements in both the available data and econometric techniques have emerged in recent years, holding out hope for improved estimates of the marginal value of risk. This subsection provides an overview of the key issues in estimating the hedonic price function.

Measurement Error

Perhaps the biggest challenge in hedonic wage studies of risk is constructing the appropriate measure(s) of risk. Most studies rely upon objective measures of fatality risk, matched to individuals based on their occupation and/or the industry in which they work. These objective risk measures, however, are subject to a variety of potential measurement errors and may bear little resemblance to the risk perceptions of either the firms or the individuals that underlie the hedonic wage equilibrium.

One of the major questions in interpreting estimates of willingness to pay for risk reduction is whether individuals perceive differences in risks across jobs and, if so, whether these perceptions are accurate. Compensating wage differentials for risk can exist in the labor market only if workers perceive differences in risks across jobs. The absence of compensating differentials need not mean that workers do not value reducing the risk of accidental death, only that they are unaware of the differences in risks; and if individuals have inaccurate estimates of job risks, then risk premia can exist. However, these risk premia will yield biased estimates of individuals' marginal willingness to pay for risk reduction unless the researcher can identify what it was that individuals thought they were buying when they accepted a particular job with its bundle of characteristics. Since most hedonic wage studies use objective measures of job risk, it is important to find out if individuals' perceptions correspond well with these objective measures.

The only evidence pertaining directly to individuals' perceptions of job-related risks is found in research that compares workers' risk perceptions with data on frequency of job-related death and injury. This evidence suggests that workers' perceptions are positively correlated with objective risks, but may overstate them. Viscusi (1979) reported a positive correlation between a dichotomous risk variable (posed as a question: "Is your job dangerous?") and accident rate data from the Bureau of Labor Statistics (BLS). In a subsequent study, Viscusi and O'Connor (1984) reported that workers in the chemical industry perceived risk of injury on

their job to be about 50 percent higher than BLS estimates. Gerking, de Haan, and Schulze (1988) also found that workers' perceptions of risk of death on the job overstated BLS accidental death rates. Liu and Hammitt (1999) found evidence that workers' risk perceptions evolve over time with their experience on the job, which would in turn influence their propensity to stay with a firm and the resulting hedonic price equilibrium.

Other studies providing evidence on the accuracy of risk perceptions compare relative frequency of deaths, by cause, with individuals' perceptions of these frequencies. Slovic, Fischhoff, and Lichtenstein (1979) found that, on average, people overestimate the likelihood of infrequent causes of death (for example, deaths due to botulism, floods, tornadoes) but underestimate the probability of deaths with higher frequencies (for example, deaths due to heart disease, cancer). However, Fischhoff et al. (1981) noted that one must distinguish between an individual's perception of the relative frequency of death in some population and the individual's estimate of his or her *own* risk of death. There is evidence that the latter is often underestimated (see Hamermesh 1985 and Fischhoff et al. 1981).

There is, of course, an analogous issue for firms, but there appears to be no research regarding the relationship between objective risk measures and firms' subjective perceptions of them. To the extent that the perceptions of firms and workers differ, it is not enough to include only one side's perceptions in the analysis, as it is the interaction of these two sets of agents that determines the resulting hedonic wage function. Ideally, one would want to understand the mechanisms by which objective risk measures are converted to subjective perceptions of these risks by *both* sides of the market.

Even ignoring the potential disconnect between subjective and objective risk measures, there are significant concerns regarding the objective risk measures underlying most of the hedonic wage models prior to the year 2000. Until relatively recently, the bulk of hedonic wage studies have relied on fatality data from two sources: the Bureau of Labor Statistic's Survey of Worker Conditions (BLS-SWC) and the National Institute of Occupational Safety and Health's National Traumatic Occupation Fatality Survey (NIOSH-NTOFS). In both instances, the available sources provide only fatality measures, requiring independent information on total employment in an industry and/or occupation in order to convert the fatality rates to risk measures. In the case of the BLS-SWC, the data were available only by two-digit and three-digit Standard Industrial Classification (SIC) code. The problem here is that risks can vary significantly within an industry depending on the individual's specific occupation (e.g., the steel mill worker versus clerical workers in the steel industry). The NIOSH data, on the other hand, were aggregated along different lines, providing fatality measures at the one-digit occupation or industry levels and by state. This form of aggregation, of course, creates its own problems. For both data sources, concerns have been raised regarding the accuracy of the fatality counts themselves (see, for example, Viscusi and Aldy 2003 and Dorman and Hagstrom 1998).

Black, Galdo, and Liu (2003) provided an extensive analysis of these two data sources and the implications for their use in valuing risk reductions. They argued that the data suffer from significant measurement error and that studies that fail to account for this measurement are likely to significantly understate the value of risk reduction. They went on to conclude that their findings lead them "to have severe doubts about the usefulness of existing estimates to guide public policy. These estimates are so highly sensitive to the risk measure used and the specification of the wage equation that the selection of any particular value of the price of risk seems arbitrary" (2003, 3). In another study, Black and Kniesner (2003) found that comparable risk measures from the two data sources "are not highly correlated, with a maximum correlation being 0.53," again drawing into question hedonic wage studies of risk that are based upon them.

Fortunately, several databases have emerged in recent years that seek to address the shortcomings in the BLS-SWC and NIOSH data. Of particular importance is the Census of Fatal Occupational Injuries (CFOI), which the BLS began collecting in 1992. One advantage of the CFOI is that it is available at the two- or three-digit SIC industry and occupation level. Moreover, fatalities reported in the CFOI are based on at least two independent sources, avoiding some of the reporting errors thought to plague both the BLS-SWC and NIOSH databases (see, for example, Mellow and Sider 1983 and Bound, Brown, and Mathiowetz 2001). A number of hedonic wage studies have employed this relatively new source of risk measures, including Kniesner et al. (2012), Kniesner, Viscusi, and Ziliak (2010), and Viscusi (2003, 2004). In a novel study, Lee and Taylor (2011) took advantage of plant-level injury and fatality data gathered by the Occupational Safety and Health Administration (OSHA) to construct plant-level risk measures, linking these to plant-level wages and worksite characteristics and avoiding some of the concerns with employing national measures of risk. Research based upon these alternative data sources is relatively new, but seems promising in terms of ameliorating concerns regarding measurement errors in the earlier BLS and NIOSH data.

Finally, even with these newer databases, an ongoing concern will be the inevitable imprecision with which fatality risk can be measured given the infrequent occurrence of on-the-job deaths in many industries and occupations. This problem is likely to only get worse over time with improvements in workplace safety and as researchers seek to provide a better (i.e., finely-tuned) match between the individual and the available fatality risk data. In order to alleviate this issue, it is not uncommon for researchers to average fatality risks over several years.

Omitted Variables Bias

The hedonic wage function represents the equilibrium outcome from the interaction between workers and firms. As such, it depends on the characteristics of both workers (influencing their productivity and preferences) and jobs (influencing worker utility and firm costs). Failure to control for characteristics in either category can lead to significant omitted variables bias. The classic example

in the hedonic wage literature is the importance of controlling for both the fatality *and* injury risks associated with a job. Industries with high fatality rates are also likely to have high injury rates. Since workers presumably value reductions in both risks, and the two risks are positively correlated, including only fatality risk in the estimated hedonic wage function will tend to overstate its value to workers.

Equally important is controlling for worker characteristics. For example, it is not uncommon to find that the highest paying jobs also have some of the lowest levels of risk. This (unconditionally) negative correlation between risk and wage rates should not be construed as an indication that workers are willing to pay (in the form of lower wages) for increased risk. Rather, the high salary positions are usually associated with higher levels of education and/or experience. Firms are willing to pay more (both in wages and in the costs associated with providing lower levels of risk) in order to attract such workers.

A related concern is that of *endogenous risk*, with workers sorting themselves into jobs both within and across sectors based on unobserved skills in managing risk (Shogren and Stamland 2002). Individuals with "cool hands" may choose high-risk jobs, not because they place little value in risk reduction, but because their perceived (and perhaps actual) risk is lower than the objective risk measures in the industry. Failure to control for this sorting process will tend to understate the value of risk reductions.

Finally, an important development in recent years is the availability and use of panel data techniques to control for possibly omitted variables, including worker and industry/occupation characteristics. The use of individual fixed effects, for example, will control for any unobserved worker characteristics that are constant over time, mitigating the problem of omitted variables bias. Kniesner et al. (2012), for example, found that controlling for unobserved individual characteristics reduced their estimated wage–risk premium by up to 60 percent. Other examples include Hinterman, Alberini, and Markandya (2010), Kniesner, Viscusi, and Ziliak (2010), and Scotton and Taylor (2011). A potential downside of using panel data in hedonic wage analysis is that it implicitly assumes that the market equilibrium does not shift over the course of the panel, or that it shifts in ways that can be readily accounted for (using, for example, time-fixed effects). Changes in workplace regulations and the costs to firms of reducing risk, as well as shifts in the overall market, will endanger this assumption.

Inter-Industry and Inter-Personal Wage Differentials

In their recent review of the hedonic wage literature, Bockstael and McConnell (2007) highlighted the importance of controlling for inter-industry wage differentials. They noted that there is substantial evidence that "wage premia exist for some industries, irrespective of type of job or job attributes" (2007, 201). If labor markets are segmented across different industries, and wages are not allowed to equilibrate, then potential problems will arise with inferring risk values from differences in wages and risks across industries. This will be particularly the case

if the factors impeding the equilibration (say differences in market power) also influence the levels of risk in the industry. This suggests, at a minimum, including industry dummy variables in the hedonic wage model, though it may also call into question treating such disparate industries as part of a single labor market.

A related issue can arise if there are persistent wage differentials due to discrimination. If an ethnic or gender class is effectively barred from a segment of the job market, their choices can no longer fully reflect their willingness to pay for risk reduction. Their choice of a low paying job with high levels of risk may simply reflect the more narrowly defined segment of the labor market that they have access to. Viscusi (2003) examined differences in labor market by race and the implications for these differences in terms of measuring wage–risk tradeoffs. This highlights, again, the importance of controlling for worker characteristics in estimating a hedonic price function.

Functional Form

Relatively little consideration has been given to the choice of functional form in hedonic wage literature. The vast majority of applications model log-wages, with risk typically entering linearly into the hedonic wage function. However, there have been a number of studies allowing for more flexible functional forms. Just over 20 percent of the cases included in Mrozek and Taylor's (2002) meta-analysis of the VSL literature allow risk to enter quadratically into the hedonic wage function, a term that is often found to be statistically significant. Moore and Viscusi (1988) and Shanmugam (1997) considered Box–Cox representations of the hedonic wage function that nest the semi-log and linear representations. In both cases, both the linear and semi-log representations are rejected as restrictions on the more general functional form. Given these results, and the related findings in the hedonic property literature by Kuminoff, Parmeter, and Pope (2010), it would seem prudent for researchers to consider more flexible functional forms moving forward.

Identifying Marginal Willingness-to-Pay Functions

We know that the partial derivative of the hedonic wage function with respect to each characteristic is its marginal implicit price, and that in equilibrium we can take the marginal implicit price to be a point estimate of each worker's marginal willingness to pay for that characteristic. However, as explained in Chapter 10, a second stage of analysis is required in order to identify the marginal willingness-to-pay function for each characteristic. As in the property value literature, there have been very few efforts to identify the willingness-to-pay function for job characteristics.

Two studies have dealt with the identification problem in somewhat different ways. In both cases, the authors imposed structure on the problem by making explicit assumptions about the form of the underlying utility functions. In one

study (Biddle and Zarkin 1988), identification of the marginal willingness-to-pay function for reducing risks of an accidental injury on the job was achieved by an instrumental variables approach. The second study (Viscusi and Moore 1989) is noteworthy for two reasons. The first is that the authors identified the marginal willingness-to-pay function for reduced risks of death by making use of interregional variation in the hedonic wage function. To identify the parameters of the marginal bid function, the marginal price of job risk was estimated for different regions of the United States, thus assuring variation in marginal price that is independent of the variables entering the marginal bid function. The second reason is that the authors explicitly took into account the age of each worker and the number of expected life-years at risk.

Viscusi and Moore specified a lifetime utility maximization problem, which was a simplification of equation (7.17) in Chapter 7 of this book, in that the exogenous probability of death was constant across all time periods. This assumption made it possible to derive a simplified expression for the first-order conditions and the marginal willingness to pay for any given explicit utility function. The model also made it possible to derive estimates of the implied discount rate on life-years. Viscusi and Moore used data from the 1982 wave of the University of Michigan Panel Study of Income Dynamics, supplemented by National Traumatic Occupational Fatality data published by NIOSH. Depending upon the specification of the utility function in the estimation technique, the implied discount rate ranged from 11 to 17 percent for the sample as a whole (see also Moore and Viscusi 1990).

Conclusions

There is an extensive empirical literature suggesting the existence of a positive wage–risk premium (Kniesner and Leeth 1991; Liu, Hammitt, and Liu 1997; Shanmugam 2000), though the size of the premium varies substantially depending upon the data set being used, the time period analyzed, and modeling assumptions employed (Mrozek and Taylor 2002; Viscusi and Aldy 2003). It appears that (a) workers perceive differences in risks across jobs; (b) these perceptions are correlated with objective measures of on-the-job risks; and (c) workers prefer jobs with lower risks, other things being equal, and are willing to pay for safety in the form of reduced wages. Yet, serious concerns have been raised about the accuracy of the available empirical estimates, particularly those prior to the year 2000. While drawing on the best data available at the time (such as the BLS-SCW and NIOSH data), these studies are plagued by measurement and omitted variables bias stemming from the disconnect between both firms' and workers' perceptions of risks and available objective measures, the accuracy of the available risk data that are aggregated over occupational groupings and/or industry categories, and the ability to control for other individual and job-related determinants of wage differentials. Recent advances in econometric techniques, as well as the advent of new data sources (such as

the BLS-CFOI), provide promise in terms of mitigating these concerns, though more work clearly remains.

It has also become clear that there is no one single value for risk reduction; rather, individuals' values depend upon such things as age and income, and even the type of fatality risk the person is exposed to. Studies that have examined the role of age in the wage–risk premium include Aldy and Viscusi (2007, 2008) and Evans and Smith (2006, 2008). Scotton and Taylor (2011) provided one of the few studies to differentiate the value of risk reduction according to type of risk, finding that the risk premium associated with homicide risk is substantially higher than that associated with traditional workplace risk. Understanding these individual determinants of behavior toward risk and the preferences and values that lie behind them is critical to their continued use in the policy arena. Indeed, this issue represents a potentially fundamental concern with the use of the hedonic wage model to value environmental risk. As Scotton and Taylor noted, "the application of VSL estimates from labor market studies, which our results clearly indicate are driven by traditional sources of workplace risks (e.g., electrocutions, falls, traffic accidents), to policy contexts involving reducing latent cancer risks, or premature mortality from acute asthmatic events, for example, is very likely inappropriate" (2010, 394). Similarly, if the value of risk reduction is age dependent, then estimated values for risk reduction based solely on hedonic wage models will be suspect, relying as they do on attitudes of working-age individuals and not on the preferences of young children (or their parents) and the elderly that typically face the highest risks from environmental pollutants.

Interurban Wage Differences and the Value of Amenities

All of the studies discussed in the previous section share the common characteristic of assuming that the location of the job is unimportant; that is, they do not control for differences in urban amenities and the potential effects of amenities on wage levels across cities. However, in a series of papers Smith and Gilbert have included location-specific characteristics such as air pollution levels in their hedonic wage equations (see Smith 1983, and Smith and Gilbert 1984, 1985). More recently, Rehdanz and Maddison (2004) examined the tradeoff between wages and climate amenities. The modeling of determinants of interurban wage differentials is discussed in this section.

Those cities that are more desirable places to live and work in will attract workers from less desirable cities and regions. The in-migration of labor will exert downward pressure on wage rates in the desirable city. An equilibrium occurs when wages have fallen to the point where the marginal worker is indifferent between moving to this city and staying in his or her next-best alternative location. The difference in wages between this city and the next-best alternative is a compensating wage differential.

The possibility that such compensating differentials could be used as estimates of the monetary value of amenity differences spawned a series of empirical studies during the 1970s (Nordhaus and Tobin 1972; Hoch and Drake 1974; Hoch 1977; Meyer and Leone 1977). In these studies, measures of average wage rates, earnings, or income were regressed on variables reflecting such things as climate (temperature, humidity, frequency of rain, for example), environmental quality (say, measures of air pollution, water pollution, and access to recreational resources such as beaches), cultural amenities (for example, number of museums, newspapers, and universities), the disamenities of urban life (for example, crime rates, population density), and city size itself. Some of these studies controlled for differences in the occupational structure of urban labor forces by estimating separate equations for individual occupations. Others used more aggregated measures such as average earnings. In none of these early studies was the estimating equation derived from a formal model of individual choice, interurban migration, or supplies and demands of labor in a system of interconnected urban labor markets.

Rosen (1979) was apparently the first to attempt to provide a formal model for deriving the structural equations relating wages to urban amenities and disamenities. Such a formal model is necessary to provide a welfare-theoretic interpretation of regressions that explain wage differences across cities. Rosen pointed out that there are really two hedonic markets in which individuals are making choices—one for labor, and one for land or housing. A decision to work in one city is also a decision to purchase housing services in that city. As individuals are drawn toward the more desirable cities and push wages down in those cities, they are also pushing out the demand for land and housing and increasing their prices. Not only are there compensating wage differentials, then, but there are also compensating land rent and housing price differentials across cities. The labor market model must also provide a coherent explanation for why firms in some cities are able to pay higher wages and still compete in markets for goods traded among cities. A formal model of both sets of markets is required in order to draw inferences about amenity values and willingness to pay from data on wage differentials and housing prices.

A number of such formal models have now been presented in the literature. See, for example, Rosen (1979), Cropper and Arriaga-Salinas (1980), Cropper (1981), Henderson (1982), Hoehn, Berger, and Blomquist (1987), Blomquist, Berger, and Hoehn (1988), and Roback (1982, 1988). Bartik and Smith (1987) reviewed most of these models, discussed similarities and differences in their theoretical structures, and described some of the major empirical results (also see Palmquist 1991). These models share the common feature of explicitly dealing with the interaction between markets for labor and for land across urban areas. However, they differ in the way in which they model certain specific features of the determination of the interurban equilibrium. For example, some models treat the size of each city as fixed, while others allow the city boundary to expand to accommodate higher populations. Some models treat the costs of firms as

exogenous, while other models allow costs to be affected by environmental amenities and the population of the city itself. Also, some models treat amenity levels as uniform within each city, while others allow for variation in amenity levels within, as well as among, urban areas.

Apparently models that allow all of the relevant variables (such as city size, firm location, and cost) to be determined endogenously are analytically intractable. As a consequence, the results of these models depend upon the specific features of the model. None of these models fully captures all of the complexity of the general interurban equilibrium. The models produce what Bartik and Smith have termed "partial descriptions" of the more complex reality (1987, 1232).

This section does not try to explain all of the models and their different results. Instead, a simple model of individual choice is first introduced to show how the interaction between wages and rent or housing price affects our ability to interpret regression coefficients as welfare measures. One version is then sketched of a simple model of the equilibrium of the land and labor markets that permits the estimation of a marginal amenity value. The discussion ends by examining in a more qualitative manner the implications of constructing richer models with more endogenous variables and more interactions.

The Welfare Measure with Two Hedonic Markets

Most of the early studies of interurban wage differences were based either implicitly or explicitly on the assumption that wage differences measure the values of amenity differences. For example, Nordhaus and Tobin (1972) and Meyer and Leone (1977) used their results to make adjustments to the national income accounts for nonpriced positive and negative welfare effects of urbanization. Clark and Kahn (1989) interpreted their coefficient on an interurban wage equation as a marginal implicit price and used it in a second-stage estimation of a marginal bid function for an amenity. However, a very simple model of individual choice of a city to live and work in can be used to show that the assumption that wage differences measure amenity values is not valid in general. This is because the wage differential is also affected by an interaction between the markets for labor and land.

To see this, consider the simplest case of an individual who derives utility from the consumption of a numeraire good, z, the quantity of housing consumed, h, and the level of an urban-specific amenity, q. By selecting a city to live in, the individual determines her annual wage, p_w, and the level of the urban amenity. Assume that q_j is the same at all locations within city j. Let z be normalized with a price of one that is constant across cities, since z is a nationally traded good. The individual selects q_j along with z and h so as to maximize

$$u = u\left(z, h, q_j\right) + \lambda\left[p_w\left(q_j\right) - p_h\left(q_j\right)\cdot h - z\right] . \tag{11.6}$$

Note that both wages and the price of housing vary across cities according to the level of the amenity in each city. If one city is relatively more desirable as a place to live and work, other things being equal, workers will move to that city;

the increase in the supply of labor will push down wages in that city; and, since people must live in the same city in which they work, the increased demand for housing will bid up its price. Of course, a complete model must aggregate across individuals as well as specify the supply sides of these markets in order to solve for the wage and housing price equations (more on this below).

The first-order conditions for the individual choice problem are:

$$\frac{\partial u}{\partial z} = \lambda \tag{11.7}$$

$$\frac{\partial u}{\partial h} = \lambda \cdot p_h \tag{11.8}$$

$$\frac{\partial u}{\partial q} = \lambda \left(h \cdot \frac{\partial p_h}{\partial q} - \frac{\partial p_w}{\partial q} \right) . \tag{11.9}$$

Using the first-order conditions, the individual's optimum choice of z and q can be found. Graphically, the optimum combination is at the tangency between the individual's indifference curve for z and q and the budget line giving the terms at which z and q can be exchanged, both holding the quantity of housing h constant (see Figure 11.1). The marginal rate of substitution (MRS) between z and q is

$$MRS_{zq} = \frac{\partial u / \partial q}{\partial u / \partial z} = -\left. \frac{dz}{dq} \right|_{u=u^0} . \tag{11.10}$$

The slope of the budget line is found by taking the total differential of the budget:

$$\frac{dz}{dq} = \frac{\partial p_w}{\partial q} - h \cdot \frac{\partial p_h}{\partial q} . \tag{11.11}$$

The marginal rate of substitution can also be interpreted as the marginal willingness to pay for q or w_q . In equilibrium, w_q can be inferred from knowledge of the slope of the budget line:

$$w_q = h \cdot \frac{\partial p_h}{\partial q} - \frac{\partial p_w}{\partial q} . \tag{11.12}$$

This marginal willingness to pay has two components. The first component is the willingness to spend more on housing in a higher quality city with a higher price for housing; and the second is the willingness to accept a lower wage in the higher quality city.

This analysis makes it clear that knowledge of the hedonic wage gradient across cities is not sufficient for inferring marginal amenity values. The hedonic wage gradient is also shown in Figure 11.1. The budget line is more steeply sloped than the wage gradient because it includes the negative term for the effect of q on the price of housing. This means that the marginal implicit price of q in the labor

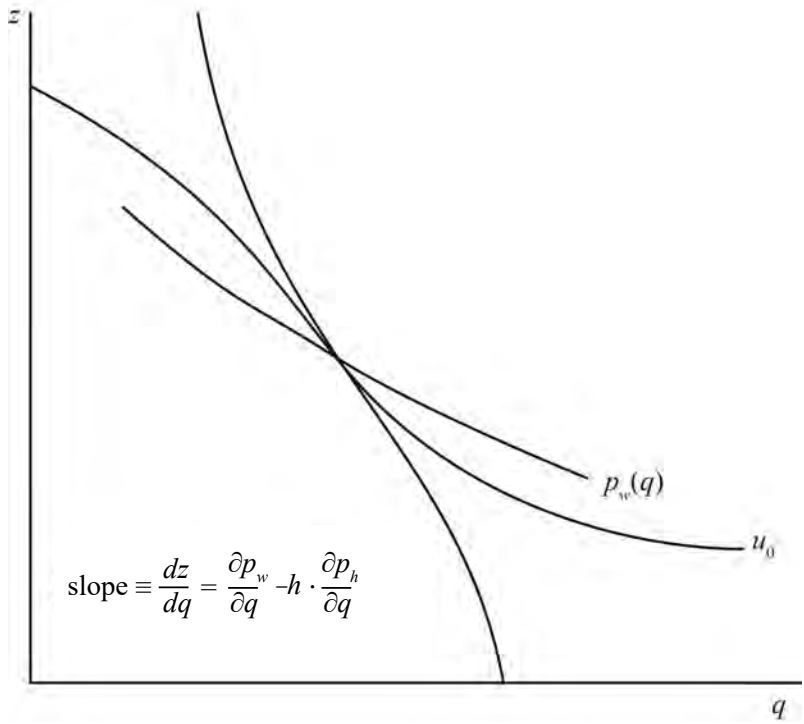

Figure 11.1 The individual's choice of an urban amenity

market is an underestimate of the individual's marginal valuation of q when q is in fact purchased simultaneously through two hedonic markets.

A General Model of Interurban Equilibrium

In order to estimate a welfare measure such as equation (11.12), it is necessary to develop a formal model of the interurban equilibrium so that the hedonic wage and housing price equations can be properly specified. The model presented here does not capture all of the complexities of the interurban location equilibrium problem, but it does show the general features of most of the models that have been presented in the literature. This model is patterned most closely after those of Hoehn, Berger, and Blomquist (1987) and Roback (1982), and incorporates the following features:

- It takes account of the variation in housing prices within cities caused by the spatial character of a city. Specifically, since land rents vary inversely with commuting costs, distance from the city center is an argument in the rent function.
- It treats the size of a city as endogenous. As cities that are more desirable draw workers to them and the price of housing is bid up, it becomes

profitable to convert some of the surrounding agricultural land to housing. This helps to dampen the upward pressure on housing prices.

- It treats the number of firms in each city as endogenous, and firms' costs are made to depend on the level of one or more amenities and disamenities and, perhaps, on city size, in order to reflect agglomeration effects.

This model assumes that all individuals have identical preferences and identical endowments of wealth, which in turn are assumed to be zero for simplicity. From the initial assumptions, it follows that an equilibrium must be characterized by equal levels of utility for all individuals. Each individual's preferences are represented by the utility function

$$u = u(z, a, q), \tag{11.13}$$

where z is the numeraire good traded in a "world" market at a price of one, a is the quantity of land (acres) occupied by the individual for housing, and q is the amenity that varies across cities but is uniform within each city. Let there be $\mathcal{J}$ cities in this economy. Each city offers a wage of p_{w_j} and an amenity level q_j, $j = 1, \ldots, \mathcal{J}$.

Each city also has the standard circular form in which all jobs are located at the city center. Individuals choose a residential location at distance d, and incur commuting costs of $t \cdot d$ per period, where t is the unit per period transportation cost. In order to compensate for higher commuting costs, the price of land p_a will decline with increasing distance from the city center. The boundary of the city will be at distance d^* where in equilibrium $p_a(d^*)$ is equal to p_{a^*}, the rental price of undeveloped or agricultural land.

Each individual chooses a wage and amenity package by selecting a city in which to live and work, a location for his or her residence, a quantity of residential land, and a quantity of the numeraire. Formally, each individual's choice problem is to maximize

$$u = u(z, a, q) + \lambda\left[p_w(q_j) - t \cdot d - p_a(d) \cdot a - z\right]. \tag{11.14}$$

The solution to this problem gives the indirect utility function:

$$v = v\left[p_w(q_j) - t \cdot d, p_{a_j}(d), q_j\right]. \tag{11.15}$$

An equilibrium is achieved when wages and land prices have adjusted so that all individuals are indifferent as to the choice of a city and their location within the city chosen. From equation (11.15), this equilibrium must satisfy

$$v\left[p_w(q_j) - t \cdot d, p_{a_j}(d), q_j\right] = v^* \equiv v\left[p_w(q_j) - t \cdot d_j^*, p_{a^*}, q_j\right]. \tag{11.16}$$

The left-hand side of equation (11.16) can be solved for each city's land price schedule, $p_{a_j}(d, q_j, t, v^*)$. The right-hand side can be solved for the city's physical size, $d_j^*(t, p_{a^*}, q_j, v^*)$. The application of Roy's identity to equation (11.16) gives the individual's demand function for land:

$$a = -\frac{(\partial v/\partial p_a)}{(\partial v/\partial p_w)} = a\left(p_w - t \cdot d, p_{a_j}, q_j\right) , \tag{11.17}$$

and since the supply of land at any distance is given by $2\pi d$, the total number of people that any city can accommodate is

$$\mathcal{N} = \int_0^{d^*} \frac{2\pi d}{a(\cdot)} dd \ . \tag{11.18}$$

The model must be closed by specifying the production sector of the economy, which also determines the demand for labor. Assume that the industry producing good z is competitive and subject to constant returns to scale. Then price, which has been normalized at one, equals average cost as well as marginal cost. If firms purchase materials and capital at fixed world prices, then some economic characteristic of each city must be an argument in the firms' cost functions; otherwise, all firms would be forced by competition to pay the same wage. Hoehn, Berger, and Blomquist (1987) assumed that q and total population are arguments in the cost function, the latter through an agglomeration effect. Alternatively, Roback (1982) assumed that firms use land in production, so that p_a is an argument in the cost function, along with q.

This system of equations can be solved to determine p_w and $\mathcal{N}$ for each city, and p_a and d for each individual within the city. These variables depend upon the exogenous q_j , t, and p_{a^*} , which are the variables to be included in hedonic wage and land price equations. Because of all of the interdependencies, the comparative statics of this type of model can be fairly complex. Hoehn, Berger, and Blomquist (1987) analyzed the comparative statics of their model with respect to changes in q. The results depend on how q and $\mathcal{N}$ affect the unit cost function.

To derive the marginal willingness to pay for q, return to equation (11.15) , take its total differential, and set it equal to zero, obtaining

$$dv = \frac{\partial v}{\partial p_w} dp_w + \frac{\partial v}{\partial p_a} dp_a + \frac{\partial v}{\partial q} dq = 0 \ , \tag{11.19}$$

and

$$w_q \equiv \frac{(\partial v/\partial q)}{(\partial v/\partial p_w)} = -\frac{(\partial v/\partial p_a)}{(\partial v/\partial p_w)} \cdot \frac{dp_a}{dq} - \frac{dp_w}{dq} \ . \tag{11.20}$$

Again, employing Roy's identity, this becomes

$$w_q = a \cdot \frac{dp_a}{dq} - \frac{dp_w}{dq} \ . \tag{11.21}$$

This expression is similar to the one derived above from the model of individual choice; see equation (11.12). Here, each individual's marginal willingness to pay for q consists of two components, the change in the expenditure on land associated with an increase in q, and the willingness to accept a lower wage rate for an improvement in q. The comparative static analysis of Hoehn, Berger,

and Blomquist (1987) showed that it is possible for dp_a/dq to be negative or for dp_w/dq to be positive. It should be noticed that although the level of q is uniform within each city, individuals can have different marginal willingness to pay for q, depending upon the other elements in their consumption bundle, in particular, the quantity of land.

Other models of the interurban equilibrium have been developed to examine other forms of interaction. The additional features that can be captured in these models include the following:

- Intra-urban variation in amenity levels. Some amenities and disamenities that vary on average across cities also vary systematically within each city. For example, crime rates and air pollution tend to be higher in the center of each city than at the city boundary. This variation in amenity levels within cities will affect the spatial pattern of land rents and housing prices within each city (see Cropper 1981).
- The existence of both traded and nontraded goods. With zero transportation costs, traded goods sell at the same prices in all cities. However, the prices of nontraded goods can vary across cities, leading to differences in the cost of living. The explicit treatment of nontraded goods is necessary to determine how cost of living differences should be treated in the specification of the hedonic wage function. Cropper (1981) has shown that housing prices should not be included in the index of prices used as an argument in hedonic wage functions.
- Variation in the cost of supplying housing across cities. Although the simpler urban models are formulated in terms of land rent, housing prices are easier to observe for purposes of hedonic price estimation. Since the concept of interurban equilibrium is inherently long run, it is necessary to model the supply side of the housing market. The cost of producing housing will depend on, among other things, the price of land (which varies within and across cities) and the wage rate (which also varies across cities). For examples, see Roback (1982) and Hoehn, Berger, and Blomquist (1987).

Alternative Sorting Models

As noted in the previous chapter, a variety of sorting models have emerged in recent years as alternatives to traditional hedonic pricing models of quality-differentiated products. Not surprisingly, this line of research has made its way into models of migration. This section closes with a brief look at several examples of these efforts.

The first example is Bayer, Keohane, and Timmins (2009). The authors developed a discrete choice model of migration, with the utility from choosing a specific location depending upon local wage scales, housing conditions, and an environmental amenity of interest (in their application it is air quality). A central contribution of the paper is that the authors relaxed the assumption, underlying

conventional hedonic techniques, that moving is costless. Instead, households are assumed to incur a psychological cost from moving outside the area in which they are born, a cost that is estimated as a part of the discrete choice model. The argument is that these costs create a barrier to migration and that ignoring them will yield biased estimates of the marginal willingness to pay (MWTP) for changes to the environmental amenity. Indeed, in their application they found that the MWTP for air quality improvements (measured in terms of ambient concentration of particulate matter) is three times larger when migration costs are accounted for than when they are ignored.

Timmins (2007) allowed for migration costs as well, but the focus of the paper is somewhat different. Whereas Bayer, Keohane, and Timmins (2009) were interested in estimating the MWTP for air quality improvements, Timmins' (2007) goal was to model the impact of nonmarginal environmental changes. To do so, he drew on the horizontal sorting models of Bayer, McMillan, and Reuben (2004) and Bayer and Timmins (2007). In his application, measuring the cost of climate change in Brazil, Timmins found that migration costs play an important part in determining both the overall impact of climate change and how the impacts are distributed in the population.

Finally, Kuminoff (2012) extended the vertical sorting model of Epple and Sieg (1999) to a dual-market setting, and developed a structural model of a household's interrelated choices of housing location and job. The framework recognized that the individual's ultimate set of choices may involve compromises between the two decisions—for example, accepting a lower-paying job (or one with few workplace amenities) in exchange for a location with higher local amenities (such as cleaner air or more recreational opportunities). In his application, modeling the MWTP for improved air quality in Northern California, Kuminoff found that accounting for the interaction between job and housing choices, rather than focusing on housing choice alone, led to an increase in the MWTP by up to 110 percent.

Both the vertical and horizontal sorting models that have emerged in recent years are promising tools for understanding the partial and general equilibrium impacts of changing environmental amenities and integrating the interrelated housing and labor market choices. The inclusion of moving costs allowed for in the horizontal sorting models seems particularly important. At the same time, these sorting models are relatively new and often quite complex, both in terms of their structure and in terms of the econometrics required for estimation. Additional research remains to be done in order to understand the implications of the assumptions required to close these models, particularly in terms of the specific functional forms used to represent utility and in the assumed distributions of both underlying preferences and the unobserved factors thought to influence them.

Summary

This chapter has examined two different types of application of hedonic price theory to labor markets. In one application, the focus is on the intrinsic characteristics of jobs. These applications seek to infer willingness to pay for changes in characteristics such as risk of accidental death or injury on the job. In these applications, it is usually assumed implicitly that the location of the job is unimportant. An exception is the work of Smith and Gilbert (1984, 1985) described above. In some wage–risk studies, regional dummy variables may be included to control for differences in wage levels across broad regions of the country, but the location of the job and the amenities or disamenities that go with living near the job are not explicitly considered. Recent advances in both the available data and in econometric techniques have helped to alleviate concerns regarding measurement errors and omitted variables bias in estimating the wage–risk premium. However, these advances do not eliminate these concerns, nor do they resolve the problem that these are best suited for working-age individuals, and not the elderly and infants most substantively impacted by environmental disamenities.

In the other application, the focus is on the choice of a city in which to live and work and, in some cases, where in the city to live. Jobs are typically treated as homogeneous, except for their location. These applications seek to infer willingness to pay for urban amenities. In some of these applications, job characteristics are controlled to some extent by examining interurban differences within broad occupational categories. A key feature of applications of this kind is the need to model explicitly the interaction between the two hedonic markets— those for labor and for land or housing. Introducing heterogeneous jobs into this interurban labor market does not require adding another hedonic market into the analysis. However, it does require a more detailed treatment of the determinants of wage differences within a multi-city, multi-characteristic market for differentiated labor. This is an important subject for further theoretical and empirical research.

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Chapter 12

Stated Preference Methods for Valuation

Chapters 9 through 11 described methods for inferring individuals' values for environmental amenities from their observed choices of related market goods. These approaches are broadly categorized as "revealed preference" (RP) techniques because, it is argued, a person's actions in the marketplace reveal information about his core preferences, including his preferences for public goods.[1] However, discussions in Chapter 4 also showed that there are many circumstances under which value measures cannot be derived from market transactions. This is the case, for example, when individuals are thought to place value in an environmental amenity that they do not directly use (that is, so-called "passive-use" values). Because such values are not tied to the use of a related market commodity, they leave no footprint in the marketplace from which their magnitude can be inferred. Even in the case of use values, revealed preference techniques may be hampered by a lack of sufficient independent variation in the amenity of interest from which to infer its impact on behavior and preferences. In the context of recreation demand models, for example, it may be difficult to discern the marginal value associated with a specific site amenity (e.g., a reduction in lake algae) when there is limited variation in the amenity across existing sites, or a high degree of correlation with other site attributes (e.g., fish catch rates).

This chapter examines a broad class of alternative techniques that can be used to assess the welfare impact of changing environmental conditions, either in lieu of the RP approaches described thus far, or as a complement to them. These methods have in common their source of data for analysis: individuals' responses to questions about hypothetical situations such as "Would you pay $\$X$ for …?", "What is the most that you would be willing to pay for …?", "What would you do if …?", or "Which of the following alternatives do you prefer …?" Since values are inferred from stated responses to such questions, these methods are now commonly referred to as stated preference (SP) methods.

Although not all authors use the same terminology, the term stated preference methods has come to refer to any survey-based study in which respondents are

1 The tight linkage between behavior and core preferences has come into question in recent years by both behavioral economists and psychologists. See, for example, Sugden (2004), Beshears et al. (2008), and Vatn (2004).

asked questions that are designed to reveal information about their preferences or values. The term encompasses three broad types of questions. The first type involves questions that elicit monetary values for a specified commodity or environmental change. These are usually referred to as *contingent valuation* (CV) methods, as the survey responses are assumed to be "contingent" upon the presented hypothetical scenario. A wide range of elicitation formats fall within the CV framework. A relatively straightforward CV question simply asks people what value they place on a specified change in an environmental amenity or the maximum amount they would be willing to pay to have it occur. Also referred to as an "open-ended" elicitation format, the responses to these questions, if truthful, are direct expressions of value, and would be interpreted as measures of compensating surplus (*CS*). Another major type of CV question asks for a yes or no answer to the question, "Would you be willing to pay $\$X$ …?" Each individual's response reveals only an upper bound (for a no) or a lower bound (for a yes) on the relevant welfare measure. Questions of this sort are termed *binary discrete choice* (or "closed-ended") questions. By comparing the responses to such questions across individuals facing different values of X, estimated willingness to pay or indirect utility functions can be obtained.

The second and third major types of SP methods do not reveal monetary measures directly. Rather, they require some form of analytical model to derive welfare measures from responses to questions. In the second approach to questioning, respondents are asked how they would change the level of some activity in response to a change in an environmental amenity. If the activity can be interpreted in the context of some behavioral model, such as an averting behavior model or a recreation demand model, the appropriate revealed preference valuation method can be used to obtain a measure of willingness to pay. These are known as contingent behavior (CB) questions.

In the third type of SP question, individuals are given a set of hypothetical alternatives, each depicting a different bundle of environmental attributes. Respondents are asked to choose the most preferred alternative, to rank the alternatives in order of preference, or to rate them on some scale. Responses to these questions can then be analyzed to determine, in effect, the marginal rates of substitution between any pair of attributes that differentiate the alternatives. If one of the characteristics has a monetary price, then it is possible to compute the respondent's willingness to pay for the attribute on the basis of the responses. Studies based on this form of question are usually referred to as choice experiments or sometimes attribute-based methods (Holmes and Adamowicz 2003) and have their roots in the marketing literature (Louviere, Hensher, and Swait 2000).

The use of the SP methods for environmental valuation has been controversial. This controversy became especially heated after the Exxon Valdez oil spill in March 1989 when it became known that a major component of the legal claims for damages was likely to be based on CV estimates of lost passive use values. In response to the Exxon litigation, National Oceanic and Atmospheric Administration (NOAA) convened a Blue Ribbon Panel, chaired by two Nobel Prize Economists, Kenneth Arrow and Robert Solow, to review CV in the context of assessing damages to

natural resources in support of litigation and provide guidelines for best practice. The Panel's report (Arrow et al. 1993), while not quelling the debate over the validity of SP techniques, had the salutary effect of stimulating a substantial body of new research on both SP practice and on the credibility or validity of SP-based estimates of value. A good overview of the issues raised at the time is contained in the three essays published as a symposium in the *Journal of Economic Perspectives* (Portney 1994; Hanemann 1994; Diamond and Hausman 1994). To get a sense of the intensity of the controversy, see the collection of essays sponsored by Exxon Corporation—especially the transcripts of the discussions by audience members and authors at the public presentation of these essays (Hausman 1993). A reprisal of this debate was recently published in the *Journal of Economic Perspectives* (Kling, Phaneuf, and Zhao 2012; Carson 2012; Hausman 2012). Interestingly, despite nearly two decades of research, some of the positions held back in 1994 have hardly changed.

The remainder of this chapter is structured as follows. The next section describes in more detail the major types of SP question formats and, where appropriate, explains how responses can be analyzed to obtain measures of welfare for changes in the environmental amenity. A major question regarding all stated preference methods concerns the validity and reliability of the data—that is, whether the hypothetical nature of the questions asked inevitably leads to some kind of bias or results in so much "noise" that the data are not useful for drawing inferences. The second section of this chapter considers approaches to assessing the validity of measures of value obtained with stated preference methods, where validity refers to the degree of correspondence between the measure obtained and the theoretical concept of value. The chapter concludes with an assessment of the current "state-of-the-art" of the stated preference methods.

The treatment in this chapter of the issues surrounding stated preference methods is selective. Relatively little attention is given to a number of design and implementation problems, such as the form of contact with respondents (mail, telephone, personal interview), survey testing, and sampling design. While these issues are essential to successful stated preference research, they are treated extensively elsewhere in the literature. Mitchell and Carson's (1989) pioneering treatise is still a primary reference for CV studies, especially for design and implementation questions. Two more recent works that focus on best practice and empirical estimation for CV, and nonmarket valuation more generically, are Bateman and Willis (1999) and Champ, Boyle, and Brown (2003), respectively. Louviere, Henscher, and Swait (2000) provided a primer for choice experiments. Other important references are: Bjornstad and Kahn (1996) for a review of theoretical and empirical issues that includes assessments by both proponents and critics of stated preference methods; Haab and McConnell (2002) for a discussion of econometric issues in non-market valuation; Bateman et al. (2002) for a user's guide to stated preference methods; Carson and Hanemann (2005) for an overview of contingent valuation; Carson (2011) for a comprehensive overview of the history of contingent valuation and an extensive bibliography of papers in the area; and Kopp, Pommerehne and Schwarz (1997).

Stated Preference Approaches to Valuation

Contingent Valuation

Contingent Valuation (CV) is perhaps the best known of the stated preference techniques employed in valuing environmental amenities, with applications dating back to Davis (1963). At their heart, these techniques seek Hicksian measures of the welfare impact of hypothetical changes in environmental conditions, or at least bounds on such measures. The specifics of the survey design vary, but as Carson and Hanemann (2005, 825) noted, based on current best practices, the survey instruments typically contain the following elements: "(1) an introductory section identifying the sponsor and general topic, (2) a section asking questions concerning prior knowledge about the good and attitudes toward it, (3) the presentation of the CV scenario including what the project was designed to accomplish, how it would be implemented and paid for, and what will happen under the current status quo situation if the project were not implemented, (4) question(s) asking for information about the respondent's WTP/WTA for the good, (5) debriefing questions to help ascertain how well respondents understood the scenario, and (6) demographic questions." In what follows, variations in the elicitation format (that is, item 4) are described.

Open-Ended Questions

Data obtained from open-ended value questions, taken at face value, are the simplest to interpret. Each respondent is typically asked to state his or her maximum willingness to pay (WTP) for an environmental improvement (compensating surplus) or to avoid a loss (equivalent surplus).[2] There are several ways to elicit this number. Many of the earliest studies used an iterative technique, which has come to be called the bidding game. In the bidding game, individuals are first asked whether they would be willing to pay $\$X$. If the individual answers yes, the question is repeated with a higher "price." The procedure is repeated until the individual answers no. The highest price with a yes response is interpreted as the maximum willingness to pay. If the original response is no, the iteration proceeds downward until a yes response is received. Bidding games, however, fell out of favor in the literature when researchers found that the starting point used in the bidding process influenced the individual's purported maximum WTP. See Whitehead (2002) for a recent example. The specific source of this so-called "starting point bias" is unknown, though possible explanations include respondent fatigue (with the individual saying "yes" or "no" to simply stop the line of questioning) or the respondent's interpreting the starting point as conveying information about either the cost of provision or a "reasonable" estimate of the good's value.

2 Minimum willingness to accept (WTA), as compensation for an environmental degradation or in lieu of an environmental improvement, is sometimes also asked for, but is less common in the literature for reasons discussed later in this chapter.

An alternative elicitation technique is simply to ask open-ended questions of the form: "How much would you be willing to pay?" One concern with this approach is that it confronts people with an unfamiliar problem. In most real market settings, individuals are faced with choices among sets of goods with listed prices. Seldom are they asked to offer a one-shot bid that may be either accepted or rejected by the seller.[3] People appear to have difficulty dealing with the open-ended form of direct question as evidenced by high rates of nonresponse to the valuation question and/or high proportions of implausibly high or low stated values.

A variation on the open-ended approach is to show respondents a card with a range of alternative payment values on it and ask them either to pick a number from the card or to state their own value if that is not to be found on the card. Some authors have also experimented with payment cards that indicate the amounts that typical respondents are paying in the form of taxes for such public programs as police protection, health care, and national defense. Rowe, Schulze, and Breffle (1996) conducted a study to see if changes in the ranges covered by payment cards affected the distribution of expressed maximum WTPs. They found no effect as long as the payment card did not truncate the upper range of values.

Regardless of the specifics of the elicitation process, the key advantage of the open-ended format is that it provides a specific welfare number (W_i) for each survey respondent. If these numbers accurately reflect the individual's preferences, subsequent analysis of the data is relatively straightforward. An estimate of the total value of the welfare change for the population from which the sample is drawn can be obtained by calculating the sample mean and multiplying by the total population. Alternatively, the responses can be regressed on income (M_i) and other socioeconomic characteristics ($\boldsymbol{S}_i$) to obtain a bid function for the proposed policy scenario:

$$B_i = B(M_i, \boldsymbol{S}_i). \tag{12.1}$$

If the survey design includes variation in the size or composition of the environmental changes ($\Delta \boldsymbol{Q}_i$) across the sample, then the bid function

$$B_i = B(\Delta \boldsymbol{Q}_i, M_i, \boldsymbol{S}_i) \tag{12.2}$$

can be estimated and used to calculate values for alternative scenarios of environmental or resource change.

Two econometric issues typically arise in estimating either type of bid function. First, the parameter estimates can be sensitive to outliers. Many surveys obtain at least a few bids that are so large relative to the sample mean, or individual income levels as to be of questionable validity. Varieties of procedures have been suggested in the literature to deal with these observations, including trimming or censoring

3 This statement, while generally true in much of the industrialized world, is perhaps less of a concern in countries where haggling in local market places is more commonplace. See Whittington (2002) for a discussion of the use of contingent valuation methods in developing countries.

extreme values (e.g., Mitchell and Carson 1989, 226–227). As these procedures involve judgment calls regarding what constitutes an "extreme value," sensitivity analysis should be reported. Second, open-ended questions will often yield a large number of zero values. Censored regression models, such as Tobit, or double-hurdle models, are typically used to accommodate the mass of observations at zero. See Haab and McConnell (2002) for additional details.

The primary concern regarding the open-ended elicitation format is whether the reported welfare measure W_i accurately reflects the respondent's true preferences. There are two related issues here. First, if the survey respondent does not believe that their answers will have any impact on them, either because the survey will not ultimately influence policy or because they believe they will not have to pay for the policy change, then the survey itself is *inconsequential* and they have no incentive to respond truthfully.[4] Second, even if the respondent believes that the survey is *consequential*, Carson and Groves (2007) argued that the open-ended elicitation format is *incentive incompatible*; that is, agents have an incentive to exaggerate their reported WTP. The problem arises in part because it is not credible that actual payment for the program will be tailored to the individual's reported W_i. Consider the simple case in which the respondent believes her portion of the program's cost is fixed, say at C_i. In this setting, she has an incentive to report an arbitrarily high WTP if $W_i > C_i$ (thereby encouraging the policy's adoption) and report a zero WTP if $W_i < C_i$. The direction of the bias will depend upon individual beliefs regarding C_i. While conventional wisdom early on was that open-ended questions would lead to overstated WTP estimates, the reverse appears to be the case.

