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Assessment and Valuation of Forest Ecosystem Services: State of the Science Review

Seth Binder, Robert G. Haight, Stephen Polasky, Travis Warziniack, Miranda H. Mockrin, Robert L. Deal, and Greg Arthaud

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Abstract

This review focuses on the assessment and economic valuation of ecosystem services from forest ecosystems—that is, our ability to predict changes in the quantity and value of ecosystem services as a result of specific forest management decisions. It is aimed at forest economists and managers and intended to provide a useful reference to those interested in developing the practice of integrated forest modeling and valuation. We review examples of ecosystem services associated with several broad classes of potentially competing forest uses—production of timber, sequestration of carbon, regulation of the quality and quantity of water, provision of residential and recreational amenities, and protection of endangered species. For each example considered, we briefly describe what is known about ecological production functions and economic benefits functions. We also highlight the challenges and best practices in the creation and use of this knowledge. In the final section, we discuss the process, strengths, pitfalls, and limitations of utilizing integrated models for benefit-cost analysis of proposed forest management activities.

Quality Assurance

This publication conforms to the Northern Research Station's Quality Assurance Implementation Plan which requires technical and policy review for all scientific publications produced or funded by the Station. The process included a blind technical review by at least two reviewers, who were selected by the Assistant Director for Research and unknown to the author. This review policy promotes the Forest Service guiding principles of using the best scientific knowledge, striving for quality and excellence, maintaining high ethical and professional standards, and being responsible and accountable for what we do.

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INTRODUCTION

Ecosystems provide many goods and services that enable and enrich human life, from traditional natural resources, such as timber, fish, and edible plants, to the aesthetic qualities and characteristics of a place, to clean water and air (Daily 1997). Human ingenuity has enabled people to refine, re-allocate, and intensify the production of many goods and services by combining natural processes with human-created tools and labor. This has led to extraordinary advances in longevity and material well-being. However, it has also led to declines in some forms of natural capital and many non-marketed ecosystem services (Millennium Ecosystem Assessment 2005).

Scientists, policymakers, and land managers increasingly recognize the varied contributions of healthy, multi-functional ecosystems to human well-being and seek to develop the tools and knowledge necessary to manage these systems to best meet societal objectives. Within the last decade, several major academic and governmental initiatives related to ecosystem services have emerged. These include the publication of the Millenium Ecosystem Assessment (Millennium Ecosystem Assessment 2005), a National Research Council report on valuing ecosystem services (National Research Council 2004), a report from the EPA Science Advisory Board (U.S. Environmental Protection Agency 2009), a report on the economics of ecosystem and biodiversity (Kumar 2010), and the establishment of the new Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES). The Executive Office of the President (EOP) recently released a memorandum directing Federal agencies to factor the value of ecosystem services into Federal planning and decision-making. An additional EOP memorandum outlines research needs to assess ecosystem services in coastal green infrastructure. Within the EOP Office of Science and Technology Policy, an interagency Ecosystem Service Working Group was formed to facilitate cooperation among relevant agencies.

A consistent finding among these publications is that economic valuation of ecosystem services and comprehensive benefit-cost analyses are important tools to help decisionmakers manage ecosystems. This review focuses on the assessment and valuation of ecosystem services from forest ecosystems—that is, our ability to predict changes in the quantity and economic value of ecosystem services as a result of specific forest management decisions. It is aimed at forest economists and managers of public and private forest land, with the intention of providing a useful reference to those interested in developing the practice of integrated forest modeling and valuation. To this end, we review examples of ecosystem services associated with several broad classes of potentially competing forest uses—production of timber, sequestration of carbon, regulation of the quality and quantity of water, provision of residential and recreational amenities, and protection of endangered species. For each ecosystem service, we review a selection of ecological and economic research related to ecological production functions and economic benefits functions, and highlight challenges and best practices in the creation and use of this knowledge. In the final section, we discuss the strengths, pitfalls, and limitations of utilizing integrated models for benefit-cost analysis of proposed forest management activities. We supplement this discussion with a more quantitative treatment for relatively simple decision problems of optimal land use (i.e., preserve, harvest, or develop a given forest area) and optimal rotation age.

The academic literature on ecosystem services is vast and we limit our scope to services of non-urban forests, public or private, that are amenable to economic valuation. We do not cover cultural ecosystem services such as cultural heritage or spiritual significance that are difficult to quantify and whose value is often thought to be antithetical to consideration in monetary terms. These cultural services have value in their own right, and they have played an important role in motivating public support for the protection of ecosystems. Daniel et al. (2012) review research on relationships between ecological structures/functions and cultural values including landscape aesthetics, cultural heritage, outdoor recreation, and spiritual significance. We also do not cover ecosystem services provided by urban forests, wetlands, lakes, and undeveloped areas (e.g., McPhearson et al. 2014) where the beneficiaries are primarily urban residents.

Methods have been developed to estimate the economic value of urban forests based on their effects on air quality (Nowak et al. 2014), water

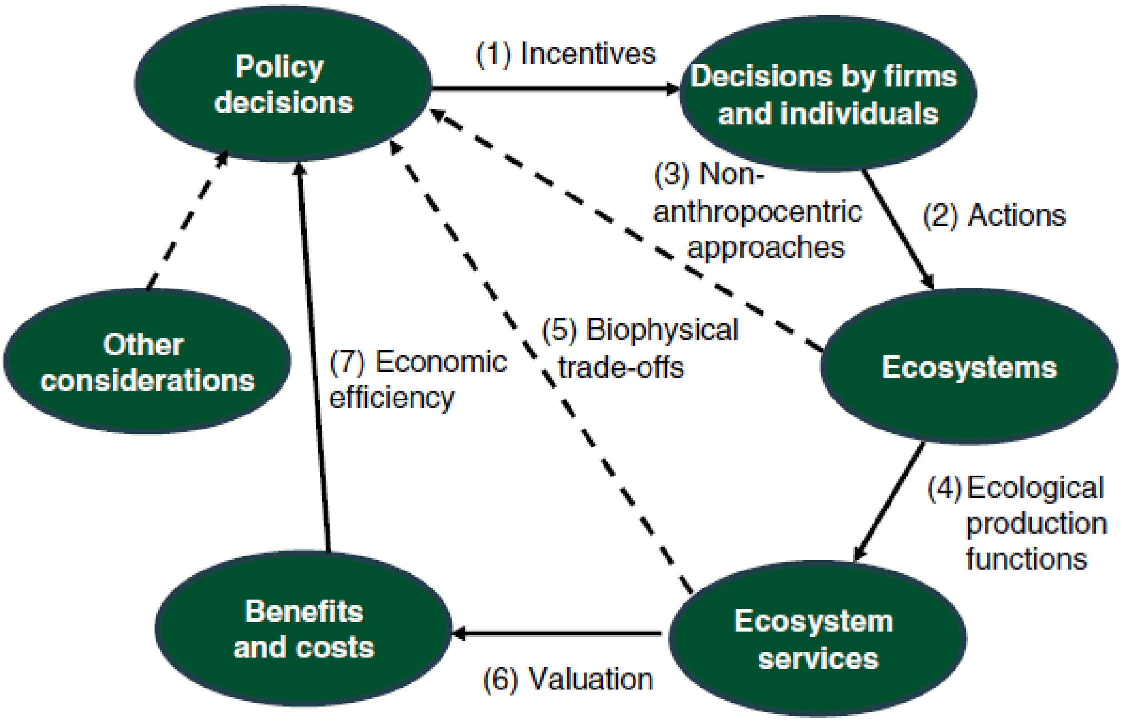

Figure 1.—Conceptual diagram of the links among changes in ecosystem management decisions, the production of ecosystem services, and resulting benefits and costs to society. From Polasky and Segerson (2009).

quantity and quality (Hobbie et al. 2014, Keeler et al. 2012, McPherson et al. 2005), residential energy consumption (Akbari 2002), and aesthetic amenities (Sander et al. 2010). Non-urban forests may affect aesthetic amenities of nearby residents and so we do discuss hedonic property value studies (e.g., Sander et al. 2010) as a way to estimate the value of forests for the provision of aesthetic amenities.

Framework for Identifying and Valuing Ecosystem Services

Science-based decision support tools have the potential to provide information to Federal agencies, States, and private landholders about the benefits and tradeoffs among various forest uses. These tools require understanding—and quantitatively modeling of—the chain of relationships that link changes in forest policy or management to changes in human well-being (Fig. 1). Government agencies control some important management decisions directly (e.g., land use within National Forests). In this case, the agency would start the analysis by considering the effect of its management decisions on ecosystems (Link 2). In other cases, government agencies set policies that provide incentives to private landowners (e.g., the Conservation Reserve Program). Here the agency would need to predict how the policy would affect private landowner decisions (Link 1) and then what impact these decisions have on ecosystems (Link 2). From here, analysis must consider how changes in ecosystem structure and function translate into changes in ecosystem services (Link 4). Ecological production functions (NRC 2005, Polasky and Segerson 2009, Swallow 1990) capture these relationships. An example of an ecological production function is an empirically estimated equation that predicts the abundance of a wildlife species that people care about (the ecosystem service) as a function of the age, species composition, slope, and elevation of the forest stand in which the wildlife population lives. Ecological production functions can be used to estimate production potential and identify biophysical tradeoffs between alternative ecosystem services. Some analysts may prefer to base policy decisions on consideration of impacts on the flow of various ecosystem services, recognizing the potential for tradeoffs in those flows (Link 5) rather than assessing them in terms of the public's preferences.

In many cases, inefficiencies in existing management imply the possibility of identifying alternative "win-win" management scenarios that increase all ecosystem services. In other cases, however, decisionmakers require additional information to help navigate tradeoffs among ecosystem services. Here, understanding the relative values of increases or decreases in different services can help managers or policymakers select the option that brings the greatest benefits to society. This requires the estimation and use of economic benefits functions, which quantify in monetary terms the relationships between changes in the provision of ecosystem services and changes in human well-being (Link 6). The benefits function for our example above would be an equation that translates changes in wildlife abundance into a dollar value, based perhaps on an economic valuation study of recreational demand for wildlife abundance.

Economic valuation of an ecosystem's goods and services represents an attempt to estimate changes in people's economic well-being—as measured by their own preferences—due to incremental (*marginal*) changes in the ecosystem's components. When ecosystem goods are traded in markets (e.g., timber), the market price (e.g., U.S. dollars/cubic meter) is a measure of the benefit people get from a unit of the good. Since most ecosystem services are not traded in markets, and therefore do not have observable prices, economists estimate the value of changes in ecosystem services by leveraging the information conveyed by individuals' observable decisions. Information obtained from observable decisions in hypothetical markets created by the analyst is known as stated-preference data. In contrast, revealed preference data is obtained from observable decisions in actual markets for a *weak complement* to the non-market ecosystem service. In both cases, the choices and tradeoffs people make reflect their *willingness to pay* (WTP) to access or obtain ecosystem services or their *willingness to accept* (WTA) some amount in exchange for a reduction in services. Their WTP or WTA is a monetary measure of the benefits they get from a change in the service. Economic benefits functions estimate WTP or WTA based on the nature and extent of changes in ecosystem components, the availability of substitute or complementary goods or services, and beneficiaries' income and other demographic characteristics.

It is important to point out that methods for economic valuation of ecosystem services differ from survey methods, such as public participation geographic information systems (PPGIS), for assessing public preferences for ecosystem services. Analysts using PPGIS methods typically ask selected individuals to locate ecosystem services (e.g., aesthetic, recreation, economic, and ecological services) that they value within a given landscape (e.g., Brown and Reed 2009). The maps of landscape values are then analyzed to determine their relative importance as an estimate of people's preferences (e.g., Brown and Donovan 2014). Economic valuation methods go further than PPGIS methods by estimating how much people are willing to pay for an incremental change in the level of any given service, based on their stated or revealed preferences in hypothetical or actual markets.

Together, ecological production functions and economic benefits functions form an integrated assessment model that links management decisions to their full costs and benefits (Daily et al. 2009, Nelson et al. 2009). This scientific approach to analysis, when transparent, will provide policymakers and managers with valuable information to support their decisions. However, working across disciplines and integrating models that were not necessarily designed to be compatible is no easy task. One of the biggest stumbling blocks can be an understanding and operationalization of the ecosystem service concept.

Ecosystem services have been variously defined as the benefits people obtain from nature (e.g., recreational fishing) (Millennium Ecosystem Assessment 2005), the end products of an ecosystem that are directly used or consumed by people (e.g., the fish anglers seek for their recreational benefit) (Boyd and Banzhaf 2007), or the processes by which ecosystems produce resources (e.g., nutrient cycling that enhances fish populations) (Ecological Society of America 2012). While all of these definitions make useful connections between ecology and human well-being, we use the definition advanced by Boyd and Banzhaf (2007) because it facilitates measurement, integrated modeling, and valuation. Ecosystem services are components of nature, directly enjoyed, consumed, or used by people for their well-being. As noted by Boyd and Banzhaf (2007), this definition has several important features. First, ecosystem services are end products of nature that are directly consumed, enjoyed, or used for human benefit. We distinguish between a benefit and an ecosystem service because benefits are often produced with capital and labor in addition to biophysical inputs. For example, flood control is a benefit that depends on the construction of levees, canals, and other engineering features in addition to the peak flow of water downstream from the forest. We think it is important to identify the end product of the ecosystem (peak flow of water) so that it can be measured and valued in the context of the benefit to which it contributes. Second, the distinction between end products and intermediate products or processes is important in welfare accounting. Because the value of intermediate goods or processes is embodied in the value of final goods, only the value of the final good need be counted. (However, it is still possible and may sometimes be desirable to estimate the value of a change in an intermediate ecosystem service.) Third, ecosystem services are components of ecosystems, which means they are ecological things or characteristics, not the functions or processes that support or produce the end products. Fourth, ecosystem services are measured by their quantities or physical units, which subsequently can be paired with estimates of the monetary value of changes in these quantities. We emphasize that other definitions and lists of ecosystem services, such as Daily's (1997), were constructed to illustrate the connection between ecology and human well-being, not to facilitate measurement, integrated modeling, and valuation of services.

For each forest use, Table 1 provides a subset of forest benefits and associated ecosystem services that we illustrate in the following sections. Because a forest ecosystem consists of all the biological organisms in a woodland functioning together with all of the non-living physical components of the woodland, goods and services may be biotic (e.g., timber, trout) or abiotic (e.g., stream water, carbon) components of the ecosystem. The examples given in the table and covered in this review are not exhaustive of either the benefits arising from different forest uses or the ecological services associated with a particular benefit. Rather, they are examples for which integrated biophysical and economic modeling techniques have been used for service valuation. Each row of the template represents a unique forest benefit and beneficiary. For each benefit, the template identifies: 1) the ecosystem service (i.e., the ecological end product) that can be measured or modeled in biophysical assessments and that directly affects human well-being; 2) the ecological production function that models how changes in ecosystem structure and function translate into changes in ecosystem services; and 3) the economic benefit function or method to estimate the monetary value of changes in the ecosystem service that result from changes in forest management.

