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Volume 1



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Advances in Threat Assessment and Their Application to Forest and Rangeland Management

Volume 1

**John M. Pye, H. Michael Rauscher, Yasmeen Sands,
Danny C. Lee, and Jerome S. Beatty, Technical Editors**

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Abstract

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In July 2006, more than 170 researchers and managers from the United States, Canada, and Mexico convened in Boulder, Colorado, to discuss the state of the science in environmental threat assessment. This two-volume general technical report compiles peer-reviewed papers that were among those presented during the 3-day conference. Papers are organized by four broad topical sections—Land, Air and Water, Fire, and Pests/Biota—and are divided into syntheses and case studies. Land topics include discussions of forest land conversion and soil quality as well as investigations of species' responses to climate change. Air and water topics include discussions of forest vulnerability to severe weather and storm damage modeling. Fire topics include discussions of wildland arson and wildfire risk management as well as how people perceive wildfire risk and uncertainty. Pests/biota topics include discussions of risk mapping and probabilistic risk assessments as well as investigations of individual threats, including the southern pine beetle and *Phytophthora alni*. Ultimately, this publication will foster exchange and collaboration between those who develop knowledge and tools for threat assessment and those who are responsible for managing forests and rangelands.

Keywords: Environmental threats, threat assessment, environmental risk analysis, disturbance, wildfire, pests, forest and rangeland management.

Preface

Danny C. Lee¹ and Jerome S. Beatty²

In July 2006, more than 170 researchers and managers from the United States, Canada, and Mexico convened in Boulder, Colorado, to discuss the state of the science in environmental threat assessment. The 3-day conference explored the latest information on environmental threats, bringing together people who develop knowledge and tools for threat assessment and management and those responsible for managing forests and rangelands. The event included more than 100 oral and poster presentations on topics ranging from severe weather and climate change to risk mapping and forest pests.

The year preceding the conference, 2005, was one of extremes and provided an ideal backdrop for a discussion of environmental threats. More hurricanes were tracked that year than had ever before been reported, including Katrina, one of the costliest U.S. hurricanes on record and one of the deadliest. Also in 2005, entomologists confirmed the capture of a female sirex woodwasp in a sample collected in New York in late 2004. It was the second of its kind identified in the United States and has pine managers across the country understandably concerned given the species' ability to cause severe tree mortality. The wildfire season that year also made history, claiming more than 8 million acres and breaking the record that had been set in 2000 for total acreage burned.

Given this context and, more generally, the complexity of assessing and managing the myriad threats that face North America's wildlands, the conference's organizers were committed to continuing the exchange of information and collaboration fostered during the event long after its final session. To that end, nearly 50 of the synthesis and case study papers presented were adapted to form the initial content of the Encyclopedia of Forest Environmental Threats (<http://www.threats.forestencyclopedia.net>)—an online resource that promises to deliver to researchers, land managers, and policymakers the scientific knowledge about environmental threats they need to achieve their objectives. The threats encyclopedia—like the others in the Forest Encyclopedia Network, of which it is part—connects scientific results, conclusions, and impacts with management needs and issues. Designed for scientists and practitioners alike, the encyclopedia serves as a growing online compilation of scientific knowledge relating to environmental threats and their assessment and management.

The conference's content is also preserved here, in a traditional and more permanent form. The peer-reviewed papers featured in this volume represent the scope of environmental threats and underscore the complexity of their assessment and management. As these papers show, environmental threats often act in concert and with no regard for land ownership and administrative boundaries, making them as difficult to identify and anticipate as they are to manage and control. In response, researchers and managers are developing a growing foundation of knowledge, which can help to assess or minimize these threats. This volume represents a significant contribution to this effort.

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The “Advances in Threat Assessment and Their Application to Forest and Rangeland Management” conference and this publication were made possible by the contributions of many people and organizations. Participating organizations include the USDA Forest Service’s Pacific Northwest (PNW) and Southern Research Stations (SRS); the Western Wildland Environmental Threat Assessment Center (WWETAC); the Eastern Forest Environmental Threat Assessment Center (EFETAC); the Cooperative State Research, Education, and Extension Service; Southern Regional Extension Forestry; and the Southern Forest Research Partnership. The conference’s organizing committee members included John M. Pye, Jerome S. Beatty, Danny C. Lee, H. Michael Rauscher, Yasmeen Sands, Gregg DeNitto, Charles G. Shaw, James P. Shephard, and David A. Weinstein. Special thanks to the moderators who oversaw the extensive peer-review process—Gregg DeNitto, William Bechtold, Charles G. Shaw, Borys Tkacz, William D. Smith, Kurt Riitters, David Weinstein, Becky Kerns, Alan Ager, Charley Luce, and Kerry Overton—and to the many peer reviewers who helped to ensure the quality of the science. This publication would not have been possible without the editing expertise of Alan Salmon, Gary Benson, Aimee Tomcho, Charmaine Rini, and Sharon DeLaneuville. Matt Howell and his University of Georgia team provided excellent Web development and hosting for the Web version of this publication’s content, available online at <http://www.threats.foresten-cyclopedia.net>. Funding for the conference, this publication, and its Web counterpart was provided by WWETAC, EFETAC, PNW, and SRS.

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Conversions of Forest Land: Trends, Determinants, Projections, and Policy Considerations

Ralph Alig, Susan Stewart, David Wear, Susan Stein, and David Nowak

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Abstract

Forest land conversion leads to ecological effects (e.g., changes in water quality and wildlife habitat) and socioeconomic effects (e.g., expanding urban-forest interface, reduced long-term timber production possibilities and loss of open space). Socioeconomic drivers of land use change such as population totals and personal income levels have increased substantially since World War II. Human land use is the primary force driving changes in forest ecosystem attributes. Land use changes affecting forests since 1990 have been heavily concentrated in the South. Nationwide, more than 60 percent of housing units built in the 1990s were constructed in or near wildland vegetation. More than 44 million acres of private forest are projected to experience housing density increases between 2000 and 2030, with the majority of the most heavily impacted watersheds in the East. The United States population is projected to grow by more than 120 million people by 2050, and deforestation associated with this growth is projected to exceed 50 million acres. Fragmentation of remaining forests is also projected and expected to be concentrated in distinct subregions; in the South, these include urbanizing areas and areas close to interstate highway corridors. As urban lands expand into surrounding areas, retaining trees can have significant benefits. Current benefits of urban vegetation on environmental quality nationally are on the order of several billion dollars per year.

Keywords: Deforestation, development, fragmentation, land-use change, population growth.

Forest Land Conversion and Recent Trends

Forests cover about one-third of the United States and range from wildland forests to urban forests. These diverse ecosystems provide a variety of habitats for wildlife; help to cleanse the air and water; supply timber, fuelwood, and other harvested products; serve as places for recreation; help to mitigate the effects of global climate change; and provide other essential goods and environmental services.

Forests are vulnerable to conversion to other land uses. An increasing number of houses and other buildings in and near forests portend growing costs and complications in fire suppression and potential loss of many values derived from forests. Long-term assessment of the condition of forests and of the relationships between forest conditions and socioeconomic factors is the key to defining policy questions and actions needed to sustain forest-based services.

In this synthesis, we survey recent trends, determinants, and projections of forest land conversion in the United States. Examples with more detailed treatments, supporting tables, and figures are available in Alig and others (2003, 2004, 2010) and Alig and Ahearn (2006). Forest land conversion is a persistent issue for managers and policy-makers; for example, a recent position statement concerning loss of forest land by the Society of American Foresters (2004) lists ecological effects (e.g., effects on water quality and wildlife habitat) and socioeconomic effects (e.g., expansion of the urban-forest interface, reduction of forest recreation opportunities, reduction of long-term timber production possibilities, and loss of open space) as important implications of forest loss. We examine how socioeconomic drivers of land-use change, such as population totals and personal income levels, have increased substantially since the Second World War and led to changes in forest ecosystem attributes. We summarize determinants of land-use changes, focusing on the societal and private tradeoffs of retaining land in forests. Our projections reflect population

Table 1—Databases used in studies of different types of land base changes

Land base change	Coverage	Databases
Deforestation	48 contiguous States; 1982 to 1997, with national updates for 2001 and 2003	USDA NRCS 2001
Afforestation	Annual tree planting by State, 1980 to 1998; periodic estimates of reversions to forests	USDA Forest Service Tree planting reports; periodic and annual FIA surveys
Forest fragmentation	1992, with another in progress	National land cover database
Forest parcelization	Periodic owner surveys	National forest landowner survey
Structure additions to forests	Decadal national census; special studies	USDC Census Bureau; Oregon structure counts (contact: Dept. of Forestry)
Urban forest changes	National	USDC Census Bureau

growth that spurs demand for land for developed uses at the same time that demands for some forest products and other forest benefits are increasing. Risk and policy considerations necessitate that creating effective policy in this area will require careful deliberation concerning private and social viewpoints. For example, some forest benefits (such as wildlife habitat and other ecosystem services) can most effectively be produced at scales greater than the individual private parcel scale and because market imperfections can cause some social forest benefits to be undersupplied when this is the case (Kline and others 2004a).

Five categories of significant changes affecting forest area are:

- Afforestation
- Deforestation
- Forest fragmentation
- Forest parcelization
- Increased numbers of structures, such as houses, on forest land

This analysis does not address changes in forest cover type. For an example of a national analysis on this topic, see Alig and Butler (2004). Examination of intensification of land management is illustrated by the 2001 RPA Timber Assessment (Haynes 2003).

Table 1 lists supporting major databases, and Table 2 lists examples of studies that have examined land base dynamics in the United States. In the United States, millions of acres of land shift uses each year (USDA NRCS

2001) reflecting billions of choices made by individuals, corporations, nongovernmental organizations, and governments. Next, we look at recent trends in those five categories of changes in the land base. Examining historical trends provides guidance for identifying key factors that are likely to influence forest land condition and associated natural resources in future years. Discussion of historical trends is a foundation for considering projected changes.

Forest Area Changes in Total

From 1953 to 1997, a majority (26) of States had a loss in forest area according to periodic surveys by the USDA Forest Service, e.g., Smith and others (2004). Nine States had net losses of at least 1 million acres each, ranging up to 6.3 million acres. In descending order of net loss amount, the States are Texas, Florida, California, Oklahoma, Louisiana, Washington, Alaska, Missouri, and Minnesota. Seven States had net gains of more than 1 million acres, ranging up to 4.1 million acres. In descending order of net gain amount, the States are New York, Ohio, Pennsylvania, West Virginia, Mississippi, Alabama, and Kentucky. The only regions with net gain in forest area were the North, where a relatively large amount of pastureland reverted naturally to forest, and the Intermountain Region, where a large number of acres were reclassified from pasture or rangeland to forest over time.

Note that much of the shift from pastureland or rangeland to forest use is due to reclassification over time.

Table 2—Example studies of determinants of land base changes involving forest land

Land base change	Data	Studies	Bases
Deforestation	Econometric	Alig and Healy 1987, Alig and others 2004, Kline and Alig 2001, Hardie and others 2000	USDA NRCS 2001; FIA surveys
Afforestation	Econometric	Lee and others 1992, Kline and others 2002	USDA Forest Service tree planting reports
Forest fragmentation	Econometric and statistical	Butler and others 2004, Wear and others 2004, Alig and others 2005	National land cover database
Forest parcelization	Statistical	Butler and Leatherberry 2004	National forest landowner survey
Structure additions to forests	Statistical	Hammer and others 2004, Radeloff and others 2005, Stewart and others 2003	USDC Census Bureau changes
Urban forest changes	Statistical	Nowak and Walton 2005	USDC Census Bureau

As trees grow, they expand to reach the 10-percent canopy cover used to define forest land, which changes the pasture-land classification to forest use. Even though now classified as forest, the land may still be used for grazing. Shifts between grazing land and forest uses are common, and although they are technically considered as shifts into and out of agriculture, they really represent multiple, overlapping uses.

Deforestation

The long-term loss in United States forest area since the early 1950s has been due to a combination of factors, but, in more recent decades, has been primarily due to conversion to urban and developed uses. Deforestation is conversion from forest to nonforest use, and between 1982 and 1997, 23 million acres were deforested on non-Federal land in the United States. Here we concentrate on private lands and secondarily on other non-Federal lands, for which more data are available (e.g., USDA NRCS 2001). The destination of about half of the converted forest acres was to urban and developed uses (Figure 1). Between 1982 and 1997, more than 10 million acres of non-Federal forests were converted to developed uses, an area larger than the combined current forest area of five Northeastern States (Connecticut, Delaware, Maryland, New Jersey, and Rhode Island). In the most recent data remeasurement period (1992–97), the proportion

of deforestation due to urban and developed increased to 55 percent (USDA NRCS 2001).

Net changes (area into forest minus area out of forest) are typically much smaller than total or gross changes (area into forest plus area out of forest). Gross change in area of non-Federal forests in the contiguous 48 States between 1982 and 1997 totaled about 50 million acres (USDA NRCS 2001). The gross change in forest area was 14 times as large as the net change in forest area.

Conversion to Developed Uses—

Development spans a broad range of population density associated with settlement patterns, and definitions of development can depend on the data source and the purpose for which the data are analyzed. Two major data sources both show a steady increase in developed uses over recent decades. Estimates from the U.S. Census Bureau extend furthest back in time and show a 130-percent increase in census-defined urban area between 1960 and 2000. Census urban area comprises all territory units in urbanized areas and in places of more than 2,500 persons outside of urbanized areas. The census measure of urbanization labels as “built-up” some land that is still to some extent available for rural productive uses, thereby probably erring on the side of overgenerous inclusion (Alig and Healy 1987). Although the term “paved over” has frequently been used to describe

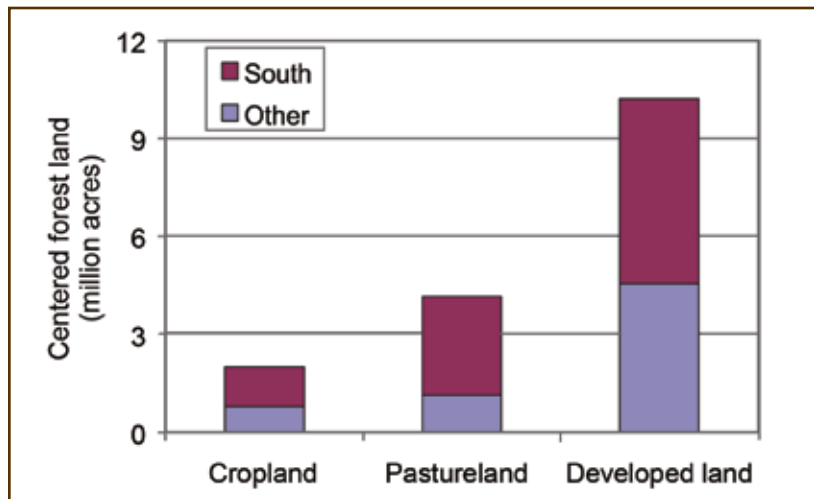


Figure 1—Conversion of nonfederal forest land by destination, and South vs. non-South, 1982-97 (USDA NRCS 2001) (note does not include rangeland and other miscellaneous uses).

urban land, only a small fraction of the land so classified is literally paved.

The other major data source is the National Resource Inventory (NRI) (USDA NRCS 2001), and it covers a shorter period (1982-97). The NRI estimate of U.S. developed area increased 34 percent between 1982 and 1997, with an acceleration in the 1990s that was more than 50 percent higher than that of the previous 5 years of measurement. Between 1982 and 1997, developed area as a percentage of the total land area in the 48 contiguous States increased from 3.9 percent to 5.2 percent. Forests were the largest individual source of developed land.

One important feature of the NRI data classification in contrast to the census urban data is the attempt to exclude areas devoted to agricultural crops, forestry, or similar purposes when they are within a parcel or contiguous area that is otherwise built-up. Outside urban areas, the NRI also includes developed land occupied by nonfarm rural built-up uses (e.g., rural transportation land), which are not included in the census urban category. Including transportation infrastructure can be important in that new roads open land to development, alter the environment (e.g., facilitate invasion of certain species), can create congestion, and can degrade the quality of life. Changes in rural land use have historically been and remain connected with changes in motor vehicle use, technology, and policy.

A significant amount of low-density development has been part of the expansion in developed area. Rural America is home to a fifth of the Nation's people, and rural residential lots tend to be larger than housing lots in urban areas. One factor in the relatively greater increase in rural residential land use is that it is generally land extensive compared with the land-intensive residential use in urban areas. Rural residential lots, although fewer in number than urban lots, tend to be larger, averaging nearly 3 acres per household, compared with less than a half-acre per household in urban residential areas (USDA ERS 2006). Forty-four million acres, 60 percent of all rural residential lands, are in the largest lot-size category, over 10 acres. Rural land in this category is 3 1/2 times the area of urban land in this category. The wide acreage disparity between rural and urban large-lot categories is likely attributable to relative land values—lower land prices in rural areas make large lots more affordable (USDA ERS 2006).

The low-density housing development in rural areas means more people living closer to remaining forest land. A measure added in recent periodic Forest Inventory and Analysis (FIA) surveys conducted by the USDA Forest Service has been the identification of forest lands by rural-urban continuum class. Based on nationwide rural-urban continuum classes (Smith and others 2004), 13 percent of United States forest land now is located in major

metropolitan counties, and 17 percent in intermediate and small metropolitan counties and large towns, together making up 30 percent of all U.S. forest land (Smith and others 2004, p. 47). Between 1997 and 2002, the forest area in major metropolitan areas increased by 5 percent, or more than 5 million acres, as the United States developed area expanded considerably. Consider that for the whole United States, more than one-quarter of counties are currently classified as metropolitan. That compares with less than one-tenth 50 years ago.

Amount of urban land per additional person is higher for non-metropolitan counties. Many Americans prefer to live in less-congested areas and will commute additional minutes or hours to realize their goals, taking advantage of the United States' excellent road system. Moreover, an increasing population of retirees has augmented out-migration from central cities and suburbs to rural areas that offer aesthetic amenities. Natural amenities may be a more important determinant of county-level immigration than nearness to metropolitan centers or type of local economy (McGranahan 1999).

South

The largest increases in U.S. developed area between 1982 and 1997 were in the South, a key timber supply region (USDA NRCS 2001). There, the amount of land in urban and other developed uses increased more than 50 percent since the 1960s. The South had one-third of its developed area added during those 15 years, equal to about half of the U.S. total of developed area added during that period. Between 1982 and 1997, the South had 7 of the 10 States with the largest average annual additions of developed area according to the NRI. The top three—Texas, Florida, and North Carolina—each added more developed area than did the country's most populous State, California. Over a more recent period, 1992–97, 6 of the 10 States that lost the most cropland, forests, and other open spaces to urban development were in the South. These six southern States in descending order of amount converted were Texas, Georgia, Florida, North Carolina, Tennessee, and South Carolina (USDA NRCS 2001).

In the Southeast, the concentration of development has been in the area of the urban Piedmont Crescent, extending from Richmond to Atlanta. Within this area are the Interstate 85 and Interstate 40 corridors, the backbone of job growth in the Southeast. Many of the smaller cities are adjacent to larger urban areas, resulting in population concentrations in larger metropolitan areas. The urban areas of the Piedmont are likewise expected to witness the fastest growth, whereas the mountains and the Coastal Plain will experience most of their growth in nonmetropolitan areas.

Several factors contribute to expansion of developed area in the South:

1. Above average county population growth due in part to climatic factors and attractiveness to immigrants (Glaeser and Shapiro 2001).
2. Above average marginal consumption of land per additional resident.
3. Income growth.

The Southern Forest Resource Assessment (Wear and Greis 2002) identified urbanization as one of the primary threats to forests in the region.

North

Areas of urban and developed uses steadily increased in the North since 1982. Between 1992 and 1997, the area of urban and developed area in the Northeast increased from 10.4 to 11.9 percent of the land base. Corresponding increases in the North Central subregion were from 6.7 to 7.3 percent.

The North had about one-third of the total addition to U.S. developed area between 1982 and 1997. The North had 3 of the 10 States with the largest average annual additions of developed area according to the NRI.

West

The West—Great Plains, Southwest, California, and Pacific Northwest—accounted for less than one-fifth of the total national addition to NRI developed area between 1982 and 1997. However, recent growth in the region has been above the national average. A growing number of “ranchettes” and large-lot subdivisions characterize housing growth in the Rocky Mountain region, resulting in the highest amount of developed area per additional person between 1992 and 1997 (Alig and others 2004, USDA NRCS 2001).

The largest percentage of change for a major land use in the contiguous three Pacific Coast States (California, Oregon, and Washington) was the 262-percent increase in urban area between 1960 and 1997 (Alig and others 2003). Urban area as a percentage of total land varies notably by State: 5.9 percent for California, 3.2 percent for Washington, and 1.0 percent for Oregon (Vesterby and Krupa 2001). The State of Washington illustrates the importance of migration for regional population growth and the concentration of growth in coastal areas (Alig and White 2007). Between 1990 and 2000, net migration to western Washington was 180 percent of the national increase (births minus deaths). Approximately 3.5 million people (59 percent of Washington State residents) live within 10 miles of coastline (including the Pacific Ocean and sounds).

Conversion to Agriculture

More than 8 million acres of forest land were converted to agricultural uses between 1982 and 1997 (USDA NRCS 2001). About half of the converted land has gone to pasture use, with the remainder fairly evenly split between crop use and rangeland. Forest land contributed 55 percent of the land that shifted into agriculture from 1982 to 1997, as land continued to be converted from less intensive uses, like forest, to agricultural uses, like cropland and pasture.

As with the conversions of forest land to developed uses, the majority of forest to agricultural conversions was in the Eastern United States and concentrated in the South. The South had the majority of forest land involved in either conversion to agriculture or gained from agriculture. In the South, land is often suitable for multiple land uses, given relatively gentle topography and ease of access.

On net, forestry gained 14.4 million acres from agriculture between 1982 and 1997. Of total land shifting out of agriculture, 22.7 million acres (46 percent) shifted into forest use, with about 17 million acres being former pastureland. Much of the shift from pastureland to forest use is due to reclassification over time. Factors associated with afforestation can differ by region and over time (in the North, for example, some land formerly used for dairy operations has reverted to forest cover). Most of the afforestation across the Nation—forestation either by human

or natural forces of non-forest land—has been of a passive nature, e.g., reclassification as forest cover increased primarily through natural succession. However, tree planting has played a role especially in the South where 25 million acres of pine plantations have been established since 1952, (Alig and Butler 2004) mostly on land formerly in pasture and range use.

Forest Fragmentation

Land use change can lead to forest fragmentation—the transformation of a contiguous patch of forest into disjunct patches. Forest fragmentation is widely considered to be a primary threat to terrestrial biodiversity (Armsworth and others 2004), and recent analysis of the fragmentation of continental U.S. forests indicates that it is so pervasive that edge effects potentially influence ecological processes on a majority of forested lands (Riitters and others 2002).

Definitions of forest fragmentation differ and are influenced by the questions or policy issues of interest. One major distinction is between treating fragmentation as a process and treating it as a pattern (e.g., Alig and others 2000). Here we discuss forest fragmentation as a pattern. Fragmented forests may occur naturally across the landscape (as in the Great Basin, NV), or this pattern may be a result of human activities, resulting in edge, core or interior habitat, and interspersed changes (Butler and others 2004). Forest fragmentation can be quantified spatially using various indices of landscape structure, with different metrics for different scales of analysis and measurements of interest. Although many fragmentation statistics are available, none provide a definitive indicator of landscape fragmentation, only a means for comparing the characteristics and relative degree of fragmentation across landscapes or periods. Numerous biophysical studies have provided snapshots of forest fragmentation, primarily for the East (Table 1).

Forest Parcelization

Forest parcelization is the subdivision of forest tracts into smaller ownerships. This phenomenon can have profound impacts on the economics of forestry and lead to reduced forest management, even when land is not physically altered. Land ownership can influence forest land

management and investment practices. In addition, per unit costs of forest management practices will increase if economies of scale are lost.

Many of the forest-related increases in population density have been on nonindustrial private forest (NIPF) lands, the ownership class most subject historically to land use changes (e.g., Smith and others 2004). Because NIPF owners are aging and have descendants who live farther from the forest and for whom timber management is not a primary objective, dealing with real estate appreciation may be more central to family succession planning now than it was in the past. Critical wildlife habitat is often provided by NIPF ownership, as in the Pacific Northwest, where lowlands and riparian areas critical to threatened and endangered species are primarily in NIPF ownership (Bettinger and Alig 1996). Family forests are a large component of the NIPF ownership class; the number of family forest owners increased from 9.3 million in 1993 to 10.3 million in 2003, and these owners now control 42 percent of the Nation's forest land (Butler and Leatherberry 2004).

Recent shifts in the ownership of the most intensively managed forests in the United States could lead to a substantial increase in parcelization. In the sales of large forest properties, there often is a spinoff for real estate development purposes (highest and best use), and overall, the amount of large industrial forest ownership has been reduced materially in a relatively short time. A large share of the forests long held by consolidated forest products companies has recently been sold to institutional investors. Many of these transactions have occurred in the South. Institutional investors currently hold about 8 percent of the investable U.S. timberland (Wilent 2004). By the end of 2003, the top 10 timberland investment organizations (TIMOs) managed about 9 million acres of U.S. timberland, and some analysts predicted that TIMOs and other investor groups (e.g., Real Estate Investment Trusts, or REITs) will purchase another 10 to 15 million acres in the next decade (Wilent 2004).

Two main types of investment models are pursued by TIMOs: separate accounts and closed-end funds. Whereas separate accounts tend to be managed for the long term, closed-end accounts are typically held for a more limited

period of 10 to 15 years before being sold. One estimate is that one-half of all TIMO investments are of the closed-end type (SAF 2004). In terms of forest fragmentation and conversion, it is the closed-end accounts that may exacerbate rates of deforestation. When TIMOs sell land, they pursue the highest value they can receive, which will most likely be for development and real estate. Currently, sales and acquisitions of forest industry (FI) lands continue to be active as market forces, globalization, and consolidation impact the forest sector.

The emergence of timberland holding firms with timber production objectives but no link to processing facilities has created some difficulties in this traditional taxonomy. Lacking processing facilities, these firms would be grouped in the NIPF class. Yet their timber management behavior is more closely akin to that of the integrated firms in FI. Shifts from traditional integrated FI ownership to the TIMO/REIT class were extremely rapid in the late 1990s and early 2000s, and it is likely that the future will see still further decline in traditional, integrated FI ownership.

With a substantial amount of prime U.S. timberland shifting from being a personal or industrial asset to being a financial one, more frequent turnover in forest ownership may be part of a new era in forest ownership. This warrants increased attention in data collection and land base monitoring because such changes have implications for a broad range of forest-based ecosystem goods and services owing to the influence of changing forest ownership patterns on forest conversion, fragmentation, and parcelization.

Increased Numbers of Buildings and People on Forest Land

A significant proportion of forest land undergoing development each year is used for dispersed residential development in fringe suburbs and smaller cities, commonly known as sprawl. Sprawl is characterized by low-density residential and commercial settlements, and increases in housing density on or adjacent to forests can result in changes to the forest's quality and function and changes in forest investment (e.g., Kline and others 2004b). Forest lands are very popular as residential building sites; forests provide homeowners with shade, screening from neighbors,

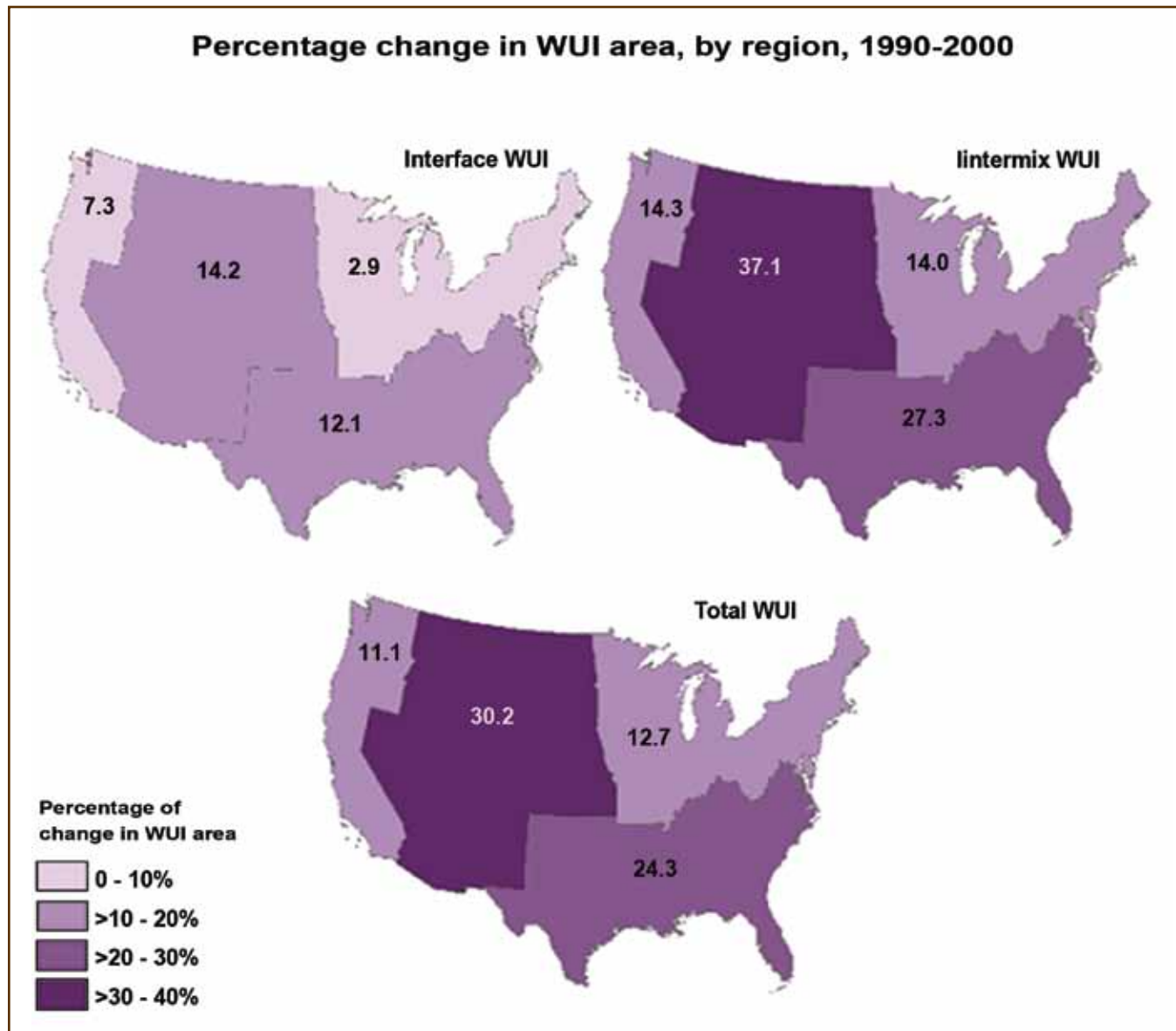


Figure 2—Wildland-urban interface in the United States and percentage change in area by region, 1990-2000.

scenic views, wildlife and bird watching opportunities, and often, easy access to forest-based recreation opportunities. New communication and transportation technologies reduce the isolation of remote locations and make possible long-distance commuting and a wide variety of remote work arrangements. These developments effectively reduce the costs associated with living far from cities and towns. Many areas of the United States are experiencing residential growth in the forests and pressure to develop remaining forests.

The colocation of houses and forests (as well as other wildlands) is captured in the national map of the Wildland Urban Interface (WUI), which was created to aid analysis of the national wildland fire situation (Figure 2). The WUI definition that guided creation of this map and analysis is found in the Federal Register (USDA and USDI 2001) and specifies minimum housing density of 1 structure per 40 acres (or 6.17 structures per km²) and either co-location with, or close proximity to, wildland vegetation. National Land Cover Data and Census Bureau housing data are used

together to determine where these conditions exist. Two main types of WUI—intermix and interface—are identified. Intermix exists where housing and wildland vegetation (at least 50 percent of all pixels in the census block are forests, grasslands, or shrublands) coincide. Areas that meet the housing density minimum but where the wildland vegetation is less dense are considered interface if they are within 1.5 miles of extensive wildland vegetation (defined as an area larger than 5 km² or 1,235 acres with >75 percent wildland vegetation). Together, the intermix and interface make up the WUI (Radeloff and others 2005).

In areas where forest conditions, weather, and climate make wildfire possible, the WUI is a zone where the threat of loss from wildfire is high, because fires can be carried into this zone where they will threaten homes and lives. Consequently, the WUI has high priority for wildfire hazard reduction treatments. The WUI is also the area where wildland fire outreach programs focus their attention. Resource managers and their outreach partners work with communities and property owners to mitigate wildfire hazards and to plan for evacuation and other emergency measures in the event of wildfire.

Across the United States, the 1990s were a period of rapid housing growth, with a net gain of 13.5 million housing units, a rate of 13 percent growth. The WUI was clearly a preferred setting for new housing; overall WUI growth was 22 percent, and intermix growth was 37 percent. The growth patterns for the United States were consistent across the regions, with growth slower in the non-WUI and faster in the intermix WUI (Figure 2). Most of this WUI housing growth took place in areas already designated as WUI in 1990. The growth in WUI area owing to new neighborhoods reaching the housing density minimum for WUI was just 1.5 percent nationally, though it expanded much more in the South and North than in the West.

Across the United States as a whole, the distribution of housing units across the high-, medium-, and low-density interface categories changed little over the decade. Growth in the intermix WUI occurred at high, medium, and low densities, with biggest gains in medium-density intermix. This finding is consistent with adding housing units to existing WUI areas (since existing WUI already had at least

low-density housing in 1990) at a greater rate than adding new areas to the WUI. However, in the West, housing growth was strongest in high-density intermix.

In the South, housing increased by 18 percent, almost as much as it increased in the Rocky Mountain region and presented an even greater contrast between non-WUI housing growth (9 percent) and WUI housing growth (29 percent). Over 3 million housing units were added to the WUI during this decade, and the WUI expanded to cover 17.1 percent of the land area, a greater share than in any other region.

Housing grew more slowly in the North than in any other region at just 9 percent. However, the intermix WUI gained nearly 1.2 million new housing units, an increase of 21 percent over the decade. The area of the WUI also expanded; by 2000, over 15 percent of the North was WUI.

Housing increased by 23 percent in the Rocky Mountain region more than in any other region. Once again, WUI housing growth was even stronger (37 percent), whereas intermix WUI housing grew by 75 percent. Although the 2000 WUI makes up just 1.4 percent of this region's land area, it contains 45.7 percent of the housing units.

Housing growth in the West Coast region was 12 percent overall, with over 1 million new WUI housing units, an 18-percent increase in the number of WUI homes. WUI area also expanded from 5.8 to 6.5 percent of the three-State area. Of the 16.1 million housing units in this region, over a quarter (4.5 million) are located in the interface WUI.

Analyzing housing growth within the WUI classification provides insight about more than the location and density characteristics of recent change; it also indicates the impact of this growth on forests, grasslands, and other wildland vegetation. More than 60 percent of housing units built in the 1990s were constructed in or near wildland vegetation. Although the fire management community originated the WUI concept as an approximation of where values are at risk from wildland fire, the WUI zone is significant for a broader range of ecosystem services. Clean water, timber, recreation, and other services and outputs from undeveloped land are at risk when development encroaches, and encroachment was significant during the 1990s.

Trees in Urban and Developed Areas

The extent of urban forest has grown appreciably in recent decades. As urban lands expand into surrounding areas, natural resources are often affected or displaced. Between 1990 and 2000, most urban expansion in the United States was on forested or agricultural land (Nowak and others 2005). Urban areas in the United States, as defined in the 2000 census, contain approximately 3.8 billion trees with an average tree canopy cover of 27 percent (Nowak and others 2001). The impact of current urban vegetation on environmental quality nationally is on the order of several billion dollars per year (e.g., Nowak and Crane 2002, Nowak and others 2006).

Urbanization concentrates people, materials, and energy into relatively small geographical areas to facilitate the functioning of an urban society. Urbanization often degrades local and regional environmental quality as natural landscapes are replaced with anthropogenic materials. Byproducts of urbanization (e.g., heat, combustion, and chemical emissions) affect the health of local and regional landscapes, as well as the health of people who visit or reside in and near urban areas. Urban vegetation, through its natural functioning, can improve environmental quality and human health in and around urban areas, with benefits including improvements in air and water quality, building energy conservation, cooler air temperatures, and reduction in ultraviolet radiation.

Forest-Land Dynamics

Five types of land base changes (afforestation, deforestation, forest fragmentation, forest parcelization, and increased number of buildings and people on forest land) have significantly altered U.S. forests over the last half century. Although net changes in total forest area are relatively small from a national perspective, many more forest acres are actually involved in land use changes as the gross amount of change is more than 10 times the net amount. Small net changes in forest cover do not necessarily equate to small net changes/losses in services provided by forests. The gross changes reflect the combined outcome of the five types of land-base changes, which often result in spatial rearrangement of land uses and land covers. The South,

which now provides more timber harvest than any other region of the country, in particular, has seen many forest-related land-use changes resulting from population growth and economic activity. In addition to deforestation, substantial forest ownership changes in the South include the shifting of prime timberland from a personal or industrial asset to a financial one. Further, many remaining forest acres are affected by addition of houses on them or nearby. For some forests that are converted to urban uses, there are opportunities to manage urban trees to reduce some of the adverse environmental and health effects associated with urbanization. The South also has a relatively large number of afforestation opportunities, including land suitable for biofuels production as part of global change mitigation strategies.

Determinants

Based on land use theory, empirical testing utilizes real world data to quantify model parameters and test for consistency with underlying hypothesized behavioral relationships. Empirical models can be used to predict how land use will change in response to changes in economic conditions and policies. A method increasingly reported in the literature is econometric modeling, which is based on statistical methods that are used to quantify relationships between land uses and hypothesized determinants such as landowners' profit from land management. Determinants to be tested are drawn from the interaction of biophysical, ecological, and socioeconomic processes and forces, often operating at a variety of scales. For example, market forces tend to operate at much larger scales than biophysical processes commonly studied at micro levels such as stands or reaches of a stream. Most econometric models of land use developed to date have been regional in nature, although Lubowski (2002) recently developed a national land use model.

Econometric land use models typically are estimated with sample plot data for a random sample of parcels or aggregate data such as county-level observations of land use (e.g., Ahn and others 2002, Alig 1986, Hardie and Parks 1997, Kline and Alig 1999, Kline and Alig 2001, Lubowski and others 2006, Parks and Murray 1994, Plantinga 1996, Wear and others 1996). With the advent of satellite imagery

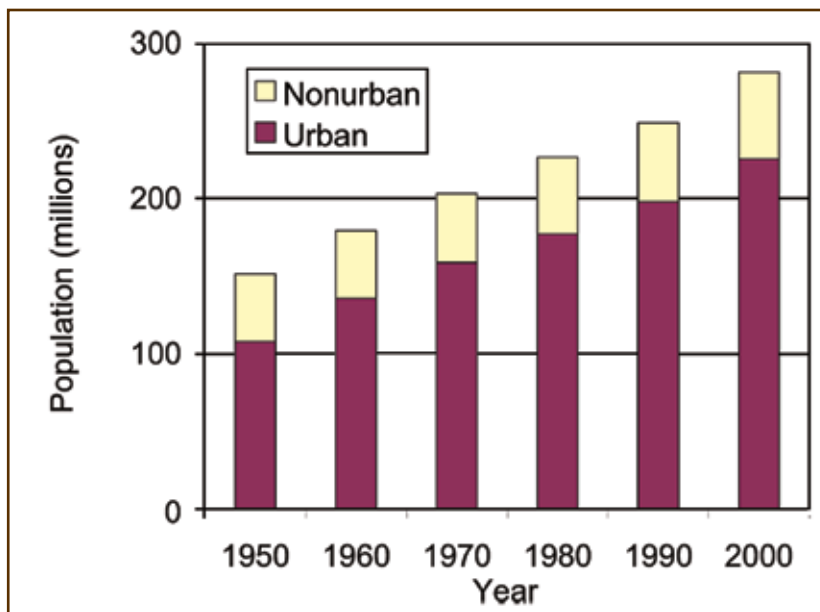


Figure 3—United States population by urban and nonurban components, 1950 to 2000.

and geographical information systems (GIS), econometric land-use models have been estimated using spatially referenced plot or parcel-level data (e.g., Bockstael 1996, Irwin and Geoghegan 2001, Kline and Alig 2001, Wear and Bolstad 1998). Examples of explanatory variables in such models are rents (or its proxies) for forestry, agriculture, and urban/developed uses.