Another concern with open-ended questions is that the sample could contain invalid zero responses, so-called *protest zeros*. Protest zeros occur when respondents reject some aspect of the constructed market scenario by reporting a zero value even though they place a positive value on the amenity or resource being valued. Simply trimming a portion of the zeros is not a solution to the problem, since it is not known a priori what proportion of the zeros to trim. Rather, some means must be found to identify protest zeros for deletion before applying the procedure for deleting outliers. One approach commonly used in the literature is to follow up the valuation question with a question regarding the individual's motive for his response. For example, respondents could be asked:

Which statement best expresses your reason for your response?

- I can't afford to pay for the good.
- The good is not important to me.
- I don't think that I should have to pay for the good.
- The proposed program is unrealistic.

4 The notion of consequentiality is discussed in greater detail in Carson and Groves (2007). This issue is discussed again later in the chapter.

Responses of those choosing either the third or fourth statement would be classified as protest zeros and deleted from the sample, while responses of those choosing the first and second would be considered valid responses. Unfortunately, the deletion of protest zeros can also lead to item nonresponse bias if those who protest are systematically different in some respect from those who give proper responses. At a minimum, it would be desirable to examine whether the individuals in the protest group differ in observable ways from the general population.

Binary Discrete Choice Questions

Perhaps the most commonly used elicitation format in contingent valuation is the single-shot binary discrete choice question, also known as the dichotomous choice format. The questions are typically couched in the form of a referendum. After presenting the survey participant with the proposed environmental changes and the cost (also referred to as the "bid amount") that they would bear if the changes were implemented, the individual is asked if they would vote in favor of the referendum. If a respondent answers yes, that person has indicated a willingness to pay that is greater than or equal to the specified cost. If the response is no, then that sum of money can be taken as an upper bound on true willingness to pay. Respondents are assigned randomly to different subsamples, with each subsample being asked to respond to a different bid amount. It is then possible to test the hypothesis that the proportion of yes responses decreases with an increase in the price of the environmental good. These data can then be analyzed with a model of discrete choice to obtain estimates of indirect utility functions or bid functions.

The single-shot binary choice format has at least three advantages relative to open-ended formats. First, it places people in a relatively familiar social context. Many private market transactions involve goods offered on a take-it-or-leave-it basis in which the individual decides whether or not to purchase the good at the offered price. Furthermore, if the payment vehicle is a tax, the discrete choice question simulates a true referendum of the sort found everywhere from small New England town meetings to statewide votes on highway bond issues. The second advantage is that, since only a yes or no answer is required, the discrete choice question format poses a relatively simple decision problem for individuals. This may result in lower levels of item nonresponse and fewer refusals to participate in the survey. Third, in at least some circumstances, it is incentive-compatible; that is, respondents' best strategy is to be truthful in answering the question. This point is returned to below.

The primary disadvantage of the binary choice format is that it yields relatively little information from each survey respondent. Specifically, one learns only whether the individual's WTP for the proposed program lies above or below the bid value they are presented with. Consequently, relatively large sample sizes are required in order to accurately characterize central tendencies and the distributional characteristics of WTP in the population. This is all the more the case if one is interested in recovering a bid function analogous to (12.2) above.

In order to convert data on yes or no responses to a discrete choice question into a monetary measure, it is necessary to employ some explicit utility-theoretic model of choice. The discrete choice model introduced in Chapter 4 is well suited to the task. As Cameron (1988) and Cameron and James (1987) have pointed out, the variation in prices across the sample makes it possible to explain individuals' choices in terms of a willingness-to-pay function rather than in terms of differences in indirect utility. McConnell (1990) has compared the Cameron model with the model based on utility functions outlined by Hanemann (1984) and by Sellar, Chavas, and Stoll (1986). He showed that the two models can be derived from the same underlying utility-theoretic framework. In a deterministic formulation, the two models yield the same predictions about behavior and choice. Since the stochastic forms of the models introduce random components in different ways, the two models are not in general equivalent, although there are special cases in which the two models are dual to each other. Moreover, as Cameron (1988) showed, the willingness-to-pay function model permits the straightforward calculation of marginal values for all of the arguments in the willingness-to-pay function, while this is not possible with the utility function model.

In what follows, a model of the individual's response to a single dichotomous choice question is first presented using the utility difference framework of Hanemann (1984), showing how welfare measures can be derived. This is followed by the parallel development based upon the willingness-to-pay function approach.

Consider an individual who must decide whether to answer yes or no to the following question: "Would you vote for a program to permanently increase environmental quality from q^0 to q^1 if the total cost to you was $\$T$?" Let the indirect utility function be $u(q, M, \boldsymbol{S})$, where $\boldsymbol{S}$ is a vector of individual characteristics and the vector of market prices, $\boldsymbol{P}$, is omitted since these prices are assumed to be constant. The individual responds yes if

$$u\left(q^1, M - T, \boldsymbol{S}\right) > u\left(q^0, M, \boldsymbol{S}\right) , \tag{12.3}$$

and no otherwise.

Let $v(\cdot)$ denote the analyst's representation of consumer preferences, sometimes also referred to as the "observable" component of utility. The probability of a yes response is then given by

$$\Pr(Y) = \Pr\left[v\left(q^1, M - T, \boldsymbol{S}\right) + \varepsilon^1 > v\left(q^0, M, \boldsymbol{S}\right) + \varepsilon^0\right] , \tag{12.4}$$

where the $\varepsilon^j \equiv u\left(q^j, M, \boldsymbol{S}\right) - v\left(q^j, M, \boldsymbol{S}\right) \left(j = 0,1\right)$ are the random, unobserved components of utility.[5] As discussed in Chapter 4, if the random terms are

5 The error terms in this random utility model capture a myriad of possible errors on the part of the analyst in characterizing preferences, including characteristics of the individual or the choice alternatives that are unobserved by the analyst, errors in the functional form used by the analyst to represent utility (i.e., v), or measurement errors on the part of the analyst. It is important to keep in mind, however, that these error terms are not capturing errors or uncertainty on the part of the respondent. We discuss this point again later in the chapter.

independently and identically distributed with a Type I Extreme Value distribution, then this probability can be expressed as

$$\Pr(\mathcal{Y}) = \frac{\exp(\Delta v)}{1+\exp(\Delta v)}, \tag{12.5}$$

where $\Delta v \equiv v^1 - v^0$. Also, reversing the sign on the probability difference gives the expression for the probability of rejecting the offer:

$$\Pr(\mathcal{N}) = \frac{1}{1+\exp(\Delta v)}. \tag{12.6}$$

The willingness to pay for q^1 (CS) is defined implicitly by

$$u\left(q^1, M - CS, \boldsymbol{S}\right) = u\left(q^0, M, \boldsymbol{S}\right), \tag{12.7}$$

or equivalently as

$$v\left(q^1, M - CS, \boldsymbol{S}\right) + \varepsilon^1 - \varepsilon^0 = v\left(q^0, M, \boldsymbol{S}\right). \tag{12.8}$$

In terms of the observable utility function, CS is thus a random variable, because of the term $\varepsilon^1 - \varepsilon^0$. The probability of accepting the offer is also, then, the probability that $CS \geq \mathcal{T}$, and the probability of rejecting the offer is the probability that $CS < \mathcal{T}$. This is a cumulative density function (c.d.f.), denoted here as $F(\mathcal{T})$, and shown in Figure 12.1, which plots the probability of a "no response," $\Pr(\mathcal{N})$, as a function of $\mathcal{T}$. As Hanemann (1984) has pointed out, the expected value of the random variable CS can be found from the c.d.f. as follows:

$$E[CS] = \int_0^\infty \left[1 - F(\mathcal{T})\right] d\mathcal{T}. \tag{12.9}$$

Graphically, in Figure 12.1, the expected value of CS is the shaded area above the c.d.f. and below $\Pr(\mathcal{N}) = 1$. For example, for any given $\mathcal{T}$, an individual with a lower willingness to pay for the change in q would have a higher probability of rejecting the offer, and the shaded area would be smaller. Alternatively, an offer of $q^2 > q^1$ at any $\mathcal{T}$ would decrease the probability of an individual rejecting the offer. So $F(\mathcal{T})$ would be shifted down and $E[CS]$ would be larger.

Specifying a functional form for the observable component of utility makes it possible to estimate the parameters of the utility difference in equations (12.5) and (12.6). For example (following Hanemann 1984), if $v = a + b \cdot \ln M + c \cdot \ln q$ (ignoring the $\boldsymbol{S}$ vector for simplicity), then

$$\Delta v = \left(a^1 - a^0\right) + \left[b \cdot \ln\left(1 - \frac{\mathcal{T}}{M}\right)\right] + \left[c \cdot \ln\left(\frac{q^1}{q^0}\right)\right]. \tag{12.10}$$

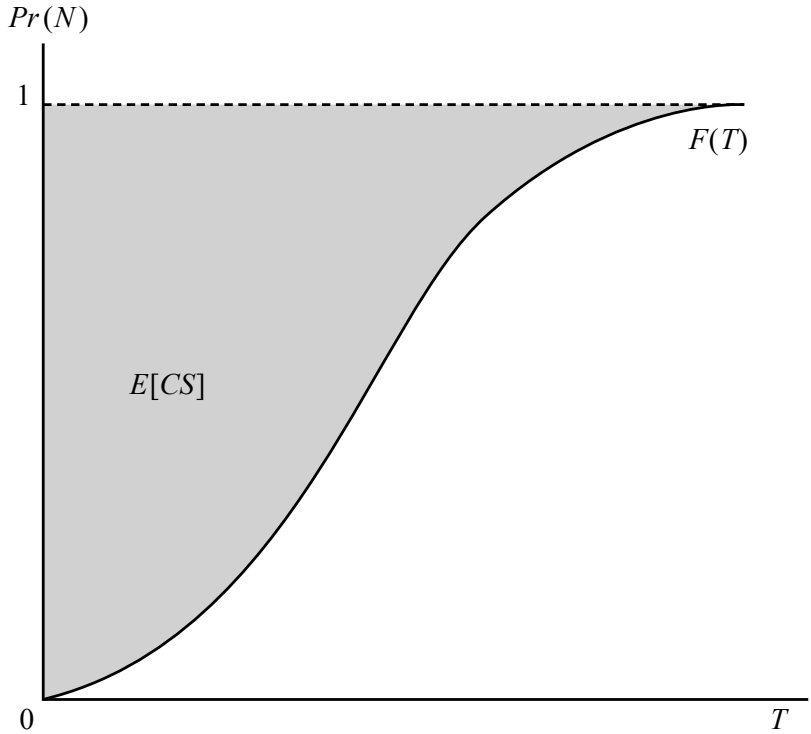

Figure 12.1 The cumulative density function for rejecting the offer of q^1 at T and the expected value of CS

Then with the parameters from equation (12.10), equations (12.5) or (12.6) and (12.9) can be used to calculate $E[CS]$ (for example, Hanemann 1984 and Seller, Chavas, and Stoll 1986).

As equation (12.19) shows, in principle $F(T)$ is integrated over the range to infinity. In order to avoid the implausibly high estimates of $E[CS]$ that can sometimes result, some researchers have truncated the integration of $F(T)$ at some "reasonable" finite value for T, such as some fraction of the individual's income.[6] Estimates of $E[CS]$ can be highly sensitive to the value of $T_{\max}$ chosen for truncation, at least in some cases (Bishop and Heberlein 1979; Hanemann 1984; Mitchell and Carson 1989, 103, 196–197). However, T should, at a minimum, be bounded above by income, recognizing the individual's budget constraint.

There is an alternative approach to dealing with the sensitivity of estimates of CS to outliers in the distribution. As Hanemann (1984) has suggested, the median willingness to pay for the sample can be found by setting $\Pr(Y)$ in equation (12.5) equal to 0.5 and solving for T. This gives the value for T that makes

6 Indeed, Δv in equation (12.10) becomes undefined for $T > M$.

the representative individual indifferent as to accepting or rejecting the offer. Generalizing to the sample, one would expect half the respondents to accept (and half to reject) the offer, other things (including income and $\boldsymbol{S}$) being held constant. This procedure is less sensitive to extreme values in the data.

The alternative model for analyzing discrete choice data is based on the bid or willingness-to-pay function. This function can be derived from the expenditure function as follows:

$$B\left(q^0,q^1,u^0,\boldsymbol{S}\right)=e\left(q^1,u^0,\boldsymbol{S}\right)-e\left(q^0,u^0,\boldsymbol{S}\right). \tag{12.11}$$

Cameron (1988) called this a valuation function, and McConnell (1990) called it a variation function, because it can be defined for either compensating or equivalent welfare measures. The formulation here gives the compensating surplus measure of value for an improvement. Other versions can easily be specified for losses and for *ES* measures. See Carson (1991, especially 143–144) for further discussion of this model.

The individual will respond yes if

$$B\left(q^0,q^1,u^0,\boldsymbol{S}\right)\geq T\ , \tag{12.12}$$

and no otherwise. The probability of accepting the offer of q^1 at T can be expressed in terms of the function

$$\Pr(Y)=\Pr\left[B^*\left(q^0,q^1,u^0,\boldsymbol{S}\right)+\eta\geq T\right]\ , \tag{12.13}$$

where B^* is the observable component of the bid function and η is the unobserved random component of willingness to pay. In other words,

$$B\left(q^0,q^1,u^0,\boldsymbol{S}\right)=B^*\left(q^0,q^1,u^0,\boldsymbol{S}\right)+\eta. \tag{12.14}$$

The next step is to make some assumptions about the distribution of the random component of the bid. Cameron and James (1987) assumed that η is normally distributed with zero mean and standard deviation σ. The result is a form of probit model in which

$$\Pr(Y)=\Phi\left[\frac{B^*\left(q^0,q^1,u^0,\boldsymbol{S}\right)-T}{\sigma}\right], \tag{12.15}$$

where $\Phi(\cdot)$ is the standard normal cumulative density. It is typical in the literature to assume that $B^*=\boldsymbol{X}\beta$, where $\boldsymbol{X}$ is a row vector including characteristics of the individual ($\boldsymbol{S}$), the choice alternative (e.g., $\Delta q=q^1-q^0$) and their interactions, while β is a conformable vector of parameters. The bid function in (12.14) can then be rewritten as

$$B=\boldsymbol{X}\beta+\sigma\eta_s\ , \tag{12.16}$$

where $\eta_s\sim\mathcal{N}(0,1)$ denotes a standard normal random variable. In this case, (12.15) reduces to

$$\Pr(\Upsilon) = \Phi\left[\boldsymbol{X}\left(\frac{\beta}{\sigma}\right) - \frac{1}{\sigma}T\right]. \tag{12.17}$$

Once the parameters of equation (12.17) have been estimated, calculations of total and marginal bids for individuals and in the aggregate are straightforward using (12.16). The precision with which the parameters will be estimated will depend on the sample size, the variability of preferences in the population (represented by σ), and the bid design (that is, the range and spacing of bids presented to different subsegments of the sample, see, for example, Alberini 1995).

Alternatively, if η is assumed to be distributed as in the logistic model, with scale parameter μ, then (following Cameron 1988)

$$\Pr(\Upsilon) = \Lambda\left[\frac{B^*\left(q^0, q^1, u^0, \boldsymbol{S}\right) - T}{\mu}\right], \tag{12.18}$$

where

$$\Lambda(x) \equiv \frac{\exp(x)}{1 + \exp(x)} \tag{12.19}$$

denotes the standardized logistic cumulative distribution function. Again, estimation of the parameters of the bid function makes it possible to calculate individual total and marginal willingness to pay as well as the probabilities of accepting an offer.

The logit and probit specifications above, while convenient, impose a considerable structure on the distribution of the bid function in the population, including symmetry around the observable portion of the bid function, B^*. To avoid these assumptions, nonparametric procedures exist that can be used to characterize the distribution of B (see, for example, Kriström 1990), and to estimate a lower bound on the mean WTP, such as using the Turnbull estimator (Turnbull 1976). For a more thorough discussion of these and other statistical issues associated with the analysis of discrete choice responses, see Hanemann and Kanninen (1999), Haab and McConnell (2002), or Carson and Hanemann (2005).

Just as in the case of open-ended valuation questions, respondents might express rejection of the discrete choice market scenario described in the survey instrument. *Protest questions*, similar to those described at the end of the discussion on the open-ended format, can be used to identify these individuals. Respondents who signal that their vote is a protest response can be deleted from the sample, though again it would be preferable to understand how these individuals differ from the general population so as to avoid nonresponse bias.

Alternative CV Elicitation Formats

The elicitation formats discussed thus far represent extremes of sort in the CV literature. The open-ended format provides a precise, though potentially biased, value in terms of the individual's WTP for the proposed policy scenario. The single-shot binary choice referendum format, at the other end of the spectrum, yields only limited information on WTP (in the form of upper or lower bounds), but is argued to be easier for the survey respondent to answer and, under certain sets of assumptions, is incentive compatible. The choice between the two represents a classic variance-bias tradeoff, though the NOAA Blue Ribbon Panel argued that binary choice referenda are preferred.

In the years following the NOAA Panel report, researchers have sought a middle ground, trying to squeeze more information out of each survey respondent, while avoiding the problems with pure open-ended questions. One popular variation is the double-bounded discrete choice (DBDC) format. The approach augments the standard binary choice format with a follow-up question, asking the individual to further narrow the range of the willingness to pay, W_i. For example, suppose an individual responds yes to an initial question asking if they would vote for a referendum given a cost of T. The follow-up question would ask if they would still vote for the referendum given a higher cost, say $T_H > T$. On the other hand, if they respond no to the first question, they are then asked if they would vote yes if the cost were lowered to $T_L < T$. The responses, if truthful, provide tighter bounds on the individual's WTP, with

$$\begin{aligned}
&(\text{no,no}) \Rightarrow W_i \in (-\infty, T_L) \\
&(\text{no,yes}) \Rightarrow W_i \in [T_L, T) \\
&(\text{yes,no}) \Rightarrow W_i \in [T, T_H) \\
&(\text{yes,yes}) \Rightarrow W_i \in [T_H, \infty).
\end{aligned} \tag{12.20}$$

These tighter bounds, in turn, yield more precise parameter estimates.

Unfortunately, more data is not the same thing as more useful information. In early applications of the double-bounded format, researchers treated the responses to the first and second questions as independent draws from the same distribution, for example, Hanemann, Loomis, and Kanninen (1991). Subsequently, others subjected this assumption to statistical tests and found that it was not consistent with the data (Cameron and Quiggin 1994), and can lead to biased welfare estimates.[7] In general, the WTP calculated from both questions together is often less than the WTP based on responses to the first question alone. Carson and Groves (2007) offered an explanation based on an examination of the incentive properties of this

7 In some sense, this result should not be surprising. The double-bounded format is essentially a structured version of the bidding game approach used earlier in the literature and found to be potentially biased.

question format, a point returned to below. Herriges and Shogren (1996) provided an explanation based on anchoring.

Other formats have been suggested in the literature, including a one-and-one-half-bound format (Cooper, Hanemann, and Signorello 2002) and the multi-bounded discrete choice format (Welsh and Poe 1998). In each case, the formats try to elicit additional bounds on preferences. However, in doing so, the formats risk the incentive compatibility of the survey questions and open up the possibility of framing effects driven by the structure of the questions themselves. How substantial these effects are remains an empirical question.

Contingent Behavior

As noted in the introduction to this chapter, a fundamental limiting factor often encountered in applications of revealed preference techniques is the lack of sufficient and independent variation in the environmental attributes of interest. Without such variation, it is not possible to identify the causal impact that these attributes have on individual behavior and welfare without imposing additional assumptions on consumer preferences. For example, suppose one had used travel cost data to estimate a demand function for visits to a single recreation site, but one wanted to know the value of a change in one of the environmental attributes of that site. In the absence of an observed variation in the environmental attribute, it might not be possible to predict the shift in the demand curve for visits to the site. Even in those studies with substantial variation in the environmental amenity of interest, either through spatial or temporal variation, it is often difficult to isolate the impact of that amenity. For example, most recreation demand studies have data on only a small number of site attributes, such as a measure of water clarity or fish stock. The problem is that these measures are likely correlated with the many unobserved site attributes, leading to the classic omitted variables bias problem for the included variables.

Contingent behavior studies provide one resolution to this problem by asking survey respondents, not how much they would value proposed policy scenarios, as one would in contingent valuation, but rather how they would change their behavior if the policy changes occurred. For example, in the single site example above, individuals could be asked how their visitation behavior would change if the environmental attribute were to change in a specified way. This contingent change in visitation rates could then be used to estimate the shift in the demand curve for visits. McConnell (1986) used this approach to estimate the benefits to visitors of local beaches if the pollution of New Bedford Harbor, Massachusetts, by polychlorinated biphenyls (PCBs) could be eliminated. For more recent applications, see Cameron et al. (1996), Azevedo, Herriges, and Kling (2003), and Egan and Herriges (2006).

The advantage of the contingent behavior approach is that, if it is an accurate reflection of what individuals would actually do under the hypothetical scenario presented, it not only fills in information that is missing on consumer

reactions to alternative environmental conditions, but also it does so in a way that is independent of other unobservable factors. Thus, while actual recreational trips may be impacted by both observed water quality conditions and those that are unobserved, making it hard to identify the effect of water quality alone, the contingent behavior questions can ask how trips would change given *just* a shift in water quality—that is, *holding fixed* all other conditions at the site (see von Haefen and Phaneuf 2008).

The concern with contingent behavior data is that the incentives for truthful responses are less clear. Consider, for example, a proposed policy scenario involving an improvement in water quality at a given site with a fixed cost to consumers (in particular, the cost is not in the form of an entrance fee). An individual who would see the policy as welfare improving might exaggerate their purported future use of the site if they perceived decision-makers would value such use through, say, its economic impact on the surrounding community. There have been relatively few studies examining the consistency between actual trips and those reported through contingent behavior questions (e.g., Azevedo, Herriges, and Kling 2003; Jeon and Herriges 2010; von Haefen and Phaneuf 2008). Generally, these studies have rejected consistency between the two data sources, though the divergence, while statistically significant, has not always been substantial. More research in this area would seem warranted.

Choice Experiments

Attribute-Based Methods

Chapter 4 described the use of the discrete choice model to estimate the parameters of a utility function from individuals' choices of one alternative from a set of $\mathcal{J}$. The discrete choice contingent valuation format described above is a simplified version of this approach in that individuals make their selection of one alternative from a set of two (the alternatives of yes or no). Choice experiments or attribute-based methods of estimating values ask individuals to provide more information about their preferences by giving them more alternatives than the discrete choice approach and by asking them either to select their most preferred option or to rank alternatives in order of preference.[8] Each alternative is described in terms of a series of attributes. Normally one attribute would have a monetary dimension—for example, a price—in order to facilitate calculation of monetary values.

In this respect, choice experiments bear more than a passing resemblance to the random utility models used in the analysis of recreation demand (see Chapter 9). In both cases, the objects of choice are differentiated by embodying different levels

8 Some researchers have asked respondents to rate the alternatives on some scale. See, for example, Roe, Boyle and Teisl (1996) and Boyle et al. (2001). Holmes and Adamowicz (2003) also discussed this approach but conclude that the pick one or ranking formats are to be preferred because people may not be able to translate differences of strength of preference into numerical ratings.

of a set of attributes. By focusing on the tradeoffs among attributes, both methods yield estimates of the marginal rates of substitution between pairs of attributes and, where price is one of the attributes, the marginal willingness to pay for the attribute. Furthermore, with the stated choice method the analyst has experimental control through the design of the attributes presented in the choice set.

One of the earliest applications of choice experiments in economics was that of Beggs, Cardell, and Hausman (1981). They used an ordered logit model to estimate the values of characteristics of alternative models of cars, including electric vehicles. Their respondents were asked to rank sixteen alternative vehicle designs. Each design had nine attributes, including purchase price and fuel costs per mile. The method has subsequently been used in the environmental realm to value rural visibility (Rae 1983), water quality (Smith and Desvousges 1986), and the avoidance of diesel odor (Lareau and Rae 1989). The Smith and Desvousges study was perhaps the simplest for respondents: they were asked to rank only four alternatives, each of which involved only two attributes, a description of water quality and an annual fee. Other studies have posed the simpler task of asking respondents to indicate only their most preferred alternative. Examples include Adamowicz, Louviere and Williams (1994) and Morrison, Bennett, and Blamey (1999).

There has been a significant resurgence of interest in choice experiments in recent years. This stems in part from the rich array of data that it provides relative to the single-shot binary discrete choice contingent valuation exercise. The binary choice CV is typically designed to consider a single specific policy scenario, with each survey respondent providing a single bound on her WTP for the proposed program. In contrast, choice experiments ask respondents to compare a series of alternatives, typically presented using a sequence of pairwise or three-way comparisons (with one alternative often consisting of the status quo). By systematically varying the attributes both across the sequence of alternatives presented to any one individual and across individuals, the analyst can, in theory, estimate the marginal willingness to pay for each attribute and how that WTP interacts with the level of other attributes.

The additional data provided by choice experiments, however, does not come without a cost. Presenting individuals with a sequence of comparisons can result in respondent fatigue, with participants ignoring portions of the information provided and drawing on simpler decision rules to process the complex set of alternatives (see, for example, DeShazo and Fermo 2002). There are also concerns about order effects in the presentation of the choice attributes (see Day et al. 2011 and Boyle and Özdemir 2009). Finally, the conditions under which choice experiments are incentive compatible are more stringent than those required for the single-shot binary choice referendum (Vossler et al. 2011).

Analysis of Stated Choice Responses

The analytical model used to extract information about preferences from the stated choice responses is a straightforward extension of the discrete choice model. This

model provides that set of parameter weights on the attributes that maximizes the likelihood of realizing the observed choice (the most preferred alternative or the complete rank ordering). Provided one of the attributes is a money measure, parameter weights can then be used to calculate the marginal willingness to pay for an attribute (the marginal rate of substitution between income and the attribute) or the willingness to pay for a nonmarginal change in its level.

Suppose that an individual is asked to pick the most preferred alternative or to rank a set of $\mathcal{J}$ alternatives, with the *j*th alternative described in terms of price p_j and a vector of K nonprice attributes $\boldsymbol{Q}_j = \left(q_{1j},\ldots,q_{Kj}\right)$, where q_{kj} denotes the level of the *k*th attribute for alternative *j*. The individual's indirect utility associated with alternative *j* consists of two components, with

$$u\left(p_j,\boldsymbol{Q}_j,M,\boldsymbol{S}\right) = v\left(p_j,\boldsymbol{Q}_j,M,\boldsymbol{S}\right) + \varepsilon_j\,, \tag{12.21}$$

where $v(\cdot)$ denotes the portion of utility that is a function of factors observable by the analyst, whereas ε_j captures the remaining portion of $u(\cdot)$ that is unknown to the analyst and assumed to be random. If the ε_j are independently and identically distributed with a Type I Extreme Value distribution, the probability of the individual picking alternative 1 or ranking it first is

$$\Pr\left[u_1 > u_j;\text{for all } j \neq 1\right] = \frac{e^{v_1}}{\sum_{j=1}^{\mathcal{J}} e^{v_j}}\,. \tag{12.22}$$

In the case of contingent ranking, suppose that the individual values the alternatives in the following order from most preferred: $r_1,\ldots,r_{\mathcal{J}}$, where r_m denotes the number of the alternative given rank *m*. The probability of this ranking is the probability of ranking r_1 above the remaining $\mathcal{J}-1$ alternatives times the probability of ranking alternative r_2 above the remaining $\mathcal{J}-2$ alternatives, and so forth. That is,

$$\Pr\left[u_{r_1} > u_{r_2} > \cdots > u_{r_{\mathcal{J}}}\right] = \frac{e^{v_{r_1}}}{\sum_{j=1}^{\mathcal{J}} e^{v_j}} \cdot \frac{e^{v_{r_2}}}{\sum_{j=2}^{\mathcal{J}-1} e^{v_{r_j}}} \ldots = \prod_{j=1}^{\mathcal{J}} \left[\frac{e^{v_{r_j}}}{\sum_{l=j}^{\mathcal{J}} e^{v_{r_l}}}\right]. \tag{12.23}$$

If, as is typically the case, the survey respondents are asked to make a sequence of comparisons of such choices or rankings, the corresponding probability expressions become more complex, particularly if one allows for correlations across the responses made by the same individual across different sets of comparisons (as one should). Discussion of these issues can be found in Holmes and Adamowicz (2003).

As in the other applications of the discrete choice model, once a functional form for the observable component of the indirect utility function has been specified, its parameters can be estimated from data on the choices or rankings of a sample of individuals using maximum likelihood methods. Welfare measures can then be

derived by computing the marginal rate of substitution between income and any of the elements of $\boldsymbol{Q}$.

Assessing the Validity of Stated Preference Welfare Measures

As Mitchell and Carson (1989, 190) put it, "The validity of a measure is the degree to which it measures the theoretical construct under investigation." The theoretical construct of interest here is the individual's true value, her true probability of accepting an offer or selecting one of the alternatives offered, her true ranking of alternatives, or her true change in the level of an activity. Ideally, one would like to assess the validity of a stated preference measure by comparing it with the true measure. But at least in the cases of monetary values (especially those including passive use values), probabilities, and rankings, the true measure cannot be known, so this option is not available. In the case of contingent behavior questions, we will not often be able to observe "true" changes in activity levels under precisely the same conditions that prevailed when the question was asked. So, much of the discussion of validity in the literature focuses on less direct approaches to validity assessment, such as asking whether the results are consistent with economic theory (construct validity) or whether the study was conducted in accordance with generally accepted principles (content validity).

In this section, four types of validity are described along with the approaches to assessment for each type. These are not mutually exclusive alternatives but should be seen as focusing on different aspects of the way a study is conducted and of the information generated by the study. For each type of validity, a summary is provided in terms of what can be said at present about the validity of stated preference methods in general, as distinct from the validity of particular studies. For additional discussion of validity and a review of the evidence available as of the date of writing, see Mitchell and Carson (1989). The reader should also consult Kling, Phaneuf, and Zhao (2012), Carson (2012), Carson and Hanemann (2005), and some of the essays included in Bjornstad and Kahn (1996) and Kopp, Pommerehne, and Schwarz (1997). Bishop (2003) also includes an insightful discussion of concepts of validity.

Criterion Validity

The Concept

Assessing criterion validity involves comparing the value as measured by the stated preference method with some alternative measure that can be taken as the criterion for assessment. It is tempting to think of the "true" value as being the appropriate criterion. However, the "true" value is a theoretical construct: the sum that creates a state of indifference between two alternative states of the world; and as such, it is not directly observable (Bishop 2003). Instead, researchers have turned to experimental

methods, in field or laboratory settings, to create simulated markets in which individuals can engage in actual transactions. In simulated or experimental markets, transactions actually take place and respondents live with the consequences of their choices. These experiments allow the comparison of responses to hypothetical questions with responses in similarly designed experimental or simulated markets. These authors have used the observed values from these transactions as the criterion for comparison with stated preference responses.

The Evidence

One of the pioneering studies of this type was conducted by Bishop and Heberlein (1979) who made hypothetical closed-ended offers to purchase special goose hunting permits from a sample of hunters. At the same time, they made actual cash offers for a limited number of permits from a different sample, so they could compare the values revealed. The mean value of a permit calculated from the responses to the hypothetical offer (a WTA measure) was about 60 percent higher than the mean revealed by the real cash offers. Similar findings have emerged in subsequent lab and field experiments, including studies by Neill et al. (1994), Cummings, Harrison, and Rutström (1995), and Cherry et al. (2004). Indeed, Harrison and Rutström (2008) noted that roughly three-fourths of such studies find that stated preference values are biased upwards.

List and Gallet (2001) provided one the first systematic analyses of this line of research, reviewing the results of 29 studies dealing with the provision of both private and public goods, including that of Bishop and Heberlein. They used a meta-analysis to explain the reported calibration ratios; that is, the ratios of minimum, median, and maximum hypothetical values to the corresponding values from the real transactions. Although the median calibration ratio for their sample was about 3.0, they found that using a WTP rather than WTA question and using a first-price sealed bid auction mechanism both had statistically significant effects in lowering the calibration ratio. A more recent meta-analysis by Murphy et al. (2005) came to similar conclusions, though they found a substantially smaller median calibration ratio of 1.35. In any case, the persistent gap between hypothetical and real cash transactions documented in both lab and field experiments has come to be referred to as "hypothetical bias" and has been a central point of concern regarding the application of stated preference methods in welfare analysis.

The response to the experimental evidence of criterion invalidity (in the form of hypothetical bias) has taken two forms in the stated preference literature. First, some authors have investigated whether changes in the design or interpretation of stated preference surveys can mitigate or eliminate hypothetical bias. Second, and more recently, a number of authors (Carson and Groves 2007; Vossler and Poe 2011) have come to question the value in comparing purely hypothetical and actual transactions, arguing that stated preference questions require an element of consequentiality in order to be incentive compatible. These two forms of responses are discussed in turn.

REDUCING HYPOTHETICAL BIAS

A number of strategies have emerged in the literature designed to ameliorate hypothetical bias in stated preference surveys. Ex ante approaches seek to alter the survey process so as to encourage truthful answers from the survey respondent, while ex post strategies use the survey data itself to adjust either individual responses or the aggregate measure of WTP derived from the survey.

The most popular of the ex ante strategies is the use of a so-called "cheap talk" script. With this procedure, survey participants are explicitly warned of the tendency for hypothetical bias in stated preference surveys and asked to treat the survey "as if" it were real. Cummings and Taylor (1999) found that, in an experimental setting, their cheap talk script was successful in inducing hypothetical valuation responses that were not statistically different from comparable responses involving real cash payments. However, subsequent applications have found the efficacy of cheap talk to be somewhat idiosyncratic, varying with the length of the cheap talk script used (Aadland and Caplan 2003), the experience of survey respondents (List 2001), and other factors. See Morrison and Brown (2009) for a useful summary. Taking a different tack, Jacquemet et al. (2013) used an oath script, asking respondents to promise to tell the truth in responding to the survey questionnaire. They found that the oath script was successful in inducing responses that are consistent with parallel real transaction responses. In the case of both "cheap talk" and "oath scripts," however, more research is needed into why it does (or does not) work before it can be reliably used to close the gap between hypothetical and actual transactions responses.

A number of ex post approaches to dealing with hypothetical bias have also emerged in the literature. One tack is to adjust values obtained from a stated preference study by a calibration factor (see, for example, Diamond and Hausman 1994, 54) to reflect the observed disparity between hypothetical and real transactions. While appealing in its apparent simplicity, it is not clear which calibration factor one would use in a given setting. Experimental comparisons of hypothetical and real transactions can provide some guidance, though these are often conducted using private goods, while stated preference work is particularly useful in the context of public goods. Moreover, the meta-analyses of List and Gallet (2001) and Murphy et al. (2005) suggest that the calibration factors can vary substantially across settings. Using data from their field experiment, List and Shogren (1997, 194) concluded that "calibration is good- and context-specific."

A second ex post approach to controlling for hypothetical bias involves recoding survey responses based upon the self-reported certainty of the respondent in answering discrete choice referendum questions. A variety of formats have been used to allow individuals to express their degree of confidence, from qualitative ratings (such as the "definitely sure" and "probably sure" categories used by Blomquist, Blumenschein, and Johannesson, 2009) to numerical scales (such as the "1 very unsure" to "10 very certain" scale used by Champ et al. 1997). Numerous studies have found that recoding some or all "uncertain" yes responses to no responses yields a better match between hypothetical surveys and their

actual transactions counterpart (see, for example, Champ et al. 1997; Champ and Bishop 2001; Blumenschein et al. 1998; Poe et al. 2002), though the precise cutoff used for recoding varies across the studies.

There are a number of concerns with this type of ex post recoding. First, it is not clear why it works and which specific cutoff is appropriate in a given setting. The argument is typically that the "uncertain" yes's do not represent a true commitment to purchase the policy option being offered, but rather are a form of "yea-saying." The fact that it "works" in experimental settings is relatively little comfort, since a variety of other bases for recoding would do the same, as long as they reduced the proportion of yes responses in the survey. One might, for example, argue (facetiously) that older survey respondents are easily confused by complex survey instruments and recode all their yes's to no's. If age was not typically a factor in WTP, there would exist an age cutoff that would successfully yield a match between hypothetical and actual referenda responses, but hopefully few would actually buy the senility argument. Second, if uncertainty is indeed a cause of hypothetical bias, then the random utility maximization (RUM) model typically used to analyze such discrete choice responses becomes problematic in that the RUM framework assumes that individuals know precisely the utility (or at least the expected utility) associated with each option. Understanding the nature of respondent uncertainty, and how it varies across individuals, becomes important in modeling the decision process used in choosing among alternatives. This remains an area in need of both theoretical and empirical research.

FOCUSING ON INCENTIVES INSTEAD

In recent years, a number of authors (Carson and Groves 2007; Vossler and Poe 2011) have argued that experimental comparisons between purely hypothetical and real transactions should not be the point of departure in evaluating stated preference techniques. Rather, one should start by asking what incentives a given elicitation procedure creates for truthful revelation of the individual's preference. In particular, Carson and Groves (2007, 183) distinguished between two broad categories of survey questions:

- Consequential survey questions: If a survey's results are seen by the respondent as potentially influencing an agency's actions and she cares about the outcomes of those actions, she should treat the survey questions as an opportunity to influence those actions. In such a case, standard economic theory applies and the response to the question should be interpretable using mechanism design theory concerning incentive structures …[and]
- Inconsequential survey questions: If a survey's results are not seen as having any influence on an agency's actions or the agent is indifferent to all possible outcomes of the agency's actions, then all possible responses by the agent will be perceived as having the same influence on the agent's welfare. In such a case, economic theory makes no predictions.

There are two important implications of this dichotomy. First, purely hypothetical stated preference surveys are uninformative and should not be used. Indeed, Carson and Groves (2007, 183) went out of their way to "formally reject the notion, sometimes advanced by proponents of preference surveys, that when a respondent perceives no gain or loss from how a preference survey is answered, the respondent always answers truthfully. While such an assumption may be true, there is no basis in economic theory to either support or deny it." A corollary to this is that the experimental comparisons between purely hypothetical and actual cash transactions that have proliferated in the literature shed little, if any, light on the validity of properly designed and consequential stated preference surveys.

Second, alternative elicitation formats used in a consequential survey instrument should be evaluated in terms of well-known results from mechanism design theory. A consequential survey instrument is not necessarily incentive compatible—that is, it need not encourage truthful revelation of preferences. However, understanding the incentives of respondents is key to understanding the direction of bias, if any, one should expect from their responses. For example, Carson and Groves (2007) argued that, when asked about their WTP for a proposed *private* good, respondents have an incentive to say yes to a given bid (T), not because their WTP for the new good necessarily exceeds T, but by saying yes they encourage the supply of the good and create the option to consider its purchase at a later date. Thus, one should expect an upward bias in the estimated WTP derived from such a survey instrument.

Drawing on the results from mechanism design theory, Carson and Groves (2007) went on to examine several of the elicitation formats commonly used in the stated preference literature. They concluded that many of the formats, including open-ended, double-bounded, and multinomial choice formats, are *not* incentive compatible. However, the single-shot binary choice referendum format is incentive compatible if (a) the probability that the proposed project is implemented is increasing in the proportion of yes votes, and (b) the agency proposing the policy scenario can enforce payment at the proposed price (T). Experimental evidence to date appears to corroborate this claim (see, for example, Carson et al. 2004; Landry and List 2007; Vossler and Evans 2009). Herriges et al. (2010) provided additional support in the context of a single-shot dichotomous choice referendum question. More research, however, is needed in terms of how to design survey instruments that meet the incentive compatibility conditions. Vossler et al. (2011) provided a similar discussion regarding the conditions for incentive compatibility in the context of discrete choice experiments.

Convergent Validity

The Concept

A related form of validity assessment is the comparison of stated preference values with measures derived from some revealed preference method. This provides an assessment of convergent validity. This strategy is as old as the contingent valuation

method itself (see Knetsch and Davis 1966). The problem with this measure of validity, of course, is that if a discrepancy between the stated preference value and the alternative is discovered, it could be because of the lack of validity of the stated preference measure, the lack of validity of the alternative measure derived from observation, or both (see, for example, Azevedo, Herriges, and Kling 2003).

The Evidence

Early reviews of this evidence can be found in Cummings, Brookshire, and Schulze (1986) and Mitchell and Carson (1989). More recently, Carson et al. (1996) provided a large study of convergent validity of contingent valuation estimates of WTP for quasi-public goods (goods provided by the government for which exclusion of users is possible). The estimates covered were for three types of goods: access to outdoor recreation activities, environmental amenities, and changes in health risks. They found that the ratio of CV to RP values averaged 0.89 for the whole sample with a 95 percent confidence interval of 0.81–0.96. Based on their evidence, the authors argued against systematic calibration of CV values and for convergent validity.

Construct Validity

The Concept

Construct validity asks whether stated preference responses are related to variables that economic theory suggests should be predictors of WTP, and invariant to factors that theory suggests should be irrelevant. For example, it is usually thought that values should be an increasing function of income, other things being equal, and that the estimated income elasticity of WTP should be "reasonable." When an experimental design involves facing different individuals with different quantities and/or implied prices, the choices should also be consistent with the axioms of revealed preference, such as transitivity (see, for example, Adamowicz and Graham-Tomasi 1991). Construct validity can be assessed by regressing expressed values on characteristics of the good being valued and characteristics of the respondent. Depending upon the stated preference approach, these assessments may involve comparisons *across* randomly assigned groups that were exposed to different treatments (such as the price, payment mechanism, and the scope of the proposed policy scenario). Other elicitation formats that expose participants to a series of choices (such as in discrete choice experiments) allow *within* individual validity checks.

While there are a myriad of possible validity checks one might consider, the literature has focused on three particular areas of concern, involving the sensitivity of stated preference valuations to

- income,
- scope, and
- the use of WTP versus WTA measures.

Each of these topics is considered in turn below.

Evidence on Sensitivity to Income

It is generally thought that environmental amenities represent luxury goods, with income elasticities that would exceed one. The fact that WTP is often found to be inelastic with respect to income has been cited as evidence of the construct invalidity of stated preference techniques (McFadden 1994). However, Flores and Carson (1997) showed that the relationship between the income elasticity of demand for a quantity constrained good and the corresponding income elasticity of WTP is not as simple as one might think. They showed that in the case of more than one quantity-constrained good, the income elasticity of WTP for a good depends not only on the income elasticities of demand for all the quantity-constrained goods but also on the cross-price substitution effects for those goods, and on the expenditures for market goods as a percentage of virtual income. This latter term will always be less than one, leading to a likelihood that the income elasticity of WTP will be less than the income elasticity of demand. However, they also showed by example that plausible values for the three determinants of the income elasticity of WTP can lead to values greater than the income elasticity of demand, or even negative values for so-called normal goods. So very small or even negative values for the income elasticity of WTP for a nonmarket good are not prima facie evidence of lack of construct validity.

Evidence on Sensitivity to Scope

It is generally assumed that an increase in the "quantity" of the good being valued should produce an increase in the magnitude of expressed values. This increased quantity can take the form of a literal increase in the number of environmental amenities provided (such as the number of birds saved or acres of wetlands preserved). Alternatively, the increase can be in the number of distinct subprograms proposed (going from, say, a program to provide improved rescue equipment and trained personal to a package of programs in education, safety, etc., that includes the safety equipment and personnel as a component, as in Kahneman and Knetsch 1992).[9] In either case, tests of the hypothesis that value increases with the size of the proposed policy have come to be known as scope tests. The NOAA Panel (Arrow et al. 1993) included conducting a scope test in its recommended guidelines.

Inadequate response of expressed WTP to changes in scope is often cited as a reason to reject stated preference techniques by critics of the methods, for example Diamond and Hausman (1994) and Hausman (2012). Yet, several authors have reviewed the evidence from scope tests and found that they can reject

9 This latter type of scope change is also referred to as "nesting."

the hypothesis of inadequate responsiveness of expressed values to changes on scope (see Smith and Osborne 1996, Kopp and Smith 1997, Brouwer et al. 1999, and Carson 1997). Ojea and Loureiro (2011) found scope sensitivity in their meta-analysis of biodiversity valuation studies, and Lew and Wallmo (2011) found scope sensitivity in a choice experiment setting. Moreover, a number of the early studies purporting to show scope insensitivity have come under criticism in terms of both survey design and analysis (see, for example, Harrison 1992). Powe and Bateman (2004) also provided experimental evidence that an erroneous scope insensitivity finding may result from survey respondents questioning the realism of the larger programs and not because the programs themselves are not valued.

Heberlein et al. (2005) suggested that the current emphasis on scope testing is misplaced and that, in many instances, a finding of scope insensitivity is perfectly consistent with both psychological and economic theory. Using an example of species preservation, they note that an individual may be willing to pay for the preservation and protection of 300 wolves, but would be reluctant to support a population growth to 800 wolves, fearing the increased potential for negative externalities. Through a series of such scenarios, Heberlein et al. (2005, 20–21) illustrated the fact that values can depend in complex ways on scope and how it is communicated, and that "more is better" is too simplistic a characterization of preferences. They conclude "that, by itself, failure to pass a scope test is neither a necessary nor sufficient condition to invalidate any given CV study. And certainly the scope test failures reported so far in the literature do not undermine the validity of the entire CV method."

Evidence on the WTP/WTA Gap

SP questions can be asked either in WTP or WTA formats. A large body of evidence suggests that WTP and WTA formats for stated preference values lead to quite different results. WTA responses are typically several times larger (and, in at least one case, two orders of magnitude larger) than WTP responses for the same change. See Cummings, Brookshire, and Schulze (1986, especially 35) for a review of earlier studies comparing WTP and WTA. Brookshire and Coursey (1987) also found large differences. Some have argued that the disparity between WTP and WTA responses is evidence that SP methods lack construct validity.

There are two basic responses to this concern in the literature. The first response is that, as discussed in Chapter 3, there are theoretically plausible reasons why there could be large differences between WTP and WTA, at least in the case of environmental resources and amenities. For example, Hanemann (1991) noted that the gap between WTP and WTA will be larger, all else equal, if there are few substitutes for the commodity in question, which is likely to be the case for many environmental amenities. Shogren et al. (1994) provided empirical support for this argument using laboratory experiments. Zhao and Kling (2004), on the other hand, argued that dynamic considerations, involving uncertainty and the opportunity for learning, can also drive a wedge between WTP and WTA. While both of these

arguments are valid, the counterargument is that it is the size of the gaps, not their direction, that are implausible. Specifically, Sugden (1999) derived the relationship

$$1-\frac{\text{WTP}}{\text{WTA}} \approx \frac{\partial \text{WTP}}{\partial M} \tag{12.24}$$

where M denotes income. Using this result, Horowitz and McConnell (2003, 539) examined the WTA/WTP ratios available in the literature and concluded that they "imply income effects that are implausibly high, much higher than income effects found in valuation studies; and much higher than income effects estimated in the set of studies from which the WTA/WTP results are drawn. Based on this evidence, we reject the claim that observed WTA/WTP ratios are consistent with a standard neoclassical model."

The second response to the disparity between WTA and WTP is to note that such differences are not limited to stated preference settings, but have also arisen in lab and field experiments involving real transactions. Indeed, in their review of WTA/WTP ratios found in the literature, Horowitz and McConnell (2002) found that real experiments do not yield significantly lower ratios. Thus, the observed disparities between WTP and WTA are not prima facie evidence of lack of construct validity for stated preference techniques per se, but perhaps raise broader concerns regarding neoclassical theory.

Content Validity

The Concept

Content validity refers to the extent to which the design and implementation of the survey conform to the generally recognized best practice or state of the art. It should involve an examination of the survey instrument, including the scenario specification, the elicitation question—especially its incentive properties—and the payment vehicle, as well as procedural matters such as sample size and design and the analysis of data. For a more complete discussion of the assessment of content validity, see Bishop and McCollum (1997).

Practitioners in the field of stated preference valuation methods have identified a number of problem areas and issues that can affect the validity of responses. There is general agreement, for example, that the form of the valuation question is important. Discrete choice questions work better than the bidding game format. WTP questions work better than WTA questions, and a number of sources of potential bias have been identified. Thus, much can be learned from a systematic examination of the survey instrument.