TIMBER PRODUCTION

Forests provide timber for the wood products industry (Table 1), and timber production has long been an objective of public and private forest management. We begin with a description of timber management systems and ecological production functions that project timber yields. Then, we describe economic benefit functions, including the market processes that determine timber prices on public and private land and the net present value criterion to evaluate timber management systems. While we focus on timber, we emphasize that forest management is inherently a problem of joint production: decisions intended to produce timber affect the production of other forest ecosystem services. Here, we briefly note how ecological production functions and economic benefit functions for timber can be extended to assess changes in other ecosystem services and their values. In a later section, we give a detailed account of integrated modeling and benefit-cost analysis for the joint production of multiple ecosystem services.

Ecological Production Functions for Timber

Forests that are managed for timber production are subdivided into stands, which are administrative units bounded by physical and geographic features such as roads and property boundaries and by discontinuities in forest vegetation. Each stand is managed using a particular management system, typically even-age or uneven-age management (Smith 1962). A stand management system is a planned program of silvicultural treatments during the life of the stand. Treatments include regeneration cuttings that are used to establish a new stand by means of natural or artificial regeneration. Intermediate cuttings such as thinnings or selection harvests are used to control the density and growth of existing trees. Stand management systems are classified based on their regeneration method. The most well-known management system uses the clearcutting

Table 1.—Template for identifying and valuing ecosystem services from forests. For each forest use, we identify one or more benefits, beneficiaries, and ecosystem services. Ecosystem services are the valued end products of the forest ecosystem that contribute to the production of benefits and may be affected by forest management activities. The ecological production functions and economic benefits functions form an integrated assessment model that links management decisions to their full costs and benefits.

Forest use	Benefit	Beneficiary	Ecosystem service (valued end product of the forest ecosystem)	Ecological production function	Economic benefits function
Timber production	Timber for wood products	Industrial wood producers	Merchantable timber (stumpage)	Stand simulation models (e.g., Forest Vegetation Simulator) Forest landscape simulation models	Market price of timber (stumpage price)
Carbon storage	Climate regulation	Everyone	Sequestered carbon	Carbon budget simulation models (e.g., FORCARB2) Stand simulation models	Social cost of carbon
Water regulation	Irrigation water	Farmers	Flow of water downstream	Paired watershed studies Forest hydrological models (e.g., Distributed Hydrology Soil Vegetation Model)	Market price for water Shadow price of water Hedonic price model for farmland
	Flood control	Homeowners	Peak flow downstream	Paired watershed studies Forest hydrological models	Avoided damage
	Coldwater fishing	Anglers	Trout abundance	Energy transfer models for stream temperature coupled with trout population models	Hedonic travel cost model Discrete choice RUM
	Clean drinking water	Local water consumers	Amount of sediment in water	Erosion prediction models (e.g., Water Erosion Prediction Project model)	Household demand model for water, avoided costs of treatment, replacement cost
	Safe navigation	Commercial navigators	Amount of sediment in water	Erosion prediction models	Avoided costs of dredging
	Clean drinking water	Local water consumers	Amounts nitrate and phosphorus in water	Nutrient and chemical movement models (e.g., Soil and Water Assessment Tool)	Household demand model for water, avoided costs of treatment, replacement cost
Aesthetic amentity	Aesthetic amenity	Homeowners near forest	Forest cover in viewshed	Stand simulation models Forest landscape simulation models	Hedonic property price model
	Aesthetic amenity	Leisure travelers and commuters	Forest cover in viewshed	Stand simulation models Forest landscape simulation models	Recreational demand model
Recreation	Recreational hiking, camping, and biking	Hikers, campers, bikers	Old growth area, forest density, burned area	Stand simulation models Forest landscape simulation models	Recreational demand model Discrete choice RUM
Wildlife	Recreational hunting	Hunters	Game abundance	Demographic models of wildlife abundance and viability (e.g., RAMAS)	Recreational demand model Discrete choice RUM Hedonic pricing of licenses
	Protecting rare and endangered species	Everyone	Species survival probability	Demographic models of wildlife abundance and viability	Contingent valuation method

reproduction method and produces even-aged stands. In this case, even-aged management is a cycle of events (rotation period) that includes clearcutting a mature stand, planting a single cohort of trees, periodically thinning the new crop, and clearcutting the crop at a specified rotation age. Uneven-age management uses a selection harvesting method that involves the periodic harvest of trees in specified size classes. Selection harvests are conducted to control the spacing and growth of the remaining trees and to enhance natural regeneration from seeds produced by the remaining mature trees. Since selection harvest and regeneration take place simultaneously, uneven-aged stands include a mixture of trees in a wide range of size and age classes.

For both even-age and uneven-age management systems, timber yields are projected using stand simulation models (i.e., ecological production functions), which have been developed since the 1960s to provide forest managers with accurate growth and yield information for planning (see Munro 1974 for review). Stand simulation models include stage-class models or individual-tree simulators, both of which use discrete time difference equations to project tree growth, mortality, and regeneration. The models differ in their characterization of stand structure and their tractability for optimization.

In stage-class models, trees are classified into species and stem-diameter classes. Equations for tree growth, mortality, and regeneration project the changes in the number of trees in each class over time as a function of stand density (see Getz and Haight 1989 for review). Stage-class models are constructed by first estimating the parameters of each of the component equations separately and then inserting the equations into a matrix framework to project stand growth. This matrix framework facilitates the application of linear and non-linear programming algorithms for optimization of stand treatments and harvests (Getz and Haight 1989, Haight 1987).

Individual-tree simulators describe the stand with a list of tree records where each record contains the current tree dimensions (e.g., stem diameter, height, and crown length) and an expansion factor representing the number of trees of its kind in the stand (see Liu and Ashton 1995 for review). Equations for diameter growth, height growth, and crown development predict changes in the tree attributes as a function of stand density. An equation for tree mortality adjusts the expansion factors, and an equation for regeneration creates new records. Like stage-class model construction, equations for changes in tree dimensions are estimated separately and then inserted in a simulation shell for projecting stand growth. Major modeling efforts in different regions of the United States have created individual-tree simulators, and many of those have been incorporated into the Forest Vegetation Simulator, which is a family of forest growth simulation models supported by the USDA Forest Service (see Crookston and Dixon 2005 for review).

An important property of individual-tree simulators is the detail in which a stand and its growth processes are described. Since individual-tree simulators explicitly project the dimensions of hundreds or even thousands of trees in a stand, they provide a detailed picture of stand structure and composition over time. This detail facilitates the simulation of many forest-related processes including wildlife population dynamics, insect and disease dynamics, wildfire intensity and spread, and carbon sequestration (Crookston and Dixon 2005). Recently, progress has been made connecting individual-tree simulators with optimization algorithms to analyze stand treatment and harvest decisions in relation to the ecosystem services produced (e.g., Hyytiäinen et al. 2004, Rämö and Tahvonen 2014).

Forest landscape simulation models (FLSMs) have been developed to project the effects of forest management options on the spatial configuration, composition, and heterogeneity of vegetation, including community types, tree species age classes, and aboveground biomass (see Scheller and Mladenoff 2007 for review). In contrast to stand simulation models, FLSMs are applied to extensive areas of forest. They subdivide the landscape into cells (or polygons) and project attributes of the vegetation in each cell. Projections are based on vegetation attributes within and across cells and disturbance processes that are endogenous (e.g., wildfire) or exogenous (e.g., land-use change or timber harvesting) to the model. The extent and detail of vegetation attributes that are projected with FLSMs allow for the projection and analysis of many forest benefits including carbon storage, recreation, wildlife abundance, and water yield.

Economic Benefit Functions for Timber

Wood products (e.g., lumber, paper, structural panels, and fuel) have well-defined markets and significant economic value, contributing $280 billion in 2007 to the U.S. economy (United States Census Bureau 2009). The economic benefit of timber for wood products is measured by stumpage value—the amount per unit area that a commercial wood cutter is willing to pay for an area of standing trees (Helms 1998). It is the product of the stumpage price and the amount of timber offered for sale. To understand how stumpage price is determined, it is useful to first describe the ownership of forest lands in the United States. Forest lands are primarily owned by private entities, including nonindustrial owners without processing facilities, forest industry owners with processing facilities, and various types of forest land investment organizations. Nonindustrial and forest land investors own over two-thirds of U.S. forest land; combined with the forest industry, the U.S. private sector owns roughly 75 percent of U.S. forest lands and produces over 90 percent of the industrial wood harvest. The remaining 25 percent of forest lands is owned by government agencies—Federal, State, and local—which produce less than 10 percent of the industrial wood (Sedjo 2006).

Stumpage prices for private timber are market-determined (Sedjo 2006). They depend on the industry's aggregate demand for trees (which is based in part on market-determined prices for wood products, which in turn depend on domestic and international trade) and the aggregate supply of industrial wood from private landowners (which is based on the current and expected future stumpage prices and age of the forest). Stumpage prices for a given ownership also depend on wood quality—itself dependent on timber age, species, and condition—and cost considerations associated with timber accessibility, mill distance, terrain, and other factors reflecting extraction, transport, and processing costs. Stumpage prices for most sales of public timber are determined through a competitive auction process, which reflects the market information on stumpage prices of private timber (Sedjo 2006).

The decision about which management system to use in a particular stand and the attributes of the management system is often made using a net present value criterion (e.g., Haight 1987). For even-age management, the problem is to determine: the timing and intensity of silvicultural treatments for the current stand; the time when the stand is clearcut and replaced with a plantation, if it is currently under natural forest cover; and the timing and intensity of silvicultural treatments and clearcut age for the plantation. For uneven-age management, the problem involves determining the sequence of selection harvests that converts the current stand to the desired uneven-age steady state. The net present value of each management system is calculated based on the discounted value of timber yields and costs of management (e.g., planting, weeding, and pruning) over all future harvests (Faustmann 1849, Samuelson 1976).

We emphasize that forest management systems intended to produce timber affect the production of other forest ecosystem services. For example, stands managed with selection harvests maintain plant understory species richness and abundance, which are important for wildlife habitat (Deal 2001), while maintaining stand growth and providing industrial wood (Deal and Tappeiner 2002). The stand structures that develop after selection harvests create structurally complex, multi-layered forest canopies that were much more similar to old-growth forests than the uniform young-growth stands that develop after clearcutting (Deal et al. 2010). The value of non-timber ecosystem services can be incorporated into the calculation of net present value of forest management systems. Integrated assessment models that account for the production and value of multiple ecosystem services have been developed and used in forest management since the 1980s (e.g., Bowes and Krutilla 1989).

CARBON STORAGE

Forest ecosystems provide a climate regulation benefit (Table 1) because forests store carbon in the soil and in biomass that might otherwise be released into the atmosphere in the form of CO_2. Carbon storage is a valuable ecosystem service because reducing atmospheric carbon reduces the intensity of future climate change. Reducing the likelihood of damage associated with more intense climate change will benefit everyone. In this section, we review biophysical modeling approaches to estimating forest carbon stocks and fluxes, and we discuss economic approaches to estimating the value of storing additional carbon through enhanced sequestration or release prevention.

Ecological Production Functions for Carbon Storage

When discussing regional, national, or global carbon stocks and fluxes, most papers report carbon in teragrams (Mt, megatons, million metric tonnes) or megagrams (Mg, one metric tonne). At the stand or forest levels, megagrams (Mg) per hectare of carbon are used. These measures are carbon mass, not CO_2 mass, because carbon is a standard currency and can easily be converted to any other unit. Many reports give stocks and fluxes of the mass of CO_2. To convert C mass to CO_2 mass, multiply by 3.67 to account for the mass of the O_2.

The evaluation of forestry opportunities for carbon storage received intense analysis beginning in the 1990s. Birdsey (1992) was the first to provide comprehensive estimates of carbon storage and accumulation in U.S. forests. Carbon was estimated separately for trees, soil, forest floor, and understory vegetation for major forest types and plantation species in eight geographic regions. For trees, carbon estimates were based on merchantable volumes in existing forest inventory data, which were converted to carbon based on conversion factors for total tree volume, total biomass, and carbon as percentage of dry mass. Estimates of carbon stored in soil, forest floor, and understory estimation were obtained from the ecological literature. Birdsey (1992) also estimated changes in carbon storage over time based on changes in live trees. The conversion factors and carbon estimates of Birdsey (1992) were incorporated into carbon budget models to examine the effects of forest management practices on carbon storage in U.S. timberlands (e.g., Adams et al. 1999, Alig et al. 1997).

In the early 2000s, work began to replace Birdsey's (1992) conversion factors for merchantable tree volume with estimators for forest biomass and carbon. Jenkins et al. (2003) developed a consistent set of aboveground tree biomass equations as a function of tree diameter for over 100 species in the United States. These individual-tree equations were then applied to inventory plot data to estimate equations for biomass density (Mg/ha) of live and standing dead trees as a function of merchantable volume (m^3/ha) for broad forest types and regions of the coterminous United States. (Smith et al. 2003a). Tree biomass is about 50 percent carbon, so carbon estimates can be derived from estimates of biomass by multiplying by 0.5. The equations for forest biomass were incorporated in the U.S. Forest Service carbon budget simulation model (FORCARB2), which provides inventory-based estimates of U.S. forest carbon stocks (Smith et al. 2004). The model includes separate, non-overlapping components of total forest ecosystem carbon pools, including live trees, standing dead trees, understory vegetation, down dead wood, forest floor, and organic carbon in soil. The model is applied to plot-level inventory data, where merchantable tree volume (m^3/ha) and age are used to estimate tree and forest floor carbon, respectively. FORCARB2 was used in U.S. Greenhouse Gas Inventory (United States Department of Agriculture 2008) and Forest Service studies (e.g., Heath et al. 2011), which provided managed forest carbon estimates for 253 million ha of U.S. forest land.

In addition to tools like FORCARB2 that estimate carbon storage at the county, state, and national levels, simulation models of forest growth have been developed to predict changes in carbon storage at the national, regional, and stand levels. Further, they are used to estimate net carbon storage under alternative forest policy or management scenarios relative to a baseline scenario to evaluate the carbon impacts of changes in policy (Richards and Stokes 2004). For example, Wear and Coulston (2015) develop a forest carbon projection model based on observations from over 350,000 permanent monitoring plots across the United States that are part of the USDA Forest Service's Forest Inventory and Analysis Program. Their model, which is developed for regional or national projections, includes estimates of carbon densities by forest age class, forest sequestration rates by age class, areal extent of forest by age class, and age transition probabilities aggregated at the state or regional level, including disturbance and management effects. Forest dynamics are applied as transition probabilities to current estimates of areal extent by age and define changes in forest structure. Carbon sequestration is then estimated using observed carbon stock densities and sequestration rates applied to the new forest structure. Wear and Coulston (2015) project the effects of national-level policies intended to boost forest carbon sequestration, including reducing deforestation and development, increasing afforestation and reforestation, and reducing wildfire.