Findings from econometric studies indicate that drivers of deforestation differ notably from those of afforestation and reforestation activities. Major determinants for deforestation associated with conversion to urban and developed uses in the United States are population totals and personal income levels. The rate and extent of urbanization are typically governed by such determinants, which shift demand toward urban and developed uses. Revealed behavior by landowners indicates that values for developed uses (e.g., residential uses) are generally higher than those for rural uses (e.g., forestry and agriculture) (Alig and Plantinga 2004, Alig and others 2004). Within the rural land base, relative land rents for forestry and agriculture affect deforestation (i.e., forest converted to agriculture), afforestation, and reforestation decisions. A number of econometric studies offer insights about determinants of afforestation (e.g., Plantinga 1996) and reforestation activities (e.g., Alig and others 1990, Lee and others 1992, and Kline and

others 2002), including tests of government subsidies. For example, Lubowski (2002) found that rising government subsidies for agricultural crops restrained an increase in forest area in the Mississippi Delta area by 10 percent from 1982 to 1997.

Population

A key determinant in land use change is population growth, which affects the demand for land. But, population growth has potentially contradictory effects on forest land conversion; it can increase demand for land for residential use, while also increasing demand for (and thus, price of) wood products. Rising prices for wood products will tend to increase the relative rents associated with keeping land in forest rather than converting it to residential use.

Figure 3 shows population growth for the United States since 1950. The population has increased almost fourfold since 1900. The distribution of population has also changed over time. For example, decentralization of population relative to city centers has involved a downward trend in the percentage of U.S. population within 3 miles of city centers. Around 1900, about 80 percent of the population lived within 3 miles of those centers; now less than 30 percent of the population does so. However, the proportion of people who live in more broadly categorized “urban areas” has

steadily increased since 1950 and is now about 80 percent. The largest increases in population between 1980 and 2000 have been in metropolitan edges (Heimlich and Anderson 2001). At the same time, the populations of some non-metropolitan counties adjacent to metropolitan ones have increased as well.

The location of population is important in connection with vulnerability of communities and assets (e.g., houses). More than half of the U.S. population lives in coastal areas (within 50 miles of a coast), part of a growing trend. This has implications for vulnerabilities to extreme weather events, such as last year's hurricanes on the Gulf Coast. More broadly, coastal ecosystems are increasingly being stressed by factors that include lowland development in States such as Florida and Texas.

The Census Bureau projects that the Nation's population will increase by more than 120 million by 2050—more than a 40-percent increase over the 2000 population size. The two major components driving the United States population growth are fertility (births) and net immigration. Almost one-third of the current population growth is caused by net immigration. Net immigration remains constant at 880,000 per year, whereas the Census Bureau recognizes that there is considerable uncertainty about the future flow of migrants. By 2050, the Nation's population is projected to be 82 million people larger than it would have been without growth through migration. In fact, about 86 percent of the U.S. population growth during the year 2050 may be due to the effects of post-1992 net immigration.

Population redistribution (i.e., regional growth and decline within the United States) is due in part to amenity migration. The National Forests of the United States have rich scenic and recreational resources that have induced amenity growth in nearby rural areas of the country over the past three decades (Garber-Yonts 2004). The 2000 U.S. Census showed that national forest (NF) counties (counties where >10 percent of land area is national forest, $n = 454$) had higher population growth rates than other counties. This is true especially in those NF counties that were nonmetropolitan ($n = 386$), where the 1990 growth rate was 18.1 percent versus 10.3 percent in nonmetropolitan, non-NF counties. Migration accounted for most growth

across all NF counties, even metropolitan ones, despite the much stronger role of natural increase (i.e., more births than deaths) in metropolitan county growth generally.

Personal Income

Average family income (in real or inflation-adjusted dollars) increased by more than 150 percent from 1950 to 2000, giving individuals more income to spend. The U.S. per capita disposable income in 1998 was \$22,353, which represents more than a 10-percent increase, in real terms, during the 1990s alone.

Further increases in personal income are projected, but not at the level of increase in the 1990s (USDA Forest Service 2001). Even with constant tastes and preferences, a larger population base with higher income levels will result in greater consumption and demands for developed space. For example, consumers may demand more shopping space, as between 1990 and 2000 when the United States shopping area increased by 27 percent and number of shopping centers by 24 percent (USDC Census Bureau 2001).

Incomes From Rural Land Uses

More than 90 percent of land use changes on non-Federal lands in recent decades have been among rural land uses (USDA NRCS 2001): forests, crops, pasture, or range. Where climate and physiography permit, these rural uses can compete for the same land. For example, a reduced supply of agricultural land due to urbanization can result in "replacement" conversion of forest land to agricultural uses (Alig and Healy 1987). Market forces often result in shifts in the use of rural lands between agricultural production and forest production. Increasingly global markets are also affected by technological improvements, and, since World War II, increases in cropland yields per acre have generally been larger than corresponding increases in forestry yields. Enhanced productivity has the effect of concentrating agricultural uses on a smaller land base and easing demands for conversion of forests to agriculture.

Net income from forestry enterprises is affected by prices for products including timber. Over the 50 years from 1952 to 2002, real prices of softwood lumber, hardwood lumber, and paper rose (at compound rates of 0.8 percent,

0.4 percent, and 0.3 percent, respectively), whereas prices of softwood plywood, oriented strand board (OSB) (since 1976), and paperboard fell (Haynes and others 2007). Recent timber market projections for the USFS Resources Planning Act (RPA) Assessment (Haynes and others 2007) indicate that prices for some forest products will increase over time but at a slower rate. Prices of softwood lumber, hardwood lumber, and OSB are projected to rise slowly (at compound rates of 0.2 percent, 0.3 percent, and 0.1 percent, respectively), and prices of softwood plywood, paper, and paperboard are projected to remain stable or fall. Slow product price growth is reflected in many categories of timber prices (which determine returns to landowners). Sawtimber stumpage prices in the South and interior West are expected to decline slowly after 2010, while those in the PNW and North are expected to rise at about 0.2 percent and 0.8 percent per year. Southern hardwood pulpwood prices rise in the projection as hardwood inventories contract. Southern softwood pulpwood prices oscillate in response to the changing fiber mix, ending the projection near recent levels.

On the agricultural side, real prices have declined for major agricultural crops grown on land also suitable for forestry. Toward the end of the 20th century, farmers and ranchers were increasingly caught in a cost-price squeeze. The ratio of the Prices Received Index to the Prices Paid Index fluctuated considerably over the past hundred years (Ahearn and Alig 2006, USDA NASS 2006). Commodity prices spiked upward during both World Wars and plunged during the Great Depression. Prices again shot upward during the early 1970s, spurred on by sharply increased world demand. Technological improvements in agriculture, such as in yields per acre, have generally outpaced those in forestry. Although this has resulted in an increase in the use of land for agricultural cropland in some areas, the increase in aggregate crop yields and downward pressure on agricultural market prices resulted in land saving for farmers and a net switch from agriculture to forestry at the national scale.

The outlook for agricultural income often involves substantial uncertainty in land use studies because of the cyclical nature of the agricultural economy, effects of government programs, and technological developments (e.g., genetically modified materials) (Alig and Ahearn

2006, Alig and others 2003). Emergency aid provided to farmers through legislation in 1998 through 2001 suggests that the direction of the current policy transition remains uncertain. Under the Federal Agricultural Improvement and Reform Act of 1996, the Federal Government is moving further away from direct involvement in farm commodity markets. However, threats of droughts and other elements that inject volatility in agricultural and forestry production have prompted some annual adjustments in the government intervention plans. With lower agricultural prices than in the mid-1990s, the 2002 farm bill was debated during a period when agricultural prices were low. The debate showed that there was some interest in changing the thrust of the 1996 farm bill by introducing new countercyclical policies. Policy shifts in this area could affect the dynamics of the link between land use patterns and market prices.

Projections

Projections from different studies (e.g., Resources Planning Act Assessments) are summarized next and compared where appropriate. Land use projections are generally prepared by obtaining projections of the independent variables and then simulating the impacts of projected conditions on future land uses (Table 3). Projections can be implemented to contrast the potential effects of policy or market changes with historical usage (counterfactual simulations, e.g., Lubowski and others 2006) or to project future land uses over a range of scenarios (e.g., Alig and others 2003). Projection exercises have indicated that future land uses are especially sensitive to changes in population density, income, and agricultural and timber prices and production costs.

Developed Uses

Urban and developed areas are projected to continue to grow substantially in line with the projected population increase of more than 120 million people over the next 50 years (Alig and others 2004). This will be part of a global increase in population, as the world's population is projected to grow from 6 to 9 billion by 2050. The U.S. developed area is projected to increase by 79 percent, raising the proportion of the total land base that is developed

Table 3—Sources of land use projections, by region (note: no large-scale forest parcelization projections were located)

Land base change	Approach	Studies	Projected amount in U.S.
Deforestation	Econometric and mathematical optimization	Alig and Healy 1987, Alig and others 2004, FASOM (Alig and others 2002) Alig and Plantinga 2004	More than 50 million acres deforested by 2050, mostly for developed uses
Afforestation	Econometric and mathematical optimization	Lee and others 1992, Kline and others 2002, FASOM (Alig and others 1998, 2002), Alig and others 2003, Alig and Plantinga 2004	About 20 million acres converted from agriculture to forest by 2050 (FASOM)
Forest fragmentation	Econometric	Wear and others 2004	South projected to lose about 2 million acres of interior forest by 2020
Structure additions to forests	Statistical	Stien and others 2005	44 million acres of private forest with substantial increase in housing density by 2030
Urban forest changes	Statistical	Nowak and Walton 2005	29 million acres of forest to be urbanized by 2050

from 5.2 percent to 9.2 percent. Projections based upon Census Bureau data indicate similar substantial increases in urban area. Urban land in the United States is projected to increase from 3.1 percent in 2000 to 8.1 percent in 2050, an area of about 97 million acres, which is larger than the state of Montana. Most of the urban growth is projected to occur around the more heavily urbanized areas, with significant expansion in the East and along the west coast.

Population and income pressures on land uses are not uniform across the Nation. For example, population has shifted from the North to the South and the West in recent decades. Because much of the growth is expected in sensitive areas already burdened by anthropogenic impacts, such as some coastal counties, implications for landscape and urban planning include potential impacts on sensitive watersheds, riparian areas, wildlife habitat, and water supplies. Although providing additional living space and infrastructure, added development may also diminish agricultural output by reducing farmland and changing ecological conditions by converting and fragmenting forests and other natural landscapes. The projected developed and built-up area of about 175 million acres in 2025 represents

an area equal to 38 percent of the current U.S. cropland base, or 23 percent of the current U.S. forest land base. In line with recent historical trends, the South is projected to continue to have the most developed area through 2025 (Alig and others 2004).

Forest Land

Total forest land area in the United States is projected to decrease on net by approximately 23 million acres, or 3 percent between 1997 and 2050 (Alig and others 2003). Projections of forest land area are related to those above for the other major land uses. The main reason for the projected reduction in forest land area is conversion to urban and developed uses associated with the projected increases in population and income discussed earlier.

The projected reduction in forest land is consistent with earlier studies (e.g., USDA Forest Service 1988). The notable reductions in the South are generally consistent with the regional assessment of the southern forest resource situation, indicating that urbanization, among all forces of change, will have the most direct, immediate, and permanent effects on the extent, condition, and health of

forests in the South (Wear and Greis 2002). Projections estimate that tens of millions of acres of forests in the region will be lost to urbanization from 1992 to 2040. The 13 States in the South are projected to have an overall net decline of only 2 percent in forest area because some farmland will be converted to forests. Timberland area is projected to increase in the South-Central region, where much land is suitable for use in either agriculture or forestry, as a consequence of conversion of agricultural land to forest. Such conversion is to be expected if it is assumed that real prices for agricultural commodities fall and agricultural subsidies and related programs are reduced.

The largest forest area losses in the South are projected for the Southeast. Forest land at the periphery of urban areas is likely to be developed. For example, in Georgia about 5.6 million acres of forests may be converted to developed uses by 2010. It is conservatively estimated that as much as 26 percent of the timber-growing stock measured in the 1997 Georgia inventory could be affected (Wear and Newman 2004).

Projections for other regions of the country largely follow recent historical trends (Alig and others 2003). Most of the losses are projected to be on NIPF lands.

Comparison of Land Use Projections—

We compared land use projections from models by Lubowski and others (2006) and Hardie and others (2000). We use these models to project land uses in the Southeastern United States to the year 2020. Using two separate models allows us to examine potential differences based on modeling structure. The Lubowski and others model estimates transitions or changes in land uses, whereas the Hardie and others model estimates the equilibrium land use shares in each period. Both models summarize land uses at the county level and are based on measures of land uses from the National Resource Inventory (NRI). The 1997 NRI survey serves as the base year for projections.

Projections for the South as a whole show a significant continuation of urbanization and are consistent with the Nowak and others (2006) projections and separate projections by Alig and others (2004). Urban area in the South is projected to increase under all scenarios.

Year 2020 forest land area projected by the model of Lubowski and others is similar to those generated by the model of Hardie and others under a high timber price scenario. The Hardie and others model projects that forest land area in the South could fall as much as 20 million acres under a low timber price scenario. Econometric land use models are generally effective tools for projecting forest area; but an important consideration is whether there are any changes in the underlying structural relationships over the historical or projection periods, analyzed using statistical methods to test for changes (e.g., significant policy environment alteration) in model parameters over time (Ahn and others 2000).

Projections of Forest Fragmentation—

Relatively few studies have projected forest fragmentation, especially at larger scales. One example of a broader scale study is in Wear and others (2004) where changes are forecast in interior forest for each county in the South, a region where recent trends include significant land use change. Wear and others, who based forest fragmentation projections on population density forecasts to the year 2020, assumed that relative returns to agricultural and timber production would remain at current levels. Under this scenario, the South as a whole is forecast to lose 747 000 ha (1.85 million acres), or about 2.12 percent of interior forest cover.

These changes are not constant across the region. Among ecological sections, the Southern Appalachian Piedmont would lose the greatest area of interior forest cover (173 166 ha or 427,903 acres). The gulf prairies and marshes in Texas, which have very little interior forest, would lose the greatest proportion of interior forest (56.7 percent) (Wear and others 2004). The second and fourth greatest reductions are projected for the eastern and western Florida coastal lowlands, respectively. Aggregation to the ecological province level indicates that the Outer Coastal Plain would experience the greatest reduction in interior forest. All ecological sections with losses forecast at greater than 2 percent are located either on the Atlantic and Gulf Coastal Plain or in the upland areas of North Carolina, Virginia, Tennessee, and Kentucky.

The distribution of forecast losses of forest interior between urban and rural counties was examined further. Urban counties were defined as those attached to metropolitan statistical areas (MSAs) by the Office of Management and Budget, and rural counties were defined as the remainder. The MSA counties contain 492 690 ha (1,217,463 acres) or 66 percent of the total forecast loss of 747 744 ha (1,847,716 acres). (Heavily impacted MSAs are concentrated in Florida. The Tampa-St. Petersburg-Clearwater MSA is forecast to lose 34.5 percent of its interior forest, and 7 of the 10 MSAs with the highest percentage losses are found in Florida. Columbia (South Carolina), Atlanta (Georgia), and Raleigh-Durham-Chapel Hill (North Carolina) round out the top 10.

Housing Density—

Projections of housing density increases on forest land were made in the Forests on the Edge study discussed in “Forest land Conversion and Recent Trends.” This project has ranked watersheds across the conterminous United States according to the percentage of each watershed that contains private forest projected to experience increased housing density (Stein and others 2005, Theobald 2005). Three housing density thresholds were identified: rural (no more than 15 units for every square mile); urban (at least 64 units per square mile); and ex-urban (16 to 63 units per square mile). Areas identified as having a substantial increase (44 million acres in total) were those where housing was projected to increase from either rural or ex-urban to urban (22 million acres) or from rural to ex-urban (22 million acres). Watersheds included in the assessment had at least 10 percent forest cover with a minimum of half private land.

Note that for the WUI research, projections of WUI growth, 2010 to 2030, will be made by Northern Research Station scientists and collaborators as soon as the housing density projections on which they are built have been completed. The WUI projections will assume that vegetative cover will remain constant through 2030.

Most watersheds projected to experience the greatest amount of change were located in the East, although some were located in the Great Lakes area, California, and the Pacific Northwest. The greatest change will be in 12 States

in the Northeast and South (Stein and others 2005). A study in progress is identifying watersheds where private forests contribute most to water quality, timber, interior forest, and at-risk species habitat and determines where these contributions may be most affected by factors such as housing increases, fire, air pollution, insect pests, and disease. As discussed in a case study at this conference, private forested watersheds most affected are generally found in the East and along the West Coast (Stein and others, this volume). Stresses on forest environmental conditions can be compounded if more people live on the remaining forest land as the U.S. population density continues to increase. The United States had about 80 people per square mile of land in 1999 (USDC Census Bureau 2001) in comparison to about 5 people per square mile in 1790.

Risk and Policy Considerations

A broad complement of research studies is consistent in projecting continued development of forests or increases in the housing density of remaining forests in the future, or both. Key assumptions in such studies include projected increases in population and income, which are generally viewed as more likely to approximate future conditions over the next five decades. In contrast, other assumptions are viewed as having larger bands of possible outcomes, and examples of such assumptions are future changes in the agricultural sector, technological changes, changes in forest practice regulations, and global climate change. The relative sensitivity of projected land use changes to such assumptions has been tested in a number of studies (e.g., Alig and others 2003, Haynes 2003, USDA Forest Service 1988).

It appears that tens of millions of acres of forest are at risk of being converted to nonforest and that many more acres remaining in forest cover will have houses and other structures added over the next several decades. To the extent that this loss of forest land may have detrimental impacts on the social values derived from forests, it seems reasonable to ask how changes in policy might affect this outcome. We next discuss several types of policies that could have some influence on the future of forests in the United States.

Government Policies

Government policies that can contribute to development include Federal/State expansion of highways, income tax subsidy for home ownership, and extension of public utilities. Policy responses to any perceived problem can involve local, regional, State, and national responses. An example of local influence is zoning and development impact fees. Regional responses may include regional governance or tax sharing. State policies include urban growth boundary approaches as in Oregon. Conservation easements have been increasing in popularity as a tool for encouraging the protection of forests and other lands. Landowners receive tax benefits or are paid a lump sum in exchange for restricting the type and amount of development and other uses that may take place on their property. Easement restrictions are identified in a legal agreement signed by the landowner and a conservation recipient (usually a public agency or land trust). To qualify for tax benefits, an easement must be perpetual, with future owners bound by the same restrictions.

National Level—

At a national level, the USDA (1983) is an example of a Federal agency with a major land use responsibility. Departmental policy is to promote land use objectives responsive to current and long-term economic, social, and environmental needs. This policy recognizes the rights and responsibilities of State and local governments for regulating the uses of land under their jurisdiction. It also reflects the department's responsibility to:

1. Assure that the United States retains a farm, range, and forest land base sufficient to produce adequate supplies, at reasonable production costs, of high-quality food, fiber, wood, and other agricultural products that may be needed.
2. Assist individual landholders and State and local governments in defining and meeting needs for growth and development in such ways that the most productive farm, range, and forest lands are protected from unwarranted conversion to other uses.
3. Assure appropriate levels of environmental quality.

Contemporary land use policies as a whole are multi-objective in nature, as is evident in the policy directive of USDA (1983). Implementation of multiobjective policies is laden with tensions. For example, although the USDA policy directive was written nearly two decades ago, major challenges still exist in attaining a balance that satisfies USDA's many constituents. One challenge involves a major contemporary focus of land use policies—the management of the direction of development. Urban sprawl has been cited as one of the leading concerns of Americans (Pew Center 2000). According to the Pew report, approximately 1,000 measures aimed at changing planning laws and at making United States development more orderly and conserving were introduced in State legislatures in the late 1990s. Concerns about sprawl originate from both the disamenities associated with increased congestion as well as the loss of productive land for agriculture and forestry uses. Although the recent and current situation in agriculture is one of surpluses and depressed markets, agriculture is historically cyclical in the long run. With a rapidly growing world population, food and fiber demand is likely to increase in the future. There has not been sufficient confidence that the current land market is capable of appropriately discounting the future value of farmland to account for this eventuality. Long-term loss of prime agricultural land and forest land to urban uses arises in part because lands that are highly suitable for agriculture or forestry and for urban expansion are often one and the same—gently sloped, fertile valleys, and flood plains. Urban conversion is generally one way; the land is usually irretrievably lost for less intensive use within typical planning horizons.

State and Local Levels—

State and local governments use a variety of tools to protect farm and forest lands as productive resource bases. These tools include agricultural zoning, differential farm tax assessments, right to farm laws, agricultural districts, purchase of development rights, transfer of development rights, comprehensive land use planning, and urban growth boundaries.

At a State level, Oregon has a statewide land use policy involving urban growth boundaries (e.g., Kline and Alig 1999). For example, Portland, Oregon, in the 1990s had

30 percent of new housing as infill and redevelopment. Boulder, Colorado, has an urban containment policy. The city restricts new development, and the majority of the workforce lives outside city limits.

Two USDA programs, the Farmland Protection Program for agricultural lands and the Forest Legacy Program for forest lands, complement State and local government programs that purchase development rights. The purchase of development rights gives government agencies the option of conserving open space for future use in farm or forest production without government acquisition. The land will not necessarily be required to stay in a current farm or forestry use, but under a program that purchases development rights, a landowner will not be allowed to develop the parcel. Because the cost of cultivating undeveloped land is considerably less than the expense associated with reversing development, purchasing development rights is viewed as an investment in food and forestry security for future generations. Conservation easements and other partial interests in land have also been increasingly used to accomplish particular natural resource protection goals such as maintaining open space that provides scenic beauty and wildlife habitat.

Most land use protection programs are designed to conserve urban open space. Few are focused directly on working forests. Managed by the USDA Forest Service in partnership with State governments, the Forest Legacy Program is designed to encourage the protection of privately owned forest land and promote sustainable forestry practices by purchasing development rights, including conservation easements. Legacy parcels continue to produce forest commodities and noncommodity ecological values such as healthy riparian areas and fish and wildlife, as well as scenic, aesthetic, cultural, and recreational resources, on landscapes otherwise likely to be shifted to nonforest use. As of 2006, the program has protected 1.15 million acres in 33 States. Interest in the program has grown, with 46 States and Territories now enrolled in the program, up from 24, 5 years ago, and with over \$200 million in requests each year. In 2006, Federal appropriations for the program were \$56 million, with 91 percent of this money directed to conservation projects. Conservation easements in general have

been increasing in popularity as a tool for encouraging the protection of forests and other lands, and are also used by NGOs, such as the Pacific Forest Trust. Landowners receive tax benefits or are paid a lump sum in exchange for signing a legal document that restricts the use of their land. These restrictions might include development as well as certain other forms of land use.

Risk, Hazard, and Land Use Change

The focus of this conference is primarily on threats to ecosystems, and we discuss the role of land use change and housing growth in creating and amplifying threats to ecosystems. A related issue worth exploring is the relationship between human settlement patterns and vulnerabilities to natural disasters. Natural disasters have many varied consequences, including damage to ecosystems and human communities. Recent trends in land use and housing growth not only create stresses on natural ecosystems, they also increase society's vulnerability to natural hazards.

Housing growth is perhaps the single most important factor behind increasing economic losses from natural disasters. The threat posed by most natural disasters has not changed significantly over time. Wildland fire is an exception, to some extent; many ecologists argue that forest management policies contribute directly and indirectly to increasing the severity of wildland fires. Global climate change has also been indicted in recent catastrophic weather events, and although scientific opinion is mixed regarding its role in current patterns, scientists agree that there is potential for significant change in the future. However, in the short run, i.e., over the past 50 years, the likelihood of natural hazards has been relatively stable, but losses in the United States have increased because our vulnerability to these hazards has increased. More houses and more wealth concentrated in regions of the country facing significant hazard levels describe the trend in the United States over the past 50 years (Cutter and Emrich 2005, De Souza 2004, van der Vink and others 1998).

Regional patterns of growth and decline in the United States have shifted population and property value to more vulnerable areas (van der Vink and others 1998). By 1970, population and housing growth had shifted away from the

cities of the Northeastern United States and into two regions facing considerable natural hazards: the Southeast, with its exposure to hurricanes from both the Atlantic and the Gulf; and the West where a wide range of hazards are present. Colorado and California stand out as States where population and housing growth have been substantial (California earlier in the period, Colorado later) and both face the threat of catastrophic wildland fires. In California, earthquakes and landslides are also major threats to heavily populated cities. Coastal Oregon and Washington are exposed to tsunami risks; Southwestern States (Arizona, and later in the period, New Mexico and Nevada) have grown tremendously and have active fire regimes.

Selective urban deconcentration, which has been the overarching pattern of settlement change in the late 20th century (Johnson and others 2005), has brought growth to many rural communities and to suburbs more distant from the urban core. This change from the centuries-long urban concentration pattern contributes to vulnerability in two ways. First, isolated communities and especially unincorporated areas have less infrastructure (e.g., roads and water supply systems) and fewer resources for providing protection services (e.g., police and fire protection). Rapid growth exacerbates the difficulties of providing adequate infrastructure. Second, wildland fire is a meaningful threat to homes in the wildland urban interface, which is typically found around the outer edges of metropolitan areas and throughout the countryside, the same areas where housing growth has been most dramatic.

United States society's response to natural disasters has been more oriented to reaction than to planning. When losses to human communities are substantial, the outcome is often new policy, reallocation of public spending, and regulation. To take an example familiar to the resource management community, the Healthy Forests Restoration Act can be seen as a policy response to the bad fire seasons of 2000 and 2002. With human community vulnerability and loss comes the prospect of more changes such as these. To date, few incentives or policies have addressed the root cause—the development of new housing units without regard to landscape patterns, ecological processes, or hazard exposure.

Research to Improve Analyses of Risk of Forest Land Conversion: Expected Benefits and Costs

Improved analyses of threats from conversions of forest land require additional data and research pertaining to:

1. Expected benefits and costs regarding likelihood of land use conversion for a particular unit of land.
2. Environmental impacts, losses in commodity production, and other costs that may arise if a unit of land is developed, such as increased costs for fire suppression with houses in the wildland-urban interface.
3. Estimates of opportunity costs of retaining forest land, such as land values, to provide a sense of what it may cost to transfer development rights, implement a conservation easement, or undertake some other policy action.

Here we point to several studies as examples where improved information is being pursued, with a more detailed discussion outside the scope of this paper.

At a national scale, the RPA Assessments have examined costs and benefits associated with land conversion for several decades (e.g., Alig and others 2003, Nowak and Walton 2005, USDA Forest Service 1989). With a growing wildland-urban interface, other national studies have focused on housing growth (e.g., Stein and others 2005, Stewart and others 2003). Other national studies have focused on global climate change and relationships to land use and land cover changes (e.g., Alig and others 2002). At a regional scale, two studies for the South illustrate the changing nature of the informational requirements and analytical approaches: the study of the South's "fourth forest" (USDA Forest Service 1988) and the *Southern Forest Resource Assessment* (Wear and Greis 2002).

Forest conditions have received increasing attention in recent years, with bioregional assessments implemented to examine conditions at an ecoregional rather than a jurisdictional level: e.g., the Southern Appalachian Assessment (USDA Forest Service 1996), the Southern Forest Resource Assessment (Wear and Greis 2002), and the Coastal

Landscape Analysis and Modeling Study (Kline and others 2003). One part of improving such resource-centric studies is better tracking of ecological structures that are changing in response to human population growth and economic developments at broad scales across a region (Wear and others 2004). The identification of specific conservation targets within these broad areas would require additional work at a finer scale. Nevertheless, the methods described by Wear and others (2004) could provide a mechanism for defining conservation priorities for the region at a broad scale. Forecasts of interior forest change can be viewed as a risk measure indicating where human activities are more likely to change ecosystem structure, similar to the risk indicators developed at a much finer scale by Theobald (2003). This forecast provides a first step in setting priorities: defining where the structure is likely to change and where it is likely to be relatively stable. Wear and others (2004) identified four ecological sections of the South where 5 percent or more of existing interior forest is forecast to be lost by 2020, but they also identified 16 ecological sections, or about half of the sections in the Southeastern United States, where less than 1.5 percent would be lost. Areas that are found to be essentially stable could be excluded from further detailed analysis, allowing analysts to focus their efforts on that portion of the landscape that is more likely to change without some intervention.

A second step in defining conservation priorities involves examining the ecological condition of the broad areas with relatively high threat levels. Indicators such as numbers of imperiled species highlight where ongoing change may have the most impact on biodiversity. Although a detailed assessment of ecological scarcity is beyond the scope of this paper, the Southern Forest Resource Assessment (Wear and Greis 2002) illustrates the approach. For lands that are urbanized, a tool developed to assess urban forests structure and functions is the Urban Forest Effects (UFORE) model (Nowak and Crane 2000; www.itreetools.org).

Different types of land base changes can result in different forest ecosystem conditions because acres exiting (e.g., through deforestation) or entering (e.g., through afforestation) the forest land base can represent quite

different forest conditions. This change becomes even more important when there is a relative acceleration in one type, as in deforestation, as occurred in the 1990s, when about 1 million acres of forests were converted to developed uses per year (USDA NRCS 2001).

With a projected increase of more than 120 million people in the United States over the next 50 years, the different projections of additional developed land area and housing growth all point to significant increases that represent threats of forest conversion. Demand for wood products is expected to keep growing, driven by the same population increases and economic development that affect demands for other major land uses. Given dynamics of the changing population and social values, some forest conversion can adversely impact provision of public goods by forests, such as the environmental service of storing terrestrial carbon to mitigate climate change, which falls outside private decisionmaking. Measuring and evaluating multiple forest benefits associated with public goods can be difficult owing to a general lack of information describing forest outputs and their values. This lack of information is especially true when it comes to valuing benefits accruing from ecosystem services, a set of values clearly needed to fully value open space and other ecosystems services provided by private forests. Efforts to better align commercial uses of forests with conservation objectives have led to increased interest in what is being called sustainable forestry, although there are similar efforts tied to other major competing interests in the land, such as sustainable agriculture or sustainable communities. Land use will continue to change as private decisionmakers and society examine options to adjust to changing demands for and supplies of renewable resources (e.g., biofuels for energy security and to address climate change) (e.g., White 2010) and ecosystem services from the Nation's forest and aquatic ecosystems. Sustainability analyses will be enhanced if both land use and land investment options are examined. Analyses should be explicit as to timing of tradeoffs and market-level impacts, to help promote enhanced integrated macro analyses of land base changes using a balanced mixture of spatially explicit data and other information.

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Soil Quality Is Fundamental to Ensuring Healthy Forests

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Abstract

Government agencies, industrial landowners, and private landowners often strive to maintain soil quality after site management activities in order to maintain site productivity, hydrologic function, and ecosystem health. Soil disturbance resulting from timber harvesting, prescribed fire, or site preparation activities can cause declines, improvements, or have no effect on site productivity and hydrologic function. In many cases, detailed soil resource data can be used to determine the stress level and ecosystem health of stands and may be one method used to determine the risk of disease or insect outbreak. Currently, organic matter accumulations in many forests exceed historical levels. Fire suppression or fire exclusion has produced numerous overstocked stands. When this condition is combined with increased climatic variation, drought, and type conversion, these stands have a high risk for catastrophic wildfire. The resulting large, high-intensity, and high-severity fires could contribute to changes in soil quality and lead to outbreaks of insects and diseases in many ecosystems. Changes in ecosystem processes can also be associated with changes in overstory properties that alter forest stand resilience. For example, loss of western white pine to blister rust infection in the Northwestern United States has caused a type conversion to forest species that are not tolerant of root diseases, are not fire resistant, and sequester nutrients in the surface mineral soil and tree crown that can later be

lost through logging or fire. These relationships, and others, can be used in conjunction with soil resource data bases to assess susceptibility to threats and to help develop management strategies to mitigate disturbances. Development of monitoring strategies that use common methods that can be utilized by a variety of land management agencies and specialists is a key component for relating forest health to soil changes after fire or other land management activities.

Keywords: Fire suppression, forest health, soil indicators, sustainable forestry.

Soil Quality

Soil quality and function are interrelated concepts that represent the range of soil properties and their associated ecological processes. The National Forest Management Act of 1976 and related legislation direct U.S. Department of Agriculture Forest Service managers to maintain the productivity potential of national forest land. The British Columbia Ministry of Forests uses professional assessment to evaluate the impacts of management practices on organic matter (OM) losses (British Columbia Ministry of Forests 1997). Even with these mandates and laws, the concept of soil productivity has not been well defined, and the impact of timber removal or fire on the productive potential of soils is not well understood or easily measured (Powers and others 2005). Soil quality has been defined as the capacity of a soil to function within an ecosystem to sustain biological productivity, maintain environmental quality, and promote plant and animal health (Doran and others 1996). In addition, soil health definitions include maintaining the integrity of nutrient cycling and resilience to disturbance or stress (O'Neill and others 2005). Tree or stand growth has often been used as an indicator of soil productivity changes, but growth reductions attributable to management practices may take >20 years to become manifest in many North American ecosystems (Morris and Miller 1994). The forest floor is likely a key element in maintaining healthy ecosystems, but it is also the one most impacted by fire and forest management (Tiedemann and others 2000). Maintaining site organic matter at or near the ecosystem baseline levels may help reduce nutrient losses (McNabb and Cromack 1990), insect (Fellin 1980a) and disease (McDonald and

others 2000) outbreaks, and may ultimately reduce many forest health problems. For example, Page-Dumroese and Jurgensen (2006) described baseline organic matter levels in 13 undisturbed forests around the Northwestern United States. The levels include measurements of downed wood, forest floor and mineral soil organic matter (OM), carbon (C) and nitrogen (N), and they can be used to determine when a site has excess or deficient organic matter stores. Carbon accumulation, as measured by forest floor depth or amounts of downed wood, can be a useful indicator of forest health because forests with OM levels above their historical baseline levels are at risk from increased insect and disease activities or high-intensity fires (Oliver and others 1994).

Wildfire Impacts

Active fire suppression during the 20th century has increased OM volume on the soil surface in forest stands that historically had supported a regular fire-return interval (Oliver and others 1994). It has been suggested that active fire suppression, together with selective harvesting of seral species, has resulted in a shift in dominance to shade-tolerant Douglas-fir (*Pseudotsuga menziesii*) and grand fir (*Abies grandis*) in many western forests (Mutch and others 1993, Swetnam and others 1995), and a build-up of fuels in other forest types such as ponderosa pine (Covington and Sackett 1984, DeBano and others 1998). A consequence of the advance in succession in some western forests and the suppression of fire in others is the increased accumulation of aboveground biomass and nutrients in standing live trees, standing dead trees, downed wood, and forest floor (Keane and others 2002, Major 1974). Particularly in the Western United States, this increase in OM biomass in fire-suppressed or fire-excluded forests has led to forest floor C accumulation far in excess of normal conditions. In the absence of fire, critical nutrients are tied up in this excess plant debris, possibly causing the site to become nutrient limited (Harvey 1994). Accumulations of woody residue and surface OM from fire suppression activities are also undesirable because of the increased risk from high-severity wildfires and slower OM decomposition rates (Covington and Sackett 1984).

Deep accumulations of organic material (those in excess of decomposition) are generally lost through fires (Oliver and others 1994). If the fires are frequent and of low severity, few organic matter (or nutrient) losses occur (Neary and others 1999, Page-Dumroese and Jurgensen 2006), but infrequent, high-severity fires can be catastrophic to soil productivity and forest health if significant amounts of biomass have accumulated (Habeck and Mutch 1973). Neary and others (1999) outlined the threshold temperatures for biological disruptions in soils. The cumulative impact of a catastrophic fire may directly affect belowground processes because it can alter nutrient inputs (soil macro- and microfauna), increase soil temperatures, increase erosion, alter evapotranspiration rates, and decrease moisture availability (Neary and others 1999). These detrimental impacts may also exacerbate insect and disease outbreaks (Harvey and others 1989, Jurgensen and others 1990).

Management Impacts

The Healthy Forest Restoration Act of 2003 was designed to help alleviate the accumulation of OM by using partial cuts and prescribed fires to remove small-diameter trees and surface OM from many forest stands. Prescribed fire and harvesting operations are important variables in determining soil OM losses because they both influence the removal of organic matter, C, and N on the soil surface and influence the amount of OM within the mineral soil profile. However, frequent repeated burns and multiple entries by mechanical equipment to reduce wildfire risk may impact ecosystem processes, soil quality and productivity, and site sustainability at a variety of scales (e.g., a cutting unit or an entire watershed).

Prescribed Fire

Prescribed fire, as a site preparation method or for underburning intact stands, is a major component of the restoration effort to reduce fuel levels in many forested ecosystems (McIver and Starr 2001). Prescribed fires produce a wide range of fire intensities, depending on fuel loads, fuel moisture content, slope position, and slope aspect (Brown and others 1991, Huffman and others 2001, Little and Ohmann

1988, Oswald and others 1999, Vose and others 1999). Fire severity is a term used to describe the impact of fire on both above- and belowground stand components (DeBano and others 1998, Keane and others 2002). Various burn indices have been developed to evaluate fire effects on ecosystem processes and soil productivity (Neary and others 1999), but three classes are commonly used: **low severity**—a non-lethal, low-intensity surface fire in which patches of surface OM are lost, **moderate severity**—a patchy fire that creates a mixed mosaic of fire intensities and all small-diameter (<7.6 cm) OM is consumed, and **high severity**—a stand-replacement fire that kills more than 90 percent of the trees, consumes most surface OM, and some OM in the mineral soil has been lost (mineral soil changes color) (Keane and others 2002). Information is needed on the impact of prescribed fire on soil OM content and distribution to evaluate the effects of fire management practices on residual fuel loads, soil erosion potential, and long-term site productivity (Elliot 2003, Neary and others 2000).

Because of the range of fire conditions possible in any given stand, the range of soil conditions will also be variable (Landsberg 1994). Often, reduced productivity or health of the remaining stand is influenced by the amount of injury to remaining trees, crowns, and roots, reductions in microorganisms in the surface mineral soil, and changes in C and other nutrient pools (Klemmedson and Tiedemann 1995, Page-Dumroese and others 2003). In addition, the impacts of prescribed fire are dependent on forest type and past management (Schoennagel and others 2004). Historically, dry forests of the Western United States (e.g., *Pinus ponderosa*, dry *Pseudotsuga menziesii*, etc.), which had a relatively short fire return interval and low fire intensity (Agee 1998), did not usually develop disease problems (like *Armillaria* spp.) when the fire return interval remained short (McDonald and others 2000). However, because these dry stands are water limited and have relatively shallow forest floors (Page-Dumroese and Jurgensen 2006), as fire exclusion and suppression increase, so does the stress and competition between trees for limited water and nutrient resources (McDonald and others 2000).

The combined effects of harvest operations and prescribed burning on the remaining forest slash may severely

impact mesofauna living in the forest floor by either directly killing them or removing their desired food source (Fellin 1980b). This includes both pests and beneficial insects. For instance, predators and parasites of spruce budworm that live in the forest floor may help to regulate budworm numbers at low levels so that existing populations do not reach epidemic proportions (Fellin 1980b). Stands affected by low-severity fires, which can leave many unburned areas of the forest floor, may provide a favorable location for maintaining important insects in the forest floor and ensuring that decomposition and nutrient cycling processes continue (Fellin 1980b).