Mitchell and Carson (1989, especially Chapter 11) provided a classification of ways in which scenarios can affect individuals' responses so as to introduce systematic error or bias into the value measures that are derived from these responses. Systematic errors arise from several sources. One is scenario misspecification. Were the question and the information provided with it to establish its context

understood by the respondent in the way intended by the investigator? A second source is implied value cues. To what extent did the information provided to the respondent predispose her toward a particular response? A third is the choice of a payment vehicle. Did the payment vehicle tap into attitudes about government and taxation or business pricing policies that might bias responses? Finally, were there incentives for misrepresentation of values? Part of the art of asking stated preference questions involves avoiding problems such as these. In what follows, each of these sources of error is taken up in turn.

Scenario Misspecification

The various types of scenario misspecification discussed by Mitchell and Carson (1989, 246–258) all have the effect of creating a divergence between what the respondent understands about the choice situation and what the investigator intends him to understand. The investigator can increase the likelihood that the questions will be understood as intended by developing an awareness of how people think about the issues at hand and what language and terminology they use in talking about it. This can be accomplished through the use of focus groups, pilot surveys, pretests, and the like. Focus groups involve structured interviews with small groups of individuals from the population to be surveyed. Focus groups can be used to investigate perceptions of environmental problems, to determine which terminologies have particular salience, and to try out various descriptions and question formats. Pretests and pilot surveys may be useful in testing different scenarios and question formats, especially if respondents are queried afterwards to identify questions and problems they may have with specific wording.

Implied Value Cues

If a respondent has well-formed preferences regarding the hypothetical choice problem posed to him, he should find the task of formulating a response to the question fairly easy. However, if the choice problem posed to the individual is unfamiliar and the individual is not clear about her preferences, she might, quite unconsciously, seek clues regarding the "correct" choice or value from the information supplied as part of the scenario specification. If such clues are present, they may systematically bias individuals' choices and values. Such clues are termed implied value cues. The most well-studied implied value cue is the starting point in bidding game versions of direct expression of value questions. Other forms of direct value questions may avoid starting point biases.

Implied value cues can be found outside of the scenario itself. For example, respondents who are asked about the value of access to a recreation site or to a fishing experience may anchor their responses to the implied value provided by the normal admission fee or fishing license fee. The task for the investigator is to find ways to thwart this tendency, perhaps by specifying a payment vehicle that focuses attention away from the customary admission fee or license fee. For

example, recreationists might be asked about their willingness to pay higher travel costs to reach a site.

Stated choice and contingent behavior questions appear to be less susceptible to implied value cues. However, there could be a tendency for the results of a stated choice question to depend upon the order in which options are initially presented for ranking. Order effects can be examined by randomizing the sequence in which alternatives are presented to respondents, allowing the researcher to test for order effects explicitly. If order effects are determined to exist, it may only be possible to bound the values associated with an individual alternative.

Payment Vehicle

Another feature of scenario design that may be important is the specification of a payment vehicle for collecting the respondents' willingness to pay. In order to make contingent market scenarios plausible to respondents, it is necessary to inform them of how their stated value will be collected. Examples of payment vehicles used in past studies include an increase in the prices of all goods resulting in a higher expenditure of $\$X$, an increase in taxes of $\$X$, and a surcharge on utility bills of $\$X$. If the vehicle involves a higher price for a market good, the question must be clear about whether it refers to a change in expenditure or a change in price. If the question is about a change in price, then the welfare calculation must take account of the reduction in quantity that a price increase will cause. In some studies where respondents were asked for a maximum increase in a price per unit, total willingness to pay was calculated by multiplying the increase in price by the observed quantity. However, assuming that respondents accurately stated the maximum price increase they would accept, the correct welfare measure is the area to the left of the demand curve between the two prices. This will be less than the original quantity times the increase in price.

There is evidence that the specification of a payment vehicle can exert an independent influence on bids, a phenomenon that has been termed "vehicle bias" in the literature. However, judging whether this influence is a bias in the usual sense of the word is a more difficult matter, since the respondent's "true" value is probably not known. The influence of the payment vehicle on bids is one example of the importance of what Fischhoff and Furby (1988) have called the social context of the hypothetical transaction. They argued that in both designing and interpreting surveys that elicit hypothetical values, researchers must be sensitive to how individuals understand and interpret the environmental good being valued, the payment they are being asked to consider, and the social context in which the exchange takes place.

Conclusions

Economists and other social scientists have made a great deal of progress in the design and implementation of stated preference methods since the earliest

contingent valuation studies of the mid-1960s and early 1970s. One major advance was the development and examination of a wide range of elicitation formats, broadly grouped as contingent valuation methods, choice experiments, and contingent behavior methods. Each of these approaches has its strengths and limitations relative to the others; but together they provide a more powerful array of tools for dealing with a variety of valuation problems.

A second advance was the development of a much greater awareness of, and sensitivity to, the ways in which scenario specification can influence responses and thereby affect the validity of the value measures. Practitioners in economics owe a debt to researchers from other fields, especially those in cognitive psychology, for their contributions to our understanding of how people respond to questions and of the importance of how questions are asked.

A third advance was the development of more sophisticated approaches to the analysis of data and the calculation of welfare measures, especially with respect to discrete choice models. Fourth, there has developed a much greater sensitivity to a variety of sampling issues and related problems such as the need to deal with item nonresponse and to identify and treat outliers and protest zeros. Finally, researchers are developing a variety of methods for combining stated preference and revealed preference data into one statistical model, a topic that is covered in greater detail in Chapter 13.

All of these developments have helped to make the set of stated preference methods a more powerful tool for dealing with amenity and resource valuation problems. Yet, stated preference methods continue to be controversial. On the one hand, the NOAA "Blue Ribbon Panel" concluded that "under those conditions [that were discussed earlier in the Report], CV studies convey useful information. … [and] can produce estimates reliable enough to be the starting point of a judicial process of damage assessment" (Arrow et al. 1993, 4610). This conclusion is in sharp contrast to that of Diamond and Hausman (1994, 62), "that contingent valuation is a deeply flawed methodology for measuring nonuse values, one that does not estimate what its proponents claim to be estimating." Indeed, Hausman's position has changed little over the past two decades. He concluded recently that "'no number' is still better than a contingent valuation estimate… [U]nless or until contingent value studies resolve their long-standing problems, they should have zero weight in public decision-making" (Hausman 2012, 54).

Our own assessment of SP methods is cautiously optimistic. On the one hand, stated preference techniques should not be viewed as a panacea. They do not provide a "simple" alternative to the daunting, and sometimes impossible, task of valuing changes to environmental amenities based on revealed preference data alone. Designing, implementing, and analyzing a stated preference survey requires careful attention to detail, with an unusual mix of both art and science to be successful. Reading this chapter alone is certainly not sufficient to equip one to do a SP study well. There is no substitute for a careful reading in the now substantial and rapidly growing literature—including, but not limited to, the sources cited here. Having said that, the preponderance of evidence suggests that a carefully

executed SP study provides valuable insights into the tradeoffs that individuals are willing to make to secure or avoid changes to the environment. To ignore these tradeoffs, or to rely solely on expert opinion regarding such tradeoffs (as Diamond and Hausman 1994, 56, suggested), would seem counterproductive.

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Chapter 13

Additional Topics

This chapter covers four additional topics that do not fit easily into any of the previous chapters. The first is the use of values borrowed from studies of values in other settings to estimate the value of the environmental change of interest. This practice has come to be known as "benefits transfer." The second is the use of data from both revealed preference and stated preference analyses to estimate a single valuation model. The third is the estimation of the values of services provided by ecosystems. The fourth is a brief discussion of some of the implications of behavioral economics for the task of estimating nonmarket values for environmental goods and services.

Benefits Transfer

Benefits transfer refers to the practice of applying nonmarket values obtained from primary studies of resource or environmental changes undertaken elsewhere to the evaluation of a proposed or observed change that is of interest to the analyst. Examples include:

- using values per day for recreational angling at one lake obtained from a travel cost demand study to value an increase in the same activity at another lake;
- using the willingness to pay to avoid an asthma attack obtained from an averting behavior study to value the prevention of a day of respiratory symptoms; and
- using the willingness to pay to preserve a square mile of tropical rain forest in one country estimated from a stated preference study to value the preservation of a square mile of rain forest in another country.

The practice of benefits transfer became common in the economic analysis of environmental regulations in the United States in the mid-1980s, even before any systematic development of either terminology or procedures and protocols and certainly before any rigorous testing of the validity of the practice. All of this changed in 1992 with the publication of a set of nine papers on benefits transfer

in a special issue of *Water Resources Research*. See Brookshire and Neill (1992) for an introduction and overview of these papers.

It is common to refer to the environmental policy being evaluated as the "policy site" and the source of the values being used as the "study site," even though many of the policies for which benefits transfer are used are not site specific. In principle, the values at the policy site can be different from those of the study site for two sets of reasons: differences in the characteristics of the two environmental features being valued ("supply side" factors) and differences between the populations making use of, or at least valuing, the resource change ("demand side" factors). The latter can include differences in income, tastes, and preferences, and other relevant socioeconomic characteristics. Thus, we must consider the question of how values can be adjusted in the transfer process to reflect these two types of differences. To examine this question, we look at the types of procedures available for benefits transfer from the simplest to the most sophisticated. There are several useful sources for more detailed information on how to design and implement a benefits transfer. One (Rosenberger and Loomis 2003) is a fairly basic "how to do it" that walks the reader through the steps involved, using three case studies. Likewise, Loomis and Richardson (2008) provides guidance on the process in a manual that accompanies an on-line toolkit (Loomis et al. 2007). Desvousges, Johnson, and Banzhaf (1998) provide a more comprehensive and rigorous treatment that includes discussions of several econometric issues in analyzing and summarizing study site data.

The simplest procedure is to select the study site that in the analyst's judgment is most similar to the policy site in terms of both demand-side and supply-side characteristics. If there is more than one suitable study site (or more than one estimated value for a given study site), one could use a range, calculate a mean or median value for the distribution, or perhaps subjectively weight the values based on judgments of their quality before taking the mean. If the study site and policy site values are for different years, the study site value should be adjusted for changes in prices using a suitable price index. However, adjusting for inflation in this way involves the assumption that the budget shares of expenditures of both the study site and policy site populations are similar to the weights used in constructing the price index (Eiswerth and Shaw 1997). If there is information available on the income levels of the two populations, then it would also be appropriate to adjust the study site WTP using an estimate of the income elasticity of WTP. The relevant elasticity, however, is not the income elasticity of demand for the good. As Flores and Carson (1997) showed, the relationship between the two elasticity concepts is complex.

This simple procedure does not allow for more systematic adjustments of study site values to account for differences in site characteristics and populations. To make such adjustments, it is necessary to have information on how values are related to the relevant characteristics. Some information of this sort is available if the study site data includes a valuation function or WTP function—that is, a statistical relationship between the WTPs of individuals in the sample and their socioeconomic characteristics. The adjusted policy site value can be calculated by plugging mean values for the policy site population into the valuation function.

A more sophisticated approach is to apply meta-analytical techniques to a larger sample of study site data that represents a variety of population and site characteristics. The meta-analysis equation can also include variables that reflect differences in methods, such as revealed preference versus various forms of stated preference methods. Examples include Smith and Kaoru (1990) for recreation and Mrozek and Taylor (2002) for the value of statistical life. For a more detailed analysis of meta-analysis for benefits transfer, see Bergstrom and Taylor (2006). Desvousges, Johnson, and Banzhaf (1998) and Rosenberger and Loomis (2003) contain references to additional examples.

More recently, in a series of papers, Klaus Moeltner has described several studies showing how to utilize meta-regression techniques and Bayesian models to deal with problems such as unobserved heterogeneity and small sample size in the underlying valuation studies. See Moeltner et al. (2007, 2009), Moeltner and Rosenberger (2008), and Moeltner and Woodward (2009). Phaneuf and van Houtven (forthcoming) recently added to this line of research.

Finally, Smith and others developed and applied a method, based on data from several studies using different valuation methods, to "calibrate" the parameters of an assumed underlying preference function over site characteristics. They then used the calibrated preference function to calculate values for a variety of policy site proposals. See Smith, van Houtven, and Pattanayak (2002), Smith, Pattanayak, and van Houtven (2006), and Pattanayak, Smith, and van Houtven (2007).

An important question concerns the validity of using benefit transfer values as a substitute for doing original valuation research for the policy being evaluated. Any assessment of the validity of benefits transfer requires a yardstick. The yardstick used by those who have investigated this question is an estimate of value obtained by stated preference or revealed preference methods at the policy site, so these are tests of convergent validity, not criterion validity (see Chapter 12).

Rosenberger and Loomis (2003) and Shrestha and Loomis (2001) both reviewed studies of validity (see papers for references). Also, Shrestha and Loomis (2001) performed a validity test of benefits transfer of outdoor recreation activities across countries based on a meta-analysis of more than 600 recreation values from U.S. studies. Overall, the results are mixed, but one finding stands out: convergent validity is generally higher when the transfer is done using either a valuation function or meta-analysis. This confirms the importance of using systematic methods to adjust study site values for differences in population and site characteristics. Nevertheless, significant differences between transfer values and original values often occur. See Lindhjem and Navrud (2008) for a review of the more recent literature on validity testing and their own test of convergent validity of benefits transfer involving non-timber values in Norway, Sweden, and Finland. They found substantial transfer errors, and accordingly, they were skeptical about the use of meta-analysis for international benefits transfers.

In recognition of the popularity of using benefits transfer and the potential savings in analytical costs that are possible, several organizations have created and made available on the Internet large databases of valuation studies that can be

used in benefits transfer. They provide not only the estimated values from each study but also other data on study methodology in a standardized format so that studies can be evaluated for their suitability for the transfer exercise. The largest and most detailed of these is the Environmental Valuation Reference Inventory (EVRI), created and maintained by the Canadian federal environmental agency, Environment Canada. It is a searchable storehouse of empirical studies on the economic value of environmental benefits and human health effects. Access to EVRI is free for Canadian residents, and subscription information is available at the EVRI website (www.evri.ca/Global/Splash.aspx).

There are also more specialized databases. For example the University of California has created a database of economic values for beneficial uses of water (http://buvd.ucdavis.edu/); the U.S. Fish and Wildlife Service maintains a sports fishing value database (www.indecon.com/fish/); and Texas A&M has compiled a database for valuing ecosystem services relevant to the Gulf of Mexico.

Combining Revealed Preference and Stated Preference Data

It seems natural to think of revealed preference (RP) and stated preference (SP) methods as substitutes or alternative ways of estimating values for a given change in environmental or resource conditions. Thought of in these terms, RP methods have served as a natural comparison point for convergent validity tests of SP as described in Chapter 12 (see Whitehead et al. 2010 for a current example). However, several authors have shown that it is possible to use RP and SP methods as complements by combining RP and SP data to estimate a single valuation model. These combined models embody the hypothesis that the RP and SP data come from the same structure of preferences or utility function. Testing this hypothesis can be viewed as a kind of convergent validity test.

In the first published study using this approach, Cameron (1992) specified a travel cost demand model to explain the numbers of fishing trips taken in a year by individuals in her sample of Texas anglers. She also specified a stochastic utility difference model to explain the same individuals' responses to a contingent behavior question of the form "If the travel cost of your trips were $\$X$ more, would you stop visiting these sites altogether?" The functional forms of the two equations were derived from an assumed underlying quadratic utility function. She constrained the common parameters of the two equations to be equal, estimating them using a simultaneous equation technique. She then used the estimated utility parameters to generate welfare measures for various changes in the conditions of access to the sites to illustrate the method.

Adamowicz, Louviere, and Williams (1994) designed a survey explicitly to generate both RP and SP data of recreation choices. Their RP data were used to estimate a random utility model of recreation site choices. This was then combined with responses to a set of SP questions about site choice based on the same set of

site attributes. The SP questions were designed to eliminate collinearity among attributes. They estimated the random utility model and SP models separately and then combined the two data sets for a joint estimation. Their results supported the hypothesis that both sets of data came from the same underlying preference structures. However, von Haefen and Phaneuf (2008) have since re-examined these data using more recent econometric techniques and found inconsistencies in preferences implied by the RP and SP data sources.

McConnell, Weninger, and Strand (1999) also combined RP and SP data on recreation activities to estimate willingness to pay for a recreation trip. Their method allowed differences in the preferences generating the RP and SP data because the SP responses came after individuals had visited the sites in question. However, they could not reject the hypothesis that the parameters in the two models were equal. More recently Earnhart (2001, 2002) combined SP data with hedonic housing discrete choice data to examine housing choices and to estimate values for environmental amenities linked to residential locations.

As Kling (1997) showed through simulation studies, combining SP and RP data can reduce bias and improve the precision of welfare estimates, assuming of course that the data generating processes (including preferences) are the same for both data sources. Also, the design of the choice sets for the SP part of the study presents an opportunity for examining portions of individuals' response surfaces or preference structures for which there is no observed behavior.

A review of the literature by Whitehead et al. (2008) showed there has been extensive use of combinations of various types of RP and SP data in the fields of marketing (predicting demands for new or changed products) and transportation policy. This review also covers the types of models and econometric techniques that can be used to combine RP and SP data and discusses ways in which combined RP and SP data can help to overcome some of the problems and limitations of relying on either RP or SP data alone. Combining RP and SP has become increasingly popular with recent applications to wetlands (Grossmann 2011), professional sports (Whitehead et al. 2013) and food safety (Morgan et al. 2013).

Valuing the Services of Ecosystems

An ecosystem can be defined as "an assemblage of organisms interacting with its associated physical environment in a specific place" (Barbier 2011, 28, and references therein). It is a human construct whose size and scope depend on how an investigator draws the spatially explicit boundaries around the objects of interest. The unit of analysis can be as small as the gut of an insect and as large as the delta area of a major river system such as the Amazon or Mississippi. Each ecosystem is characterized by its spatial dimensions, its species composition, the functions or processes that it carries out, and the services that it provides to people. Ecosystem functions or processes are the normal characteristic actions or activities of the system that are necessary for its self-maintenance (Whigham 1997, 231). Ecosystem services have been defined in various ways in the literature.

EPA has defined them as "those ecological functions or processes that directly or indirectly contribute to human well-being or have the potential to do so in the future" (U.S. EPA 2004, 4). Ecologist Gretchen Daily defined them as "the conditions and processes through which natural ecosystems sustain and fulfill human life. They maintain biodiversity and the production of *ecosystem goods*, such as seafood, forage, timber … and their precursors … In addition to the production of goods, ecosystem services are the actual life-support functions, such as cleansing, recycling, and renewal, and they confer many intangible aesthetic and cultural benefits as well" (Daily 1997, 3).

A number of authors have provided lists of ecosystem services. Gretchen Daily's (1997, 3–4) list is representative and includes:

- purification of air and water;
- mitigation of floods and droughts by regulation of hydrological cycles;
- detoxification and decomposition of wastes;
- generation and renewal of soils;
- pollination of crops and natural vegetation;
- control of agricultural and other pests;
- nutrient recycling;
- partial climate stabilization; and
- providing aesthetic beauty and intellectual stimulation.

Other authors (for example, Costanza et al. 1997) have added such things as erosion control, habitat or refugia for species and preservation of biodiversity, and production of food and raw materials. Some of these services affect humans directly as in the cases of food and raw material production, and these direct services can be either market or nonmarket services. Other services affect people only indirectly as in the cases of nutrient recycling and the regeneration of soil. In these cases, it is necessary to establish the link between the ecosystem service being valued and the channel through which it affects people. More will be said on this point later.

The Millennium Ecosystem Assessment (2003, 8) added to this conceptual framework by identifying four categories of service streams: provisioning, regulating, supporting, and cultural. Provisioning services are the flows of goods such as food, fiber, fuels, and so forth that stem from the primary and secondary productivity of ecological systems. These service streams are usually easily identified and the most apt to be in the form of private goods and to be governed by market transactions. Regulating services are the benefits people obtain from the regulation of ecosystem processes such as the maintenance of air quality and climate regulation. Supporting services are those that are necessary for all other ecosystem services. Supporting services provide the tools (for example, oxygen and nutrients), while regulating services do the work (atmospheric regulation). Supporting services are sometimes called "ecosystem functions." Finally, cultural

services provide benefits through recreation and other highly subjective activities like spiritual enrichment and aesthetic experience.

A careful inspection of this list and these definitions shows a strong possibility of double counting of service flows, especially between some of the supporting services and the provisioning services. For example, waste decomposition is the source of the nutrients that support the bottom of the food chain and whose value is eventually embodied in the value of the plant and animal species that are harvested for human consumption. Similarly, the value of pollination services is eventually embodied in the value of harvested plant crops. In recognition of this potential for double counting, Boyd and Banzhaf offered a more limited definition:

> Ecosystem services are components of nature, directly enjoyed, consumed, or used to yield human well-being.
>
> (Boyd and Banzhaf 2007, 619)

For example, while many economists recognize nature-based outdoor recreation as an ecosystem service, Boyd and Banzhaf pointed out that what we count as recreation (a period of time spent at a site engaged in some activity) involves the combining of human inputs (time and perhaps equipment) with inputs from the ecosystem—for example, populations of flora and fauna that add value to the experience. Not all economists accept this narrow approach to defining ecosystem services. See, for example, Polasky and Segerson (2009, 412) for further discussion.

This emphasis on final *versus* intermediate services also calls attention to another issue to be addressed in the identification and valuation of ecosystem services. Most of the provisioning and cultural services identified by the Millennium Ecosystem Assessment are actually produced by combining ecological outputs with human labor and capital. The costs of these nonecological inputs must be deducted from the value of the food, fiber, fuels, and recreation in determining the value of the ecosystem service itself. This deduction is necessary in arriving at the *value added* by the ecosystem service. Thus, the value of the ecosystem service is essentially the net economic rent attributable to this dimension of nature; and those economists estimating the values of fish and forest harvests have been engaged in valuing one type of ecosystem service.

Valuing Service Flows

Given the anthropocentric definition of ecosystem services, the economic concepts of willingness to pay and willingness to accept compensation provide the conceptual basis for defining the economic values of these services; and the methods and models described in this book provide the means for estimating these values, at least in principle. Estimating the economic value of an ecosystem service involves three steps (see for example, Barbier 2011, 33). The first is determining the nature and size of the environmental change affecting ecosystem structure and function. The change could be in the spatial area of a particular type of

habitat, for example, freshwater wetlands, in the populations of species present, or in the fluxes of energy or nutrients through the system. The second step involves determining how these changes affect the quantities and qualities of ecosystem service flows to people. The third step involves using existing economic methods where available to assess the changes in people's well-being, as measured in dollars.

When an ecosystem service supports the production of a marketed commodity, the value of a change in that service is the sum of the changes in consumers' and producers' surpluses in that market. For example, an increase in the population of a pollinating insect could increase the output of agricultural crops, resulting in lower prices to consumers and/or greater quasi-rents to producers. For example, Ricketts et al. (2004) examined the economic value of the pollination services provided by feral bees to coffee planters in Costa Rica. They measured the variation in yield in areas adjacent to, proximate to, and distant from patches of forest that provided habitat for pollinating bees. The increased yield valued at the price received by planters provided an estimate of the added producers' surplus generated by the free pollination service of the bees.

Tidal wetlands are known to shelter the young of commercially valuable fish species. Changes in the area of wetlands have been related to changes in commercial harvest of blue crabs and finfish (Lynne, Conroy, and Prochaska 1981; Bell 1989). Barbier, Strand, and Sathirathai (2002) found a similar relationship between areas of mangroves and fisheries harvests in Thailand. The methods described in Chapter 8 are available for this type of ecosystem service.

When ecosystem functions support nonmarket environmental services, we may be able to draw on the tool kit of nonmarket valuation methods to determine the economic values of changes in these service flows. For example, when a change in an ecosystem service results in an improvement in the quality of outdoor recreational experiences, recreational demand models can be used to estimate the value of the service flow (see for example, Adamowicz et al. 2011), and stated preference methods might be used to value aesthetic services.

In order to estimate the economic value of a basic ecosystem function, we need to know the link between that function and the ecosystem service flows that it supports, which will not always be easy to uncover. One approach to establishing this link is to think of the relevant components of the ecosystem as being involved in a production process. For discussions of the production function approach, see Barbier (1994, 2000, and 2011, 53–54). Under this approach, the ecosystem is assumed to be an equilibrium system that can be subjected to comparative static analysis to determine changes in service flows in response to changes in ecosystem conditions. Crocker and Tschirhart (1992) developed a formal model of an ecosystem and economic system to demonstrate this approach. More recently, researchers have been developing empirically based integrated assessment models of spatially explicit multiple ecosystem services to investigate the effects of land use changes and policy interventions on the flows and values of ecosystem services. For examples, see Kareiva et al. (2011).

One complication in developing such models is that in terms of production theory ecosystems are multiproduct production systems in which jointness in production is likely to be a dominant feature. For example, a species of bird might be valued both for its pollination of a commercial fruit species and for its control of insects that damage some other commercially valuable plant. The value of the bird species is the sum of the values of all of its services. However, the jointness in production must be taken into account when estimating the values of individual service flows (see Chapter 8). Complex spatially explicit models are being developed to generate predictions of the supplies of a variety of ecosystem services in response to changes in such things as land use management plans. For an example, see Polasky et al. (2011).

Another complication is that the responses of ecosystems to perturbations might display nonlinearities, discontinuities, multiple end points, and even chaotic behavior, especially for changes in the populations of species and fluxes of energy or nutrients (Levin and Pacala 2003; Dasgupta and Mäler 2004). In fact some aspects of ecosystem behavior might be fundamentally unpredictable (Huisman and Weissing 2001). See also Botkin (2012) who challenged the whole notion of stability and equilibrium of ecosystems generally. For these reasons, economists may have more success in estimating the values of changes in the spatial extent of an ecosystem than changes in other characteristics of the system.

A recent and ambitious example of incorporating the value of ecosystem service flows into land use alternatives demonstrates both the complexity of doing so and the potential policy importance that such an effort may produce. Bateman et al. (2013) used a set of spatially explicit biophysical models for the United Kingdom in conjunction with nonmarket values estimated for recreation, open space, carbon emissions and sequestration, and wildlife diversity to evaluate economically optimal land use. Their findings make clear that using market prices alone can lead to suboptimal decision making and that the tools to evaluate a wide range of ecosystem services at a relatively fine spatial scale are available.

It is sometimes suggested that the cost of replacing a function of an ecological system with a human engineered system can be used as a measure of the economic value of the function itself. Replacement cost can be a valid measure of economic value only if three conditions are met: the human-engineered system must provide services of equivalent quality and magnitude, the human-engineered system must be the least costly alternative, and individuals in aggregate must, in fact, be willing to incur these costs if the natural service were not available (Shabman and Batie 1978). Note that when these conditions are not met, there is no presumption that replacement cost is either an overestimate or an underestimate of true economic value—all that can be said is that the two numbers are measures of different things.

In a classic example of the replacement cost approach, Gosselink, Odum, and Pope (1974) used an estimate of the cost of a tertiary sewage treatment as the economic value of the nutrient removal function of a tidal wetland. In another widely cited article, Chichilnisky and Heal (1998) used this method to estimate the

value of water purification service provided by a large protected watershed area in the Catskill Mountains of New York to be $6–8 billion. Chichilnisky and Heal argued that in the Catskill Mountain case, these conditions were met, since the City of New York would have been forced to spend $6–8 billion building a water filtration plant for the city water supply if it did not invest in watershed protection. However, the philosopher Mark Sagoff (2005) convincingly argued that the record does not support the assumption that the $6–8 billion filtration plant was actually required.

Valuing the Wealth of Nature

The value of that part of the natural world that contributes to human well-being or welfare through the provision of ecological goods and services can be called the wealth of nature, or the value of natural capital (Costanza et al. 1997). A large-scale collaborative effort to operationalize the concept of natural capital is the Natural Capital Project at Stanford University (Kareiva et al. 2011, 37).[1] There are two ways to think about how to measure the value of the stock of natural capital. The first is to think of the value of this stock as the discounted present value of the economic values of all of the streams of service flows from all of the ecosystems of the world. The problem is how to measure the total value of each service flow—that is, the value of moving from a zero flow to the current set of flows.

The second way to think about the value of the stock of natural capital is borrowed from the way the Bureau of Economic Analysis in the U.S. Department of Commerce develops estimates of the tangible wealth of the U.S. economy (for example, Katz and Herman 1997). These estimates are based, in principle, on counts of the numbers of each type of productive asset and a set of unit values or "prices" for each asset that are assumed to be constant. The measure of wealth is the summation of the price times quantity calculations for each asset type. It has long been understood that the result of the price times quantity calculation does not represent the total value of the stock of wealth any more than gross domestic product (also a price times quantity measure) represents the value that people place on the nation's output. Both calculations use marginal values or unit prices and have no way to capture the value of inframarginal units.

Behavioral Economics

The term behavioral economics refers to the study of behavior that is inconsistent with the standard economic model of rational choice with stable and purely self-interested preferences, and to efforts to explain these anomalies with findings from other fields, especially psychology and sociology. For an overview of this

1 Its web page states, "Our mission is to align economic forces with conservation by mainstreaming natural capital into decisions." See: http://naturalcapitalproject.org/home04.html.

relatively new field, see Camerer, Loewenstein, and Rabin (2004). Shogren and Taylor (2008) provided a cogent discussion of the ways in which behavioral economics ideas relate to environmental economics with a particular emphasis on the divergence between willingness to pay and accept.

This section raises the question of the possible implications of behavioral economics for the methods for estimating environmental and resource values that are discussed in this book. Briefly, the methods for estimating individuals' values from observations of revealed or stated choices are all based on models of behavior and choice that start with the assumptions of stable preferences and rational behavior. However, if choices are not consistent with preferences and do not reflect increases in individuals' well-being, this could call into question the values obtained from using these methods.

Examples of the kinds of behavioral anomalies that could affect estimates of values include the following: the endowment effect leading to large differences between WTP and WTA (see Chapter 3) and hyperbolic discounting (see Chapter 6). In addition to these, Shogren and Taylor add coherent arbitrariness (the anchoring of values on arbitrary levels), problems in self-control, and altruistic behavior to the list of deviations from rational choice theory that may have important implications for nonmarket valuation. Another way to categorize the issues is suggested by Kling, Phaneuf, and Zhao (2012) who noted that there are two types of findings from behavioral economics that are relevant to nonmarket valuation: (a) preferences that deviate from neo-classical preferences, and (b) failure of individuals to optimize. The endowment effect, hyperbolic discounting, framing effects, and the issues identified by Shogren and Taylor above largely fit within the first category. An example of a behavioral anomaly consistent with the second is the observation that consumers sometimes appear to hold "mental accounts" where they arbitrarily constrain their spending on goods to stay within predetermined bounds.

In response to the observed anomalies, some analysts have suggested that policy analysts should reject the premise of revealed preference and substitute expert judgments about what will really enhance individuals' well-being when conducting policy evaluations. For example, Bernheim and Rangel (2007, 9) argued that "if one knows enough about the nature of decision making malfunctions, it may be possible to recover tastes by relying on a selective application of the revealed preference principle." Kerry Smith (2007, 154), after quoting from Bernheim and Rangel, countered with, "Unfortunately we cannot resolve the problems that these authors suggest arise when people do not make what they (the authors) think is the appropriate choice. There is one small detail. Who decides what is a *suitably defined* welfare improvement—elected officials, policymakers, experts?" He goes on to say, "The skeptic should ask who decides what choices are mistakes and when enough are made so that the evidence compels a conclusion that preferences in these dimensions are incoherent" (2007, 156). There is now a growing literature dealing with these differing perspectives. In addition to Bernheim and Rangel and Smith, see, for example, Robinson and Hammitt (2011), Smith and Moore (2010),

and Hammitt (2013), as well as references therein. Some authors point to the fact that observed behavioral anomalies often appear to depend on the context in which the choices are observed and the constraints that people face (for example, Smith and Moore, 2010); and this raises questions about their significance for environmental valuation.

There is no doubt that research in behavioral economics has raised important questions for nonmarket valuation, and welfare economics in general, and students of nonmarket valuation should remain engaged in this fascinating literature. However, as Shogren and Taylor point out "the evidence from behavioral economics remains insufficient to support the wholesale rejection of rational choice theory" (2008, 41), and until there are compelling alternatives, benefit-cost analysis will continue to rely on its neoclassical roots (see the work of Bernheim and Rangel 2009, for one avenue). As the literature continues to develop, welfare analysts should recognize that in addition to challenges, there are likely to be many useful insights from behavioral economics findings that can strengthen the methods used to elicit preference information (from stated preference surveys for example) and for the interpretation of our data (comparisons between revealed and stated preference information for example).

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Chapter 14

Conclusions

State of the Art and Research Needs

The State of the Art

In the concluding chapter of *The Benefits of Environmental Improvement: Theory and Practice*, Freeman claimed:

> Suppose the administrator of EPA wished to know the magnitude of the benefits accruing from a given pollution control policy. If asked, I believe an economist could specify the economic theory and models he would use, the data he would like to have, and the empirical techniques he would apply to the data to obtain measures of benefits.
>
> (Freeman 1979, 248)

Freeman added five qualifications to this optimistic assessment of the state of the art. Briefly, they were:

1. Where RP methods were not available (for example, for measuring non-use values), the economist would have to resort to SP measures. He was concerned about "the accuracy of responses of individuals in the necessarily hypothetical situations they pose" (Freeman 1979, 249).
2. In the valuation of reductions in mortality risks, there did not appear to be broad acceptance of the idea that individuals' behavior could or should be the basis of welfare measures.
3. Because all welfare measures are conditioned upon the existing distribution of income, to accept these welfare measures for making public policy choices involved accepting the existing distribution as satisfactory.
4. Economic measures of welfare change "must be built upon noneconomic data such as dose–response functions" (Freeman 1979, 250), which often were not available.
5. The data required to implement many of the models and methods would often be difficult and costly to obtain.

In looking back over this list of qualifications nearly 35 years later, there seems to be much less reason to be concerned about the first two of them. Regarding the first qualification, substantial progress has been made in the development of SP methods. We can now be more optimistic about the ability of properly framed questions to generate meaningful data on preferences and values. This does not mean that all the investigator has to do is go out and ask people some questions. However, well-designed SP studies have a place in economists' toolboxes. As to the second qualification, although there is still some debate about it, there is now fairly wide acceptance of the use of RP measures of the value of risk reduction for evaluating public policies that affect health and safety. Such measures are sanctioned, for example, by the U.S. Environmental Protection Agency for use in performing regulatory impact analyses (U.S. EPA 2000). As noted in Chapters 8 and 11, there is a substantial body of evidence on the magnitude of risk premiums revealed in various markets. Rather, the terms of the debate have shifted; the debate now revolves around whether, in valuing risk reduction policies, differences in age, income, health status, cause of death, and the level of baseline risk should be taken into account, and if so, how. The third qualification remains true (as it does for any economic analysis). However, the welfare measures themselves can be used to assess the distributional impacts of any given policy.

Having said this, however, there is also a new qualification to be added to the list. It has to do with the process of fitting data to a model to generate a welfare measure and the effects of the choice of a model and functional form on the welfare measure. There has been a virtual explosion in the number of models available to be exploited, and a number of rich data sets have been created. Several studies show that welfare measures can be sensitive to the choice of a model or the choice of a functional form for a specific model. Some of these studies involve simulations in which the "true" welfare measure is known and can be compared with estimates derived by fitting the simulated data to alternative models. For examples, see Cropper et al. (1988, 1993), Kuminoff, Parmeter, and Pope (2010) and Kling (1988, 1989). Other studies have involved examining the sensitivity of welfare estimates to changes in some aspect of the specification of the model being applied to real data; for an example, see Cooper and Loomis (1992) and von Haefen and Phaneuf (2003). Because the true model cannot be known, we must add model uncertainty to the list of sources of uncertainty in welfare measures. An important area for future research is to obtain a better understanding of the sources and properties of model uncertainty.

The first chapter of this book discussed several ways of classifying the types of environmental and resource service flows for which value measures might be desired. The book concludes here by briefly outlining the types of models that can be applied to value each type of service flow. This information is summarized in Table 14.1. The first thing to note is that SP models apply to the estimation of all types of service flows except those that work through changes in market prices (producers' and consumers' surpluses). SP methods are all that is available for estimating nonuse values.

Table 14.1 Models for estimating the values of environmental services

	Type of Model						
Type of environmental service flow	*Averting Behavior (Chapters 4 and 7)*	*Health production function (Chapter 7)*	*Changes in producers' and consumers' surpluses (Chapter 8)*	*Recreation Demand Models (Chapter 9)*	*Property Value Models (Chapter 10)*	*Hedonic Wage Models (Chapter 11)*	*Stated Preferences (Chapter 12)*
Human health:							
Mortality	X				X	X	X
Chronic morbidity	X				X		X
Acute morbidity	X	X			X		X
Other direct impacts on humans (amenities, visibility, noise)	X				X	X	X
Economic productivity of ecological systems			X				
Other ecological services (recreation)				X			X
Effects on nonliving systems, such as materials	X		X		X		X
Nonuse values such as ecological stability and biodiversity							X

For valuing mortality risks, some version of the hedonic wage model is probably the method of choice, although averting behavior and SP studies have been conducted in this area with some success. Some form of averting behavior model would be most appropriate for valuing the nonfatal health effects as well. Even the health production function model can be considered a special case of the averting behavior models in that it is based on substitutability among market and nonmarket inputs into the production of health.

Hedonic property value models capture the value of differences in amenity and disamenity levels across residential locations, but they might also measure other environmental effects caused by the same environmental agent. For example, air pollution may reduce visibility and cause adverse health effects and damages to household materials, all of which would be reflected in property prices. Hedonic wage models can be used to value differences in the average levels of amenities or disamenities among cities or regions.

Human exploitation of natural and managed ecosystems in agriculture, commercial fisheries, and forestry is largely managed through market mechanisms. Thus, the values of changes in these ecological services will be reflected in changes in prices and incomes. The cost function and production function models described in Chapter 7 provide the basis for estimating the resulting changes in producers' and consumers' surpluses. However, these models are also applicable where changes in other environmental services affect the productivity and costs of firms. Examples include the effects of water quality on processing costs and the effects of air pollution on manufacturing costs and on the costs of repair and maintenance of structures.

Finally, humans use both natural and managed ecosystems for various forms of recreation. A suite of recreation demand models, detailed in Chapter 9, are available to exploit information on the implicit price of access and on the cost of traveling to a site to estimate the values of sites and changes in their characteristics.

Research Needs

In many ways, both the theory and the empirical methods used in measuring environmental and resource values have matured significantly since the first edition of this book. Stated preference techniques have undergone rigorous evaluation and evolution and have become an essential tool in the area. Recreation demand models have evolved from simple site selection and participation models to complex representations of consumer preferences often requiring elaborate econometric techniques to estimate them. Property and wage models have seen both advances in the underlying theory used to model quality differentiated products, such as those emerging from the equilibrium sorting literature, and a rapid growth in the data available for use in estimation. Yet, there remain numerous areas in which additional research is needed. In this last section of the book, we seek to identify a few of these areas, in the hopes of spurring further research into the field. The list, of course, is not exhaustive, but rather suggestive and in no particular order of importance.

Behavioral Economics – Theory and Implementation

Documenting and understanding departures from neoclassical theory, both in the lab and in the field, remains an active area of research. The implications of behavioral economics for economics broadly, and for welfare analysis in particular, is still unclear; but work in this area should and will undoubtedly continue. Of particular interest from the perspective of this book is the conceptual work on behavioral welfare analysis by Bernheim and Rangel (2009), and more recently Fleurbaey and Schokkaert (2013), with an accompanying need to develop practical approaches for implementing the welfare metrics being proposed.

Modeling Choice Under Uncertainty and Risk

Many of the models used in nonmarket valuation are based on the Random Utility Maximization (RUM) framework. An underlying assumption of this model is that the decision-maker *knows* the conditional utility associated with a given choice alternative. If there is risk (i.e., a variety of possible states of the world, each with a known subjective probability of occurring), the structure implicitly assumes that the individual is able to integrate out that risk to form an aggregate value for each available choice alternative. This can be done on the basis of expected utility (EU) theory or some other non-EU model (such as loss aversion). The difficulty here is in choosing or discerning which aggregate value and decision rule is being used by a survey respondent and eliciting relevant information that they use in making that decision, including what they perceive to be possible states of the world and what subjective probabilities they associate with those states. The challenges become even more daunting in the setting of uncertainty (or ambiguity), when the decision-maker is no longer assumed to know subjective probabilities for the different states of the world. It is clear that more research is needed into understanding the decision-making process under both risk and uncertainty and in developing practical methods for gathering the requisite information from decision-makers. De Palma et al. (2008) provide an excellent introduction to these issues.

Stated Preference Surveys and the Role of Consequentiality

As noted in Chapter 12, there has been a paradigm shift in evaluating stated preference approaches to nonmarket valuation. Whereas much of the research to date assessing the criterion validity of SP methods has compared purely hypothetical surveys with real transactions in experimental settings, Carson and Groves (2007) emphasized the need instead to understand and evaluate the underlying incentives provided by the survey instrument to induce preference revelation. An important component in making a survey incentive compatible is making it consequential from the perspective of the survey respondent. To date, relatively few studies have tested the role of consequentiality in SP survey responses and none that we are

aware of have done so using the increasingly popular choice experiment format. Moreover, additional research is needed into how best to convey consequentiality and assess the efficacy of a particular consequentiality script. More research is also needed to understand the role of incentive compatibility in the choice experiment framework.

Validity Tests of Revealed Preference Methods

Kling, Phaneuf, and Zhao (2012) suggest that much could be learned by applying to revealed preference methods the same type of validity tests that stated preference methods have been subjected to. With the exception of Bishop and Heberlein's (1979) early validity tests using goose hunting permits (where recreation demand estimates were included with contingent valuation estimates to compare with actual transactions), almost all such tests have focused exclusively on stated preference methods. Given the many sources of potential error in hedonic methods, recreation demand models, averting behavior estimates, and other revealed preference models, it is somewhat surprising that so little attention has been paid to this question.

Defining the Choice Set

In modeling the demand for recreation, analysts typically assume that the choice set being considered by the individual is either the complete choice set available or one limited in a consistent fashion across individuals (e.g., alternatives within a fixed distance of the individuals' home). However, knowledge of the full choice set is likely to differ across individuals (for example, due to their time living in a given region of the country) and to evolve endogenously over time as they seek to learn more about the available alternatives. One strand of research in this area seeks to distinguish the full set of alternatives from the options the decision-makers seriously consider (that is, the so-called "consideration set"). In some cases, authors have tried to elicit the consideration set from survey respondents (see Hicks and Strand 2000), while others have treated an alternative's inclusion in the consideration set as a latent variable to be estimated econometrically (see Ben-Akiva and Boccara 1995; Haab and Hicks 1997; von Haefen 2008). While this represents a potentially useful line of research, there are concerns regarding the feasibility of extracting the desired choice set through a survey instrument (see, for example, Horowitz and Louviere 1995). At the same time, econometrically characterizing the choice set without such information relies heavily on distributional assumptions in order to identify the consideration set. More research and data development are clearly needed in order to better understand the dynamics underlying the formation of the relevant choice set. Analogous problems exist in transportation, marketing, and labor market research—areas that represent potentially fruitful literatures for the cross-fertilization of ideas. Finally, it should be noted that the problem of identifying the choice set is not restricted to recreation demand models; it is an

issue for property value models (Banzhaf and Smith 2007) and in modeling stated preference survey responses.

The Opportunity Cost of Time

In inferring the value associated with recreational sites and changes in the associated amenities, one of the key elements is the opportunity cost of time. As noted in Chapter 9, this topic has received considerable attention in the literature, with a variety of approaches developed to account for differences in the constraints and opportunities facing an individual. Nonetheless, this an area where additional research would seem worthwhile given the pivotal role it plays in recreation demand modeling. Palmquist, Phaneuf, and Smith (2010), for example, consider the fact that leisure time is often broken down into component parts that cannot easily be combined to create the large blocks of time needed for certain types of recreation. This in turn has implications for the opportunity cost of time. Essentially, rather than facing a single time constraint, households face a series of constraints, with elemental components of time that are not readily substitutable for one another.

Sustainability and Resilience

The concepts of sustainability and its cousin, resilience, are quickly gaining a foothold in many government agencies and private businesses as organizing principles for planning. A commonly cited working definition of sustainability in the context of development comes from the Brundtland Report (World Commission on Environment and Development, 1987): "sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs." Others discuss sustainability in the context of three pillars: economic, environmental, and social. The term resilience has not received as much attention to date, but is increasingly discussed as a beneficial characteristic of a system. As economists, a focus on the sustainability and/or resilience of a system seems quite incomplete without considering them in the context of a broader objective addressing overall social welfare. Research on the role of sustainability and/or resilience in evaluating the optimality of a development path or use of the environment is needed. The tools of nonmarket valuation should be able to provide insight on possible tradeoffs between sustainability, resilience, and conventional social welfare optimization. Further, the tools of nonmarket valuation may prove useful in constructing metrics of sustainability of resilience.

Valuing Risk Reduction (Mortality/Morbidity)

As noted in Chapters 7 and 11, significant work is needed to improve and update the values of reducing risks that lead to morbidity and/or mortality. The extremely

old studies being relied upon for many U.S. government benefit-cost analyses of regulations are sadly out of date and reflect risk and population characteristics that are often not policy relevant. Further, it is now generally appreciated that the value of risk reduction will be a function of many features of the type of risk faced and the populations subjected to that risk. Finally, the puzzle concerning why stated preference estimates of risk reduction lie almost universally below the revealed preference estimates is sorely in need of study.

Ecosystem Services

As noted in Chapter 13, the concept of ecosystem services has gained ground over the past decade. The notion that the environment and ecosystems provide services to human beings is compelling and clear from a general perspective, but clearly identifying the full suite of services provided by a particular environmental change or alteration of an ecosystem remains elusive. This is an area where economists must work closely with ecologists to identify the services and develop methods to measures the services in ways that are meaningful to human usage and values.

General Equilibrium

In nonmarket valuation, and in environmental economics more generally, there has been increased interest in the general equilibrium effects stemming from environmental changes (driven either by policy or by other forces). This has clearly been the case in the property valuation literature, with the emergence of the equilibrium sorting literature (see Kuminoff, Smith, and Timmins, 2013), where the models seek to characterize the equilibrium process in a housing market given a shock to the system. These equilibrating factors need not be restricted to market forces, but can also include feedback effects from environmental and biological factors, such as fishing stocks or habitat conditions (see, for example, Phaneuf, Carbone, and Herriges 2009, and Smith et al. 2010). Further research along these lines would seem beneficial, both because policy programs often induce feedback effects that cannot be adequately captured by marginal analysis and because these effects often require an understanding of the physical processes impacted by large-scale policy initiatives. Incorporating both wage and housing market equilibrium processes may be an important area for such research—see Kuminoff (2012).

Combining Revealed and Stated Preference Data

As noted in Chapter 13, a large and growing literature has been developing in which economists combine revealed and stated preference information to jointly estimate the parameters of a common utility or demand function. Whitehead et al. (2008) provide an excellent review of this work. One area that seems to us to be in need of additional work is the combination of stated and revealed preference data to estimate nonmarket values when the property of weak complementarity

cannot be reasonably assumed to hold. As discussed in Herriges, Kling, and Phaneuf (2004), when the property of weak complementarity does not hold, the traditional approach of computing use value from revealed preference data and identifying it as a lower bound cannot be assumed to be accurate. However, including appropriately designed stated preference data has the potential for providing the missing components of information (Ebert 1998; Eom and Larson 2006). Returns to research in this area could be quite substantive.

Valuation in Developing Countries

Much, though not all, of the valuation literature has evolved in industrialized countries, where market economies are the norm. Yet there are many environmental issues that impact developing countries, such as those stemming from climate change, and an understanding of the costs and benefits of alternative initiatives would facilitate policy design. Adapting both revealed and stated preference methods to these settings will require understanding how the local economy and social structures alter both individuals' incentives and their perceptions regarding the tradeoffs available to them.

Dynamic Welfare Measures

There are several places in the book where dynamic aspects of welfare measurement are referred to, including the theoretical basis for welfare measures that incorporate uncertainty and learning in Chapter 5, empirical recreation models that attempt to incorporate dynamic behavior in Chapter 9, and equilibrium sorting models in housing and wage markets that recognize to some degree a temporal response to a change in environmental conditions. In many cases, modeling and addressing the dynamic nature of welfare measures may be unnecessary as when it can reasonably be expected that markets are in long-run equilibrium. However, when working with individual observation data such as typically collected in stated preference surveys or recreation demand data sets, there may be elements of dynamic behavior that are important to understand and identify with respect to policy. It will be necessary to carefully design data collection efforts to study the dynamic components of welfare measures, but the potential for improved understanding of fundamental valuation puzzles such as the willingness-to-pay and willingness-to-accept disparity seems high (Kling, List, and Zhao 2013).