At the stand level, carbon projections can be made with the Forest Vegetation Simulator (FVS), an individual-tree, distance-independent, growth and yield model that predicts changes in tree diameter, height, crown ratio, and crown width, as well as mortality, over time (Crookston and Dixon 2005). FVS has been calibrated for geographic areas of the United States and can simulate a wide range of silvicultural treatments for most major forest tree species, forest types, and stand conditions. The model includes equations to predict the biomass of live trees, dead trees, down dead wood, understory, and forest floor. Biomass, expressed as dry weight, is assumed to be 50 percent carbon. FVS is used to estimate the potential carbon consequences of forest management actions, including planting densities, thinning regimes, and rotation age (Hoover and Rebain 2011).

Economic Benefit Functions for Carbon Storage

Evaluating the economic benefits of carbon storage by U.S. forests depends on identifying a monetary value per ton of carbon removed from the atmosphere. Monetary units are especially helpful because they can then be compared with monetary costs of carbon policies and programs. The value to society of sequestering or preventing the release of additional carbon dioxide can be viewed as the avoided economic damages or costs of additional carbon in the atmosphere. The value of carbon is not fully (or even mostly) reflected in market prices. Although both voluntary and compulsory carbon markets exist, the prices in these markets reflect a demand for mitigation that falls far short of estimates of the actual economic damage avoided by preventing the release of additional carbon to the atmosphere. It is this value—the value of avoided damage—that forest managers should use in decision-making if they wish to maximize benefits to society and to treat the benefits of carbon storage on par with the benefits of other ecosystem services, for which valuation methods are designed to capture the full social marginal value.

Economists use the term *social cost of carbon (dioxide emissions)* (SCC) to describe the marginal damages from a ton of carbon emitted to the atmosphere (Tol 2008). Storing an additional ton of carbon (above a given baseline) leads to less intense climate change and damage. The value of permanently storing that additional ton is equivalent to the SCC. This value depends on the specific trajectory of emissions, economic production, and climate change over time (Nordhaus 2008). Only if the world adopted an optimal incentive-based carbon policy would the resulting price of carbon exactly equal the SCC. In the absence of a complete, compulsory market for terrestrial carbon storage, a coherent approach to forest management that accounts for the full value of carbon storage will require coordination across agencies to choose a common methodology for the estimation of the SCC, and to jointly adopt revised estimates and management plans as economic conditions change.

The SCC depends fundamentally on estimates of total damages arising from a given change in climate over a specified period of time. As Tol (2009) notes, despite a proliferation of SCC estimates, there exist only 13 different studies of total damage estimates upon which to base estimates of the SCC, of which only nine had been used at the time of Tol's writing. Estimation of total damages remains an important and active area of research. Given the need for convergence in the selection of IAM approaches, features, parameter values, and underlying damage cost estimates, it is clear that estimates of the SCC will continue to evolve.

Several methods have been used to estimate the SCC, with important differences in the choice of model, scope, and parameterization. The academic literature lacks consensus in regard to these choices, and differences in approach have led to wide differences in SCC estimates. In a survey of the literature, Tol (2008) finds more than 200 different estimates of the SCC, with a wide and highly skewed distribution: mean, median, and mode values are 127, 74, and 35 $U.S. 1995 per Mg CO_2, respectively. The standard deviation in estimates is $243 per Mg CO_2.

Perhaps the most widely referenced SCC estimates come from the Dynamic Integrated Climate-Economy (DICE) model (Nordhaus 1993) and the Regional Integrated Climate-Economy (RICE) model (Nordhaus and Yang 1996) developed and continuously updated by William Nordhaus. Other integrated assessment models (IAMs) include FUND (Tol 1995), PAGE (Hope 2006), and WITCH (Bosetti et al. 2007). Various scholars have adjusted the basic DICE/

RICE approach (though not all report changes in SCC): Sohngen & Mendelsohn (2003) incorporate the mitigation potential of forest carbon storage; Buonanno et al. (2003) and Popp (2004) include endogenous technical change; Sterner and Persson (2008) account for relative price changes and the consumption of non-market goods; de Bruin et al. (2009) integrate the costs and benefits of adaptation; Lemoine and Traeger (2012) incorporate tipping points and ambiguity aversion; and Cai et al. (2012) utilize a continuous-time framework to improve the temporal resolution and reliability of analysis.

Large differences in SCC estimates can be obtained even from the same model, depending on the choice of international equity weighting (Fankhauser et al. 1997) or intertemporal discount rate. Equity weighting is an attempt to correct differences in estimates of people's WTP for reductions in damages from climate change (e.g., a reduction in human mortality risk). These differences in WTP may arise because of differences in income or other socio-economic conditions, which may be considered unfair. Equity weighting can significantly increase aggregate (global) damage figures, although some specifications of weighting functions also imply reduced estimates (Fankhauser et al. 1997).

Two camps have emerged with regard to the appropriate choice of the intertemporal discount rate. The first insists on a parameterization that is consistent with an ethical framework that values future generations on par with the present (Stern 2007), which implies a very small pure rate of time preference (PRTP) that exceeds zero only to reflect the very small probability of non-existence of future generations. The second camp insists on a parameterization that is consistent with observed behavior and other economic model parameters (Nordhaus 2007), which implies a much larger PRTP. Kaplow et al. (2010) and Goulder and Williams III (2012) argue that these approaches can be reconciled: the PRTP used for evaluative purposes (for example, in a planner's social welfare function) need not be the same as the PRTP used for predictive purposes (i.e., in a positive economic model).

An important point of recent consensus among leading economists is that the discount rate should generally decline over time as a result of uncertainty, regardless of the choice of PRTP (Arrow et al. 2013). This means that the certainty-equivalent value of benefits and costs realized further out in time should be discounted at progressively lower rates. More concretely, in a three-period context, the value of consumption in the third period would be discounted back to the second period at a lower rate than that at which consumption in the second period is discounted back to the first. The "term structure" of declining discount rates should apply equally to all costs and benefits relevant to a given decision context, so the forest manager should ensure that the structure underlying any off-the-shelf estimate of the SCC to be incorporated into an integrated assessment model is consistent with the structure applied to other market and non-market benefits.

In 2009 the U.S. Government convened an interagency working group to estimate the SCC for use in regulatory analysis (Greenstone et al. 2013, U.S. Interagency Working Group 2010). They assumed a global perspective and used three IAMs (DICE, PAGE, and FUND). The estimates were subsequently updated in 2013 (U.S. Interagency Working Group 2013). Using a 3 percent discount rate, the working group estimated that SCC increased from $44 to $72 per Mg of CO_2 from 2015 to 2044 measured in 2016 U.S. dollars.

In addition to navigating the complications of SCC estimation and the alignment of discount rates, forest managers must also account for potential leakage and the impermanence of terrestrial carbon storage in calculating carbon values for large-scale policy decisions. Carbon leakage occurs when management actions that successfully lead to greater carbon storage locally indirectly create incentives for greater carbon emissions elsewhere. For example, a prohibition on harvest on all National Forest land would dramatically increase carbon storage in the National Forests. However, on a global timber market, the reduction in timber supply from National Forests may be partially or fully offset by increases in harvest on forest lands elsewhere, leading to additional carbon emissions that would partially, fully, or even more than offset the additional carbon stored locally. In contrast, an afforestation or reforestation project on marginal lands would produce carbon storage benefits with little risk of leakage.

For any given management plan, the value of carbon should be assessed on the carbon expected to be stored net of leakage. Murray et al. (2004) investigate the extent of leakage in the United States associated with forest set-asides (prohibitions on harvest), avoided deforestation for conversion to agriculture, inducements for afforestation, and an integrated afforestation-avoided deforestation program. They find leakage rates that range from minimal (<10 percent) to enormous (>90 percent) depending on the activity and region. Further, for small projects, leakage is usually small in absolute terms but larger in proportion to the direct project benefits compared with the leakage rate of larger projects. They conclude that leakage effects should not be ignored in accounting for the net level of greenhouse gas offsets from land-use change and forestry mitigation activities.

Aside from the problem of leakage is the issue of impermanence in assessing the value of stored carbon. In the context of industrial production, the benefits arising from adopting a technology that decreases carbon emissions by one ton for a fixed amount of output are equivalent to the SCC. This equivalence stems from the fact that the ton of carbon that would otherwise have been emitted is permanently kept out of the atmosphere. However, in the context of forest and range land management, additional carbon storage may not be permanent. For instance, increasing timber rotation ages leads to greater carbon storage, but this carbon will eventually be re-emitted to the atmosphere after harvest.

Managers can account for impermanence in one of two ways. The first is to value both the carbon sequestered (i.e., new, additional carbon stored) and the carbon emitted using the SCC at the time of sequestration and emission, discounting all costs and benefits back to the present (van Kooten et al. 1995). The second approach is to calculate a carbon rental value (Sohngen and Mendelsohn 2003). Conceptually, the rental value is equal to the interest one could earn on the proceeds obtained by selling the asset (a ton of stored carbon) at its current price (the SCC), minus any expected capital gains due to changes in the SCC.

WATER REGULATION

Forest structure and composition affect the quality of aquatic ecosystems, which in turn affect many different benefits, from irrigation water to clean drinking water (Table 1). We focus on the effects of forest management within the riparian zone and surrounding hillsides of the riverine system. We divide the section according to four environmental drivers of aquatic ecosystem health: flow regime, thermal/light inputs, sediment flux, and chemicals, nutrients, and pathogens.

Flow Regime

Flow regime refers to the quantity, rate, timing, and pathways of water through the watershed. It is characterized by base flow, seasonal timing and annual variation, frequent (e.g., 2 year) floods, and rare or extreme (e.g., 100 year) floods. Floods and droughts create a patchiness of riparian landscape important for variation in species and age class of species. In semi-arid regions, extreme floods bring large wood into riparian zones and rivers; in wet regions, large floods add wood to rivers by eroding banks and causing trees to fall into the channel (Naiman et al. 2008). After entering the channel, large wood helps retain organic matter, forms deep pools, and promotes nutrient uptake in the river. More frequent small floods serve to flush large wood and sediment down the river and eventually out of the system (Latterell and Naiman 2007). Flow regimes can be affected by diversions, dams, stream channelization, timber harvests, and wildfire.

Riparian and aquatic species have adapted to these natural flow regimes, creating locally distinct habitats (Lytle and Poff 2004, MacDougall and Turkington 2005). In the Pacific Northwest, the lives of salmon are lockstep with the hydrograph: high flows in fall cue spawning and create the necessary spawning habitat; baseflows in the dry season maintain juvenile habitat; and elevated flows in spring improve emigration out of the river. In the snowmelt-dominated streams of the Rockies, willows and cottonwoods release their seeds during the recession of spring floods when seeds find scoured ground and moist substrate needed for germination (Scott et al. 1997). Maintaining natural flow regimes on forests is an effective means of managing invasive species, providing adequate habitat, and sustaining human uses of water on the forest and in downstream communities.

Two main methods are used for studying the relationship between changes in forest management and changes in flow regime: paired watershed studies and forest hydrology models. Paired watershed studies compare the flow regimes of two or more watersheds with similar physical characteristics (climate, soil, etc.). During the study, one set of watersheds undergoes a management action (e.g., clearcutting, prescribed fire) and one set of watersheds remains undisturbed to serve as a control. Ideally, comparisons of changes (or lack of changes) between watersheds allows one to tease out effects of land cover, management actions, and disturbances on changes in the flow regime. Such studies have shown a wide range of effects from clearcuts and partial cuts of forested watersheds on streamflow. Beschta et al. (2000) looked at three small watersheds and six large basins in the western Cascades (H.J. Andrews Experimental Forest). During the period of study, two of the three small watersheds experienced typical forest management actions, including road building, clearcutting, cable logging, and site preparation. The third watershed was left undisturbed and served as the control site. They found peak flow increased 13-16 percent in the treated watersheds for 1-year recurrence interval events and 6-9 percent for 5-year recurrence interval events. Swank et al. (2001) examined changes over a 20-year period in a mixed hardwood covered watershed in the southern Appalachians (Coweeta Hydrologic Laboratory) following clearcutting and cable logging, compared to an untreated control watershed. They found annual flow increased 28 percent during the first year following logging but continued to decrease until virtually no effect was seen after the fifth year. Brown et al. (2005) reviews other paired watershed experiments that look at changes in water yield resulting from alterations in forest vegetation. Most studies find significant effects shortly after the vegetation change, but the effects are short-lived and dependent on the length of time it takes for the vegetation to grow back. Another review by Troendle et al. (2010) concluded that streamflow response varies by climate, species composition, and percentage change in vegetation density. Wetter regions like the southeast, northeast, north central, and northwest United States are likely to see increases in streamflow an order of magnitude larger than the arid southwest. Furthermore, due to variation in annual streamflow, about 20 percent of the basal area of the vegetation must be removed to see a statistically significant effect on streamflow (Bosch and Hewlett 1982, Hewlett and Hibbert 1967, Stednick 1996).

When long run observations or control sites are not available, the potential effects of management actions can be evaluated using forest hydrological models. Forest hydrological models are computational representations of the watershed that allow managers to run counterfactual experiments. Typical models account for soil-vegetation-atmospheric transfer, canopy interception of precipitation, evapotranspiration, channel routing, and effects of management actions like roads and culverts. There is usually a complexity vs. usability tradeoff in these models. Simple models use straightforward applications of the U.S. Soil Conservation runoff curve number for calculating the relationship between precipitation and runoff (e.g., Mishra and Singh 2003). The runoff curve number is a relatively straightforward formula that gives surface runoff as a function of hydrologic soil type, land use, treatment, and hydrological condition. By comparison, the Distributed Hydrology Soil Vegetation Model (DHSVM), widely used in the Pacific Northwest, is capable of modeling the effects of vegetation change in watersheds up to 10,000 km^2 in size at 100 m resolution, sub-daily timescales, and multi-year simulations (Wigmosta et al. 1994). Models like DHSVM are feature rich but are often maintained as research models, with limited user support and without emphasis on user-friendly interfaces. Model choice also depends on the scale of the management decision being evaluated. The Soil and Water Assessment Tool (SWAT) is a widely used tool capable of modeling at the river basin scale but not useful for finer details like the effect of roads on an area's hydrology. A good review of forest hydrological models is given in Beckers et al. (2009), and an inventory of forest hydrological models is maintained by Texas A&M University at http://hydrologicmodels.tamu.edu/models.htm.