Results of studies of repeated prescribed fires on soil quality and forest health are mixed. Annual or biannual prescribed fires have been shown to reduce pools of C, N, and sulfur (S) in the forest floor after 30 years (Binkley and others 1992). Burning on a 2-year interval for 20 years reduced N from both the forest floor and mineral soil (Wright and Hart 1997). However, a study of prescribed fire at intervals of 1, 2, 4, 6, 8, and 10 years in ponderosa pine resulted in an increase of soil C and N (Neary and others 2002). These long-term changes in nutrient status may or may not affect long-term site productivity (Jurgensen and others 1997), but on some sites they could affect insect and disease outbreaks (Harvey and others 1989). For example, the introduction of white pine blister rust (*Cronartium ribicola*) in the Western United States has reduced the number of 5-needled pines (e.g., *Pinus monticola* or *P. lambertiana*) in many ecosystems (Monnig and Byler 1992). These pines, along with ponderosa pine (*P. ponderosa*) and western larch (*Larix occidentalis*) tend to be broadly adapted species (Rehfeldt 1990) and are relatively tolerant to many native pests. However, in the absence of fire, they are strongly reduced and are more susceptible to nonnative pest outbreaks (Harvey 1994). This shift from more tolerant species has also reduced nutrient cycling within the surface organic matter and mineral soil (Harvey 1994). Typically, more carbon is held in aboveground biomass when there is a compositional shift toward less tolerant species, and this can result in more plant stress when available moisture is low (McDonald 1990).

Mechanical Soil Disturbance—

Soil displacement (removal of surface organic and mineral soil) is most often measured by the amount of forest floor removed. Loss of surface OM either by equipment or through accelerated erosion may produce detrimental changes if it is moved off site, is unavailable to tree roots, or if mineral soil removal results in exposing subsoil horizons. However, careful placement of harvesting and yarding layout in ground-based units could mitigate some detrimental displacement. For instance, McIver and others (2003) noted that when displacement along the edge of trails is included (i.e., displacement caused by the harvester moving close to trails to cut logs) in the inventory, displacement can be extremely variable (5 to 43 percent). However, if soil displacement along trail edges was excluded from the site inventory, displacement was no greater than USDA Forest Service guidelines of 15 percent. The report by McIver and others (2003) and the information in Curran and others (2005) both stress that it is imperative to have uniform terms for describing soil disturbance to improve our techniques for tracking the consequences of forest practices on soil productivity and forest health.

Often soil erosion is not a significant problem on slopes that have some soil cover. Using information about slope, amount of the hillside with some soil cover, and local precipitation values, Page-Dumroese and others (2000) noted that in many cases, soils with at least 50-percent soil cover did not produce more than 2 to 4 mg ha⁻¹ of soil erosion. In some cases, removal of soil from the upper slope to somewhere downslope may reduce upslope productivity, but increase downslope productivity. However, if soil is moved off site, productivity is reduced permanently (Elliot and others 1998). Both onsite and offsite soil movement results in lower soil productivity for part of the slope because of loss of nutrients, water-holding capacity, and rooting depth, and it may also impair forest health on that portion of the landscape. Combining soil-cover loss with compaction can accelerate erosion rates above the natural soil formation rates (Elliot and others 1998). Soil loss is only one problem associated with accelerated erosion. Often N, C, and cation exchange capacity are also moved offsite with the moving soil (Page-Dumroese and others 2000). Erosion rates are

usually highest immediately after soil is disturbed mechanically or as a result of fire (Robichaud and Brown 1999).

Minimizing soil compaction during harvesting and mechanical site preparation operations on forested lands is critical for maintaining the productive capacity of a site (Powers and others 2005). Compaction increases soil bulk density and soil strength, decreases water infiltration and aeration porosity, restricts root growth, increases surface runoff and erosion, and alters heat flux (Greacen and Sands 1980, Williamson and Neilsen 2000). These changes can lead to substantial declines in tree growth and forest health (Froehlich and others 1986, Gomez and others 2002) or, conversely, have little impact (Powers and others 2005). Significant changes in soil physical properties occur more often on fine-textured soils than on coarse-textured soils (Page-Dumroese and others 2006), and knowing basic site conditions like texture and soil moisture content along with designating skid trails or providing operator training may help reduce undesirable soil conditions and maintain long-term productivity (Quesnel and Curran 2000).

Most forests are dependent on a variety of biological processes to regulate nutrients and cycle organic matter. For instance, forest diseases such as *Armillaria* and *Annosus* root diseases are a key ecosystem process to recycle carbon (Harvey 1994), but they can also expand to epidemic proportions if conditions are favorable. Trees not adapted to a site, wounded during thinning operations, growing on compacted soils, or in areas of disturbed hydraulic function, are at risk from both disease infection (Goheen and Otrosina 1998, Wiensczyk and others 1997) and insect outbreaks (Larsson and others 1983). In a study in loblolly pine (*Pinus taeda*) plantations, *Annosum* was positively correlated with areas of higher bulk density that had stressed numerous trees (Alexander and others 1975). Low vigor, stressed forest stands are also susceptible to insect attacks (Larsson and others 1983). The recent outbreak of mountain pine beetle (*Dendroctonus ponderosae*) in British Columbia, Canada, and the Pacific Northwest of the United States have caused significant changes in water flow, soil moisture, and groundwater levels in many forest stands (Uunila and others 2006). These changes in water abundance and timing can affect the health of subsequent stands by changing annual

water yields, times of low flow and peak flows (Uunila and others 2006).

Acidic Deposition and Forest Health

The risks posed by atmospheric deposition to forest health are complex because of distinct regional patterns of deposition and temporal variations in air quality (McLaughlin and Percy 1999). However, air pollution stresses in North American forests and their impact on insect and disease incidence are a major consideration in both the United States and Canada (McLaughlin and Percy 1999), particularly in the eastern part of North America. One potential threat from changes in air quality on many forested lands is the altered nutrient balance and soil acidity (Adams and others 2000), which then affect insect and disease outbreaks. Soil impacts and threats to forest health can be caused by nitrogen (N) saturation (Aber and others 1989), depletion of basic cations due to increased leaching (Federer and others 1989), and altering nutrient availability (Robarge and Johnson 1992). Combined, all these nutrient stresses may cause long-term declines in site productivity (Likens and others 1996) and may increase the number and severity of disease or insect outbreaks (McLaughlin and Percy 1999). Soil processes altered by deposition of air pollutants also alter the way forests grow and respond to biotic and abiotic stresses within their regional environments. The potential for large-scale forest health changes becomes greater over time (McLaughlin and Percy 1999).

Monitoring

Current soil monitoring efforts primarily address changes in soil quality through measures of compaction, pH, water infiltration, hydrologic function, water availability, and plant-available nutrients, but do not address ecological function of a site (van Bruggen and Semenov 2000). Ecological function may be difficult to assess because of season, management, or climate variables. Potential approaches to assess ecological function can include techniques that extract DNA to determine microbial species composition (Zhou and others 1996), determine substrate utilization (i.e., Biolog) to trace microbial communities (Bossio and Scow 1995), or decomposition of a wood substrate (Jurgensen and

others 2006). In addition, levels of root pathogens or disease suppression organisms could also be considered as potential bioindicators of soil health (Hoeper and Alabouvette 1996, Visser and Parkinson 1992).

In North America, several forest monitoring programs, [i.e., Forest Health Monitoring (FHM, U.S.A.), Acid Rain National Early Warning System (ARNEWS, Canada), North American Maple Project (NAMP, joint United States and Canada), Forest Inventory and Analysis (FIA, U.S.A.)] have been developed as a result of studies that indicated widespread changes in forest health (McLaughlin and Percy 1999). Many of these “tree” monitoring programs also include soil indicators in their protocols (O’Neill and others 2005). In addition, the Montréal Process (an international effort to develop criteria and indicators of forest sustainability) is working to provide a common framework for describing soil changes by using two important soil characteristics: significantly diminished organic matter and significant compaction (Montréal Process Working Group 1997). The USDA Forest Service is mandated to monitor management impacts on soil productivity, site resiliency, and long-term productivity (Johnson and Todd 1998, Jurgensen and others 1997, Page-Dumroese and others 2000) and is part of an international, multiagency effort to pursue correlation and common definitions of practical standards for soil disturbance and maintenance of forest productivity capacity (Curran and others 2005).

There are numerous tools available to monitor a forest site or soil for changes over time or after wildfire, prescribed fire, or mechanical disturbances. Together, forest and soil monitoring data can provide a backdrop for current conditions and trends in forest health. Successful forest and soil monitoring programs must:

- Be simple to use.
- Have repeatable methods that are useable by nonspecialists.
- Provide meaningful results.
- Comprise collaborative efforts between both scientists and various government agencies.
- Provide a compelling link between soil properties and forest growth or health.

Summary and Management Implications

Solutions to the current forest health problems are neither clearcut nor easily managed (Tiedeman and others 2000). However, maintenance of the forest floor during prescribed fire or mechanical forest treatments will help to maintain soil quality, soil health, and ultimately forest health (Fellin, 1980a, Page-Dumroese and Jurgensen 2006). The indirect influence of changing microclimatic conditions of the forest floor will depend on the kinds of site treatment and the condition of the previous stand. Overcrowded and fire-suppressed stands will likely differ in their response to treatment than more open-grown stands. Similarly, stands with seral species will respond differently to nutrient, water, insect, and disease stressors than those stands occupied by less tolerant, narrowly adapted species (Harvey 1994). Forest health can be successfully managed if we shift the focus from just “tree health” to entire ecosystem health. Current levels of pest problems are not due only to a single organism, but to the complex of environmental conditions and stand histories that determine stand resilience to pest outbreaks. Healthy and resilient forest soils can also help limit the extent of forest pest problems.

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Assessing the Threat That Anthropogenic Calcium Depletion Poses to Forest Health and Productivity

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Abstract

Growing evidence from around the globe indicates that anthropogenic factors including pollution-induced acidification, associated aluminum mobility, and nitrogen saturation are disrupting natural nutrient cycles and depleting base cations from forest ecosystems. Although cation depletion can have varied and interacting influences on ecosystem function, it is the loss of calcium (Ca) that may be particularly limiting to tree health and productivity. Calcium plays unique roles in plant cell function, including environmental signal transduction processes that allow cells to sense and respond to stress. Considering this, Ca depletion could impair plant response systems and predispose trees to reduced growth and increased decline. Controlled experiments with red spruce (*Picea rubens* Sarg.) and other tree species provide mechanistic support for the hypothesis that Ca deficiencies predispose trees to decline. Importantly, several examples of species declines in the field also suggest that injury is often greater when Ca depletion and stress exposure co-occur.

Connections between contemporary species declines and Ca depletion highlight the need for monitoring forests for indicators of change, including Ca loss. Direct measures of soil and plant Ca concentrations provide one traditional means of assessing the Ca status within forests. Although these measures are often valuable, substantial variation among soils and species and a lack of comparative historical data provide obstacles to the use of these measures for evaluating Ca depletion across the landscape. An alternative approach for assessing Ca depletion is to model critical loads and exceedances of pollutant additions that lead to

net losses in Ca pools and likely disrupt ecosystem Ca cycles within forests. For example, spatial associations of Ca cycling and loss to broad-scale data on forest health and productivity were recently conducted for portions of the Northeastern United States. A steady-state ecosystem process model was coupled to extensive spatial databases and used to generate maps identifying forest areas likely to experience Ca depletion. Sustainable Ca supplies in forest ecosystems are functions of forest type, timber extraction intensity, prior land use, atmospheric deposition rates, and site factors including climate, hydrology, soil mineral type and weathering rates. Considering the unique vulnerability of Ca to leaching loss and its vital role in supporting tree stress response systems, the model focused on how changes in Ca pools may influence forest health conditions. Initial comparisons within New England indicated that the model-based Ca deficiency metric was a good predictor of field-based indicators of forest health and productivity. Models such as this show promise for evaluating the threat Ca depletion poses to forest health and productivity in an integrated and spatially explicit manner in North America. This approach has already proven valuable to policymakers and managers in Europe when evaluating alternative pollution reduction or mitigation options.

Keywords: Anthropogenic depletion, calcium, critical loads, stress response, sustainable nutrient supply, tree health.

Anthropogenic Cation Depletion

Cations are naturally occurring, positively charged elements that are important constituents of soils and surface waters and play unique and critical roles in biological systems. Among many functions, cations serve as important co-factors influencing the activity of biomolecules, act to modify charge balances within cells and organelles, and serve as signaling agents that help regulate cell physiology (Buchanan and others 2000, Marschner 2002). In forested ecosystems, the presence and availability of cations is governed through the interplay of numerous natural processes, including atmospheric additions, mineral weathering, soil

formation, plant uptake and growth, forest stand dynamics, and leaching losses (Likens and others 1998). However, mounting evidence indicates that a variety of anthropogenic factors are altering biogeochemical cycles and depleting base cations such as calcium (Ca) and magnesium (Mg) from terrestrial ecosystems. Chief among these drivers of cation loss are processes directly or indirectly associated with atmospheric pollution.

Pollutant Drivers

Through industrial activity and the increased combustion of fossil fuels over the past century, humans have dramatically increased gaseous emissions of sulfur dioxide (SO_2), nitrogen oxides (NO_x), and ammonia (NH_3) and particulate emissions of acidifying compounds (Driscoll and others 2001). Recent pollution controls have reduced emissions of sulfur (S)-based compounds in Europe and North America, resulting in moderate reductions in S deposition, but there has been little change in nitrogen (N) deposition (Driscoll and others 2001, UNECE 2005). In contrast to North America and Europe, with rapid economic development and economic growth, Asia—and most notably China—has significantly increased fossil fuel combustion in recent years (Liu and Diamond 2005). As a result, emissions of SO_2 , NO_x , NH_3 , and associated compounds have increased greatly in the region (Carmichael and others 2002, Liu and Diamond 2005, Richter and others 2005). In fact, pollutant deposition of S and N compounds now affects a quarter of China's land area, making China one of the countries most influenced by these pollutants (Feng and others 2002, Jianguo and Diamond 2005).

Through the atmospheric conversions of SO_2 and NO_x to the acids H_2SO_4 (sulfuric) and HNO_3 (nitric) as well as the release of hydrogen ions (H^+) during the oxidation of ammonium (NH_4^+) by soil microbes, S- and N-based pollutants act to acidify forest systems (Driscoll and others 2001). Among other impacts, this acidification increases the leaching of base cations from soils (Kirchner and Lydersen 1995; Likens and others 1996, 1998; Schulze 1989), and enhances the availability of aluminum (Al), which reduces base cation availability for plant uptake (Cronan and Scholfield 1990, Lawrence and others 1995). In addition to the atmospheric

production of acids from pollutant constituents, N inputs can lead to N saturation (the availability of N in excess of biological demand), which can deplete cations as excess N leaches from forest soils (Aber and others 1998). It has even been hypothesized that pollution-associated climatic warming could enhance rates of natural acidifying process, further exacerbating soil cation loss (Tomlinson 1993). In addition to pollution-associated cation loss, a side effect of existing pollution controls has been the reduced emission of particulates that contain base cations such as Ca (Hedin and others 1994). Reduced inputs and increased removals of cations from forests have resulted in net depletions that have been documented in a variety of ways, including long-term changes in stream chemistry, the analysis of archived soils, and the chemical analysis of tree xylem cores.

Evidence From the Field

Long-term data of stream water chemistry at watersheds such as those at the Hubbard Brook Experimental Forest, New Hampshire, have documented changes consistent with the pollution-induced leaching of base cations from soils (Likens and others 1996, 1998). Early stream monitoring revealed an increasing flush of base cations (Ca and Mg) that paralleled SO_4^{2-} and NO_3^- concentrations—evidence that pollutant inputs were leaching stored soil cations into surface waters (Likens and others 1996, 1998). However, after 1970, mass balance calculations identified ever-reducing concentrations of cations, particularly Ca, coincident with decreases in SO_4^{2-} and NO_3^- —a pattern suggesting the depletion of available cations following long-term leaching (Likens and others 1996, 1998). The connection between cation losses and pollutant inputs was reinforced by data indicating that these same trends in stream chemistry occur in old-growth forests where the potentially confounding effects of land use disturbance (and associated acidification) were avoided (Martin and others 2000). Furthermore, European data indicate that the largest losses of Ca and Mg occur at sites with the most acid loading (Kirchner and Lydersen 1995).

Calculated reductions in soil cation storage inferred from the chemical analysis of stream water have recently

been bolstered by studies from the United States and Europe that directly measured reductions in soil Ca storage following long-term exposures to acidic deposition. Bailey and others (2005) measured the cation concentration of soils at four forested sites in the Allegheny Mountains of Pennsylvania in 1997 and compared these to data from archived soil samples from these same sites collected in 1967. At all four sites there were significant reductions in Ca and Mg concentrations and pH over the two sample periods, and, at most sites, documented losses of Ca and Mg were much larger than could be accounted for by biomass accumulation—suggesting leaching losses as a more likely cause. In a separate analysis, Lawrence and others (2005) measured the cation contents of soil samples collected in 1926, 1964, and 2001 near St. Petersburg, Russia. They found that concentrations of exchangeable Ca in the upper 30 cm of soil decreased about tenfold from 1926 to 1964 but remained stable thereafter. In contrast, exchangeable Al showed a small decrease in the upper 10 cm of soil from 1926 to 1964, but a tenfold increase in the upper 30 cm from 1964 to 2001. They interpreted these results as reflecting a two-stage acidification process: (1) from 1926 to 1964 when inputs of acidity were neutralized by the replacement of exchangeable Ca by H, and (2) from 1964 to 2001 when the neutralization of continued acidic inputs shifted from cation exchange to weathering of solid phase Al (Lawrence and others 2005). Here, too, changes in soil Ca concentrations were not attributable to biomass accumulation of Ca, but appeared better related to pollution-induced soil Ca depletion.

Consistent with measured reductions in soil Ca, several studies have noted reduced Ca concentrations in the stemwood of trees following the advent of elevated pollution loading (Bondietti and others 1990, Likens and others 1996, Shortle and others 1995). An initial increase in Ca concentration is often noted within wood for the decades with the greatest increases in acidic deposition that likely mobilized soil cations, increasing their availability for root uptake and leaching loss (e.g., Shortle and others 1995). However, the reduction in stemwood Ca in recent decades may better reflect the long-term depletion of Ca from soils (Shortle and others 1995). Importantly, reductions in Ca concentrations

within wood also suggest that pollution-induced changes in soil Ca levels are being transferred to living organisms.

Potential Contributions from Harvesting

In addition to pollution-associated depletion, tree harvests have the potential to exacerbate cation depletion within forests if they contribute to net cation losses that exceed long-term inputs (Adams 1999, Federer and others 1989, Huntington 2000, Mann and others 1988, Nykvist 2000). Sequestration in aboveground woody biomass is an important cation sink within forest systems (Federer and others 1989, Mann and others 1988), and this is particularly true for Ca, which is highly concentrated in woody cell walls (Marschner 2002). Because of this, tree harvests can lead to the disproportionate removals of Ca relative to other cations (e.g., Adams 1999, Federer and others 1989). Harvests can also affect nutrient cycling through increased site acidification and leaching (Federer and others 1989), and reduced stocking following harvest may diminish stand-level transpiration and associated Ca uptake, further promoting Ca loss via leaching (Hornbeck and others 1993). In addition, varying methods of harvest can differentially alter Ca loss. For example, in one study, whole-tree (stems and branches) harvests removed up to 530 kg/ha, whereas sawtimber sales (bole wood only) removed about 442 kg/ha (Mann and others 1988). The frequency of tree harvest may also influence overall cation removal. Calculations from one study estimated a 15-percent loss of Ca from leaching even with no harvest, a 28-percent loss of Ca with one harvest (at 80 years), and a 41-percent Ca loss for an equal-intensity harvest performed in two stages: once at 40 years and once at 80 years (Adams 1999).

Calcium Depletion as a Biologically Unique Threat

Although the depletion of any of the essential base cations can have varied and interacting influences on ecosystem function, the loss of Ca may be particularly limiting to tree health because the unique distribution and physiology of Ca suggests that the depletion of this key cation could specifically weaken plant stress response systems and predispose trees to decline.

Calcium Distribution and Physiology

In contrast to many cations, Ca is highly compartmentalized within plant cells and tissues, and this partitioning is a defining characteristic of its physiological function. Although Ca is an essential micronutrient, it is toxic in its free form within the cell cytoplasm because it precipitates with inorganic phosphate (e.g., Bush 1995, Knight 2000). Thus, in order to assure phosphate availability for energy metabolism and other essential processes, Ca is actively pumped from the cytoplasm and is sequestered in inaccessible locations and chemical forms, including insoluble oxalate crystals outside the plasma membrane (Fink 1991). Because Ca can only exist in very low concentration in the cytoplasm, it is functionally immobile in the phloem (which relies on cytoplasmic transport). Thus, unique to other cations, Ca cannot be redistributed within plants to overcome localized deficiencies.

Localized concentrations of Ca support at least two important functions: (1) they add to the structural stability of cell walls and membranes, and (2) labile Ca is a key constituent in the pathway that allows cells to sense and respond to environmental stimuli and change (Marschner 2002). This second function appears particularly relevant to tree health concerns related to Ca depletion. Ca serves as an important second messenger in the perception and transduction of environmental and stress signals (Bush 1995, Pandey and others 2000, Roos 2000, Sanders and others 1999). Because extremely little free Ca exists in the cytoplasm of cells, environmental stimuli that temporarily alter the permeability of the plasma membrane allow labile Ca to flow into cells along a steep concentration gradient (Sanders and others 1999). Once in the cytoplasm, Ca quickly binds to Ca-specific proteins such as calmodulin, which then initiate a chain of physiological modifications (e.g., changes in enzyme activity, gene transcription, etc.) that help cells adjust to the environmental conditions that triggered the response cascade. This entry of Ca into the cytoplasm acts as a messenger of environmental information for cells and appears to be an essential first step in triggering a wide range of physiological responses needed by plants to successfully adjust to environmental change or defend against pests and pathogens. Numerous independently

conducted studies have concluded that Ca plays a critical message perception and transduction role in response to an array of environmental stresses, including low temperature (DeHayes and others 1997, 1999; Monroy and others 1993), drought (Sheen 1996), fungal infections (Hebe and others 1999), and insect infestations (McLaughlin and Wimmer 1999).

Implications to Forest Health

Given the fundamental role Ca plays in plant stress response systems, biological Ca depletion could create a scenario analogous to the suppression of animal immune systems (Schaberg and others 2001). For example, there are numerous circumstances (e.g., HIV infection, chemotherapy treatment, etc.) that impair the normal function of human immune systems. An immuno-compromised person may appear, feel, and ostensibly function as if they were healthy. Nonetheless, when exposed to a disease agent, they can experience declines in health that are exaggeratedly large relative to a person with a fully functioning immune system. In this same way, it is possible that depletions of biologically available Ca could suppress the ability of plants to adequately sense and respond to changes in their surroundings and make them more vulnerable to decline (Figure 1). This suppression would predispose plants to disproportionate decline following exposure to perhaps even normal levels of stress (e.g., pathogens or drought) that would otherwise pose no catastrophic threat if biological response systems were fully functional. Importantly, under this scenario plants might initially appear to be normal and healthy even though their biological response systems were compromised (Schaberg and others 2001).

Experimental Evidence of Tree Health Impacts

Although based on basic understandings of the distribution and physiology of Ca in plants, experimental evidence that Ca deficiencies could reduce stress tolerance in trees has only recently surfaced. This evidence was first documented for the well-studied phenomenon of winter injury in red spruce, but was later shown to be relevant to other tree species and stresses other than freezing injury.

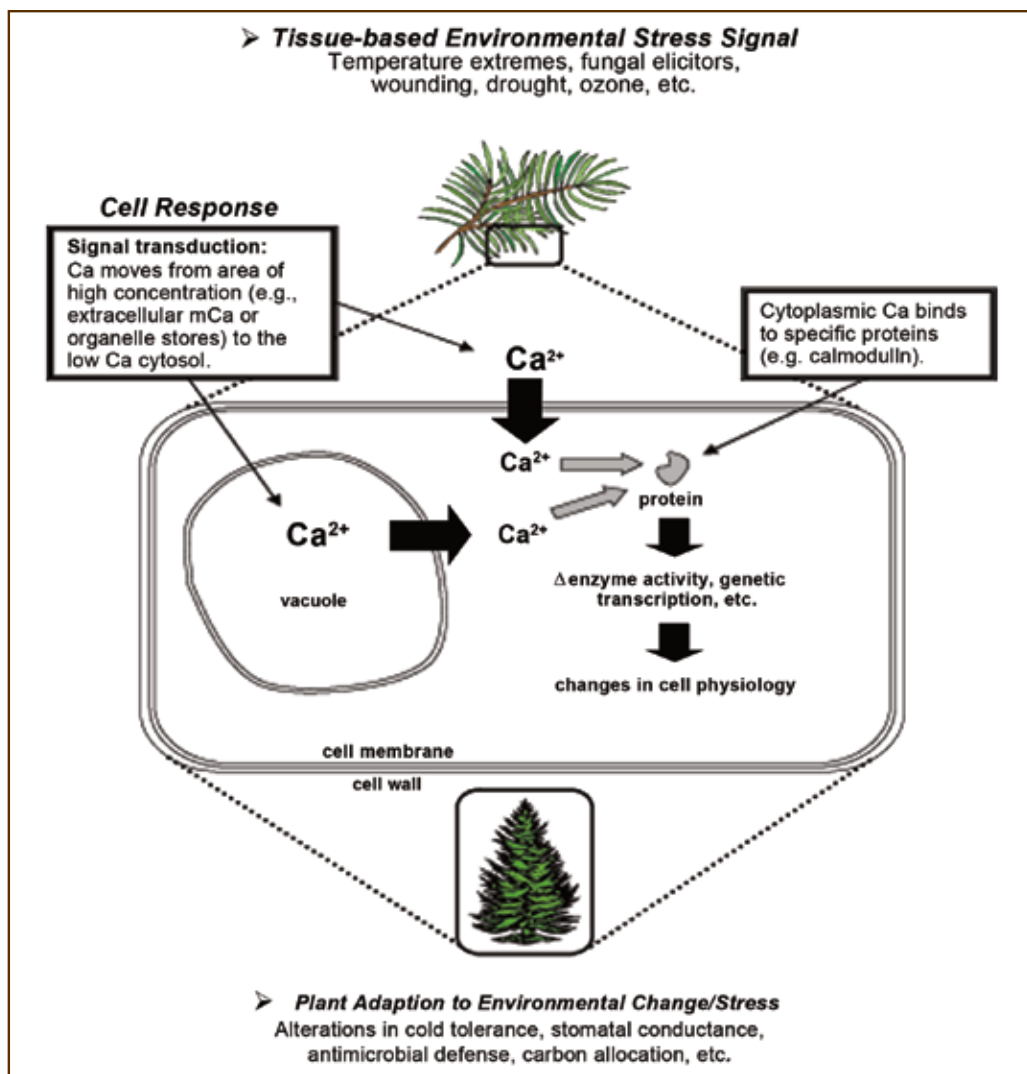


Figure 1—Schematic representation of how biological calcium (Ca) depletion may suppress plant stress response systems and predispose trees to decline. Environmental stimuli (e.g., a stress event) triggers the movement of Ca from areas of high concentration (i.e., extra-cellular and organelle pools) into the cytoplasm where Ca concentrations are 100 to 1,000 times lower. Once in the cytoplasm, Ca can bind to and activate Ca-specific protein complexes that interact with other cellular components (existing enzymes, DNA, etc.) to modify cell physiology in response to the instigating environmental cue. Depletion of biologically labile Ca may perturb signal transduction and diminish the ability of plants to sense and respond to environmental change/stress (Schaberg and others 2001).

Mechanism of Red Spruce Winter Injury

Red spruce winter injury is the reddening and mortality of the foliage in late winter followed by its abscission in late spring (DeHayes 1992). Injury is caused by freezing and is likely the result of various stresses, including low temperatures (DeHayes and others 1990), freeze-thaw cycles (Hadley and Amundson 1992, Lund and Livingston 1998),

and rapid freezing (Perkins and Adams 1995). The current-year foliage of red spruce is more vulnerable to injury than older foliage age classes or foliage from sympatric species because it is less cold tolerant (DeHayes and others 1990). In addition, certain anthropogenic inputs such as acidic or prolonged N deposition can further reduce foliar cold tolerance and increase the risk of freezing injury (Schaberg

and DeHayes 2000). Heavy foliar loss and potential bud mortality from winter injury disrupts the carbon economies of trees, leading to growth declines and potential mortality (DeHayes 1992, Lazarus and others 2004). Winter injury was linked to the widespread decline of red spruce observed in the Northeastern United States from the 1960s through the 1980s (Friedland and others 1984, Johnson 1992), and severe winter injury events persist within the region (Lazarus and others 2004).

Beginning in the late 1980s, a series of studies were published showing that acid mist exposure significantly reduced the cold tolerance of red spruce current-year foliage, increasing the risk of foliar winter injury (e.g., DeHayes and others 1991, Fowler and others 1989, Vann and others 1992). The physiological mechanism for this acid-induced reduction in cold tolerance remained unresolved, however, until a new method for measuring Ca specifically associated with cellular membranes was used in conjunction with controlled acid mist exposure experiments. Using these new methods for measuring membrane-associated Ca (mCa), a series of experiments documented that acid mists preferentially leached mCa from the outside of mesophyll cells, whereas other cations and forms of Ca were leached less, presumably because they were concentrated within the protective membrane barrier of cells (DeHayes and others 1999, Jagels and others 2002, Jiang and Jagels 1999, Schaberg and DeHayes 2000, Schaberg and others 2000). Furthermore, these studies showed that this loss of mCa destabilized membranes, depleted a source of Ca needed for stress signaling, reduced foliar cold tolerance, and predisposed trees to the secondary freezing injury responsible for decline (DeHayes and others 1999, Schaberg and DeHayes 2000, Schaberg and others 2000). Later work verified that soil-based Ca depletion initiated the same mechanistic sequence of physiological disruptions documented for foliar Ca leaching (Schaberg and others 2002).

Pertinence to Other Tree Species and Stressors

The bulk of experimental evidence elucidating the influence of Ca depletion on tree nutrition and stress response has involved winter freezing injury of red spruce. However,

recent evidence indicates that the same basic mechanism of physiological disruption documented for this species and syndrome are pertinent to other tree species and stressors. For example, Schaberg and others (2001) treated red spruce, eastern hemlock (*Tsuga Canadensis* (L.) Carr.), balsam fir (*Abies balsamea* (L.) Mill.), and eastern white pine (*Pinus strobus* L.) seedlings with acid mist and compared the nutritional and physiological responses of the newly evaluated species to those well-documented for red spruce. Although there was insufficient tissue to make all measurements on each species, results showed that acid mist reduced mCa levels (in eastern hemlock), decreased cell membrane stability (in balsam fir), and reduced foliar cold tolerance (in white pine) similar to red spruce (Schaberg and others 2001). In a separate experiment with red spruce, Borer and others (2005) examined the influence of acid mist exposure on stomatal closure following tissue desiccation—a stress response to drought that is also dependent on Ca signaling (Knight 2000). Red spruce seedlings were exposed to pH 3 or 5 mists and then measured for foliar Ca concentrations and rates of stomatal closure as foliage desiccated following shoot harvest. As with past experiments, acid mist exposure reduced the Ca available in foliage, but here the loss of Ca was also accompanied by a 15 percent slower rate of stomatal closure as tissues desiccated (Borer and others 2005). Results of experimental trials like this support the theory that anthropogenic Ca depletion could deplete biological Ca pools enough to suppress stress response systems and predispose trees to decline.

Field Evidence of Influences on Tree Health

Controlled experiments like the ones outlined above have provided valuable insights into the biological mechanism through which Ca depletion may influence tree physiology and health. However, such studies by themselves do little to inform us of the threat Ca depletion may pose to native forests. Instead, evidence from numerous field studies has supported experimental findings and implicated Ca depletion as a contributing factor in the real-world decline of tree species in the United States and abroad.

Winter Injury of Red Spruce

The role that foliar winter injury has played in red spruce decline in Northeastern United States and adjacent Canada has long been understood (see DeHayes 1992). In addition, experimental evidence has provided a detailed understanding of the mechanism through which acid deposition can deplete biologically available Ca and predispose foliage to damage (see “Mechanism of Red Spruce Winter Injury”). However, it was only following the severe winter injury event of 2003 that evidence indicated that acid deposition exposure in the field influences winter injury expression across the landscape. Lazarus and others (2006) measured the degree of foliar winter injury among dominant and codominant red spruce trees at multiple elevations (plots) at 23 sites in Vermont and adjacent States and used regression analyses to evaluate how injury varied with plot elevation, latitude, longitude, slope, and aspect. They found that injury was significantly greater in western portions of the study area, west-facing slope, and higher elevations—areas that have historically received higher levels of acidic and N deposition (Lazarus and others 2006). Although these findings support the hypothesis that acidic or N deposition or both act on a landscape scale to exacerbate winter injury, it was an ancillary evaluation that more specifically implicated Ca depletion as a modifier of injury expression in 2003. Hawley and others (2006) measured foliar nutrition and winter injury of red spruce on two watersheds at the Hubbard Brook Experimental Forest in New Hampshire: one a reference watershed that has undergone considerable Ca loss attributed to acid deposition-induced leaching (Likens and others 1996, 1998), and another watershed that was fertilized with CaSiO_3 in 1999 to replace lost Ca. Dominant and codominant red spruce on the Ca-addition watershed had significantly more Ca in their foliage and experienced about one-third the foliar injury of comparable trees on the reference watershed (Hawley and others 2006).

Sugar Maple Decline

Sugar maple (*Acer saccharum* Marsh.) decline has been documented throughout parts of the Northeastern United States and Quebec over many recent decades (Allen and others 1992a, Kelley 1988, Mader and Thompson 1969,

Wilmot and others 1995). These declines have been characterized using various measures, including crown deterioration, increased leaf chlorosis, and reduced growth. Stress factors such as drought (Payette and others 1996), freezing (Robitaille and others 1995), and insect defoliation (Allen and others 1992b) have been implicated with the decline and mortality of sugar maple. Regardless of stressor, decline has also been associated with deficiencies or imbalances of various elements including N, phosphorous (P), potassium (K), Mg, manganese (Mn), or Ca (Bernier and Brazeau 1988, Horsley and others 2000, Mader and Thompson 1969, Ouimet and Fortin 1992, Paré and Bernier 1989, Wilmot and others 1995). Although the specific elements associated with decline can differ among sites, deficiencies in Ca have been highlighted as a potential contributor to sugar maple decline in recent studies throughout the region (e.g., Ellsworth and Liu 1994, Ouimet and Camiré 1995, Wilmot and others 1996), in part because experimental additions of Ca or lime or both have been shown to reduce decline symptoms (Long and others 1997, Moore and others 2000, Wilmot and others 1995).

Consistent with these observations, Schaberg and others (2001) hypothesized that sugar maple decline may be another example of Ca depletion's influence on tree stress response systems and health (Figure 2). Variations in maple decline symptoms (crown condition and basal area growth) coinciding with differences in soil Ca status across a range of sites are consistent with this hypothesis (Schaberg and others 2006). However, as with red spruce, controlled additions of Ca at the Hubbard Brook Experimental Forest have provided a more specific test of the influence of Ca depletion on sugar maple health. For example, Juice and others (2006) compared the nutrition, reproductive success, and physiology of sugar maple seedlings on the reference and Ca-addition watersheds there. They found that seedlings on the Ca-treated watershed had higher root and foliar Ca concentrations, experienced greater survivorship, existed in greater densities, and had higher foliar chlorophyll concentrations than seedlings on the reference watershed (Juice and others 2006). Mycorrhizal colonization of seedlings was also greater in the treated than the reference watersheds (Juice and others 2006). In another study, Huggett

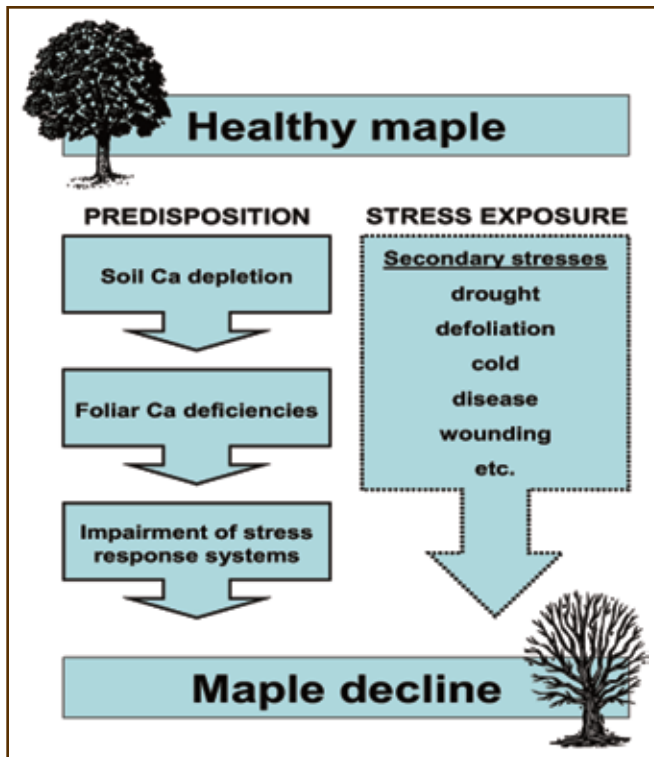


Figure 2—Proposed mechanism of sugar maple decline. This depicts how Ca deficiency may suppress stress response systems and predispose trees to decline following exposure to stress factors reported to contribute to sugar maple decline in the Eastern United States and Canada (Schaberg and others 2001).

and others (2007) surveyed and wounded forest-grown sugar maple trees in a replicated Ca manipulation study at Hubbard Brook. Similar to past studies, this study found that Ca addition increased foliar Ca levels and resulted in improved crown vigor, reduced branch dieback, and greater basal area growth among trees (Huggett and others 2007). However, new were findings that Ca addition particularly improved the growth release of trees following a severe ice storm and significantly increased stem wound closure—a capacity particularly important to a species that is regularly wounded as part of maple sugar production (Huggett and others 2007). These new findings are particularly noteworthy because they more specifically test the influence of Ca nutrition on the ability of sugar maple trees to respond to environmental change (release from competition) and stress (wounding).

Anthracnose Damage to Flowering Dogwood

Acidic deposition exposure significantly increases the susceptibility of flowering dogwood (*Cornus florida* L.) trees to injury by the fungal pathogen dogwood anthracnose (*Discula destructive* Redlin) (Anderson and others 1993, Britton and others 1996). Furthermore, controlled studies have implicated acid-induced nutrient deficiencies in this altered disease susceptibility (Britton and others 1996). Elevated disease susceptibility has also been associated with environmental conditions (e.g., shade, low temperatures, wet cool summers, etc.) that result in low transpiration rates and thereby reduce the accumulation of Ca in plants (McLaughlin and Wimmer 1999). The association of Ca deficiency and anthracnose susceptibility is also supported by an increased resistance to this disease following lime application (USDA FS 1991). Based on this and other evidence, McLaughlin and Wimmer (1999) proposed that Ca deficiencies driven by low soil Ca concentrations, low transpiration rates or accelerated foliar leaching, or both, reduce the natural resistance of dogwood to anthracnose infection.

Emerging Examples in the United States and Europe

In addition to the well-established connections between Ca depletion and tree health outlined above, new associations between the Ca status and health of trees periodically emerge—particularly in regions that experience continued pollution-induced Ca leaching. Differences in the susceptibility of eastern hemlock trees to damage by the hemlock woolly adelgid (HWA; *Adelges tsugae* Annand) in the Northeastern United States may provide an example of this. The HWA is a small, aphid-like insect that was likely introduced to the mid-Atlantic States from Asia in the 1950s and has since expanded its range and influence, devastating hemlock forests over an ever-widening portion of eastern hemlock's native range (Orwig and Foster 1998). Although it was first believed that little or no variation in susceptibility to HWA damage existed among hemlock trees (McClure 1995), recent work has shown that differences in site conditions and the presence of other stressors are associated with differential damage and decline (Orwig and Foster

1998, Sivaramakrishnan and Berlyn 1999). In particular, Pontius and others (2006) recently concluded that foliar chemistry was linked to the infestation and susceptibility of eastern hemlock to HWA. Among other evidence, results of a regional monitoring effort showed that concentrations of Ca, K, N, and P were strongly correlated with HWA densities (a driving factor in hemlock decline). From this and other findings, they hypothesized that, whereas foliar N and K concentrations may influence hemlock decline through an alteration in insect behavior due to palatability issues, Ca and P concentrations may deter severe HWA damage through an alteration in tree physiology (Pontius and others 2006). Experimental tests are needed to assess whether these changes in physiology involve Ca-induced alterations in plant stress response systems.