Valuing Ecosystem and Other Environmental Effects of Climate Change

Climate change and its impacts on ecosystem services, invasive species, biodiversity, and a host of other ecological endpoints are poorly understood in the scientific community. This makes nonmarket valuation of these effects even harder.

Economic models that assess welfare losses from climate change do not include more than a cursory assessment of nonmarket values (Nordhaus 2008; Stern 2009). Yet, without such measures, assessment of the impacts of climate change on the welfare of current and future generations cannot be fully understood. While daunting, the returns to research that better quantifies the welfare losses from climate change via its impact on the environmental are likely to be high.

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ECOLOGY

Spatial models reveal the microclimatic buffering capacity of old-growth forests

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Sarah J. K. Frey,[1]* Adam S. Hadley,[1] Sherri L. Johnson,[2] Mark Schulze,[1] Julia A. Jones,[3] Matthew G. Betts[1]*

Climate change is predicted to cause widespread declines in biodiversity, but these predictions are derived from coarse-resolution climate models applied at global scales. Such models lack the capacity to incorporate microclimate variability, which is critical to biodiversity microrefugia. In forested montane regions, microclimate is thought to be influenced by combined effects of elevation, microtopography, and vegetation, but their relative effects at fine spatial scales are poorly known. We used boosted regression trees to model the spatial distribution of fine-scale, under-canopy air temperatures in mountainous terrain. Spatial models predicted observed independent test data well (r = 0.87). As expected, elevation strongly predicted temperatures, but vegetation and microtopography also exerted critical effects. Old-growth vegetation characteristics, measured using LiDAR (light detection and ranging), appeared to have an insulating effect; maximum spring monthly temperatures decreased by 2.5°C across the observed gradient in old-growth structure. These cooling effects across a gradient in forest structure are of similar magnitude to 50-year forecasts of the Intergovernmental Panel on Climate Change and therefore have the potential to mitigate climate warming at local scales. Management strategies to conserve old-growth characteristics and to curb current rates of primary forest loss could maintain microrefugia, enhancing biodiversity persistence in mountainous systems under climate warming.

INTRODUCTION

Macroscale climate patterns are well known to influence range-wide suitability for biota. However, local-scale climate (hereafter microclimate) is often most relevant to animal behavior and demography (*1*). Reconciling this mismatch between global climate models and the scale at which organisms experience their environment should therefore improve our understanding of biodiversity responses to climate change (*2*, *3*). Furthermore, in heterogeneous mountain landscapes with complex thermal regimes (*4*), climate-sensitive species have the potential to disperse to, and persist in, favorable microclimatic conditions (*5*). Coarse-scale climate data are not as accurate for predicting trends in mountains, influencing our ability to assess climate impacts (*6*). Identification of factors that generate particular microclimates will help focus conservation efforts to lessen the impacts of climate change on biodiversity (*7*), which are expected to be particularly substantial in mountainous regions (*8*). However, to date, the coarse resolution of most land cover and climate data has precluded such analysis (*9*).

The decoupling of the surface temperature conditions from those of the troposphere is commonly attributed to two main factors in mountainous areas: (i) local air-flow dynamics, such as cold air drainage and pooling, and (ii) variations in slope and aspect (microtopography) (*7*). However, vegetation also has the potential to influence microclimatic patterns via its effects on solar radiation, wind exposure, interception of precipitation, and retention of understory humidity. Indeed, the influence of vegetation on microclimate has long been recognized (*10*, *11*) and is the reason why long-term weather stations are situated in open areas. Unfortunately, such sampling strategies have precluded inferences about the relative influences of microtopography and vegetation structure on mediating microclimate (*2*).

If particular vegetation structural characteristics can abate the effects of regional climate change (*12*), land management has the potential to either amplify or buffer these effects on biodiversity (*13*). Given the rapid global changes in land use (*14*), it is critical to understand the degree to which management influences microclimate. An increasing proportion of the world's forests are secondary, transformed forests (*14*, *15*). Therefore, it is essential to understand the implications of secondary forests for climate and biodiversity. Here, we examine whether the structural characteristics present in old-growth forests (for example, heterogeneous canopies, high biomass, and complex vertical structure) increase site-scale thermal buffering capacity over more structurally simple, but mature plantation forest stands. The substantial biomass associated with western old-growth forests might be expected to result in slower rates of warming during summer months (*16*). Alternatively, the closed and homogeneous canopy conditions of old (>50-year-old) forest plantations could prevent rapid site-level warming through reduced solar radiation, thereby moderating climate (*17*). Although previous work has examined the effects of substantial differences in canopy cover on microclimate (*4*, *18*–*22*), to our knowledge, ours is the first broad-scale test of whether subtle changes in forest structure due to differing management practices influence forest temperature regimes (Fig. 1). Given that old-growth forests continue to decline globally (*23*) and that plantations continue to proliferate (*24*), understanding microclimatic impacts is of great conservation importance.

In 2012 and 2013, we collected understory air temperatures at high spatial resolution across a complex mountainous landscape at the H. J. Andrews Experimental Forest (HJA) in Oregon, USA. We obtained fine-resolution (5 m) data on topography and vegetation structure using LiDAR (light detection and ranging). We then used machine-learning techniques [that is, boosted regression trees (BRTs)] to map predicted

[1]Forest Biodiversity Research Network, Department of Forest Ecosystems and Society, Oregon State University, Corvallis, OR 97331, USA. [2]U.S. Forest Service, Pacific Northwest Research Station, Corvallis, OR 97331, USA. [3]Geography, College of Earth, Ocean, and Atmospheric Sciences (CEOAS), Oregon State University, Corvallis, OR 97331, USA.
*Corresponding author. E-mail: sarah.frey@oregonstate.edu, sjkfrey@gmail.com (S.J.K.F.); matt.betts@oregonstate.edu (M.G.B.)

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Fig. 1. The high biomass, tall canopies, and vertical structure of old-growth forests are associated with lower spring maximum temperatures than in mature plantations. Photos of old-growth (**A**) and mature plantation (**B**) forest stands at the H. J. Andrews Experimental Forest (HJA) in Oregon, USA. [photo credit: Matthew Betts, Oregon State University].

thermal properties at the landscape scale and to test the hypothesis that vegetation structure mediates under-canopy microclimates.

RESULTS

All models performed well when tested on independent data (table S1). The cross-validation correlations were high, showing substantial congruence between training and test data [mean ± SD (range): 2012, r = 0.87 ± 0.09 (0.69 to 0.98); 2013, r = 0.84 ± 0.11 (0.64 to 0.96)]. This performance was not caused by spatial dependency in the data (table S1). These results indicate that BRTs, which are now used extensively in species distribution models (SDMs) (*25*), seem to offer a powerful new approach to examining spatial distributions in abiotic conditions. As in SDM applications, the advantage of such machine-learning methods lies in their capacity to incorporate many independent variables and their flexibility to include nonlinearities and variable interactions. Although parametric alternatives are available (for example, generalized linear models), our results indicate that BRTs also represent a promising option for distinguishing the relative importance (RI) of complex climate drivers and for generating detailed spatial climate predictions (fig. S1).

Elevation was the dominant predictor for most of the temperature metrics (Fig. 2), including cumulative degree days (CDDs); monthly minimum, maximum, and mean temperatures during spring-summer (April to June); and minimum temperature of the coldest month (Fig. 3A). High-elevation sites were generally cooler and had fewer CDDs. Microtopographic features showed high RI for predicting CDDs during winter-spring (January to March; RI 2012: 45.9%; RI 2013: 53.9%) and maximum temperature of the warmest month (RI 2012: 37.9%; RI 2013: 32.2%; Fig. 3C). Steeper, exposed, south-facing sites were generally warmer and more variable, and accumulated more CDDs (fig. S2).

Among all temperature metrics, maximum temperature of the warmest month (RI 2012: 35.5%; RI 2013: 39.8%) and variability in weekly spring-summer temperature (RI 2012: 36.7%; RI 2013: 10.2%) were most strongly influenced by vegetation structure (that is, canopy height, biomass, understory cover, and vertical structure; Fig. 2). Vegetation structure also had a strong effect on mean monthly maximum temperature from April to June (RI 2012: 20.4%; RI 2013: 18.9%; Fig. 2). Furthermore, vegetation structure was an important predictor for temperature variability (RI 2012: 28.9%; RI 2013: 31.6%) and CDDs during winter (RI 2012: 31.2%; RI 2013: 34.3%). Sites with old-growth forest traits (for example, taller canopies, higher biomass, and more complex vertical structure) had reduced temperatures and greater temperature stability (fig. S2). Old-forest characteristics, such as taller canopies, more canopy cover >10 m, and biomass >500 Mg/ha, reduced maximum temperature of the warmest month and mean monthly maximum temperature from April to June (fig. S3, A to C). Old-forest traits also had an important influence on climate variability; for example, increased coefficient of variation in canopy height and greater midcanopy cover (2 to 10 m) both reduced variability in mean weekly temperature from January to March (fig. S3, D and E). Areas with the lowest biomass variability (for example, even-aged stands such as plantations) showed more microclimate variability in this winter-spring transition (fig. S3F).

However, topographic position (valleys versus topographically exposed sites) appeared to accentuate the effect of forest structure; exposed sites with low variability in biomass (for example, plantation stands) accumulated the most degree days (fig. S4A). A vegetation-elevation interaction revealed that maximum monthly temperature during spring-summer was lowest at high elevations with high amounts of canopy cover surrounding a site (fig. S4C). In both years and for most temperature metrics, variables at the local scale (25-m radius) had a greater relative influence than variables averaged across a 250-m radius [overall average RI at the local scale (25 m) across all metrics: 2012: 60.7 ± 10%;

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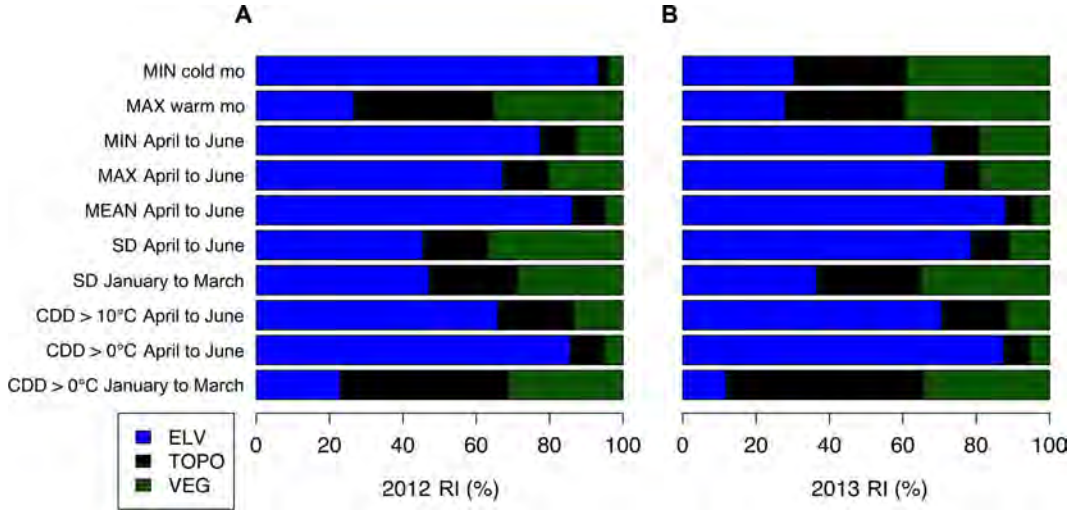

Fig. 2. Relative influence (RI) of variables describing elevation (ELV), microtopography (TOPO), and vegetation structure (VEG) for each temperature metric. RI values for 2012 (**A**) and 2013 (**B**) were derived from the number of times each variable was selected in the process of model building using boosted regression trees (BRTs). Overall, elevation had the strongest influence on air temperature patterns in the HJA, but microtopography and vegetation also exerted important effects, particularly for maximum temperature of the warmest month, variability measures, and cumulative degree days (CDD) in the winter-spring transition.

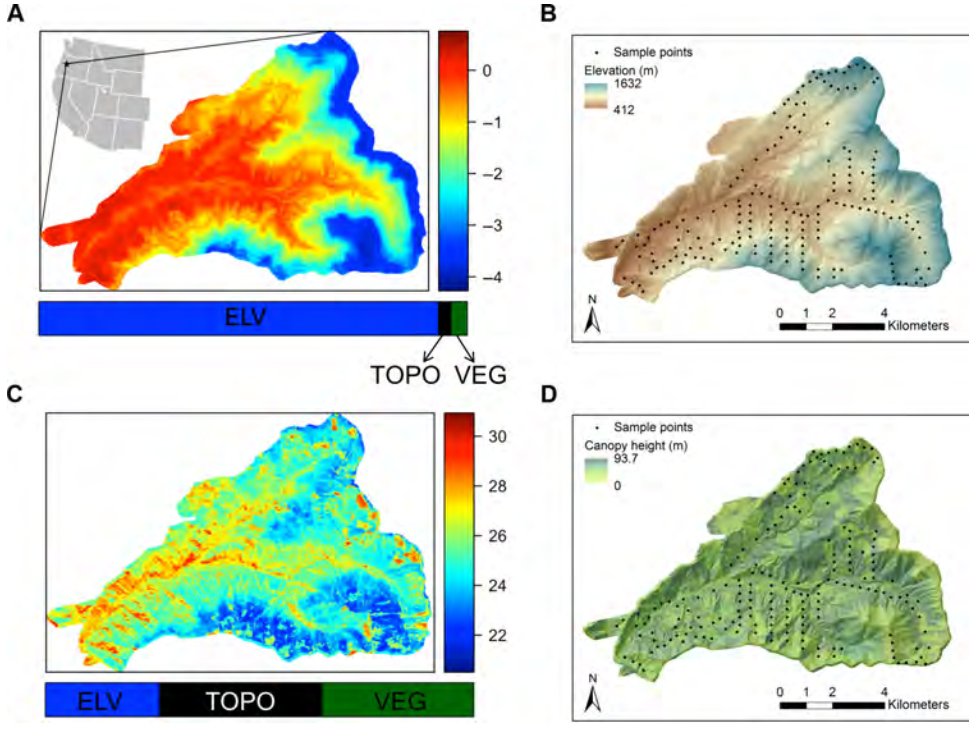

Fig. 3. Spatially predicted maps of minimum temperature of the coldest month and maximum temperature of the warmest month (in degrees Celsius) based on BRT models. Minimum temperatures (**A**) were primarily influenced by elevation (**B**), but maximum temperatures (**C**) were primarily a function of vegetation and microtopography (**D**). Maps of the elevational gradient (B; in meters) and canopy height (D; in meters) based on LiDAR from 2008 at the HJA. Black dots show the 183 temperature sampling locations. The location of the HJA in the western United States is shown in (A).

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2013: 60.4 ± 10.1%]. However, vegetation metrics tended to have more influence at the broader (250-m) spatial scale (fig. S5). Despite interannual differences in the RI of variables, we found remarkable between-year consistency in both predicted and observed under-canopy thermal conditions across sites (7 of 10 variables with $r > 0.9$; table S2 and figs. S6 and S7). This interannual consistency in site-level conditions, which occurred despite substantial differences in annual climatic conditions, lends support to the notion that thermally buffered sites may provide temporally consistent refugia for biodiversity (*7*).

Despite being well suited to prediction given our complexity of input variables, BRTs do not provide information on effect sizes (that is, regression coefficients) (*25*). Therefore, we used principal components analysis (PCA) to integrate vegetation structure variables into a reduced number of components (table S3). The first two principal components of the PCA explained 74.7% of the variability [principal component 1 (PC1) = 44.7%; principal component 2 (PC2) = 30%]. The first component (PC1) strongly reflected a gradient in forest structure from closed-canopy plantations to mature/old-growth forests (Figs. 1 and 4A); this gradient represents the predominant forest types on federal land in the region (*26*). Sites with low PC1 values had less biomass (mean and SD), lower canopies (mean and SD), and less cover (2 to 10 and >10 m) (table S3). The individual LiDAR metrics effectively distinguished between plantation sites and mature/old-growth forest sites (table S4); a discriminant function analysis (*27*) showed that prediction accuracy was 85.3% for plantation sites and 90.4% for mature/old-growth forest sites. Furthermore, our LiDAR metrics were congruent with previously

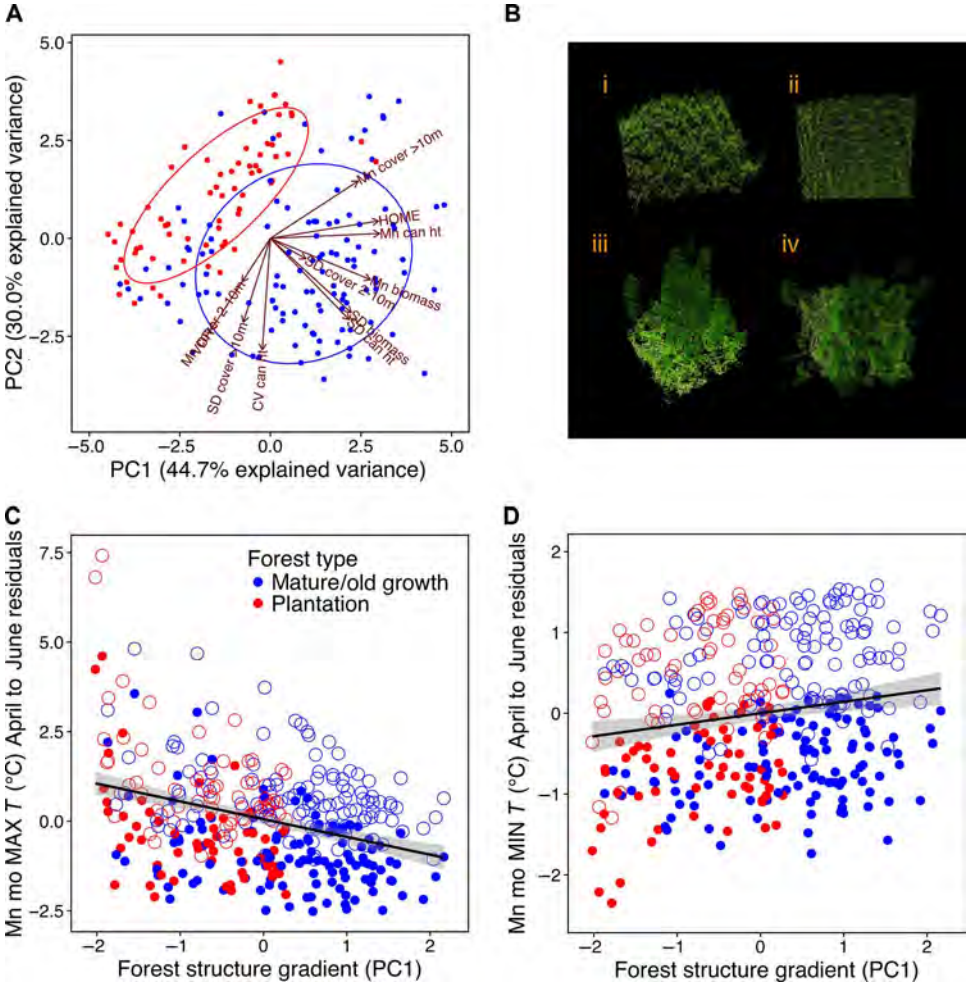

Fig. 4. Differences in microclimate conditions across a gradient in forest structure. (**A**) Principal components analysis (PCA) showing how vegetation structure metrics differ between mature/old-growth forest sites and plantations. The ellipses represent 68% of the data assuming a normal distribution in each category (plantation and mature/old growth). (**B**) Three-dimensional LiDAR-generated images of plantation forests [(i) side view; (ii) overhead view] and old-growth forests [(iii) side view; (iv) overhead view] at the Andrews Forest. (C and D) Results from generalized linear mixed models show the modeled relationship between forest structure [PC1, the first component of a PCA on forest structure variables (A)] and the residuals from an elevation-only model of mean monthly maximum during April to June (**C**) and mean monthly minimum during April to June (**D**) after accounting for the effects of elevation. Closed circles represent 2012 and open circles represent 2013. Maximum monthly temperatures (C) decreased by 2.5°C (95% confidence interval, 1.7° to 3.2°C) and observed minimum temperatures (D) increased by 0.7°C (0.3° to 1.1°C) across the observed structure gradient from plantation to old-growth forest.

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Table 1. Generalized linear mixed model results for the relationship between temperature metrics and the first component of a PCA (PC1) representing a gradient in vegetation structure. Data from 2012 and 2013 were combined and "site" was included as a random effect in all models. Lower PC1 values indicate forest plantations and higher values indicate old-growth forests. "Change in temperature metrics" reports the average difference in temperature (°C) or degree days (dd) across the range of PC1 values. The effect of old-growth structure (PC1) on microclimate was consistent between years for most variables ("No year effects"). Effects of old-growth forests were stronger in 2012, and the direction of old-growth effects remained consistent for all but SD in weekly temperature. We included elevation (ELV) in all models to statistically account for elevation differences. Elevation had a significant effect on all models ($P < 0.0001$), except for SD in weekly temperature from January to March ($P = 0.062$). Coefficients from the interaction models include the 2012 intercept ($\hat{\beta}_0$ 2012), the slope of the 2012 PC1 effect ($\hat{\beta}_1$ PC1 2012), the 2013 intercept presented as the difference from the 2012 intercept ($\Delta\hat{\beta}_0$ 2013), and the 2013 PC1 effect presented as the difference from the 2012 PC1 effect (PC1 $\Delta\hat{\beta}_1$ 2013). *P* values in boldface indicate a statistically significant effect of PC1 on temperature metrics at $P < 0.05$. LCL, lower 95% confidence limit; UCL, upper 95% confidence limit.

Variable	Intercept		PC1			Change in temperature metrics ~ PC1			
	$\hat{\beta}_0$	SE	$\hat{\beta}_1$	SE	*P*	Units	Change	LCL	UCL
No year effects									
CDD > 0°C January to March	178.66	3.66	−0.47	3.92	0.9051	dd	−1.96	−37.01	14.25
CDD > 0°C April to June	820.27	4.17	−10.25	4.47	**0.0229**	dd	−42.89	−82.83	−2.95
Mn mo MEAN *T* April to June	8.98	0.05	−0.11	0.05	**0.0235**	°C	−0.47	−0.92	−0.03
Mn mo MAX *T* April to June	13.68	0.08	−0.59	0.09	**<0.0001**	°C	−2.47	−3.24	−1.70
Mn mo MIN *T* April to June	5.29	0.04	0.16	0.05	**0.0006**	°C	0.68	0.26	1.10
Significant year effects									
	Intercept 2012		PC1 2012			2012 Change in temperature metrics ~ PC1			
	$\hat{\beta}_0$	SE	$\hat{\beta}_1$	SE	*P*	Units	Change	LCL	UCL
CDD > 10°C April to June	115.06	1.73	−7.84	1.85	**<0.0001**	dd	−32.82	−49.36	−16.28
SD wkly *T* January to March	1.62	0.03	−0.07	0.03	**0.0341**	°C	−0.27	−0.55	0.00
SD wkly *T* April to June	3.78	0.01	0.01	0.01	0.4512	°C	0.03	−0.04	0.13
MAX *T* warmest mo	25.22	0.14	−0.68	0.15	**<0.0001**	°C	−2.82	−4.15	−1.49
MIN *T* coldest mo	−1.14	0.06	0.33	0.06	**<0.0001**	°C	1.39	0.88	1.91
	Intercept 2013		PC1 2013			2013 Change in temperature metrics ~ PC1			
	$\Delta\hat{\beta}_0$	SE	$\Delta\hat{\beta}_1$	SE	*P*	Units	Change	LCL	UCL
CDD > 10°C April to June	73.36	0.71	2.72	0.71	**0.0002**	dd	−21.43	−37.97	−4.90
SD wkly *T* January to March	0.98	0.04	0.13	0.04	**0.0014**	°C	0.26	0.02	−0.53
SD wkly *T* April to June	1.08	0.01	−0.04	0.01	**0.0071**	°C	−0.11	−0.21	−0.02
MAX *T* warmest mo	1.17	0.06	0.19	0.06	**0.0047**	°C	−2.05	−3.38	−0.72
MIN *T* coldest mo	−0.61	0.08	−0.26	0.08	**0.0013**	°C	0.32	0.16	0.84

reported structural differences between old-growth forests and secondary Douglas-fir (*Pseudotsuga menzisii*) forests in the Pacific Northwest (*28*).

After we statistically controlled for the effects of elevation, PC1 was associated with most temperature variables (8 of 10 variables; Table 1). Effects were largest for maximum and minimum temperatures, as well as for CDDs in the spring and summer months. Temperature differences were substantial across the gradient in forest structure; for instance, in 2012, maximum spring monthly temperatures decreased by 2.5°C (95% confidence interval, 1.7° to 3.2°C; Fig. 4C) across the observed gradient in forest structure (from structurally simple plantations to complex old-growth forests). Minimum temperatures during this same period were 0.7°C (95% confidence interval, 0.3° to 1.1°C; Fig. 4D) warmer across the same gradient. Overall, these influences of old-growth forests on thermal conditions were consistent between years (although we found statistical evidence for year × PC1 interactions, parameter estimates of the interactions tended to be small and only resulted in a sign change for variability in temperature from January to March; Table 1).

DISCUSSION

Elevation was a powerful predictor for air temperatures across years, variables, and scales, confirming the importance of macrotopography

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in temperature patterns (*7*, *20*). However, although elevation predicted most temperature metrics well, it was less effective in predicting temperature variability and degree-day accumulation from January to March—both of which are microclimatic factors that are likely to influence species behavior and demography (*29*). Microtopographic variables, including slope, aspect, and relative topographic position, also influenced temperature patterns, but this effect varied markedly by time of year. Depressions and other topographically sheltered areas are thought to contribute to the decoupling of surface temperatures from regional patterns, thereby potentially generating microrefugia in complex terrain (*7*). These topographic features exerted a large influence on understory microclimate during winter, when persistent cold air pools form in depressions and valleys.

Vegetation characteristics associated with older forest stands appeared to confer a strong, thermally insulating effect. Older forests with tall canopies, high biomass, and vertical complexity (Fig. 1A) provided cooler microclimates compared with simplified stands (Fig. 1B). This resulted in differences as large as 2.5°C between plantation sites and old-growth sites, a temperature range equivalent to predicted global temperature increases over the next 50 years (*30*). This effect was potentially attributable to large differences in biomass between forest types (*16*), rather than canopy cover, as we observed less variation in canopy cover between old-growth sites and plantation sites (table S5). Although previous studies have shown strong influences of vegetation on microclimate, most of these demonstrated differences between significantly different vegetation types or stages, such as mature forests versus grasslands (*18*, *31*) or clearcuts (*19*, *32*). At the global scale, forests have been shown to have a broad-scale cooling effect (*33*); however, to our knowledge, this is the first evidence that subtler structural differences within mature forest types (that is, mature plantations versus old-growth forests) mediate under-canopy temperature regimes.

Our findings indicate that management practices that result in single-species, even-aged plantations are likely to reduce the thermal buffering capacity of forest sites, potentially limiting the availability of favorable microclimates for some species. Unlike most predictors, which were primarily useful when summarized at fine spatial scales (25-m radius), vegetation at broader scales exerted the strongest influences on temperature. This is consistent with results from studies that examine the thermal edge effects of high-contrast cover types (*34*); smaller forest patches tend to be more susceptible to changes in temperature (*20*), and such edge effects also limit microclimatic buffering of tropical forests (*22*). In jurisdictions where biodiversity maintenance is the goal, conservation and restoration of structures associated with old-growth forests are more likely to sustain favorable microclimates (*35*) and to reduce climate change impacts on temperature-sensitive species. Recent work shows that the understory microclimate differences documented here could be highly relevant to biodiversity conservation in temperate forests; cooler forest types have attenuated the widespread loss of cool-adapted understory plant species (*13*) and have promoted tree recruitment (*36*). Amphibians, lizards, insects, and even large mammals are shown to take advantage of microclimate conditions when regional climate moves beyond the range of thermal preferences (*5*, *37*, *38*). However, our findings apply to species inhabiting forest understory. Although a high proportion of forest biodiversity is found in this stratum (*39*–*41*), species associated with upper canopies may not benefit from the microclimate buffering capacity of old-growth forests. Furthermore, because older seral stages provide the highest levels of buffering, management options may be limited for species inhabiting early successional forests (*42*), unless they are able to take advantage of the microclimatic buffering of older forests or cooler microclimates that are near old-forest edges (*34*). Currently, early seral species are of high conservation concern in the Pacific Northwest, largely as a result of habitat loss (*42*); given that early seral forests may not have equivalent thermal refuges, we predict synergistic negative effects on these species when combined with climate.

We conclude that the substantial influence of vegetation structure on microclimate presents the opportunity to manage for conditions that favor the persistence of biodiversity (*43*). By conserving or creating forest conditions that buffer organisms from the impacts of regional warming and/or slow the rate at which organisms must adapt to a changing climate, it may be possible to ameliorate some of the severe negative effects of regional warming. Given the time frame for forests to acquire old-growth structural conditions, understanding thresholds in forest structure where important ecosystem services are lost is of pressing concern (*44*). With ~3.5 million ha of old-growth forests remaining in the coastal region of the Pacific Northwest (much of which is now protected under the federal Northwest Forest Plan) (*45*), there is substantial potential—in this region at least—to reduce the effects of warming on native populations of forest species.

Understanding the fine-resolution effects of vegetation structure on mediating microclimate could also improve predictions about biodiversity responses to climate change. Without considering the detailed information on forest structural conditions reported in this paper, published model estimates of changes in species' climatic niches (and extinction risk) (*46*, *47*) could be substantially underestimated or overestimated, depending on the amount of old forests in a landscape. We demonstrate how combining new remote sensing technologies (*48*) with machine-learning techniques provides an effective option to develop high-resolution, spatially explicit models of under-canopy temperature characteristics. As under-canopy temperature data become more readily available, microclimate variability can thus be mapped across broad spatial scales. Under-canopy temperature modeling, coupled with models predicting future vegetation dynamics (*49*), offers the potential to enhance our understanding of microclimate and species persistence in the face of climate change.

MATERIALS AND METHODS

Experimental design

We collected fine-scale temperature data using calibrated HOBO pendant data loggers (Supplementary Materials) at 183 locations across the 6400-ha HJA in the Cascade Mountains of central Oregon, USA (44°12′N, 122°15′W). Data loggers were placed 1.5 m above the forest floor; this ground- and shrub-level forest stratum is characterized by particularly high biodiversity in Douglas-fir forests of the Pacific Northwest (*39*–*41*). The HJA spans an elevational gradient from 410 to 1630 m above sea level. It is a conifer forest mosaic that is largely composed of a mix of old-growth forests, mature forests, and 40- to 60-year-old Douglas-fir plantations. Sampling locations were stratified across elevation, forest type, and distance to roads, ensuring that the full environmental gradient was sampled (Fig. 3) with a minimum distance between sampling points of 300 m (Supplementary Materials). We collected air temperature data at the 183 sample locations from January 2012 to July 2013. In total, we used 7,417,320 temperature loggings to calculate summary metrics (table S6).

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Environmental predictor variables

To model temperature metrics, we selected 19 predictor variables (table S7) that we hypothesized to be important for influencing air temperatures in forested mountain landscapes and categorized these into three main groups: (i) elevation (ELV), (ii) microtopography (TOPO), and (iii) vegetation structure (VEG). We derived all vegetation variables from LiDAR data collected at the HJA in August 2008 during the leaf-on period (*50*). LiDAR is a relatively new technology that allows for fine-scale mapping of forest structure across broad spatial extents (*51*, *52*). The VEG category variables described vegetation structure using metrics relating to (i) canopy height, (ii) cover at multiple strata, and (iii) vertical distribution of canopy elements.

Spatial scales

We assessed the importance of our predictor variables at two spatial extents around each sample point because the scale at which drivers of microclimate act is largely unknown (*29*): (i) at a radius of 25 m, which represents local-scale predictors, and (ii) at a radius of 250 m, which represents stand-scale conditions. The relative roles of the biotic and abiotic aspects of the environment could influence microclimate differently at these two spatial scales (*3*).

Response variables

Our 10 response variables included CDDs, mean monthly minimum and maximum temperatures, and variability in temperature (table S6). CDDs are linked closely to timing of spring plant bud burst, leaf out, and flowering, as well as insect emergence, and therefore have the potential to influence higher trophic levels (*53*). Variability in weekly temperature in both time periods (as measured by SD) may also determine the quality of sites (*54*). We chose the two time periods of January to March (winter-spring transition) and April to June (spring-summer transition) because they are relevant to the phenology of many organisms on our landscape. In temperate regions, phenological events during spring also have direct implications for reproduction and growth in both plants and animals (*55*). We also included the minimum and maximum temperatures of the coldest and warmest months, respectively, as they represent extremes at the annual time scale.

Statistical analysis: BRTs

We used a machine-learning approach (BRTs) to explore the relationship between our suite of predictor variables (19 × 2 spatial scales = 38 total predictor variables; table S7) and air temperature at our 183 sample locations. BRTs have recently been used extensively in species distribution modeling because of their capacity for uncovering nonlinear relationships between predictors and response variables, as well as their flexibility in testing interactions among predictors (*25*). BRTs can also handle large numbers of predictor variables and the colinearity between them (*25*), which is advantageous in studies such as ours, where there are many categorized predictor variables (table S7) but little prior information about which are most important or at which spatial scales. This modeling method afforded the flexibility to explore multiple potential correlates of microclimate without arbitrarily restricting our predictor set. We used the R program version 3.0.1 (*56*) in combination with the "dismo" package version 0.8-17 (*57*) for all analyses (Supplementary Materials). We used predictive deviance, measured as the mean deviance from the held-out data in all folds, as our primary measure of model performance. We also tested the correlation between predicted and observed temperature metrics via a built-in 10-fold cross-validation that uses 10% independent "test" data. These tests thus represent an entirely independent test of model performance. If overfitting occurred, these tests should show low correlations between predicted and observed values. We also tested for spatial autocorrelation in our temperature data set (Supplementary Materials).

We assessed the contribution of each group of predictor variables (ELV, TOPO, and VEG) to the explained variance in temperature metrics by summing the RI values of the variables in each category (Supplementary Materials). To determine the direction and nature of the relationships between the temperature metrics and the most influential individual predictor variables (>2% RI), we examined the partial dependence plots for the visualization of the fitted functions.

Statistical analysis: PCA and generalized linear mixed models

We performed a PCA on all of our LiDAR-derived vegetation variables at the 25-m scale (13 variables) to test whether we could reliably differentiate between plantations and older forests (table S3). This also aided in determining whether our vegetation structure variables captured the gradient in forest structure present across the landscape. A weakness of BRTs is that they do not produce effect sizes (that is, regression slopes) that can be easily related to differences in response variables (degrees Celsius or degree days). Therefore, we used generalized linear mixed models [R package "nlme" (*58*)] to examine the relationship between values of principal components (reflecting a gradient in forest structure) and temperature metrics. We combined data from both years and included "site" as a random effect to account for a lack of independence within location between years. Elevation was included as a covariate in all models. We tested for a year effect on the influence of PC1 on microclimate via an interaction model. In this fashion, we were able to quantify differences in temperature across the old-growth forest structure gradient and to test for consistency between years.

SUPPLEMENTARY MATERIALS

Supplementary material for this article is available at http://advances.sciencemag.org/cgi/content/full/2/4/e1501392/DC1

Supplementary Materials and Methods

fig. S1. Fine-resolution (5 m) spatial predictions of temperature metrics at the HJA based on BRT models.

fig. S2. Partial dependence plots showing the relationship between selected microtopographic variables and microclimate.

fig. S3. Partial dependence plots showing the relationship between selected vegetation structure variables and microclimate.

fig. S4. Key interactions identified from BRT models testing the effects of elevation, microtopography, and vegetation structure on microclimate.

fig. S5. RI of variables measured at 25- and 250-m scales for each temperature metric in both years.

fig. S6. Comparison of observed microclimate data by year.

fig. S7. Comparison of predicted microclimate metrics by year.

fig. S8. Photo of the HOBO temperature sensor in the field.

table S1. BRT model settings (learning rate, number of trees), performance diagnostics (deviance, deviance SE, CV corr, CV SE), and tests for spatial autocorrelation in the BRT model residuals (Moran's *I* and *P*).

table S2. Pearson's correlation coefficients (*r*) and associated *P* values for both observed and predicted values between years.

table S3. Results from a PCA of all vegetation structure predictor variables.

table S4. Summary statistics and *t* tests showing differences in LiDAR metrics between mature plantations and mature/old-growth forests.

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table S5. Results from Welch two-sample *t* tests comparing measures of biomass and canopy cover for plantation sites and mature/old-growth forest sites.
table S6. Temperature metrics used in our study and associated summary statistics.
table S7. Predictor variables used to predict patterns in microclimate metrics.
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Spatial models reveal the microclimatic buffering capacity of old-growth forests

Sarah J. K. Frey, Adam S. Hadley, Sherri L. Johnson, Mark Schulze, Julia A. Jones and Matthew G. Betts (April 22, 2016)

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Memorandum:

Sediment Concerns in Headwater Streams on State and Private Forests in the Pacific Northwest: A Brief Review of Directly Pertinent Science

Chris Frissell, Ph.D.
39625 Highland Drive
Polson, MT, 59860
Email: leakinmywaders@yahoo.com

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Abstract

Recent studies help elucidate the potential importance of Pacific Northwest logging practices on "dispersed" erosion and sediment delivery to headwater streams, apart from the more widely recognized point sources of roads and landslides. Rashin et al. (2006) show that near-term and immediate effects of logging on direct soil disturbance are substantial and pervasive, but can largely be avoided through no-yarding, no-felling, no-cut buffers strips extending a minimum 30 m from the stream margin, extended wider to encompass steep inner gorge slopes where present.

Reid et al. (2010), Klein et al. (2012), and Keppeler (2012) taken together show that even when buffer strips are left, logging of upland slopes results in increased runoff, which in turn can cause channel and gully erosion, stream network expansion into previously unchanneled headwater swales, and persistently elevated suspended sediment. Expanded channel networks post-logging generate new sediment, and also infiltrate sediment sources that were previously unconnected to surface waters. These changes and related geomorphic adjustments may cause recurring episodes of turbidity many years after logging. Turbidity impacts generally propagate to downstream receiving waters. Channel expansion can partially but not fully mitigated by riparian buffers. Where stream and wetland densities are not high, it could potentially be avoided only by limiting logging rate and pattern within and among first-order catchments to minimize the marginal hydrologic stresses of logging, in the face of past and future natural vegetation disturbances.

Introduction

Headwater streams by definition contribute water and sediment to downstream areas, hence are critical in determining water quality and habitat conditions for aquatic resources in receiving waters across the landscape (Likens and Borman 1974, Lowe and Likens 2005). But headwater streams themselves are by linear extent the most widespread freshwater habitat in Pacific Northwest landscapes. Small streams, whether permanent or ephemeral in flow, constitute important water resources and habitats for many animal and plant species. Olson et al. (2007) reviewed the importance of sediment and erosion and deposition processes in affecting stream-associated amphibians and determining sedimentation, water quality, and habitat features in downstream, fish-bearing waters. While the effects of logging, including increased sediment delivery to headwater streams can be at least partly mitigated by riparian buffers (within which logging operations, including ground disturbance and tree removal, are excluded), some larger-scale effects of logging across catchments tend to be pervasive, and not fully mitigated by narrow forest buffers. Moreover, current state rules governing forest practices on private lands do not require forest buffers on many--perhaps most-- headwater streams that are not fish-bearing, especially those that lack permanent or continuous flow (Olson et al. 2007).

Headwater streams lie at the erosional interface of the stream network and hillslopes. Not only are steep headwater streams the recipients of erosion and sediment generated by hillslope processes, they are themselves formed and maintained by erosional and sediment transport processes. Changes in the dynamics of erosion and sediment delivery, then are dramatically and rapidly expressed as changes in channel stability, morphology, and turbidity in headwater streams. Gomi et. al. (2005) reviewed the variety of processes and conditions that initiate and mediate the consequences of suspended and fine sediment in headwater streams in the Pacific Northwest. While the roles of forest roads and by landslides in delivering sediment to streams have received much scientific and some policy attention in recent decades in the Pacific Northwest, we have long known that forest practices also introduce sediment to streams via direct ground disturbance, and by way of alteration of catchment-wide hydrology following vegetation removal. Recent scientific information points to a need to re-examine current forest practices in light increasing understanding of these "dispersed" erosion and sediment sources.

In this short memorandum I focus on three relatively recently-published journal articles that provide important new information on sediment sources, sediment delivery, fine sediment effects, and the effectiveness of stream buffers in mitigating sediment delivery assocated with logging practices in forested watersheds of the Pacific Northwest. These studies are directly relevant to key questions that receive insufficient attention in policy debates: the need for buffers on headwater streams, critical design considerations for such buffers, and the need to regulate harvest rates within and across catchments.

1) Rashin, E. B., C.J. Clishe, A.T. Loch, and J.M. Bell. 2006. Effectiveness of Timber Harvest Practices for Controlling Sediment. Journal of the American Water Resources Association 42:1307-1347.

Based on systematic field surveys of ground and channel conditions, Rashin et al. (2006) examined immediate effects of logging-related ground disturbance on slope erosion, sediment routing, and headwater stream channel condition in forested industrial forestlands of Washington. The slope erosion study employed a between-sites-comparison design with unlogged sites as controls, and stream channel condition surveys used a before-and-after-logging design. Results were presented as a direct examination of near-term BMP effectiveness under forest practices rules that at the time allowed a variety of logging actions along non-fish-bearing, headwater streams, but did not require continuous riparian forest buffers.

This study found that riparian vegetative buffers (*ca*. 10m or more wide on each side of channel), including leave tree areas and with required directional felling and restrictions on log yarding or ground-based equipment use, were wholly or partially effective in preventing near-term sediment delivery from logging disturbance. Area of exposed soil that delivered sediment to stream channels was an order of magnitude higher in sites logged without stream buffers compared to those logged with buffer.

Effectiveness of stream buffers was breached at yarding corridors through buffers (associated with ground scars from cable yarding), and where selective logging occurred on steep inner gorge slopes within (or possibly adjacent to) the buffer areas. Stream buffers were generally effective at reducing near-term sediment delivery to streams regardless of whether surrounding logging was by clearcut or partial cut. However, yarding corridors measurably compromised the effectivenes of stream buffers in preventing sediment delivery.

Stream buffers were most effective where timber felling and yarding activies were kept at least 10m from streams and kept outside of steep inner gorge slopes. Simple exclusion of logging-caused ground disturbance from near-stream areas and hillslopes with a light likelihood of delivery of sediment to channels accounted for most of the effectiveness of stream buffers.

The primary observed causes of soil disturbance and erosion were skid trails from tractor yarding and yarding scars from cable yarding. Relict but ongoing erosion sources associated with roads, skid trails, and landslide scarps that had existed prior to the latest logging were also observed, but with lower frequency.

Windthrow of leave trees in stream buffers was frequent, but a far less important contributor of sediment than yarding scars and other ground disturbance, because the localized, self-draining topography of root throw pits discouraged sediment routing to nearby streams. This contrasted with the linear, down-slope oriented, and often larger features caused by direct logging disturbance. Windthrow of trees growing on the immediate streambank was seen to contribute sediment to streams, but this apparently was not the predominant location of blowdown. Surrounding logging prescriptions was seen to affect the windfirmness of stream buffers. Incidence of windthrow within buffers adjacent to clear cuts was an order of magnitude higher than within buffers adjacent to partial cuts, and also greater than riparian area windthrow within unlogged control watersheds. Recently, Schuett-Hames et al. (2012) reported the results of a study of windthrow in buffers along non-fish-bearing, perenial streams in western Washington. While they found a high incidence of windthrow within the first five years after logging, similar to Rashin et al. they observed only a modest incidence of sediment delivery to streams from windthrown trees within stream buffers.

Rashin et al. reported that where riparian vegetative buffers were not left, erosion and channel response were clearly linked to surrounding logging practices. Namely, clearcuts tended to exhibit more sustained active erosion and sediment delivery than partial cuts. In winter-cold, interior forests, logging over snow and frozen ground appeared to substantially reduce observed erosion and sediment delivery.

Despite that most streams in the study were small headwater channels with steep channel slopes (exceeding six percent), surveys revealed numerous instances of fine sediment accumulation on streambeds, both in pools and across the entire bed, associated with localized sediment sources (and in some cases, accumulations of logging slash). In addition, the extent of actively eroding streambanks increased in streams logged without buffers. These observations contradict the oft-repeated but seldom-tested presumption that sediment entering steep headwater streams is “rapidly flushed out” and therefore is assumed to have little effect in instream biota and water quality. Gomi et al. (2005) have previously commented that sediment routing and fate in headwaters streams in the Pacific Northwest has received insufficient study relative to the inherent risks fine sediment poses to aquatic resources.

Note the Rashin et al. study is conservative in the sense that “controls” were previously-logged, second-growth sites, and hence “background” or baseline incidence of erosion was likely elevated over natural rates in watersheds not previously disturbed by logging and roads (see below, Keppeler 2012).

Nevertheless the analysis provides a useful basis for evaluating future management of the existing, largely second- or third-growth, road-private forest landscape.

The most important caveat is that this study only evaluated localized erosion and sedimentation within the immediate two years after logging. Many potential sources of impact to streams, including those that result from hydrologic change caused by vegetation removal, and those that propagate over time and space, were not accounted for in the study design. Hence it is also important to recognize that some additional effects could occur that might not be fully mitigated by riparian buffers. Such processes include headward channel incision or gullying, landslides that increase with root strength depression occurring several years after harvest, and streamside erosion increases resulting from debris flow scour or the passage of coarse sediment waves.

2) Reid, L.M., N.J. Dewey, T.E.; Lisle, and S. Hilton. 2010. The incidence and role of gullies after logging in a coastal redwood forest. *Geomorphology* 117: 155-169. http://naldc.nal.usda.gov/download/40745/PDF

Although expansion of headwater channels has been an often-suggested cause of post-logging erosion, this key paper is apparently the first comprehensive and systematic study to assess the quantitative extent and sediment source contributions of this phenomenon in the Pacific Northwest. In Caspar Creek, north coastal California, Reid et al. (2010) found that second-growth logging of a redwood-dominated forest was followed by a substantial headwater expansion of stream channels density and coalescence of pre-existing discontinuous channels in headwater swales.

Despite "robust" riparian buffer strips left in the second round Caspar Creek logging during this study, suspended sediment yields in instrumented tributaries increased significantly after logging. Reid et al. documented that channel expansion was caused by observed headward migration of existing channel knickpoints and subsequent channel incision and enlargement, as well as sapping and collapse of subsurface flow macropores and pipes. Acceleration of surface and subsurface channel-forming processes was apparently associated with increased antecedent moisture conditions, soil saturation, and runoff caused by the abrupt reduction of forest canopy interception and evapotranspiration following logging. In addition, back erosion of extant channels increased in linear extent, possibly reflecting increased channel-forming flows possibly coupled with impingement of hillslopes that could have been creeping at faster rates in the years immediately following logging (e.g., see Swanston et al. 1988).

Channel heads (the initiation point of discrete or continuous, definable channels, with evident bedload movement) after logging were located substantially upslope from their pre-logging conditions. Observed channel incision through the extensive root frameworks of aboriginal redwood stumps and through weathered bedrock layers (saprolite) suggest that the present channel network has expanded well beyond historical pre-logging conditions in Caspar Creek. Time–trend monitoring and reconstruction indicate stream density increased by about 28 percent after the most recent epsisode of logging.

Expanded channel networks are associated with persistent increases in peak flow magnitude, which may result from more rapid translation of slower subsurface to rapid surface flow during storms. Erosion, both primary and secondarily associated with expanding or expanded channel networks, may be responsible for sustained elevation of suspended sediment yield and turbidity in Caspar Creek (reported in this study, Keppeler 2012, Klein et al. 2012, and discussed as a regional concern in the review by Gomi et al. 2005). Expanded channel networks increase surface water connectivity to and sediment delivery from pre-existing erosion sources like landslide scarps and roads, and can itself initiate additional mass erosion through bank collapse and trigging of channel-adjacent landslides.