A different set of tools may be more appropriate for evaluating effects of fire on flow. Hydrophobic soils and loss of ground cover following extreme fires increase runoff, with the potential for small rain events to cause large-scale flooding. These effects are measured by comparing historic flow rates with gauges and weirs and by computer simulations, as shown in: Moody and Martin (2001) for postfire analysis of the Buffalo Creek Wildfire; Robichaud et al. (2002) for postfire analysis of the Hayman Wildfire; Rosgen et al. (2013) for postfire analysis of the Waldo Canyon Wildfire; and Jarrett (2009) more broadly for the Colorado Front Range. Moody and Martin observed 140 percent increases in peak flows following the Buffalo Creek Fire. Jarrett (2009), using physical measurements, observed up to 400 percent increases in peak flows on the Colorado Front Range following wildfires. Rosgen et al. (2013) used computer simulations with the WRENSS Water Yield Model to project the effect of lost vegetation on runoff following the Waldo Canyon Fire. They expect over 1.4-2.9 inch increases in water yield for watersheds affected by the fire.

The benefits associated with a change in flow regime could include increases in the provision of water for downstream irrigation, reductions in risk of floods, better control of invasive species and preservation of native species, improved maintenance of fish and wildlife habitat, and better maintenance of river corridors important for recreation. This wide range of benefits implies a wide range of valuation methods. Here we focus on the value of the flow regime for downstream irrigation and flood control. Estimating non-consumptive value of instream water to provide recreation opportunities or protect endangered species habitat requires the use of non-market valuation methods described in the sections on recreation and wildlife.

When they exist, prices charged in competitive water markets can provide signals of the value of increased flow. Brown (2006) reviews 1,380 transactions in Western water markets between 1990 and 2003. Over half the sales were to municipal areas to satisfy the needs of fast growing cities, such as along the Colorado Front Range, near Las Vegas, and near Reno, Nevada. Over half of the sellers were irrigators. The median sale prices were $2,120 per ML (mega liter or 1 million liters) for municipal uses and $1,917 for irrigation. Eleven percent of water rights purchases studied in Brown (2006) were for environmental purposes and sold for a median price of $706. Most of these (105 of the 113) purchases were by government entities to maintain instream flows for the protection of aquatic species. Instream water is also valued for its contribution to recreation and for riparian and wetland restoration.

When water markets do not exist, benefits of water for commercial uses can be found by estimates of the *shadow price* of water, that is, the change in net profits from a small change in water use. Estimating the shadow price of water across producers is done frequently in production economics with mathematical programming (Scheierling et al. 2006), field experiments, and hedonic methods (Young 2005). Estimating a full production function for the relevant beneficiaries typically requires a great deal of information. Pattanayak and Butry (2005) demonstrate that when changes in flow induce a change in the derived demand for a weak complement in the production process (e.g. labor), the value of the change in flow can be calculated as the difference in the firm's surplus between the old and new derived demand curves. These demand curves may be simpler to estimate than the production function. In this and other methods it is important to account for spatial correlation among observations for downstream beneficiaries (Pattanayak and Butry 2005).

The value of reducing flood risk is often measured with avoided costs or avoided damages methods--essentially an accounting of downstream values at risk and the change in probability of flooding that results from changes on the land. While such valuation is conceptually straightforward, implementation can be complicated. Watson et al. (2016) demonstrate how predicted changes in the hydrograph for a given location along a river or stream due to changes in upstream land use can be used to predict the extent and depth of flooding in a residential area. The authors estimate the difference in total damages using GIS data on the location, type, and value of structures in conjunction with a flooding depth-damage function estimated on the basis of historical insurance claims. To calculate the expected annual value of the difference in flood risk, they apply their comparative valuation method to a series of historical flooding events of known exceedance probabilities; estimate a continuous probability-damage function; and evaluate the expectation of that function.

Thermal and Light Inputs

Temperature of streams is closely related to flow regime, and it is a major concern in forest management. Volume of water determines the dispersal rate of thermal inputs, and vegetation in the riparian zone helps regulate water temperatures. At the most basic level, vegetation shades the stream, reducing solar radiation and the stream's heat load (Davies and Nelson 1994; Hostetler 1991). Riparian vegetation also provides a number of buffering roles like trapping air next to the stream surface and regulating the temperature and flow of groundwater. These buffering roles are important in shallow saturated groundwater systems, where shade cools groundwater temperatures and as streams become larger (Hewlett and Fortson 1982). When aquifer recharge and floodplain storage occurs during winter or spring snowmelt, new additions are made when stream water is coldest. The cold water is later released back to the stream during periods of low flow, when stream temperature is likely to be the highest. Permanent changes in stream temperature can make formerly suitable habitat unusable for native species (Holtby 1988). Species like bull trout that rely on cold waters for spawning are already seeing loss in habitat due to climate change and increased fire in the high elevation streams they occupy (Isaak et al. 2010).

Understanding stream temperatures is important to forest managers for two reasons. First, many streams in the United States are already impaired due to temperature. Restoring vegetation along streams is an important management action in such watersheds. Second, as atmospheric temperatures increase due to climate change, water temperatures will also increase. Many aquatic species will become temperature-limited in their habitat, and threshold effects may cause dramatic changes in fish populations from small changes in stream temperature. In these cases, establishing refugia will be an important tool in preserving species and biodiversity (Isaak et al. 2015).

Models for projecting how stream temperatures change as a function of forest management fall into two broad categories, reviewed in Caissie (2006): statistical models using observed data on stream temperature, stream properties, and environmental variables; and models of energy transfer between the stream and the environment. Statistical models regress observed stream temperatures on a wide variety of stream and atmospheric variables to predict stream temperature and assess habitat suitability for aquatic species (Isaak et al. 2014). For example, Jeppesen and Iversen (1987) used air temperature, solar radiation, and depth of water in a statistical model to predict stream temperatures. A new class of spatial statistical models, called spatial statistical network models, account for spatial dependence between observations in a stream network (e.g., temperature in a downstream segment of river is highly dependent on temperature in upstream segments) and have been shown to significantly improve predictive power (Isaak et al. 2014). Energy transfer models are based on a system of physical relationships for heat transfer between streams and other factors in the environment. Models are calibrated by adjusting system parameters for variables like solar radiation and wind speed to minimize the errors between observed and predicted stream temperature.

Generally, regression models are very good at predicting changes in temperature at a given site within the historic range. Energy transfer models are more often used when evaluating impacts outside of the historic range or when historic data is not available, such as when evaluating the impact of thermal effluent from power plants, coldwater releases from reservoirs, and large-scale vegetation removal (Kim and Chapra 1997, Sinokrot and Stefan 1993, Vugts 1974, Younus et al. 2000).

Stream temperature is not typically valued *directly* as an ecosystem attribute. Instead, changes in temperature are linked to changes in an attribute of ultimate interest to beneficiaries. For example, Isaak and Hubert (2004), in a study of trout populations in the Salt River, examined the effect of temperature on cutthroat, brown, and brook trout abundance. They found populations peaked near 12 degrees Celsius, with viable ranges between 3 and 21 degrees Celsius. Similar findings were found for trout in Michigan and Wisconsin rivers (Wehrly et al. 2007) and for fish abundance in Michigan rivers (Creque et al. 2005). Changes in abundance of aquatic species, particularly species targeted for recreational fishing, can then be valued with non-market valuation methods, which are described in detail in the sections on recreation and wildlife and Appendix 1. McKean et al. (2010) use a travel cost method to estimate a demand curve for steelhead trout in the Snake River.

Dalton et al. (1998) use contingent valuation methods to value hypothetical increases in trout populations, and Sorg et al. (1985) use contingent valuation methods to value hypothetical increases in the likelihood of catching coldwater fish. Values from these studies range from $100 per trip in Dalton et al. to about $32 per trip in Sorg et al. (1985).

Wu and Skelton-Groth (2002) offer an interesting example of when restoration focused on decreasing stream temperatures does not make sense. They examine optimal investments in streamside vegetation in Oregon's John Day River Basin, where flow alterations and reduced vegetation have produced water temperatures unsuitable for native rainbow trout and Chinook salmon. They concluded that four out of 10 streams studied should not be targeted for restoration either because their current temperatures were so high that achieving suitable temperatures was near impossible, or because other water quality conditions were too poor to sustain fish even if suitable temperatures were achieved.

Sediment Flux

River systems require sediment and organic matter to form habitat and provide nutrients to the stream. Runoff carries necessary sediment to the streams, and periodic flooding of rivers deposits sediment within the floodplain. Deposited sediment rebuilds wetlands and provides rich soil for vegetation, which in turn slows runoff and captures sediment, allowing the "right" amount of sediment to enter the stream (Cummins 1974). In small streams, too much sediment often arises due to deforestation and roads that increase surface runoff. Increased rates of water and sediment delivered to the river accentuate high and low flow conditions (Bormann and Likens 1979), leading to the widening and shallowing of stream channels (Knapp and Matthews 1996, Richards et al. 1996, Sidle and Sharma 1996). When stream bottoms accumulate too much sediment, they choke important plants and damage spawning grounds. About 15 percent of all impaired waters in the United States have sediment loads above EPA water quality standards (U.S. Environmental Protection Agency 2016).

Several models exist for projecting the effect of land management on sediment delivery. Most of these models use a variant of the Universal Soil Loss Equation (USLE). The USLE calculates the amount of long-term annual sheet and rill erosion, given by

$$A = R \times K \times LS \times C \times P$$

where A is the computed annual erosion, R is a rainfall and runoff factor, K is a soil erodibility factor, LS is a topographic factor, C is a vegetation and management factor, and P is a conservation factor. USLE, at least in its early forms, was limited to estimating annual averages of sediment from a hillslope and was not suitable for estimating sediment delivery to offsite streams or reservoirs via stream networks, nor was it capable of channel erosion. These shortcomings led to the development of the Water Erosion Prediction Project (WEPP) model. The WEPP model is a physically based model, meaning it uses sub-models to simulate physical processes tied to erosion, including infiltration, runoff, sediment transport, deposition, plant growth, and residue decomposition (Flanagan et al. 2007). The model is parametrized based on ongoing field studies and is constantly being integrated into new technologies and models. Soto and Diaz-Fierros (1998) found WEPP model predictions were very close to actual measured sediment in watersheds following a controlled burn and a wildfire. The Sediment Tool, similar to WEPP, uses USLE with a routing model to project total sediment yield in medium-sized watersheds. Riedel and Vose (2002) use the Sediment Tool to estimate reductions in sediment from restoration along the Conasuaga River in northern Georgia and southeastern Tennessee. The Conasuaga River is an important drinking water source, is extensively used for recreation, and has high sediment due to erosion from agricultural lands, streambanks, and forest roads.

For large watersheds, the SWAT may be more appropriate. SWAT has been used extensively to model sediment and agricultural chemical movement in large complex river basins, with several studies targeting transportation of nonpoint source pollutants (e.g., Santhi et al. 2001). In general, process models like WEPP and SWAT tend to underestimate erosion from large events and overestimate erosion from small events. Shen et al. (2009) compare WEPP and SWAT; they find the two models produce very similar results for sediment yield, so the choice between the two may depend on other uses of the model. SWAT, for example, may be preferable as nutrient or chemical transport is also of interest.

The Watershed Assessment of River Stability and Sediment Supply (WARSSS) is a procedure developed by the EPA (Rosgen 2007) meant to guide land managers in watershed scale assessments of erosions. WARSSS begins with an initial survey of the watershed to identify likely areas of erosion and to rule out watersheds that are not likely to be sources of erosion. Then, a more intense look at the watershed is aimed at matching the scale and scope of assessment and management activities with the resources available to the land manager. Final assessment is then done on the areas deemed high priority. The assessment phase could use any of the tools listed above, but WARSSS also recommends a few user-friendly spreadsheet-like tools including the Bank Erosion Hazard Index (BEHI) and the Near Bank Stress estimation tool for streambank erosion, the Hillslope Processes tool for sediment from hillslopes and roads, and the WRENSS model for mass wasting.

The value of reducing sedimentation, in places where there is too much, may be measured with avoided costs or damages. One of the key costs associated with too much sediment is the increased cost to municipal water suppliers for dredging reservoirs and removing sediment in drinking water. Moore and McCarl (1987) detail persistent costs of soil erosion in the Willamette Valley in northwestern Oregon. They found that a 1 percent reduction in turbidity reduced water treatment costs by one-third of 1 percent, or about $3,385 annually across the region. Additional costs of sediment in the Willamette Valley were $4.22 million annually to clean ditches and culverts and $0.85 per ton to dredge and remove accumulated sediment from the Port of Portland. Forster et al. (1987) found that a 10 percent reduction in soil erosion in Ohio's corn belt would reduce water treatment costs by 4 percent. Dearmont et al. (1998) found that a 1 percent reduction in turbidity led to a one-fourth of 1 percent reduction in treatment costs across water suppliers in Texas. Using a dataset of 430 of the largest water utilities in the United States, Holmes (1988) found a 1 percent increase in turbidity leads to seven hundredths of one percent increase in operating and maintenance costs.

Perhaps the largest collection of values for reductions in sediment is given in Hansen and Ribaudo (2008). They give values for reductions in erosion for 14 categories of economic and ecological benefits in each of the 2,111 eight-digit hydrologic unit code watersheds in the contiguous United States. Included benefits range from improvement in catch rates for marine fisheries to avoidance of groundings by shipping fleets to better water for recreation. And though the estimates were intended for reductions in erosion from agricultural fields, they are easily transferable to other sources of erosion in the same watershed. The values are estimated at a variety of spatial scales using damage functions, replacement costs, travel costs, and averting behavior models.

Chemicals, Nutrients, and Pathogens

Nutrients used in agriculture are the leading cause of water impairments on rivers and lakes in the United States, and they are the second leading cause of impairments to wetlands. Pathogens such as Cryptosporidium, Giardia, and coliforms enter waterways from drainage of lands with animal feces or open sewage systems. Fertilizers and pesticides used on agricultural fields include phosphates, nitrogen, and potassium from manure, chemical fertilizers, and sludge. High nutrient levels cause algae blooms, lead to organic enrichment and oxygen depletion in water bodies (fourth leading cause of impairment), taint drinking water, and causes fish die-offs from too little oxygen. Drinking water with nitrates can cause methemoglobinemia, which can be fatal in infants.