Evidence of acid deposition damage to forest health and productivity in Europe is anything but new or emerging. Indeed, media attention and resulting public concern about the possible connection between pollution exposures and forest death (Waldsterben) helped spur initial efforts to control acidifying pollution additions (Kakebeeke and others 2004). Research eventually identified pollution-induced cation depletion (particularly Mg and Ca) as an instigating component of forest decline (Schulze 1989). Although the mechanism through which acid deposition influences the health of European forests has not been explicitly defined, it is generally agreed that it acts as a predisposing agent, weakening forests and making them more susceptible to damage by other stresses including insect attack, extreme climate events, or storm damage (Materna and Lomský 2002, UNECE 2005). This scenario is strikingly similar to the Ca depletion and stress response suppression hypothesis developed primarily using evidence from declines in the United States (Schaberg and others 2001). In fact, a recent example of forest damage in Europe that is thought to be predisposed by pollution exposure involves the reddening and abscission of foliage of Norway spruce in the late winter, presumably due to freezing injury (Lomský and Šrámek 2002, Materna 2002). Patterns of injury (preferentially impacting the youngest foliage with the intensity and extent of damage increasing with elevation, Lomský and Šrámek 2002, Materna 2002) are identical to those documented

for red spruce winter injury (DeHayes 1992, Lazarus and others 2004), which has been mechanistically linked to Ca depletion (see “Mechanism of Red Spruce Winter Injury”).

Limitations of Current Indicators of Calcium Depletion

Evidence from laboratory experiments and a growing number of field assessments indicate that anthropogenic Ca depletion may pose a unique threat to forest health and productivity. In particular, connections between contemporary species declines and Ca depletion highlight the need to monitor forests for indicators of change, including Ca loss. Direct measures of soil and plant Ca concentrations provide one traditional means of assessing the Ca status of forests. Although often valuable, these measures alone may not support a comprehensive and practical assessment of the biological threat posed by Ca depletion across the landscape for several reasons. Chief among these are difficulties associated with the high spatial variability of Ca storage and availability within forest soils, substantial differences in Ca uptake and nutritional requirements among tree species, and a lack of historical data to serve as a reference to gauge the timing and extent of Ca depletion.

Soils

As noted in “Evidence From the Field” (p. 38), direct evidence of cation depletion is limited to a few retrospective studies that were able to use archived soil samples and contemporary measurements of the exact field locations to determine changes in soil chemical properties (Bailey and others 2005, Lawrence and others 2005). These studies, in combination with a 9-year watershed acidification study in Maine that showed sizable reductions in exchangeable Ca and Mg compared to the control watershed (Fernandez and others 2003), and long-term watershed mass balance observations (Likens and others 1996), support the theoretical basis of cation depletion due to acid deposition (Reuss 1983). However, application of these results to larger landscapes is highly problematic because of the large spatial variability of soil properties. Plant-available Ca status within soils is primarily determined by the mineralogy of the parent material from which the soil was formed and the

period of time that the rooting zone has been exposed to weathering. Thus, the underlying parent material and soil classification information are useful in identifying regions that may be at risk to Ca depletion (e.g., areas of the White Mountains in New Hampshire or the Adirondacks in New York). However, within susceptible regions, soil properties are highly variable, both vertically and horizontally, which make it challenging to conduct site-specific evaluations. This spatial variability can be dealt with by using a large sample size and sampling the soil by genetic horizon (Bailey and others 2005); however, this requires the assistance of trained soil scientists and incurs high costs for analysis. Consequently, collection of soil nutrition data to support regional assessments would be expensive and requires the expenditure of considerable time and effort. In part to overcome the inherent difficulties in assessing soil nutrition, but also considering the theoretical value of assessing living organisms when considering biological deficiency thresholds, vegetation chemistry has also been monitored to test for Ca deficiencies.

Vegetation

Ca deficiency thresholds for trees in native forests exist for some species with established nutritional vulnerabilities. For example, based on greenhouse (Swan 1971) and field studies (Borer and others 2004, DeHayes and others 1999), minimal sufficiency and deficiency Ca thresholds have been determined for red spruce foliage (1200 and 800 $\mu\text{g g}^{-1}$, respectively). The deficiency threshold is associated with reduced cold tolerance and growth and increased winter injury of trees (Borer and others 2004, DeHayes and others 1999, Swan 1971). Similarly, based on surveys of forest-grown trees, a foliar Ca deficiency threshold of about 5000 $\mu\text{g g}^{-1}$ has been established for sugar maple (e.g., see Kolb and McCormick 1993). Foliar Ca concentrations below this threshold have been associated with increases in crown dieback and reduced growth of trees (Schaberg and others 2006), reduced growth following overstory release, and impaired stem wound closure (Huggett and others 2007). However, an analysis of these thresholds reveals that threshold concentrations are not uniform among species and can differ greatly (here more than fivefold).

Obviously, not all tree species access or require Ca in equal amounts. Depending on tree rooting habits (e.g., depth of roots, possible mycorrhizal association, etc.), access to soil Ca can differ greatly. Furthermore, perhaps because Ca cycling within forests has historically been adequate to amply supply this critical nutrient, specific thresholds of Ca depletion have been developed for only a few key species (such as red spruce and sugar maple). Even for well-studied species, internal chemical sequestration of Ca (e.g., the precipitation of Ca as insoluble oxalate crystals within cell walls; Fink 1991), may mask biological Ca deficiencies and complicate the establishment of universally relevant deficiency thresholds. Indeed, discovery of the mechanism through which acid deposition reduces the cold tolerance of red spruce foliage relied on the differentiation of biologically labile Ca from total foliar pools in order to remove the confounding influence of foliar Ca sequestration (DeHayes and others 1999, Schaberg and others 2000). Thus, owing to great species-to-species variation in Ca nutrition and use, combined with questions of tissue sequestration and biological availability, foliar assessments of Ca are not universally valuable in assessing Ca deficiency. However, despite limitations, measurements of foliar Ca concentrations may have value if conducted as part of monitoring efforts to gauge spatial and temporal changes in Ca nutrition, thereby assessing trends in Ca accrual or depletion.

Lack of Historical Data

One-time measures of soil or plant Ca concentrations can provide useful information of the current status of a site, particularly if Ca concentrations are clearly aberrant relative to limited established standards. However, short of this, they provide little insight into trends in Ca availability or potential disruptions in Ca cycling. In contrast, repeated measures (and the archiving of samples to allow for reanalysis should measurement protocols change over time) provide baseline data needed to make necessary trend assessments. To date, comparatively few such databases and archives have been established. Long-term research at the Hubbard Brook and other experimental forests (e.g., Adams 1999, Likens and others 1996, 1998), as well as a few recent studies (notably Bailey and others 2005, Lapenis and others

2004, Lawrence and others 2005) highlight the unique benefit of repeated measures for detecting meaningful changes in Ca nutrition. Recognition of the value of such data sets has bolstered efforts to compile broad nutritional data sets that encompass a variety of sites, tree species, and time periods (e.g., see the foliar chemistry database compiled by the Northeastern Ecosystem Research Cooperative: <http://www.folchem.sr.unh.edu/>), and establish sample archives as described by Lapenis and others (2004) and Lawrence and others (2005). However, given practical (especially funding) limitations, even if such data sets and archives were dramatically expanded, appropriate historical data would exist for just a fraction of the land area impacted by pollution loading. Thus, alternatives for assessing the location and extent of Ca depletion across the landscape must be developed, tested, and employed.

Critical Loads as an Alternative Approach

Given the limitations of static measures of Ca status, an alternative approach has been developed that models the level of pollutant additions that will likely lead to net losses in Ca pools within forests, therefore disrupting Ca cycling and leading to reduced plant Ca availability, and the physiological consequences described above.

Description and Overview

The critical load of a pollutant is classically defined as a quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge (Nilsson and Grennfelt 1998). This definition implies some sort of measurable threshold value of ecosystem condition (termed the critical limit) that is demonstrably influenced by the pollutant(s) in question. The critical load is the pollutant deposition load, which can be shown to perturb a system such that the critical limit condition is attained. In the context of the adverse impacts of S and N deposition, many different critical limits have been proposed based on values of the Ca/Al ratio of soil water, the Ca/Al ratio of soil exchange sites, soil percentage base saturation, soil water nitrate concentrations, and soil and plant carbon (C)/N ratios. The relationship between

pollutant loading and the value of the ecosystem condition being evaluated can be established empirically or by mechanistic modeling. For mechanistic modeling, both steady-state and dynamic-process models have been employed. Important modeling considerations are appropriate data availability at the spatial scale desired and the existence or lack of data that can be used for dynamic model calibration and evaluation.

Examples from Europe and Canada

In 1984, a protocol to the UNECE 1979 Convention on Long-Range Transboundary Air Pollution was established providing for long-term financing of the Cooperative Program for Monitoring and Evaluation of the Long-Range Transmission of Air-Pollutants in EUROPE (EMEP) (<http://www.unece.org/env/lrtap/welcome.html>). This protocol permitted the development of the scientific and administrative structure necessary for coordinated evaluation of critical loads and exceedances for acidifying pollutants in Europe. For over a decade, European countries have used the critical loads concept to set emissions reduction targets for S and N under the UNECE Convention on Long-Range Transboundary Air Pollution (Posch and others 1995). Initially, the critical loads for S and N were developed for effects on freshwater aquatic ecosystems. This work was rapidly expanded to include the developing understanding of the role of S and N deposition in producing European forest declines.

Canada followed Europe's lead and developed critical loads modeling programs first for aquatic and then forest ecosystems (Ouinmet and others 2001, 2006) that have been incorporated into the periodic Canadian Acid Rain Assessments (Morrison and others 2005). Until recently, few have attempted to apply the critical loads approach to large forest landscapes in the United States.

Application to New England

One approach for applying the concept of critical loads to assessing the sensitivity of forests to Ca depletion grew out of the work of the Conference of New England Governors and Eastern Canadian Premiers (NEG/ECP). In 1998, the NEG/ECP developed an Acid Rain Action Plan that

called for (among 44 action items) a regional assessment of the sensitivity of Northeastern North American forests to current and projected S and N emissions levels. This assessment was intended to identify the location and extent of forested areas most sensitive to continued S and N deposition and estimate deposition rates required to maintain forest health and productivity under current rates of forest resource utilization.

Conference of New England Governors and Eastern Canadian Premiers

Considering the unique vulnerability of Ca to leaching loss and its vital role in supporting tree stress response systems, the model focuses on how atmospherically deposited S and N act to produce changes in Ca pools that may influence forest health conditions. Review of the literature (discussed above) indicated that the most appropriate critical limit that could be modeled as a function of S and N deposition was whether an ecosystem could maintain a sustainable supply of the nutrient base cations Ca, Mg, and K or if the system was experiencing long-term depletion of these elements. Systems in a chronic state of cation depletion will eventually exhibit the Ca deficiency-related problems discussed above. Because the goal of the assessment was to provide estimates of the forest area potentially impaired if pollution remained at current S and N deposition levels, a steady-state modeling approach was selected. It was determined that adequate estimates of the parameters required for steady-state modeling could be developed regionwide at an appropriate spatial scale, whereas the data requirements for dynamic modeling could only be met in a few locations. It is anticipated that the results of the steady-state modeling assessment will direct future data collection efforts to high-value, high-risk areas where the cost of data collection for dynamic modeling may be justified.

A steady-state ecosystem process model was coupled to extensive spatial databases and used to generate maps identifying forest areas likely to experience Ca depletion (Miller 2005, 2006; Ouimet and others 2006). Sustainable Ca supplies in forest ecosystems are functions of forest type, timber extraction intensity, prior land use, atmospheric

deposition rates, and site factors including climate, hydrology, and soil mineral weathering rates (NEG/ECP 2001). The ecosystem model and several submodels simulate these processes. The crucial determinant of an ecosystem's ability to tolerate S and N deposition without declining Ca supplies is the rate at which primary minerals (e.g., hornblende, plagioclase, calcite) chemically decompose, liberating the nutrient cations Ca^{2+} , Mg^{2+} , and K^{+} to plant-available pools and, thus, replenishing nutrients lost via timber removals and acid-induced leaching. A geochemical model based on the work of Sverdrup and Warfvinge (1993) was used to estimate the weathering rate of primary minerals. Considerable field and modeling efforts were required to develop the spatial data layers needed to apply this model to the New England region. The annual demand for nutrients required to regrow the biomass exported via harvesting was estimated from timber extraction rates and wood-nutrient content. This information was generally available for the New England States from combinations of State and Federal sources. Atmospheric deposition of S, N, chloride (Cl), Ca, Mg, sodium (Na), and K was estimated for a 5-year period (1999-2003) in order to provide some smoothing of year-to-year variations in climate and patterns of atmospheric transport. Total deposition, including precipitation, cloud droplet interception, and dry deposition, was estimated using atmospheric chemistry data from the US NADP, CASTNet, and NOAA-AirMon deposition monitoring networks and Ecosystems Research Group, Ltd.'s High-Resolution Deposition Model (Miller 2000, Miller and others 2005, NEG/ECP 2001).

Although only results for Vermont and New Hampshire are presented here as examples, this assessment methodology has been applied to all of New England. Critical loads of S plus N ranged widely in New Hampshire and Vermont (0 to 21 $\text{keq ha}^{-1} \text{y}^{-1}$) as a result of the diverse geology and climate of the region (Figure 3). Areas of Ca-rich rocks and soil materials scattered throughout the region support the highest critical loads, often in excess of 3 $\text{keq ha}^{-1} \text{y}^{-1}$. The lowest critical loads were found primarily in northern New Hampshire where soils are developed in thin and patchy tills derived from base-poor rocks. The range of S deposition was between 3.2 and 18.9 (average 5.1) $\text{kg ha}^{-1} \text{y}^{-1}$ and N

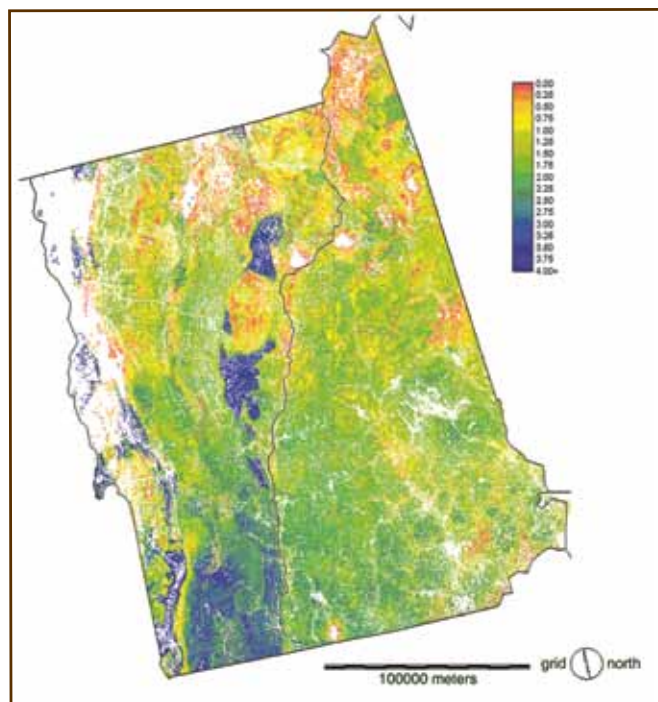


Figure 3—Critical loads of sulfur plus nitrogen to upland forests; S + N atmospheric deposition rates higher than the critical load result in greater exports of nutrient cations (Ca^{2+} , Mg^{2+} , K^{+}) than inputs of these nutrients, leading to the eventual deterioration of soil fertility, forest health, and forest productivity. Critical loads are expressed in kilo-equivalents per hectare per year in this map of New Hampshire and Vermont; N deposition includes both ammonium + nitrate forms. White areas depict locations with nonforested land or water.

deposition (ammonium + nitrate) ranged between 3.3 and 25.2 (average 8.4) $\text{kg ha}^{-1} \text{y}^{-1}$, producing an aggregate acidifying and nutrient-leaching potential of 0.43 to 2.7 (average 0.92) $\text{keq ha}^{-1} \text{y}^{-1}$ (Figure 4). The highest elevation areas received the highest S + N deposition from orographically enhanced precipitation and cloud water inputs. Deposition was also high in the southern and western parts of the region owing to proximity to emission sources.

Using critical load and atmospheric deposition estimates, a deposition index can be calculated to help evaluate the relative risk for ecosystem health problems resulting from Ca limitation (Figure 5). The deposition index is calculated as the ecosystem critical load minus the atmospheric deposition. In this index, positive values reflect the capacity of a forest ecosystem to tolerate additional acidic deposition, whereas negative values correspond to the reduction in S and N deposition required to eliminate

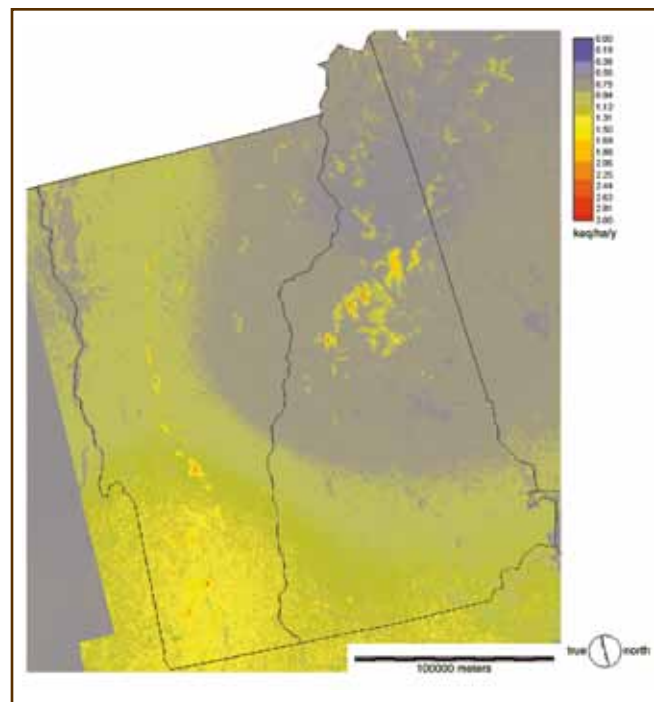


Figure 4—Average annual atmospheric deposition of sulfur and nitrogen (1999-2003). Total deposition (particle + SO_2 + precipitation + cloud water) expressed in terms of kilo-equivalents of charge per hectare per year in this map centered on New Hampshire and Vermont; N deposition includes both ammonium + nitrate forms. This represents the total base-neutralizing and cation-leaching power of S and N atmospheric deposition.

present or deter the development of nutrient limitations. Atmospheric deposition of S and N during 1999-2003 exceeded the critical load in approximately 18 percent of the forested area of NH and 30 percent of the forested area in VT (Figure 5 and Figure 6). Critical loads were frequently exceeded where deposition was moderate (Northeast) to high (South) and where critical loads are low. An additional 10 percent of the forested area in each State experienced deposition rates during 1999-2003 that were less than 0.2 $\text{keq ha}^{-1} \text{y}^{-1}$ below the critical load. Some locations within these areas with shallower soils and more intense harvesting than the average values used in this assessment are likely to also be at risk of Ca depletion.

Forest tree species occupy different portions of the landscape as a function of climate, soil conditions, and land use history. This distribution results in some types of forests being more severely impacted than others by the nutrient

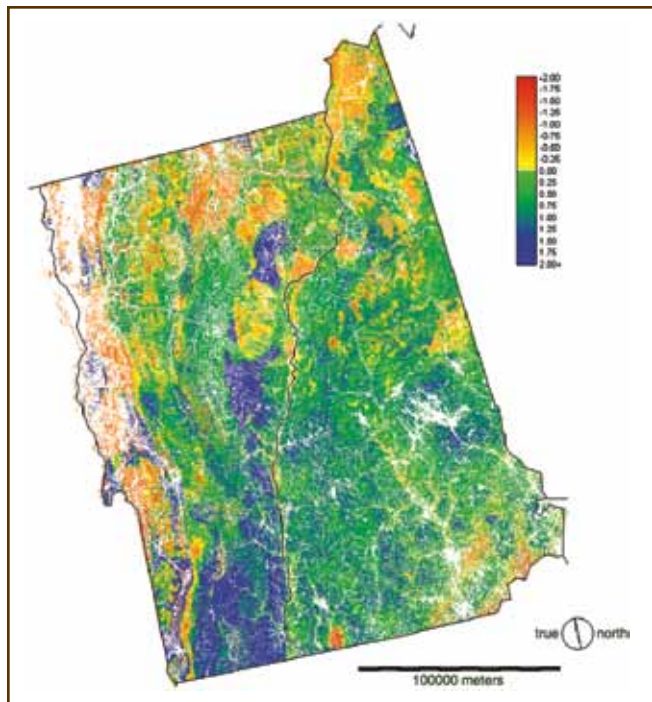


Figure 5—The deposition index for atmospheric sulfur and nitrogen deposition (1999–2003) with respect to Ca depletion. Positive values of the deposition index reflect the capacity of a forest ecosystem to tolerate additional acidic deposition. Negative values of the index correspond to the reduction in S and N deposition required to eliminate present or deter the development of nutrient limitations. Red-orange-yellow areas in this map of New Hampshire and Vermont indicate current S and N atmospheric deposition rates greater than the critical load. The deposition index is expressed in terms of kilo-equivalents of charge per hectare per year. N deposition includes both ammonium + nitrate forms. White areas represent areas with nonforested land or water.

cation depletion caused by S + N deposition. For example, critical loads are exceeded in 49.6 percent of New Hampshire's central hardwood forests, but in just 2 percent of the State's northern hardwood forests. This discrepancy exists because the central hardwood forests (6.1 percent of total forest area) tend to occur more frequently on poor sandy soils. The northern hardwood forest (18.9 percent of forest area) occurs on somewhat richer sites. Stands dominated by sugar maple (11.4 percent of forest area) occupy mid-elevation sites and have the highest Ca requirement of the northern hardwood forest variants. We estimate that the critical load is exceeded in 39.8 percent (88 167 ha) of New Hampshire's sugar maple stands, and deposition is within 10 percent of the critical load in an additional 3.6 percent (7826 ha).

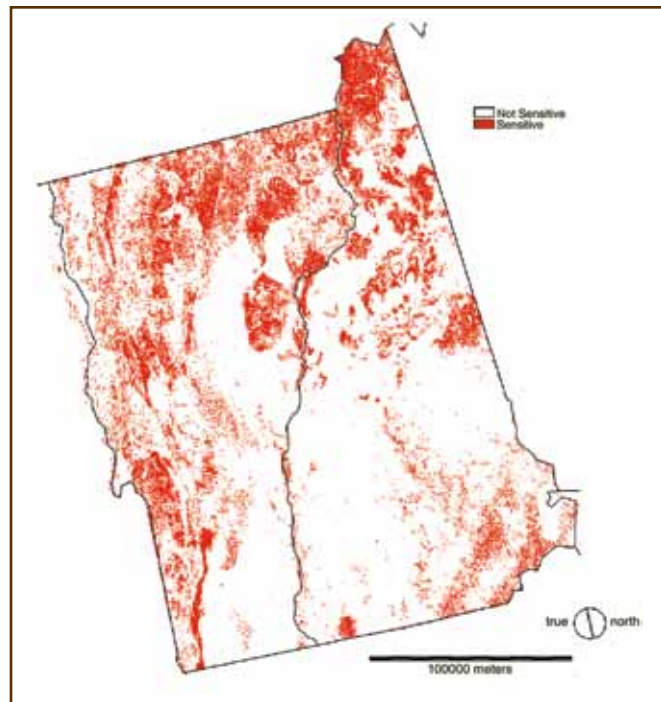


Figure 6—Areas with sulfur plus nitrogen deposition in excess of the critical load. Red-shade shows areas in New Hampshire and Vermont where S+N deposition exceeded the critical load, and forests would be likely to experience Ca depletion under 1999–2003 deposition rates.

Relationships to Forest Health

The model-based nutrient deficiency metric appears to be a good predictor of independent, on-the-ground indicators of current forest health and productivity. For oak and pine forests in Massachusetts, tree height and canopy transparency were significantly related to foliar Ca levels and to the modeled rate of base cation depletion. Canopies were more transparent and grew to lower heights where base cation depletion rates were higher, and foliar Ca status was lower. At these same sites, root-zone soil base saturation, pH, and Ca/Al ratio were correlated with modeled base cation depletion rates. A separate evaluation also showed promising results: a comparison of model results with multiple-year aerial surveys of forest damage in Vermont indicated that both the frequency of damage and size of damaged areas were related to modeled base cation depletion. Forests in Vermont also showed greater canopy transparency and higher percentages of trees with chronic dieback where modeled base cation depletion rates were higher.

The critical loads framework can also be used to assess the recovery of ecosystems owing to changes in pollution levels and management strategies. Miller (2005) estimated that the cation depletion that likely previously occurred in 126 000 ha of forest in New Hampshire and 133 775 ha in Vermont was probably reversed by the 36-percent reduction in S deposition that occurred in the region from the late 1990s to the early 2000s. These forest areas should be experiencing recovery of base cation pools.

Development and Relevance of Future Models

Initial results like those described for New Hampshire and Vermont indicate that models of critical loads and forest sensitivity show real promise in helping to empirically define the threat posed to forests in an integrated and spatially explicit manner. Risk assessment will be assisted by the expansion and refinement of existing data sets and models, as well as the development of more detailed dynamic models applicable to specific, high-interest sites. Ultimately, refined models should help policymakers and managers improve their analyses and recommendations regarding possible options to prevent or mitigate anthropogenic Ca depletion across the forested landscape.

Steady State versus Dynamic Models

Steady-state models are most appropriate for regional studies because of their lower data requirements. Steady-state models can provide reasonable estimates of long-term average Ca depletion rates covering large areas at a reasonable cost. These models can provide estimates of the extent, location, and average intensity of base cation depletion. Where additional measures of base cation pools are available or can be obtained, comparison of steady-state depletion rates with base cation pools can provide first-order estimates of the time to ecosystem degradation or recovery (Miller 2005).

Dynamic models offer important insight into how and when ecosystem changes will occur in response to changes in pollution loadings and management activities. However, dynamic models are data intensive and require calibration and evaluation data that take years to obtain at considerable

expense. Calibrated dynamic models are typically only applicable to the specific locations for which they were calibrated. Thus, dynamic models seem most appropriate for assessment of high-value or high-interest sites where the additional cost of data gathering can be justified. Dynamic models may be used opportunistically where suitable pre-existing data exist; however, great care must be taken to understand how representative such opportunistic studies are of the broader landscape.

Policy Implications

First and foremost, modeling approaches such as those involving the estimation of steady-state or dynamic critical loads could be of benefit to policymakers when evaluating the possible consequences of various pollution mitigation options as they relate to Ca cycling, forest health, and productivity. Analyses of trends in data from the field have been used to evaluate whether implementation of Phase I of the Clean Air Act Amendments of 1990 resulted in sufficient reductions in pollutant inputs to protect environmental resources (e.g., Butler and others 2001, Likens and others 2001). However important, this retrospective approach does little to inform decisionmakers of the possible future or long-term impacts of pollutant deposition. In contrast, a noted benefit of critical loads and associated forest sensitivity modeling is the flexibility provided to project trends forward and estimate the location and extent (steady state) and timing (dynamic models) of pollution-induced impacts. Furthermore, these models can be used to estimate differences in pollution loads and impacts among various pollution reduction plans, thereby allowing for a more detailed and reasoned analysis among policy alternatives.

Management Implications

Results from modeling efforts may also inform resource managers regarding the need for possible intervention and protective strategies to reduce or mitigate the impacts of air pollution. For example, forest sensitivity maps could be used to identify forest tracts where N or S loads remain above critical loads and where liming may be needed to sustain the health and productivity of forests. In recent decades, lime has been routinely applied to forests in

Europe to counter pollution-induced losses of base cations (e.g., Materna 2002, Meiwes 1995, Saarsalmi and Mälkönen 2001), and experimental trials in the United States have shown reductions in decline symptoms when sugar maple forests received lime applications (Long and others 1997, Moore and others 2000, Wilmot and others 1996). However, lime addition can be costly and difficult to apply over forest landscapes. Modeling results could help identify areas of particular vulnerability to Ca depletion and better target regions where interventions such as lime addition may be most beneficial and cost effective. Another management option is to modify harvesting intensity in particularly vulnerable areas by extending rotation lengths or limiting the biomass removed (stem only vs. whole-tree harvesting). A reduction in harvesting intensity could be used to lower the export of Ca from the forest during the time required for air pollution reductions to be achieved.

Worldwide Relevance

Concerns about the influence of Ca depletion on forest health exist for industrialized regions around the world including Europe, Eastern North America, and increasingly China (Driscoll and others 2001, Duan and others 2000, Kirchner and Lydersen 1995, Likens and others 1996, Schulze 1989, Tao and Feng 2000). Growing experimental evidence and examples from the field indicate that the threat posed to forest ecosystems from anthropogenic Ca depletion is real and potentially widespread. Knowledge of the influence of pollution loading on the cation pools that sustain forest health and productivity provides further scientific grounding and impetus for policymakers to modify existing pollution control measures. In addition, an increased recognition of the potential consequences of Ca depletion has functional relevance to managers in the field. Especially in regions with low inherent soil fertility or high precipitation leaching, or both, management options that either add Ca to systems or decrease its removal are increasingly being examined and employed. Indeed, in some tropical forests where native precipitation-based leaching has historically been high, and pollution-induced acidity is on the increase, sustainable forestry is not currently possible without the

addition of Ca after logging (Nykqvist 2000). The output of tested models that simulate the complexities of Ca cycling could greatly enhance the capabilities of policymakers and managers to integrate the influences of factors adding to or removing Ca from forest ecosystems. As such, these models could play a critical role in guiding adaptive policy and management decisions that prevent or mitigate pollution-induced damage to forest ecosystems.

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LAND

Case Studies

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Assessment of Oak Wilt Threat to Habitat of the Golden-Cheeked Warbler, an Endangered Species, in Central Texas

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Abstract

A major epidemic of oak wilt, caused by *Ceratocystis fagacearum* (Bretz) Hunt, has been killing trees in Central Texas for at least 40 years. This has created large and expanding canopy gaps in the vast, homogenous live oak woodlands (*Quercus fusiformis* Small) in the Edwards Plateau region of Texas. The changes in stand structure may have detrimental consequences for an endangered migratory songbird, the golden-cheeked warbler (GCW, *Dendroica chrysoparia* Sclater & Salvin). More information is needed to assess the direct impact of oak wilt on the GCW and how oak wilt control measures might affect bird populations. In our study, two surveys with different objectives were conducted at the Fort Hood Military Installation in Central Texas. In 2001, IKONOS 1-meter pan-sharpened satellite imagery was used to assess the incidence and severity of oak wilt. The disease was found to be the cause of mortality in 69 percent of the sampled plots. Only a small proportion of the oak wilt centers (12 percent) were located in designated GCW habitat. A second survey was conducted in 2003-04 to determine the key characteristics of GCW nesting sites and how they compare to those of oak wilt centers. This systematic survey was based on randomly selected cluster sample plots stratified in five resource categories based on the presence or absence of oak wilt, GCW habitat, or GCW nesting sites, or both. Stand densities ranged from 90 trees/ha (GCW habitat, no oak wilt) to 1,298 trees/ha (GCW habitat, nesting site). Juniper (*Juniperus ashei* Buchh.) to oak ratios ranged from 0.24:1 (GCW habitat, no oak wilt) to 6.57:1 (GCW habitat, no oak wilt). Classification tree analysis was conducted to identify independent variables associated with the presence of nesting sites in

GCW habitat. Key variables in the resulting model included road density, selected Landsat and SPOT 10 satellite imagery bands, elevation, and distance to roads. In terms of tree mortality, the impact of oak wilt on GCW home ranges may be minimal. Further analyses are needed to evaluate the impacts of other site disturbances caused by oak wilt, such as fragmentation and alterations in stand composition. The results of this project will be used to aid natural resource managers when conflicts occur between endangered species management and oak wilt control.

Keywords: *Ceratocystis fagacearum*, classification tree analysis, endangered species, Fort Hood, golden-cheeked warbler, oak wilt.

Introduction

Woodlands were sampled on Fort Hood Military Installation that were typical of the oak-juniper savanna ecosystem in central Texas. The tree disease oak wilt, caused by *Ceratocystis fagacearum* (Bretz) Hunt, is a common disturbance throughout the region. Oak wilt management is a viable option for reducing losses from the disease, but the decision to implement control options is not always obvious. Further information is needed to assess the benefits of controlling oak wilt when compared to the costs of deploying expensive and disruptive management tactics. Specifically, the objective of this project was to determine whether oak wilt is having a detrimental impact on endangered species habitat. This information would presumably be useful to natural resource managers responsible for oak wilt management decisions. The following topics describe the study site, the status of an endangered species that may be influenced by oak wilt management decisions, and the disease.

Characterization of Central Texas Ecosystem

Central Texas is a unique, fragile ecosystem increasingly pressured by multiple land use objectives. A description of this ecosystem is important for understanding the complex issues being faced by natural resource managers throughout the region. Central Texas is dominated by the Edwards

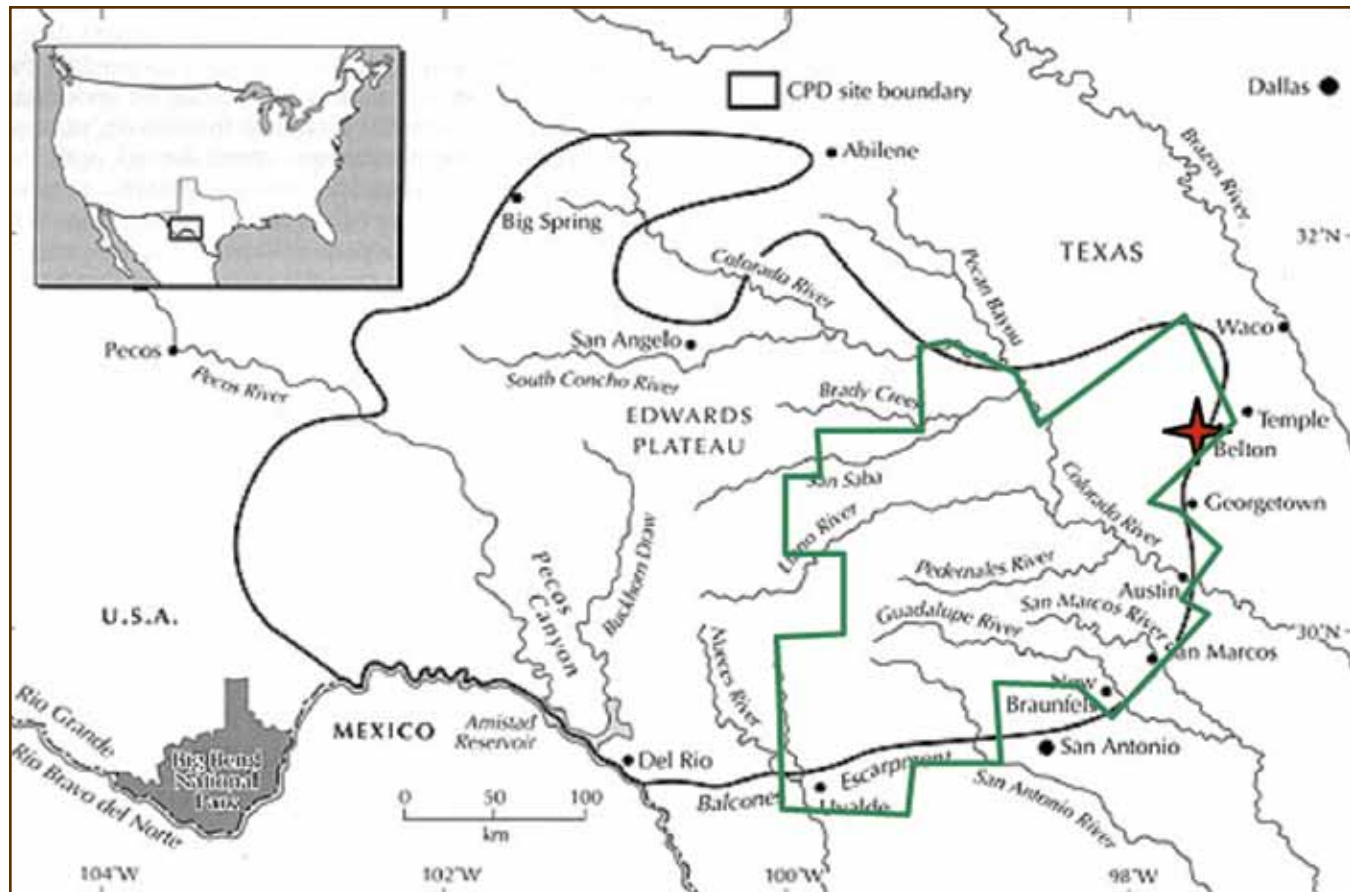


Figure 1—This map shows the region in Texas called the Edwards Plateau. Fort Hood is located with a red star, and the hill country is outlined in green.

Plateau, a limestone-layered tableland lying between latitudes 29°–32° N and longitudes 97°30'–102°30' W (Figure 1). The region is known as the Texas Hill Country because the topography is highly dissected by canyons separated by flat or sloping divides. There are prodigious outcrops of Cretaceous limestone, and the thin soils are mostly stony clay loam (Davis and others 1997, Jordan 1970). Rainfall is sparse at around 35 in per year, resulting in a low, semiarid temperate, semievergreen forest interspersed with grassland savanna. Historically, the Edwards Plateau was on the southern range of the Great Plains grassland prairie, but fire control and overgrazing have significantly changed the landscape (Reisfeld 1992). The dominant trees occur in large expanses and include live oak (*Quercus fusiformis* Small), Ashe juniper (*Juniperus ashei* Buchh.), and mesquite (*Prosopis glandulosa* Torr.) (Burns and Honkala 1990, Hayden and others 2001). Primary land uses are ranching,

tourism, retirement urbanization, wildlife and hunting leases, and farming. In the past few decades, relatively low land costs have encouraged a doubling of population size owing to light industrial development and subdividing formerly large ranches into small ranchettes for retirement and tourism. The Edwards Plateau is a region of biological transition at the limits of the natural ranges of numerous plant and animal species. Many of these are threatened and endangered species.

Characterization of the Study Site: Fort Hood, Texas

There are no significant Federal parks or reserves in the Texas Hill Country. Opportunities for large-scale conservation management are limited. One exception is the largest Army installation in the United States, Fort Hood, covering 88,500 acres in Bell and Coryell Counties (Figure 2).

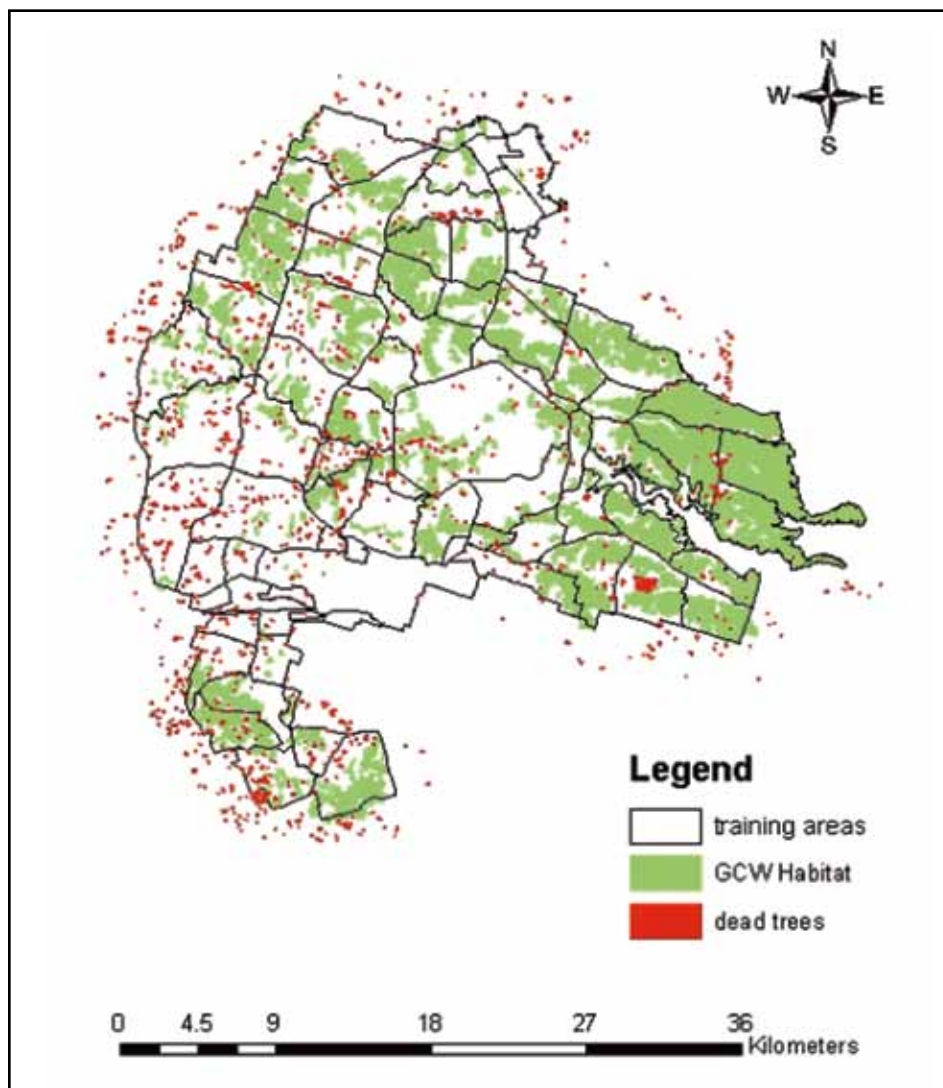


Figure 2—Photointerpreted polygon, training areas, and golden-cheeked warbler habitat at Fort Hood, Texas.