If headwater streams are to be effectively protected against logging-related erosion, buffer design must anticipate headwater channel expansion, with a soil disturbance exclusion zones and vegetative buffers left in swales well above pre-logging channel head locations. However as Reid et al. observed, boles and living tree roots can hinder, but not entirely prevent channel expansion. Fully controlling channel expansion effects on streamflow, erosion, and sedimentation would require limiting the overall rate of logging within small catchments over time, moderating silvicultural treatments to promote more rapid hydrologic recovery (e.g., via partial cutting rather than clearcutting), and careful consideration of past and future natural events, including wildfire, windthrow, and disease which, independent of or interactively with logging, also alter the hydrologic effects of vegetation. In other words, substantial changes of silvicultural methods and reduction of overall rates of vegetation disturbance would likely be necessary for logged watersheds to remain within unimpaired, natural erosional and water quality limits.

In Caspar Creek, overall erosion in upland areas, including diversion-induced mass wasting, incision of crossings along untreated roads and skid trails, and in-channel gullying of headwater streams and zero-order basins, remains active. These sources have caused a sustained, second peak in suspended sediment yield about 20 years after initial logging (the first peak and temporary “recovery” occurred within the initial decade after logging) (Keppeler 2012).

3) Klein, R.D., Lewis, J., Buffleben, M.S. Logging and turbidity in the coastal watersheds of northern California. Geomorphology, Volumes 139–140, 15 February 2012, Pages 136-144, ISSN 0169-555X, 10.1016/j.geomorph.2011.10.011. Downloadable from: http://www.sciencedirect.com/science/article/pii/S0169555X11005277

Examining continuous turbidity records from 26 coastal watersheds in northern California, Klein et al. found that the extent (or rate) of recent logging activity (mean annual percent of watershed area logged within the last decade) was the strongest predictor of the duration of chronic turbidity. Road indices, and a large set of physiographic watershed characteristics were not correlated with chronic turbidity, except for a weak positive association with drainage area.

The study implicates the importance of headwater sources associated with ground disturbance and catchment-scale hydrologic change that can lead to channel expansion and gullying, as well as landslides; the authors conclude that "Despite much improved best management practices, contemporary timber harvest can trigger serious cumulative watershed effects when too much of a watershed is harvested over too short a time period."

Discussion

To my knowledge, the concerns raised in the Caspar Creek research (and previously identified as important in other areas, such as the Idaho Panhandle National Forest) have not previously been considered in Oregon Forest Practices discussions. This is an important scientific oversight. I have personally observed abundant evidence for the processes Reid et al. describe from Caspar Creek in my own research in the Oregon Coast Range and Klamath Mountains. (E.g., my Master's research at OSU was focused on exactly these kinds of headwater stream conditions in logged and unlogged watersheds of the Coast Range west of Corvallis, and what I observed is entirely consistent with Reid's description and inferences.) By contrast, the relatively immediate and direct erosion impacts observed by Rashin et al. (2006) are more widely recognized, but their consequences are commonly overlooked or minimized by pointing to the evidently larger and more spectacular sediment impacts associated with road and landslide sources. Currently in many areas of moderate or high road density, road-related impacts on streams likely are of such magnitude as to indeed obscure dispersed erosion and catchment-scale hydrologic effects. However, if road restoration and remediation becomes more widely applied, with road systems rationalized and road densities reduced, the pervasive effects of other disturbances associated with logging and forest management will almost certainly become more evident.

It's worth noting that the Oregon Department of Forestry's summary report, "The Oregon Forest Practices Act Water Protection Rules: Science and Policy Considerations" (Lorensen et al. 1994, p. 14) states, without attribution or citation,

that “Streamside buffers have been found to be relatively unimportant in preventing or reducing sediment delivery to streams.” When Oregon’s 1994 Forest Practices rules were finally adopted lacking continuous buffers on the vast majority of non-fish-bearing streams, it had been widely assumed, and was an oft-repeated meeting mantra, that "overland flow is relatively uncommon in the Pacific Northwest on forest soils,” and that somehow as a consequence of this, erosion simply was not an issue, except perhaps where roads intersected with streams. At that time, buffer design criteria focused on shade and woody debris recruitment to stream channels, not erosion or sediment delivery. This construct appears to be rooted in an early, agronomy-dominated perspective view that soil erosion occurs only as a function of classically defined overland flow (plus rainsplash detachment of exposed soil particles). However, geomorphologists and forest soils scientists have long recognized that erosion in steep, forested landscapes is dominated by a host of processes other than overland flow. For example, the erosion and sediment delivery observed in Rashin et al. (2006) and Reid et al. (2012) was initiated or propagated by piping, sapping, channel incision and gullying by headcut migration, mass erosion, accelerated creep or solifluction, root throw, and direct physical disturbance and displacement of soils, debris, and stream channels by equipment or logging debris. Considering this list of processes, all have either been shown to be altered by logging activity, have been identified as having that potential, or are defined as being directly the product of it.

Roads may be highly correlated with watershed condition, but it is important to recognize that such a correlation does not necessarily mean that “fixing” roads will alleviate all of the correlated effects. Road density integrates at least two major and separate categories of phenomena that contribute to erosion and sediment delivery (Trombulak and Frissell 2000). The first is erosion and sediment that is generated by the road itself and operations on it, and runs off into surface waters. In this category we can include secondary hydrophysical effects of roads, including landsides and gullies that initiate because roads disturbed natural drainage pattern, and maintenance-related runoff. This first category is targeted by road remediation and mitigation measures that reduce erosion or sediment delivery to streams from roadways.

The second category is indirect: the erosion and sedimentation that are generated by land use actions and practices that are either supported by or incidental to the road network. Those phenomena in the second category that pertain to dispersed erosion and sediment delivery in forested watersheds are the subject of this memo: primarily, they are direct ground disturbance from felling and yarding, accelerated windthrow around cutting unit margins, and channel extension, gullying, and bank erosion initiating as a consequence of catchment-wide vegetation removal. These erosion and sediment sources are not mitigated by road management measures.

On private industrial forest lands there is commonly a very high correlation between between known (or “open”) road density and the spatial extent of vegetation disturbance (primarily by logging, but also herbicides, grazing, and fire) among

watersheds, but that correlation gets weaker in some Coast Range watersheds that were in the past cut over in a short period of years, are now in an advanced second growth state and have seen relatively little thinning, alder conversion, "sanitation" cutting, or other logging disturbance in recent decades. Erosion may not be completely mitigated by reducing hydrologic connectivity of the road network, nor necessarily by directly by reducing road density through decommissioning, because dispersed erosion could continue if logging practices continue. However, road remediation and forest management planning that produce a major and lasting reduction in road density at whole-watershed or cross-watershed scales might secondarily limit the pattern and frequency of future logging and other vegetation management activity in ways that also limit hydrologic alteration and its resultant dispersed erosion.

Olson et al. (2007) suggested the possibility of adopting "islands," large forest patches that include stream networks, as reserve areas that are not exposed to the effects of logging that occurs elsewhere on the landscape. Such reserves would encompass whole headwater catchments, but also lap across ridgelines to provide terrestrial microhabitat connectivity to adjacent watersheds. While consolidating and protecting some measure of biological habitat integrity locally, this approach alone would not comprehensively protect headwater streams, or downstream receiving waters, from the harmful effects of erosion and sediment delivery. To meet such broader goals or requirements, an "island" strategy would have to be coupled with comprehensive buffer protection for headwater stream networks (including perennial, ephemeral, intermittent, incipient channels in swales or zero-order basins, as well as forested wetlands).

Where stream density is very high (including some highly dissected coastal terrain and interior wetland-rich areas), including a larger total area of each headwater catchment within an extensive network of unlogged riparian buffers along headwater streams, un-channeled swales and wetlands would by default result in a restricted pattern of logging. Such a reduced total footprint of forest disturbance could substantially limit the extent of hydrologic alteration from logging at the catchment scale, thereby greatly diminishing the magnitude and spatial extent of gully erosion and channel expansion as sediment sources. In landscapes with lower stream and wetland density, catchment-wide processes would be more prone to push erosion processes even when expansive headwater riparian buffers were left. Where stream and wetland densities are not high, this impact could potentially be fully mitigated only by limiting logging rate and pattern within and among small catchments to minimize the marginal hydrologic effects of logging in the face of natural vegetation disturbances.

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The Geography of Freshwater Habitat Conservation: Roadless Areas and Critical Watersheds for Native Trout.

Chris Frissell and Gary Carnefix

Senior Staff Scientist and Conservation Director (Acting), and Research Associate, respectively, Pacific Rivers Council, PMB 219, 48901 Highway 93, Polson, MT, 59860. phone (406) 883-1503, fax (406) 883-1504, email hanfris@centurytel.net and gcarnefix@aol.com, respectively.

ABSTRACT— Inventoried roadless areas on National Forest and Bureau of Land Management (BLM) lands have been the subject of sustained controversy and legal and policy machinations. The importance of presently unprotected roadless areas for conservation has received mention, but little formal analysis. Research from the northern Rocky Mountain Region of the U.S. helps put roadless lands in conservation perspective. We examine how roadless lands spatially integrate with watersheds of known high conservation value for freshwater species and habitats. Roadless areas can be small and fragmented, but can accrue to a large fraction of critical landscape. In the Upper Missouri Basin in Montana, within the 37% of the landscape with watersheds classified as highest value for freshwater conservation, almost one-half occurs within unprotected federal roadless areas; just 7% is inside wilderness and parks. In western Montana, bull trout *Salvelinus confluentus* abundance increases with watershed roadless proportion. Roadless lands tend to occupy middle to lower elevations compared to protected Wilderness, where they more directly interface with high-value fish habitat; a Montana statewide "fine-filter" assessment revealed remarkably high occurrence of native trout populations associated with roadless areas, even within watersheds that are otherwise compromised. Most roadless areas contain steep lands with expanses of erosion-prone soils. We conclude that the value of roadless areas for native trout and biodiversity conservation continues to receive insufficient evaluation and disclosure in roadless policy debates and decisions.

INTRODUCTION

Inventoried roadless areas on National Forest and Bureau of Land Management (BLM) lands, officially recognized in the RARE I and RARE II planning processes of past decades (USDA Forest Service 1972, 1979), continue to be the subject of sustained controversy and extraordinary machinations of policy and law. Scientists and fish and wildlife managers across the West recognize that native fish and high-quality waters are often positively associated with watersheds having low overall road density and large proportions of roadless area (Quigley and Arbelbide 1997; Trombulak and Frissell 2000; WNTC 2001). In public debate, however, the conservation value of roadless areas for native fish and water, while occasionally mentioned, has seldom been the focus of rigorous analysis or thorough consideration and public disclosure when decisions about roadless area or national forest management are made.

The Forest Service commonly prefers to consider roadless area decisions primarily in the context of how roads affect offsetting "motorized versus nonmotorized" recreational values—not their ecological and biological conservation values that are an intrinsic function of the lack of roads and many associated forms of human disturbance of land and water (Forman and Alexander 1998; Jones and Swanson 2000; Trombulak and Frissell 2000). The Clinton Roadless Rule

(USDA Forest Service 2000) was the first forum where the environmental values of Forest Service roadless lands were more broadly accounted for (see also Gucinski et al. 2001 on National Forest roads), and in so doing, it was explicitly in accord with numerous federal court rulings about roadless area significance. However, as a national rule this analysis did not document the case-by-case natural resource values sustained by roadless areas, and more recent agency efforts press for devolution of roadless area decisions to state and local authorities. Hence, it is increasingly important that a full local accounting of the natural resource conservation value of roadless lands be made.

In this paper we examine scientific evidence at three regional scales to illustrate how National Forest roadless lands in their current state contribute to the conservation of freshwater species and habitats, including native trout, in the state of Montana.

Roadless Areas and Freshwater Values: Statewide Analysis

We used USDA Forest Service digital mapping of roadless areas as the basis for our analysis. These data were "cleaned up" via adjustment for some errors, detached polygons, and redundant entries. We found the Forest Service inventory rosters 223 uniquely named roadless areas, encompassing a total of 26,359 km^2 in Montana. These areas are mapped statewide in Figure 1.

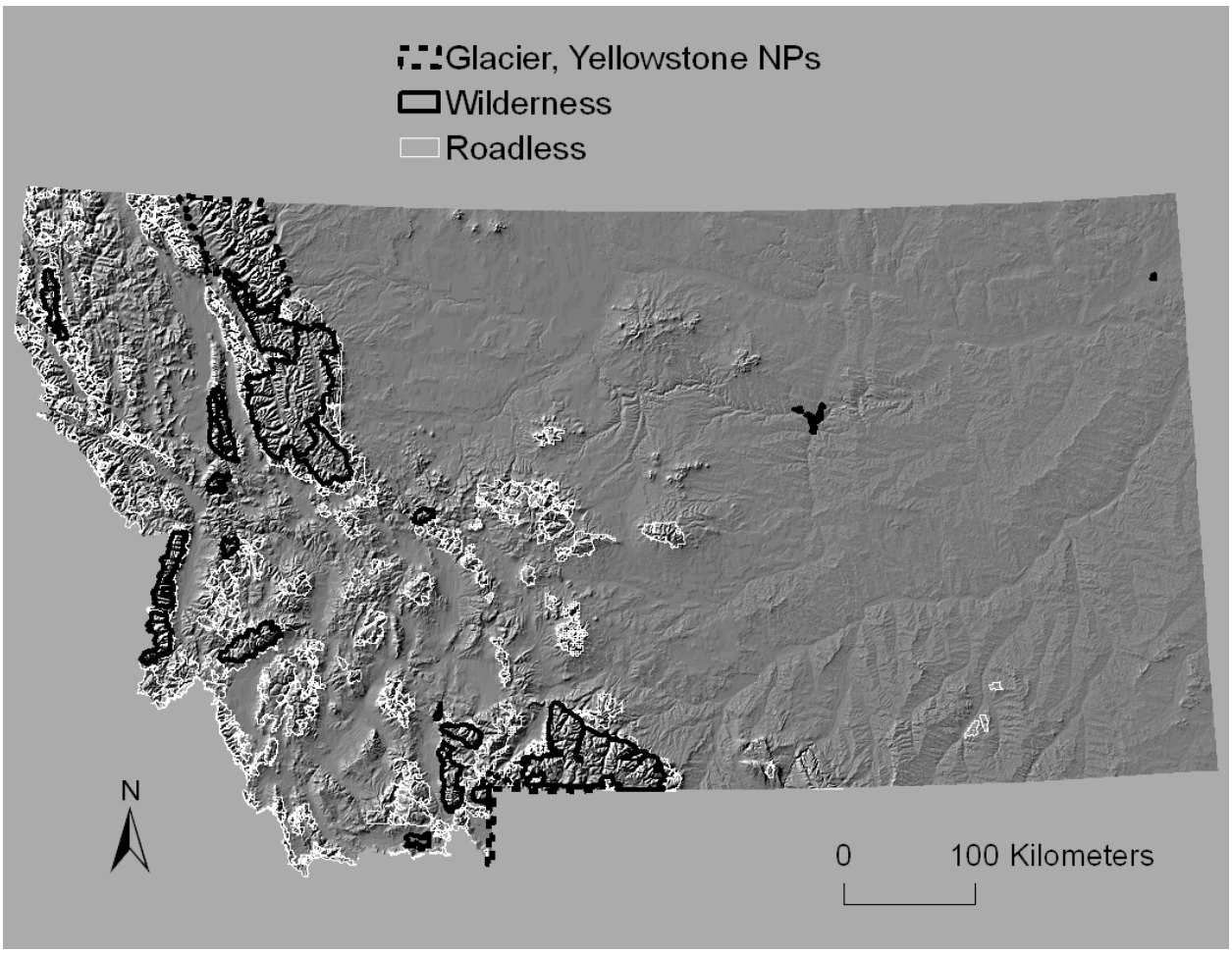


Figure 1. Distribution and relative topographic relationship of inventoried roadless areas (white outlines) on National Forest lands in Montana, USA. Black outlines delimit existing protected lands in congressionally designated Wilderness and National Parks

Native Fishes

By spatially intersecting Forest Service roadless area maps with stream and fish species data obtained from Montana Fish, Wildlife and Parks (Frissell et al. In Preparation; statewide fish distribution data with genetics for some species, Natural Resource Information System, Montana State Library, Helena, MT; note these data were adjusted by eliminating occurrences outside the native range of each species) with ArcInfo software, we identified a total of 1,282 occurrences of fish populations from the seven target taxa within inventoried roadless areas (IRAs) state-wide. Native westslope cutthroat trout *Oncorhynchus clarkii lewisi*, Montana's official state gamefish, occurred in 1,220 records, occupying 138, or 62% of the total list of IRAs. Known genetically pure populations (free of hybrid contamination from introduced rainbow trout *Oncorhynchus mykiss* or other trout) occurred in 92 IRAs. In other words, 41% of roadless areas support known pure populations of westslope cutthroat trout—an extraordinary figure because pure populations are thought to occupy far less than 10% of the present range of this subspecies (Allendorf et al. 2004). Bull trout *Salvelinus confluentus* occurred in 286 locations occupying 80, or 36% of IRAs statewide, with known genetically pure populations in 19 of these IRAs (many populations have not been tested). Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri* were found in 54 occurrences within 26 IRAs statewide, and eight of these areas support populations known to be free of hybridization. Columbia River redband trout *Oncorhynchus mykiss gairdneri* occurred in 20 places within 15 roadless areas, with two IRAs supporting known genetically pure populations. Arctic grayling *Thymallus arcticus* had 28 occurrences in 16 IRAs, and native lake trout *Salvelinus namaycush* occupied two IRAs in the Upper Missouri River basin.

Although the spatial extent of the native range of these species within Montana varies, within their ranges all species appear to show a substantial affinity for watersheds with a high proportion of roadless area. Native trout populations within roadless areas tend to be more frequently free of hybrization influence and demographically robust enough to be considered "strong" in population status (e.g., Quigley and Arbelbide 2000). Comparatively many streams in designated Wilderness and National Parks drain headwater lake basins that were historically targeted for introduction of nonnative trout; hence, native trout in some Wilderness and National Parks are more extensively hybridized or displaced by nonnative species (Hitt and Frissell 1999; Adams et al. 2001) than are those in many National Forest roadless areas.

Elevation and Slope Hazards in Roadless Areas

To examine how roadless areas relate to the remainder of the landscape with regard to general geomorphic features and landscape setting, we sought a range of data sets concerning soils and slope stability. Ultimately, within the time available and with a need for statewide coverage, the primary source we found to be useful was a digital elevation model (DEM), which we used to characterize the elevation and slope of roadless areas relative to other parts of the landscape. Our source of DEM data for this analysis was the USDA Forest Service Region 1's 200-m grid Digital Elevation Model (http://www.fs.fed.us/r1/gis/thematic_data/dem_region1_200m.zip). DEM data were processed and related to roadless areas and other land categories using ArcEditor mapping and GIS software. Slopes were derived from the DEM using ArcGIS's Slope tool, then classified with the desired class breaks using the Reclassify tool. Inventoried roadless areas were downloaded from http://roadless.fs.fed.us/documents/feis/data/gis/coverages/, as were Forest Service lands boundary data (nfsland_us_dd.zip). Then IRAs and FS lands within Montana were extracted using a state boundary polygon dataset. We used Montana designated Wilderness areas mapped by the Montana Natural Heritage Program (http://nris.mt.gov/nsdi/nris/shape/wild.zip),

masked with the Forest Service lands layer to extract only Forest Service Wilderness areas for these slope and elevation comparison analyses.

Figure 2 compares the elevation distributions of inventoried roadless areas, roaded portions of the National Forests, and National Forest lands protected as Wilderness. The graph clearly shows that the previously managed, roaded portions of the National Forests are concentrated at lower elevations. This is important because soils, climate, growing seasons, and other elevation-related factors often confer resilience to disturbance that is lacking in higher-elevation ecosystems. Roadless areas comprise a large proportion of high-elevation lands, as do existing Wilderness areas, but also clear in Figure 2 is that roadless areas contain a higher-than average concentration of land within the middle elevation range from 2000-6000 meters. Land in this elevation range receives relatively high snowpack, but tends to have shorter growing seasons and less resilient soils than lower-elevation terrain. A geographer might characterize lands in this elevation range as the hydrologic "bread and butter" of watersheds across the region.

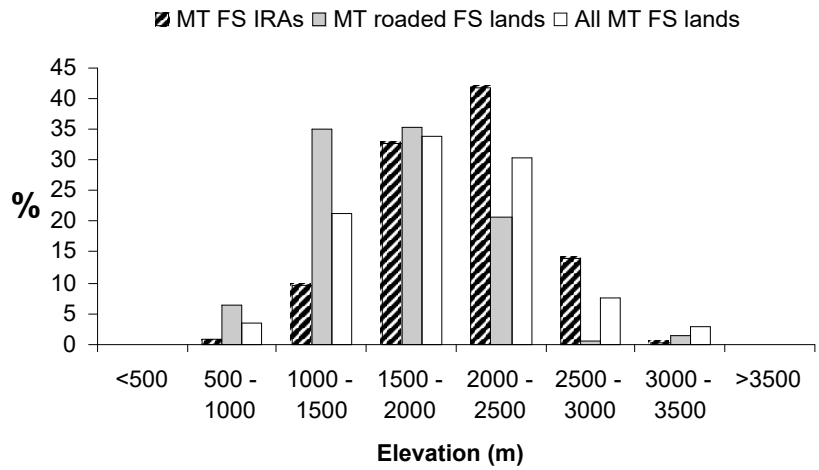


Figure 2. Elevation distribution of aggregated inventoried national forest roadless areas statewide (crosshatched bars, left), roaded national forest lands (gray shaded, middle bars), and all national forest lands in Montana (wilderness, roadless, and roaded, open bars).

Steep hillslopes are prone to mass failure, channelized debris flows, increased vulnerability to surface erosion and gully incision, and longer transport distances of detached sediments, yielding a greatly increased probability of delivery of sediments to streams compared to gentler slopes. All of these erosion types are significantly aggravated and sometimes initiated by disturbance of vegetation and soil surfaces through logging, grazing, mining, fire suppression activities, and other management practices that are supported by roads, as well as by construction, maintenance, drainage alteration by, and use of the roads themselves. We could not find direct and integrated metrics of soil surface erosion and landslide hazard, but the comparison of land categories by slope steepness distribution is revealing (Figure 3).

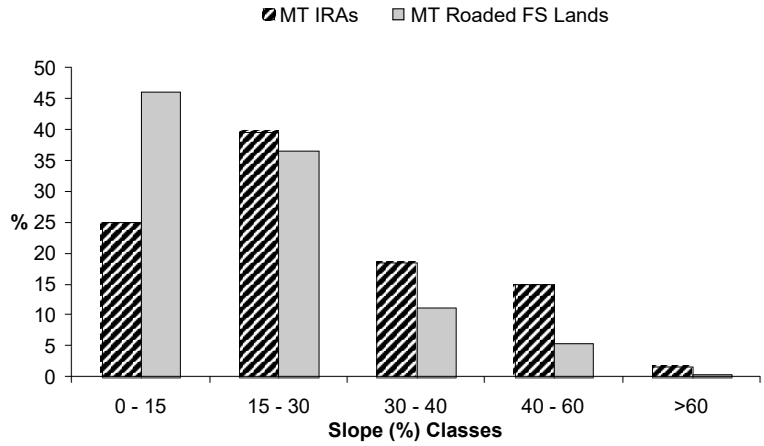


Figure 3. Distribution of slope steepness in Montana inventoried national forest roadless areas in aggregate (crosshatched) and roaded national forest land (shaded).

First, note that the roaded landscape of the national forests in Montana—sometimes referred to as "the working landscape" is dominated by gently sloping terrain. Using an arbitrary slope threshold of 40% for comparison, just under 6% of the managed, roaded landscape exceeds 40% slope angle; 94% is comparatively gentle or moderate in slope. By contrast, almost 17% of the area of inventoried roadless lands exceeds 40% in slope. Keep in mind, slope classifications based on DEMS of this resolution are biased downward for steep slopes, and the bias increases with average landscape slope. Hence, the difference is likely greater than depicted in these data. In simple terms, hazardous conditions are at least three times more prevalent in remaining roadless areas than they are on the already-roaded portion of the national forests. This is but one piece of evidence that risk of incremental harm to watersheds and fish habitat caused by management disturbance is severely elevated compared to that in the portion of the landscape where Forest Service managers have previous experience. In fact, the history of nearly every national forest and ranger district in Montana is fraught with one or more incursions of roads and logging projects into steeper portions of the landscape that led to immediate, disastrous consequences, followed by formal or informal decision to halt further development on those lands. This history is precisely the origin of the majority of inventoried roadless areas today.

Roadless Areas at the Basin Scale: the Upper Missouri

In 2002 a consortium of nongovernmental organizations sponsored a region-wide assessment of watersheds in the Upper Missouri drainage of Montana, spanning most of the Rocky Mountain Front and the headwaters of the Big Hole, Beaverhead, Jefferson, Madison, and Gallatin rivers (Oechsli and Frissell 2002). In the assessment process, we compiled comprehensive spatial data on native fish species distribution and status, nonnative species introduction and fish stocking, Montana Natural Heritage Program occurrences of aquatic-, riparian-, and wetland-dependent plant and animal species, and road networks as an indicator of landscape and habitat disturbance. All sub-watersheds were scored and ranked across these factors using different models, with the final result based on behavior across multiple models. The result is a map of this large region of central and southwest Montana showing variation in the relative ecological integrity of its streams, rivers, and watersheds (http://www.y2y.net/science/aia/AIA_UMfinal.pdf).

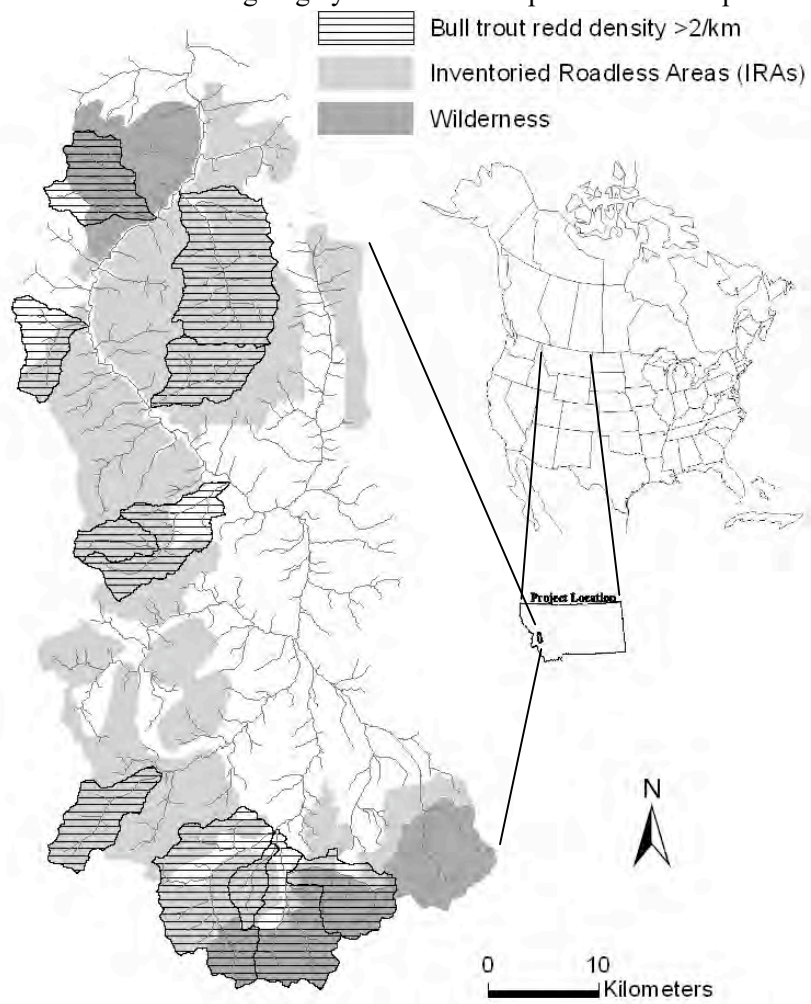
For this analysis, using data from Oechsli and Frissell (2002) we selected the highest-ranked 37% of the landscape where watersheds were classified as highest value for freshwater conservation and evaluated its land management status relative to national forest roadless areas. We found that just under one-half of this high-value acreage occurs within unprotected federal roadless areas. By contrast just 7% of this highest-priority area lies inside designated Wilderness and National Parks that can be considered to afford comprehensive watershed protection (Oechsli and Frissell 2002). This case analysis shows the extreme importance of roadless areas in maintaining the integrity of watersheds that support high conservation value. These are the watersheds needed to serve as the cornerstones of effective regional species recovery and aquatic restoration programs (Moyle and Sato 1991; Frissell and Bayles 1996; Frissell 2000).

Roadlesss Areas and Fish Abundance within Basins: Rock Creek

Bull trout are less likely to use highly roaded basins for spawning and rearing, and if present, were likely to be at lower population levels (Quigley and Arbelbide 1997, USFWS 1999). Baxter et al. (1999) reported an inverse correlation of bull trout abundance and roads in the Swan River Basin that strongly implicated the importance of protecting remaining roadless areas. Similarly, extensive habitat and population surveys on the Clearwater National Forest, Idaho, found that

with few exceptions, native salmonid abundance was higher and nonnative brook trout *Salvelinus fontinalis* were scarce or absent in streams draining largely un-roaded compared to road-impacted landscapes (Huntington 1995). Baxter et al. (1999) found that bull trout population trend over time was strongly negatively associated with road density.

In Rock Creek, tributary to the Clark Fork River in western Montana, we first indexed bull trout spawner abundance from redd survey counts made in 19 tributary sub-basins within the Rock Creek drainage (Frissell and Carnefix 2002). We compared response metrics of spawner abundance against a large suite of environmental variables, including measures of geomorphology, summer stream temperature, and land management. We used pairwise correlation of environmental variables with redd density to select 44 candidate predictor (independent) variables across 11 major categories such as geology, basin relief, channel slope, climate, size, and land use/management disturbance, for entry into stepwise and backward model selection routines.



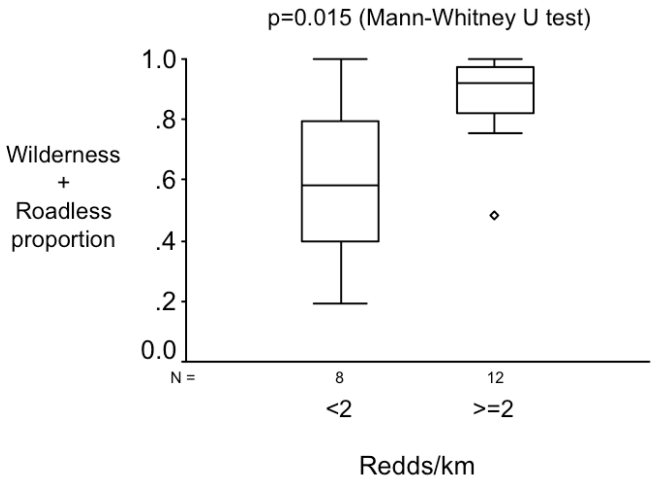


Figure 5. Top: Map of Rock Creek, Clark Fork Basin, Montana, showing the association between national forest Inventoried Roadless Areas (gray shading), designated Wilderness (black shading), and subwatersheds with high bull trout spawner density (crosshatched). Unshaded (white) portions of the basin are roaded areas and a small area of non-Forest Service land. Bottom: Box plot of proportion of un-roaded land area (Wilderness plus roadless) within basins of low (left) and high (right) bull trout spawning density.

We iterated multivariate analyses to compare effects of alternate aggregation and stratification methods for both response and environmental variables. Across iterations, significance tests revealed several robust results: spawner abundance increased with channel or sub-basin slope and extent of bounded alluvial valley geomorphology, and declined with maximum stream temperature. Abundance increased with increasing proportion of sub-basin comprised by wilderness and roadless areas, and the roadless variable was not significantly correlated with the other environmental correlates. Surprisingly, catchment road density did not correlate with bull trout spawning, but the range of road density among Rock Creek sites was far lower than regional average and one order of magnitude lower than in a previously published analysis for Swan River tributaries (Baxter et al. 1999).

Based on the Rock Creek results, we hypothesize that proportional roadless area, a variable that reflects the dispersion of road disturbance within a catchment, is an important measure of human disturbance at lower road density, but at moderate and higher road densities prevailing across the bull trout's range, total road density tends to saturate or override the effect of spatial distribution of roads within the catchment. These results are highly consistent with contention that undeveloped roadless areas are of special importance for sustaining, among other regional biological values, bull trout populations of relatively high abundance. In other words, even in landscapes marked by relatively limited road development, roadless areas appear to provide a critical and prominent function as refugia to sustain robust local populations of native fish. Protecting roadless areas will be a key element determining the success of any recovery or restoration plan.

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Hydrobiological & Landscape Sciences

Christopher A. Frissell
39625 Highland Drive, Polson, Montana USA 59860
Email: leakinmywaders@yahoo.com
Web: www.researchgate.net/profile/Christopher_Frissell
Mobile: 406.471.3167

Comments on Fisheries and Water Quality Issues in the US Forest Service Draft Environmental Impact Statement for the Alaska Roadless Rule, December 2019.

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1. Introduction

1.1 Scope and Importance of Review

I was commissioned in November and December 2019 by The Wilderness Society to review the US Forest Service's Draft Environmental Impact Statement for the Alaska Roadless Rule (https://www.fs.usda.gov/nfs/11558/www/nepa/109834_FSPLT3_4876629.pdf) (herein after, "DEIS"). The Society asked me to prepare comments on water quality and fisheries effects on the Tongass National Forest, as they are addressed, or not addressed in the DEIS, based on best available scientific information and my professional opinion as an aquatic scientist with expertise in freshwater ecology, fish conservation, watershed processes, environmental impact assessment and land and water resource planning. The observations and opinions in this document are expressly my own.

The resource at risk from logging and road construction on roadless lands of the Tongass National Forest is considerable at a regional and national scale (Byrant 2011, Halupka et al. 2003, Bryant and Everest 1998, Everest et al. 1997). Freshwater habitat on the Tongass National Forest produced roughly 25% of Alaska's commercial salmon catch in the past decade, with an average annual dockside landed value of US$88 million (Johnson et al. 2019). Despite recognized harms to salmon habitat in some watersheds from past timber harvesting and road construction, the Tongass National Forest produces

more wild salmon by far than any other national forest in the nation. This globally impressive productivity is in large part attributed to the extensive area of unlogged, roadless watersheds on the national forest, where ecological integrity water quality, biophysical diversity, and the productive capacity of freshwater habitat for salmon remain high (Halupka et al. 2003, Bryant and Everest 1998, Everest et al. 1997).

The proposed Alaska Roadless Rule would exempt the Tongass National Forest from the 2001 Roadless Area Conservation Rule and thereby remove that rule's prohibitions against road construction and timber cutting on all of the 9.2 million acres of inventoried roadless areas in the Tongass. The DEIS evaluates several other alternatives that provide varying levels of protection for Tongass roadless areas, but none are as protective as the No Action alternative.

The Forest Service's evaluation of impacts to fish habitat and salmon harvest are summarized as follows: "Overall effects to fish habitat are expected to be negligible under all alternatives, because of the strong protections to fish habitats provided by Forest Plan LUDs, Forest-wide standards and guidelines including the riparian management strategy, and the lack of old-growth harvest or associated road construction allowed in the T77 watersheds and TNC /Audubon Conservation Priority Areas" (DEIS ES-15). The DEIS further states that "localized effects on fish habitat may occur, but these are expected to be minimal overall" (DEIS ES-15). Consequently, according to the DEIS, "None of the alternatives are expected to have a significant change to the commercial fishing or fish-processing industries" (DEIS ES-13).

For reasons discussed below, these erroneous assumptions and conclusions in the DEIS are based on a grossly inadequate consideration of the best available science regarding the effects of road construction and logging on aquatic ecosystems.

1.2 Qualifications

I am a consulting aquatic ecologist and watershed scientist with expertise in land management and conservation and restoration strategies for fishes and amphibians, with extensive experience with Pacific salmon, native trout and charr. I also serve as Affiliate Research Professor at Flathead Lake Biological Station, the University of Montana. My expertise is outlined in my CV, which is appended to this declaration.

My education is as follows. I hold a Bachelors degree in Zoology from the University of Montana, and Masters and PhD degrees in Fisheries Science from Oregon State University, where the focus of my graduate research was the cumulative effect of land use and watershed disturbance on freshwater ecosystems and fish populations.

I have 30 years of experience as a research scientist in the field of aquatic ecology, fishery and conservation biology, and watershed science, having held research faculty positions at The University of Montana and Oregon State University. I have more than 40 scientific and technical publications in aquatic ecology, fishery and conservation biology,

and watershed science, in professional journals, symposia, books, and book chapters, and also am author of more than 30 research reports for various institutions and agencies. I have served as peer reviewer or reviewing editor for more than a dozen professional journals and government research publications. I have served on 13 professional and government panels that provided technical guidance about stream and river protection to state and federal wildlife and forest management agencies in three states, including technical panels that advised Oregon state agencies on water temperature standard development, and forestry landslide prevention rulemaking. I later served on Montana governor's scientific panel to inform that state's restoration strategy for threatened bull trout, and participated in Forest Service expert panels assessing the efficacy of regional plans for conservation of freshwater species, including amphibians. I have commented or served as an expert witness in litigation of numerous national forest plans and federal forest project and programmatic NEPA efforts since about 1980. In Alaska, I sponsored a PhD student who studied ecology and conservation headwater trout populations in southeast Alaska (Hastings 2005); contracted with USEPA to evaluate impacts of roads and pipelines in possible mine development in Bristol Bay; and reviewed environmental impact statements for mine and mine road development in Bristol Bay and the Ambler Mining District of the Brooks Range.

While on the faculty as a researcher at Oregon State University, I was funded to lead a 6-year research project on salmon habitat protection in Oregon coastal rivers. In 1992 I completed my doctoral dissertation on the cumulative effects of land use on salmon habitat in Oregon South Coast rivers. That research focused on the full spectrum of threats to physical habitat of salmon in coastal watersheds, including water temperature, sediment conditions, landslides and road erosion, large wood, and channel dynamics. As the dominant land use in the region, forestry was a primary topic of that research.

For ten years I was a full-time Research Assistant Professor and Research Associate Professor at the University of Montana's Flathead Lake Biological Station, where I continued to conduct research on salmon ecology and freshwater habitat conservation. For 11 years I held the positions (alternately) of Senior Staff Scientist or Conservation and Science Director with the Pacific Rivers Council, where I worked specifically on the interface of scientific information and land management, with considerable involvement in forest management policy development for stream protection and salmon and trout recovery, including in coastal Oregon. My work in particular has focused on the scientific adequacy of federal forest land planning and aquatic conservation policies, and I have special expertise in the manifold impacts on freshwater habitat and salmonid fishes of roads and road development in roadless forested watersheds.

1.3 Overview of Documents Reviewed

In preparing these comments I reviewed relevant portions of the DEIS and other Forest Service planning documents and other reports and articles from the scientific literature, as cited in the text below. In particular in the DEIS, I reviewed material in section 1 on aquatic habitat, soils, and water quality impacts; in section 2 on expected change in

salmon harvest and fish habitat; and in section 3 on soils and water, salmon harvest, fisheries, and transportation and roads.

2. Priority Watersheds and the Long-Term Conservation of Salmon Ecosystems

Although salmon in southeast Alaska represent five relatively widely distributed species, homing to natal habitats in combination with a diversity of habitat configurations and conditions has provided a ripe evolutionary field for the emergence of many distinct, locally adapted ecotypes within these species in southeast Alaska (Halupka et al. 2003). This diversity of habitats and locally adapted ecotypes is the very basis of salmon species productivity (Brennan et al. 2019, Schindler et al. 2010). This diversity of habitats and populations serves in turn as the basis of the large trophic and ecological roles that salmon play in ecosystems (Armstrong et al. 2019). This means the conservation of salmon and the manifold roles of salmon in the natural ecosystem and the human economy of southeast Alaska are directly dependent on protection and, where past degradation has occurred, restoration of the full natural diversity of aquatic habitats across the region.

Loss of diversity through increased footprint of human disturbance of watersheds will inexorably reduce the productive capacity of southeast Alaska, and especially the pristine, now roadless watersheds of the Tongass National Forest, for salmon. This fact is well-recognized in the scientific literature (see many aspects of the problem reviewed and cited in the text below), but it is obscured, if not overtly denied, in this DEIS. It seems the DEIS is premised on a covert, unstated, and utterly undocumented assumption that road-building and logging can occur in currently roadless watersheds with no risk of significant harm to aquatic habitat and fisheries. History and the available scientific literature establish clearly that this assumption is wholly untenable. The assumption is also at complete odds with Forest Service planning and policy documents of the past three decades, yet this departure is not explained or reasonably defended in the DEIS.

2.1 Protection of Priority Watersheds is in Question

In recent years the conservation of salmon in the Tongass National Forest has been strategically pinned to the concept of strict protection of a subset of watersheds in the region that are known to have high ecological and fishery values. One iteration is the Tongass National Forest Priority Watershed Classification (https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd622074.pdf), and another is the so-called "T77" watershed network proposed by a coalition of public interest and fishing industry groups (http://www.americansalmonforest.org/the-details.html, http://ak.audubon.org/sites/default/files/t77_subsection_seak_atlas_ch07_human_uses_200dpi.pdf). The DEIS falls short in failing to adequately account for the potential effect of removal of roadless area conservation protections and reclassification of timber suitability on road building and logging in these watersheds, which are heavily keyed to existing roadless areas where habitat, water quality, and watershed conditions remain

optimal. Habitat losses and fish populations impacts in these watersheds could disproportionately affect near term salmon production. Other than stating that old-growth harvest will continue to be disallowed in T77 watersheds under the 2016 Tongass Plan, the DEIS is wholly unclear as to the level and kind of protections these priority watersheds would receive under the alternatives. It appears the DEIS is designed to allow new road construction within the boundaries of conservation priority watersheds in order to access timber in adjacent areas, which could be highly detrimental to salmon habitat in these watersheds (see review of the impacts of roads below).

That said, in my opinion the shifting spatial distribution of salmon productivity demonstrated in recent "salmon portfolio" research (e.g., Brennan et al. 2019) calls into question whether a conservation strategy based primarily on protection of these selected watersheds is tenable in the long term. Watersheds that are productive for a given salmon species at the present time may not be those most productive for that species in past decades or centuries, and may not be those that will be most productive in future decades. A triage-based strategy that prioritizes a subset of extant habitat for conservation is warranted when one is considering a tattered landscape with few remaining productive habitats and populations, and the managing agency is in restoration mode. But when the subject is a relatively intact region, and the planning is to program actions that bring intrinsic risk of highly persistent adverse impacts to that habitat (e.g., roadbuilding and logging of primary and old growth forest), protection or restoration will not be the outcome. In fact, the outcome will explicitly be a net loss of habitat and population productivity--with possibly less loss of habitat and populations than if no protection priorities at all had been in place. And the shifting productivity/portfolio research on salmon ecosystems all points to our fundamental inability to anticipate where future production will come from, at least across relatively ecologically intact landscapes such as southeast Alaska.

The portfolio research tells us ultimately that a fixed reserve subset is not a viable means of protecting an existing productive salmon ecosystem, and that characterization certainly applies to the Tongass National Forest. Effective conservation of salmon on those forests will require comprehensive protections that assure no net loss of watershed condition relative to current conditions. That is plainly not the policy put forth in this DEIS, though the DEIS does not make that clear. Rather, the proposed action would risk degrading many watersheds that are currently in pristine roadless condition, while offering no reasoned assurance or defensible evidence that such widespread degradation would be compensated by habitat improvement or restoration elsewhere. Despite efforts in the DEIS to minimize effects through omission and tacit denial, the proposed action is in fact a massive, regional-scale step backward from the level of conservation that salmon enjoy under present forest plans, including the regulatory protection provided by the Roadless Rule.

3. Insufficiency of Riparian Management Areas to Protect Streams from Logging and Roads

While somewhat tacit and not stated in a plain way that could be subject to scrutiny and review, it is clear to an informed observer that the DEIS rests on an unfounded grand underlying assumption that logging and roadbuilding can be pursued in roadless areas with no significant or systematic impact on watershed processes, water quality, fish habitat and fish populations. One rationale for this vague and broad assumption is presumably that riparian protections offered in the Tongass forest plan are themselves sufficient or more than sufficient to fully mitigate any harms that might arise from road building, road use and maintenance, and logging. This is the context within which I evaluate the relevant literature in the following section. Virtually all of the following information is not considered in the DEIS; therefore, these potential and known impacts of logging and roadbuilding are not disclosed to the public therein—despite that they are widely documented in the Forest Service's own research (as cited below, and further within the reference sections of many of the papers and reports cited) and in the agency's own past planning documents.

Leaving unlogged riparian forests is insufficient to mitigate for the effects of upland logging on streams, contrary to the implications in the DEIS. In the sections below I discuss edge effects on windthrow or blowdown, mass erosion and channel erosion resulting from hydrologic changes caused by logging, the effects of roads altering hydrology and erosion processes, and alteration of groundwater temperature by logging. Each of these categories of impact poses consequences for fish habitat and water quality that need to be analyzed on a regional scale to account for potential cumulative impacts of multiple logging projects that we know, from past experience and common sense, can result from a systematic forest plan policy change, such as proposed removal of roadless areas from protection forest-wide. The DEIS arbitrarily and capriciously dismisses, and fails to substantively and accurately address, the environmental effects I discuss below.

3.1 Soils and Water Quality: Unreasoned Assumptions Wholly Inconsistent with Past NEPA Assessments, Plans and Policies.

The DEIS identifies aquatic habitat and the fisheries supported by that habitat as a "key issue" (DEIS 1-7). However, the document proceeds immediately to eliminate soils and water quality from detailed analysis (DEIS 1-8), with only sparse and grossly inadequate explanation. DEIS takes this inexplicable step despite that the mechanisms by which road construction, road use and management, and logging adversely affect soil erosion and water quality are well understood, and are the very mechanisms that in turn impact aquatic habitat and fish populations. This is the first of many inexplicable and wholly unreasoned skips of logic that allow the Forest Service to skirt the issues of risk of impact to salmon habitat and populations of the proposed action and alternatives in the DEIS. I offer a more detailed review of science pertaining to how salmon habitat is affected by alterations of vegetation, soil and water quality that occur when roadless areas are logged.

Specifically, the DEIS (1-8) states that a "preliminary review" of potential soil impacts found that Alternative 6 would increase the amount of land with "high hazard" soils that would be open for commercial logging by 38 percent. This is consistent with a nationwide pattern of relatively high concentrations of "high hazard" or high-erosion risk soils in national forest roadless areas. Indeed, vulnerability of soils to erosion and landsliding is among the major reasons the Forest Service has deferred road construction and logging and in these areas in the past. It is among the principal reasons they remain roadless today. Neverthess, the DEIS fails to address the environmental consequences of the increased area of "high hazard" terrain in lands allocated for logging on the Tongass. Inexplicably, the Forest Service simply claims that "From a broad standpoint, the impacts to soil characteristics and composition from the proposed alternatives would be the same as disclosed in the 2016 Forest Plan Amendment FEIS due to similar harvest levels and Forest Plan standards and guidelines" (DEIS 1-8), then capriciously denies that further analysis is needed. This claim in the DEIS stands in direct contradiction to the increase in "high hazard" soils in the commercial timber base. In my opinion it is near certain any increase in "high hazard" soils within areas open for commercial logging substantially increases the likelihood of damage to water quality and fisheries from post-logging soil erosion and sedimentation, as further described in my comments below.

The fact that the PTSQ remains unchanged, the reason given by the Forest Service as to why environmental effects related to soil erosion will ostensibly not increase under any alternative (DEIS 1-8) does not mitigate against potential increases in mass failure and soil erosion, for several reasons. One reason is that PTSQ is a "soft target" that does not in fact cap the total area logged in any given time period. For example, the same volume of timber can be drawn from a smaller area of concentrated larger trees, or a larger area of lower-volume and lower value trees. Another is that neither the PTSQ nor any other forest-level timber volume target regulates the specific areas logged within the overall area allocated to timber production. By knowingly including more high-hazard soils in the commercial timber base, the Forest Service inexorably increases the likelihood of triggering and increasing the incidence of erosion and landslides through errors of identification of erosion-prone sites and inadequate implementation of necessary mitigation measures (those being primarily *avoidance* of logging in high-hazard locales, see comments below).

For the reasons above, and because soil erosion hazard (including surface erosion, mass failure, and debris flows propagated by landslides) are central causal factors in the harms done by forestry operations to water quality and fishery resources, in my opinion it is arbitrary and utterly indefensible for this DEIS to fail to analyze, consider, and disclose the effects of commercial timber land reallocation and loss of roadless area protection on Tongass National Forest soil, water, and fishery resources.