Studies show riparian vegetation routinely removes as much as 90 percent of nitrates in the groundwater (Hill 1996). The USDA's National Agroforestry Center (NAC) advocates the use of forested buffers along agricultural fields to diversify farm income, reduce soil erosion, improve water quality, and increase wildlife habitat. NAC's tool Buffer$ (http://nac.unl.edu/tools/buffer$.htm) provides site-specific cost and benefit measures of forested buffers compared to traditional farming methods. The SWAT model, described above, also models nutrient and chemical movement through the watershed. If both sediment and nutrients are of interest, using one model for both may make the most sense. The USGS's SPARROW model models instream measurements of water quality as a function of upstream landscape characteristics within the watershed. SPARROW is unique in that it models the source of nutrients in a given body of water rather than the flow of nutrients through a watershed. Applications of SPARROW have mostly been on very large watersheds. It has been used to identify the sources of nutrients in U.S. streams (Smith et al. 2003b), to identify sources of phosphorous and nitrogen entering the Gulf of Mexico from the Mississippi River basin (Alexander et al. 2007a), and to assess the role of headwater streams in downstream water quality in the Northeast (Alexander et al. 2007b). Because of the scale of many of these applications, SPARROW may be useful for large regional collaborative projects but may not be useful for most forest management activities.

Similar to valuing the benefits from reduced sediment in a water body, one way to value the contribution of forest and riparian vegetation to drinking water quality is to use an estimate of the cost avoided if the ecosystem service is preserved. That cost should include expenditures to avert or mitigate the increased pollution as well as any damages that arise from the portion of additional pollution that is not mitigated downstream. Many studies approximate avoided cost by estimating the cost of (fully) replacing the functional benefits of the ecosystem service. For example, Kapoor and Viraraghavan (1997) estimate the costs of various treatment processes for the removal of nitrates from drinking water. Replacement costs have been used to value the preservation of ecosystem services that result in filtration avoidance waivers, which allow water treatment plants to use lower cost treatment methods. The 1996 amendments to the Safe Drinking Water Act required all surface water systems to be filtered for Giardia, viruses, bacteria, and turbidity unless managers could control human activities in the watershed. Any water treatment plant capable of demonstrating high quality of raw water at their intakes could forgo many of these treatment measures. In the case of New York City's Catskill/Delaware water supply system, the largest unfiltered water supply system in the county, the city's investment in land purchases and conservation easements resulted in an avoided expenditure of $6-8 billion in capital costs plus operation and maintenance costs of $300 million annually (Chichilnisky and Heal 1998).

When achieving a standard mandated by law, replacement cost may be the most important factor for decisionmakers. However, in these (and other) cases, the replacement cost may exceed the economically efficient expenditure on averting or mitigating pollution downstream. If some policy or management alternatives that achieve the legal water quality standard involve changes in other ecosystem services, it may be useful to have estimates of the benefits per se of water quality changes in order to compare with the value of changes in other services. In this case, the so-called "value of statistical life" (VSL) can be applied to quantify the benefits of small changes in the probability of mortality due to changes in chemical or nutrient flux (e.g., Hanley 1990). The EPA uses a variety of approaches to estimate a VSL (currently at $7.4 million in 2006 dollars) to evaluate costs and benefits of regulations. A review of VSL practices is given in Viscusi and Aldy (2003).

AESTHETIC AMENITIES

Forests provide aesthetic amenities to residents and visitors (Table 1). Attributes of individual trees and the forest itself, like color, size, texture, and shape, affect scenic quality as well as peoples' feelings of shelter and security associated with trees and forests. There is a large literature on the assessment of forest amenities, which relates biophysical features of the landscape such as forest composition and structure to design- or perception-based parameters that are assumed to be measures of aesthetic quality (see Ribe 1989 and Daniel 2001 for reviews). These design- or perception-based measures can then be used to define desired landscape features and cost-effective ways of attaining them (e.g., Brown 1987, Ribe et al. 2002). Rather than focus on assessment, here we focus on economic benefit functions to estimate people's WTP for aesthetic amenities, as expressed through stated or revealed preference methods.

Stated preference methods are quantitative techniques for eliciting an individual's preferences by asking the person to choose from different hypothetical alternatives. Different types of choice experiments have been used to estimate people's WTP for programs that increase the scenic quality of forests (e.g., Biénabe and Hearne 2006, Haefele et al. 1992). One type of choice experiment is called the contingent valuation method, which presents a respondent with a simple yes/no decision for the provision of a particular environmental service at a particular price. A more general type of choice experiment presents the respondent with a menu of options of different environmental services at different prices (Adamowicz et al. 1998). Stated preference methods are discussed in detail in the section on wildlife valuation. Forest insect and disease control are examples of programs that aim to improve aesthetic and recreation quality. Several studies have used stated and revealed preference methods to estimate people's WTP for pest control programs and are reviewed in Rosenberger et al. (2012).

Revealed preference methods attempt to estimate individual preferences based on observable decisions in actual markets for a good whose value depends on the non-market ecosystem service. Revealed preference methods have been used to estimate the value of aesthetic amenities in urban and rural forests based on home sales prices in different housing markets and analyses of how aesthetic amenities of forests affect those prices. Urban and rural forests improve the quality of life of nearby residents. Forests affect the scenic quality of a neighborhood, provide privacy, reduce stress, shelter residents from the negative effects of undesirable land uses, and improve retail areas by creating environments that are more attractive to consumers (Dwyer et al. 1992). The value of changes in the quality of residential amenities can be estimated by relating home sale value to forest characteristics in hedonic price models (Rosen 1974). The hedonic technique treats properties as bundles of amenities and statistically decomposes sale value into the measurable quantity of each amenity multiplied by its implicit price or marginal value. While economists continue to improve statistical methods of hedonic estimation (e.g., Bajari et al. 2012, Bayer et al. 2016, Bishop and Timmins 2011, Kuminoff et al. 2010), the method itself has a relatively long and robust tradition of use in economics.

For houses in Portland, Oregon, Netusil et al. (2010) find that increasing the acreage of forest patches with greater than 76 percent canopy closure within a quarter mile increases sales prices in neighborhoods with low levels of such cover but decreases the value of homes in neighborhoods where cover is already high. The authors hypothesize that in the latter neighborhoods, increasing cover offers little additional benefit and instead may block desirable views. Indeed, Sander and Polasky (2009) find no statistically significant evidence that increasing the *view* of forest cover increases home values, though Sander et al. (2010) find that marginal increases in canopy cover within 250 meters (approximately 0.15 miles), and especially within 100 meters, of homes in the Minneapolis-St. Paul metro area lead to statistically significant but practically very small increases in sale value (at the mean, a 10 percent increase in cover increases sale value by 0.48 percent).

In these and other cases, it is difficult or impossible to disentangle the value of aesthetic amenities per se from that of other benefits related to the same forest attributes. For example, Stetler et al. (2010) find that views of areas burned by wildfire depress home values in Montana by as much as 14 percent, depending on proximity to the burn. Because proximity mattered, and because the effect on home prices appeared persistent, the authors suggest that part of the loss in value may be attributed to the perceived risk of property damages from future wildfires. In this case, forest structure and composition regulate fire occurrence and behavior. Changes in home sales prices associated with changes in forest structure and composition may represent homeowners' pricing of both aesthetic amenities and fire risk. Similarly, Kovacs et al. (2010) find that Marin County (California) homes near oak woodlands subject to infestation of Sudden Oak Death suffer declines in property value on the order of 3 to 6 percent, which may be attributable to a combination of diminished aesthetic amenities, higher risk of damage from dead trees, or reduced recreational opportunities. As a notable exception, Kim and Johnson (2002) identify effects that are likely exclusively aesthetic. They find lower sale prices for Oregon homes from which forest clearcuts (-$16,381) or pasture (-$25,994), rather than mixed-species standing timber, were visible at the time of purchase.[1]

Ambiguity regarding the services associated with a quantitative estimate of marginal value from hedonic price models can confound interpretation for management purposes. For example, Kim and Wells (2005) find that increasing the area of medium density forest within 0.5 km of homes near Flagstaff, Arizona,

[1] Numbers in parentheses correspond to 1995 USD; the authors do not report average sale prices.

increases their value, while increasing the area of high density forest decreases their value. The authors appeal to literature on the determinants of scenic beauty to argue that these values reflect the aesthetic benefits of lower density. Consequently, they argue that their estimates should be added to the value of reduced fire risk in assessing the total benefits of potential fuel treatments that reduce forest density. However, it is quite plausible that the estimated hedonic prices of forest density also (or even exclusively) capture the value of changes in fire risk, whether in whole or part.

Similarly, the value of recreation opportunities in nearby forests may or may not be capitalized into housing values, and this ambiguity can complicate benefit-cost analysis. To the extent that the value of recreation opportunities is reflected in property values, the marginal value of a change in forest quality will be captured by the hedonic method. For instance, Kovacs (2012) finds that improvements in the quality of regional parks near Portland, Oregon, increases home values within 5 to 10 miles. However, adding an estimate of the "amenity" value obtained through hedonic price analysis to an estimate of the "recreational" value obtained through one of the travel cost-based methods will then double-count some portion of the recreational value and thus overstate the benefits of management or policy interventions meant to enhance forest quality (McConnell 1990).

Another example of ambiguity comes from the literature on the hedonic valuation of landscape configurations. Several hedonic price studies have used forest landscape metrics such as forest patch size, density, and diversity. In pioneering work, Geoghegan et al. (1997) estimate the hedonic value of changes in indices of landscape diversity and fragmentation in Maryland's Patuxent watershed. The authors argue that the value of changes in these landscape metrics obtained from hedonic values represent homeowners' WTP for a suite of services. The problem is that the authors do not establish the relationship between the landscape metric and the quality of services, which may depend on forest attributes unrelated to the landscape metrics. As a result, the policy relevance of these landscape metrics for the management of forest ecosystem services is not clear. Even the use of more straightforward measures, such as canopy cover within a given radius of the home (e.g., Netusil et al. 2010, Sander et al. 2010), is subject to important ambiguity when it does not distinguish among forests with different uses and ownership types (Mansfield et al. 2005).

These ambiguities could be reduced by using landscape metrics that are significant in perception-based measures of visual quality (e.g., Ribe et al. 2002).

While forest cover provides amenity benefits to nearby residents, forest cover along travel routes, particularly backcountry and scenic byways, provides aesthetic value to leisure travelers and commuters (Ben-Akiva et al. 1984). Despite the potential importance of these benefits and the mandate of Federal agencies, such as the BLM, to manage lands for their scenic value, little research has been done to estimate the economic value to travelers of changes in particular forest attributes. Alivand et al. (2015) estimate a route choice model that accounts for a variety of route and view characteristics. Though not a formal valuation exercise, they find that forest cover contributes significantly to travelers' choice of scenic versus faster routes.

RECREATION

Forests provide many types of recreational benefits—opportunities for hiking, camping, mountain biking, and more. A recent analysis suggests an average net economic value of access to National Forests of approximately $90 per person per trip (Bowker et al. 2009). Estimates such as these rely on the travel cost approach (Clawson and Knetch 1966), which estimates recreationists' WTP for access to a site based on their opportunity cost of travel to the site. Relating data on trip frequency to the implicit price of a trip (the travel cost), the analyst estimates a demand curve for trips to the site in question. The demand curve can then be used to estimate the consumer surplus or net economic value of access to the site.

While forest management decisions may be as straightforward as allowing or prohibiting access to a given area, in many cases management decisions affect the quality—not the possibility—of the recreational experience. Altering attributes of the forest site, either directly or indirectly, can influence the attractiveness and value of a site for recreation. Assessing the recreational value of changes in site attributes requires a more sophisticated approach to valuation. In this section, we describe economic benefit functions that have been developed to estimate the monetary value of changes in forest attributes that affect recreation quality.

Economists have used a variety of valuation techniques to estimate the marginal values of forest attributes to recreationists (see Appendix 1 for descriptions and discussion of these methods). Studies have focused on several different attributes, including the size of trees (basal area), stand age, and species composition. For example, Englin and Mendelsohn (1991) estimate hikers' WTP for trail attributes in Washington State wilderness areas using the hedonic travel cost method (HTCM) (Brown and Mendelsohn 1984). These attributes include clearcuts, old growth, and species type. The authors find that the marginal social value of old growth for recreationists varies substantially from site to site and can even be negative (presumably due to over-satiation). Similarly, they find that the social value of a clearcut along a trail is positive in some cases. The positive values of clearcuts may be due to enhancing the populations of certain wildlife species that hikers value for viewing, which were unobservable to the analysts; clearcuts per se may not necessarily have positive aesthetic benefits.

Pendleton et al. (1998) and Englin et al. (2006) both find that the marginal value of changes in forest attributes varies substantially from site to site. Pendleton et al. use HTCM to measure marginal social values of forest attributes along hiking trails in the southern Appalachian Mountains, including wilderness areas, state parks, and Great Smoky Mountain National Park. They find that average basal area has an average marginal social value ranging from \$8.50 per m^2/ha in North Carolina to \$372 in Georgia. The marginal value varies from site to site depending on current basal area as well as the characteristics and origins of trail visitors. Englin et al. (2006) use the random utility model (RUM) to estimate the value of changes in the length of backcountry trail segments running through different age class and species combinations in Jasper National Park (Canada). They find that the average recreationist places additional value on "truly ancient" forest (i.e., greater than 300 years in age). Notably, the authors stress that perhaps the most important insight from their study is the fact that welfare effects due to changes in forest characteristics vary over sites and users.

A critical challenge for predicting the full impact of a policy or management intervention is the valuation of natural changes in forest characteristics associated with stand growth over time. Hilger and Englin (2009) account for dynamic effects on site choice of fire across multiple trails, allowing for correlation across sites and over time. Englin et al. (2001) estimate the change in recreational benefits as forest areas recover from fire in the Intermountain West. They find that the effect of fire on recreational benefits is non-linear, with early benefits that gradually decline and then increase again as the forest matures and returns to normal. The authors note that early benefits are likely due to the novelty of postfire vegetation and wildlife. Managers should be careful to consider how these novelty-based benefits change if they are substantial and if fires become more frequent as a result of fire management.

In many cases, insufficient information on stand ages or condition leads the analyst to define recreationists' preferences for stand attributes simply as preferences over discrete age classes (e.g., 30 to 50 years versus 50 to 100 years). The consequence of this for projecting out the impacts of a policy intervention over time is that the estimated value of a stand for recreation will suddenly jump in an unrealistic fashion when it crosses an age class threshold. Englin (1990) avoids this by employing a piece-wise linear interpolation to approximate the dynamic path of recreational value of a stand from clearcut to old growth, based on consumer surplus estimates associated with different age classes obtained from an HTCM study. However, when preferences are defined over, say, length of trail passing through forest of a certain age or type, as in Englin and Mendelsohn (1991), changes in individual stands cannot be evaluated in isolation: the marginal value of a mile of trail through a given stand type depends on the existing length of trail through that stand type.