Fort Hood, the home to two U.S. Army divisions, has the advantage of being under a single management authority and operates under the auspices of the Endangered Species Act. Fort Hood was originally established on privately held ranchland, consisting of 65 percent perennial grassland and 30 percent forest/woodlands (Hayden and others 2001). A full range of military training operations is conducted at Fort Hood, including large-scale troop and vehicle movements, live-fire weapons exercises on extensive training ranges, and realistic air attack and air transport missions. Also, there are large expanses of cattle grazing under lease through cattlemen's associations. Sections of Fort Hood are

available for public recreation, including water sports, hunting leases, mountain biking, off-road vehicles, and hiking. Soil compaction, vegetation damage, and erosion are just a few of the disturbances having an impact on the fragile Fort Hood topography (Chenault 2005). Land management activities at Fort Hood are conducted under plans designed to protect and mitigate effects on the habitats of a number of endangered species while repairing rangeland and adapting sites for military training activities. These multiple land use objectives often conflict, providing an ideal location for studying the consequences of controlling oak wilt within endangered species habitat.

Association between Golden-Cheeked Warbler and Oak Wilt

A migratory songbird inhabiting Fort Hood, the endangered golden-cheeked warbler (GCW, *Dendroica chrysoparia*), is of particular interest to conservation specialists, wildlife experts, and military planners. The original listing of the GCW as threatened and endangered was in 1990 (USFWS 1990). The breeding and nesting requirements of the GCW are particularly dependent upon certain characteristics of the oak/juniper savannas of central Texas (Kroll 1980). As a migrating species, the GCW overwinters in Central America and southern Mexico and returns to Texas in the spring for 3 to 4 months (Ladd and Gass 1999). While in Texas, the warbler inhabits woodlands comprising mature junipers with shedding bark that is used for nesting. Oaks are required for foraging because they support high populations of Lepidopteran insects during the breeding season (Kroll 1980). Feeding and breeding GCW habitats may be considered one and the same. These warblers forage for insects in oak tree canopies within their home ranges (Kroll 1980, Pulich 1976, Simmons 1924, Smith 1916, Wahl and others 1990), and nests have been found in Ashe juniper, Texas red oak (*Quercus buckleyi* Buckl.), post oak, (*Q. stellata* (Wangenh.)), Texas ash (*Fraxinus texensis* (Gray) Sarg.), and live oak trees in Fort Hood (Hayden and others 2001). Suitable habitat usually consists of steep canyon slopes or rugged terrain (Moses 1996) with some proximity to a source of water. About 21 850 ha, or 24.7 percent of the total installation, is designated GCW habitat (Dearborn and Sanchez 2001). Urbanization, fragmentation of breeding habitats for agricultural purposes, and predators are the primary reasons given for the decline in GCW numbers throughout its northern range (Moses 1996, USFWS 1990). Although the additional woodland disturbance caused by oak wilt is mentioned in the GCW recovery plan as a factor with the potential to impact GCW populations, it needs further study (Keddy-Hector 1992).

Ceratocystis fagacearum is a destructive pathogen causing enormous losses of oaks throughout Central Texas (see Web site of the Texas Forest Service <http://www.texasoakwilt.org>). In terms of numbers, live oak is the species most severely affected by oak wilt. The fungus

grows through connected root systems of live oak resulting in large, expanding patches of dead and dying trees. In red oaks (gen. *Quercus*, subgenus *Erythrobalanus*) such as Texas red oak, the pathogen can grow briefly as a saprophyte forming fungal pads under the bark, thus making spores available to insect vectors. These two modes of transmission, through roots and by insect vectors, strongly influence the spatial distribution of the disease. Root transmission in live oaks kills larger numbers of trees, but insect transmission initiates new disease centers.

Oak Wilt Management Concerns and Objectives

Oak wilt control on a landscape scale involves removal of large numbers of trees, both healthy and diseased, and digging deep trenches on the perimeters of disease centers. These measures prevent inoculum formation and spread of the pathogen through root connections (Appel 1995). They are expensive and result in a great deal of environmental disruption in order to successfully control the disease. Resource managers must therefore be able to assess the potential impact of the disease and the benefits resulting from costly control measures. Given the conflicting land management objectives throughout the central Texas region, the decision to undertake oak wilt control on the landscape level can be difficult to make. We have initiated long-term studies on Fort Hood to assess the impact of oak wilt on GCW habitat and to contribute this knowledge in the oak wilt management decision process.

Methods and Materials

Separate surveys were conducted to assess the incidence of oak wilt on Fort Hood and to determine the effects of the disease on GCW populations. Each of the surveys incorporated satellite imagery into a geographic information system, ground surveys, and data analysis with various statistical approaches.

2001 Survey

The goal of the 2001 survey was to estimate the incidence of oak wilt at Fort Hood. In order to complete a survey of the entire installation with a minimum of personnel, IKONOS 1-m pan-sharpened satellite imagery was obtained for Fort

Hood that included a buffer area extending 1 mi beyond the boundary (Pacific Meridian Resources, Emeryville, CA 94608). Survey lines separated by 330 m were transposed on the images utilizing the geographic information system ArcView (ESRI, 380 New York St., Redlands, CA 92373). Fort Hood is parceled by training areas (Figure 2), which were also used to aid photointerpretation. Images of the entire post were interpreted by a trained technician to select and map live and red oak mortality, presumably from oak wilt. Attempts were made to exclude brush clearing, wildfire, and obvious sources of mortality other than oak wilt. The imagery was coregistered to Orthophoto Quarter Quadrangles (DOQQS), and the mortality polygons were transposed to maps for ground truthing. A random sample of 10 percent of the photointerpreted polygons was selected for diagnosis. Oak wilt was diagnosed according to recognized symptoms of the disease in the field and laboratory isolation of the pathogen when necessary (Appel 2001).

2003–04 Survey

One of the goals of the second survey was to characterize typical GCW nesting sites and assess the threat posed by oak wilt to GCW populations. This goal was part of a larger project conducted in cooperation with the USDA Forest Service (USFS) Forest Health Technology Enterprise Team (FHTET, Fort Collins, Colorado) that focused on methodology to model and predict oak wilt incidence and severity. The tool being tested for these purposes was binary classification and regression tree analysis (CART) (Baker and others 1993, De'ath and Fabricius 2000). This nonparametric statistical technique results in a classification tree intended to explain variation of a dependent or response variable by a collection of independent, or explanatory variables. The dependent variable for the FHTET model was the presence or absence of oak wilt. Cluster sample plots ($n = 80$) were randomly selected using a Sample Points Generator (SPGen), an ArcView application, from four land classification categories: (1) GCW habitat, non-oak wilt; (2) GCW habitat, oak wilt; (3) non-GCW habitat, oak wilt; and (4) non-GCW habitat, non-oak wilt. A fifth category consisting of known nesting sites (GCW/NS) was subsequently added as an additional dependent variable

for a separate CART analysis. Plots in this category were known to have been occupied by GCW nesting pairs during 2002–03, in contrast to plots in the other four categories. In these latter plots, GCW habitat was designated according to stand characteristics based on aerial photography and ground surveys. Designated GCW habitat is characterized as having high densities of mature junipers, the availability of deciduous hardwoods (primarily oaks), and a proximity to water (Hayden and others 2001). Independent variables for the model were derived from two sources of satellite imagery (2003 Spot 10 and Landsat TM satellite imagery) and geographic information system (GIS) files in the format of grid themes to be used in the ArcView program, e.g., slope, elevation, aspect, soils, distance to roads, road density, distance to streams, streams density, distance to lake, forest savanna, and landform. Ancillary data for Fort Hood were obtained from the Natural Management Branch office at Fort Hood. The classification tree was fitted to the spatial information database using the S-PLUS[®] statistical software package (Insightful Corp., Seattle, WA 98109).

Systematic surveys by ground crews from the USFS, the Nature Conservancy, the Texas Forest Service, and Texas A&M University were conducted in the summers of 2003 and 2004. Cluster sample plots were distributed throughout the four sampling categories. Each sample consisted of a 20-m by 20-m fixed plot subdivided into four subplots. Data collected for plots and subplots consisted of tree diameters, tree species identification, symptom development of infected trees, dominant overstory and understory species, and average tree height. In 2003, 80 systematic fixed plots were surveyed with equal numbers of plots in each of the four categories. In 2004, an additional 33 fixed plots were surveyed to increase sampling intensity, bringing the total to 28, 21, 32, and 32, in each of the four categories, respectively. Unlike the 2001 survey, all plots were located within the boundaries of Fort Hood. For the present study, the response or dependent variable was the presence of a GCW nesting site. The 24 GCW nesting site locations from 2002 and 2003 were obtained from the Nature Conservancy and surveyed in 2004 in the manner identical to the procedure described for the sample cluster

plots. These plots were treated as a separate category resulting in five different categories for the analysis.

A preliminary analysis of the total survey data was conducted to determine frequencies of species, their sizes, and their densities in the various habitat categories. Tree species included deciduous hardwoods other than oaks (DH), live oaks (LO), red oaks (RO), shin oaks (*Q. sinuata* Walt.) and other white oaks (SO/WO), and junipers (J). Typical DHs at Fort Hood include cedar elm (*Ulmus crassifolia* Nutt.), walnut (*Juglans* spp.), hackberry (*Celtis* spp.), and Texas ash (*Fraxinus texensis* (Gray) Sarg.). The dominant red oaks are Texas red oak and blackjack oak (*Q. marilandica* Muenchh.). The primary white oak, other than shin oak, is post oak. The ratio of juniper to oak was also calculated for each habitat category.

Classification Tree Analysis

Tree-based modeling is an exploratory technique for uncovering structure in data (Clark and Pregibon 1992) and has been used with ecological data that are complex, unbalanced, and contain missing values. Classification trees explain variation of a single response variable by one or more explanatory variables. The response variable can either be categorical (classification trees) or numeric (regression trees) (De'ath and Fabricius 2000). The categorical response variable used in this analysis of Fort Hood data was the presence or absence of GCW nesting sites. The tree is constructed by repeatedly splitting the data in two mutually exclusive groups, each of which are as homogeneous as possible. The objective for classification trees is to partition the response variable (GCW nesting sites) into subsets of homogeneous groups while also keeping the tree reasonably small (De'ath and Fabricius 2000). The tree is shown graphically in Figure 3 and consists of three parts: the root node, branches, and leaves. The root node represents the undivided data at the top, the branches and leaves each represent one of the final groups beneath. To keep the trees as accurate as possible, cross validation is a widely used technique to look at the independent variables from the tree and calculate the amount of error produced by iteratively combining the independent variables. It starts with one variable, then adds another, and keeps adding independent

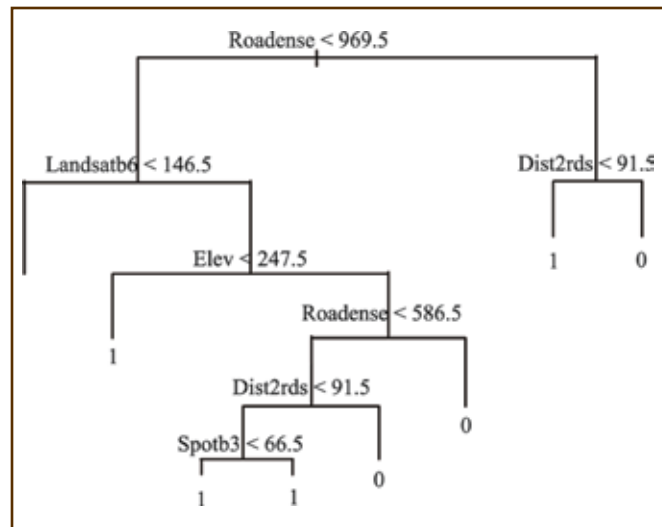


Figure 3—Classification tree model of golden-cheeked warbler (GCW) nesting-site characteristics.

variables until all independent variables have been included. The result of the cross validation is plotted with the x-axis as the number of terminal nodes and the y-axis as the misclassification error. The best model chosen is the one with the greatest number of terminal nodes with the least amount of misclassification error. The original tree was then pruned to the best model chosen from the cross validation results.

S-PLUS[®] statistical software was used to fit the classification tree to the Spatial Information Databases for each model (TREE, S-PLUS[®], Statistical Sciences 2000). Twenty-two independent variable grid themes and 25 field data categories were used to construct the classification tree to describe the GCW nesting sites in Fort Hood. The independent variable grid themes consisted of each of the seven bands exported as grid themes from Landsat 5 TM and each of the four bands exported as grid themes from SPOT 5. The remaining 11 variables included slope, elevation, aspect, soils, distance to roads, road density, distance to streams, streams density, distance to lake, forest savanna, and landform. ERDAS Imagine Software Grid Export function was used to create the individual grid themes from each band in the imagery (ERDAS Inc. ERDAS Imagine V8.5. 2001, Atlanta, GA). The classification trees were run comparing the nesting site data with the data from GCW habitat with no oak wilt present (GCW/non-OW). The comparison was run with both the grid-theme data and

Table 1—Diagnostic results for a sample of tree mortality locations randomly selected from photointerpretation of satellite imagery of Fort Hood, Texas

Cause of mortality	No. of centers	Total
		<i>Percent</i>
Oak wilt	82	69
Military ops	1	0.8
Unknown	8	6.7
Brush piles	23	19.3
Blow down	1	0.8
Fire	4	3.3

field-collected data (total data), grid-theme data only (independent data), and field-collected data only (field data).

Results

2001–Survey

There were 1,164 polygons delineated as dying oak trees on the IKONOS satellite imagery. A sample of 119 polygons, or 10 percent of the total, was randomly selected for ground truthing (Table 1). Oak wilt was found to be the cause of mortality in 82 (69 percent) of the centers. The major factor, other than oak wilt, delineated as dying or dead trees on the imagery, was brush-clearing operations (19.3 percent), where piles of dead trees resemble the crowns of dead, standing live oaks. With two exceptions, all of the brush piles consisted of Ashe juniper cut and stacked during land-clearing operations. Relatively few other causes of mortality were found, including fire, wind damage, and damage to trees caused by military operations. At eight of the sites, trees identified as oaks were actually some other species, or causes of mortality were not readily identified.

Of the 1,164 polygons, 821 fell within the perimeter of Fort Hood. Of those 821 polygons, 144 or 18 percent were located in designated GCW habitat. Of the total 82 oak wilt centers

Table 2—Species composition and average diameters of trees located in sample cluster plots for the 2003–04 survey

Habitat ^a	Total no. plots	Trees / ha	Tree type ^b	Total no. trees	Proportion of total	Ave. d.b.h. ^c <i>Inches</i>
GCW/non-OW	28	886	DH	115	0.12	3.8
			LO	38	0.04	6.6
			RO	58	0.06	5.5
			SO/WO	20	0.02	3
			J	762	0.78	6.9
GCW/OW	21	639	DH	55	0.11	5.3
			LO	103	0.21	10.6
			RO	72	0.15	6.6
			SO/WO	6	0.01	2.3
			J	301	0.62	5.6
non-GCW/OW	32	570	DH	190	0.26	5
			LO	260	0.33	11.5
			RO	165	0.23	8.1
			SO/WO	10	0.01	3.3
			J	105	0.13	5.1
non-GCW/non-OW	32	90	DH	42	0.37	4.4
			LO	8	0.07	9.0
			RO	13	0.11	4.5
			SO/WO	0	0	0
			J	52	0.45	6.1
GCW/NS	24	1,298	DH	185	0.15	5.36
			LO	34	0.02	6.76
			RO	122	0.1	7.7
			SO/WO	99	0.08	3.53
			J	806	0.65	6.98

^a GCW/non-OW = golden cheeked warbler habitat, no oak wilt; GCW/OW = golden cheeked warbler habitat with oak wilt present; non-GCW/OW = non-habitat, oak wilt present; GCW/NS = habitat with nesting site present.

^b DH = deciduous hardwood, LO = live oak, RO = red oak, SO/WO = shin oak or other white oak.

^c d.b.h. = diameter at breast height.

identified in the survey (including the 1-mi buffer), 60 were located within the post perimeter. Only 7, or 12 percent of the oak wilt centers found within the Fort Hood perimeter, were located in designated GCW habitat.

2003–04 Survey

The highest stand densities, 1,298 trees/ha, were found at the nesting sites within GCW habitat (GCW/NS) (Table 2). Stand densities were

Table 3—Juniper to oak ratios for each of the four sampling categories at Fort Hood, Texas

Habitat ^a	J:O ratio
GCW/non-OW	6.57:1
GCW/OW	1.66:1
non-GCW/OW	0.24:1
non-GCW/non-OW	2.48:1
GCW/NS	3.16:1

^a GCW/non-OW = golden-cheeked warbler habitat, no oak wilt; GCW/OW = golden-cheeked warbler habitat with oak wilt present; non-GCW/OW = no golden-cheeked warbler habitat, oak wilt present; non-GCW/non-OW = no golden-cheeked warbler habitat, no oak wilt present; GCW/NS = habitat with nesting site present.

also relatively high in habitat where there was no oak wilt (GCW/non-OW = 886 trees/ha). At oak wilt locations both within and outside of habitat, the stand densities were relatively lower. There were also notable trends in the species among the various categories. The proportion of juniper was far lower outside GCW habitat where oak wilt was present (13 percent) than in the plots located within GCW habitat (62 percent) (Table 2). Live oak density was greater in oak wilt locations, whether they were within (21 percent) or outside of habitat (33 percent), than in the uninfected plots within habitat (4 percent) or nesting sites (2 percent).

The juniper to oak ratios (J:O) varied widely among the four sampling categories. The highest J:O ratio was 6.57:1 in the GCW habitat where there was no oak wilt (Table 3). The lowest was 0.24:1 in oak wilt centers outside of GCW habitat.

Classification Tree Model

The classification tree model was developed using the plot survey data from nesting sites (GCW/NS) in 2004 and tested with plot survey data from one of the four habitat categories (GCW/non-OW) collected in 2003 and 2004. The nesting sites were assigned a value of 1, and the plots used to test against were assigned the value of 0. The analysis included the total data (field and independent data) from both the nesting-site plots and from the GCW/non-OW plots. Preliminary analysis indicated that the accuracy of the classification tree could be increased by modeling the soil types separately from the other independent data. The resulting classification tree had an accuracy of 98.2 percent

with 8 terminal nodes (Figure 3). Discriminating variables included road density, Landsat band 6, elevation, distance to roads, and Spot band 3. Deviance was calculated for each variable by dividing the total deviance of the model variance by each of the variables produced (Kelly 2002). Road density explained the most variance (62 percent) in nesting site habitat location, followed by Landsat band 6 (43 percent), elevation (30 percent), road density (16 percent), distance to roads (6 percent), and Spot band 3 (3 percent). Plots that had a road density of less than 969.5 m per mi² had a higher probability of being in a site for GCW nesting-site habitat. Of all combinations of forest habitat conditions for this test, GCW habitat nesting sites are more likely to occur in areas having low road density (roadense < 969.5 m per mi²), an elevation greater than 247.5 m, and a distance from roads of less than 91.5 m.

Discussion

Photointerpretation of the satellite imagery for the 2001 survey proved to be fairly accurate in identifying oak wilt. Brush piles resulting from roguing Ashe juniper were the features most often confused with oak wilt. Improved training would probably reduce many of these errors, but may not eliminate them altogether. Juniper clearing is a common practice in the Hill Country, and, in many cases, piles were visually indistinguishable from the crowns of dead live oaks.

Oak wilt was found to be a prominent feature and a major cause of oak mortality throughout Fort Hood. No other cause of mortality came close to the level found for oak wilt. However, the survey was not designed to determine the volume or extent of the other major cause of tree mortality—fire. It was noted that in certain locations, fire was a dominant cause of mortality and probably far exceeded the extent of oak wilt as a disturbance. A detailed analysis of the comparative effects of fire and oak wilt on habitat is warranted and will be considered in future studies.

Habitat requirements and tree species composition associated with GCW populations have been addressed in previous studies (Kroll 1980). The dependency of GCW on Ashe juniper bark as a source of nesting materials is

a well-described phenomenon (Kroll 1980, Pulich 1976). There appear to be preferences for the sizes and densities of junipers. Kroll (1980) characterized good habitats as those with juniper-oak ratios of 1.35 to 1, and poor habitats with ratios of 2.27 to 1. We also found trends for the selection of nesting sites at Fort Hood. The juniper-oak ratios ranged from a high of 6.57:1 to a low of 1.66:1 in designated GCW habitat (Table 3). But, our results indicate that preferred nesting sites were in areas with a juniper-oak ratio of 3.16:1. Other variables determined to be characteristic of good habitats by Kroll (1980) were older Ashe junipers at wider spacing and relatively lower densities than those of poor habitats. In a previous study conducted at Fort Hood, Dearborn and Sanchez (2001) made pairwise comparisons of 13 vegetation variables between nest locations and nearby non-use vegetation patches. The only significant variable was canopy closure, which was greater at nesting sites than at the paired nest-free location. A stand density equivalent to 487 stems per ha for junipers and hardwoods combined was found at nesting sites, and junipers dominated hardwoods in all size classes. Nesting sites were characterized as having dense vegetation and nearly complete canopy closure dominated by junipers (Dearborn and Sanchez 2001). The nesting sites in our survey appear to have higher stand densities than those surveyed by Dearborn and Sanchez (2001), but we included smaller diameter stems in the survey protocol. The trends in both surveys are consistent. Of all habitats surveyed, our results confirm that GCWs prefer dense vegetation with high juniper densities.

Live oaks dominated sites where oak wilt occurred outside of GCW habitat. Within GCW habitat, the highest levels of live oaks also occurred in oak wilt centers, with a large decrease of live oak density at nesting sites and randomly selected habitat sites. The average diameters of live oak were larger in oak wilt centers than in healthy plots. A similar trend, although not as pronounced, was observed for the deciduous oaks. Oak wilt appears to be less likely to occur in places where the proportion of oaks is relatively low, such as GCW nesting sites. The incidence of live oak depends, for the most part, on availability of susceptible hosts, availability of inoculum, occurrence of infection courts (fresh wounds), and existence of nitidulid

vectors (Appel 2001). Red oak density was fairly consistent among the different plot types, so inoculum availability in the form of fungal mats was potentially the same. The most likely explanation relates to vector behaviors in the live oak-dominated stands, but this suggestion would need to be confirmed with trapping studies. Our results also suggest that the site requirements for oaks may not coincide with sites preferred by GCW. The oak wilt threat to critical habitat may therefore be less than anticipated. A comparison of the GCW classification tree model developed in the present study for nesting sites, with the oak wilt model developed by FHTET, should be useful in determining whether oak wilt is a threat to GCW breeding habits.

Classification tree modeling proved to be a useful technique for establishing the site factors influential in determining the habitat for GCW nesting sites. When the comparison was made between designated habitats classified by The Nature Conservancy and nesting sites, the classification tree revealed that low road density was needed for ideal nesting-site locations. This agrees with other research findings that GCW prefers to have large unfragmented habitat for breeding and territory ranges (Kroll 1980, Ladd 1985, Moses 1996). There are, however, conflicting opinions on the GCW preferences for large blocks of unfragmented habitat or for sites bounded by edges of different vegetative composition (Moses 1996). In one study in Travis County (Texas), the estimated territory required per breeding pair of GCWs was 1.9 to 2.7 ha/pair (Ladd 1985). Kroll (1980) estimated ranges for breeding pairs were 4.49 to 8.48 ha/pair in a Texas state park. These estimates were noted to be larger than those from previous research, which resulted in estimates of territory sizes ranging from 0.81 ha to 2.55 ha per breeding pair (Kroll 1980). One effect of oak wilt is to fragment contiguous tree stands into treeless patches and expanding edges (Appel and others 1989). Dispersal distances for adult males averaged 223 m in a study conducted at Fort Hood during 1991–96, whereas juvenile dispersal distances were greater, averaging 4040 m (Jette and others 1998). These dispersal distances, patch expansion, and the creation of edges by oak wilt requires further research to determine how the disease relates to the GCW beyond the consequences of direct loss of trees.

The effects of oak wilt on the landscape go beyond the destruction of trees. Gaps and edges are created, tree composition is changed, and woodland stand structure is altered. All of these effects may influence GCW populations and will require further analysis to confidently decide whether to manage the disease in the vicinity of GCW nesting sites. Oak wilt control activities need not be disruptive because they can be implemented when the birds are migrating. However, because oak wilt appears to fall in areas where oak densities are greater than those found in preferred GCW habitats, the simple loss of trees may not be sufficient justification to undertake expensive and disruptive oak wilt control methods.

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A Methodology for Assessing Annual Risk of Southern Pine Beetle Outbreaks Across the Southern Region Using Pheromone Traps

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Abstract

An operational system to forecast infestation trends (increasing, static, declining) and relative population levels (high, moderate, low) of the southern pine beetle (SPB), *Dendroctonus frontalis*, has been implemented in the Southern and Eastern United States. Numbers of dispersing SPB and those of a major predator (the clerid beetle, *Thanasimus dubius*) are monitored with multiple-funnel traps baited with the SPB aggregation pheromone frontalin and host volatiles. One to three traps are placed in each county or national forest ranger district to be surveyed for 4 consecutive weeks during the spring, to coincide with the long-range dispersal of SPB. The average number of SPB per trap per day and the ratio of SPB to total catch of SPB and clerids in the current and previous year for the same trapping location are the variables used for predicting infestation trends and population levels for the remainder of the year. An analysis of predicted and actual SPB infestation trends and population levels for 16 States and up to 19 consecutive years (1987-2005) documents the accuracy of the annual prediction system. From 1987 to 1998, predictions at the State level, validated by subsequent infestation detection records for the specific year, proved accurate 68 percent of the time (range 42 to 83 percent) for SPB infestation trend and 69 percent of the time (range 42 to 92 percent) for population level. From 1999 to 2005, the mean accuracy of predictions of SPB infestation trend improved to 82 percent for all States combined (range 71 to 100 percent); mean predictions for population level for States increased in accuracy to 74 percent (range 43 to 100 percent). Despite system limitations, forest managers have come to depend on this early warning system to predict pending outbreaks

or collapses of SPB populations. This represents the first effective and validated prediction system for outbreaks of a bark beetle species.

Keywords: Bark beetles, *Dendroctonus frontalis*, prediction, surveys, *Thanasimus dubius*.

Introduction

Forest managers and pest control specialists have long needed a reliable and practical early warning system to detect developing outbreaks of the southern pine beetle (SPB), *Dendroctonus frontalis* (Coleoptera: Curculionidae: Scolytinae), the most destructive forest pest of southern pine forests (Thatcher and others 1980). Because aerial surveys to detect SPB infestations do not become effective until late spring or summer (Billings and Doggett 1980), a more efficient system for monitoring SPB populations early in the season has value. Early detection of pending outbreaks gives forest pest managers valuable insight for scheduling detection flights and planning suppression programs.

In 1986, the Texas Forest Service began development of an operational system using pheromone-baited traps for predicting SPB infestation trends (increasing, static, declining) and population levels (high, moderate, low). In cooperation with State and Federal pest management specialists, the system was implemented throughout the South beginning in 1987 (Billings 1988, 1997). Validation and refinement of the system have been conducted periodically with use of year-end SPB infestation detection records. Here, we evaluate the accuracy of this prediction system, based on results from 16 States, most of which have now deployed the system for 20 consecutive years (1987-2006).

The southern pine beetle's range extends from New Jersey to Florida and west to Texas, coinciding with the distribution of its major hosts, loblolly (*Pinus taeda* L.) and shortleaf pine (*P. echinata* Mill.). SPB populations also are found from southern Arizona south to Nicaragua. Outbreaks of this insect tend to occur somewhere within its range every year, with peak populations occurring every 6 to 9 years in many Southern States (Price and others 1998).



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Figure 1—Small (A) and large (B) expanding southern pine beetle (SPB) infestations (spots) in east Texas, as seen from the air. Beetles emerging from the yellow-crowned host trees (faders) will tend to attack uninfested, green-crowned trees on the spot periphery in response to aggregation pheromones, leading to continuous spot growth during summer months.

The location and intensity of SPB outbreaks may vary greatly from year to year. Unlike most other destructive bark beetles of the genus *Dendroctonus*, SPB completes up to seven generations per year in Gulf Coastal States and infests host trees in distinct and predictable patterns that vary with the seasons. For example, most new multiple-tree SPB infestations (spots) are initiated during the spring, following long-range dispersal of overwintering beetle populations (Hedden and Billings 1979, Thatcher and Pickard 1964). In the late spring and summer, adult beetles tend to attack host trees on the periphery of the same spots from which they emerged (Gara 1967; Hedden and Billings 1977, 1979; Thatcher and Pickard 1964). This behavior leads to expansion of previously established spots, rather than the initiation of new ones. These expanding infestations (Figure 1) are easily recognized in summer detection flights by the presence of pines in various phases of crown discoloration (Billings and Doggett 1980). Emerging SPB adults tend to disperse again in the fall, redistributing the population among scattered single trees, nonexpanding spots, and established infestations prior to winter.

Materials and Methods

Standardized procedures for conducting the annual SPB prediction survey are available online at http://texasforests.tamu.edu/pdf/forest/pest/tutorial_for_predicting_spb.pdf. Basically, from one to three multiple-funnel traps (Lindgren 1983), available from Phero Tech, Inc., Delta, British Columbia, and other sources (e.g., Synergy Semiochemicals Corp., Burnaby, British Columbia, Canada) are placed in pine forests within a county or national forest ranger district in early spring. The traps consist of 12 funnels superimposed over a collection cup (Figure 2A) that is partially filled with soapy water. The number of counties or ranger districts surveyed differs from State to State, depending on availability of pest management personnel, supplies, travel distances, and other factors.

The survey is initiated when flowering dogwood (*Cornus florida* L.) begins to bloom or loblolly pine pollen appears. These physiological events coincide with the long-range dispersal of SPB (Billings, unpublished data). Each spring, when ambient temperatures consistently exceed 59 °F, (the flight threshold for SPB and its predators) (Moser

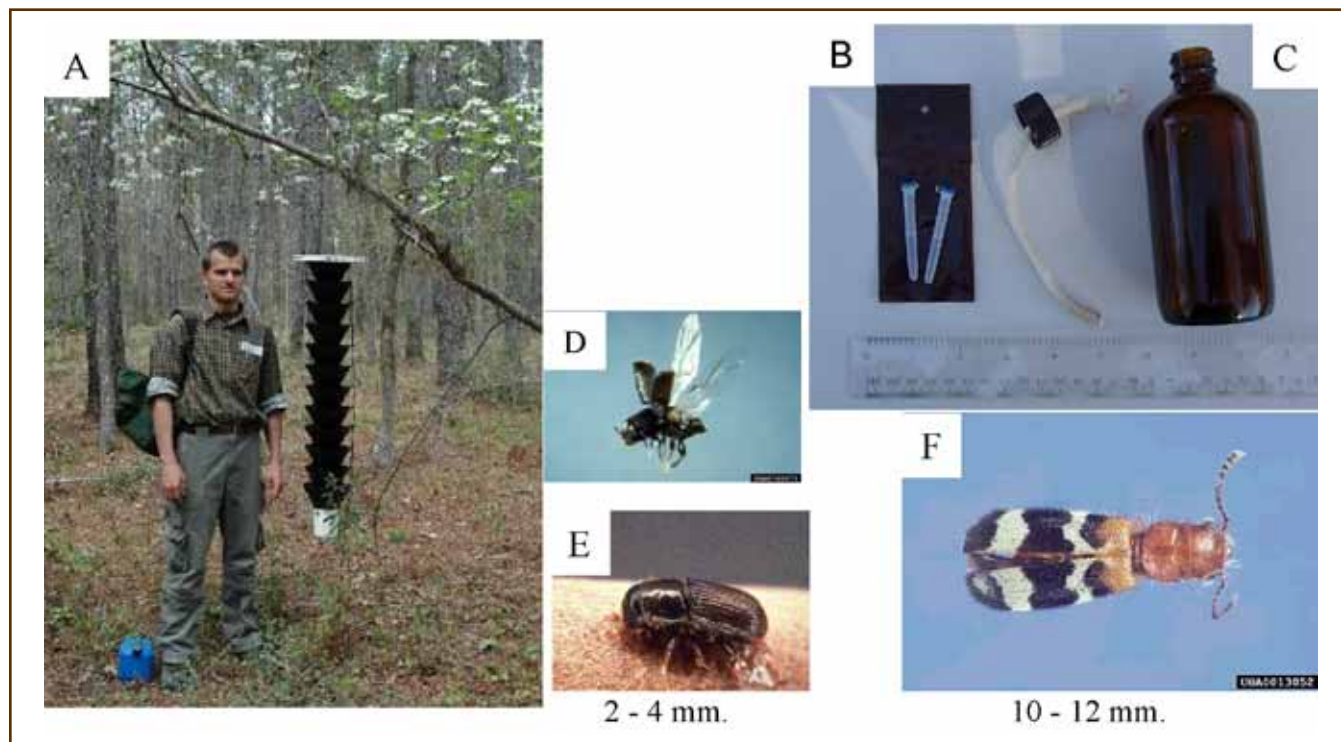


Figure 2—The Southern Pine Beetle Prediction System uses 12-funnel survey traps. The traps are (A) placed in the field for 4 weeks in the spring starting when dogwoods bloom. Each trap is baited with the SPB pheromone frontalinal (B) and a bottle of steam-distilled pine turpentine (C), an attractive bait combination that draws in flying adult SPB (D) and the clerid beetle, *Thanasimus dubius* (F), a major SPB predator. Both the number of SPB (E) and number of clerids caught in traps are used to forecast SPB infestation levels for the current year. (Photos A-C by R. Billings; photo D by Erich G. Vallery, USDA Forest Service, Bugwood.org; photo E by Texas Agricultural Extension Service Archive, Texas A&M University, Bugwood.org; photo F by Gerald J. Lenhard, Bugwood.org)

and Dell 1979), populations of SPB disperse from overwintering sites in search of weakened pine trees in which they initiate new infestations (Billings and Kibbe 1978). Because of regional variations in climate, trapping usually occurs from early March to mid-April in the Gulf Coastal States and in April or May in States in the northern portions or mountain regions of the beetle's range. It is during this spring dispersal period that airborne populations of adult SPB and associated insects can be most effectively monitored with pheromone-baited traps for predictive purposes (Billings 1988).

Each trap is baited with a single pheromone packet containing two 400 microliter “bullet” capsules of racemic frontalinal (Phero Tech, Inc., Delta, British Columbia) (Figure 2B), the SPB aggregation pheromone (Kinzer and others 1969, Payne and others 1978), and a rapid-release dispenser of steam-distilled southern pine turpentine (Billings 1985) (Figure 2C). From 1987 to 2005, the turpentine

dispenser consisted of a 250 ml amber Boston round bottle (Fisher Scientific Company, Pittsburgh, PA) with a cotton wick. Various brands of commercial-grade southern pine turpentine (Klean-strip™ (1987-1997), Star-tex™ (1998), and Hercules™ (1999-2006)) were used, depending on availability. These insect- and host-produced volatiles attract in-flight adult SPB (Figures 2D, E) as well as a major predator, the clerid beetle *Thanasimus dubius* (Coleoptera: Cleridae) (Figure 2F) (Billings and Cameron 1984, Moser and Dell 1980, Payne and others 1978, Vite and Williamson 1970). More than any other associated insect, *T. dubius* is believed to play a major role in the population dynamics of SPB (Moore 1972; Reeve and Turchin 2002; Thatcher and Pickard 1966; Turchin and others 1991, 1999).

Each trap is installed in a pine-forested area, preferably in stands having sawtimber trees (> 30 cm in diameter at breast height) with a sparse hardwood understory. The traps are purposely placed outside of SPB infestations to

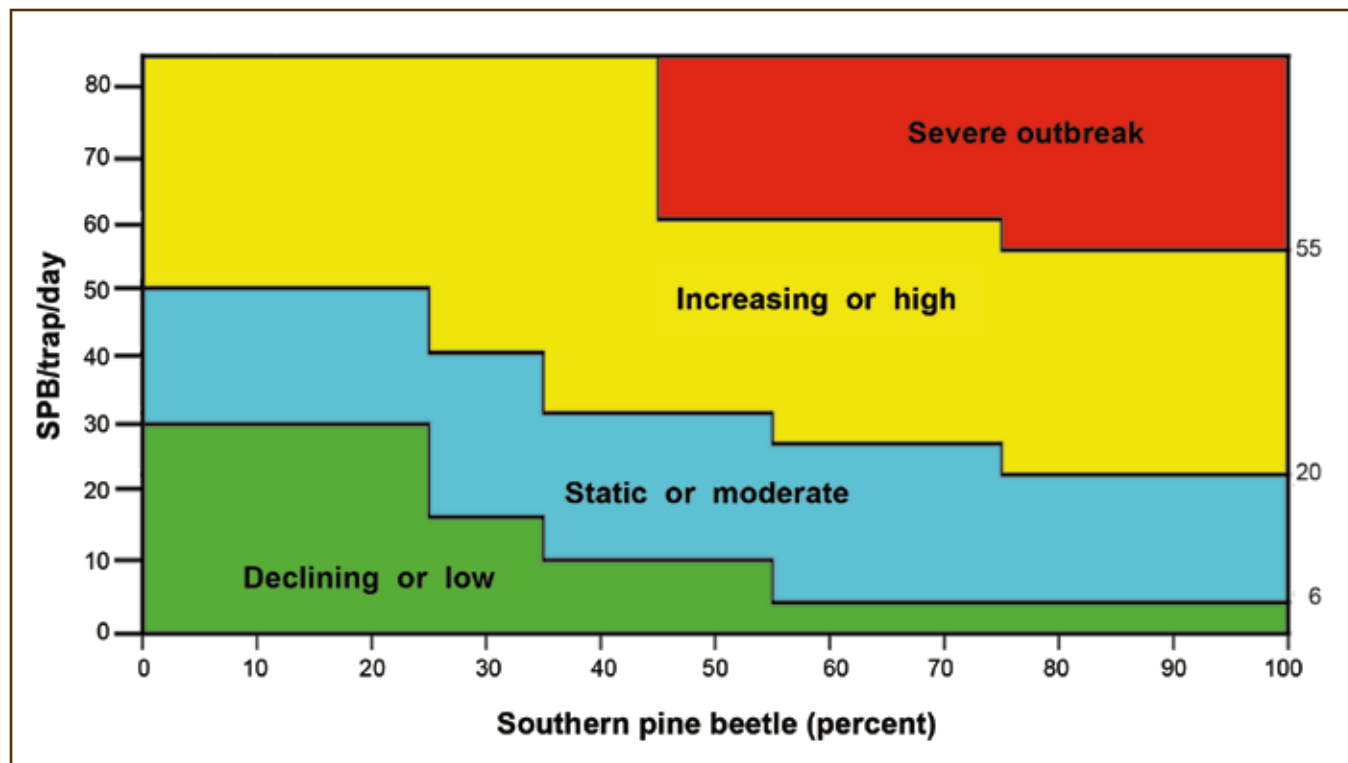


Figure 3—Southern pine beetle prediction chart. The prediction of SPB infestation trend and level for the current year is derived by plotting the mean number of SPB per trap per day and percent SPB for the current year and comparing these data to that for the previous year (if available) for a given locality. Percentage SPB = $(\text{No. SPB} \times 100) / (\text{No. SPB} + \text{No. Clerids})$.

attract long-range dispersing SPB and its predators. Traps are suspended from a 3-m metal pole or from a cord strung between two hardwood trees, at least 10 m from any live pine (Figure 2A). If multiple traps are deployed in a county or ranger district, they should be at least 1 mi apart. The turpentine bottle is placed within the top funnel of the trap, and the frontalinal packet is hung from a funnel stanchion near the middle of the trap.