3.2 Windthrow in Riparian Forests

Logging adjacent to riparian management areas alters the disturbance regime of riparian forests and streams in ways that can adversely affect fish habitat and populations (Moore

and Richardson 2012). Logging adjacent to riparian forests often results in increased windthrow of tree within riparian areas (Tongass National Forest Annual Monitoring Report 2007, Tongass National Forest Annual Monitoring Report 2013, Moore and Richardson 2012, Bahuguna et al. 2010, 2012, Rollerson and McGourlick 2001, Everest et al. 1997). Windthrow increased over natural background rates can result in exposure of channels to solar insolation and increased summer temperatures (Macdonald et a. 2003), reduction of future large tree recruitment, and increased channel bed and bank erosion, including landsliding and debris flows (Bahuguna et al. 2010, 2012, Lewis 1998, Mcdonald et al. 2003).

3.3 Landslides Originating from Upslope Cutting Units

Mass failures, including both shallow rapid landslides and deeper, often slower-moving slump-earthflow failures, are common across the Tongass National Forest, and it is well-established that the incidence of landslides is magnified by logging (Johnson et al. 2000, Everest et al. 1997, Swanston and Marion 1991, Wu and Swanston 1980, Wu et al. 1979). Logging not only directly disturbs soils, but associated vegetation removal renders soils vulnerable to mass movement and mass failure by reducing canopy interception and dispersion of rain and snow, by greatly reducing evapotranspiration and causing increased soil moisture conditions, and by destroying root strength that contributes to soil cohesion on forested slopes. The DEIS fails to consider and disclose how logging in currently protected roadless areas will impact mass-erosion-prone slopes, hence altering the frequency, magnitude, and distribution of landslides relative to salmonid habitats across the Tongass National Forest.

It is important to recognize that vegetation removal by logging—whether by clearcutting or thinning-—not only causes many landslides on recognized high-erosion risk terrain (which generally includes the steepest part of the landscape) but also increases the incidence of landslides on parts of the landscape that are usually considered to be of moderate or even relatively low risk of landslide erosion (most often because they are not as steeply sloping). This is a critical point, because the only effective means of preventing large increases in landslide occurrence is by identifying locations prone to failure and prohibiting vegetation removal on those sites, and in up-slope areas that contribute drainage to those sites. Some landside-prone sites occur on areas of the forest with moderate slopes and that are typically not mapped as highly landslide-prone. In many cases no clear surface evidence exists in the field that allows such sites to be identified prior to logging. Because complete avoidance of sensitive sites is impossible, logging will inevitably and cumulatively increase the incidence of landslides in salmon watersheds. The only question is how large the magnitude of increase in landslide erosion will be relative to unlogged watersheds. Previously unlogged roadless areas are likely to show the highest rates of landslide erosion increase if they are logged, because for the most part slopes in those areas have not previously experienced deforested or low-tree-density conditions in recent decades or centuries.

3.4 Headward Channel Expansion Caused by Altered Hydrology

Expansion of headwater channels has been an often-observed cause of post-logging erosion, but has been seldom quantified in Pacific coast watersheds (Frissell 2012). The one study I know of that focused comprehensively on this phenomenon, Reid et al. (2010), makes clear this is a seriously unexamined and too-often overlooked source of sediment delivery to Pacific Coast streams. Reid et al. (2010) reported that second-growth logging of a redwood-dominated forest in Caspar Creek, north coastal California, was followed by a substantial headwater expansion of stream channel density and coalescence of pre-existing discontinuous channels in headwater swales. Despite "robust" riparian buffer strips left in the second round Caspar Creek logging during this study, suspended sediment yields in instrumented tributaries increased significantly after logging. Channel expansion was caused by headward migration of existing channel knickpoints and subsequent channel incision and enlargement, as well as sapping and collapse of subsurface flow macropores and pipes. Acceleration of surface and subsurface channel-forming processes was apparently associated with increased antecedent moisture conditions, soil saturation, and runoff caused by the abrupt reduction of forest canopy interception and evapotranspiration following logging. In addition, back erosion of extant channels increased in linear extent, possibly reflecting increased channel-forming flows possibly coupled with impingement of hillslopes that could have been creeping at faster rates in the years immediately following logging (e.g., see Swanston et al. 1988). Reid et al. (2010) found that channel expansion led to stream density increasing by about 28 percent after logging.

Given that logging of any dense forest cover greatly reduced evapotranspiration of soil water, it is extremely likely the same processes drive erosion, channel expansion and sedimentation of streams after logging of forests of southeast Alaska. Expanded channel networks are associated with persistent increases in peak flow magnitude, which may result from more rapid translation of slower subsurface to rapid surface flow during storms. Erosion, both primary and secondarily associated with expanding or expanded channel networks, may be responsible for sustained elevation of suspended sediment yield and turbidity in Caspar Creek (reported in Reid et al. 2010, Keppeler 2012, Klein et al. 2012, and discussed as a regional concern in the review by Gomi et al. 2005). Expanded channel networks increase surface water connectivity to and sediment delivery from pre-existing erosion sources like landslide scarps and roads, and can itself initiate additional mass erosion through bank collapse and triggering of channel-adjacent landslides.

Reid et al. (2010) observed that boles and living tree roots in riparian forest buffers can partially hinder, but not entirely prevent, channel expansion. Fully controlling channel expansion effects on streamflow, erosion, and sedimentation would require limiting the overall rate of logging within small catchments over time, moderating silvicultural treatments to promote more rapid hydrologic recovery (e.g., via partial cutting rather than clearcutting), and careful consideration of past and future natural events, including wildfire, windthrow, and disease which, independent of or interactively with logging, also alter the hydrologic effects of vegetation.

Roadless areas preserve natural vegetation dynamics and disturbance regimes that maintain catchment hydrology and stream networks within a natural range of variability. Logging as an exotic disturbance in roadless areas is highly likely to alter hydrology such that accelerated stream erosion and stream network expansion result, over a larger area and larger number of watersheds than would occur if roadless areas are protected from logging.

Post-logging fluvial erosion, gullying and channel expansion is a scientifically recognized cumulative impact of logging that affects sediment supply and could potentially degrade salmonid habitat quality in connected waters downstream of headwaters if roadless areas of the Tongass National Forest are logged. This environmental impact has not been addressed or disclosed in the present DEIS.

3.5 Effects of Roads on Hydrology, Erosion and Sedimentation

Roads are well known to alter hydrology and erosion regimes in watersheds of the Tongass National Forest (Everest et al. 1997), just as they do elsewhere (Wemple et al. 2001, Luce and Black 2001, Jones et al. 2000, Trombulak and Frissell 2000). Landslides and gulley erosion initiating at or associated with the hydrological alterations caused by roads and landings not only can penetrate and deliver sediment through even very wide riparian forest buffers, they often initiate debris flows that can travel and directly impact aquatic and riparian habitat a great distance downstream from the point of origin. In either case, riparian forest buffers only confer limited protection against the harmful effects of road-caused mass failures, and in larger events, mass failures can virtually obliterate riparian forests, exposing streams to extremes of summer solar insolation and winter freezing, as well as redistributing large wood, scouring existing habitat structure away or burying it under large sediment deposits, and simplifying habitat structure in runout zones.

It is important to recognize that roads not only cause many landslides on recognized high-erosion risk terrain (which generally includes the steepest part of the landscape) but roads and landings often trigger landslides on parts of the landscape that are considered to be of moderate or even relatively low risk of landsliding under natural conditions. This results from the inexorable distortion of flow paths of both surface water and subsurface water caused by distortions of natural slopes and soils by road construction, use, and maintenance. The result is that road system expansion will inevitably expand both the number and area of occurrence of mass failures and associated debris flows and sediment deposits that adversely affect downstream fish habitat on a large scale. The DEIS utterly fails to consider, explain or disclose what the impact will be of road system expansion into currently roadless areas, many of which contain extensive areas of landslide-prone terrain.

Roads also cause chronic, on-going delivery of sediment at road crossings of small and large streams (Wemple et al. 2001, Jones et al. 2000), and sediment delivered even in the

smallest headwater streams can be rapidly transported downstream to harm salmonid spawning and rearing areas (Trombulak and Frissell 2000, Everest et al. 1997, Furniss and others 1991). Seldom can sediment discharges at road crossings be completely eliminated; to do so requires extreme care in crossing design and intensive, frequent within-season road maintenance. In fact, some road maintenance activities that are necessary to reduce the risk of catastrophic failure of forest roads, as well as actions to decommission or remove existing forest roads, themselves generate sediment runoff that can impact streams (Switalski et al. 2004, Luce and Black 2001b). Variability in the cause-effect relations between forest roads and stream sedimentation complicates both remedial practices and preventative practices in road construction and management, resulting in continuing high level of uncertainty about the effectiveness of so called "best management practices" (Al-Chokhachy et al. 2016). In contrast to this recognized uncertainty, the DEIS purports, while offering virtually no evidence, that the impacts of new roads on water quality and fisheries in and downstream of roadless areas will be somehow nonexistent.

The widespread, systemic failure or inadequacy of existing road maintenance resources to mitigate harm from sediment pollution (see Gucinksi et al. 2001), especially at road crossings and other near-stream road segments, is one of the major reasons the US Forest Service implemented the Roads Policy and Roadless Rule nationally (USDA Forest Service 2000). All national forests, including the Tongass, remain unable to adequately maintain the existing road system to reduce its ongoing and future harmful impact on aquatic resources and fisheries. It is clear on the face of it that proposed elimination of Roadless Rule protection on the Tongass, and potentially the Chugach National Forest, is intentionally designed to allow expansion of the existing road network. The DEIS offers no rationale for how harms to the overall road system will be reduced in the face of road system expansion that is supported by the proposed suspension of the Roadless Rule.

Roads may be correlated with watershed condition, but it is important to recognize that such a correlation does not necessarily mean that "fixing" roads will alleviate all of the correlated effects (Al-Chokhachy et al. 2016, Frissell 2012, McDonald and Coe 2007). Road density integrates at least two major and separate categories of phenomena that contribute to erosion and sediment delivery (Trombulak and Frissell 2000). The first is erosion and sediment entering surface waters that is generated by the road itself and operations on the road. This category includes secondary hydrophysical effects of roads, including landsides and gullies that initiate because roads disturbed natural drainage pattern, and maintenance-related runoff. This first category is targeted by road remediation and mitigation measures that reduce erosion or sediment delivery to streams from roadways (Al-Chokhachy et al. 2016, Switalski et al. 2004). The second category is indirect: the erosion and sedimentation that are generated by land use actions and practices that are either supported by or incidental to the road network, as discussed above. Those phenomena in the second category are direct ground disturbance from timber felling and yarding, accelerated windthrow around cutting unit margins, and channel extension, gullying, and bank erosion initiating as a consequence of extensive vegetation removal in the catchment. These erosion and sediment sources are not mitigated by road management measures.

The spatial arrangement of road networks on the landscape relative to slope stability, soil erosion proneness, and stream network locations act to codetermine the extent of impairment of downstream fish habitat by road-generated erosion and sedimentation (Al-Chokhachy et al. 2016, MacDonald and Coe 2007, Jones et al. 2000, Trombulak and Frissell 2000). Within the Pacific Coastal mountains and the Pacific Northwest more broadly, existing roadless areas are often associated with the highest-quality fish habitat, in part because of the limited spatial extent of road impacts and relatively few road crossing locations in their catchments. As a result, watersheds with a high proportion of roadless area tend to be relatively high in fish abundance, salmonid diversity and production, and roadless areas thus are of extreme value in the long-term conservation of salmon and trout populations throughout their ranges (Dellasala et al. 2011, Frissell and Carnefix 2007, Hitt and Frissell 2004, Loucks et al. 2003, Trombulak and Frissell 2000, Baxter et al. 2000). Despite that the proposed suspension of the Roadless Rule is explicitly intended to allow the expansion of the logging road network into presently roadless areas in Tongass National Forest watersheds, the DEIS utterly fails to explain how road system expansion will not be associated with more widespread impacts of salmon streams and more extensive deterioration of high-quality salmonid habitat.

Because road systems span multiple watersheds across large areas of national forest, because their adverse impacts cannot be completely avoided or remediated, and because harms to aquatic ecosystems accrue over many decades and are often triggered or exacerbated by natural events like winter storms and summer drought, as well as by climate change that affects storms and drought at regional scales, the cumulative impacts of expansion of road systems must be addressed at the scale of the national forest or a major portion of a national forest. That is, the cumulative effects of road system expansion into presently roadless areas on fish habitat and fisheries simply cannot be adequately analyzed, disclosed, or effectively remediated at the scale of individual timber or road construction projects (Selva et al. 2015, Hitt and Frissell 2004, Trombulak and Frissell 2000). For example, in many cases existing Forest Service roadless areas act in concert with National Parks, Wilderness, or other permanent land protections to secure fish habitat and other conservation values in a larger downstream stream and river network (e.g., Frissell and Carnefix 2007, Hitt and Frissell 2004, Loucks et al. 2003, Martin et al. 2000, Noss et al. 1999). This fact is a major underlying reason for the Forest Service's decisions to implement the Roadless Rule (Turner 2006, Martin et al. 2000, USDA Forest Service 2000) and Roads Policy as directives systematically augmenting national forest plans and planning procedures across the nation.

3.6 Water Temperature Alteration from Upslope Logging

Logging alters the evapotranspiration demand by directly removing vegetation. At least for the initial decade after logging, until vigorously growing second-growth trees attain significant cover, soil and groundwater tend to increase because vegetation is using less water. Moreover, the removal of canopy cover can expose soils to direct solar heating,

and areas of shallow groundwater may warm to a greater degree than they did under full forest cover.

Pollock et al. (2009) found that mean and summer mean and maximum temperature across 40 small streams on the Olympic Peninsula in Washington was substantially higher in streams draining watersheds with a higher proportion of cumulative logged area catchment-wide. The catchment area logged relationship was significantly stronger than the relationship to riparian forest removal by logging. Many streams with high canopy shade warmed substantially when more than half of their catchment area was logged. The results strongly suggest that factors other than direct canopy shade over the stream can drive water temperatures; these may include canopy opening from landslides and debris flows, or may indicate warming of shallow groundwater after extensive loss of soil canopy cover, or both. In either case, riparian buffers failed to protect streams from substantial temperature changes associated with logging.

Macdonald et al. (2003) found that headwater tributaries in BC logged with buffer strips of a wide range of widths all warmed 4-6 degrees C in summer compared to streams in unlogged watersheds. Part of this warming was associated with shade loss and post-logging windthrow, but a significant fraction of warming was unexplained by canopy shade, and is thought to have been associated with catchment-scale changes in shallow groundwater temperature or flow rates.

Research especially in long-term paired watershed studies in BC has shown that putative modest changes in daily mean, maximum, or minimum stream temperature associated with logging can result in biologically significant changes in cumulative thermal exposure. These in turn result in shifts in development rates of and timing of fish population life history events, such as time of emergence of young-of the-year from streambed gravels (Macdonald et al. 1998, Holtby 1988, Holtby and Newcombe 1982). Such developmental rate changes are known to alter salmon survival rates, and can result in population decline or collapse (Bryant 2009, Holtby 1988, Holtby and Newcombe 1982).

The DEIS ignores and fails to consider or disclose these known relationships between logging and alteration of temperature regime in streams that can cause substantial adverse cumulative effects on fish life history and population productivity, especially in Pacific salmon.

4. Climate Change and Resilience of Roadless Watersheds

Watersheds with a large proportion of primary forest and roadless area are likely to be among the most resilient salmonid habitats to the stresses imposed by ongoing and future climate change (Bryant 2009, USDA Forest Service 2000). One principal category of recurring and lasting impact from roads and logging is to introduce stressors that reduce resilience and increase the volatility of watershed responses to climatic stresses like flood and drought. Examples include the increased incidence of landsliding in the face of

winter storms or rain-on-snow events, and the potential depletion of stream base flows by a combination of increased water demand by second growth forest and increased drought stress. Another major and extensive source of impact from climate change is likely to be the marine inundation of current estuaries from rising sea level (although in a few cases new estuaries may be created or existing estuaries expand in the face of sea level increases).

The dominant vectors of expected change in climate (Bryant 2009) and the effects of road development and logging in roadless watersheds inexorably increase the vulnerability of freshwater habitats, and the fish populations dependent upon them to recurring climatic stresses like floods and drought. Their inherent resilience to climate variability and extreme weather events is one of the reasons that watersheds associated with roadless areas are considered "safe havens," refugia, or core areas for conservation of salmonid fishes and other sensitive species (Bryant 2011, Dellasala et al 2011, Frissell and Carnefix 2007, Baxter et al. 2000, USDA Forest Service 2000, Bryant and Everest 1998).

Despite the recognized imperative that climate changes places on land managers of coastal and northern regions (Bryant 2009), the DEIS critically fails to consider or analyze the likely effects of road development and logging on the response of currently roadless watersheds to future climate change.

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Curriculum Vitae *19 June 2018*

CHRISTOPHER A. FRISSELL

Research Ecologist & Freshwater Conservation Biologist
39625 Highland Drive
Polson, Montana 59860

Phone: 406.883.3891/Mobile 406.471.3167
e-mail: *leakinmywaders@yahoo.com*
WWWeb: https://www.researchgate.net/profile/Christopher_Frissell/

Birth: 1 December 1960, Chehalis, Washington

Education: Ph.D. in Fisheries Science, Oregon State University, 1992
M.S. in Fisheries Science, Oregon State University, 1986
B.A. with High Honors in Zoology, University of Montana, 1982

Professional Appointments:

Principal Scientist and founder, Frissell & Raven Hydrobiological and Landscape Sciences, LLC, Polson, MT, 2012-present (affiliated with Kier & Associates, M. Scurlock & Associates, and Pacific Watershed Associates).

Affiliate Research Professor and summer field course instructor, Flathead Lake Biological Station, The University of Montana, 2016-present

Director of Science and Conservation and Senior Staff Scientist, The Pacific Rivers Council, 2000-2012

Research Associate Professor, The University of Montana, Flathead Lake Biological Station, 1998-2000

Research Assistant Professor, The University of Montana, Flathead Lake Biological Station, 1993-1998

Research Assistant Professor, Department of Fisheries and Wildlife, Oregon State University, 1994-1997

Postdoctoral Research Associate (Faculty), Department of Fisheries and Wildlife, Oregon State University, 1992-1994

Research Assistant (Faculty), Oak Creek Laboratory of Biology, Department of Fisheries and Wildlife, Oregon State University, 1985-1992

Fields of Interest:

- Land-aquatic ecosystem linkages and cumulative impacts of natural processes and human activities on stream habitat and stream biota.
- Ecology, biogeography, and conservation biology of fishes and freshwater biota in relation to landscape and hydrologic change.
- Aquatic ecosystem conservation and restoration strategies.
- Geomorphic, hydrophysical, and landscape ecology considerations in design of integrated conservation reserves.
- Restoration and recovery planning and design for freshwater ecosystems and species.

Theses and Dissertations:

Frissell, C.A. 1992. Cumulative effects of land use on salmon habitat in southwest Oregon coastal streams. Doctoral dissertation, Oregon State University, Corvallis.
Frissell, C. A. 1986. A hierarchical stream habitat classification system: development and demonstration. M.S. thesis, Oregon State University, Corvallis.
Frissell. C. A. 1982. Colonization and development of community structure in coexisting Ephemerellid mayflies (Ephemeroptera, Ephemerellidae). Senior Thesis, Watkins Scholarship Program, The University of Montana, Missoula.

Professional Societies:

Society for Conservation Biology, 1991-present
American Fisheries Society, 1985-present
Ecological Society of America, intermittent
North American Benthological Society, intermittent

Graduate Students Mentored

Cavallo, B.J. M.S. in Organismal Biology and Ecology, The University of Montana, 1997. Thesis title: *Floodplain habitat heterogeneity and the distribution, abundance, and behavior of fishes and amphibians in the Middle Fork Flathead River Basin, Montana.*
Adams, S. B. Ph.D.in Organismal Biology and Ecology, The University of Montana, 1999. Dissertation title: *Mechanisms Limiting a Vertebrate Invasion: Brook Trout in Mountain Streams of the Northwestern USA*.
Hitt, N.P. M.S. in Organismal Biology and Ecology, The University of Montana, 2002, *Distribution and potential invasion of introduced rainbow trout in the upper Flathead River drainage.*
Carnefix, G. M.S. in Organismal Biology and Ecology, The University of Montana, 2002. Thesis title: *Movements and ecology of bull trout in Rock Creek, MT.*
Hastings, K. Ph.D.in Organismal Biology and Ecology, The University of Montana, 2005. Dissertation title: *Long-term persistence of isolated fish populations in the Alexander Archipelago.*

Reviewer for Journals and Agency Publications:

Canadian Journal of Fisheries and Aquatic Sciences, Conservation Biology, Ecological Applications, Environmental Management, Fisheries (AFS), Freshwater Biology, North American Journal of Fisheries Management, Oikos, Transactions of the American Fisheries Society, Fundamental and Applied Limnology, USDA Forest Service General Technical Reports

Member of Board of Editors for Journals:

Conservation Biology, 1996-2000

Appointments to Review Panels and Scientific Advisory Committees:

USEPA Bristol Bay Watershed Assessment Team, Subcontractor on road and pipeline impacts, through University of Alaska Anchorage and NatureServe, 2011-2012.

Independent Expert Review Panel for King County Water and Land Resources Division's Project Scoping and Implementation Practices. 2011-12. Subcontractor to MWH (Montgomery Watson Harza) for King County Dept. of Natural Resources and Parks, Seattle, WA. http://www.kingcounty.gov/environment/dnrp/publications/wlrd-expert-review-report.aspx

Umpqua Watersheds Science Advisory Council, Sponsored by Umpqua Watersheds, Inc., 16-17 November 2010, Roseburg, OR.

Wychus Creek Restoration Monitoring Plan Review Panel, sponsored by Upper Deschutes Watershed Council and Bonneville Environmental Foundation. 2 October 2009, Bend, OR.

Landscape Pattern Task Group, *State of the Nation's Ecosystems* report. 2003-2007. H. John Heinz III Center For Science, Economics and the Environment. Washington, DC. http://www.heinzctr.org/Programs/Reporting/Working%20Groups/Fragmentation/index.shtml

Science Review Team, King County Normative Flow Studies Project. 2002-2005, Seattle, WA. http://dnr.metrokc.gov/wlr/BASINS/flows/science-review-team.htm

Science Advisory Panel, Westside. Governor's Salmon Restoration Funding Board, Washington State, February 2000.

Ecological Work Group, Multi-species Framework Process and Subbasin Assessment Process, Northwest Power Planning Council 1998-2000.

Peer review panelist for U.S. Environmental Protection Agency/National Science Foundation Water and Watersheds Grants Program for 1997. 7-9 May 1997.

Scientific Group for the Governor's Bull Trout Restoration Team, State of Montana, 1994-2000

Oregon Department of Environmental Quality, 1992-95: Temperature Standards Review Subcommittee of the Technical Advisory Committee, Triennial Water Quality Standards Review

Scientific Assessment Panel for amphibian species, Eastside Oregon-Washington and Upper Columbia Basin EIS, US BLM and US Forest Service, 1994

Oregon Department of Forestry, 1990-93: Technical Advisory Group for the Forest Practices Monitoring Program; Wetlands Technical Group; Stream Protection Advisory Panel.

Peer-Reviewed Articles Published in Scientific Journals:

Hand, B.K., C. G. Flint, J. A. Stanford, C. A. Frissell, C. C. Mulhfeld, S. P. Devlin, B. P. Kennedy, R. L. Crabtree, W. A. McKee, Gordon Luikart. In Press. The Importance of a Social-Ecological Perspective for Riverscape Management in the Columbia River Basin**.** *Frontiers in Ecology and the Environment*.

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Frissell, C.A. 2008. Ecological impacts of roads in an era of climate change (Abstract). Watershed Restoration and Forest Roads Symposium, Pacific Rivers Council, 4 April 4, Tacoma, WA. http://pacificrivers.org/conservation-priorities/land-management/roads/watershed-restoration-and-forest-roads-symposium

Frissell, C.A., and G, Carnefix. 2007. (Abstract) Spawning abundance of bull trout (*Salvelinus confluentus*) in relation to geomorphology, temperature and roads in tributaries of Rock Creek Basin (Missoula and Granite Counties), Montana, US. Annual Meeting of the Montana Chapter of the American Fisheries Society, 13-16 February, Missoula, MT. http://www.fisheries.org/units/AFSmontana/2007%20MCAFS%20Annual%20Meeting%20Program.pdf

Frissell, C.A. 2007. Setting regional priorities for watershed restoration. 25th Salmonid Restoration Conference, Salmonid Restoration Federation, 9-10, Santa Rosa, CA.

Frissell, C.A. 2006. Post-fire management effects on streams. NCSSF Disturbance, Management, and Biodiversity Symposium, **National Commission for Science and Sustainable Forestry,** 26-27 April, **Denver, CO.**

Frissell, C.A., and G. Carnefix. 2005. (Abstract) Indicators of landscape pattern for freshwater ecosystems. 20th Annual Symposium of the US-International Association for Landscape Ecology, 12-16 March, Syracuse, NY.

Frissell, C.A. 2004. Managing risk and uncertainty: National Forest management and freshwater conservation. Regional Centennial Forum: The Forest Service In the Pacific Southwest Region. US Forest Service, 5-6 November, Sacramento, CA.

Williams, J.E., D.A. DellaSala, J. F. Franklin, C,D.Williams, and C. Frissell. 2004. Scientific findings require a new vision for successful wildlfire preparation. News briefing at the Society for Conservation Biology Annual Meeting, Aug. 2, 2004., Columbia University, New York, NY. http://www.conbio.org/Media/Fire/

Frissell, C.A. 2001. (Abstract) What to do first with limited time, money, and staff. Watershed Restoration Workshop: Integrating Practical Approaches. Oregon Chapter of the American Fisheries Society, 13-15 November, Eugene, OR.

Ebersole, J.L., Colden V. Baxter, Hiram W. Li, and William J. Liss, and Frissell, C.A. 2001. (Extended abstract) Detecting temporal dynamics and ecological effects of smallmouth bass invasion in northeast Oregon streams. *In*: Proceedings, American Fisheries Society Special Symposium: Practical Approaches for Conserving Native Inland Fishes of the West. Montana Chapter and Western Division of the American Fisheries Society, 6-8 June, The University of Montana, Missoula, MT.

Selected Papers and Seminars Presented Since 1993, continued:

Carnefix, G., C. Frissell, and E. Reiland. 2001. (Extended abstract) Complexity and stability of bull trout (*Salvelinus confluentus*) movement patterns in the Rock Creek drainage, Missoula and Granite counties, Montana. *In*: Proceedings, American Fisheries Society Special Symposium: Practical Approaches for Conserving Native Inland Fishes of the West. Montana Chapter and Western Division of the American Fisheries Society, 6-8 June, The University of Montana, Missoula, MT.

Frissell, C.A. 1999. (Abstract) Groundwater processes and stream classification in the montane West. Invited paper, Symposium #7: Aquatic Classification Schemes for Ecosystem Management: Making the Transition from Methods Development to Application and Validation. Annual Meeting of the Ecological Society of America 7-12 August, Spokane, WA.

Frissell, C.A. 1999. Fisheries and watershed processes: strategies for protection and restoration. Invited paper, Annual Meeting of the Cal-Neva Chapter of the American Fisheries Society, 24-27 March, Redding, CA.

Frissell, C.A. 1999. Surface-subsurface flow linkages in rivers and their importance for river flow conservation. Invited paper, Symposium on Water Quality and Hydropower Re-licensing, Annual Meeting of the Cal-Neva Chapter of the American Fisheries Society, 24-27 March, Redding, CA.

Frissell, C.A. 1999. Dams, uncertainty, and the salmon ecosystem. *Keynote Address*, Annual Meeting of the Idaho Chapter of the American Fisheries Society and The Wildlife Society, 4-6 March, Boise, ID.

Frissell, C.A. 1998. Climate forcing of thermal habitat in Pacific Northwest rivers: Buffering effects of floodplain forests and hyporheic processes. (Abstract) Symposium on Climate Change Impacts to Freshwater Fish Habitats, Annual Meeting of the American Fisheries Society, 23-27 August, Hartford, CT.

Frissell, C.A. 1998. Ecosystem concepts in large-scale restoration. (Abstract). Montana Chapter of the American Fisheries Society, 3-5 February, Helena, MT.

Frissell, C.A. and B.J. Cavallo 1997. Aquatic habitats used by larval western toads (*Bufo boreas*) on an intermontane river floodplain and some landscape conservation implications (Abstract). Annual Meeting of the Ecological Society of America, 10-14 August, Albuquerque, NM.

Stanford, J.A. (presented by C.A. Frissell). 1997. Conservation and enhancement of alluvial rivers: the importance of hyporheic linkages. (Abstract). Symposium on Ecological Effects of Roads, Society for Conservation Biology, 7-10 June, Victoria, British Columbia, Canada.

Frissell, C.A., and G.C. Poole . 1997 Management of Riparian Zones in Western Montana: Present Issues and Emerging Challenges. (Abstract). Annual Meeting of the American Fisheries Society, 23-28 August, Monterey, CA.

Frissell, C.A., and J.T. Gangemi. 1997. Roads and the conservation of aquatic biodiversity and ecological integrity. (Abstract). Society for Conservation Biology, Victoria, British Columbia, Canada, 7-10 June.

Selected Papers and Seminars Presented Since 1993, continued:

Frissell, C.A. 1997. Spatial assessment of biological status and biodiversity loss. Invited seminar, National Research Center for Statistics and the Environment, University of Washington, Seattle, WA, 14 January.

Frissell, C.A., and B.J. Cavallo 1996. Thermal and hydrologic diversity of aquatic habitats mediated by floodplain complexity and hyporheic flow exchange in an alluvial segment of the Middle Fork Flathead River, Montana, USA. (Abstract). Annual Meeting of the N. Am. Benthological Society, Kalispell, MT, 3-8 June.

Frissell, C.A. 1995. Ecological principles for watershed restoration. (Abstract). Invited paper for Workshop on Watershed Restoration: Principles and Practices, Annual Meeting of the American Fisheries Society, Tampa, FL, 27-31 August.

Frissell, C.A. 1995. Managing native fish and their ecosystems: let's get (spatially) explicit! (Abstract). Invited panel presentation at Montana Chapter of the American Fisheries Society, Chico Hot Springs, MT, 6-10 February.

Frissell, C.A. 1995. Birth in the fast lane: sediment transport, human disturbance, and reproductive strategies of salmonid fishes in Pacific Northwest streams. (Abstract). Invited paper for Symposium on Influence of Geomorphic Processes on Terrestrial and Aquatic Ecosystem patterns and Processes, Annual meeting of the Ecological Society of America, Snowbird, UT, 31 July-3 August.

Frissell, C.A. 1995. Resource management impacts on bull trout populations. Invited panel presentation for Searching for Solutions: Solving the Bull Trout Puzzle Science and Policy Conference, Andrus Center for Public Policy, Boise State University, Boise, ID, 1-2 June.

Frissell, C.A. 1995. Watershed dynamics: natural pattern and process and some consequences for ecosystem management. Invited presentations at Managing Terrestrial Ecosystems Relative to Past and Present Disturbances: A Workshop Integrating Fire, Range, Fish and Wildlife Habitat and the Practice of Silviculture in the Northern Region. U.S. Forest Service, Missoula, MT, 14-16 March.

Ebersole, J.L., C.A. Frissell, and W.J. Liss (co-presenters). 1995. Invasion of non-native fishes in northeast Oregon and western Montana streams: potential impacts of climate change. (Abstract). Oregon Chapter of the American Fisheries Society, Ashland, OR, 15-17 February.

Frissell, C.A. 1994. Watershed restoration strategies. (Invited presenter and session convenor) Watersheds '94 Expo, US Environmental Protection Agency and Center for Streamside Studies, University of Washington. Bellevue, WA, 27-30 September.

Frissell, C.A. 1994. A hierarchical approach to restoration of riverine ecosystems. Invited paper at Symposium on Aquatic Habitat Restoration in Northern Ecosystems, Alaska Chapter of the American Fisheries Society, Girdwood, AK, 20-22 September.

Selected Papers and Seminars Presented Since 1993, continued:

Frissell, C.A. 1994. An integrated, biophysical strategy for ecological restoration of large watersheds (Abstract). Annual Conference of The Universities Council on Water Resources, Big Sky, MT, 3-5 August.

Frissell, C.A., and J.A. Stanford. 1994. Designing a watershed reserve network to protect and restore aquatic biodiversity in the northern Rocky Mountains (Abstract). Annual meeting of the Montana Chapter of the American Fisheries Society, Billings, Montana, Billings, MT, 9 February.

Frissell, C.A. 1994. The Endangered Species Act: principles for the protection and recovery of fishes. Invited panel presentation, annual meeting of the Idaho Chapter of the American Fisheries Society, McCall, ID, 24-26 February.

Frissell, C.A., W.J. Liss, B. Doppelt, and D. Bayles. 1993. A new, ecologically based restoration strategy for Pacific salmon in the Pacific Northwest (Abstract). Annual meeting of the American Fisheries Society, Portland, OR, 29 August-2 September.

Technical Workshops Organized (selected):

Lead organizer and facilitator, New Science Implications for the Aquatic Conservation Strategy of the Northwest Forest Plan. Sponsored by the Coast Range Association, 2-3 December 2013, Portland, OR.

Co-organizer, with M. Scurlock and R. Kattelmann: SNEP Plus 15 Years: Ecological & Conservation Science for Freshwater Resource Protection & Federal Land Management in the Sierra Nevada. Sponsored by Pacific Rivers Council, Sierra Forest Legacy, UC Berkeley School Environmental Design, UC Davis Center for Watershed Science, and CaliforniaTrout; 12-13 December 2011, Davis, CA.

Organizer and facilitator, Workshop on Science for River and Watershed Conservation. Sponsored by Campaign for Montana's Headwaters, 7 October 2010, Flathead Lake Biological Station, Polson, MT.

Co-convener, with M. Scurlock and Kristen Boyles: Technical Workshop on Science for Forest Planning. Sponsored by Pacific Rivers Council and Earthjustice, 29 June 2010, Seattle, WA.

Organizer and panelist, Umpqua Independent Science Council. Sponsored by Pacific Rivers Council, 2010-2011.

Co-organizer and panelist, with Deanne Spooner and David Bayes: Workshop on Economics of ESA Critical Habitat Policy, sponsored by Pacific Rivers Council and San Francisco State University, October 4-5, 2007, San Francisco, CA.

Organizer and coordinator of Science Panel on Roads and Watersheds, sponsored by Pacific Rivers Council, 10-11 November 2006, Forest Grove, OR.

Organizer and coordinator of the Recovery Science Panel for the Western Native Trout Campaign. Sponsored by Pacific Rivers Council, meeting 2-3 March 2002, Portland, OR.

Technical Workshops Organized (selected), continued:

Organizer and coordinator of Biodiversity Workshop, Consortium for the Study of North Temperate Montane Ecosystems. A cooperative research venture of The University of Montana and Montana State University, supported by the NSF EPSCoR program. 4 February, 1997 Missoula, MT.

Scientific Workshop on Large Basin Restoration: Grande Ronde River (co-organizer). 21-22 March 1993, La Grande, OR. Sponsored by The Pacific Rivers Council.

Scientific Workshop on Large Basin Restoration: South Umpqua River. 16-18 September 1992, Roseburg, Oregon. Sponsored by The Pacific Rivers Council.

Scientific Workshop on Large Basin Restoration: Lower Rogue River. 21-23 October 1992, Gold Beach, OR. Sponsored by The Pacific Rivers Council.

Other Panels and Workshops Attended by Invitation since 1994 (selected):

Invited Review Panelist, Workshop on Linking Habitat Characteristics to Salmon Data. 29-30 September 1999, National Marine Fisheries Service, Northwest Fisheries Science Center, Seattle, WA.

Invited participant, Yellowstone to Yukon Aquatic Conservation Science Workshop. 20-22 August 1999, Flathead Lake Biological Station, The University of Montana, Polson, MT.

Invited Panelist, Workshop on Options for Restoring Salmon Habitat in the Mainstem Snake and Columbia Rivers. Pacific Northwest National Laboratory-Battelle, 19 August 1999, Kennewick, WA

Panelist at State of Oregon/National Marine Fisheries Service Memorandum of Agreement Committee Workshop: Cumulative Effects of State and Private Forest Practices on Salmon Habitat. 21April 1998, Salem, OR.

Invited participant in a scientific workshop, Multiple Stressors in Ecological Risk Management. Sponsored by the Society for Environmental Chemistry and Toxicology and the USEPA, 13-18 September 1997, Pellston, MI.

Society for Conservation Biology Workshop: Communicating with the Media (panel member). 9 June 1997, Victoria, British Columbia, Canada.

Invited speaker for a workshop, Continuing Education in Ecosystem Management. Sponsored by the University of Idaho. Catchment scale processes and linkages between landscape and stream conditions. 31 January 1997, Moscow, ID.

The Nature Conservancy, Aquatic Classification Workshop (invited presenter). 9-11 April 1996, Cedar Creek Farm, MO.

Kenai River Community Forum (keynote speaker and panelist). The Nature Conservancy of Alaska, USEPA and USFWS, 19-21 April, Soldotna, AK.

Conservation Biology and Management of Interior Salmonids (invited presenter and session co-moderator). USDA Forest Service Intermountain Research Station and Utah State University, 4-5 October 1995, Logan, UT.

Eastside Ecosystem Planning Workshop. Sierra Club Legal Defense Fund, 16 December 1994, Portland, OR.

Other Panels and Workshops Attended by Invitation since 1994, continued:

Co-instructor at workshop series on Watershed Restoration and the "Rapid Biotic Response Strategy" for Riverine Ecosystem Restoration, sponsored by The Pacific Rivers Council, 1993-95, California, Oregon, and Washington.

Fire/Salvage and Aquatic Ecosystems Policy Workshop. The Pacific Rivers Council, 15 December 1994, Portland, OR.

Panel on Forest Health Issues, Native Forest Network annual conference, 13 November 1994, Missoula, MT.

Workshop on Watershed/Fisheries Cumulative Effects Analysis, sponsored by Headwaters, The Pacific Rivers Council, USDA Forest Service, and Bureau of Land Management. 29 September-2 October, 1994, Ruch, OR.

Boise Funders' Scoping Meeting, sponsored by Bullit, Harder, and Lazar Foundations, 30-31 August 1994, Boise, Idaho.

Scientists Briefing for U.S. Senate staff on post-fire logging and forest management and freshwater resources. Washington, D.C., 18-19 September 2006.

Other Presentations and Outreach (Selected):

Invited testimony on federal land management and the future of salmon and aquatic biodiversity in the Pacific Northwest, to the U.S. House of Representatives, Subcommittee on National Parks and Public Lands, Washington, D.C., 11 March 1993.

Briefing for Congressional representatives and staff on federal lands management and conservation and recovery of salmonid fishes and riverine ecosystems, Washington, D.C., 22 January 1993.

Invited testimony to the 1991 Oregon State Legislature, on panel representing the Oregon Chapter of the American Fisheries Society, on the status of native fishes, impacts of forest practices on fish habitat, and the need or changes in environmental regulation.

Invited testimony to the Oregon Board of Forestry Forest Issues Forum, December 1990, on cumulative impacts of forest practices on native aquatic species and the need for changes in forest management.

Worked with Oregon Public Broadcasting to describe our research project and its significance in a 15-minute segment of the television program, Oregon Field Guide, first aired in June 1990.

Presented seminars, informal presentations, lectures, and discussions at research review meetings, as guest speaker in classrooms and public interest groups, at state board meetings, at workshops, and on field trips with foresters, geotechnical personnel, fishery and watershed managers, and conservationists.

Roadless in the Pacific Northwest: Ecology and History

James R Furnish, U.S. Forest Service, Iowa City, IA, United States

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Abstract

A great values clash unfolded in the late 1990's as the US Forest Service sought to resolve a long-festering environmental conflict over the fate of roadless public lands, comprised of 25 million hectares largely in the American West and Alaska; fully 2% of the entire land area of the United States. Conservationists had bitterly opposed the continued logging of these natural areas while the Forest Service relied on unlogged roadless lands to supply a beleaguered timber industry. The clash was particularly noteworthy in the Pacific Northwest, home of most of the remaining intact forests, following immediately on the heels of the spotted owl crisis that was resolved in favor of conservation interests to severely limit further clearcutting of old-growth forests. The Roadless Area Conservation Rule (2001) again resolved a consequential issue in favor of conservation interests by limiting commercial logging and roadbuilding in roadless areas. Lengthy litigation followed, culminating in 2013 when the Tenth Circuit Court of Appeals supported the Rule. A notable but unexpected outcome of this dramatic policy shift is a significant increase in stored forest carbon following logging reductions. Such carbon gains team with other "free" environmental services like clean water and air, and high quality wildlife and fish habitat to illustrate important non-commercial forest values.

The Untold Story of Roadless Area Protection

The legitimacy and relevancy of the US Forest Service stood at a crossroad: one road led down a familiar, increasingly arduous and costly path—that of pursuing logging and roadbuilding into untouched reaches of public forest; the other road led in a new direction—turning away from a utilitarian mindset to protection of 23 million hectares of natural lands (2% of all US lands), as yet untouched by commercial logging.

The answer emerged in due time; the Roadless Area Conservation Rule took effect on January 6, 2001 in one of President Clinton's last official acts.

What follows is a glimpse into the unseen bureaucratic battle to craft this regulation, as seen from my perspective as Deputy Chief of the Forest Service when I played a key role in bringing the regulation to fruition.

The Context

I assumed my duties as deputy chief in May 1999, and I had come prepared with my short list of priorities representing what I hoped would engage my best efforts to move the agency's national forests in a new direction.

I soon learned that the Forest Service Chief, Mike Dombeck, also had a short list that subsumed mine. I knew of his focus on watershed restoration, such as salmon recovery and protecting forest headwaters that supplied much of America's domestic waters. In fact, my interest and accomplishments in this area had everything to do with my being selected as deputy chief.

A major surprise lay in my path. Within days of my arrival, Dombeck informed me that the Forest Service would undertake promulgation of a regulation to protect roadless areas. Although I had a hint of this proposal from an earlier moratorium on road construction within inventoried roadless areas, the audacity and sweep of this proposal stunned me.

For decades, roadless areas had been the source, or pool of lands, from which Congress designated wilderness. Roadless lands also served as the best source of new timber because they generally had little or no prior logging. Roadless areas enjoyed no official status until the Forest Service undertook a national Roadless Area Review and Evaluation, begun in 1967 and completed in 1972. The RARE review, as it was called, recommended that 5 million hectares be set aside for wilderness. However, the accompanying environmental impact statement was challenged in court and judged to be deficient.

USDA's assistant secretary Rupert Cutler, a former executive with The Wilderness Society, initiated a subsequent Roadless Area Review and Evaluation (RARE II) during the Carter presidency in 1977, seeking to improve on the original study. RARE II recommended 6 million hectares for wilderness and another 10.8 million for further study.

Given this heightened interest, subsequent Forest Service logging and road-construction activities in roadless areas were often stopped by environmental lawsuits. The environmental community attached great importance to protecting the wildness of roadless areas, and they regarded the Forest Service as a threat to protecting roadless-area values.

This values conflict echoed the lengthy battles over cutting old-growth forests. Forest Service dogma called for logging to convert old forest to "healthy" stands of young, vibrant trees. Yet the agency's own research increasingly revealed that old-growth forests contained incredible diversity and complexity and served vital ecosystem functions. Far from being worthless, old growth held astounding value if one looked beyond mere timber products. The notorious spotted owl (*Strix occidentalis caurina*) controversy that dominated environmental politics in the Pacific Northwest in the late 1980's had been the point of the spear in halting the systematic elimination of old-growth forests.

Similarly, roadless areas had long been regarded by the Forest Service as merely the next place one goes to log more trees. Even so, in 2000, after nearly 100 years of Forest Service management, there still remained about 23 million hectares of inventoried roadless areas— equivalent to an area the size of Wyoming. Congress had already created about 14.5 million hectares of wilderness. Thus, in aggregate, about half of the 78 million total hectares of national forest remained undeveloped.

Wilderness areas were predominantly high-elevation lands. Roadless areas, in contrast, occupied much more of the niche of mid-elevation lands that tended to be more heavily forested. The timber industry, as well as many people in Forest Service leadership, continued to view roadless-area protection as antithetical to multiple-use mandates. The Forest Service seldom championed the wilderness cause, and could even be said to resist it.

For example, in 1984, after a crippling recession had hurt companies with Forest Service timber contracts, Oregon's Democrat Senator Mark Hatfield steered a bill through Congress that traded wilderness protection for a "release" of some roadless areas for immediate timber harvest. I recall, fresh upon arriving on the Siuslaw National Forest in 1992, looking at a road leading right into the bulls-eye of what had been one of the Siuslaw's few remaining large roadless areas near Hebo, Oregon. When built, the road (which accessed several 1985-era clear-cuts) permanently nullified the area's unmanaged, untrammeled character.

These roads were legal, thanks to Hatfield's legislation. However, the "release" language did not require that roadless lands be compromised by logging and road construction. The swift action taken by the Forest Service eloquently symbolized the agency's operational values. Protect roadless areas? Not if there were logs to be had. Furthermore, acting swiftly to compromise the area's roadless character eliminated the nuisance factor, making future management less complex.

Roadless-area protection and timber production had been engaged in a slow-motion collision for at least 20 years. You've seen bighorn rams size each other up in slow-motion? There will be a crash! Yet, it seemed clear to me that the Forest Service controlled both variables (protection and production) and could have engineered a durable solution. However, the Forest Service made itself—quite intentionally—the greatest threat to roadless areas. When Chief Dombeck proposed that the Forest Service develop a regulation for permanent protection of roadless areas, agency traditionalists felt downright insulted.

But Chris Wood, Dombeck's policy advisor, said that, in opting to protect roadless areas, "The Forest Service is becoming relevant again." I won't forget that remark. The sharp contrast between insulting and relevant made me ponder. I took Wood to mean that roadless areas were a huge issue to a vast constituency of environmentalists who viewed the Forest Service as an abuser rather than a protector of roadless areas and public lands in general. I thought this perception should have greatly alarmed agency leaders, but it didn't seem to.

Agency Dinosaurs?

The fissures that appeared after Dombeck's proposal revealed the agency as something less than one big happy family. Gloria Flora, Humboldt-Toiyabe National Forest supervisor in Nevada, made a rather public offhand remark about "dinosaurs" in Forest Service leadership who needed to retire. This outraged many forest supervisors who demanded that Chief Dombeck disavow her remarks.

Dombeck called me on the phone. "Could you come see me right away?" I took the back stairs to his office, which was just above mine. He asked me to read a letter he'd just received from Mike Lunn, Oregon's Siskiyou National Forest supervisor. Dombeck told me the letter had been electronically copied to "almost everybody" in the Forest Service. He asked what I thought he should do.

The letter was a direct, personal appeal to the chief outside the normal chain of command—except that it spoke to agency culture, not land management. Mike Lunn took umbrage at Flora's statement, and he gave the chief a hard-edged testimonial as to how the "dinosaurs," including himself, had built and sustained a great organization. The letter was a forceful defense of the old guard, tinged with defiance, contempt, and disrespect. The letter was also honest. Lunn was clearly calling Dombeck out as to how the chief felt about dinosaurs—like Lunn himself. The chief had an upcoming interview with *Time* magazine and now expected questions on this brouhaha.

Lunn's letter demanded an immediate response, and I suggested that Dombeck first call Lunn to thank him for his letter, for having the guts to tell him what he was thinking. I told him, don't argue; just be open, and listen if he's got more to say. Then reassure him that you value agency leaders and all they do and have done for the Forest Service. I said that Lunn needs to hear that the chief of the Forest Service doesn't consider him and his colleagues to be dinosaurs. Then, I said, write him a brief response, reiterating these few points. Address the reply to Lunn only. If Lunn wants to mail it around, let him. Don't escalate, but don't be dismissive.

Dombeck did this. In a few days, many thousands of agency employees had seen Lunn's letter as it spread via email. In these days before blogging, hundreds of commentators piled on via email using a "reply all" command, so that the effect multiplied

geometrically. Folks generally piled on adulation for Lunn's "telling it like it is!" I recall almost no recognition of Dombeck's response.

Emboldened, Lunn wrote yet another letter, basically doubling down and getting in more licks. Dombeck took the high road—no response this time. But the Lunn letters boldly revealed in our field leadership a lack of trust in Dombeck and feelings of betrayal. Hard stuff.