What we are in the habit of measuring or can measure easily is not necessarily the valued forest attribute of a given recreational service. For example, stand age or basal area may not be the valued attributes of recreationists; however, they may be proxies for other characteristics that recreationists do value. Pendleton and Shonkwiler (2001) deal with the problem of highly collinear site attributes, which one might expect in comparing trails or sites associated with different ecosystems. These "bundled" attributes may not be valued individually but rather combined in unknown ways to represent characteristics of value to the recreationist. Moreover, without sufficient independent variation, reliable estimation of preferences for these individual attributes may not be statistically possible or policy-relevant. The authors jointly estimate the latent relationship between site attributes (e.g., basal area, number of large trees, species type) and ecosystem characteristics ("riparian" and "upland") along with recreationists' preferences over these characteristics and other site attributes. This allows the indirect, but more reliable, estimation of the value of changes in site attributes, which are subject to management action.

Another important source of ambiguity is that what may be an ecosystem service at one level of provision may be a disservice at a different level. For example, increasing crown cover from low to moderate levels may be valuable to hikers, while increasing it from, say, 90 percent to 99 percent may actually decrease the value of the recreational experience. This potential for "over-satiation" can make it difficult to differentiate between results that faithfully capture such nuance and results that are simply implausible. A further and potentially related source of ambiguity is that some ecosystem characteristics may represent services to some recreationists while they are disservices or irrelevant for others. For instance, Hesseln et al. (2003) and Loomis et al. (2001) find that welfare effects of changes in forest characteristics due to fire differ across hikers and mountain bikers.

WILDLIFE

Forests provide habitat for wildlife populations, which contribute to a wide range of benefits (Table 1). Many wildlife species provide benefits related to their recreational, aesthetic, or spiritual value. Many species also play an important role in disease control, pest control, pollination, nutrient cycling and soil formation, or maintenance of genetic diversity. Wildlife populations may also have costs, for example, if they are vectors of disease, damage native vegetation, prey on livestock, or are a threat to human safety. In this section, we describe approaches to projecting attributes of wildlife populations (i.e., ecological production functions) and to estimating the monetary value of changes in wildlife populations (i.e., economic benefit functions). Our examples focus on terrestrial fauna, which has received the most attention in the literature.

Ecological Production Functions to Predict Changes in Wildlife Abundance

The fields of wildlife management and conservation biology contain a long line of research on demographic models of population abundance and viability, which are commonly used to inform wildlife management decisions (see Akcakaya et al. 2004, Beissinger and Westphal 1998, and Beissinger et al. 2009 for reviews). This research has spawned commercial software packages such as RAMAS (Akcakaya 2005) and Vortex (Lacy 1993) that allow users to build, run, and analyze models for species with virtually any life history. Here, we briefly describe the major types of model structures and give some early examples of their use in wildlife management. We conclude with concerns about the reliability of model projections and some ways to address them.

Deterministic population models use information about the age, stage, or social structure of the wildlife population, age or stage of first reproduction, and estimates of reproductive success and survivorship for different ages or stages to create a matrix model that projects the distribution of individuals over age or stage classes in discrete time periods (Getz and Haight 1989). When demographic parameters such as survival and fecundity are related to habitat attributes using resource selection functions (Boyce and McDonald 1999), matrix models can be used to assess the impacts of habitat management activities. For example, Beissinger (1995) used matrix models to project trends in the population of the marbled murrelet, an endangered bird species in the western United States that depends on old growth forest habitat.

Stochastic population models are similar to matrix models except point estimates of demographic parameters are replaced with probability distributions. Monte Carlo methods are used to sample from the probability distributions and project a population by varying demographic rates in each time step. Each projection follows a unique trajectory and yields a different ending population size, and as a result, stochastic demographic models yield probabilistic results (e.g., probability of population exceeding some minimum level after 100 years). Model projections must be replicated many times with different random number seeds to adequately sample combinations of parameter values and obtain a precise estimate of the desired measure of population abundance. Effects of different management options can be built into stochastic population models. For example, Lindenmayer et al. (1993) projected how measures of population abundance of a threatened mammal differed under silvicultural practices that affected habitat carrying capacity.

Modeling approaches increasingly incorporate spatial structure, for both individuals and populations. Metapopulations are spatially structured groups of local breeding populations. Migration of individuals between habitat patches affects local population dynamics, including the possibility of re-establishing populations in a patch after local extinction. Metapopulation structure is incorporated into demographic models using the dispersal of individuals to link local habitat patches. Metapopulation models typically incorporate patch-specific demographic rates and dispersal rules that are based on patch size and interpatch distances. Patch quality can be represented by varying carrying capacity or reproductive output among patches. The advantage of metapopulation models over single-population models is they partially incorporate spatial realism. Thus, the effects of landscape change can be modeled, including effects of corridors, patch-specific habitat destruction, quality alterations, and changes in interpatch distances. For example, Bevers et al. (1997) built a deterministic metapopulation model of a black-footed ferret population and examined how patch-specific habitat management strategies affect projected population size. Stochastic metapopulation models usually include demographic and environmental stochasticity and catastrophes, with the added dimension that an understanding of covariation of rates between patches may be important. Model outputs can be expressed as ending metapopulation size, the likelihood of extinction for the whole metapopulation, the percentage of patches occupied, or the minimum number of patches or area required for metapopulation. For example, Haight et al. (2004) use a stochastic metapopulation model of kit foxes to predict and compare measures of population abundance under different options for habitat protection. Spencer et al. (2011) and Scheller et al. (2011) modeled fisher population health under forest succession, treatment, and wildfire in stochastic metapopulation models.

Spatially explicit population models specify the location of the desired unit (e.g., individuals or populations) within a heterogeneous landscape and define spatial relations between habitat patches and the surrounding landscape matrix (Dunning et al. 1995). Grid-based models subdivide the landscape into cells and specify demographic parameters for the local population in each cell. Movement between patches is determined by immigration and emigration rates. Individually based models keep track of the location of each individual across the landscape, and the demographic attributes are assigned based on the patches they occupy. Thus, each individual marks a trajectory over the landscape during the course of a population projection. In both grid-based and individual-based models, demographic parameters are treated as random variables and movement rules explicitly incorporate an animal's perception of the landscape. Because spatially explicit models provide a technique for modeling processes that operate from local to landscape scales, they can potentially predict population and community changes in response to land-use changes, climatic alterations, or various management strategies. For example, Lamberson et al. (1994) and Marcot et al. (2015) use individual-based models to project spotted owl populations under different forest management strategies in the Pacific Northwest. Cohen et al. (2014) analyze the effects of landscape composition and configuration on migrating songbird populations using an individual-based model.

There are several important concerns about the accuracy and interpretation of results from demographic models (Beissinger and Westphal 1998). Models use detailed demographic data, but information available may be inadequate, imprecise, and based on studies too limited in duration to properly estimate means and variances in vital rates. Long projections cannot be validated against short-term observations. Future changes in habitat quality or quantity are often unknown. Differences in model structure can have strong effects on management recommendations. Taken together, these concerns strongly suggest that one should place very limited confidence in the long-term projections generated by these models. Nevertheless, these concerns can be reduced by making short-term projections under different scenarios of habitat quality or management to estimate changes in population abundance relative to a suitable baseline. These projections of marginal changes in population abundance can then be validated using field studies, experiments, and sensitivity analyses (e.g., Wood et al. 2015). Projections from demographic models have been successful in determining resource management decisions for spotted owls and grizzly bears because models were followed by comprehensive field studies and were thoroughly reviewed and revised (Boyce 1993, Lamberson et al. 1994, Marcot et al. 2015).

Economic Benefit Functions for Wildlife Abundance

Estimating the monetary value of changes in wildlife abundance depends on whether the affected benefits are commercial or recreational in nature. Although market hunting of wildlife is largely illegal in the United States (Geist 1988), fur is a valuable commodity that is sold in markets. The harvest of furbearing animals in the United States has declined over the past several decades, with harvests averaging approximately 4.4 million pelts per year in the past decade, but harvests do increase when pelt prices rise (Flather et al. 2013). Valuing the commercial effects of changes in the abundance and spatial distribution of a furbearing animal population requires not only an appropriate ecological model but also an economic model to predict changes in the cost and quantity of harvest. The standard, information-intensive approach to valuation in this case is the specification and estimation of a full profit function for the affected producer(s), in which the wildlife species is an input to the production process. In some cases, however, it may be possible to estimate changes in producer surplus by estimating changes in the derived demand for a weakly complementary input (Huang and Smith 1998, Pattanayak and Butry 2005).

In addition to commercial effects, changing forest management practices may also have effects on the quality of recreational hunting. Sometimes the right to hunt on private land is sold in the market, and data on hunting leases can be used in hedonic analyses to estimate the monetary value of changes in the quality of hunting (Livengood 1983, Rhyne et al. 2009). Hunting leases include premiums or discounts based on density of wildlife, structure and composition of vegetation, distance to cities, degree of congestion, and size of the hunting area. The value of each site depends on the exact mix of characteristics as given by a hedonic price function. For example, Livengood (1983) estimates a hedonic price model for the value of hunting white-tailed deer in Texas from a survey of hunters during the 1978-79 hunting season. From this hedonic model, Livengood (1983) estimates the demand (marginal willingness to pay) for wildlife and finds that hunters are willing to pay \$23 for a single deer while an additional deer was worth only \$13. These values can then be used to estimate the effects of changes in management that affect deer density and hunting success. When information about hunting leases is not available, valuation of recreational hunting can be made using methods for valuing other types of recreation. For example, Knoche and Lupi (2007) use the random utility travel cost method to estimate demand for deer hunting sites on private farm land in southern lower Michigan and estimate the value of increasing public access for deer hunters to 10 percent of the private lands. The estimated aggregate WTP for this increased access was about \$39 per acre, which is much higher than the average amount paid to farmers who enroll in Michigan's Hunter Access Program.

When it comes to hunting on public land, hunters do not purchase hunting trips in a true market (much less do they purchase any particular forest attribute that determines the quality of the overall hunting experience); instead they purchase hunting licenses at fees that are set by the government and incur costs to partake in hunting (travel, supplies). Nevertheless, hunters' choices of recreational sites reflect their implicit willingness to pay for more (or less) of particular attributes (e.g., wildlife population density), and researchers have used a variety of models to estimate the monetary value of changes in the quality of hunting experiences. These are the same models used to value changes in forest attributes for other types of recreationists, including the approximately 2.3 million visitors to National Forests with the aim of viewing wildlife (Mockrin et al. 2012). Valuation approaches include the hedonic method (Livengood 1983, Rosen 1974), HTCM (Brown and Mendelsohn 1984, Mendelsohn 1987), the Kuhn-Tucker method (von Haefen and Phaneuf 2005), and both revealed and stated preference discrete choice RUMs (Boxall et al. 1996, Hanemann 1984, Morton et al. 1995). See Appendix 1 for a discussion of these methods.

Knoche et al. (2015) provide a notable application of integrated modeling and valuation to assess the benefits to pheasant hunters of spatially targeted habitat restoration efforts. Utilizing a previously developed, spatially explicit model of ring-necked pheasant sightings in conjunction with a RUM of hunter site choice, they find that the value restoration investments varies considerably across space, from near $0 to as much as $2.50 per acre (or more than twice the median value).

Beyond recreation, cultural values (aesthetics, spiritual value, and existence value) associated with wildlife populations are more challenging to quantify. Much of the research on non-market valuation has focused on threatened and endangered species as economists have long recognized that the total economic value of the majority of these species includes both recreational use and non-use (existence and bequest) values (Richardson and Loomis 2009). One way to estimate the total economic value is the contingent valuation method (CVM) (Mitchell and Carson 1989). Applying CVM to a threatened or endangered species, researchers ask a carefully constructed sample of people questions about respondents' WTP to protect the species. Estimates of individual WTP based on the sample group are then used to extrapolate to the population as a whole and estimate the total social value of protecting the species. The surveys are often designed in a yes or no referendum format put to the respondent as a vote on a specific management plan or policy to protect a particular species or set of species. Respondents are given detailed information on the species in question, the hypothetical protection measure, the expected outcome of the measure, and how the measure is to be financed (e.g. user fees, taxes, voluntary contributions). Results of CVM surveys suggest that people are willing to pay a small portion of their income toward the protection of endangered or rare species for a variety of reasons (Richardson and Loomis 2009).

Not surprisingly, methodological factors related to survey design lead to important differences in CVM estimates of the value of wildlife protection. To investigate these effects, Richardson and Loomis (2009) performed a meta-analysis of the results from 49 studies that used CVM to estimate the value of threatened and endangered species in the United States. The authors found that several features of survey design, including type of species being valued, the change in the size of the species population, payment frequency, type of respondent, and survey mode, had an effect on respondents' WTP for species existence. For instance, WTP is greater for marine mammals, fish, and birds than for land mammals and reptiles. Charismatic species, defined as those large vertebrates that are appealing to humans and gain support for conservation campaigns, had a WTP 115-180 percent higher than non-charismatic species. Surveys based on hypothetical management plans or policies with larger changes in the size of the species population lead to greater estimates of WTP for protection, although Jacobsen et al. (2012) did not see consistent patterns of paying more for larger population sizes. Surveys using a sample frame of visitors to a particular area result in higher WTP values than surveys that sample households, due to the fact that visitors have use as well as non-use values for threatened or endangered species. Surveys based on dichotomous choice, referendum format questions result in higher estimates than open-ended questions, all else constant (see also Balistreri et al. 2001andBrown et al. 1996). Richardson and Loomis (2009) conclude that the regression function from their meta-analysis provides a rough estimate of WTP for a particular threatened or endangered species under various circumstances. As an alternative to conducting an original CVM study, using the meta-analysis regression function as a benefits transfer method can play a significant role in estimating the benefits of threatened or endangered species protection. We discuss benefits transfer methods in the next section on integrated modeling and benefit-cost analysis.

Thousands of contingent valuation studies have been conducted in over 130 countries to estimate the existence value of public goods in cultural, environmental, health, transportation, and other sectors of the economy (Carson 2012). Despite advances in, and general agreement on, best practices in CVM methodology (see Hanemann 1994), it remains controversial (Mendelsohn and Olmstead 2009), with proponents asserting that reliable estimates can be generated with the use of properly structured survey instruments (e.g., Carson et al. 2001 and Carson 2012) and critics suggesting that the method is fundamentally flawed (e.g., Hausman 2012). One of the issues is hypothetical bias, which arises when respondents answer a hypothetical question with which they have no market experience. When hypothetical questions are asked about WTP, the results tend to be upward-biased (Hausman 2012). Another issue is the scope or embeddedness effect. Results of contingent valuation surveys have shown that the assessed value of a public good can vary widely depending on whether the good is assessed on its own or embedded as part of a larger, more inclusive package. Lastly, assessed values of a public good differ depending on whether respondents are asked how much they would be willing to pay to avoid a negative outcome (or achieve a positive outcome) or how much they would be willing to accept to allow a negative outcome (or not to receive a positive outcome). Economic theory says WTP and WTA should be approximately equal but CVM results point to persistent disparities (Hausman 2012).