Insects are collected from traps weekly for 4 consecutive weeks, and the numbers of adult SPB and clerids are counted and recorded. Upon completion of the survey, the data from each State and Federal cooperator are sent to the authors for compiling. With these data, the authors or cooperators or both make predictions of SPB infestation trend and relative population level for the current year, based on mean numbers of SPB per trap per day and percentage SPB. The latter is defined as the number of SPB $\times 100$ percent divided by the combined number of SPB plus clerids caught per trap (Billings 1988). The data are

plotted onto the SPB prediction chart (Figure 3). This chart was developed and refined over the years by comparing the relationship between the two independent variables (SPB per trap per day and percentage SPB) and actual numbers of SPB spots detected in a given county or ranger district for the current year to ascertain SPB population level. Actual SPB infestation trend is obtained by comparing the number of spots reported in a given locality or State in the current year with the number reported for the same locality or State in the previous year.

Where trapping data are available for the previous year from the same county or ranger district (preferably from the same specific trap location), a comparison of mean SPB per trap per day and percentage SPB for the current year with that for the previous year provides further insight into the direction of infestation trends (increasing, static, or declining). In turn, the number of SPB spots detected in a given county, ranger district, or State in the previous year is useful for predicting population levels (high, moderate, or low) in

the current year. For purposes of this paper, the severe outbreak level shown in Figure 3 was considered the same as the increasing or high level. If no spots were detected in a given county the previous year, and few SPB are caught this year suggesting a declining trend, the prediction would be static or low (rather than declining or low, since infestation levels cannot decline below 0). Trap catch data for all individual counties or ranger districts monitored within a State are averaged to make SPB predictions at the State level. Once all the data are received and processed, the Southwide predictions at the local and State levels are sent to each cooperator and also are made available on the Texas Forest Service Web page (<http://texasforests-service.tamu.edu>).

The accuracy of State-level predictions made since 1987 are summarized in this paper. The Southwide SPB Prediction System was initiated in 1987 in 11 Southern States (Arkansas, Texas, Louisiana, Mississippi, Alabama, Georgia, Tennessee, Virginia, Florida, South Carolina, and North Carolina). Federal and State pest managers in several other States joined at a later date (Maryland in 1988, Oklahoma in 1996, Kentucky in 2000, Delaware in 2001, and New Jersey in 2002) and have participated ever since. Florida did not conduct SPB pheromone surveys from 1989 to 1994, but returned as a cooperator from 1995 to the present. At the end of each calendar year, SPB cooperators provide the authors with total numbers of SPB spots detected from aerial surveys in each county or ranger district monitored with pheromone traps, as well as for the entire State. These data are used to evaluate the accuracy of each year's predictions and provide valuable information for making predictions for the next year.

For purposes of the SPB Prediction System and to evaluate its accuracy, the following variables are defined:

Trend:

- Increasing = > 25-percent increase in total spots in a county, ranger district, or State from previous year.

- Static = $\leq$ 25-percent change in total spots from previous year.
- Declining = > 25-percent decrease in spots from the previous year.

Level:

- Low = < 0.40 spots per 1,000 acres of host type in a county, ranger district, or State during a given year.
- Moderate = 0.40-1.50 spots per 1,000 acres of host type.
- High = > 1.50 spots per 1,000 acres of host type.
- Severe outbreak = combined with increasing or high level for purposes of this paper.

Error:

- Minor = discrepancy between prediction and actual trend or population level was a single category (e.g., prediction was for static trend whereas actual trend was increasing; prediction was for low level whereas actual level was moderate, etc.).
- Major discrepancy between prediction and actual trend or level was two categories (e.g., prediction was for declining trend whereas actual trend was increasing or outbreak, prediction was for low population level whereas actual level was increasing or outbreak, etc.).

Host Type:

- Total acres of loblolly, shortleaf, and slash pine (*Pinus elliottii* Engelm.) from 1990 to 2000 Forest Inventory and Analysis data for a specific State, county, parish, or national forest ranger district.

Results

Actual SPB Infestation Levels: 1987 to 2005

Total numbers of SPB spots detected in 16 States in the Southeastern United States for the period 1987-2005 are shown in Figure 4. Infestation levels on a regional basis varied significantly by year from a high of >90,000 spots in 2002 to a low of <5,000 spots in 2005. During the 12-year interval from 1987 to 1998, at least one SPB outbreak cycle (increasing, peak, declining) occurred in every participating State. SPB populations reached peak levels in Arkansas in 1995; in Texas in 1989 and 1992; in Louisiana in 1992; in Mississippi in 1988, 1991, and 1995; in Alabama in 1992

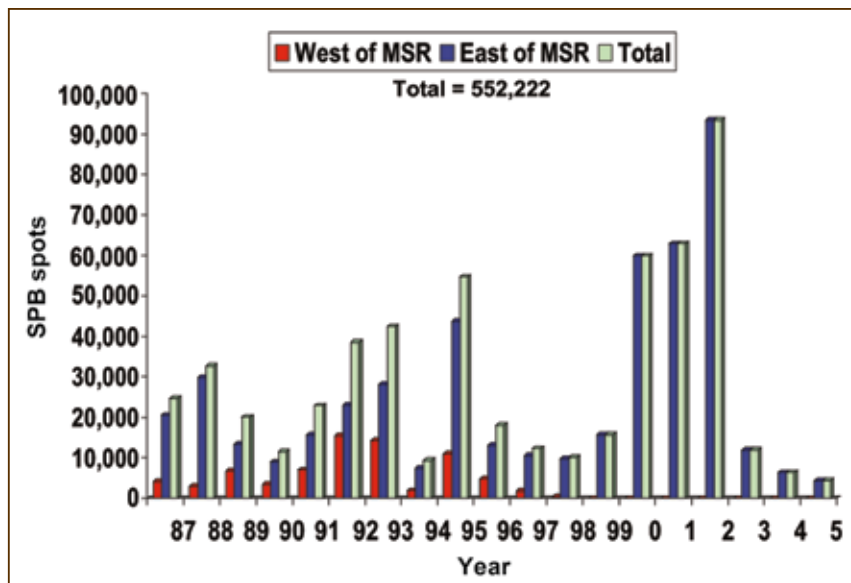


Figure 4—Total numbers of southern pine beetle infestations (spots) detected in 16 States for the period 1987-2005. This figure shows marked differences in infestation levels between those States located east and those located west of the Mississippi River (MSR).

and 1995; in Georgia in 1988; in Tennessee in 1988; in Virginia in 1993; in Florida in 1995 and 1997; in South Carolina in 1995; in North Carolina in 1995; and in Maryland in 1993. Interestingly, 9 of the 11 States that had participated in the system since 1987 reported SPB infestations in every year from 1987 to 1998. Only Florida and Maryland had at least 1 year with 0 SPB spots during this time.

From 1999 to 2005, SPB activity fluctuated widely across the South. SPB infestations totally disappeared west of the Mississippi River: Louisiana, Texas, Arkansas, and Oklahoma reported 0 spots for these years and few if any SPB were captured in pheromone traps during this 7-year period. In sharp contrast, the worst SPB outbreak on record within a single State occurred in South Carolina in 2002, when 67,127 SPB spots were detected (> 14 spots per 1,000 acres of host type). Peak populations occurred in Alabama and Virginia in 2000; in Kentucky, Tennessee, and Florida in 2001; and in Georgia, North Carolina, and New Jersey in 2002. Throughout this period, SPB trapping procedures were standardized with all cooperators using the same turpentine (Hercules™) and dispenser (amber bottle and wick). Also, in 1998 and again in 2002, the SPB prediction chart was slightly modified based on results of pheromone trapping and SPB detection records for the

period 1987–2001 [see Figure 3 versus the chart shown in Billings (1988)]. Accordingly, in certain cases, results for the period 1987–1998 were evaluated separately from those for the period 1999–2005 to determine whether accuracy of the prediction system has improved since 1999 following standardization of field trapping procedures and modifications to the SPB prediction chart.

Overall Prediction Accuracy at the State Level, Based on State Averages 1987–2005

Over the 19-year period (1987–2005), 245 State-level predictions of SPB infestation trend and population level were made by averaging the mean SPB per trap per day and percentage SPB for all locations monitored within each State for each year of participation in the Southwide survey. To determine overall accuracy of predictions at the State level, a 9 by 9 matrix was prepared to compare predicted versus actual infestation trends and population levels for every possible trend or level category (declining to low level, declining to moderate level, declining to high level, static at low level, static at moderate level, static at high level, increasing to low level, increasing to moderate level, increasing to high or outbreak level). Overall, the infestation trend and population level were both predicted accurately

Predicted ↓	Actual		
	Declining 105	Static 67	Increasing 73
Declining 107	80% 84/105	9% 6/67	23% 17/73
Static 72	10% 10/105	76% 51/67	15% 11/73
Increasing 66	10% 11/105	15% 10/67	62% 45/73
Total = 245	100%	100%	100%

Figure 5—Matrix showing accuracy of SPB trend predictions at the State level, based on 245 predictions over the 19-year period 1987–2005 in 16 States.

in 57 percent of the cases (140 of 245). The category of static or low was correctly predicted 85 percent of the time (45 of 53 cases) whereas the declining or low category was correctly predicted 73 percent of the time (60 of 82 cases). These two categories were the most common situations experienced across the South from 1987 to 2005, occurring in 55 percent of the total cases.

A 3 by 3 matrix was used to evaluate predicted and actual SPB trends alone (Figure 5). SPB infestations that increased by more than 25 percent from the previous year (increasing trend) occurred in 30 percent of the 245 cases and were correctly predicted 62 percent of the time. Those that declined by at least 25 percent from the previous year occurred in 43 percent of the cases and were correctly predicted 80 percent of the time. Static infestation trends (≤ 25 percent change from previous year) occurred in 27 percent of the cases and were correctly predicted 76 percent of the time.

A separate matrix was used for SPB population levels (Figure 6). Low SPB population levels occurred in 64 percent of the 245 cases and were correctly predicted 85 percent of the time. Moderate levels occurred in 26 percent

of the cases and were correctly predicted 54 percent of the time. In contrast, SPB populations reached high levels in 10 percent of the cases but were correctly predicted to do so only 32 percent of the time. However, 88 percent of those States that experienced high SPB infestation levels were predicted to have at least moderate levels. Major errors in both infestation trend and population level predictions occurred in only 3 percent of the cases. Major errors (those cases within red boxes in Figures 5 and 6) in trend predictions alone occurred in 11 percent of the cases (28 of 245 cases) whereas major errors in population level predictions alone occurred in only 2 percent of the cases (5 of 245 cases).

Accuracy of Predictions for Individual States, Based on State Averages: 1987–2005

Accuracy of SPB predictions at the State level for individual States for the period 1987–2005, determined by averaging data for all counties and ranger districts surveyed within the given State for a given year, is shown in Table 1. State-level trend predictions proved most accurate in Oklahoma, Texas, Mississippi, and Kentucky where accuracy exceeded 80

Predicted ↓	Actual		
	Low 157	Moderate 63	High 25
Low 160	85% 133/157	38% 24/63	12% 3/25
Moderate 70	14% 22/157	54% 34/63	56% 14/25
High 15	1% 2/157	8% 5/63	32% 8/25
Total = 245	100%	100%	100%

Figure 6—Matrix showing accuracy of SPB population level predictions at the State level, based on 245 predictions over the 19-year period 1987–2005 in 16 States.

Table 1—Accuracy of the SPB Prediction System by State: 1987–2005

State	Years Surveyed	Locations	Spots Detected	Correct Trend	Major Error Trend	Correct Level	Major Error Level
-----Percent-----							
OK	9	15	0	89	0	100	0
AR	19	123	17,004	79	5	84	0
TX	19	357	27,236	89	11	89	5
LA	19	340	30,182	79	0	79	5
MS	19	156	44,123	84	11	53	11
AL	19	97	99,956	68	5	58	0
KY	6	17	5,137	83	17	67	0
GA	19	173	50,121	58	21	63	5
TN	19	79	46,301	68	26	53	5
VA	19	61	28,706	79	11	89	0
FL	13	240	6,957	69	15	77	0
SC	19	390	156,058	58	16	47	0
NC	19	160	37,831	68	21	68	0
MD	18	51	455	78	17	83	0
DE	5	5	3	60	0	100	0
NJ	4	21	637	50	25	50	0
Total	245	2,285	550,707				
Mean				72.4	12.6	72.5	1.9
±SE				2.96	2.17	4.39	0.82

Note: Based on State averages, with Locations column indicating counties, parishes, and National Forest Ranger Districts surveyed.

Table 2—Accuracy of the SPB Prediction System for 12 States from 1987 to 1998

State	Years Surveyed	Locations	Spots Detected	Correct Trend	Major Error Trend	Correct Level	Major Error Level
-----Percent-----							
AR	12	65	17,004	67	8	75	0
TX	12	229	27,236	83	17	83	8
LA	12	181	30,182	75	0	67	8
MS	12	89	41,691	83	8	58	0
AL	12	52	47,193	67	8	42	0
GA	12	83	32,402	50	25	67	0
TN	12	39	12,710	58	17	58	8
VA	12	33	26,244	83	0	92	0
FL	6	69	1,632	67	17	83	0
SC	12	155	33,471	42	17	50	0
NC	12	70	25,978	67	25	67	0
MD	11	29	323	73	0	82	0
Total		1,094	296,066				
Mean				67.9	11.8	68.7	2.0
±SE				3.75	2.64	4.33	1.04

Note: Based on State averages, with Locations column indicating counties, parishes, and National Forest Ranger Districts surveyed.

percent. Accuracy of trend prediction was lowest for New Jersey (50 percent), South Carolina (58 percent), and Georgia (58 percent). Accuracy of predictions of SPB population level was greatest for Oklahoma (100 percent), Delaware (100 percent), Texas (89 percent), Virginia (89 percent), Arkansas (84 percent), and Maryland (83 percent). The least accurate predictions of population level were those for South Carolina (47 percent), New Jersey (50 percent), Tennessee (53 percent), and Mississippi (53 percent).

Correct predictions for all States combined averaged 72.4 percent for trend and 72.5 percent for level. It should be noted that Oklahoma, Delaware, Kentucky, and New Jersey have participated in the annual survey for less than 10 years.

Accuracy of Predictions at the State Level: 1987–1998 versus 1999–2005

We summarized State-level predictions for 12 States for the periods 1987–1998 and 1999–2005. We then compared the summarized State-level predictions for these periods to ascertain whether prediction accuracy at the State level has improved in recent years as a result of system modifications described above. Tables 2 and 3 show that the accuracy of SPB trend predictions for these States combined increased

from 68 percent prior to 1999 to 82 percent in subsequent years. Prediction of population levels increased in accuracy to a lesser extent, from 69 percent to 74 percent. Trend predictions declined in accuracy during the last 7 years only in Virginia. The accuracy of predictions of population level declined slightly for Mississippi, Tennessee, Virginia, Florida, and South Carolina and increased in the other States surveyed.

Discussion and Conclusions

The Southwide SPB Prediction System has proven to be a practical and relatively reliable means to forecast SPB activity early in the season. Three traps per county, parish, or ranger district appear sufficient to predict SPB infestation trends and probable severity at the local level. By averaging trap catch data for all counties and ranger districts monitored in a given year (20 to 100 traps per State), a reasonably reliable prediction of SPB trend and level for a given State can be achieved. There is a distinct relationship between the numbers of both SPB and clerids caught in early-season traps and subsequent SPB infestation trends during the same year, providing further evidence that *Thanasimus dubius* populations are SPB-density dependent and

Table 3—Accuracy of the SPB Prediction System for 12 States from 1999 to 2005

State	Years Surveyed	Locations	Spots Detected	Correct Trend	Major Error Trend	Correct Level	Major Error Level
-----Percent-----							
AR	7	58	0	100	0	100	0
TX	7	133	0	100	0	100	0
LA	7	159	0	86	0	100	0
MS	7	64	2,432	86	14	43	29
AL	7	42	52,763	71	0	86	0
GA	7	85	17,719	71	14	57	14
TN	7	34	33,591	86	14	43	0
VA	7	28	3,048	71	14	86	0
FL	7	168	4,953	71	0	71	0
SC	7	233	122,587	86	14	43	0
NC	7	90	11,853	71	14	71	0
MD	7	22	10	86	0	86	0
Total		1,170	254,733				
Mean	7			82.1	7.0	73.8	3.6
±SE				3.18	2.11	6.54	2.59

Note: Based on State averages, with Locations column indicating counties, parishes, and National Forest Ranger Districts surveyed.

are likely to be a major regulating factor in SPB population dynamics, as suggested by Moore (1972) and Turchin and others (1991, 1999).

Declining SPB populations tend to be more consistently predicted than are increasing populations, presumably because SPB and clerid populations are distributed across the pine-forested landscape more uniformly following an outbreak than they are at the beginning of the outbreak. Nevertheless, since its implementation in 1987, the prediction system has been used by State and Federal pest management specialists and foresters as a basis for scheduling aerial detection flights and identifying the need for suppression projects. Knowing when endemic SPB populations are about to explode or outbreak populations are soon to collapse is valuable information for more effectively managing this major forest pest.

Although 16 States currently participate in the system, levels of participation and accuracy differ from State to State. One State (Louisiana) prefers to use a single trap per parish and distribute available traps among more parishes. Interestingly, this approach doesn't seem to have substantially affected the accuracy of State-level predictions (Tables 1, 2, 3). Presumably, surveying more locations or

using more traps per location or both will lead to more reliable predictions, particularly in the early stages of a developing SPB outbreak when beetle populations are more scattered.

Continual validation using end-of-the-year detection records has enabled refinement of the system over the years. The prediction chart (Figure 3) has been modified since 1987 to better define the dividing lines between increasing or high, static or moderate, and declining or low infestations as more data points (both trap catch data and corresponding infestation trends and population levels) became available. Single sources of pheromone and turpentine have been used by all cooperators since 1999. In 2006, the standard turpentine dispenser (amber bottle and wick) was replaced in many cases by a polyethylene bag of turpentine from the same source (Hercules™). Since 2006, the standard turpentine dispenser (amber bottle and wick) has been replaced by a polyethylene bag containing 200 gm of 75 percent alpha-pinene: 25 percent beta-pinene (Synergy Semiochemicals Corp., Burnaby, British Columbia, Canada). This new dispenser is more practical for field application and has been found to be equally effective in eluting host volatiles and attracting SPB and clerids.

As with any prediction system, the SPB prediction protocol has strengths and weaknesses. The strengths include its simplicity, practicality, and relative accuracy. Data from a few strategically placed traps in early spring provide forest pest managers with insight into SPB infestation trends and levels for the remainder of the year at both local and large geographical scales. Currently, the standardized system is being implemented annually in 16 States, providing a consistent means to monitor SPB populations throughout the range of this pest in the Southern and Eastern United States. Limitations include the following:

The process of installing and monitoring traps and counting beetles is tedious and time-consuming. Numerous field crews with varying levels of training and experience are involved in setting out and monitoring the traps and identifying the beetles captured, which influences the quality of the data.

Pheromone traps placed too close to host trees may initiate infestations and inflate trap catches, leading to erroneous predictions, although such high trap catches are eliminated from the prediction data prior to analysis when infested trees are reported adjacent to a trap or trap catches exceed ca. 1,200 SPB per trap.

Variations in climate among States means that trapping data from Northern States is often not available until early June, 1 or 2 months after the results from Gulf Coastal States are completed.

In certain years, atypical weather patterns may accelerate or delay emergence of overwintering SPB and clerid populations, leading to erroneous predictions. For example, in 1989, an SPB outbreak occurred in Texas but was not forecasted. In that year, beetles emerged in large numbers in late April and May, after survey traps had been removed from the field (Billings, unpublished data). Of course, SPB outbreaks may originate in counties or locations not monitored with pheromone traps, a potential problem in those States that survey a limited number of counties (e.g., Mississippi, Alabama) or a single locality within each county or parish (e.g., Louisiana) each year. Also, predictions made in the spring for increasing SPB populations will prove wrong if adverse summer temperatures or other unanticipated factors limit SPB population growth. And,

finally, maintaining a reliable source of steam-distilled southern pine turpentine has been a problem over the years. Despite these limitations, the system has been deemed a success. This is reflected in the fact that Federal and State cooperators continue to use it.

Nevertheless, improvements in the SPB Prediction System are continually being sought. For example, monitoring SPB and clerid populations in the fall months for predictive purposes using the same protocol is currently being tested. Since 2003, the national forests in Mississippi have been conducting fall pheromone trapping during the secondary dispersal phase of SPB. These studies, though limited, have yielded relatively accurate predictions similar to those generated from the following spring survey (James Meeker, U.S. Forest Service FHP, 2500 Shreveport Highway, Pineville, LA 71360, personal communication). If effective, fall trapping would provide forest managers with additional lead time to prepare for peak infestations in the summer. The accuracy of forecasts based on fall insect numbers, however, may be dependent on weather conditions the following winter. Severe winter conditions such as freezes, prolonged cold temperatures, etc., could curtail the growth of SPB populations that were increasing in the fall, for example. In turn, SPB populations that are in the process of declining following a severe summer may be identified several months earlier if pheromone traps are deployed in the fall. Additional fall trials are needed across the Southeastern United States over multiple years to fully evaluate this approach. Indeed, a combination of spring and fall trapping may prove most effective for tracking seasonal and annual SPB population shifts.

Recent field research has demonstrated significantly enhanced attraction of SPB to trap sites additionally baited with the pheromone (+)*endo*-brevicomin (B. Sullivan, personal communication, East Texas Forest Entomology Seminar, Fall 2005). This finding offers a potentially new and alternative means of improving the precision, accuracy, and timeliness of the system. A bonus of the current SPB Prediction System is the wealth of historical trap-catch data from across the Southern United States over a 20-year period, which can help researchers better understand SPB and clerid dynamics (Tran and others 2007). Finally,

incorporation of SPB predictions into area-wide hazard maps (Billings and Bryant 1983; Billings and others 1985; Billings and others, this volume; Cook and others 2007) may lead to the development of more effective SPB risk maps. This would provide yet another tool for more effectively managing SPB.

Acknowledgments

The authors acknowledge the many cooperating State, university, and Federal pest management specialists for their dedicated contributions. The SPB Prediction System would not exist were it not for their willingness to set out and monitor traps, collect and count beetles annually in numerous counties and national forest ranger districts, and provide the data to the authors in a timely manner.

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Using Historical Photography to Monitor and Assess Threats Over Time

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Abstract

Analysis of aerial photography is perhaps the best way to assess changes in landcover conditions. In the United States, most national forests have repeat photography on approximately a 10-year cycle. Analysis of this rich photo record can reveal changes in insect damage, fuels buildup, unmanaged off-highway vehicle use, loss of open space, and other land-cover conditions. Current technologies now allow Forest Service employees to input photos into a geographic information system (GIS) accurately and easily, and important changes can be documented, analyzed, and evaluated within the GIS. This analysis discusses selected photo characteristics, available software solutions, and techniques for orthorectification of historical photos. It also discusses two cases in which historical imagery was used to document selected changing threat levels over time.

Keywords: Aerial photography, geographic information systems, loss of open space, orthorectification software, pseudocamera reports, unmanaged recreation.

Introduction

Remote sensing—aerial photography in particular—provides perhaps the best means to assess landcover changes over time. This is especially true when assessing changes over long periods of time. Aerial photography provides an invaluable record of land use and landcover conditions—frequently dating back to the 1930s. No other source of remotely sensed imagery is available prior to the 1970s. Most national forests have repeat photography on approximately a 10-year cycle. This rich photo record can indicate changing threat conditions through time, including threats such as insect damage, fuels buildup, unmanaged off-highway vehicle (OHV) use, and loss of open space.

We've long recognized and valued the quality information content of photos; however, we've also long struggled with how to get our photos into our Geographical Informa-

tion Systems (GIS). Current technologies now allow Forest Service employees to input photos into a GIS accurately and easily. Within the GIS, we can evaluate, analyze, and document important changes.

This analysis discusses selected photo characteristics, available software solutions, and techniques for historical photo orthorectification—along with examples using historical imagery to document selected changing threat levels over time.

The methods described in this paper are available throughout the Forest Service and provide the most consistent and objective method for detecting, assessing, and monitoring land-cover changes through time.

Why Photography?

Aerial photography, like all remote sensing technologies, has advantages and disadvantages compared with other remote sensing tools. The advantages of using photography to assess threats over time include a long history of repeat coverage, it is well understood, it is intuitive to the user community, it has excellent resolution, and it provides a superb stereo view.

The Forest Service has been systematically collecting aerial photography since the 1930s. Since that time, considerable effort and expense by developers have resulted in significant improvements, including faster and higher resolution films and film types, faster and higher resolution lenses and cameras, and more precise camera geometries and calibrations. On the users' side, there is a long history of effective use and technique development. These developments include a thorough understanding of photo geometry, stereoscopy, photogrammetry, and interpretation. Thus, the complexities of photography are well understood in the professional community.

Although photography is complex, it is simultaneously intuitive—both to the professional community, and to the at-large community. We've all grown up with photographs as part of our daily lives. We may not always understand the complexities and properties of the photography we see, but we still know how to interpret photographic images. Pho-

tography's intuitiveness promotes a greater understanding and acceptance of analytic results by the public—especially when the analysis techniques are easily understood.

Photography has excellent spatial resolution that is only now being challenged by other imaging systems. Even early aerial photography (1930s and 1940s), though not as resolute as post-WWII photography, has very good resolution. This allows easy comparison of vastly different dates of imagery.

An under-appreciated property of aerial photography is the ability to view the imagery in stereo. The stereo view drastically increases our ability to discern what we are viewing in the image and to perceive the topography and the varying heights of features on the ground. The stereo view has been the domain of the stereoscope and hardcopy photographs but is now becoming easily accessible in the digital domain—a trend that is certain to continue.

Why Not Photography?

Photography has its weaknesses, too. The major disadvantages of using photography to assess threats over time include: photography usually has to be converted from hardcopy (analog) to a digital format, it has non-map geometry, and it has a relatively small footprint.

All historical photography (meaning all photography before the current cycle of acquisition) is analog. Thus, one of the first steps required to get this imagery into a GIS is to scan each photo. This is a time-consuming operation affected by scanner speed, scan resolution, and the number of photographs. Time and cost of this step must be considered for each project.

The same geometry that allows for stereoscopic viewing is often viewed as a disadvantage when trying to get our photo-derived information into a GIS. Photography has single-point-perspective geometry. This means that a vertical photograph may appear map-like, but, features at differing elevations are displaced relative to one another, and the scale changes with each change in elevation. Time and money are required to orthorectify the photography (make the digital photo imagery accurately conform to a map coordinate system and projection).

A common trade-off with the various remote sensing systems is between footprint (area covered) and spatial resolution—photography is no exception. In general, the larger the area that is covered, the coarser the resolution. This holds true when comparing photography of different scales, and it is generally true when comparing photography to most satellite imagery. Compared with satellite imagery, photography is usually higher resolution but covers a much smaller area. Given the time and cost required for scanning and orthorectification of historical photography, the small footprint constrains the applicability of using photography to relatively small areas or to sampling over very large areas for detecting, assessing, and monitoring changes.

Tools for Getting Historical Photos in GIS

There is a fortunate convergence of technologies that makes it easier than ever to get historical photography into our GIS. The most important technologies are the newly available orthorectification software tools and the online data and camera reports. Also of value are the simple techniques for creating pseudocamera reports.

Although image orthorectification is not required to detect changes, orthorectification and subsequent analysis in GIS provides two major advantages:

1. Changes are much easier to detect using the built-in image display options in widely used GIS mapping applications.
2. Documenting the magnitude of change is much easier and can usually be objectively quantified.

Orthorectification Software Tools

Commercially available orthorectification software makes powerful softcopy photogrammetry tools available to all Forest Service personnel. This software has a significant learning curve, but it allows users to efficiently orthorectify large blocks (or sets) of overlapping photos.

Online Data (DOQs, DEMs, and Camera Reports)

In addition to the scanned photography, orthorectification requires horizontal and vertical reference data and information about the camera that acquired the photography. Typically, digital orthophotoquads (DOQs) are used for the horizontal reference, digital elevation models (DEMs) are

used for the vertical reference, and a USGS Optical Sciences Lab Camera Report is used for the camera information. All of this data is available online, allowing for far more efficient data preparation for orthorectification.

Creating Pseudo-Camera Reports

The USGS camera reports are not available for any resource photography prior to 1973. When one is needed, it is fairly easy to create your own pseudocamera report. The vital information needed from an official USGS camera report includes the principal point, the fiducial measurements, the fiducial orientation (with respect to the data strip), and the lens focal length. If we know the scan resolution of the scanned photo, the dimension of one pixel in the image is known (e.g., if the image was scanned at 600 dpi, each resulting pixel is 1/600 inch) and distances between fiducials on the image can be measured. We have created a simple spreadsheet that converts imaged fiducial file coordinates to millimeters in a camera coordinate system similar to USGS camera reports. For these pseudocamera reports, we have to assume the principal point is at coordinate 0,0 and we initially assume the focal length is 6 inches (152.4 mm). The fiducial orientation can be defined by the user—but then has to be applied consistently.

Obviously, these pseudocamera reports are not as accurate as official camera reports. The spatial accuracy of the resulting orthorectified photography will likely be less accurate as well. However, for many resource applications, satisfactory accuracies can be obtained.

Analysis Options

Once the multiple dates of photography have been orthorectified, there are few limits to analysis options. The imagery itself, with little or no analysis, may be sufficient for public awareness, although some interpretation is usually helpful. To detect and document spatially continuous changes over time (e.g., vegetation cover), we have used a digital dot grid technique that provides percentage of cover by cover type for each cycle of imagery. For discrete feature changes, such as OHV trails, we have used two different techniques. Our first method used a digital grid that required the interpreter to simply click a cell of the grid if an OHV trail was in the

cell. The grid was then put through a kriging process, and a map of OHV hot spots was created. The second method consisted of manually digitizing the trails as they were interpreted from the imagery. The kriging method has the advantage of being very fast. The digitizing method has the advantages of being simple, easily verifiable, and producing a map of actual OHV trails. In addition, it was a simple matter to construct graphs of the rate of change from the digitized trail's attribute tables. These methodologies have worked well for us, but the optimum methodologies may yet be awaiting discovery.

Only the most basic interpretations should be attempted from the monoscopic imagery. Most interpretations should be accomplished in stereo, which allows users to visualize relief within the image and heights of features. This can be achieved using a stereoscope and the analog (hardcopy) imagery or digitally using commercially available applications.

Examples

Following are two brief examples of using historical photography to detect and document changing conditions over time.

Loss of Open Space

Aerial photography from 1969, 1977, 1988, and 1993 of a small portion of the Black Hills National Forest boundary were orthorectified using the orthorectification tools in Arc-Map Image Analysis. Roads and buildings were digitized from each cycle of photography (within the chosen subset, there are no roads or buildings visible in 1969). Features digitized from each cycle of photography were symbolized with different colors. The results (Figure 1) clearly depict significant human-made changes that fragment the open space around the national forest boundary.

Unmanaged Recreation

Leica Photogrammetry Suite (LPS) was used to orthorectify photography from 1967, 1976, 1988, and 1995 over a small portion of the Fishlake National Forest. In addition, we added orthorectified NAIP imagery from 2004. The 1967 photography did not have a camera report, so we

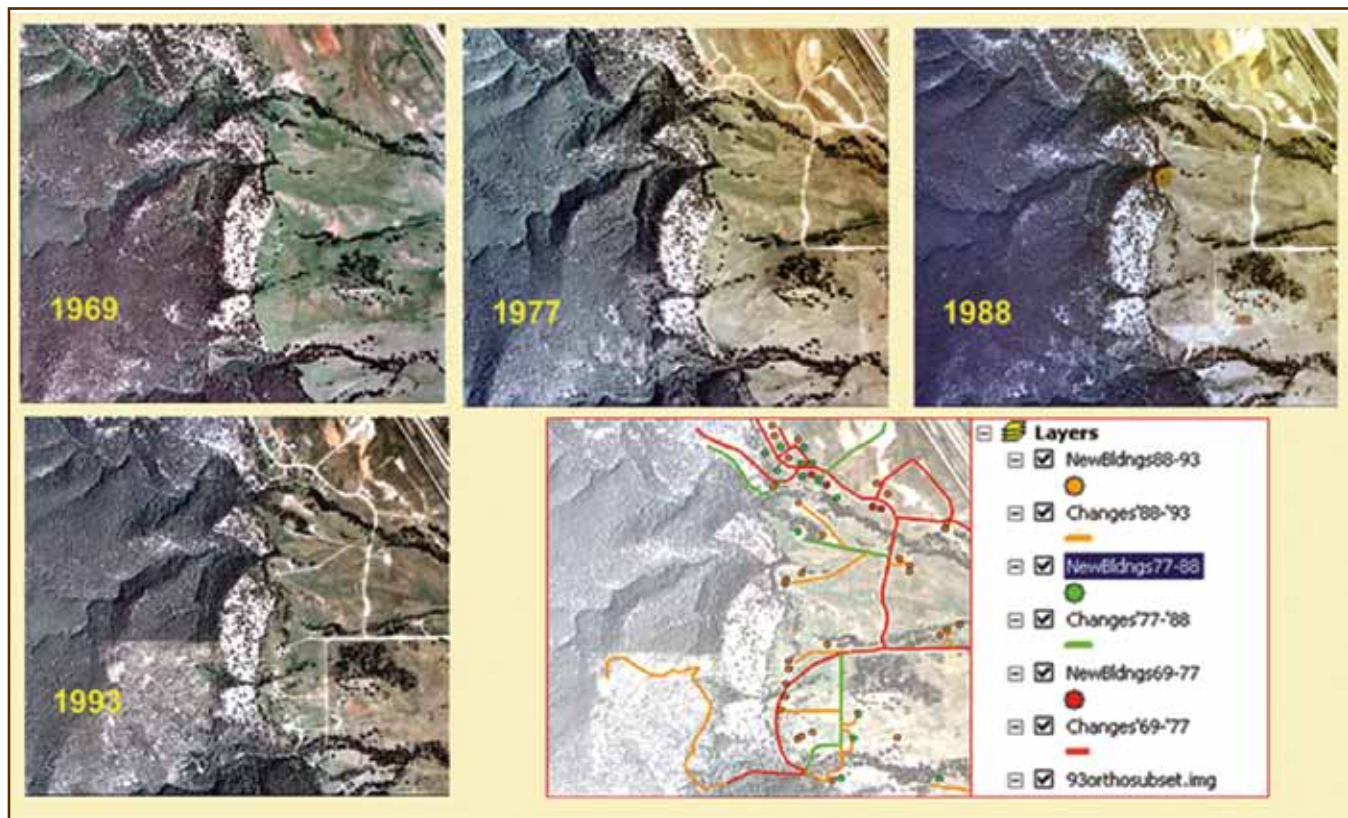


Figure 1—Aerial photography from 1969, 1977, 1988, and 1993 over a portion of the Black Hills National Forest was orthorectified. New roads and buildings were identified and digitized for each cycle of photography. The result (bottom-right of graphic) was a compelling graphical documentation of 24 years of loss of open space.

created one using methods described in above. Once all of the photography was orthorectified, we added the existing travel management plan roads layer. Then we digitized new OHV trails as they became apparent on each cycle of photography (Figure 2). This methodology was simple, easily verifiable, and produces a map of actual OHV trails. In addition, it was a simple matter to construct graphs of the rate of change from the digitized trail's attribute tables (Figure 3).

Resources

Where to Find Old Photos

Typically, each national forest receives copies of every photo acquisition over that national forest. However, there is no standard for archiving historical aerial photography. In too many cases, these old photographs are discarded by those who don't see their continued value. Start your search

locally and expand to the regional geospatial or remote sensing leader.

If photos are not available at the regional level, the USDA Aerial Photography Field Office (APFO) in Salt Lake City archives over 50,000 rolls of film. Their holdings include nearly all USDA photo acquisitions since the mid-1950s. When searching for historical photography within the APFO Web site, be aware that they have separate listings for Forest Service photography and all other USDA agencies.

Older photography is held in the National Archives in College Park, MD.

Training/Help

Countless sources for training and assistance have public access, but the following suggestions are tailored for the Forest Service and provided behind the Forest Service firewall.

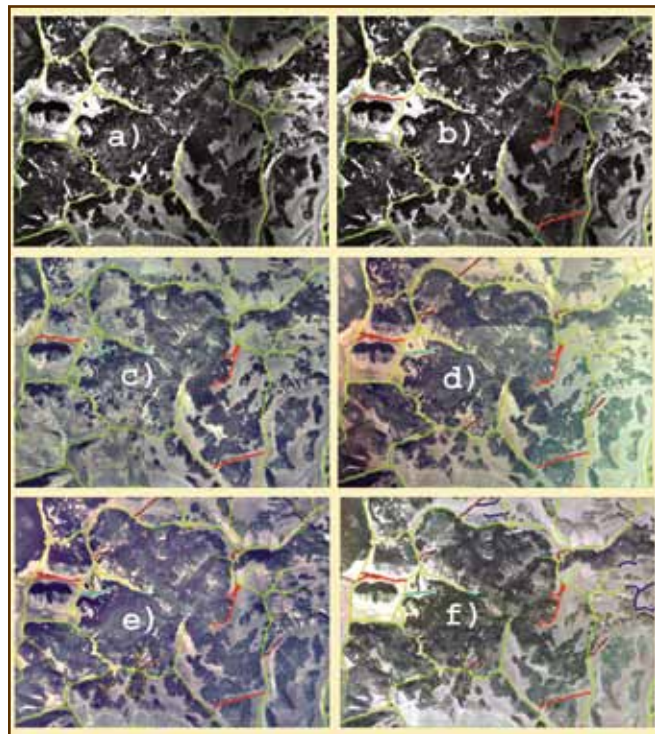


Figure 2—Aerial photography from 1967 (Figures 2a and 2b), 1976 (Figure 2c), 1988 (Figure 2d), and 1995 (Figure 2e) was orthorectified, and 2004 NAIP photography (Figure 2f) was obtained to assess unmanaged recreation over a portion of the Fishlake National Forest. The official managed trails were obtained and overlaid on each image (light green). Unmanaged recreation trails were identified on each cycle of photography and digitized from the orthorectified imagery.

The USDA Forest Service Remote Sensing Applications Center (RSAC) is an excellent source for training and help. The RSAC provides instructor-led courses in using the Leica Photogrammetry Suite—including a section on creating and using pseudocamera reports. There is no tuition for the course. Course offerings and registration are available from the Forest Service GeoTraining FS Web site (intranet) (<http://fsweb.geotraining.fs.fed.us/>). Online help for Imagine and Image Analysis is also available on this site.

The RSAC staffs the Remote Sensing Helpdesk. Access is through the Forest Service End User Support Center (EUSC). Specify that you have a remote-sensing-related question to be forwarded to the Remote Sensing Helpdesk.