Ticking Clock

Hermann Gucinski, a Forest Service research scientist in Corvallis OR, was lead author of Forest Roads: A Synthesis of Scientific Information. This comprehensive report described both the benefits and the adverse consequences of roads and road use. Other research demonstrated that roadless lands are the best landscapes in terms of water quality, fish and wildlife populations, endangered species, and healthy forests. These lands are also bulwarks against invasive species because they have an essentially robust natural character that road construction tends to degrade.

The Forest Service now found itself caught in a powerful vortex, a result of failing at the critical task of solving prominent problems. Until now, I had been at the field level striving to facilitate logging to meet agency targets (as noted, much of the timber supply came from roadless areas) while being generally aware that the agency flailed in court defending such actions. This had been going on for some two decades, and attested that the former approach failed to resolve the issue—figuratively kicking the can down the road year after year, time and money both wasted. Effectively serving the public demanded that the Forest Service craft a progressive policy to deal decisively with roadless areas and resolve the issue.

I found it remarkable that politicians in the Clinton administration, notably Chief of Staff John Podesta and Council on Environmental Quality Chair George Frampton, trusted the Forest Service for the undertaking. One overwhelming, simple reason accounted for that: Chief Mike Dombeck. The chief often lamented that the Forest Service had become extremely defensive and apprehensive trying to defend its logging legacy. "It's going to be a lot more fun playing offense," he'd say, grinning broadly. He believed roadless-area protection would be very popular and likely to win the admiration and restore the trust of a cynical environmental community. News media often editorialized about how Forest Service leadership drifted weakly in the face of public antipathy, seeming to have run off the rails. Dombeck relished the idea of sitting down in front of editorial boards from the *New York Times*, W*ashington Post*, and *Los Angeles Times* to discuss the opportunity to contend for roadless-area protection.

Several difficult realities confronted the Forest Service in resolving one of the most sweeping and controversial issues of the past several decades. Completing a regulation governing 23 million hectares would affect virtually every national forest from coast to coast. Opposition would quickly mount, particularly from the conservative western states that held the vast majority of roadless lands. As anticipated, but sadly, some of the most intense opposition would come from within the Forest Service.

And there was the clock. This was summer 1999. A presidential election loomed in November 2000. Political reality required that a regulation be completed before President Clinton left office. There were two reasons: first, the Democrat Clinton wanted credit for a key environmental accomplishment; second, we needed to sustain momentum, because we knew the election of a Republican would stop the effort dead. We submitted a project calendar with all the requisite time periods—60 days for this, 90 days for that—indicating we could complete the project by mid-December 2000, after the election and before the January inauguration of a new president. An achievable assignment, but one with no margin for error or delay.

But we had no task team organized for the roadless issue. We were figuratively entering unknown territory without a map. Dombeck approached me with the challenge to find someone to lead the roadless effort—one of the most significant Forest Service challenges in its history. The job needed to be done well, it had to be done on time, and failure was not an option. I thought to myself, "I've got just the person in mind."

Bill Sexton, my old boss on the San Juan National Forest, was now laboring in a staff position in DC. He had openly voiced his hopes for a promotion to become a prestigious regional forester. I mentally connected the dots—take on the roadless issue, get it done, and the Forest Service will owe Bill Sexton a favor. Name your dream job. It's yours! It didn't hurt that Sexton was capable of doing just about anything he put his mind to.

I spoke with Sexton and made the pitch. His response was matter-of-fact: "Not interested. There will be a Republican president in 2000. Anyone who touches this roadless thing will be dead meat."

I bristled. I had not expected such a blunt assessment; so much for nuance. In that moment, any thought I had of cajoling or persuading Sexton into leading the roadless team vaporized because we needed a leader who believed in the work without reservations. I then called Sharon Heywood, Shasta-Trinity forest supervisor in California. Not interested. I put out an SOS call to all regional foresters. I needed help, urgently. Regional foresters acted stumped. Perhaps they agreed with Bill Sexton's sentiments and didn't wish to sacrifice anyone to political suicide?

Finally, Brad Powell, regional forester for California, called to say he had someone in mind: "Scott Conroy on the Modoc."

I felt a deep anxiety. I had met and knew most of the 120-plus forest supervisors, but I'd never even heard of Scott Conroy. And few considered the Modoc National Forest (office in Alturas, in the far northeast corner of California) to be an impressive locale. It seemed we were nearing the dregs of our candidate pool, when I was seeking the crème de la crème. I reminded Powell that this was the most important thing the agency had done in decades. Can we trust this issue to Conroy? A lot hangs on the outcome.

Powell told me that he thought Conroy might be just what we needed. He was competent, very disciplined, and could be relied on to get things done.

I reported to Dombeck on my efforts to find a person to lead the roadless effort. I noted my disappointment that the regional foresters couldn't help us identify any top-flight candidates. I also mentioned the Bill Sexton episode. Dombeck noted that at least Sexton was honest and agreed that we didn't want him: "We need a believer."

I got the nod to vet Conroy. Frankly, I had few options left. Maybe I needed to trust in Conroy's competence simply because the Forest Service always tried to pick capable people for forest supervisor jobs. That, coupled with Powell's endorsement, encouraged me. Then everyone I spoke to enthusiastically supported Conroy. Now it was time to speak to him myself.

Conroy immediately impressed me with his deep desire to help the Forest Service accomplish this historic task. He knew it would be tough sledding, but he liked a challenge. My concerns that he might be unequal to the task, or worse, oblivious to the difficulties, waned. Importantly, he believed roadless areas needed protection and wanted to be a part of it.

I said there were no guarantees, but if we could pull this off, he'd likely get a good promotion. Was he interested in greater responsibilities? Yes!

We had our team leader. I called Brad Powell to thank him for his help, though I still harbored doubts that we could succeed.

Making Hay

In the next few months, bureaucratic wheels turned quickly. We published a Notice of Intent in the Federal Register to prepare an Environmental Impact Statement and regulations to achieve roadless-area protection. Scott Conroy recruited eager, enthusiastic specialists of all kinds for his task team. We swung into our public involvement effort, the most ambitious in agency history. We conducted roadless-issue meetings for all 125 national forests, as well as many formal public hearings. We cooperated with other agencies such as BLM, Park Service, and Department of Justice. It was an incredibly hectic full-on press.

Since most roadless areas were in the West, timber industry groups and western Republican governors whipped up strong headwinds to blow the effort off course or, better yet, onto the rocks (Fig. 1). Chris Wood and I traveled to Salt Lake City to meet with several governors' representatives, all suspicious of the political overtones of the effort. The meeting was hostile and tense.

"Why now?" was the trenchant question. Everybody recognized roadless protection as a potent issue. We all knew the implications of the coming election as the political clock ticked on. Wood gamely explained that roadless issues had grown and festered for decades, fostering doubt, legal uncertainty, and needless expense. It served no one's interest to leave this important issue unaddressed. While Wood spoke, I got the feeling the gang mentally searched for the best way to stop us.

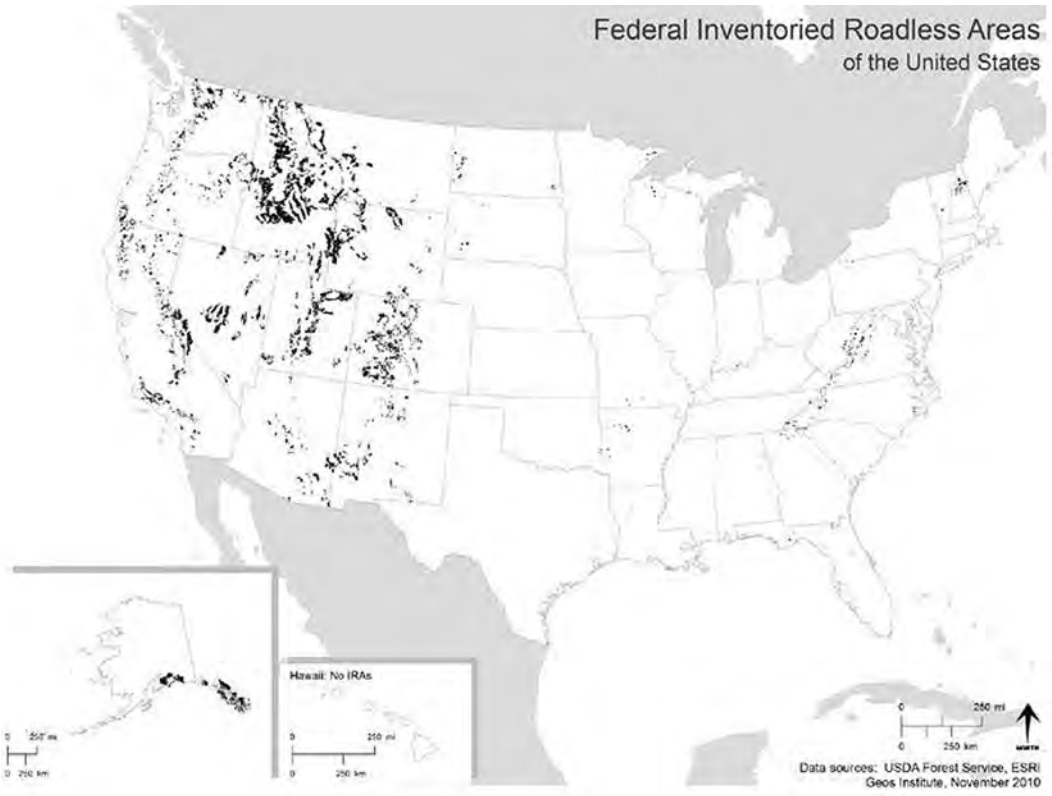

Fig. 1 Distribution of roadless areas throughout the United States, with heaviest concentrations in western states and Alaska. Source: USDA Forest Service.

Wood and I had anticipated that some of them would ask for cooperating agency status, which would essentially make them partners in our effort, a figurative bone thrown their way. But this would effectively give these prospective partners power to sabotage the project. And they did ask, but we fended them off, saying that it would be impractical and unworkable, since so many states were involved. If we made one state a cooperating agency, we'd have to do the same for all. We vowed to keep the states' governors informed throughout, but we were firm that only federal agencies would be cooperating agencies.

Pandemonium ensued. When Montana's representative pressed us hard on this issue, we asked why he counted this privilege as essential. He said, "We want to delay the process so that you can't finish the job before Clinton leaves office." They would take their chances from there on a hoped-for Republican White House pulling the plug on roadless-area protection.

Silently, I gave him high marks for honesty. After overcoming my shock at his brazen position, I stated that this was not a valid rationale for a cooperator. We left with a firm "No. You'll be hearing from us. Thanks for coming."

Other aspects of developing the rule flowed well. As the Forest Service held hundreds of public meetings, the volume of comment on the rule was massive, unprecedented. With Pew Environment Group funding a big advocacy campaign, the Forest Service received well over 1 million comments on our draft proposal. We had no precedent for processing and analyzing this volume of feedback, yet a task team heroically accomplished this as well. The trajectory of our effort held pretty tightly to the project calendar. We were still optimistic about concluding our work in December 2000.

Alaska, as usual, required special handling. Among the 23 ha of roadless areas, Alaska's two national forests, the Tongass and Chugach, contained about 5 million hectares—fully 20%. Of the two, the Tongass National Forest in southeast Alaska (remarkably, a bit larger than West Virginia) has long been a political hotbed. With 11,000 miles of coast and 57,000 miles of streams, the spectacular Tongass is regarded by the environmental community as the crown jewel of all national forests. For its part, Alaska's congressional delegation historically regarded the Tongass as its parochial domain, passing out logging and mining favors to local residents and businesses. The Forest Service is caught in a pincer-like grip between environmental constituents and powerful politicians.

With nearly 4 million hectares of Tongass roadless area distributed throughout this vast 7-million-acre archipelago of coastal rainforest (along with 3 million hectares of designated wilderness and National Monument lands), the destiny of these roadless areas loomed as a huge unresolved issue. Rather than propose any particular preference in the draft Environmental Impact Statement, we chose to defer the ultimate fate of Alaska's roadless lands to the final decision phase and simply invited public comments about the issue.

Mike Dombeck asked my opinion about the best future for Alaska's roadless lands. Having never visited Alaska, I told Dombeck I lacked a good enough feel for Alaska to provide a sensible answer. Thus, in August 2000, I made an unforgettable trip to Alaska with the chief's question in mind.

I immersed myself in the splendors of Alaska, accompanied throughout by Forest Service officials, primarily Rick Cables, the regional forester. Alaska's enormous spaces, raw exuberance, and uniqueness left me deeply touched, almost speechless. All Americans should visit Alaska to let it speak to their souls.

Over a period of several days, Cables conveyed that Alaska's roadless lands needed to remain available to local citizens for subsistence, commerce, and the basics of community sustainability. Uniquely, almost all of southeast Alaska is national forest land, meaning Forest Service management and decisions govern every aspect of life for Alaskans. In brief, Cables believed the lands' greatest value lay in utility, not protection.

Upon my return to Washington, I met with Dombeck to discuss my view on protecting Alaska's roadless lands. I described as best I could the nature and complexity of Alaska's roadless issue, ultimately siding with Cables's view that comprehensive, extensive roadless protection was unnecessary. Dombeck's face remained deadpan and he said little beyond thanking me. The few questions he did ask seemed rhetorical. I left with a sense that he disagreed with and was disappointed in my position.

Ultimately, the Forest Service decided to apply restrictions on further logging and road building to all of Alaska's roadless lands. Dombeck explained the key decision criteria to me like this: "Alaska's roadless lands are the best we have. We need to protect the best."

His logic had a simple, profound elegance that perhaps I failed to appreciate at the time. There exists much nuance between the poles of protection and wise use of lands. The choice is almost never clear, and claims are often mutually exclusive. I found the arguments of both sides compelling. Alaskans do rely heavily on the Tongass for the basics of life. Ecologically, the Tongass is of global significance. That makes the Tongass roadless area situation a wicked dilemma and why I found it difficult to see this clearly at a crucial moment.

In early December, we published the final Environmental Impact Statement and the proposed Roadless Area Conservation Rule, with a 30-day waiting period before it became effective. George Frampton, chairman of the Council on Environmental Quality (CEQ) and the president's leading spokesman on environmental matters, chaired a briefing for all federal agencies interested in the roadless proposal.

The Department of Energy was one very interested agency. The Roadless Rule proposed to ban any more road building in connection with oil and gas leases, beyond roads that existed or those needed for current leases only. North Dakota's Little Missouri National Grassland held a potentially huge but relatively undeveloped gas field for which restrictions on road building would make development, if not impossible, much more costly and difficult.

A contingent of several Department of Energy officials made an impressive (and late) entrance. The one woman among them wore a bright red dress, exuding power.

The leasing issue quickly arose. Frampton laid out the premise of the regulation on leasing. The Department of Energy quickly weighed in using strong, unequivocal terms: this must change. Frampton countered that the prohibition on new roads was a well-considered element of the rule. Acknowledging an explicit trade-off, Frampton said it was needed to protect roadless lands from the probability of more "dry holes" that would achieve little except to ruin more precious roadless lands.

The DOE contingent ratcheted up the argument, saying it would not accept any outcome other than making a last-minute change to the regulation.

It seemed clear to me that this conflict would have to come before White House Chief of Staff John Podesta and maybe President Clinton himself. And sure enough, in the next few days the regulation was rewritten to exempt any renewed lease from road-building restrictions. DOE won the argument, while CEQ gained some solace by retaining restrictions for all future leases. Prophetically, the lands in question later became a centerpiece of resurgent oil and gas industry in North Dakota as federal energy policy fostered a friendly environment for fracking.

The final rule was published in the Federal Register on January 6, 2001. Two weeks before George W. Bush assumed the presidency, William Jefferson Clinton, clad in a black overcoat, proudly made a presidential announcement at the National Arboretum, with gently falling snow dappling his shoulders. It seemed a perfect ending to a strenuous but deeply satisfying effort.

I reflected on all that happened in the past 18 months. I'd been given the opportunity to help resolve a festering conflict by fashioning regulatory protection for roadless areas. Nearing the end of my career, I had helped fashion a sweeping, bold policy for the environment. The same Forest Service that had been aggressively logging and building a vast network of roads into natural landscapes wheeled about-face to strive for durable protection—but protection from what? From the Forest Service itself.

Upon Reflection

It may prove useful to consider changes set in motion by roadless area protection that I consider to be profound: (1) a major victory for the environmental community alongside cessation of old-growth forest logging; (2) a new perspective forced on the Forest Service to consider more protective management options for roadless lands; and (3) forest carbon sequestration getting its due as vitally important to managing climate change factors.

Let's look at these changes through the lens of public land forestry in the vast Douglas fir (*Pseudotsuga menziesii)* forests of the Pacific Northwest where sharp contrasts exist between activity pre- and post-roadless protection since 2002.

The Pacific Northwest is home to much America's most intact forests, and these forests are also the most economically valuable. But any consideration of values must also include biological and even spiritual factors. The northern spotted owl controversy epitomized the clash of monetary and other at-odds values in a highly visible and protracted fight ultimately decided in the courts in 1991. Winners and losers became evident, but the courts offered no path forward. The Clinton administration resolved the legal morass by crafting a political pathway in 1994, in which commercial timber interests decisively lost ground following a decisive swing toward more ecologically conscientious policies. The environmental community declared victory but remained wary of backsliding (which did occur during the succeeding Bush administration).

Logging in the Pacific Northwest relied heavily on roadless lands for further old-growth harvests, even after the Clinton Plan reduced quotas from 9.6 million m^3 to 2.4 million per year. Thus, roadless area protection increased the bind by restricting commercial logging and road building. Whereas the spotted owl victory achieved a notable regional win for the environmental community, roadless protection achieved a monumental national victory. Remarkably, every legal challenge to the Roadless Conservation Rule failed in spite of the Bush administration electing to provide no defense. Environmental organizations intervened as friends of the court to mount a successful multi-pronged 12-year defense that ultimately vindicated the legitimacy of roadless protection.

These two remarkable environmental victories grew out of the Clinton administration's sensitivity to values other than commercial timber, and have largely endured due to changing public values and legal requirements. Yet insistent voices for the primacy of commerce on public lands remain effective, if wounded. As I write, the Trump administration prods the Forest Service via executive order to increase logging by 30% in hopes of reducing risk of catastrophic fire, such as Paradise CA experienced in late 2018. Personally, I doubt that logging one's way out of the current climate-driven fire conundrum will work.

The second point reflects, until now at least, the durability of roadless protection. Rather than logging and roadbuilding being the defining characteristics of Forest Service roadless policy, rules now legally require the agency to restrict commercial logging and new road construction in favor of protecting roadless area attributes and values like clean water and naturalness. This change clashes with a residue of traditionalist culture in the Forest Service, and has proven difficult to embrace.

The Bush administration fashioned a "work-around" whereby any state could petition to create unique roadless policies, though at considerable time and expense. Idaho and Colorado did so shortly after the original 2001 rule took effect, and Alaska and Utah began additional petition efforts in 2018–19. Forest Service officials quite eagerly made requisite cooperative gestures. In each case, the respective Governor chose to surgically loosen certain protections in the 2001 rule (such as for the ski industry in CO, and phosphate mining in ID), but the bulk of protections remain in effect currently. Thus, in spite of a few erosions, significant protections remain in effect to guide Forest Service actions in 23 million hectares of roadless areas, about 30% of the entire national forest land base.

The regressive move back to localized solutions unique to individual states reflects a long-standing reluctance to deal with national issues at a national, agency-wide level (e.g., clearcut logging, old-growth forest protection). The 2001 Roadless

Conservation Rule sought to resolve a decades-old national scale issue at a national level, after repeated local efforts had failed to pass legal muster, nor provide consistency. The environmental community exhibited enthusiastic support for a comprehensive policy that restricted commercial logging and roadbuilding—one governing all national forests—while critics (including many Forest Service officials) groused at loss of local control.

Furthermore, some Western states, notably Utah, remain fixed on converting federal lands to state ownership—a solution that would moot federal roadless area protection. The notion that federal lands should be given to states is deeply rooted in the premise that the federal government's ownership of public lands in western states is illegitimate. This notion is in direct contravention of the Property Clause of the US Constitution (Article 4), whereby the federal government reserved ownership of public lands unto itself when granting statehood to western states, and has been settled law based on Supreme Court rulings for over 100 years. The notorious 2016 occupation of the Malheur National Wildlife Refuge by "The Bundy Brothers" symbolized this fanciful movement.

Thus, although protection of roadless lands has strong and tested legal underpinnings, the political arena remains somewhat fraught.

Thirdly, the importance of forests in mitigating climate change dynamics has emerged relatively recently, both globally as well as in the Pacific Northwest region where Douglas-fir forests are among Earth's most prodigious carbon stores. Yet the relationship between climate change and forest carbon went largely ignored as recently as 2000 by those of us formulating the Roadless Area Conservation Rule. Remarkable new data from Forest Service inventories of forest carbon tells an interesting tale.

Using Oregon as an example, we see that the capacity of forest lands to store carbon increases when timber harvest declines (Fig. 2). Recall that the Northwest Forest Plan (1994) reduced federal timber harvest from 9.6 million m^3 per year to 2.4 million. That plan, in combination with restrictions on commercial timber sales in roadless areas, resulted in significant increases in storage of forest carbon between 1996 and 2012 (see Figs. 2 and 3). The annual rate of acquired carbon on national forests increased from 13 million to 34 million metric tons of CO_2 between 1996 and 2012. In fact, in the short time period 1990 to present, Oregon's federal forest lands have shifted from being a net source of carbon to being a very large sink today. This contrasts with private industrial forest lands that have reduced carbon stores in the same period, largely due to continued heavy logging.

Total national forest timber harvests have plunged from a high of about 26 million m^3 per year to the current level of about 10 million. Extrapolating from the example of Oregon, carbon stores on national forests increased as logging decreased. If carbon were to be monetized at rates considered by the Office of Management and Budget, its value would greatly exceed that of wood products. This is not to say that logging on public lands should be eliminated. But the legacy of important conservation measures like the Roadless Area Conservation Rule has been to increase forest-based carbon significantly, even though this may have been an unintended consequence. In light of the urgency of climate change, forest carbon should be an important decision criterion in weighing the relative merits of carbon sequestration versus logging in federal forest policy discussions.

In conclusion, I believe my long Forest Service career gives me legitimacy to suggest that many agency officials remain unenthusiastic about the obligation to protect, rather than log, roadless lands. Time, research, and insight have demonstrated the long-term benefits associated with conserving roadless area character and values. The Roadless Area Conservation Rule remains among the most controversial policy initiatives in Forest Service history, and also among the most environmentally consequential. The federal government chose not to defend the Rule in court, yet it prevailed against numerous challenges due to dedicated defense by environmental groups. The Bush administration tried to scuttle the Rule but failed. Once again, the Trump administration seeks to water it down in Alaska and Utah. My hope has been that as time rolls on, the wisdom of protecting roadless values would become more apparent, obvious even. In sum, the roadless rule stands as a landmark national legacy and an important example of global opportunities available for other nations if we are going to slow down climate change.

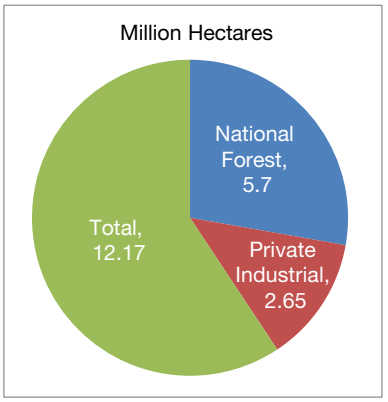

Fig. 2 Amount of forest lands in Oregon. Source: USDA Forest Service.

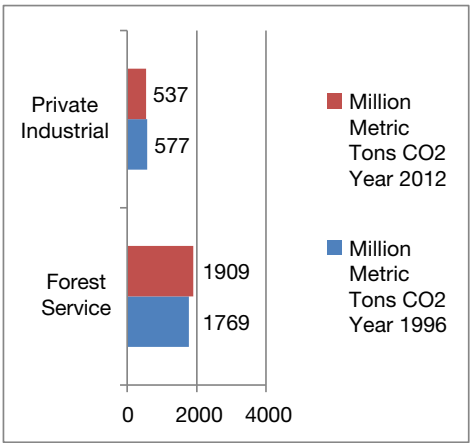

Fig. 3 Changes in forest carbon 1996–2012. Source: USDA Forest Service.

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Chapter 8

Road Construction and Maintenance

M. J. Furniss, T. D. Roelofs, and C. S. Yee

Forest and rangeland roads can cause serious degradation of salmonid habitats in streams. Numerous studies during the past 25 years have documented the changes that occur in streams as a result of forest and rangeland roads and related effects. Once the mechanisms of these changes are understood, it is possible to design roads that have less harmful effects on stream channels and their biota.

Only recently have steps been taken to minimize the negative effects of roads on streams. In the past, the primary considerations in road planning, construction, and maintenance have been traffic levels and economics, and little concern was expressed for the environmental influences of roads (Gardner 1979).

It should be recognized that only rarely can roads be built that have no negative effects on streams. Roads modify natural drainage networks and accelerate erosion processes. These changes can alter physical processes in streams, leading to changes in streamflow regimes, sediment transport and storage, channel bank and bed configurations, substrate composition, and stability of slopes adjacent to streams. These changes can have important biological consequences, and they can affect all stream ecosystem components. Salmonids require stream habitats that provide food, shelter, spawning substrate, suitable water quality, and access for migration upstream and downstream during their life cycles. Roads can cause direct or indirect changes in streams that affect each of these habitat components.

Many studies have shown how roads affect the physical environment of streams, and how the physical environment of streams affects fish. This research permits the diagnosis of problems and the design of engineering solutions to reduce negative effects.

Effects of Roads on Streams

Roads can affect streams directly by accelerating erosion and sediment loadings, by altering channel morphology, and by changing the runoff characteristics of watersheds. These processes interact to cause secondary changes in channel morphology. All of these changes affect fish habitats.

Accelerated Erosion Rates

Construction of a road network can lead to greatly accelerated erosion rates in a watershed (Haupt 1959; Swanson and Dyrness 1975; Swanston and Swanson 1976; Beschta 1978; Gardner 1979; Reid and Dunne 1984). Increased sedimenta-

Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats
American Fisheries Society Special Publication 19:297–323, 1991

tion in streams following road construction can be dramatic and long-lasting. The sediment contribution per unit area from roads is often much greater than that from all other land management activities combined, including log skidding and yarding (Gibbons and Salo 1973).

Sediment entering streams is delivered chiefly by mass soil movements and surface erosion processes (Swanston 1991, this volume). Failure of stream crossings, diversions of streams by roads, washout of road fills, and accelerated scour at culvert outlets are also important sources of sedimentation in streams within roaded watersheds.

Mass soil movement.—Where forest and rangelands occur on steep terrain, mass soil movement is often the primary mode of erosion and sediment delivery to streams from roads. Four types of mass movement common to western forest lands are described by Swanston (1991): slumps and earthflows, debris avalanches, debris flows, and debris torrents. These processes are differentiated on the basis of speed of travel and shape of the failure surface. Forest and rangeland roads can increase the incidence and severity of each type of mass movement. Several studies in the western Cascade Range in Oregon showed that mass soil movements associated with roads are 30 to more than 300 times greater than in undisturbed forests (Sidle et al. 1985) (Table 8.1).

Construction of roads can increase the frequency of slope failures from several to hundreds of times, depending on such variables as soil type, slope steepness, bedrock type and structure, and presence of subsurface water. Road location is the most important factor because it affects how much all of these variables will contribute to surface failure (H. W. Anderson 1971; Larse 1971; Swanston 1971; Swanston and Swanson 1976; Lyons and Beschta 1983). Mass soil movements triggered by roads can continue for decades after the roads are built.

The most common causes of road-related mass movements are improper placement and construction of road fills, inadequate road maintenance, insufficient culvert sizes, very steep hillslope gradient, placement or sidecast of excess materials, poor road location, removal of slope support by undercutting, and alteration of slope drainage by interception and concentration of surface and subsurface water (Wolf 1982).

Surface erosion.—Surface erosion from roadbed surfaces, drainage ditches, and cut-and-fill surfaces can severely affect streams below the right-of-way (Burns 1970; Brown and Krygier 1971; Larse 1971; Gibbons and Salo 1973; Weaver et al. 1987). In a study on the Clearwater River in Washington, Cederholm et al. (1981) found that the percentage of fine sediment in spawning gravels increased above natural levels when more than 2.5% of basin area was covered by roads. The chief variables in surface erosion are the inherent erodibility of the soil, slope steepness, surface runoff, slope length, and ground cover.

Surface erosion can be the major source of sediment delivered to streams in sensitive terrain, such as areas with soils derived from granite and from highly fractured sedimentary rocks. Surface erosion and piecemeal mass movement from landslide surfaces can prolong sediment delivery to streams after initial landslide events. Such chronic secondary erosion could be as damaging to stream biota as catastrophic or episodic sediment inputs from mass-movement events, because the particles will be finer and they will be delivered over longer time periods.

TABLE 8.1.—Comparative rates of soil mass movement with various land uses, based on landslide inventories over relatively long time periods. (From Sidle et al. 1985.)

Area (source[a])	Land use	Years of record	Number of landslides	Mass erosion: Annual rate (m^3/hectare)	Mass erosion: Multiple of undisturbed land
H. J. Andrews Forest, Oregon Cascade Range: unstable zone (1)	Forest	25	32	0.87	1.0
	Clear-cuts	25	36	2.45	2.8
	Roads	25	71	26.19	30
Oregon Cascade Range: Alder Creek (2)	Forest	25	7	0.45	1.0
	Clear-cuts	15	18	1.17	2.6
	Roads	15	75	155.65	346
Oregon Cascade Range: Blue River (3)	Forest	34	19	0.37	1.0
	Clear-cuts	22	30	3.22	8.7
	Roads	25	69	16.3	44
Oregon Coast Range: Mapleton area (4)	Forest	15		0.19	1.0
	Clear-cuts	15		0.7	3.7
Oregon Coast Range: Mapleton area, soil type 47 (5)	Forest	15	34	0.28	1.0
	Clear-cuts	10	186	1.13	4.0
	Roads	15	41	34.91	125
Oregon Coast Range: Mapleton area, slideprone soils (5)	Forest	15	42	0.32	1.0
	Clear-cuts	10	317	0.62	1.9
	Roads	15	89	15.85	50
Olympic Peninsula, Washington (6)	Forest	84	25	0.7	1.0
	Clear-cuts	6	0	0	0
	Roads	6	83	117.74	168
Coastal southwest British Columbia (7)	Forest	32	29	0.11	1.0
	Clear-cuts	32	18	0.25	2.3
	Roads	32	11	2.83	26
Idaho Batholith: Pine Creek burn (8)	Forest			0.07[b]	1.0
	Burned[c]	2	9	1.38	20
	Roads[d]	2	9	13.15	188
North Westland, New Zealand (9)	Forest			1.0[e]	1.0
	Clear-cut[f]	3	25	11.8	11.8
	Clear-cut[g]	3	72	40.4	40.4
	Roads[h]	3	32	267	267

[a]Sources: (1) Swanson and Dyrness (1975); (2) Morrison (1975); (3) Marion (1981); (4) Ketcheson and Froelich (1978); (5) Swanson et al. (1977); (6) Fiksdal (1974); (7) O'Loughlin (1972); (8) Gray and Megahan (1981); (9) O'Loughlin and Pearce (1976).
[b]Estimated from Zena Creek data (Megahan and Kidd 1972).
[c]Includes some clear-cut areas.
[d]Assumed that 6% of land area was in the road right-of-way.
[e]Estimated.
[f]Gravel substrate.
[g]Sandstone substrate.
[h]Calculations based on 4% of land area occupied by roads and 25% of observed landslides directly related to roads.

Surface erosion from road networks is usually reduced through time by natural revegetation (Beschta 1978), and it can be controlled by mulching, reseeding, and mechanical slope protection (Dyrness 1970; Megahan 1974; U.S. Environmental Protection Agency 1975; Carr 1980; Carr and Ballard 1980). Reid and Dunne (1984) found that sediment yield from roadbeds increased greatly with the amount of truck traffic. Sediment loss from road surfaces is partially a function of the surface composition and road maintenance.

Other sources.—Other erosional processes can cause accelerated erosion and sediment delivery to streams. Failure of road fills is common in steep terrain, particularly on low-standard roads where road fill is not compacted or woody material is incorporated into the fill.

Where flow restrictions such as culverts are placed in stream channels, the scouring power of streamflow is increased. This can lead to increased channel scour, streambank erosion, and undermining of the crossing structure and fill.

Failure of stream crossings can be a major source of increased sediment loading of streams. When stream crossings fail, they often do so catastrophically, causing extensive local scour and deposition and additional erosion downstream. Stream-crossing failures that divert streamflow onto nonstream areas are particularly damaging and persistent (Weaver et al. 1987).

Alterations in Channel Morphology

A stream adjusts its geometry to accommodate the water and sediment it carries. When the amount of water or sediment a stream must carry increases, channel geometry must change to accommodate the increase. When channel geometry is artificially changed, such as by a stream crossing, a stream will adjust by altering its geometry upstream or downstream of the change. The nature of the adjustment depends on the original geometry and composition of the channel, how these are changed, and the ability of the channel to reshape itself. Channel adjustments that occur, in order of smallest to largest energy requirements, are changes in channel bed form, channel bed armor, channel width, channel pattern (alignment), and longitudinal profile (Heede 1980).

Hagans et al. (1986) demonstrated that road construction and inadequate maintenance lead to substantial increases in stream-channel drainage densities and channel dimensions. Adjustments can occur quickly, but often continue over many years. The adjustments usually are detrimental to fish habitat. Therefore, road crossings that modify and restrict channel geometry least, such as bridges or low-water crossings, are likely to have the least adverse effects on fish habitat.

Channel morphology is also sensitive to indirect changes resulting from other effects that roads may have on streams. Increases in sediment loading and peak flows cause changes in channel morphology that can be detrimental to fish habitat.

Other Effects of Roads on Streams

Although sedimentation and stream-channel changes are the primary negative effects of roads on streams, roads can adversely affect streams in other ways. These include changes in rainfall–runoff relationships, hillslope drainage, potential for chemical contamination, the amount and type of organic debris in stream channels, and human access to streams and fish populations.

Roads can change the stream hydrograph and affect sediment deposition in streams. Harr et al. (1975) reported an increase in peak flows following road construction. King and Tennyson (1984) found that the hydrologic behaviors of small forested watersheds were altered when as little as 3.9% of the watershed was occupied by roads.

Hauge et al. (1979) discussed several ways that roads can affect hillslope drainage, including changes in infiltration rates, interception and diversion of subsurface flow, changes in the watershed area of small streams, changes in the time distribution of water yield to channels, and changes in the fine (micro) details of drainage. These changes combine to cause a rerouting of hillside drainage that can lead to changes in erosion and the hydrologic behavior of small streams.

Chemicals used to suppress dust, stabilize or deice road surfaces, and fertilize or control roadside vegetation can enter streams directly or can be transported by runoff water or on sediment. Little is known about the effects of these chemicals on stream biota. Furthermore, a chemical-spill hazard exists wherever roads are near streams or road drainage enters streams.

Organic debris from construction clearing and landslides caused by roads can enter and block streams. These materials can cause additional erosion and alteration of channel morphology, and can form migration barriers. However, they can also provide important cover and channel diversity for juvenile fish. Removal of large organic debris at stream crossings can eliminate important components of fish habitat.

Roads allow easier human access to streams, facilitating both legal and illegal fishing. They also give access to biologists for fish habitat and population assessment, and for habitat restoration and enhancement projects. In some cases a road can be a positive contribution to a fisheries management program for a stream, provided the road is located, designed, constructed, and maintained to protect fish and fish habitats.

Effects of Roads on Salmonid Habitats

The habitat requirements of salmonids are reviewed by Bjornn and Reiser (1991, this volume). The particular habitat requirements for each salmonid species vary with the season and life cycle of the fish. All salmonids require access to spawning areas, appropriate substrates for reproduction (including substrates that can support egg incubation, alevin development, and fry emergence), and suitable water quality. Species that rear in streams for months to years before they enter the ocean also require food organisms and shelter or cover. Physical alterations in sediment loading, channel morphology, substrate composition, riparian conditions, and other road-related changes can adversely affect all freshwater stages of these fish: migration, spawning, incubation, emergence, and rearing.

Migration

Improperly designed roads can prevent or interfere with upstream migration of both adult and juvenile salmonids in several ways. Macroinvertebrate movements can also be impaired or prevented by road-related changes to stream channels (Pearce and Watson 1983). Culverts pose the most common migration barriers associated with road networks. Hydraulic characteristics and culvert configura-

tion can impede or prevent fish passage (discussed in detail later in this chapter). Extreme sedimentation from roads can cause streamflow to become subsurface or too shallow for upstream fish movement. Likewise, stream-crossing structures can impede gravel movement in streams, leading to bed aggradation and subsurface flows that block migration. Large landslides or debris avalanches can form temporary dams that prevent fish passage (Pearce and Watson 1983). In very cold climates, large ice buildups at culverts can create barriers to migration.

Spawning

Adult salmonids have exacting habitat requirements for spawning, including requirements for substrate sizes, water depth, and velocity (Bjornn and Reiser 1991). The abundance and quality of spawning substrate can be severely affected by sedimentation. Fine sediment can be deposited in gravel interstices, even in fast-moving streams, because of the lower water velocities within the gravels. If the amount of fine material in the gravel matrix is too great, the gravels may become so cemented or indurated that fish are unable to excavate a redd.

In low-velocity stream reaches, an excess of fine sediment can completely cover suitable spawning gravel, rendering the reach useless for spawning. Excessive sediment loading of streams can also result in channel braiding, increased width:depth ratios, increased incidence and severity of bank erosion, reduced pool volume and frequency, and increased subsurface flow. These changes can result in a reduction in quality and quantity of available spawning habitat.

Gravel extraction for road construction may directly remove suitable spawning substrate. In some cases, gravel removal creates additional spawning areas, but such gravels often are hydraulically unstable. This can attract spawners to gravels that will not stay in place long enough to successfully incubate embryos.

Incubation

Successful incubation of salmonids in stream gravels depends on intragravel water flow to provide oxygen and to remove carbon dioxide and other waste metabolites (Bams 1969). If the gravel interstices are filled with fine sediments, intragravel water flow and gas exchange are reduced and egg development is slowed or halted. Fry emergence is likewise hampered by excessive fine sediments that can trap fry in the gravel (Phillips et al. 1975).

The gravels of redds must be stable throughout the incubation period. Developing embryos can be destroyed by gravel scour resulting from peak flows. Increases in peak flows and sedimentation can increase the incidence of redd destruction by scouring.

Juvenile Rearing

Increased sediment in streams can adversely affect juvenile salmonids in several ways. Most of the food items in the diets of juveniles are macroinvertebrates living in the stream. Large amounts of fine sediment reduce or eliminate much of the suitable substrate for producing macroinvertebrates, thereby limiting the food available to juvenile fish (Cordone and Kelley 1961).

Excessive sediment delivery to streams can modify the stream channel configuration, decreasing the depth and number of pools and reducing the physical

space available for rearing fish. These changes can also lead to reduced survival of juvenile fish by filling interstitial spaces in the boulder and large cobble substrates where fish reside over the winter.

Riparian vegetation provides important components of rearing habitat, including shade (which often maintains cool water temperatures), food supply, channel stability, and channel structure. Road construction near streams often removes riparian vegetation directly. Mass soil movements and channel changes resulting from roads can also eliminate or damage riparian vegetation. The essential role of large woody riparian debris in salmonid streams was reviewed by Bisson et al. (1987).

Everest et al. (1987a) reviewed the effects of fine sediment on fish habitats and fish production. They demonstrated that the effects of fine sediment on salmonids are complex and depend on many interacting factors, including species and race of fish, duration of freshwater rearing, spawning escapement within a stream system, presence of other fish species, availability of spawning and rearing habitats, stream gradient, channel morphology, sequence of flow events, basin lithology, and history of land use.

How to Prevent or Minimize Damage

The basic strategy to prevent or minimize damage from roads is to understand the physical and biotic conditions that could be affected. Then, planning should ensure that roads are designed, constructed, and maintained to reduce the risks of erosion; that the risk of eroded material entering streams is low; that disturbances to channel morphology will be reduced or eliminated; that the alteration of hillslope drainage patterns will be minimized; and that fish will be able to migrate past stream crossings.

Four general principles should be considered to control erosion resulting from roads.

- Know what the erosional processes are, how roads can affect these processes, and the appropriate measures to prevent or control changes in erosional patterns.
- Avoid building roads in areas with high erosion hazards to the greatest extent possible. Minor changes in location can often prevent major problems. This is usually the single most important consideration in preventing degradation of fish habitat.
- It is almost always less expensive and more effective to design and build roads so that erosion is prevented or minimized than to control sediment once it is mobilized. Remedial measures for major erosional events usually are much more costly than preventing the events in the first place.
- Minimize the effects of roads on streams by keeping road disturbances as far from streams as possible, and by providing buffers of undisturbed land between roads and streams.

Planning and Reconnaissance

Larse (1971) pointed out that the most important steps that can be taken to minimize the impacts of roads on streams usually occur during planning, recon-

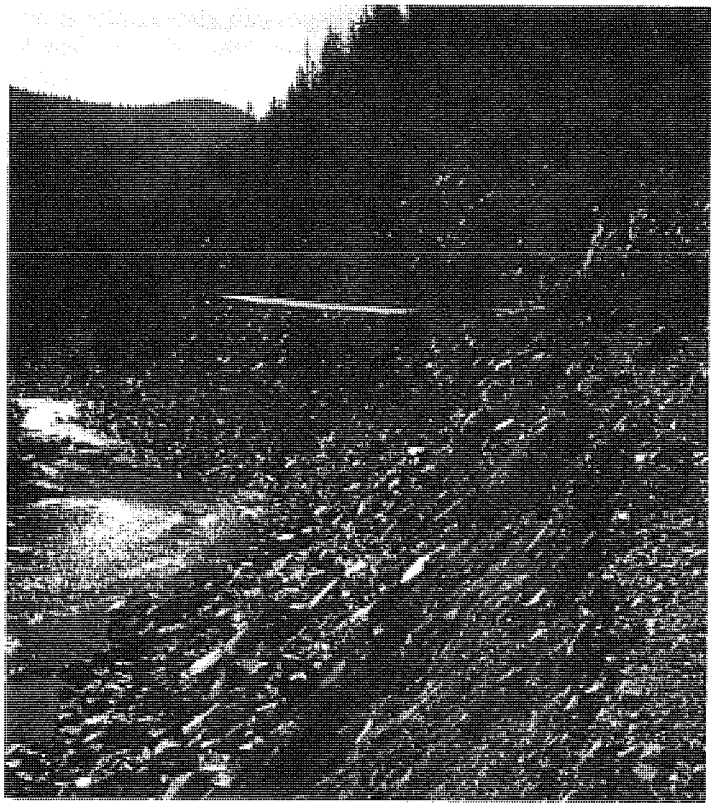


FIGURE 8.1.—Roads built near watercourses often have severe adverse effects on fish habitats. Sediments eroded from exposed and disturbed ground by rain, runoff, or streamflow move directly into stream habitats.

naissance, and route selection rather than during or after construction. Many problems can be eliminated or reduced by including on the planning team specialists such as geologists, soil scientists, hydrologists, and biologists, along with engineers. Key environmental problems and constraints are easily overlooked when routes are located and roads are designed by one person.

The following guidelines will help reduce adverse effects of roads on streams.

- Conduct long-range transportation planning for large areas to ensure that roads will serve future needs. This will result in less total road distance, roads built to appropriate standards, reduced costs of development, and fewer effects on streams.
- Use adaptable road standards to avoid sensitive areas. The prevailing planning philosophy should be to *fit the road to the landscape*. Rigid design standards, especially when limits on grade are inflexible, can severely restrict options for location, and often result in roads that pass through areas with high erosion hazards. Short grades of 14–22% can be practical for low-volume roads, and they can make it possible to avoid landslides or other sensitive areas.

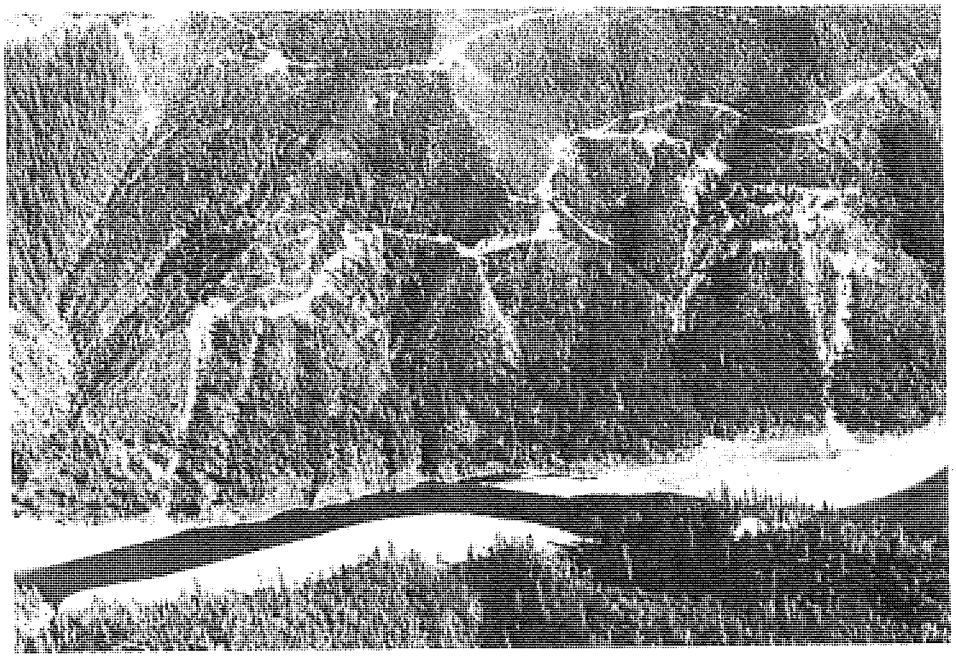


FIGURE 8.2.—Steep slopes close to waterways are high-risk locations for roads because they usually are the most erosion-prone areas in a watershed. Sediments eroded here are readily delivered to fish habitats.

- Avoid midslope locations in favor of higher, flatter areas. Ridgetop roads usually have the least effect on streams.
- Locate ridgetop roads to avoid headwalls at the source of tributary drainages. Headwall areas also are very prone to landslides (Weaver et al. 1987).
- Do not locate roads within the inner valley gorge (the oversteepened slopes adjacent to streams). These areas have the highest incidence of landsliding of any portion of an upland watershed (Wolfe 1982) (Figures 8.1, 8.2).
- Avoid slopes that show signs of excessive wetness, such as springs or water-loving vegetation.
- Avoid slopes where sidecast material could enter streams, or plan to end-haul excess material.
- Avoid slopes where large cuts and fills would be required.
- Locate roads to minimize roadway drainage area and to avoid modifying the natural drainage areas of small streams.
- Locate valley-bottom roads to provide a buffer strip of natural vegetation between the road and stream.
- Locate roads to take advantage of natural log-landing areas, such as flatter, higher, drier, and more stable terrain with good access to the timber to be removed. Good landing locations can also reduce the amount of necessary roading.
- Minimize the number of stream crossings consistent with the above considerations.

FIGURE 8.3.—A low-impact road fitted to the landscape with a narrow roadbed, narrow clearing limits, small cuts and fills, and outslope drainage. There is no inboard ditch to concentrate the erosive power of running water.

- Locate stream crossings to minimize channel changes and the amount of excavation or fill needed at the crossing.

Design

The following guidelines for road design will help minimize adverse effects on salmonid habitats.

- Use the minimum design standards practical with respect to road width, radius, and gradient (Figure 8.3).
- Minimize excavation with a balanced earthwork design wherever possible. Bench or terrace and drain natural slopes to provide a sound foundation for embankments.
- Design cut slopes to be as steep as practical. Some sloughing and bank failure is usually an acceptable trade-off for the reduced initial excavation required.
- Determine the type and extent of fish habitat before selecting criteria for drainage structure design. Bridges and arch culverts are preferred for streams with migratory fish. Where culverts are used, the gradient should be less than 1%, the culvert should be placed at or below the original streambed elevation, and

FIGURE 8.4.—Stream crossings pose the greatest risk to fish habitats of any road feature. When culverts are plugged by debris or overtopped by high flows, road damage, channel realignment, and severe sedimentation often result. The capacity of the culvert shown here was exceeded during a heavy storm, and a large slug of sediment was rapidly delivered to fish habitats downstream. Water was not diverted down the road, however. Roads should be constructed so they will not divert streams; where existing roads have the potential to do so, creation of a simple dip or "failure point" will prevent the catastrophic effects that can result from a diversion.

water depth and velocity at both low and high flows should be integrated into the design (discussed in detail later in this chapter).