INTEGRATED MODELING AND BENEFIT-COST ANALYSIS

Forest management is inherently a problem of joint production: Decisions intended to produce or preserve one type of forest ecosystem good or service affect the provision of other ecosystem services. Ideally, the value of changes in other ecosystem services is not an afterthought but rather proactively considered in decision-making. To this end, managers can utilize integrated models that account for the joint production of, and relative value of changes in, multiple ecosystem services (e.g., Bowes and Krutilla 1989, Calish et al. 1978, Creedy and Wurzbacher 2001, Grasso 1998, Nelson et al. 2009). Integrated models: 1) predict changes in ecosystem structure as a result of a given management decision; 2) predict changes in the quantity or quality of ecosystem services as a result of changes in ecosystem structure; and 3) quantify the net benefits of all changes, including but not limited to changes in ecosystem services. In short, integrated models allow a decision-maker to identify the management alternative with the highest net benefit. This section discusses use of integrated modeling and benefit-cost analysis to inform forest policy and management. It introduces cases of increasing complexity to elucidate the integrated modeling approach and demonstrate the challenges and opportunities in applying it.

Static Benefits and Costs, and Selection, Among Discrete Alternatives

When land managers choose among distinct management alternatives and net benefits do not change through time, the analysis involves a straightforward comparison of annual net benefits among alternatives. For example, Polasky et al. (2011) quantify the changes in the values of annual flows of ecosystem services, including carbon storage, water quality, and agricultural and timber production, associated with actual and alternative land-use change scenarios during 1992-2001 in Minnesota. They find that including the value of carbon storage and water quality improvements dramatically changed the ranking of the alternatives.

To demonstrate the process of integrated modeling and benefit-cost analysis, we present a basic theoretical framework and simple numerical example comparing net benefits of developing or preserving a patch of mature forest, where preservation benefits include values of carbon storage and water availability for irrigators. Each management option is associated with a set of characteristics (S) that summarize the state of the system under that option, S_D for development and S_P for preservation. Ecological production functions (EPFs) translate from the state of system (and any additional management inputs) into the quantity of a given ecosystem service: $x_j = f_j(S_i)$. Denote by $g(x_j)$ the economic benefits function (EBF) that maps the quantity of ecosystem service j into a dollar value. For marginal changes in x_j, $dg/dx_j = p_j$, where p_j is the market or shadow price of x_j. If management decisions only lead to marginal changes in the ecosystem service, then the EBF can be simplified to $g(x_j) = p_j x_j$. However, this may give a poor approximation of the value of changes in ecosystem services when marginal WTP is sensitive to changes in the provision of services over the policy-relevant range of service quantities. In that case, the analyst should make use of an integrable demand curve (or demand system) in order to capture the surplus value corresponding to a given quantity of ecosystem services.

Given the relevant EPFs and EBFs, the manager can determine whether development or preservation maximizes the sum of market and non-market values:

$$\max_{\{i=D,P\}} \quad B_D 1_D + \Sigma g_j(x_j)$$
$$\text{s.t. } x_j = f_j(S_i)$$

where B_D denotes the annual benefits of development and 1_D is an indicator function for the choice to develop.

For example, assume that developers estimate the value of development at \$20,000 per year, so that B_D = \$20,000. Assuming the forest is in equilibrium, it will produce the same quantity of ecosystem services—say, carbon storage and water yield for downstream irrigators—each year. Suppose that the forest's aboveground biomass if preserved (S_P) is 14,000 tons, and if developed (S_D) would be 4,000 tons. If the carbon content of aboveground forest biomass in this region is 50 percent, then $x^c = f_c(S) = 0.5S$, and the preserved stand stores 7,000 tons of carbon while the land with residential development would store 2,000 tons of carbon. The carbon rental value is \$2.50 per ton per year. A hydrologic model predicts that

$$x^w = f_w(S) = 136 + 0.001S,$$

so that irrigators downstream will receive a summertime yield of 150 acre-feet per year with the standing forest and 140 acre-feet with the residential development. A hedonic study of farm prices in the area reveals a shadow price of \$10 per acre-foot for rights to irrigation water. Using these EPFs and EBFs, development produces an estimated annual value of \$26,400, while the current landscape produces an estimated annual value of \$19,000. Based on this benefit-cost analysis alone, the land manager interested in maximizing net benefits would opt for development.

Temporal Changes in Benefits and Costs, and Optimization, of a Continuous Choice Variable

In the example above of development versus preservation, there are several reasons why benefits are not likely to be constant over time. First, the fate of the felled timber matters. If it has commercial value, this must be added to the overall value of the development alternative, but that value will only be realized once, not annually. Whether it will be burned, chipped, pulped, or used for long-lived wood products also matters, as this determines the timing and value of carbon emissions. Second, not all of the value of stored carbon is lost at once. Society continues to receive the benefits of some carbon storage even after the stand is gone, and this too must be added to the tally of benefits under the development alternative. Another complication arises if we no longer assume the marginal social value of ecosystem services stays constant over time. For instance, rising demand for irrigation water would increase its marginal value over time. The SCC is not fixed, but rather dependent on the concentration of greenhouse gases in the atmosphere, which will change over time. Even the annual value of the development itself may change over time with changes in the real estate market.

When the costs and benefits of forest management decisions vary over time, alternatives can be compared according to their net present value. This calculation requires projections of the consequences of current action on ecosystem processes, projections of the values of ecosystem services, and a discount rate that weighs present versus future benefits. Foresters are familiar with the well-known Faustmann (1849) formula for determining the net present value of an infinite series of timber plantations starting with bare land and accounting for the value of future timber yields and planting costs (see also Samuelson 1976). The Faustmann formula has been extended to include the discounted value of a stream of amenity benefits associated with the growing plantation (Hartman 1976).

Imagine that a land manager considers a sequence of timber harvests over time as an alternative to preservation or development. Then, a formula for the net present value (NVP) of timber management that includes the marginal values of services associated with water yield and carbon storage is:

$$NPV_T = \sum_{i=1}^{\infty} \frac{\left(\dfrac{p_1 V(T_i)}{(1+r)^{T_i}} + \displaystyle\sum_{t=1}^{T_i} \frac{p_2 W(t) + p_3 C(t)}{(1+r)^t} \right)}{(1+r)^{\sum_{k=1}^{i} T_k}}$$

where r is the discount rate, T_i is the age of the stand at the time of the ith harvest, t is the index of stand age during the ith rotation, $V(T_i)$ is the merchantable volume at rotation age, $W(t)$ and $C(t)$ are water yield and carbon storage as a function of stand age, and p_1, p_2 , and p_3 are prices or marginal values of timber, water, and carbon, respectively. Note that, for each rotation, the computation includes the values of timber harvest, water yield, and carbon storage discounted to the beginning of the rotation (numerator) and then discounted to the present (denominator). Given expected prices or marginal values and of the evolution of landscape characteristics over time, the land manager can evaluate and compare the NPV of the development, preservation, and timber management alternatives.

Notable applications of the Hartman-Faustmann model include: Calish et al. (1978), who investigate optimal rotation of Douglas-fir in the Pacific Northwest when accounting for the value of services associated with the provision of wildlife, water, and aesthetic amenities; Englin and Callaway (1993, 1995), who extend the analysis to include the value of carbon storage; and Creedy and Wurzbacher (2001), who consider the joint production and valuation of carbon storage and water yield services. None of these studies incorporates spatial interdependence among stands in the provision of services, which can greatly complicate the problem.

Spatial Interdependence of Benefits and Costs

Spatial interdependence arises when the provision or value of ecosystem services from one forest stand depends on the state of other managed lands in the vicinity. For example, a riparian buffer forest will filter out more nutrient and sediments if there are clearcuts upslope compared to having intact forest upslope. Hof and Bevers (2000) address the problem of scheduling the location of timber harvests over time to minimize sediment yield in stream segments while providing an even-flow of timber yield. Their model projects how sediment levels in different streams are affected by the timing and spatial arrangement of harvests in the vicinity of those streams. Their results suggest that maintaining forest cover adjacent to streams while clumping timber harvests in upper reaches of the watershed yields less sediment than dispersing small clearcuts across the watershed while producing the same amount of timber. While their model projects the tradeoffs between timber and sediment yield over time, it could be extended to maximize the discounted value of timber harvest and sediment reduction under given prices.

Another example of spatial interdependence involves the protection of habitat for wildlife populations whose dispersal and survival depend on the size and proximity of habitat patches. Polasky et al. (2008) utilize a spatially explicit biological model of species survival rates for 267 species that accounts for habitat preferences, area requirements, and dispersal ability between habitat patches. Their economic model incorporates site characteristics and location to predict market returns for a variety of potential land uses. Their results suggest that protecting and restoring contiguous blocks of low-elevation wetlands and old-growth conifer forests produce large increases in species survival relative to a land-use pattern that maximizes market value and converts these areas to agriculture or housing development.

Uncertainty

There is often considerable uncertainty about the provision of ecosystem services under different options or about the values of nonmarketed goods and services. Nearly every aspect of an ecological or economic model, from its specification to its parameters to the data to which it is applied, involves some uncertainty; and all of these uncertainties propagate through the modeling process to the final results. However, uncertainty does not preclude, nor even necessarily complicate, integrated modeling and benefit-cost analysis. When uncertainty (say, about a parameter value) can be bounded, the choice of high-end or low-end values might have little or no impact on the ranking of management alternatives. For example, returning to the stylized case above, suppose there was uncertainty about the shadow price of an acre-foot of water and estimates range anywhere from \$0 to \$100 per acre-foot. Over this entire range, the value of benefits under development exceeds the value of benefits under preservation. At \$100 per acre-foot, the value of development is \$39,000, while the value of preservation is \$32,500.

Uncertainty really matters when it can change the ranking of alternatives. In many such cases, it is appropriate to use probability weights within an expected utility framework and select the management option with the greatest expected value. However, the irreversibility of some land-use transitions can change the calculus. If more or better information about the costs or benefits of alternative land uses is expected in the future, there is a "quasi-option" value to receiving this information--but the value can only be positive if irreversible decisions have not yet been made, otherwise the information is useless. Thus, the rational decisionmaker may wish to postpone irreversible land-use transitions though they appear to offer the greatest expected value according to current information (Arrow and Fisher 1974).

Benefits Transfer

Benefit-cost analysis requires the use of integrated ecological and economic models to predict the net social benefits of alternative management or policy options, and the decisionmaker needs to assess the availability and applicability of scientific and economic models for the problem at hand. It is best to use EPFs and EBFs that were derived for the contexts in which they will be applied. While estimates of the SCC apply everywhere and can be used "off the shelf," the highly context-dependent nature of demand for water, recreational attributes, or residential amenities means that estimates from one place cannot be readily applied to another location.

Collecting the necessary data for a new, site-specific valuation study can be very costly (though technology has significantly lowered the cost of administering surveys and opened the door to innovative new approaches to data collection through web-scraping), and decisionmakers should take a common sense approach to the question of data collection. In many cases, sensitivity analysis of benefit-cost results using a range of simulated estimates for non-market values will shed light on the necessity of obtaining real estimates from new data: If the ranking of management alternatives does not appear to change within the range reasonable estimates for the ecosystem services in question, economic valuation is unnecessary. When rankings do change, the decisionmaker should compare the expected costs of data collection to the magnitude of potential benefits from more accurate information implied by the simulations.

If valuation would improve decision-making but the costs of new data are relatively high, the best option may be benefits transfer, the application of existing valuation studies from different locations to the focal site. Benefits transfer (BT) has garnered a great deal of attention among academic economists and practitioners (see Boyle et al. 2010 and Richardson et al. 2015 for recent reviews). There is a strong consensus that practitioners of BT ought to utilize benefits functions rather than point estimates of marginal value or consumer surplus in order to transfer information from the study context to the policy context. Simply taking a point estimate of the value of a change in an ecosystem service from one study site and using it in another (i.e., value transfer) ignores important differences and can introduce substantial error into valuation and decision-making. Averaging over multiple point estimates from different study sites generally will not alleviate this problem, since the values almost certainly will not come from a representative sample.

The methods of benefits function transfer fall into two categories: reduced-form meta-analysis and structural approaches relying on a particular utility function. Meta-analysis requires the availability of multiple studies that estimate the value of a change in a given ecosystem service using a common metric (i.e., Marshallian consumer surplus or Hicksian compensating variation). The analyst uses multiple regression to estimate a model that predicts these value estimates. In many cases, the analyst will not have a sufficient number of observations (i.e., valuation studies) to estimate a model that controls for all of the factors that drive differences in value across contexts. Boyle et al. (2010) note five dimensions along which differences between study site and policy site will contribute to differences in marginal value: 1) the studied and affected populations, 2) the physical conditions of the ecosystem service, 3) the availability of substitutes, 4) the institutional settings, and 5) other context-specific characteristics. Even when data are available, the analyst should keep in mind that the estimated model will not necessarily predict a value for the policy context that is consistent with economic theory.

Structural approaches, in contrast, produce value estimates consistent with economic theory. The most straightforward structural approach is the direct transfer of an estimated demand or utility function from a study context to the policy context. This is known as preference function transfer. Given the estimated parameters of the demand or utility function from the study context, the analyst need only plug in values for the variables corresponding to the policy context. Ideally, parameters have been estimated not only for the shape of the demand or utility function, but also for "shift" variables, such as demographic and locational characteristics, the distributions of which are likely to differ between the study and policy contexts. Boyle et al. (2010) assert that stated-preference choice models, travel cost-based site choice models, and structural hedonic models that incorporate these shift variables are most amenable to preference function transfer.

When studies for more than one similar context are available, the analyst may wish to utilize preference calibration (Smith et al. 2002) instead of relying on preference transfer, which only leverages the information contained in a single study. Like reduced-form meta-analysis, preference calibration incorporates value estimates from multiple studies, but it does so in a way that imposes consistency with standard economic theory. Using this method, the analyst assumes a particular form of the utility function and calibrates the parameters of this function to best fit existing value estimates from the literature, conditional on observed site and demographic characteristics if possible. Smith et al. demonstrate the calibration method for water quality preferences utilizing hedonic, contingent valuation and travel cost studies from quite different contexts. The multi-step process requires a strong understanding of consumer theory but provides a foundation for estimating the value marginal and non-marginal changes of the attribute of interest in the policy site.

No matter the choice of benefits function transfer method, an analyst's data requirements for the policy site remain significant. Having the data necessary to account for demographic, institutional, and locational differences is critical. The increasing richness and availability of spatially explicit (i.e., GIS) data allow an unprecedented opportunity to account for such differences, but only to the extent that such characteristics have been described and incorporated in existing studies (Boyle et al. 2010, Troy and Wilson 2006).