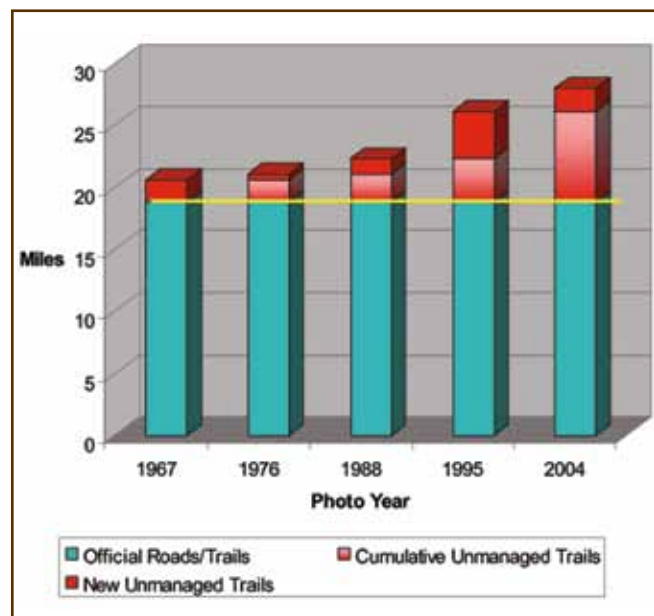


Figure 3—This bar graph documents the increase in unmanaged OHV trails for our project area on the Fishlake National Forest, 1967-2004. Official managed trails are graphed in cyan (and highlighted with the yellow line), newly interpreted unmanaged trails are shown in solid red, and the cumulative unmanaged trails are shown in the red gradient. Note: in 2004 there were approximately 9 miles of cumulative unmanaged recreation trails—a 50-percent increase over the official managed trails.

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Integrating Natural Disturbances and Management Activities to Examine Risks and Opportunities in the Central Oregon Landscape Analysis

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Abstract

We used state and transition models to integrate natural disturbances and management activities for a 275 000-ha landscape in the central Oregon Cascades. The landscape consists of a diverse mix of land ownerships, land use allocations, and environments. Three different management scenarios were developed from public input: (1) no management except wildfire suppression on federally managed lands, (2) manage Federal lands to increase multistory forests of large and very large trees, and (3) manage Federal lands to move toward historical conditions. All scenarios treated privately owned lands as if they were wildland-urban interface (WUI) areas and all recognized wilderness, reserves, and general forests within federally managed lands. Models were run for 200 years and 30 Monte Carlo simulations to include variability in fire years and other natural disturbances. Passive management on federally managed lands resulted in small increases in single-story and multistory large-tree forests and increases in high-severity wildfire and insect outbreaks. Managing toward multistory large- and very-large-tree forests resulted in minor increases in those forest types and increased wildfire and insect outbreaks. Contrary to intent, this scenario did not generate appreciable increases in multistory large- and very-large-tree forests. Managing toward historical conditions resulted in strong increases in single-story large- and

very-large-tree forests and decreases in high-severity wildfire and insect outbreaks. All three scenarios resulted in conversion of most WUI to open grass, shrub, and forest conditions.

Keywords: Forests, landscape ecology, management, modeling, natural disturbances, Oregon.

Introduction

Management of diverse landscapes in the interior Pacific Northwest requires consideration of the integrated effects of natural disturbances and management activities on natural resource conditions. The opportunities for managing lands depend on widely varying objectives of owners, vegetation conditions, environmental settings, natural disturbances, and other factors. Likewise, the risks that land managers encounter include natural disturbances, unforeseen consequences of management activities, changing political, social, and economic environments, and others. Land managers and those who influence or set land management policy need to examine the short- and long-term potential effects of different management approaches using methods that (1) integrate the effects of natural disturbances and management activities on vegetation and resource conditions; (2) consider landscapewide characteristics and trends across all ownerships; (3) maximize the effects of limited budgets and personnel through cooperation across agencies and ownerships; (4) use a modeling approach that is flexible, powerful, easy to understand, and integrative.

A partnership of Federal and State agencies and nongovernment organizations developed a shared effort to generate landscapewide vegetation data, landscape models, and related information. The Interagency Mapping and Assessment Project (IMAP) addresses several landscape assessment and analysis issues, including (1) limited and declining funds to perform landscape assessments and analyses of potential effects of various management options on resources of interest; (2) an increasing lack of highly skilled people to perform landscape analyses; (3) a desire to avoid conflicting answers to broad questions that

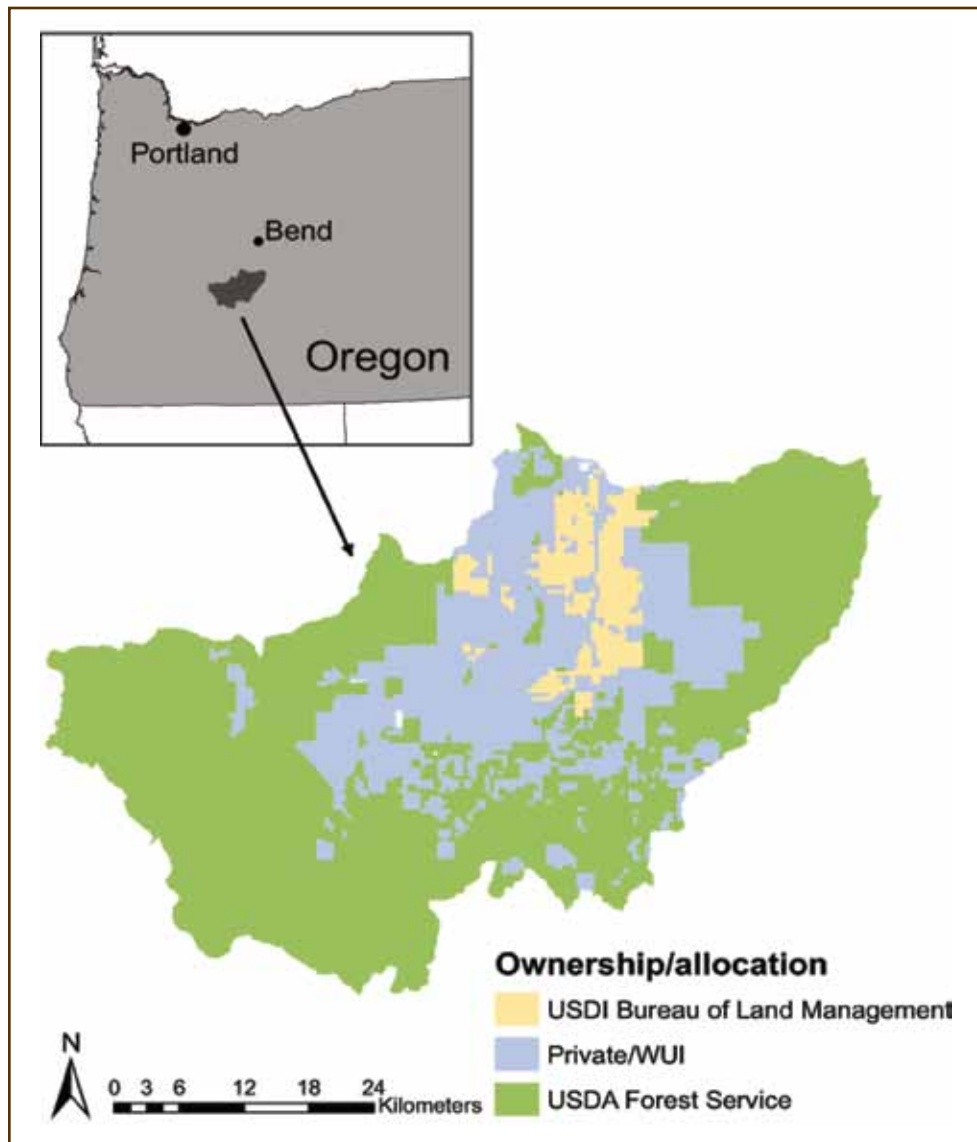


Figure 1—The Five Buttes study area in central Oregon, U.S.A. WUI = wildland-urban interface.

cross ownerships and interests; (4) the need for integrated analyses that include many management and natural disturbances across a broad range of ownerships, vegetation conditions, and environments; (5) a consistent basis for monitoring the effectiveness of management activities at achieving policy goals across large landscapes; and (6) the desire for relatively simple and understandable approaches to landscape analysis and policy evaluation. Key issues for all these landscape analysis, planning, and assessment activities include, among others, fire risks, forest conditions,

wildlife habitats, old forests, and timber products. In addition, policymakers and others want to consider long-term sustainability of landscape resources and conditions given various management approaches.

Landscape simulation models may be used to assist in understanding the potential reaction of large landscapes to various management and policy approaches (e.g., Bettinger and others 2005, Hann and others 1997, Hemstrom and others 2004, Mladenoff and He 1999, USDA and USDI 2000). Advances in modeling techniques, computer technology, and geographic information systems (GIS) have made

it possible to model large landscapes at increasingly finer scales of spatial and temporal resolution (Barrett 2001, Bettinger and others 2005). In much of the Pacific Northwest of North America, resource planning models have focused primarily on conifer succession and management while representing other ecosystem elements as byproducts (e.g., Alig and others 2000, Johnson and others 1986). Although progress has been made in the formulation of multiobjective goals in landscape simulations (Sessions and others 1999, Wedin 1999), there remain many challenges to building landscape planning models that include all of the important disturbance processes that influence change. The net, synergistic effects of various disturbances (e.g., drought, fire, insects, and management activities) across a large, ecologically diverse landscape are of particular interest to policymakers, scientists, land managers, and others. Our approach treats vegetation as discrete types and management activities and natural disturbance as transitions among those types to project the long-term net effects of alternative management scenarios across a large landscape, building on the work of Hann and others (1997) and Hemstrom and others (2004). Although we do not specifically include drought and other climatic effects, their impacts are manifest in our annual wildfire and insect probabilities.

Study Area

The study area consisted of about 276 000 ha in seven watersheds in the southern portion of the upper Deschutes subbasin (Figure 1). Vegetation ranged from low-elevation shrublands, meadows, ponderosa pine and lodgepole pine forest to high-elevation parkland and spruce-fir and mountain hemlock forests. Ownerships were mixed and include about 142 000 ha of Federal general forest, 24 000 ha of Federal late-successional forest reserves established by the Northwest Forest Plan (USDA and USDI 1994), 51 000 ha of wilderness and similar areas, and 59 000 ha of private lands.

Private lands constituted about 28 percent of the area (about 64 000 ha). These could be managed with a wide variety of treatments. For the purposes of this exercise, however, we assumed that private lands were a proxy for

wildland-urban interface areas (WUI). WUI was an important stratification because fuel treatments were generally the highest priority management activity on private lands in this landscape. A consequence of our use of private lands as a surrogate for WUI was a potential overestimate of the rate of fuel treatments and an underestimate of other treatments on private lands.

Reserves were publicly owned lands (usually managed by the Forest Service or BLM) designated for special consideration and management (3 percent or about 7000 ha). These were usually late-successional reserves under the Northwest Forest Plan (USDA and USDI 1994) or other similar areas. Under some conditions, they may be managed with thinning or other fuel-reduction treatments. However, they were generally designated to maintain old forest structure and similar conditions. Wilderness was legally designated land managed for natural characteristics and included wilderness, state parks, and similar areas (15 percent or about 34 000 ha). Only natural disturbances (wildfire and insect/disease activity) were modeled in wilderness.

We recognized eight vegetation types based on maps provided by the Deschutes National Forest, and, for gaps in those data, information gathered during the Interior Columbia Basin Ecosystem Management Project (Hann and others 1997). These ranged from the lowest elevation juniper (*Juniperus occidentalis* Hook.) woodlands to alpine parklands:

1. Juniper woodland—shrub steppe areas generally capable of supporting grass, shrubs, and juniper but not closed forest.
2. Dry ponderosa pine (*Pinus ponderosa*)—areas capable of supporting ponderosa pine forests but generally not Douglas-fir (*Pseudotsuga menziesii*) or other tree species. These were transitional between forest and juniper woodland or shrub/steppe.
3. Mixed conifer dry—grand fir (*Abies grandis*) and Douglas-fir forests at lower elevations and in relatively dry environments. Historically, these areas consisted mostly of large, open, ponderosa pine stands maintained by frequent ground fire (average 10- to 20-year fire return interval).

Table 1—Forest structure class definitions for the Five Buttes study area, central Oregon, U.S.A.

Structure class	Tree canopy layers	Overstory canopy cover	Dominant tree d.b.h.
		<i>Percentage</i>	<i>cm</i>
Grass forb	None	Tree <10, shrub <15	NA
Shrub	None	Tree <10, shrub >15	< 2.5
Seedlings/saplings	1	≥ 10	≥ 2 to 13
Pole tree – open	1	≥ 10 to 40	≥ 13 to 25
Pole tree – medium	1	≥ 40 to 70	≥ 13 to 25
Pole tree – closed	1	≥ 70	≥ 13 to 25
Small tree – open	1	≥ 10 to 40	≥ 25 to 38
Small tree – medium	1+	≥ 40 to 70	≥ 25 to 38
Small tree – closed	1+	≥ 70	≥ 25 to 38
Medium tree – open	1	≥ 10 to 40	≥ 38 to 51
Medium tree – medium	1+	≥ 40 to 70	≥ 38 to 51
Medium tree – closed	1+	≥ 70	≥ 38 to 51
Large tree – open	1	≥ 10 to 40	≥ 51 to 76
Large tree – medium	1+	≥ 40 to 70	≥ 51 to 76
Large tree – closed	1+	≥ 70	≥ 51 to 76
Very large tree – open	1	≥ 10 to 40	≥ 76
Very large tree – medium	1+	≥ 40 to 70	≥ 76
Very large tree – closed	1+	≥ 70	≥ 76

4. Mixed conifer moist—forests dominated by a variety of conifer species, including ponderosa pine, Douglas-fir, grand fir, sugar pine (*Pinus lambertiana* Dougl.), incense-cedar (*Calocedrus decurrens* (Torr.) Florin), western larch (*Larix occidentalis* Nutt.) and others. Under historical conditions, these somewhat wetter areas had less frequent natural fire than the dry mixed conifer type and were often dominated by large, widely spaced ponderosa pine.
5. Lodgepole pine (*Pinus contorta*) dry—lodgepole pine stands growing primarily on pumice soils. Soil and microsite conditions restricted other conifer species.
6. Upper montane cold—high-elevation forests dominated by Engelmann spruce (*Picea engelmannii*), mountain hemlock (*Tsuga mertensiana*), grand fir, subalpine fir (*Abies lasiocarpa*), lodgepole pine, and other species. This type occurred mostly within reserves or wilderness.
7. Upper montane moist—high-elevation forests that largely reflected westside climatic influences. Pacific silver fir (*Abies amabilis*), noble fir (*Abies procera*), Douglas-fir, and other species generally dominated.
8. Subalpine parkland—high-elevation mosaics of tree islands, alpine shrublands, and grasslands largely within reserves or wilderness.

Methods

We developed 337 combinations of vegetation structure classe (Table 1) and cover type (Table 2) to represent existing and potential future vegetation conditions. Cover types were based on the dominant species in the uppermost canopy layer and included several categories of developed land (e.g., urban, agriculture, etc.). Structure class depended on the size, tree density per unit area, and canopy layering for forests or on the dominant life form and canopy cover for shrublands and grasslands. Our structure classification was carefully designed to address important issues regarding wildlife habitats, fire and fuels, and various commercial forest products. Combinations of cover type and structure

Table 2—Forest cover type classes used in the Five Buttes study area, central Oregon, U.S.A.

Cover type	Dominant species
Not vegetated	None – rock, water, ice, etc.
Developed land	Variable – agriculture, suburban, urban, etc.
Grass/shrub	Various grass, forb, and shrub species
Juniper	Western juniper (<i>Juniperus occidentalis</i> Hook.)
Ponderosa pine	Ponderosa pine (<i>Pinus ponderosa</i> P. & C. Lawson)
Douglas-fir/white fir	Douglas-fir (<i>Pseudotsuga menziesii</i> (Mirbel) Franco) and white fir (<i>Abies concolor</i> (Gord. & Glend.) Lindl. ex Hildebr.)
Grand fir	Grand fir (<i>Abies grandis</i> (Dougl. ex D. Don) Lindl.), Douglas-fir, and other conifers
Lodgepole pine	Lodgepole pine (<i>Pinus contorta</i> Dougl. ex Loud.)
Pacific silver fir	Pacific silver fir (<i>Abies amabilis</i> (Dougl. ex Loud.) Dougl. ex Forbes), noble fir (<i>Abies procera</i> Rehd.), and Douglas-fir
Mixed conifer	Variable mixtures of white fir, Douglas-fir, Engelmann spruce (<i>Picea engelmannii</i> Parry ex Engelm.), mountain hemlock (<i>Tsuga mertensiana</i> (Bong.) Carr.), and other conifers at upper elevations
Subalpine parkland	Mosaic of subalpine fir (<i>Abies lasiocarpa</i> (Hook.) Nutt.), mountain hemlock, and Engelmann spruce at high elevations

class within potential vegetation types formed the basic vegetation state-classes in our models.

Current vegetation data was developed using Gradient Nearest Neighbor (GNN) methods as described by Ohmann and Gregory (2002). This process imputed approximately 1,600 inventory plots to 30-m pixels using a statistical relationship between LANDSAT-TM imagery and other geographic data and inventory plots. In general, GNN methods are best at predicting forest structure (e.g., diameter of dominant and codominant trees) but less accurate for canopy tree species (Ohmann and Gregory 2002). Correlation between predicted and observed quadratic mean diameter of dominant and codominant trees in the Oregon Coast Range was about 0.8, whereas that for tree species richness was about 0.53 (Ohmann and Gregory 2002). Where GNN data were unavailable, we used vegetation composition and structure attributes from Oregon GAP (2006). Cover and structure data were summarized to state-classes within strata of watershed, ownership/land allocation, and potential vegetation type. These estimates of area by state-class by stratum were the initial conditions for our modeling process.

We used state-and-transition models to project the integrated effects of natural disturbances and management treatments on vegetation. Vegetation composition and structure within plant association strata defined each state.

States were connected by transitions that indicated either the effect of successional vegetation development over time, or the effect of disturbance (Hemstrom and others 2004). This approach expanded transition matrix methods and represented vegetation development as a set of transition probabilities among various vegetative states (Cattalino and others 1979, Hann and others 1997, Horn 1975, Keane and others 1996, Laycock 1991, Noble and Slatyer 1980, Westoby and others 1989). For example, grass/forb communities might be dominated by closed forest following tree establishment over a period of time or might remain as grass/forb communities following wildfire. Alternatively, management activities or low-severity wildfire may generate more open forest conditions. State changes along the successional, time-dependent paths were usually deterministic, and, without disturbance or management, all the vegetation could ultimately accumulate in one state. Different management scenarios were developed to represent alternative landscape objectives and, hence, management treatments.

We developed and ran our models with the Vegetation Dynamics Development Tool (VDDT) (Beukema and others 2003). VDDT has been used in several landscape assessments and land management planning efforts in the Interior Northwestern United States (e.g., Hann and others 1997, Keane and others 1996, Merzenich and others 2003)

and elsewhere (Hann and Bunnell 2001, Merzenich and Frid 2005). Although VDDT is a nonspatial model, managers and others often need to understand the spatial distribution of vegetation conditions and disturbances. Consequently, we ran models using strata of land ownership and allocation and potential vegetation types within watersheds so that we could display results about the spatial distribution of landscape characteristics without implying pixel or stand-level accuracy. All scenarios were run for 200 years with 30 Monte Carlo simulations to allow the occurrence of rare events and generate estimates of long-term disturbance variability and forest development trends. We compared decadal average area treated with different treatments and disturbances across our three scenarios to examine trends that would have been more difficult to visualize in highly variable annual outputs. Average annual area in various forest types, however, was not as variable and was displayed on a yearly basis.

Forest Growth and Management Treatments

Our models include a set of assumptions and definitions that form the basis of transition rates and directions. In general, transition rates and directions were developed from a combination of inventory data and the Forest Vegetation Simulator (FVS) (Dixon 2002), the published literature, and, where necessary, expert opinion. The inventory data were tree lists from plots collected as part of the Forest Inventory and Analysis (FIA) (Barrett 2004) and Continuous Vegetation Survey (CVS) (Max and others 1996) inventories collected by the USDA Forest Service. There are over 1,600 inventory plots in the larger landscape study area. Each of these plots was assigned to one of our VDDT model state-classes, and FVS was used to project the rate and direction of growth transitions. We also modeled a set of management activities using FVS and the inventory data to estimate yield streams from management activities (Hemstrom and others 2006). We used a fixed set of silvicultural treatments to model our scenarios. The treatments we used were simplified in terms of timing, exact effects at the stand level, and other factors compared to the full suite of treatments that might be applied. However, based on discussions with local land

managers and silviculturists, our treatments represent typical, commonly implemented kinds of activities that might occur on the various ownerships and allocations in the study area. Management treatments included:

1. Regeneration harvests on private lands only.
2. Salvage following stand-replacement wildfire or insect outbreaks on Federal general forest and private land, but not in reserves or wilderness.
3. Tree planting in areas that had been regeneration harvested or salvaged.
4. Precommercial thinning from closed to open condition at age 15.
5. Commercial thinning of trees across all diameter classes to reduce stand density to open structure. We assumed that trees greater than 51 cm in diameter at breast height (d.b.h.) could be harvested on private lands but not on lands administered by the UDSA Forest Service, a reflection of current management policy.
6. Partial harvest was commercial thinning from below in closed stands to reduce stand density, favor fire-resistant tree species (e.g., ponderosa pine) and increase average tree diameter.
7. Mechanical treatments to reduce fuels were applied to closed stands beyond the age of precommercial thinning. Closed stands were converted to open, low-density conditions.
8. Prescribed fire was underburning applied to low-density stands of fire-tolerant species (e.g., ponderosa pine) to maintain open stands of fire-tolerant tree species. We assumed a small portion of these inadvertently became mixed or high-severity fires.

For the purposes of reporting, we combined regeneration, partial harvest, and commercial thinnings into a commercial harvest category that might produce enough saw-log-sized material to be of commercial interest. We combined precommercial thinning and mechanical fuel treatment into noncommercial harvest because the majority of material available from treatment would likely be too small to be used for sawtimber.

Table 3—Average wildfire return intervals under reference conditions (prior to 1850) for potential vegetation groups in the Five Buttes study area, central Oregon, U.S.A. (from LANDFIRE 2006)

Potential vegetation type	Fire severity class	Average fire interval	Average annual fire probability	LANDFIRE rapid assessment model
		<i>Years</i>		
Juniper	Replacement	1000	0.001	R#JUPlse Western Juniper Pumice
	Mixed	500	0.002	
	Surface	NA		
	All	333	0.003	
Ponderosa pine dry	Replacement	125	0.008	R#PIPOm Dry Ponderosa Pine - Mesic
	Mixed	50	0.02	
	Surface	8	0.125	
	All	7	0.153	
Mixed conifer dry	Replacement	115	0.0087	R#MCONdy Mixed Conifer - Eastside Dry
	Mixed	75	0.0133	
	Surface	25	0.04	
	All	16	0.062	
Mixed conifer moist	Replacement	200	0.005	R#MCONms Mixed Conifer - Eastside Mesic
	Mixed	150	0.0067	
	Surface	400	0.0025	
	All	71	0.0142	
Lodgepole pine dry	Replacement	125	0.008	R#PICOpu Lodgepole Pine - Pumice Soils
	Mixed	450	0.0022	
	Surface	NA		
	All	98	0.0102	
Upper montane cold	Replacement	185	0.00541	R#ABLA Subalpine Fir
	Mixed	800	0.0013	
	Surface	NA		
	All	150	0.0067	
Upper montane moist	Replacement	500	0.002	R#ABAMup Pacific Silver Fir--High Elevation
	Mixed	1100	0.0009	
	Surface	NA		
	All	344	0.0029	
Subalpine parkland	Replacement	350	0.0029	R#ALME Alpine and Subalpine Meadows and Grasslands
	Mixed	750	0.0013	
	Surface	NA		
	All	239	0.00420	

Wildfire

We used wildfire probabilities for historical or reference conditions from the interagency LANDFIRE effort (LANDFIRE 2006). Reference conditions were assumed by LANDFIRE (2006) to be the disturbance and vegetation characteristics that existed over a long period of time prior to about 1850 and, consequently, prior to wildfire suppression (Table 3). Current wildfire probabilities were estimated

from discussions with local fire managers and other experts. In keeping with the estimates provided by LANDFIRE (2006), we split wildfire into three severity levels, depending on the degree of mortality in aboveground vegetation: (1) 0- to 25-percent mortality was low severity, (2) 25- to 75-percent mortality was mixed severity, and (3) more than 75-percent mortality was high severity. We recognized that wildfire probabilities, and those of insect outbreaks, reflected larger externalities such as regional drought and

Table 4—Fire-year sequences in the entire upper Deschutes subbasin (approximately 800,000 ha) and for individual watersheds (approximately 50,000 ha).

Analysis area	Size	Fire-year type		
		Normal	High	Severe
Upper Deschutes subbasin	Fire-year frequency	80%	15%	5%
	Area multiplier	1	40	500
Huc5 watersheds	Fire-year frequency	95%	4%	1%
	Area multiplier	1	160	2000

ignition patterns but did not explicitly include drought effects.

We used random streams of fire years to model year-to-year variability in area burned owing to the effects of weather, fuel conditions, and chance. For example, regional climate may produce a series of dry years with abundant lightning ignitions. In this case, wildfire probabilities would be substantially higher compared to average years. Our modeling process included randomly drawn sequences of normal, high, and extreme wildfire years to simulate annual variability in natural disturbances. We also assumed that wildfire variability changes with landscape scale, being higher in small areas (e.g., watersheds of 14 000 to 90 000 ha) and lower in large areas (e.g., the Deschutes subbasin of over 800 000 ha). The LANDFIRE (2006) wildfire probabilities were developed for very large landscapes with, consequentially, relatively low annual variability. After discussions with local fire experts and examination of the relatively few data available, we assumed that 80 percent of years experience normal or average amounts of wildfire, 15 percent experience high amounts of wildfire, and 5 percent experience extreme amounts of wildfire at the scale of the entire upper Deschutes subbasin. Essentially, for every hectare burned in a normal year, 16 ha burned in a high year, and 200 ha burned in a severe year. At the scale of watersheds, however, we assumed that even in severe years, most wildfire would occur in large fires that impact only a few watersheds but burn most of the area within affected watersheds. We assumed that 95 percent of years produced normal amounts of wildfire, 4 percent produced

high amounts, and 1 percent produced extreme amounts at the watershed scale (Table 4). In essence, for every hectare burned in a normal year at the watershed scale, 65 ha burned in a high year, and 810 ha burned in a severe year.

Wildfire probabilities were set by vegetation cover, structure, and potential vegetation type. However, wildfire probability is not just a function of the vegetation conditions in a single state class. We used landscape condition feedback to increase overall wildfire probabilities when the landscape contained abundant dense forests and to decrease them when overall forest conditions were more open. We assumed that abundant dense forests meant highly contiguous fuels and increased landscape susceptibility to large, difficult-to-suppress wildfires.

Management Scenarios

In 2005, we held meetings in Bend and Klamath Falls, Oregon, to develop a set of management scenarios. Local members of the public and representatives from government land management agencies were invited to help us develop reasonable alternatives that might address differing perspectives about how Federal lands in the area might be managed. We used the results of these meetings to design four management scenarios for modeling. Because fuel and fire hazards in WUI were an overriding concern, fuel treatments in WUI described for scenario one were included in all three scenarios.

Scenario 1—

Active Fuel Treatment in Wildland/Urban Interface, No Management on Federal Lands

The primary emphasis was to actively treat fuels on private land (WUI). At least 25 percent of the dry lodgepole pine area was treated with partial harvests, precommercial thinning, mechanical treatment, or commercial thinning per decade. We assumed that mechanical treatments to maintain reduced fuel levels would be used on private lands, rather than prescribed fire. The long-term objective was to maintain the level of medium and dense stands on private lands at less than 10 percent of the total. No treatments other than continued fire suppression occurred on public lands.

Scenario 2—**Active Fuel Treatment in Wildland/Urban Interface, Maximize Multistory Large Tree Forests on Federal Lands**

Federal lands were managed to produce large trees and increase habitat for wildlife species associated with multi-storied stands that contained many large and very large trees (more than 51 cm in d.b.h.). With fire suppression, an understory tree layer will develop naturally on most forested environments in the study area. In these areas, early management activities included thinning prescriptions to create and maintain open stands of fire-tolerant tree species that could grow to a large size relatively quickly while reducing risk of loss to high-severity wildfire. In addition, stands dominated by large and very large trees were infrequently thinned from below to reduce stand density while retaining some smaller trees. Our treatment regime on federally managed general forest lands included:

1. Precommercial thinning of all stands at age 15.
2. Treating 5 percent of high-density and 2.5 percent of medium- and high-density stands in ponderosa pine, mixed conifer dry, and mixed conifer moist environments each year after the initial precommercial thinning to maintain open conditions until trees reached large size. After stands reached large-tree size, thinning ceased to allow development of understory trees until the stands became very-large-tree sized.
3. Lightly thinning from below in dense stands of very large trees in mixed conifer dry and mixed conifer moist types at an annual rate of 10 percent to reduce fire and insect losses while maintaining most of the multistory structure.
4. Alternately thinning and underburning open stands of smaller trees in ponderosa pine dry, mixed conifer dry, and mixed conifer moist types to reduce fuels.
5. Mechanically thinning lodgepole pine dry stands at a rate of 4 percent annually.
6. Salvaging dead wood in 25 percent of the stands that had experienced wildfire and insect outbreaks.

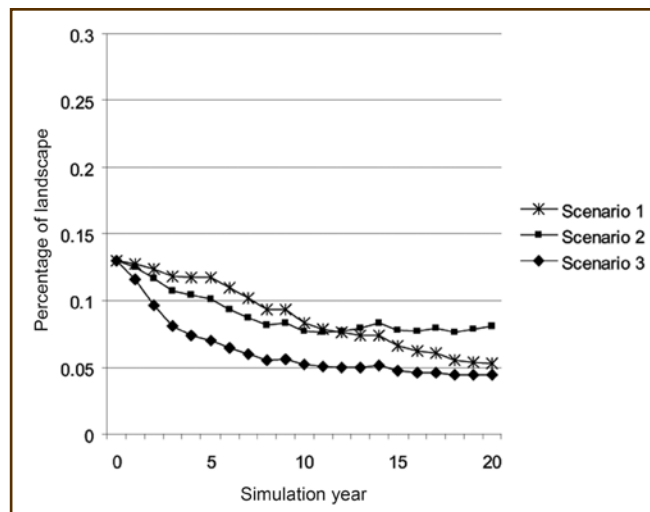


Figure 2—Percentage of the landscape in multistory large and very large tree structure classes by scenario in the Five Buttes area, central Oregon, U.S.A.

7. Treating reserves at one-half these rates and wilderness not at all.

Scenario 3—**Active Fuel Treatment in Wildland/Urban Interface, Move Federal General Forest Lands Toward Historical Conditions**

Federal general forest lands outside wilderness were managed to reduce fuels and high-severity wildfire risks while moving forests toward historical conditions, i.e., conditions assumed to be typical prior to about 1850. Management in reserves was designed to reduce stand density and fuel levels while maintaining large and very large trees in generally open forest conditions.

We used the reference condition VDDT models developed by LANDFIRE Rapid Assessment (LANDFIRE 2006) as a basis for historical disturbance regimes, including wildfire return intervals and insect outbreaks on Federal general forest lands. We added state classes to the reference condition models to reflect the variety of structural conditions required by our issues, but retained the overall wildfire return intervals by fire-severity class (Table 3). We used a variety of treatments to mimic the reference disturbance regimes and favor single-storied forests of fire-tolerant conifers, especially in the drier potential vegetation types. We applied prescribed fire to mimic historical wildfire

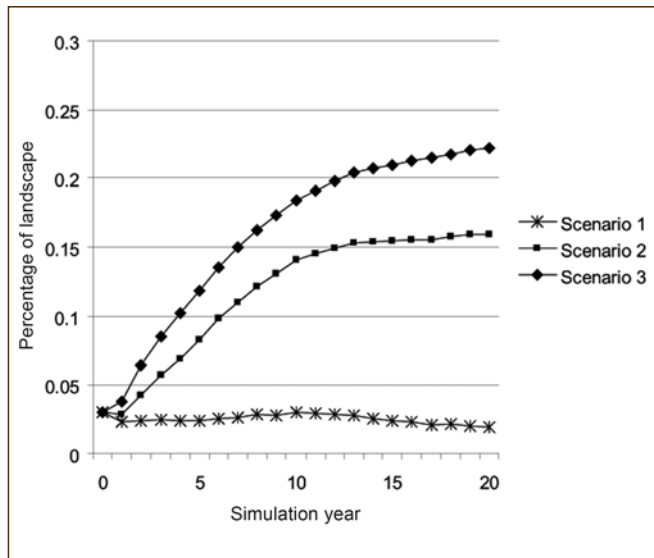


Figure 3—Percentage of the landscape in single-story very large and large tree structure classes by scenario in the Five Buttes area, central Oregon, U.S.A.

frequencies on Federal general forest lands. Reserves were treated with half the intensity of Federal general forests because the late-successional reserves are intended to provide more abundant large- and very-large-tree multistory forest habitat than Federal general forest lands. Wilderness areas were not treated except with wildfire suppression.

Multistory large- and very-large-tree forests declined over the 200-year simulation period under scenario one owing to a combination of wildfire and insect outbreaks (Figure 2). This suggests, based on our modeling assumptions, that passive management on Federal lands in the study area might produce no more than about the current abundance of large- and very-large-tree forests in the study area and that those forest conditions might decline on a long-term basis.

Single-story large- and very-large-tree forests remained relatively constant at about 2 percent of the landscape area under scenario 1, on average, over 200 years (Figure 3). The relatively low levels of single-story large- and very-large-tree forests that did occur resulted from an uncommon coincidence of slow regeneration of small trees and random low- or moderate-severity wildfire.

High-severity wildfires burned more landscape area under scenario 1 than scenarios 2 and 3 (Figure 4).

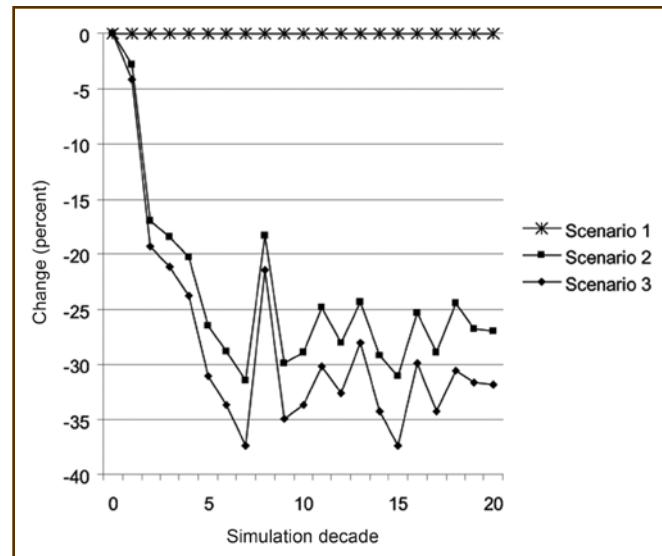


Figure 4—Percentage change in average decadal area affected by high-severity wildfire under scenarios 2 and 3 compared to scenario 1 in the Five Buttes study area, central Oregon, U.S.A.

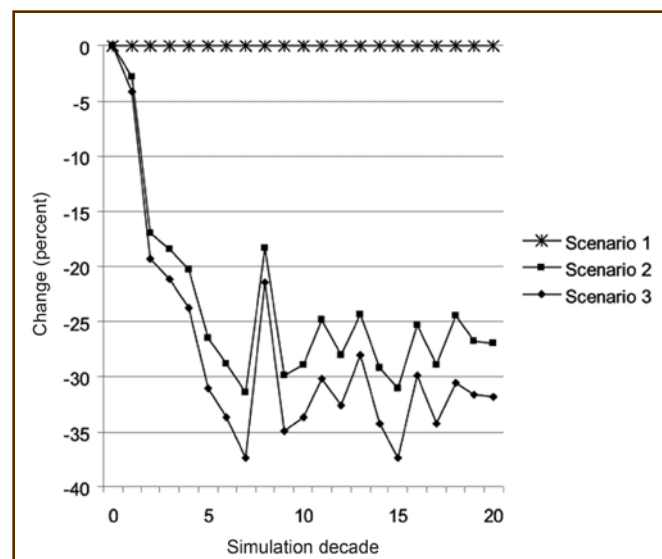


Figure 5—Average decadal area treated with mechanical fuel treatments and thinnings by scenario in the Five Buttes area, central Oregon, U.S.A.

High-severity wildfire was proportionately greatest in WUI areas dominated by grass, forb, and shrub communities. Although these communities are highly susceptible to wildfires that kill most of the aboveground vegetation, wildfires in such vegetation are much more easily controlled than those burning in dense forests. The other ownership/ allocation categories were largely forested throughout our

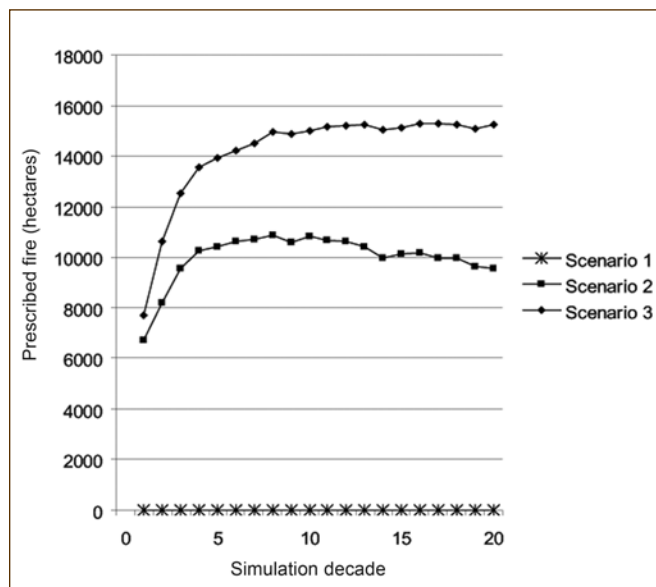


Figure 6—Average decadal area treated with prescribed fire by scenario in the Five Buttes area, central Oregon, U.S.A.

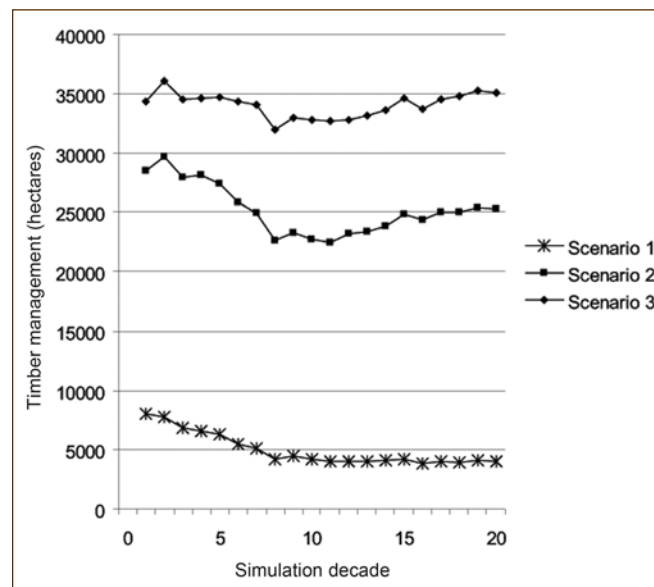


Figure 7—Average decadal area treated with management activities that may produce commercial timber products by scenario in the Five Buttes study area, central Oregon, U.S.A.

simulations. High-severity wildfire affected Federal general forests somewhat more than other ownership/allocation classes because Federal general forests were mostly in lower elevation, drier environments subject to higher fire probabilities.

Mechanical fuel treatments and stand thinnings only occurred on private lands under scenario 1 and remained below 1000 ha treated per decade (Figure 5). Likewise, prescribed fire treatments only occurred on Federal lands and were absent under scenario 1 (Figure 6). Treatments that might produce at least some commercial timber products averaged less than 10 000 ha per decade under scenario 1 and slowly declined to about 5000 ha per decade in the last ten decades (Figure 7).

Scenario 2 produced moderate amounts of multistory large- and very-large-tree forests (Figure 2). Contrary to our design objectives, scenario 2 did not increase multistory large tree forest by much compared to current conditions. In fact, scenario 2 produced lower amounts of multistory large- and very-large-tree forest than scenario 1 for the first 100 years. Both scenarios 1 and 2 simulations produced, on average, about half the current amount of multistory large-tree and very-large-tree forest at the end of 200 years. Perhaps alternative approaches to protecting and

developing multistory large-tree forests on Federal general forests could be formulated and might be more successful than our scenario 2. This also suggests, at least given assumptions in our models, that current levels of multistory large- and very-large-tree forests in the study area are perhaps an artifact of fire suppression and other factors and may not be sustainable in the study area over the long run.