- Control scouring at culvert outlets with energy dissipators such as heavy rock riprap, weirs, or gabions, consistent with fish passage considerations.
- Design drainage structures to accommodate peak streamflow based on at least a 50-year-interval flood (100-year flood for large permanent bridges and major culverts), and give consideration to the possibility that bed load and debris will restrict the flow capacity of the structure. The risk of failure can be calculated by

$$F=1-(1-1/t)^n;$$

F is the chance of failure during the design life, t is the flood recurrence interval, and n is the design life of the road or structure. For example, a culvert sized for a 50-year flood has a 33% chance of failure during a 20-year design life. Campbell and Sidle (1984) described methods for predicting peak flows and sizing culverts for small watersheds. Keep in mind that, whatever the design life, any crossing structure has a virtually 100% chance of failing over its installation life if it is not removed after the road is abandoned.

- At stream crossings, avoid channel-width changes and protect embankments with riprap, masonry headwalls, or other retaining structures. Align culverts with the natural course and gradient of the stream. Debris that floats during high

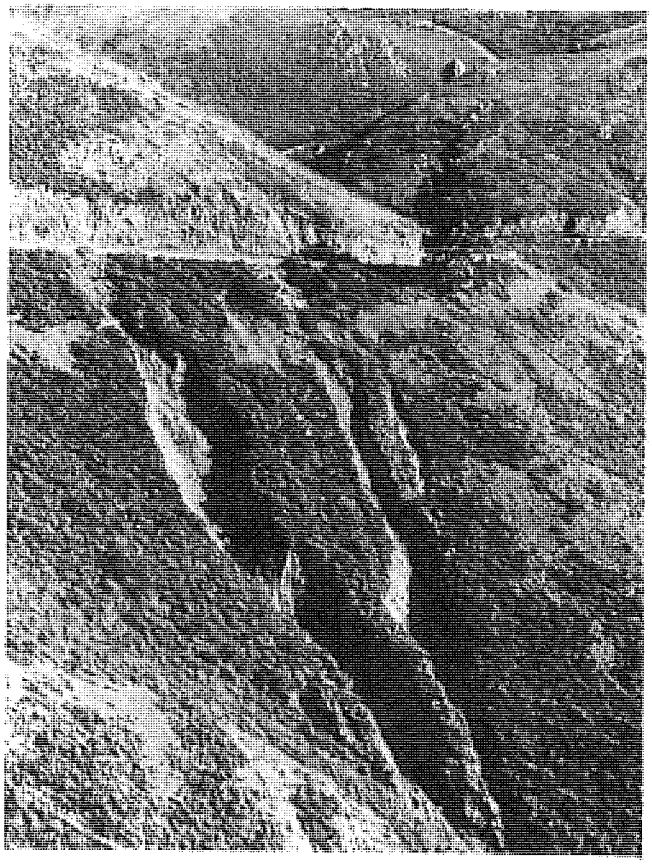


FIGURE 8.5.—If road drainage becomes concentrated, it must be discharged into places that can handle the flow without accelerated erosion. When the natural drainage of small streams is changed and concentrated flow is discharged into nondrainage areas, severe gullying and landslides can result.

streamflow can plug or restrict flow at culverts, causing severe changes in road embankments, streambanks, or channels (Figure 8.4). Trash racks can reduce culvert plugging, but can easily become barriers to fish passage. Avoid the need for trash racks by designing culverts to pass debris downstream.

- Wherever possible, disperse drainage rather than concentrating it, except in streams. Always strive to keep water flowing where it would naturally flow.
- Avoid the discharge of large amounts of concentrated runoff onto nondrainage areas (Figure 8.5).
- Do not change natural drainage areas by means of culvert or waterbar placement.
- Surface forest and rangeland roads wherever practical to control erosion and to maintain the surface drainage configuration under expected traffic conditions. Design road surfaces to remain stable and erosion-resistant during the wettest period of use. Control access on roads intended for dry-season use only.
- Use outslope drainage wherever feasible to disperse runoff. This results in the least potential for erosion and does not require as wide a road (no ditchline).

Generally, outslope drainage works well where sideslopes are greater than 20% and grades are less than about 12%, and where surface configuration can be maintained. Insloping with frequent cross-drainage is appropriate on roads with steeper grades. Cross-drainage on outsloped roads should be considered as a backup to outsloping, which can become ineffective under some traffic and surfacing conditions.

- Where inslope-and-ditch drainage is used, relieve the ditchline of drainage at frequent intervals onto areas that will not erode excessively or cause sediment to enter streams. Special care must be taken to avoid discharging drainage onto areas prone to gullying, slumping, or landsliding. Ditches along steep grades and in sensitive areas should be lined with rock to control ditchline erosion.
- When discharging drainage onto a long, erodible fill, use a discharge pipe or flume to convey the drainage to the bottom of the fill. Place energy dissipators at the outlet.
- Provide for vegetative or mechanical stabilization in areas where cut-and-fill erosion will cause sediment delivery to streams. Several publications describe erosion-control measures for cut-and-fill surfaces (Patric 1976; Carr 1980; Carr and Ballard 1980; Rothwell 1983; Swift 1986).
- Keep approaches to streams as close to right angles as possible to minimize streamside disturbances.
- Develop a specific plan for stream-crossing construction that addresses stream diversion, disturbance limits, equipment limitations, erosion control, and the operational time period when disturbances caused by construction can be most easily limited.
- Where necessary, use retaining walls with properly designed drainage to reduce excavation, contain bank material, and prevent stream encroachment.
- Design and construct stream crossings so that they will not divert streamflow out of the channel and down the road alignment if the culvert should fail or plug with debris (Figure 8.6) (Weaver et al. 1987).
- Field-check the designs before the plan drawings are complete to make certain that the design fits the terrain, that drainage needs have been met, that all critical slope conditions have been identified, and that appropriate solutions have been designed for all problem areas.

Construction

A challenge to the road builder is to construct the designed facility with the least possible disturbance of the right-of-way and without damage to or contamination of the adjacent landscape and streams. Poor construction practices can lead to severe erosion problems, toxic spills, and other water-quality problems. The following construction practices will help reduce adverse effects on salmonid habitats.

- Schedule construction during noncritical times for the local fish populations. Consult a fisheries biologist for this information. For example, avoid construction when eggs or alevins are in the gravels downstream, and do not restrict or block streamflow when adult fish are migrating upstream.
- Ensure that erosion-control measures are completed prior to rainy weather, even if construction is incomplete.

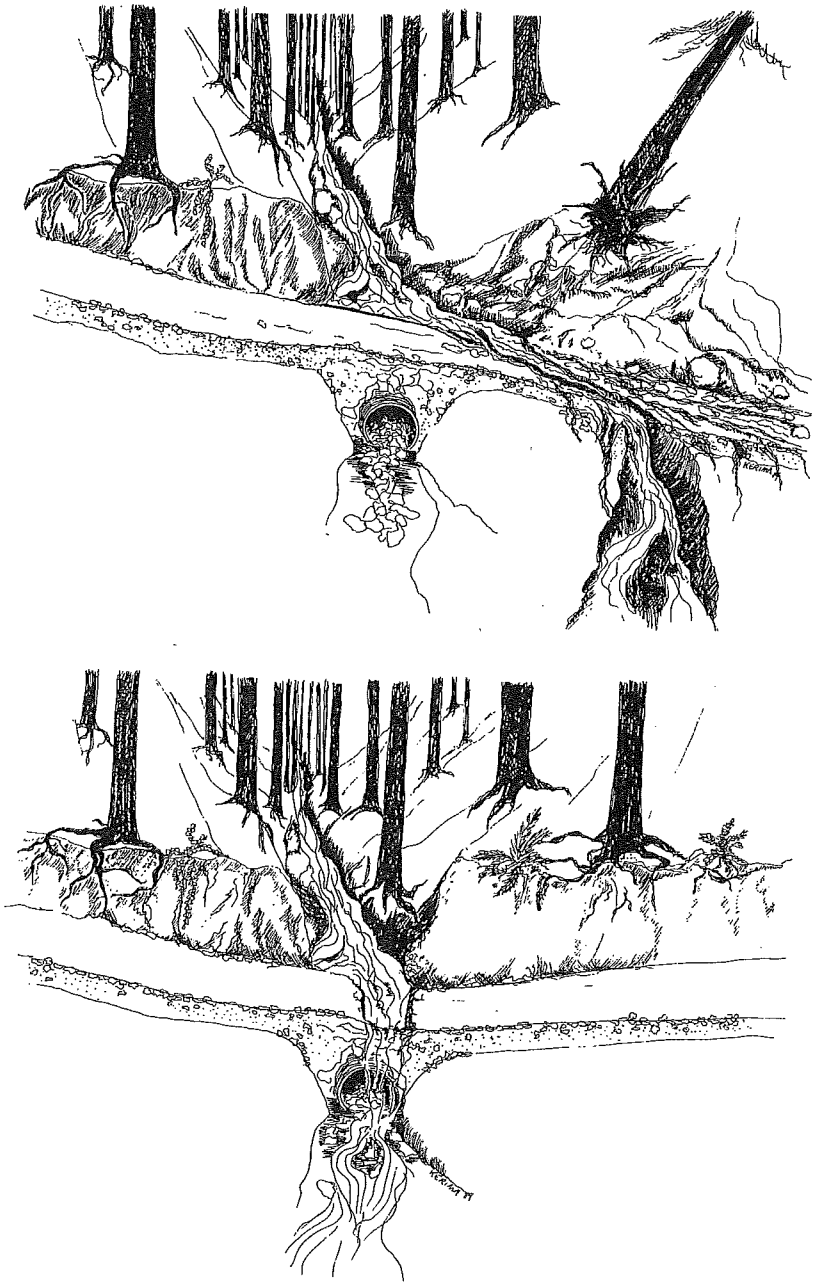


FIGURE 8.6.—Stream crossings can be considered dams that are designed to fail. The risk of failure is substantial for most crossings, so *how* they fail is of critical importance. In the upper sketch, the crossing has failed and the road grade has diverted the stream down the road, resulting in severe erosion and downstream sedimentation; such damage to aquatic habitats can persist for many years once begun. Stream diversions are easy to prevent, however, as illustrated by the lower sketch, in which the road grade was such that a crossing failure only caused the loss of some road fill.

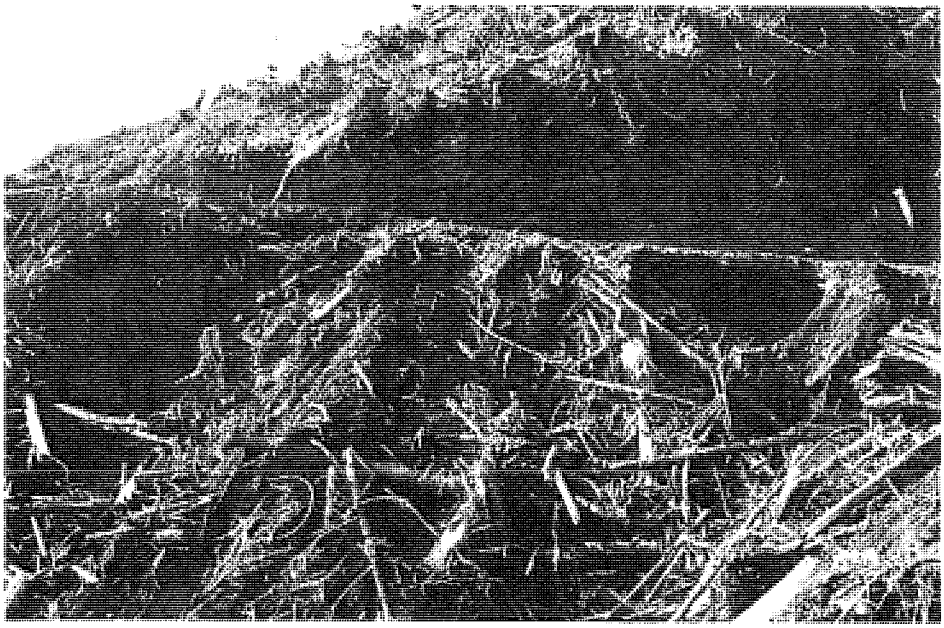


FIGURE 8.7.—Incorporation of woody debris into road fill and sidecast material inevitably leads to mass failure and surface erosion.

- Remove earth material and debris from streambanks so they cannot enter the stream later.
- Locate fuel storage areas well away from streams, and construct dikes to contain the largest possible spill. Leaks of motor oil and hydraulic fluids from heavy equipment should be monitored and controlled to prevent water contamination.
- Restrict gravel-removal operations to areas above the high-water level of the design flood. Coordinate gravel removal with a fisheries biologist. Sometimes beneficial changes can be made, but only when the fluvial system is understood and great care is taken.
- Locate and construct water withdrawal points to prevent streambank degradation and sedimentation.
- Locate road-building camps away from streams and manage wastes properly.
- Use spill-control planning and practice to keep construction toxicants out of streams.
- Do not incorporate woody or vegetative material into road fills (Figure 8.7).

Maintenance

Regular maintenance is required to keep roads in good condition and to identify and correct problems promptly. Preventive maintenance should be practiced on *all* roads, not just actively used roads. Maintenance requirements should be considered during planning and design. The higher initial costs of designing and constructing roads that weather well can be amortized by lower maintenance costs. The following practices will help reduce the adverse effects of road deterioration on salmonid habitats.

- Do not leave berms along the outboard edge of roads, unless an outboard berm was specifically designed to be a part of the road and low-energy drainage is provided for. The creation of outboard berms during road grading is a common mistake, and frequently turns low-impact roads into high-impact, chronic sediment producers.
- Grade and shape roads to conserve existing surface material. Road grading and shaping should maintain, not destroy, the design drainage of the road, unless modification is necessary to improve drainage problems that were not anticipated during the design phase.
- Inspect ditches and culverts frequently, and clean them out when necessary. Do not over-clean them, however, because excessive cleaning of ditches causes unnecessary sedimentation.
- When blading and shaping roads, do not sidecast excess material onto the fill. Periodic sidecasting can prevent fill stabilization and promote erosion.
- Do not use herbicides where they might contaminate streams, such as on areas near streams or in ditchlines that discharge into streams.
- Apply oil, other dust-abatement additives, and stabilization chemicals so they do not enter streams. Subsequent transport of these substances into water courses should be evaluated (Norris et al. 1991, this volume).
- Promptly remove debris that obstructs drainage systems.
- Close unsurfaced roads during the wet season, particularly those that can directly contribute sediment to streams.
- Close and reclaim unneeded roads. They should be put into shape to be stable and drain properly without maintenance. This usually requires earthwork for removing culverts or "dishing out" crossings that have high diversion potentials, and shaping the road for long-term stability (Figure 8.8; Eubanks 1980; Weaver et al. 1987). Where high-value fisheries are at risk from abandoned roads, more extensive obliteration and reclamation of roads should be considered.

Roads and Fish Migration

Forest and rangeland roads frequently cross streams. All crossings must be engineered to allow the efficient passage of water under the road. When the streams support fish, a means for their passage must be incorporated as well.

In western streams, anadromous salmonids migrate upstream and downstream during their life cycles, usually over long distances. Many resident salmonids and other fish also move extensively upstream and downstream to seek food, shelter, better water quality, and spawning areas.

Road crossings can be barriers to migration, usually because of outfall barriers, excessive water velocity, insufficient water depth in culverts, disorienting turbulent flow patterns, lack of resting pools below culverts, or a combination of these conditions (Figure 8.9).

The incorporation of fish-passage facilities at stream crossings should be based on assessments of the life-cycle requirements of fish species, of habitat quality, and of the accessibility of sites to fish. Natural barriers downstream or immediately upstream from the site may eliminate the need to provide fish-passage facilities. Usually, a knowledgeable fisheries biologist must be consulted to assess the habitat.

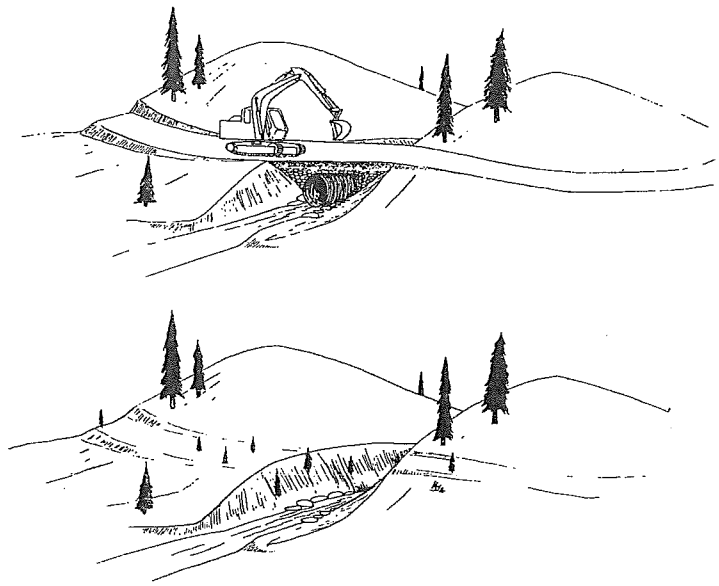


FIGURE 8.8.—When a road is abandoned, most future erosion can be prevented if roads are backfilled, stream crossings are removed, stream channels are reconstructed to stable configurations, and all bare surfaces are revegetated. Severe erosion is almost inevitable if stream crossings are left in place along abandoned roads.

Typical stream crossings involve bridges or culverts. Bridges are preferred because they usually cause less modification of stream channels than do culverts, and they are often the best way to assure fish passage. Building bridges on low-volume forest and rangeland roads is often uneconomical, however, and less-costly culverts are used. Culverts are by far the most common type of crossing device and the most likely to cause barriers to fish migration.

Low-water crossings are sometimes used where transportation requirements are seasonal and stream channel and slope configurations are suitable. Low-water crossings with concrete sills or grade-control structures can be barriers during low-flow conditions, but this can be easily mitigated in design. Low-water crossings are preferable to culverts for fish passage because high-flow migration is unimpaired and low-water migration is easy to provide for.

Consideration of fish passage during planning and design of road stream crossings can greatly reduce or eliminate the barrier effects that crossing structures can have. In many situations, structures must be designed with fish passage as the primary engineering requirement.

Where culverts are deemed necessary for stream crossings, the road designer should be aware of factors that affect the fish, and of the choices among crossing locations and structures. Close coordination between fishery biologists and engineers is critical, especially during planning stages.

The first consideration is whether or not the stream to be crossed is used by migrating fish. If not, the design problem is reduced to one of adequate discharge capacity. If the stream is used by fish that migrate at any time during their life cycles, the designer must know which species use the stream, when they migrate,

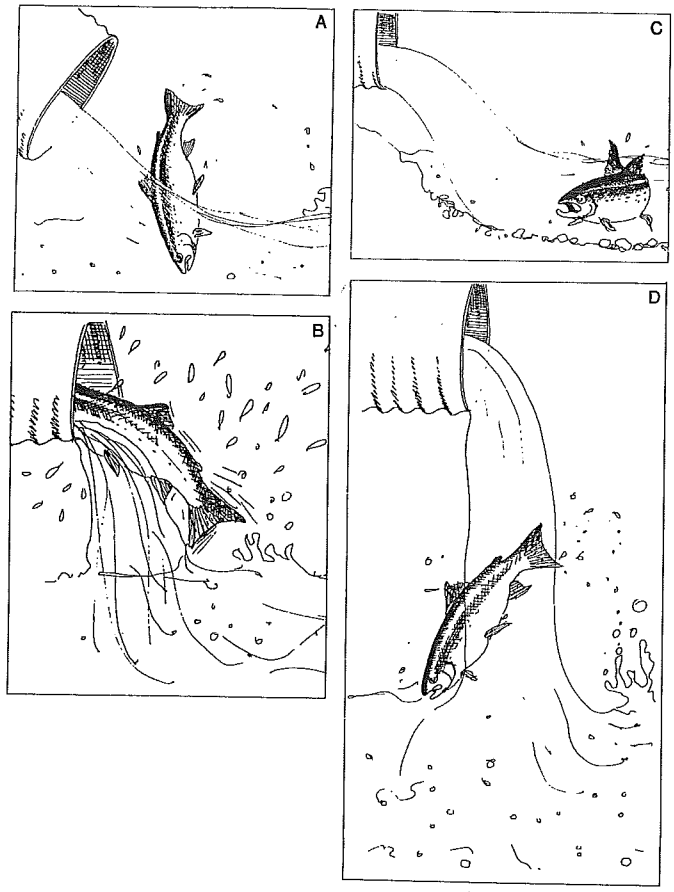
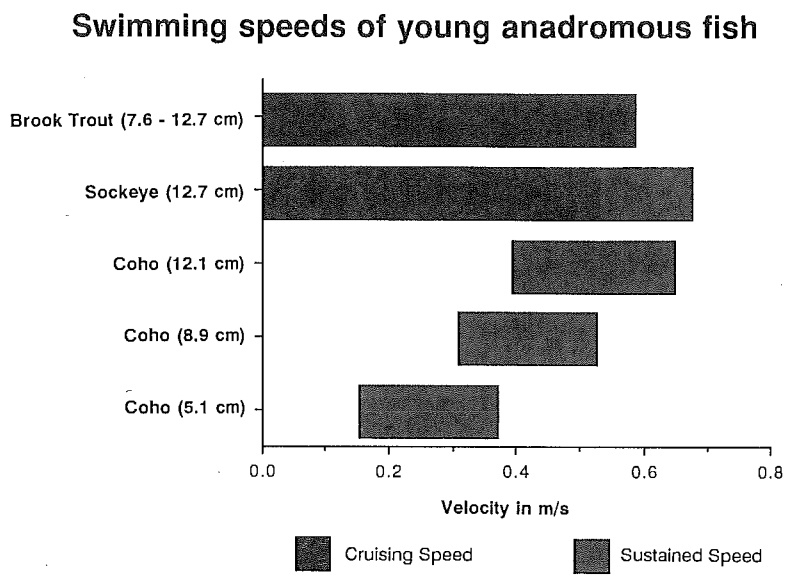


FIGURE 8.9.—Culvert conditions that block fish passage: (A) water velocity too great; (B) water depth in culvert too shallow; (C) no resting pool below culvert; (D) jump too high. (After Evans and Johnston 1980.)

and what their swimming capabilities are. Facilities for fish passage must be based on the swimming abilities of the least-capable migrating fish in the stream, not the champion swimmers.

For fish to overcome obstacles in their migration, the following conditions are necessary.

- A resting–jumping pool must be present immediately below the obstacle. This allows the fish to conserve energy and build up swimming speed to overcome the obstacle.
- Individual jumps must not be too high. For adult trout, a single vertical jump should be no higher than 0.3 m, and individual jumps in series should be 0.15 m high or less. A good working assumption is that adult salmon and steelhead can negotiate single jumps of 0.6–0.9 m or a series of 0.3-m-high jumps.
- Water depth through the culvert must be adequate for swimming. A minimum water depth of 0.15 m for trout and 0.3 m for salmon and steelhead is necessary.



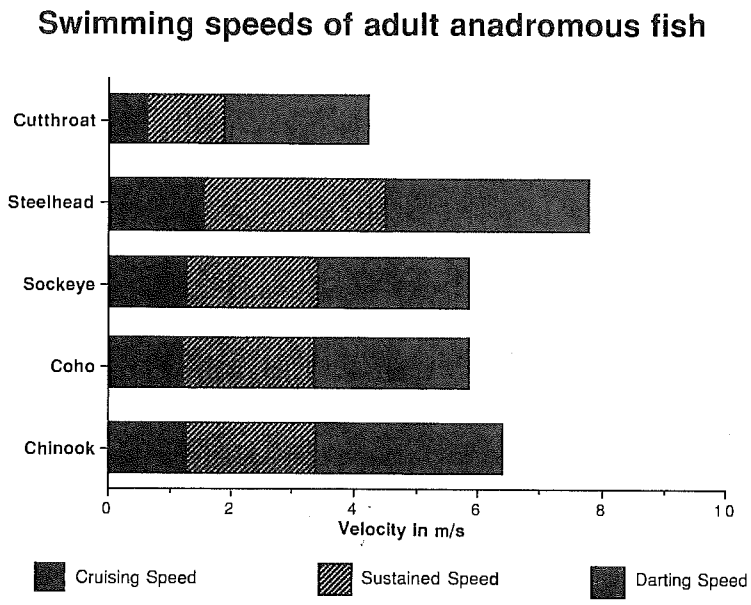


FIGURE 8.10.—Swimming speeds of young and adult anadromous trout and salmon.

- The water velocity in the culvert must not exceed the maximum sustained swimming ability of the migrating species for which passage is designed. Figure 8.10 shows the swimming speeds of adult and juvenile anadromous fish.
- Resting areas must be provided en route wherever the swimming distance through a difficult obstacle exceeds 15–30 m.

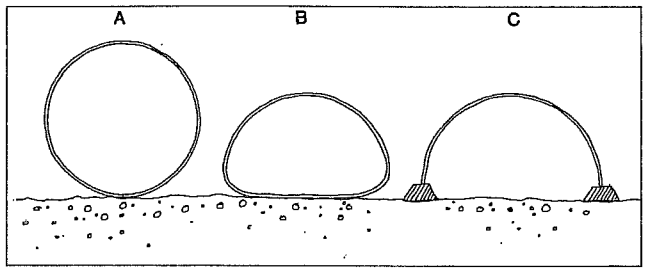


FIGURE 8.11.—Types of metal culverts used on western forest and rangeland roads: (A) corrugated-round; (B) corrugated pipe-arch; (C) structural plate-arch with concrete footings.

- A resting pool at the upstream end of a difficult obstacle is necessary so that exhausted fish are not swept downstream.

The choice of crossing location is very important in terms of both sedimentation effects and fish passage. For fish passage, preferred locations are those that do not cause large increases in velocity and have no abrupt changes in gradient or alignment of the channel. Reaches with uniform alignment, good bank stability, and uniform gentle gradients are the easiest to cross with provisions for fish passage.

Types of Culverts

Three types of metal culverts are commonly used for western forest and rangeland roads. They are classified by shape and are either standard corrugated-round, standard corrugated pipe-arch, or structural plate-arch (Figure 8.11). The first two may be prefabricated, as is usual for the smaller sizes (up to 1.5 m in diameter), or they may be of multiplate design. Structural plate-arch culverts are always of multiplate design because they are so large and are usually fabricated on site.

The structural plate-arch set in concrete footings (Figure 8.11C) is the most desirable culvert type for fish because the natural streambed is left mostly unchanged. Little narrowing of the flow occurs at either end of the culvert and there is no significant change in water velocity. Where concrete footings are not practical, split, wide-flanged, buried steel footings have been used successfully in their place. Many fisheries biologists believe that the structural plate-arch is the only acceptable culvert type where fish passage is required (Evans and Johnston 1980).

Pipe-arch culverts (Figure 8.11B) are less desirable than the structural plate-arch, but they can usually be installed to allow fish passage. Fabricated in smaller sizes, they can be used in smaller, lower fills where structural steel arches would not fit. Wherever pipe arches are used, the gradient must be kept below 1% to minimize water velocities. During periods of low flow, the water in culverts with this shape can be spread so thinly across the bottom that fish passage is impossible. Baffles may be needed to increase the flow depth through the pipe arch (baffle systems are discussed in detail later).

Although the standard corrugated-round culvert (Figure 8.11A) is the type most commonly used in western forest and rangeland roads, it is the least desirable for

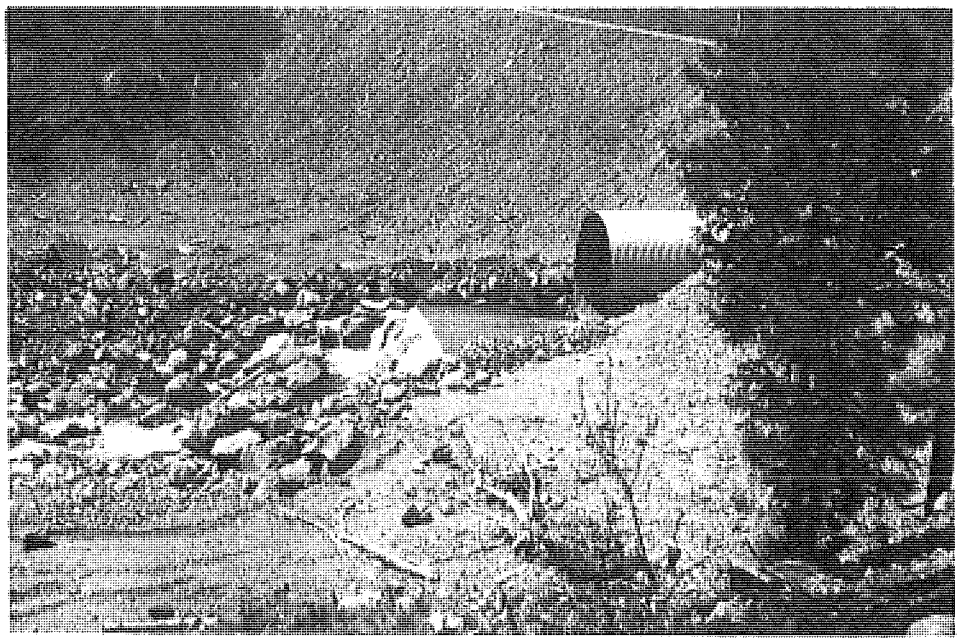


FIGURE 8.12.—Culverts installed above the grade of a stream can be a barrier to upstream fish migration. The culvert shown here is impassable to salmon and steelhead.

fish passage. The width constriction from stream channel to culvert is usually severe and the gradient of the pipe must be at or near 0% to keep water velocities within an acceptable range for fish passage. This type of culvert is also the most likely to be installed with its outfall above the tailwater elevation, producing an outfall barrier (Figure 8.12). Elevated outfalls must be avoided or mitigated.

Culvert crossings have been installed in thousands of streams with little or no thought to their effects on fish populations. A single poorly installed culvert can eliminate the fish population of an entire stream system. Poor culvert design and location can still be ranked among the most devastating problems for fish habitats on western forests and rangelands.

Following are some important considerations for culvert installation.

- Installation of round culverts should be avoided where fish passage is necessary. Install either open-arch culverts or bridges, especially if culverts longer than 30 m are required or where the stream gradient is steep (>2%).
- The two most important considerations for fish passage through culverts are maximum acceptable water velocity and minimum acceptable water depth for the migrating species.
- The diameter of culverts must be adequate to pass maximum flows and the expected debris. Washing out of culverts and their earth fills damages the road and is a source of sedimentation. Channel-bank stability upstream and downstream of culverts should be provided for. Road crossings alter the hydraulics of streams above and below the crossing for considerable distances, sometimes making banks more susceptible to erosion. Severe erosion can alter the configuration of

the stream and crossing, and can eliminate the design components that provide for fish passage.

- A single large culvert is better than several small ones because it is less likely to become plugged and it carries water at lower velocities.
- The entire length of round culverts should be placed slightly below normal stream grade, at a slope near 0%, to reduce fish passage problems.
- Any structure for fish passage must function through the range of flows during which fish migration occurs. Streams used by salmonids fluctuate widely and have occasional high peak flows. An acceptable practice in culvert design has been to not require conditions suitable for fish passage during the 5% of the year when the flows are the greatest (Evans and Johnston 1980). Fish do not normally migrate during peak flows, so little or no disruption of migration occurs. This practice often results in substantial savings in construction costs. The objective should be to ensure fish passage during all but the highest flows.
- Installations that would require baffling for fish passage should be avoided; a bridge, a low-water crossing, a pipe-arch culvert, a larger culvert, a reduced gradient, or another solution should be used instead. Baffles normally require additional maintenance and occasionally cause debris accumulations. Baffles are sometimes necessary when high water velocities are unavoidable or when fish-passage problems must be corrected at existing culverts.
- Where culverts are installed in stream sections with steep gradients, it is important to create or improve resting pools, cover, and bank protection along the stream above and below the culvert. Maintaining a stable stream bottom through the culvert-influenced area is essential.

Water Velocity in Culverts

The swimming abilities of salmonids increase with the size of the fish. Hence, the species and life history stages that must navigate the culvert determine the allowable maximum water velocity. The swimming abilities of salmonids are not adequately defined, but general information has recently become available. Figure 8.10 shows maximum, sustained, and average swimming velocities for common salmonids in the west.

Metsker (1970) pointed out that the culvert velocity a fish can overcome varies not only with fish size but also with the distance between resting pools above and below the culvert. The Oregon State Game Commission (1971) recommended maximum water velocities of 2.4 m/s for adult salmon and steelhead and 1.2 m/s for trout. The recommended velocities in Oregon, however, were for round culverts up to 30.5 m long. Water velocities in longer culverts should not exceed 1.8 m/s for adult salmon and steelhead or 0.9 m/s for trout.

To aid road designers in estimating the water velocities through culverts, both the Oregon State Game Commission (1971) and the U.S. Forest Service (Evans and Johnston 1980) have produced velocity curves based on Manning's equation for open-channel flow (Chow 1959). The Oregon State Game Commission curves are only for round culverts that range in diameter from 0.6 to 2.1 m. Gradients range from 0.25 to 5.0%. Figure 8.13 is an example of the Oregon velocity curves for a 1.8-m culvert. Fish passage through culverts normally occurs between a minimum depth of 7.6 cm and a maximum depth of two-thirds the pipe diameter; the Oregon curves cover only these depths.

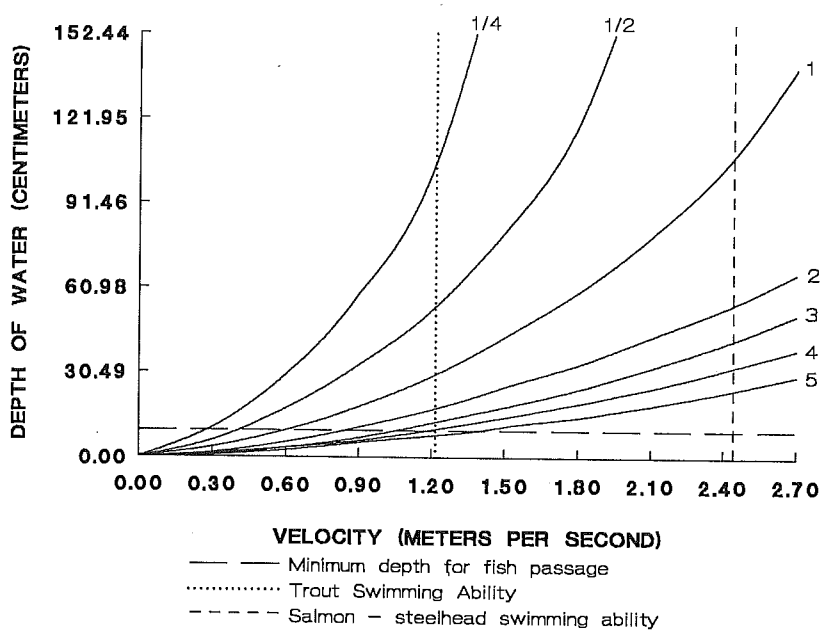


FIGURE 8.13.—Depth–velocity curves (Oregon State Game Commission 1971) for water flowing through a round, 1.8-m-diameter culvert. Curves represent gradients of 0.25, 0.5, . . ., 5%.

The U.S. Forest Service velocity curves are more detailed and comprehensive than the Oregon curves. Velocity curves are provided for round culverts (0.9–3.0 m), for concrete box culverts (0.7–3.0 m), and for corrugated metal pipe-arches (2.1 × 1.5 m to 5.1 × 3.1 m, span × rise). The U.S. Forest Service curves yield both velocity and depth of flow for any given discharge, culvert gradient, and diameter. Figure 8.14 illustrates the format of the U.S. Forest Service curves for a metal pipe-arch.

Salmonid spawning streams in the west are often mountain streams with steep gradients. Even culverts placed on the same grade as the original streambed may carry water at velocities greater than migrating fish can overcome. To control velocity in such situations, installation of baffles may be necessary. For new stream crossings, adequate water depth for fish passage should be integral to the structure design, and should not rely on baffles. Where existing crossings block migration because of inadequate water depth, baffles can be used to increase depth.

Culvert baffles are structures that impede water flow to produce pockets of lower-velocity flow in the culvert, where fish can rest momentarily, or that increase the overall depth of the flow. Impeding water flow in culverts is contrary to the primary engineering purpose of the culvert, which is to efficiently pass water under the road. The biologist and the engineer must work together to solve both problems simultaneously.

Many different baffle designs are available; little is known about their hydraulic principles, however, and additional applied research in this area is necessary. The best information on baffle design can be found in a Washington Department of Fisheries report by McKinley and Webb (1956). The principles in this report are

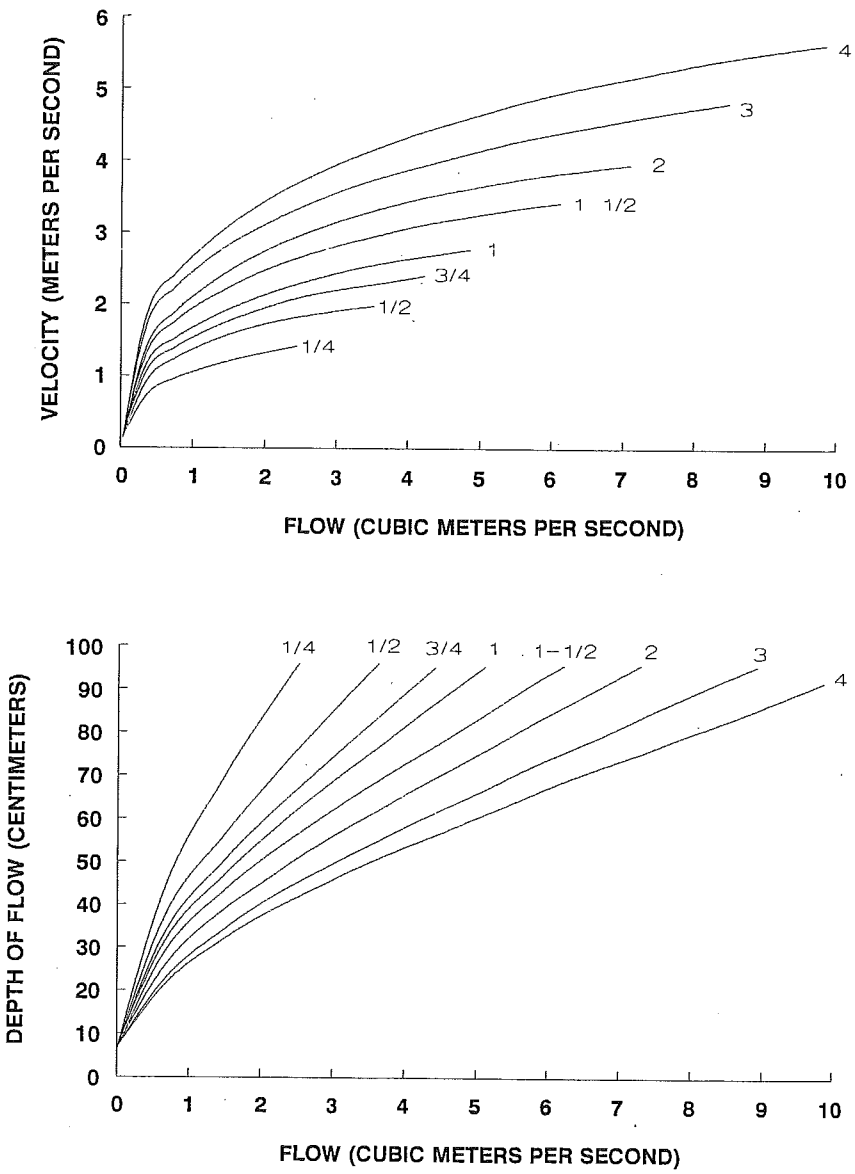


FIGURE 8.14.—Velocity–flow and depth–flow curves (U.S. Forest Service) for water flowing through a 2.1-m × 1.6-m metal pipe-arch. Curves represent gradients of 0.25, 0.5, . . ., 4%. (After Evans and Johnston 1980.)

sufficiently sound to be used as present guidelines, pending results of further research.

A few general principles apply to the use of baffles in culverts.

- Wherever possible, fish passage problems should be solved without baffles, preferably through use of bridges, low-water crossings, arch culverts, or round

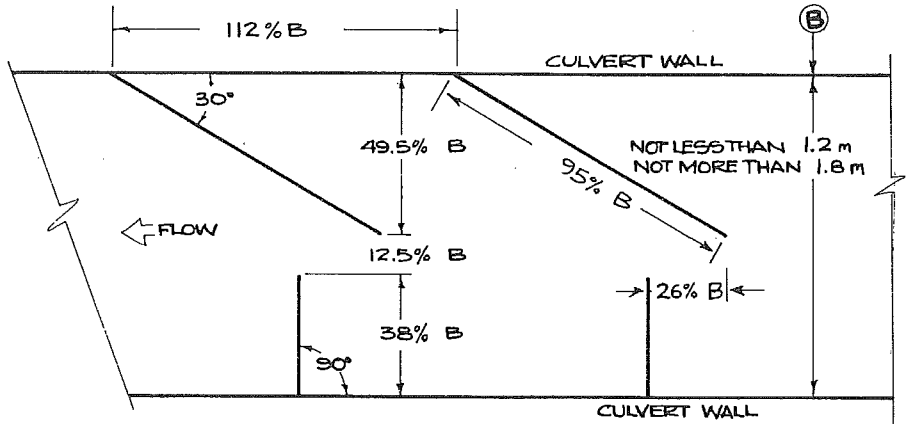


FIGURE 8.15.—Baffle design recommended by the U.S. Forest Service for culverts. Dimensions are scaled (as percentages) to culvert width (B), which can range from 1.2 to 1.8 m. (After Evans and Johnston 1980.)

culverts of adequate size and configuration. Culverts should be installed at or below streambed level such that water velocity is low.

- If higher velocities, extensive distance, or inadequate depth are unavoidable in a round- or box-culvert installation, baffles will be necessary. Baffles and the quieter waters they create allow fish to swim in short spurts through high-velocity water and to enter rest areas parallel to the higher-velocity flow.
- A large, single culvert provides better fish passage than do several smaller ones. Where multiple units are required in parallel, only one must be baffled to pass fish. Selection of the culvert for baffling is based on the route most likely to attract fish. At such installations, low flows should be diverted through the baffled culvert only.
- Large culverts, particularly the box type, can be divided so that baffles are placed in one side only.
- The baffle design illustrated in Figure 8.15 is recommended for use by the California Region of the U.S. Forest Service for solving excessive velocity problems (Evans and Johnston 1980). This design is adaptable to installations of various sizes; dimensions are given as percentages of the total width of the baffled section. Baffles should be at least 0.3 m high and 12.7–15.2 cm wide.
- Passage of water through the culvert will be somewhat impaired by baffle structures. Most culvert designs incorporate large safety factors, so the pipe diameter is considerably larger than is needed for the discharge conditions. Although the actual impairment of discharge capacity due to baffles is relatively small, a larger pipe may be necessary or a lesser safety factor must be tolerated.
- Construction materials for baffles may be wood, metal, or concrete, depending on the local situation. Concrete baffles can be precast and drilled or grouted into place. Metal baffles are usually bolted onto the culvert floor through metal plates, which give added strength.

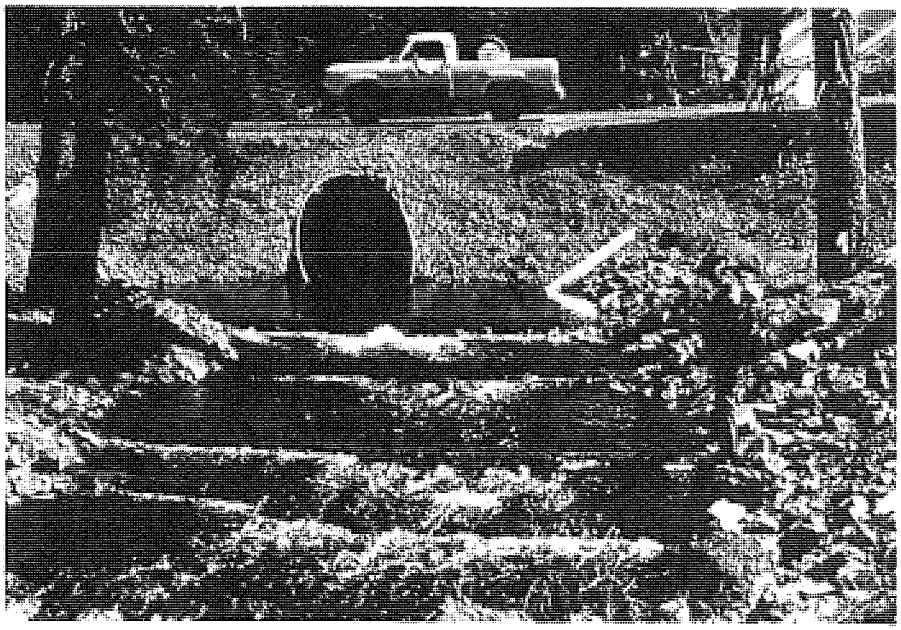


FIGURE 8.16.—Series of low-head dams that prevent a road culvert from being an outflow barrier to upstream fish migration.

- Most baffle designs work best when water flow just overtops the baffle elements. With greater water depth, baffle effectiveness is inversely proportional to the depth of water over them (Gebhards and Fisher 1972).
- Solving fish-passage problems during design and construction of new crossings is substantially less expensive than modifying existing culverts.

Culvert Outfall Barriers

Culverts can be insurmountable barriers to migrating fish when the outlet of the culvert is so far above the tailwater that fish cannot enter the pipe. This condition is termed an outfall barrier (Figures 8.9D, 8.12). When new culverts are to be installed on streams with migrating fish, every attempt should be made to avoid constructing outfall barriers. Putting a new culvert outlet below the tailwater elevation is sometimes not possible, and many existing culverts form outfall barriers.

One way to correct an outfall barrier is to provide for one or a series of low-head dams below the culvert outfall (Figure 8.16). These dams may be nothing more than hand-placed rock "reefs," wire-basket gabions filled with local rock, or concrete sills. These downstream dams raise the tailwater elevation and flood the culvert. Access by fish is not only enhanced, but water velocity in the culvert is decreased. The downstream dams should not create outfall barriers themselves and should therefore be limited to about 0.3 m in height.

For dams of greater heights, there must be a pass-through notch in the center. Because the back-flooding decreases velocity and hence discharge, a culvert of larger diameter may be necessary to handle peak flows when downstream dams

are used. Furthermore, it may be necessary to armor the downstream sides of the low-head dams to prevent scouring by the cataracts formed.

In some streams, the range of flows is so great that it is impossible not to have the culvert outlet above the tailwater at some time. Also, where severe fluctuations in flow require large culverts, fish passage may be impeded during low flows because of shallow flow over the broad culvert bottom. In such cases, stacked- or multiple-culvert installations can be used to provide fish passage. Placing the stacked culverts at different elevations assures adequate discharge capacity as well as fish passage over a wider range of flows.

Structures for Debris Control

The use of debris-control structures, such as trash or debris racks, is growing in western forest and rangelands due to the high cost of failed crossings. Trash racks, however, are detrimental to fish passage. The same freshets that often bring debris downstream are those in which many fish can move up to spawning areas. Although the protected culvert may not be a velocity or outfall barrier, a debris-laden trash rack can be impassable to fish. Debris-catching structures on streams used by migrating fish should therefore be avoided, and crossings should be designed to transmit debris downstream.

To compensate for the loss of culvert protection from a debris-catching structure, the culvert should be large enough to allow debris to pass through it. Passing debris through the culvert is a valid alternative to intercepting it above the inlet, and should not be overlooked. Of course, increasing the culvert diameter adds to its cost and sometimes may be impractical. On the other hand, when debris can be passed through the structure without clogging it, maintenance costs will be lower than when debris is intercepted and must be removed.

Summary

Forest and rangeland roads can have substantial adverse effects on salmonid habitats. These effects can be greatly reduced if the protection of fish habitat is integrated into the planning, design, construction, and maintenance of roads. The guidelines presented in this chapter for erosion control and for preventing or correcting fish migration barriers can be used by interdisciplinary teams of engineers, biologists, and earth scientists to plan and manage forest roads to maintain fish runs and habitats.

Influences of
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