Non-use Values and Distributional Analysis

Not everything we value as a society fits neatly into benefit-cost analysis. Some non-market benefits are difficult (and some critics say impossible) to evaluate in monetary terms. Species preservation is a particularly salient example. As an alternative to incorporating endangered species into the CBA framework via valuation, decisionmakers can acknowledge an obligation to preserve a species for its own sake and build in preservation to the integrated model as a constraint. They can then analyze the tradeoffs between management alternatives that produce different levels of social benefits (as measured by the values of ecosystem services produced) and species abundancy or survival probability. For example, Polasky et al. (2008) trace out a two-dimensional production possibility frontier to illustrate the tradeoffs between biodiversity (vertebrate species richness) and landscape value under different land-use regimes within the Willamette River Basin. Armed with such information, decisionmakers can choose the regulatory or management regime to achieve an outcome most consistent with society's commitment to other species.

Similarly, society may have a commitment to promote the well-being of particular stakeholders. In this case, benefit-cost analysis alone provides insufficient information to identify the most preferable management option because it ranks alternatives according to the Kaldor-Hicks potential compensation criterion, which maximizes net benefits irrespective of the actual distribution of those benefits among stakeholders. Distributional analysis can complement benefit-cost analysis in enabling a just, well-considered decision (Banzhaf 2011). It can help land managers avoid inequitable outcomes that would not otherwise be corrected through adjustments in tax or welfare policies or simply "net out" in the course of many different changes to forest management. The essence of distributional analysis is the disaggregation of net benefits by group. Whether groups are based race, income, or service beneficiary type, much of the information necessary for distributional analysis should already be on hand, as each of these variables plays an important role in recreation- and amenity-related ecosystem service valuation.

CONCLUSIONS

Forests provide an array of benefits beyond marketable commodities. The ecosystem service metaphor has brought greater attention to these benefits and encouraged a more nuanced understanding of the relationship between various forest attributes and their beneficiaries. Forest management and policymaking involves difficult decisions, with multiple and sometimes conflicting objectives and concerns about the distribution of benefits across time, space, and stakeholders. Integrating ecological and economic models to predict the benefits and costs of alternative management or policy options can assist decisionmakers in navigating these tradeoffs and make environmental decisionmaking more transparent, rational, and responsive to people's preferences. Failing to properly quantify non-market benefits for decision-making may lead to substantial tradeoffs for society—whether in the form of over-exploitation of commercial resources or overly cautious and costly restrictions on the use of forests. Despite this, economic valuation of forest ecosystem services and comprehensive benefit-cost analysis remain under-utilized in forest management. Though not without their limitations, these methods can provide decisionmakers with important information regarding the costs and benefits of management actions.

How should the Forest Service proceed given the current state of the literature on the assessment and economic valuation of ecosystem services? We have four specific suggestions:

1. Estimate the economic benefits of a given forest and associated management policy, using available methods for services related to timber, carbon, water, amenities, recreation, and wildlife. This practice is important to identifying and describing the range of benefits provided by the forest. It also provides a baseline for evaluating changes in management.
2. Estimate the change in economic benefits associated with a change in management, regulations, or incentives, or a natural disturbance. This practice is important to evaluating and prioritizing different policies, evaluating potential tradeoffs in management decisions, and assessing the damages caused by natural disturbances.
3. Enhance communication with stakeholders about the economic benefits and costs of potential changes in forest management. This practice is important because communities' preferences for different ecosystem services may be affected by estimates of economic performance.
4. Monitor the performance of agency programs. This practice is important to tracking whether the actual economic benefits and costs of agency programs are consistent with projections.

It is probably premature for the Forest Service to integrate these suggestions into every forest management plan. The literature simply does not have enough evidence to show how to do it properly for the entire forest system. A more attainable goal would be to focus on a few forests in different regions of the country and develop and evaluate alternative management plans for those forests. What are the insights of ecosystem service valuation for actively managing timber, carbon, water, amenities, recreation, and wildlife? What is the response of stakeholders to economic information about a wide range of ecosystem services? Once some concrete experience is gained, then it would be time to consider extending the plans to the rest of the system. Finally, there is a continuing need to improve EPFs and economic valuation functions of all ecosystem services as well as methods of BT.

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APPENDIX
Recreation Valuation Methods

Economists have utilized three primary revealed preference approaches to estimate the monetary value of changes in site attributes related recreation services: the discrete choice random utility model (Hanemann 1984), the hedonic travel cost method (Brown Jr and Mendelsohn 1984), and the generalized corner solution or Kuhn-Tucker model (Phaneuf et al. 1998). A brief description of each follows; readers are referred to Haab & McConnell (2002), Phaneuf and Smith (2005), Freeman et al. (2014), and Bockstael and McConnell (2007) for more comprehensive treatment of the theory and econometric methods related to nonmarket valuation.

Discrete Choice Random Utility Model

The discrete choice random utility model (RUM) applies directly to recreationists' decisions over which site to visit on a given occasion. (It is often paired with a count data model to estimate the demand for total number of trips.) Along with the implicit and explicit costs of travel to the various sites and a host of socio-economic characteristics, forest attributes thought to influence people's preferences among sites are included as explanatory factors in the site choice decision model. These explanatory factors are modeled as arguments in a utility function, which individuals are presumed to maximize in making their observed site choice decisions. In the RUM framework, the analyst specifies a utility function applicable to all individuals in the sample but recognizes that unobservable factors may differentially influence individuals' preferences among sites; the analyst models this idiosyncrasy as individual-specific random error (hence the name, random utility model). Having estimated the parameters of the choice model, the analyst can calculate the average marginal value of each forest attribute based on its contribution to utility relative to that of travel costs. In addition, because the utility function is estimated directly, the RUM allows for relatively easy, exact calculation of the welfare effects of non-marginal changes in forest attributes. It is applicable to real-world observations (i.e., as a revealed preference method) or observations of hypothetical decision-making (i.e., as a stated preference method). Recent applications of discrete choice random utility models to value forest attributes for recreation include Juutinen et al. (2014), Rolfe and Windle (2015), and Simões et al. (2013).

A key limitation of the RUM is that it does not account for individuals' preferences for variety in recreational site visits: observed choices are assumed to be independent of past or future trips. In addition, most RUM studies specify linear utility functions, despite the fact that this imposes strong assumptions on recreationists' preferences (Hanemann 1984, Mendelsohn 1987). Most notably, linear models assume that recreationists place the same marginal value on forest attributes, regardless of the current level of those attributes. Such strong assumptions are particularly unlikely to hold for attributes such as average stocking density, for which increases may be "goods" at relatively low levels and "bads" at relatively high levels; and for studies spanning different recreational "markets," in which visitors' place of origin (ZIP code area, census block, town, etc.), choice set, and implicit prices differ (Pendleton 1999). Future studies can and should explore more flexible utility specifications, as poor specifications can substantially bias estimates of ecosystem service values (Cropper et al. 1993, Pendleton et al. 1998).

Hedonic Travel Cost Method

As its name suggests, the hedonic travel cost method (HTCM) takes a hedonic approach to the question of recreation-related ecosystem service value by decomposing travel costs from each origin into the implicit prices of the characteristics of each chosen site. A different hedonic price equation is estimated for each origin or "market." If the assumptions of the hedonic model are satisfied, these implicit prices reflect the value of a marginal change in each attribute across all sites to recreationists in each market. Recovering the welfare effects of discrete changes in a site attribute, multiple attributes, or an attribute only at a particular site requires derivation of an inverse demand system based on observed choices and estimated hedonic prices. The demand equations are then used to predict changes in site choice based on expected surplus. Although this requires quite a bit of additional analyses, estimation of the demand system can reveal important substitutability or complementarity between site attributes and allows for more accurate welfare calculation than the typical linear RUM to the extent that such non-zero cross-price coefficients exist.

The HTCM faces many limitations and criticisms. As with the RUM, it cannot explain or predict the choice of different sites in the same year or season by the same individual, and demand analysis is typically conditional on recreationists' choice to make a trip. Unlike the RUM, it does not account for preference heterogeneity among demographically identical individuals. Moreover, it is not well suited for valuation when choice sets are very small and the assumption of a continuous price frontier is implausible. Although the HTCM has been criticized for producing negative implicit price estimates for some ecosystem services, this may be a feature rather than a flaw. While "too much of a good thing" rarely arises in markets, where producers have no incentive to provide costly attributes beyond the level at which consumers are willing to pay the marginal cost, recreationists may well be over-satiated by the natural supply of a particular site attribute. (Even when managers can alter the level of service provision, it may not be efficient to reduce natural supply to the level at which its marginal value would be non-negative.) However, the appeal to the natural supply of ecosystem services in justifying negative price estimates belies a more fundamental issue with the HTCM: it may not be appropriate to interpret the estimated cost function equation as a hedonic price equation at all (Smith and Kaoru 1987).

Generalized Corner Solution

The generalized corner solution or Kuhn-Tucker (KT) model is a relatively new and promising valuation method. Unlike the discrete choice RUM and HTCM, the KT model directly accounts for some recreationists' choice to take multiple trips—possibly to different sites--in a given season. It also avoids potential bias that arises in the context of estimating preferences from single choice occasions when the process is a dynamic one (Baerenklau and Provencher 2005), though estimates may still be biased if such dynamics operate over longer time scales. At the same time, the KT model accounts in a statistically and theoretically rigorous way for the fact that the number of trips a given recreationist will make to most sites is zero. These zero values, or "corner solutions," beget the name; the KT appellation refers to the (first-order) optimality conditions, which apply to both the extensive margin (site choice) and intensive margin (number of trips) and provide analytical rigor (von Haefen and Phaneuf 2008). The KT model incorporates random utility parameters to account for individual preference heterogeneity, and parameters of the Kuhn-Tucker conditions are estimated using maximum likelihood. Estimation (and subsequent welfare analysis) is computationally intensive, but possible with the use of simulation techniques (von Haefen and Phaneuf 2008). The approach has been used to estimate preferences for freshwater and coastal/marine recreation (e.g., Phaneuf et al. 2000, Kuriyama et al. 2010), as well as moose hunting trips (von Haefen and Phaneuf 2005).

Challenges

Three important methodological or informational issues confront all of the major approaches to revealed preference valuation of recreation-related forest ecosystem services. The first complication relates to the identification or selection of the set of choices that recreationists actually consider in making their decisions. This information is not generally available to the analyst relying on visitor permit data or even brief on-site surveys. Including sites and travel routes that were not part of the true choice set can substantially bias the estimates of the relative value of different site characteristics (Bell and Strand 2003, Haab and Hicks 1999). Absent information on recreationists' actual choice sets, Pendleton (1999) argues that the choice set should be restricted to sites visited by at least one person from each origin in the HTCM. It is unclear whether defining origins for this sole purpose in the context of other modeling approaches is most sensible, or whether the analyst should consider other characteristics, such as group or household size, demographic variables, and income in addition to or instead of grouping observations by proximity for the definition of choice sets.

Another challenge facing each of these valuation methods is quantification of the opportunity cost of travel time. For hourly workers free to choose as few or as many hours of work as they wish, the marginal value of their time is exactly equal to their wage rate. But recreationists rarely face labor-leisure tradeoffs of this sort. Nearly all are retired, salaried, or wage earners with limited discretion over their hours, making the identification of the marginal value of their time a much more complicated proposition (see Larson and Shaikh 2001). With few exceptions, however, analysts continue to utilize the wage rate or a fraction thereof—often one-third, as a rule of thumb. Englin and Shonkwiler (1995) treat unobserved travel costs as a latent variable, using distance, travel time, party size, and composition as indicators of this latent variable. Surprisingly, the authors find that respondents value the opportunity cost of time at 40 percent of wage rate, offering some validity to the standard, largely arbitrary assumption. Further application of this latent approach seems warranted. At the same time, other research efforts are underway to estimate the marginal value of time without appealing to the labor-leisure decision (e.g., Fezzi et al. 2014, Phaneuf 2011); these too seem promising.

A third challenge facing revealed preference valuation practitioners concerns the range of observation in quantity or quality of the ecosystem service of interest. Changes in environmental quality outside of the range currently observed cannot be evaluated reliably. For instance, if a policymaker wants to weigh the costs and benefits of opening some recreational area to harvest, but none of the sites within the choice set of local recreationists contain harvested stands, there is no way to evaluate the impact of the proposed change based on recreationists' observed behavior. Welfare calculation requires either carefully applied benefits transfer from sites where such changes fall within the range of observation or supplementing revealed preference data with stated preference data (e.g., Englin and Cameron 1996, Englin et al. 2001, von Haefen and Phaneuf 2008, Whitehead et al. 2000). This allows wider application of the revealed preference approach, subject to validation of the contingent response data.

How the results of the valuation exercise are incorporated into the benefit-cost analysis depends on the valuation technique. The goal is to obtain a measure of equivalent or compensating variation associated with the expected changes in ecosystem attributes. Equivalent variation (EV) measures how much a consumer would be willing to pay or accept for the changes to occur; compensating variation (CV) measures how much money a consumer would have to lose or receive to be just as well off as she was before the change in attributes. In the context of recreation-related services, these measures should be nearly identical. Calculation of either measure requires the use of a utility function with the relevant ecosystem attributes among the arguments. The utility function may be specified directly, as in the case of the discrete choice random utility and KT models, or it may be "recovered" from the estimated parameters of the demand system, as in the case of the hedonic travel cost model. Applications of the discrete choice RUM typically assume a convenient distribution for the random term and a utility function that is linear in ecosystem attributes, which allows a closed-form solution for the CV or EV equations. In contrast, the KT model is inherently non-linear and requires computational integration via simulation to obtain numerical solutions (see Phaneuf et al. 2000 and von Haefen and Phaneuf 2005).

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This review focuses on the assessment and economic valuation of ecosystem services from forest ecosystems—that is, our ability to predict changes in the quantity and value of ecosystem services as a result of specific forest management decisions. It is aimed at forest economists and managers and intended to provide a useful reference to those interested in developing the practice of integrated forest modeling and valuation. We review examples of ecosystem services associated with several broad classes of potentially competing forest uses—production of timber, sequestration of carbon, regulation of the quality and quantity of water, provision of residential and recreational amenities, and protection of endangered species. For each example considered, we briefly describe what is known about ecological production functions and economic benefits functions. We also highlight the challenges and best practices in the creation and use of this knowledge. In the final section, we discuss the process, strengths, pitfalls, and limitations of utilizing integrated models for benefit-cost analysis of proposed forest management activities.

KEY WORDS: Ecological production function, non-market valuation, social cost of carbon, recreation demand model, hedonic travel cost model, discrete choice model, contingent valuation model, hedonic property price model

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