Single-story large- and very-large-tree forests substantially increased across the entire landscape under scenario 2 (Figure 3). Much of the increase occurred in Federal general forest, and WUI was due to thinning to produce large trees quickly in scenario 2 and very active fuel treatments that produced open stands in scenario 3. Single-story large- and very-large-tree forests remained at very low levels in reserves and in the wilderness.

Scenario 2 produced lower levels of high-severity wildfire compared to scenario 1 (Figure 4). Even though fuel treatments were not extensive in scenario 2, some did occur on Federal general forests and in reserves to foster early development of large trees. As a result, the area burned in high-severity wildfires was, on average, about 20 percent to 30 percent less than in scenario 1, especially after the first two decades. As in scenario 1, the highest proportion of high-severity wildfire in scenario 2 was in open

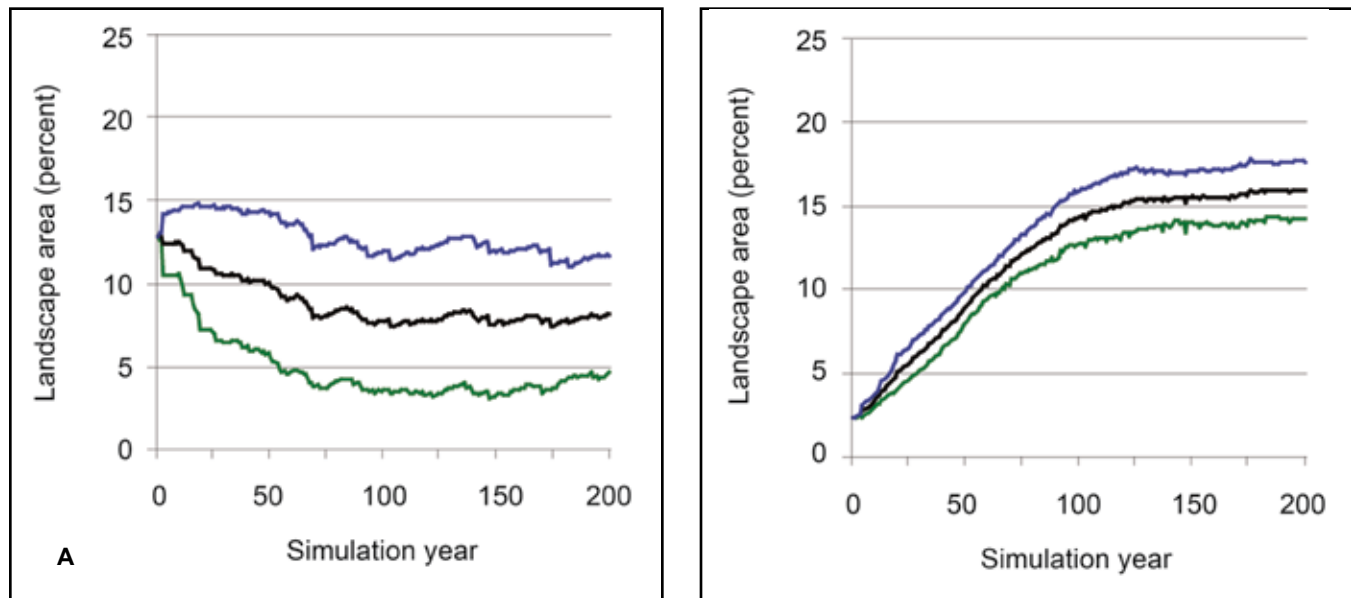


Figure 8—Variation in amounts of multi- and single-story large- and very-large-tree forests under scenario 2 for 30 Monte Carlo simulations in the Five Buttes area, central Oregon, U.S.A. Upper and lower lines are plus and minus one standard deviation from the mean.

forests dominated by grass/forb/shrub communities in WUI. High-severity wildfires are relatively easy to control in open grass/forb/shrub communities compared to high-severity wildfire in dense forest. From a wildfire protection perspective, active fuel treatment changed potential fire behavior rather than eliminating wildfire. Federal general forests experienced considerably lower amounts of high-severity wildfire compared to scenario 1, owing to fuel treatment effects, but amounts in reserves and wilderness were similar to those under scenario 1.

Management activity levels were higher in scenario 2 than in scenario 1 due to thinnings to promote large tree development in Federal general forests. Scenario 2 produced about 25 000 ha of commercial treatment activities per decade over 20 decades (Figure 7). Mechanical fuel treatment rates declined slightly after the first decade, then varied over the remaining 19 decades. Mechanical fuels and thinning were highest in the first decade as the initial round of mechanical fuel treatments peaked (Figure 5). Prescribed fire rose to about 10 000 ha per decade as fire replaced mechanical fuel treatment for fuel reduction, then remained at relatively stable levels (Figure 6).

Scenario 3 produced the lowest overall abundance of multistory large- and very-large-tree forests (Figure 2).

Multistory large- and very-large-tree forests declined from about 13 percent of the study area to a minimum of about 4 percent at the end of the simulation. Much of the decline occurred in the first 100 years as dense forests burned or were killed by insect outbreaks. Initial declines on Federal general forests were due to thinnings designed to quickly move dense forests to more open conditions followed by thinnings and fuel treatments at maintenance levels.

Conversely, scenario 3 produced abundant single-story large- and very-large-tree forests (Figure 3). Single-story large- and very-large-tree forests initially occupied less than 5 percent of the study area, but increased fourfold to over 20 percent by the end of the simulation. Increases were nearly greatest on Federal general forest lands due to active thinning and fuel treatment. Smaller increases occurred in WUI and reserves that were treated at lower rates. Though the trend was flattening after 200 years, single-story large- and very-large-tree forests were still increasing across the landscape as a whole. Landscape levels (about 20 percent) at 200 years were lower than those estimated by Hann and others (1997) for historical conditions in the southern Cascades area in Oregon and Washington (about 57 percent).

Scenario 3 also produced the lowest overall rates of high-severity wildfire (Figure 4). After the first five

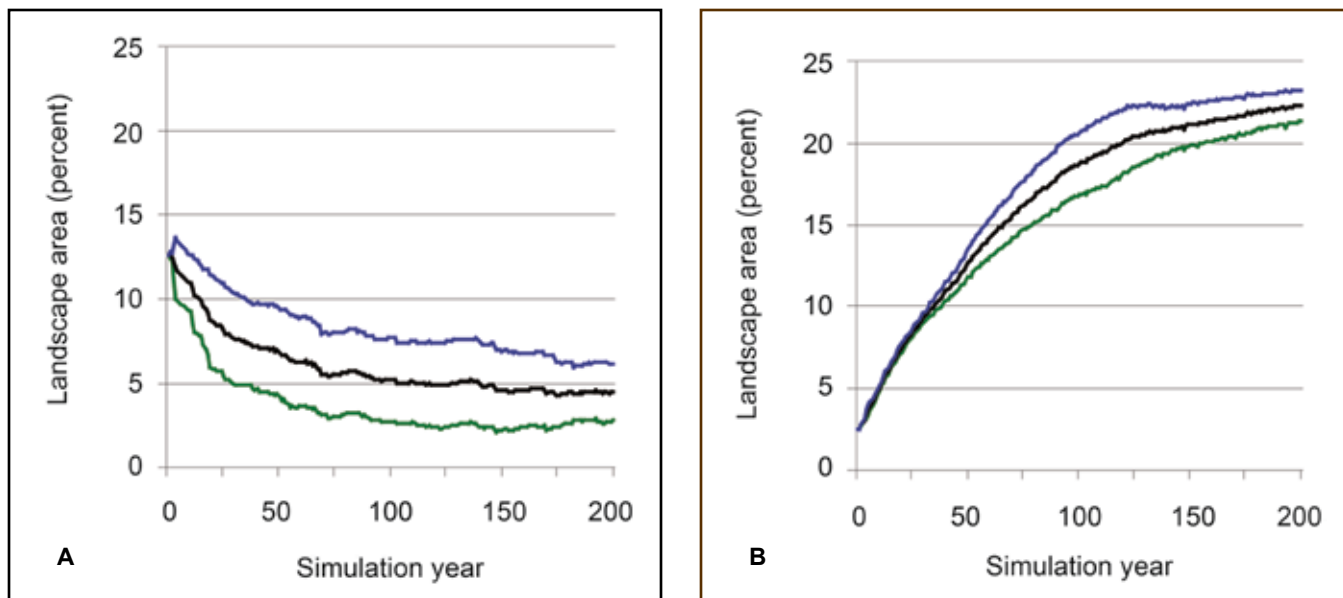


Figure 9—Variation in amounts of multi- and single-story large- and very-large-tree forests under scenario 3 for 30 Monte Carlo simulations in the Five Buttes area, central Oregon, U.S.A. Upper and lower lines are plus and minus one standard deviation from the mean.

decades, the proportion of area burned in high-severity wildfires was generally 30 percent or more below that in scenario 1 and 5 percent to 10 percent lower than that in scenario 2. WUI areas experienced the highest proportion of high-severity wildfire, again in grass/forb/shrub-dominated open forests where wildfire is most easily controlled. Of the other three ownership/allocation classes, wilderness areas were most highly impacted by high-severity wildfires, in contrast to scenarios 1 and 2. This resulted from fuel treatments that reduced wildfire outside wilderness. Federal general forests, on the other hand, experienced lower levels of high-severity wildfire compared to both scenarios 1 and 2 owing to fuel treatment effects.

Scenario 3 produced about 35 000 ha of commercial timber harvest per decade (Figure 7). Mechanical fuel treatments and thinning occurred on about 9000 ha in the first decade, then varied between 6000 and 7000 ha per decade after that (Figure 5). Not surprisingly, given the emphasis on reducing fire risks and generating open forests on Federal general forests, scenario 3 produced the high levels of prescribed fire (Figure 6). The initial ramp-up in prescribed fire took place over the first four decades after initial mechanical fuel treatments reduced fuel levels so that prescribed fire could be used for subsequent fuel treatments.

Variability

Results for several important landscape characteristics were highly variable over 30 Monte Carlo simulations in our study area. For example, whereas the multistory large- and very-large-tree forests under scenario 2 averaged about 8 percent of the landscape area at year 100, one standard deviation above and below the mean ranged from about 12 percent to less than 4 percent of the landscape area (Figure 8a). The same scenario produced lower variability for single-story large- and very large-tree-forests (Figure 8b). In this case, the mean at 200 years was about 16 percent and the standard deviation, plus or minus 2 percent. In our study area, and given the assumptions in our model, multistory large- and very-large-tree forests seem to be potentially less abundant and subject to more variability than single-story large- and very-large-tree forests, even in a scenario designed to increase multistory large- and very-large-tree forests.

Variability patterns for large- and very-large-tree forests under scenario 3 were similar to those in scenario 2 (Figures 9a and 9b). As one might expect given scenario 3 objectives, multistory large- and very-large-tree forests were much less abundant than single-story large- and very-large-tree forests overall. Single-story large- and

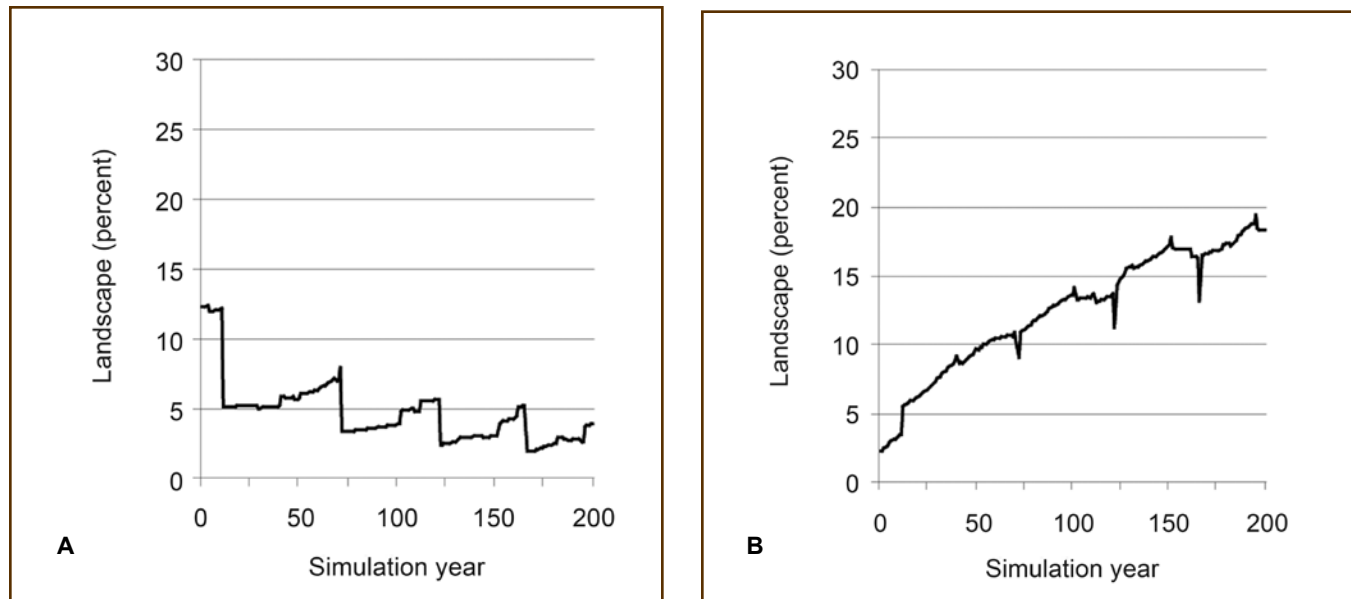


Figure 10—One randomly selected example simulation run showing amounts of multi- and single-story large- and very-large-tree forests under scenario 2 in the Five Buttes area, central Oregon, U.S.A.

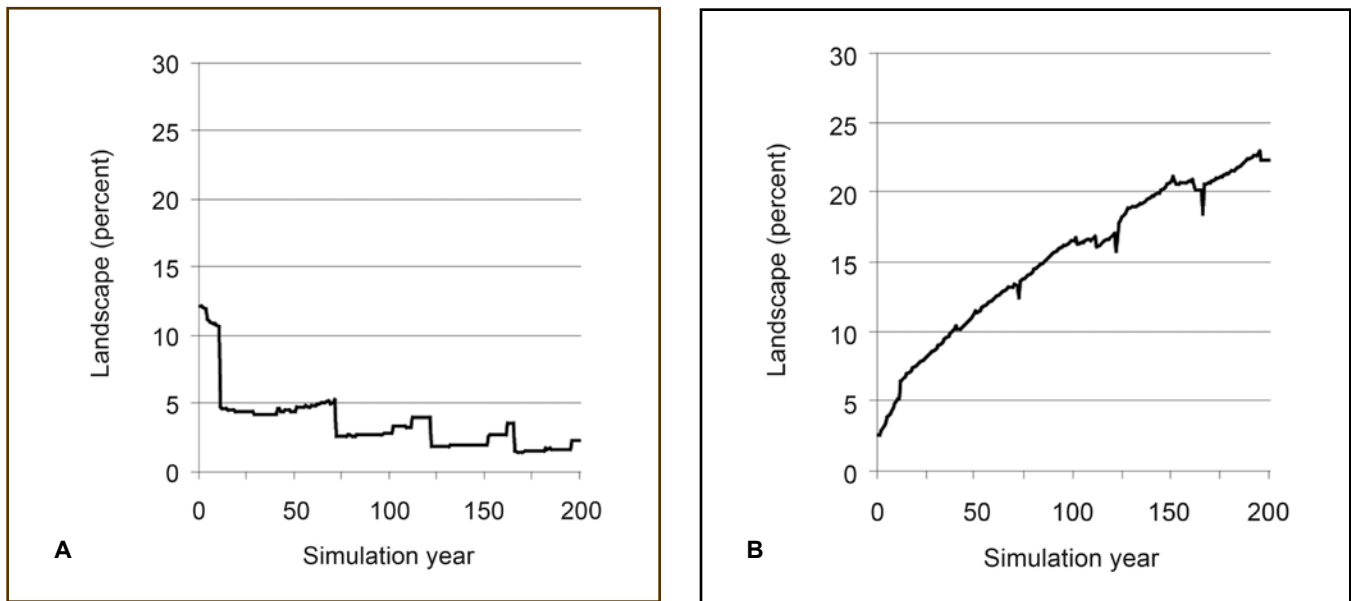


Figure 11—Variation in amounts of multistory large- and very-large-tree forests under Scenario 3 in the Five Buttes area, central Oregon, U.S.A.

very-large-tree forests increased steadily to an average of about 22 percent of the landscape area with relatively narrow variation. Large- and very-large-tree multistory forests steadily declined from current conditions to less than 6 percent of the landscape area by the end of the simulations. Our interpretation, based on our modeling assumptions, was that single-story large- and very-large-tree forests were

relatively stable and might be sustained at high abundance for many decades in much of the study area given fuel and thinning treatments, as others have suggested for similar environments (e.g., Agee 2003, Hann and others 1997, Hessburg and Agee 2003). In addition, managing to increase multistory large- and very-large-tree forests in this landscape might not succeed, and future variation

might produce very small amounts even with management designed to increase them.

In fact, none of the individual simulation runs that make up the 30 Monte Carlo set for multistory large- and very-large-tree forests under scenarios 2 or 3 looked anything like the mean trend (Figures 10a, 10b, 11a, 11b). In these examples, multistory large- and very-large-tree forests experience occasional crashes during a sequence of years with abundant high-severity wildfire or insect outbreaks. Simulated patterns, however, suggest that multistory large-tree forests may be subject to boom-and-bust abundance in the study area. Single-story large-tree forests also experienced occasional sharp drops, but to a lesser degree, and recovery was quicker. Judging from patterns in individual simulations and variation in many simulations, single-story large- and very-large-tree forest structures were the most stable older forest structure in general Federal forests in the study area.

Conclusions

Our model results may indicate some interesting landscape hypotheses in this and similar areas:

1. Fuel treatments in WUI may shift wildfire behavior as fires burn in grass, shrubs, and open forests, but not reduce overall wildfire probability. However, shift in behavior could be important because wildfires in grass, shrub, and open forest fuels are easier to control than those in closed, dense forests.
2. Efforts to increase multistory, dense forest habitats in these drier environments for particular wildlife habitats may prove difficult because increased wildfire and insect outbreak probabilities might offset gains from silvicultural manipulation. In our simulations, multistory large-tree forests didn't exceed about 15 percent of the landscape on average, and amounts declined from current conditions. Most individual model runs in our Monte Carlo set experienced boom and bust conditions such that this forest type occasionally crashed to less than 5 percent of the landscape area.
3. Scenario 3, which moved Federal general forests toward historical conditions, generated the most stable

landscape conditions, but individual simulations still produced occasional sharp declines in large-tree forests because of severe wildfire years or insect outbreaks.

The models we used and the assumptions they embody reflect how we think the landscape disturbance and management processes might work to control landscape characteristics in the study area. Our models were based on expert opinion, the existing literature, and calibration by finer scale, stand-level silvicultural models. Calibration of annual wildfire year and insect outbreak sequences with historical drought and other climatic influences with empirical data from other sources (e.g., dendroclimatology) is an area where future model improvements could be made. Stand treatment prescriptions we used need to be tested in the field to determine whether desired outcomes are achieved. In addition, processes in the future may produce results much different than our estimates because: (1) odd or unusual events could occur, (2) we may not understand the system sufficiently well, (3) there may be some undetected logical error in our models, (4) climate change may alter fire, insects/disease, and other disturbances, and (5) management direction may change.

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Moderate-Resolution Data and Gradient Nearest Neighbor Imputation for Regional-National Risk Assessment

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Abstract

This study was designed to test the feasibility of combining a method designed to populate pixels with inventory plot data at the 30-m scale with a new national predictor data set. The new national predictor data set was developed by the USDA Forest Service Remote Sensing Applications Center (hereafter RSAC) at the 250-m scale. Gradient Nearest Neighbor (GNN) imputation was designed by the USDA Forest Service Pacific Northwest Research Station (hereafter PNW) to assign a plot identifier, and, therefore, a link to associated plot data, to each pixel within a target raster. Gradient Nearest Neighbor was implemented at 30-m resolution in three separate multimillion-hectare regions of the Western United States. Concurrently, RSAC developed a set of spatial predictor surfaces at 250-m resolution for use in producing nationally consistent data products. These data have been used for modeling forest types and forest biomass for the conterminous United States and Alaska. These predictor data have also been used for large regional applications.

In this study, we substituted the 250-m predictor data for the 30-m predictor data used thus far in GNN. Our objective was to quantify the difference in performance using the lower spatial resolution predictors. We remodeled the same three regions that were mapped at 30 m with the 250-m data set and compared the error structure of the two modeling efforts. For species presence/absence models in the two areas with large environmental gradients, the Sierra Nevada and northeastern Washington, the species models

performed substantially the same at the two resolutions. For the region with reduced environmental heterogeneity and moderate environmental gradients, coastal Oregon, species models did not work well with either the 30-m or 250-m studies. Models geared towards mapping forest structure did not perform as well as the 30-m models and may be insufficient for risk-assessment use.

Keywords: Gradient Nearest Neighbor, imputation, regional analysis, species distributions, vegetation mapping.

Introduction

A great wealth of resources has been expended to inventory our Nation's forests, and an equally substantial amount of effort has gone into acquiring remotely sensed data. As such, these two data types make up the ends of a continuum of detail. Plot inventory data are extremely sparse geographically but have a high level of information content regarding the resources at the inventory plot locations. Conversely, remotely sensed data cover the entire globe but with comparatively limited information at any single location. The common approach to leverage these two forms of data has been to create thematic maps of vegetation-related classes as well as response surfaces for other target variables of interest. With regard to vegetation mapping, these thematic maps typically describe dominant vegetation and include physiognomic, floristic, or structural characteristics, or all. These thematic maps often include some additional land use classes or map land use as a separate theme. The variables available for analysis are limited to the classes included in the map, and once the analysis is complete there is no ability to develop new attributes without a new mapping effort. Recently, a more flexible approach, single neighbor imputation, has been utilized to provide sample tree lists and plot-calculated variables for all the unsampled pixels in raster maps. Although not replacing traditional mapping methods, imputed maps can greatly enhance analytical flexibility and provide information in a familiar context that is often supported by extensive simulation modeling capability. Imputed maps are not intended to suggest that each pixel is in fact occupied by the imputed plot data, but rather given

current information, what do we expect. However, developing and mapping 30-m products over broad spatial extents is a lengthy process. This project was conducted in order to evaluate the differences in using a new national spatial predictor database at a coarser resolution (250 m) instead of the 30-m data used in the previous study.

Existing Methods for Regional-National Vegetation Mapping and Risk Assessment

Traditional Methods—

In recent years, many regions within the USDA Forest Service have implemented midlevel classification and mapping programs to provide thematic maps of existing vegetation for a wide variety of analysis applications. These programs are becoming more similar as they implement the USDA Forest Service direction established by the *Existing Vegetation Classification and Mapping Technical Guide* (hereafter technical guide) (Brohman and Bryant 2005). The vegetation classifications and mapping methodologies of these programs follow the technical guide's midlevel direction (Brewer and others 2005), and most use satellite remote sensing approaches to provide synoptic coverage of the mapping areas (e.g., Mellin and others 2004). Some of these programs also utilize summary databases populated by the USDA Forest Service Forest Inventory and Analysis (hereafter FIA) data to develop quantitative map unit descriptions including estimates of common inventory variables. This approach provides thematic map products with statistically sound estimates of inventory variables. These estimates are explicitly connected to the vegetation pattern depicted in the thematic map products. The approach is designed to support midlevel and broad-level analysis applications, as well as some project-level cumulative effects analyses.

As suggested in the technical guide, these midlevel map products can be used as is or rescaled for a variety of base-level analysis applications including project support, risk analysis, and watershed analysis at the scale of 4th and 5th Hydrologic Units, (USDA FS 1995). Unfortunately, the geographic extent of these base-level analysis applications is normally too small to effectively use FIA data as the inventory data source. Given the extensive design of the FIA

program without spatial intensification, each plot represents approximately 6,000 acres. This leaves forests and ranger districts faced with the difficult choice of using the midlevel map product as is with an inadequate sample size of associated FIA data or reverting to the use of often biased and outdated stand-exam data that cannot provide defensible statistical estimates and have no explicit relationship to the midlevel map data used for forest plan revision and project cumulative effects analyses. Alternatives to this untenable choice include several expensive and logistically difficult inventory approaches including intensifying the base grid to provide an adequate sample size or implementing a new traditional two-stage sample of the map features depicting vegetation pattern.

Imputation Methods—

A primary information need of land managers is consistent and continuous current vegetation data on each and every parcel of land in an analysis area sufficient to address the principal issues and resource concerns. As discussed above, where these data do exist they are normally based on a sampling inference procedure rather than wall-to-wall inventory data. Many of the analyses needed to address multiple resource issues at the project level are essentially analyses of vegetation pattern and process relationships through time and space. Inventory data based on traditional two-stage sampling or quantitative map unit descriptions from a systematic random grid are not sufficient to address the spatial or the temporal dimensions or both of these analyses. These data are not spatially explicit enough to identify important vegetation pattern relationships and do not provide adequate thematic detail (i.e., plot-level tree list data) for simulating vegetation change through time.

The ability to simulate these vegetation pattern relationships through space and time, particularly with a variety of management and disturbance alternatives, is important for effective land and resource planning. Despite the capability of simulation models and decision-support tools, comprehensive landscape-level planning is still difficult to implement because the inventory data are rarely complete or current or both. For planning purposes, it would be convenient to be able to operate as if detailed inventory

information were available for all units in the planning area (Moeur and Stage 1995).

As an alternative to historically common statistical approaches (e.g., regression estimates or stratum averages) to populating unsampled units with data, imputation can be used. Imputation involves estimating values for variables of interest (Y variables) by supplying realistic measurements from one or more sampled units to unsampled units with similar characteristics in auxiliary (X) variable space (Hassani and others 2004, LeMay and Temesgen 2005, McRoberts and others 2002, Moeur and Stage 1995, Ohmann and Gregory 2002, Temesgen and Gadow 2004). These auxiliary (X) variables typically include biophysical characteristics such as slope, aspect, precipitation, etc., as well as data from remotely sensed imagery such as aerial photography or satellite imagery.

Imputation of inventory data from sampled areas to similar unsampled areas produces data sets that function like wall-to-wall data for planning purposes. There are many methods and variations of imputation, both univariate and multivariate; however, multivariate approaches that impute a single plot tend to produce more realistic data sets for simulation modeling because they retain the original covariance structure of actual sample units. LeMay and Temesgen (2005) provided a brief summary of common imputation approaches and a detailed comparison of variable-space nearest neighbor (NN) methods for estimating basal area and stems per hectare using aerial auxiliary variables. LeMay and Temesgen (2005) also summarized variable-space nearest neighbor methods and compared them to other estimation methods. These summaries were the most comprehensive available in current literature when this paper was written.

In recent years, two modeling approaches have been developed that could potentially address this critical need through the imputation of inventory data. The first of these approaches, Most Similar Neighbor (MSN), was developed by Moeur and Stage (1995) to impute attributes measured on some sample units (e.g., stand polygons) to sample units where they were not measured. The MSN was originally designed to use a traditional two-stage inventory of forest stands, as described by Stage and Alley (1972), imputing

stand data to unsampled stands. The second of these approaches, Gradient Nearest Neighbor (GNN) developed by Ohmann and Gregory (2002), follows the same general analytical logic, but is designed to use vegetation information from regional grids of field plots (similar to an intensified FIA grid) with remotely sensed imagery and other spatial data to produce a continuous raster surface by imputing data from sampled grid cells to unsampled grid cells.

Study Objectives

Our primary objective was to quantify the difference in GNN model performance using the lower spatial resolution predictors. We remodeled the same three regions that were mapped at 30 m with the 250-m data set and compared the error structure of the two modeling efforts. As explained below, two effects occur when 30-m data are replaced with 250-m data, and both involve averaging across multiple pixels.

One of the reasons for implementing this study is that development of spatial products and modeling at 30 m could take several years to complete a large portion of the Western United States. We anticipate much faster turnaround if 250-m modeling proves sufficient. The 250-m data products could potentially be available for large areas such as the Western United States in 1 to 2 years of production.

Methods

The Study Area

Three western regions covering temperate steppe, coastal forest, and Mediterranean ecosystems were mapped using GNN imputation for a Joint Fire Sciences Program study (Figure 1). The original study examined the feasibility of mapping wildland fuels and vegetation structure to provide data for fire and fuels management planning (Pierce and others 2009, Wimberly and others 2003). The 2.86-million-ha coastal forest site was located in the coast range of Oregon extending as far inland as the western edge of the Willamette Valley (Figure 1). The forests are primarily coniferous with hardwoods occupying riparian and disturbed areas. The 4.1-million-ha Mediterranean site was located in the central Sierra Nevada occupied by savannah,



Figure 1—The Joint Fire Sciences project covered three western sites in contrasting ecosystems.

chaparral, mixed conifer, and alpine woodlands vegetation types. The site stretches from the northern border of Sequoia National Park north through the Plumas National Forest. The 5-million-ha temperate steppe in northeastern Washington was bounded on the west by the Cascade crest and on the south by the Columbia and Spokane Rivers (Figure 1). The temperate steppe site is dominated by a combination of mixed coniferous forest and extensive shrub steppe.

Vegetation Data from Field Plots

Vegetation data from regional inventories were derived in each of the three regions from multiple sources including Forest Inventory and Analysis (FIA) plots, Current Vegetation Survey/Pacific Northwest Region (R6) plots (CVS), Pacific Southwest Region (R5), Bureau Land Management (BLM), research Ecology Plots in North Cascades National Park (NCNP) (provided by Dave L. Peterson) and Yosemite National Park (provided by Jan Van Wagtenonk). The FIA,

R5, and CVS plots were installed on systematic grids. The CVS/R6 and R5 plots covered the national forests whereas FIA installed plots on all ownerships. The FIA and CVS inventory plots used five subplot arrays within a 1-ha area. Small trees, snags, coarse woody debris line-intercept transects, and ground cover were sampled on each subplot.

Because vegetation data were derived from multiple inventories with different sampling protocols, all individual tree records were converted to per-hectare values. For plots with multiple vegetation or land cover conditions, only the forested portion was used with expansion factors adjusted accordingly. Plot-level summary variables were calculated for each plot.

Stand-summary variables included total basal area, basal area by species, trees per hectare, quadratic-mean diameter, snags per hectare, percentage of tree canopy cover, and down-wood volume. Different inventories collected down-wood data using different sampling schemes

Table 1—Comparison of spatial data between the 30-m and 250-m analyses

250-m resolution	30-m resolution (Pierce and others 2009)
Elevation, slope, and aspect	Elevation, slope, and aspect
DAYMET shortwave radiation	Potential Relative Radiation
Ecological region layers	NA
USGS mapping zones, Bailey's ecoregions, and unified ecoregions for Alaska	NA
STATSGO data layers (lower 48)	NA
Available water capacity, soil bulk density, soil permeability, soil PH, soil porosity, soil plasticity, depth to bedrock, rock volume, soil types, and soil texture	None
MODIS Vegetation indices such as EVI, NDVI	Landsat TM band ratios, 3/4, 4/5, 5/7
MODIS Vegetation continuous fields	NA
MODIS fire points developed from the MODIS active fire maps	NA
MODIS 8-day composites, multiple dates from the spring, summer, and fall	TM-5 and ETM-7 data
NA	Within footprint spectral variability
All MODIS 32-day composite imagery that is cloud-free between the years 2001–2003	NA
USGS NLCD layers, percentage tree cover, percent herbaceous cover, and percentage bare ground	NA
DAYMET temperature and precipitation, mins, maxs, aves, SD (all, yearly, monthly)	DAYMET temperature and precipitation, mins, maxs, aves (all)

and minimum sizes. As a result, we focused primarily on species basal area.

Moderate-Resolution Predictor Data

Beginning in 2003, RSAC, in cooperation with the FIA remote sensing band, developed a national predictors database to support FIA national mapping efforts (e.g., national forest type maps, and national biomass maps) (Blackard and others 2008, Ruefenacht and others 2008). The original database included about 60 layers consisting primarily of MODIS imagery. The national predictors database has been extensively used with additional data layers added each year. The current version has more than 700 layers, which includes DAYMET climate data, additional MODIS imagery, derived MODIS-based vegetation indices, STATSGO soil layers, topography, and several derived thematic products (Table 1). These data cover the entire conterminous United States as well as Alaska and Puerto Rico.

To prepare these data for Gradient Nearest Neighbor analysis, all layers were sampled with all plot locations. With 30-m data, 13-pixel footprints were used to sample the spatial data to account for plots occupying more area than a

single 30-m pixel. With 250-m data, only a single pixel was necessary to represent each plot because the plot area is less than the area of one 250-m pixel. All predictor images were split into separate bands, comprising over 700 individual predictor layers. For example, some STATSGO soil layers comprised 11 soil horizons. Those 11 horizons were split into separate predictor layers. Before statistical modeling, the 700 bands were reduced to those without a correlation of > 0.90 with any other remaining predictor. For each of the three study areas, this left about 100 to 150 predictor layers. These predictors were entered as the predictor matrix in a stepwise canonical correspondence analysis. In each case, 10 to 15 predictors were sufficient to explain the primary gradients in vegetation composition.

Satellite Imagery

For the 30-m study, Landsat Thematic Mapper imagery (TM) was mosaiced and histogram matched. For the 250-m study, several products derived from MODIS imagery were used (Table 1). These products have a spatial extent covering our entire study regions, so no image matching was necessary.

Biophysical Environment Data

For both studies, biophysical data were derived from digital elevation models, including slope, aspect, and elevation. Both studies used climate data derived from DAYMET, although the 250-m data set used many more calculated variables. These variables included year-to-year variability by month. The DAYMET data are provided at 1-km resolution, so both the 30-m and 250-m DAYMET data were interpolated or resampled to their respective resolutions.

The 250-m data set also included STATSGO data layers, such as available water-holding capacity, soil bulk density, soil permeability, and soil pH. No analog for these data existed in the 30-m data set.

The Gradient Nearest Neighbor (GNN) Method of Predictive Vegetation Mapping

Imputation is a process in which values are assigned to unmeasured locations from either measured values or a statistical summary of a few selected measured values such as a mean (Moeur and Stage 1995). Unlike regression-based predictions, the assigned value is not a product of predictor variables and coefficients. The predictor variables are used to rank sample plots as to their similarity to a target location (30-m or 250-m-square pixel). In Gradient Nearest Neighbor imputation (GNN) we used the loadings for the ordination axes and their eigenvalues from a Canonical Correspondence Analysis (CCA) to relate the target locations to the locations of sample plots. This is achieved by calculating a Euclidean distance in eight-axis ordination space between the target pixel and each plot using the ordination loadings to weigh the spatial variables and the eigenvalues to weigh each axis. The distance in gradient space between the target location and each plot is used to rank the sample plots for potential assignment at each target pixel in our study regions (Ohmann and Gregory 2002). Because we use a multivariate response, which is analogous to a community representation of a plot, the variables selected must be of similar type. Our primary modeling scenarios have been species modeling, which involves using the total basal area of each individual species, and structure modeling, which has used basal area in different size classes

of hardwoods and conifers, snag density, coarse woody debris volume, and canopy cover.

When GNN is run as a single neighbor imputation, the closest plot in gradient space is assigned to each pixel in the landscape. Once the assignment has been made, any attribute calculated for all plots can be mapped, maintaining the original covariance structure for any/all other attributes to be mapped. Additionally, the ranking of potential neighbors provides a sample neighborhood of similar plots from which natural variability and sample sufficiency can be evaluated (Pierce and others 2009).

Gradient Nearest Neighbor Results Summary from Previous Studies

In the 30-m study, coastal Oregon had favorable results for structure models but substandard results for species models. This pattern was reversed in both Washington and California, where species models worked well, but structure models did a poor job with attributes such as quadratic mean diameter and trees per hectare. The purpose of the original study was to map wildland fuels and vegetation structure. Wildland fuel components, such as coarse woody debris and snag density, were not mapped adequately in any of the three sites (However, canopy-related fuels variables were more satisfactory.) This was partially due to remote sensing not directly detecting wildland fuels, which, in general, are below the canopy. Another factor is that coarse woody debris data are collected on only a relatively short transect on FIA plots such that the resulting sample size is too small to characterize individual plots. The original design of the coarse woody debris sampling was to create estimates for a region.

We expect that species models will perform more favorably than structure models with 250-m data. In the previous study, remote sensing imagery was very important for mapping structure and has a much finer spatial grain than our climate data. Although we had climate data at 30 m, it was interpolated from 1-km-resolution DAYMET data, which are interpolated from weather stations plus higher resolution covariates (elevation, topography, etc.). Therefore, climate data would change gradually over the course of a kilometer whereas TM data can change abruptly from one

30-m pixel to the next. Therefore, our change in resolution loses much more information for predictions relying heavily on imagery than those relying on climate.

Model Evaluation and Accuracy Assessment

To evaluate the results of the 30-m study compared with the 250-m study results, we used two primary approaches. For continuous variables, we calculated 2nd nearest neighbor correlations (analogous to r-squares from cross-validation), (see Ohmann and Gregory 2002). For discrete variables, such as species presence/absence, we used standard confusion matrices. Producers accuracy, users accuracy, and Kappa statistics were calculated for each species and compared for the two studies (Wilkie and Finn 1996). The Kappa statistic accounts for the probability of randomly assigning a plot to its correct class. As such, the random probability of assigning a species with a frequency of 0.9 to any plot is very high; Kappa statistics tend to be low as the probability of improving upon randomness decreases. Producers accuracy is the proportion of sample plots with a species present in which the species is predicted to occur. Users accuracy is the proportion of plots with the predicted species occurrence that actually had the species in the inventory.

Any discrepancies derived from changing the scale of the analysis stems from two primary effects, which are both aspects of spatial averaging. First, with the 30-m product, the predicted value for a plot is the average of the 13 pixels imputed to the plot footprint. In this way, the predicted value is subject to averaging over those 13 imputations. With the 250-m imputation, only a single plot is imputed for the same ~1 ha space. Therefore, the 250-m results will be slightly more variable. The second effect involves the spatial averaging of the predictor data set. The predictor variables are the averages of the 13 pixels within the individual predictor layers. In addition, we calculate two texture indices for the remote sensing data, which provides an estimate of variability within the footprint. With the 250-m data, a single pixel covers an area considerably larger than a 1-ha plot (62 500 m² vs. 11 700 m²). Thus, the spectral values are averaged over a larger area, and we have no estimate of within-plot variability.

Results

Gradients in Species Composition

Daubenmire (1952) noted the axiomatic relationship between climate and vegetation. Our CCA modeling results are consistent with this observation and suggest that climatic variables, as well as topographic and edaphic variables indirectly related to temperature and moisture, strongly influence the patterns of species composition. The gradients described by the three CCA models were composed of the dominant patterns in temperature and precipitation. In Washington, elevation, precipitation frequency, and brightness in MODIS 8-day composites separated warmer drought-tolerant conifers from both high-elevation wet and dry forests. In California, species were separated by September growing degree days, September average air temperature, April cooling degree days, and water vapor pressure variability in July. In Oregon, the dominant environmental variables were August mean precipitation, June cooling degree days, soil permeability, and June standard deviation of water vapor pressure.

Species Mapping Performance of GNN

Species performance from confusion matrices are listed for all species occurring with a frequency of at least 5 percent in the plot data set (Tables 2 and 3). California had the highest average Kappa statistics at 0.53 for the 30-m study and 0.48 for the 250-m study (Table 2). Washington was second with 0.46 and 0.43 (Table 3), followed by Oregon with 0.32 and 0.25 (Table 4). Patterns for producers and users accuracy were similar across sites, as were the actual values. In each case, the 30-m study had higher producers accuracy than the 250-m study by about 22 percent, whereas for users accuracy, the 250-m study was actually about 3 percent higher with an average across sites of 55 percent.

Structure Mapping Performance of GNN

To date we have only developed structure models for the Washington and Oregon study sites. Second nearest neighbor correlations for structure variables were generally low in both sites. In Washington, total basal area had an r-square of 0.06 compared to 0.17 for the 30-m analysis, 0.04 for snags per hectare compared to 0.16, and 0.01 for quadratic

Table 2—California species presence/absence results, N = 1,835 plots for species of ≥ 5 percent frequency on plots

Species	Kappa		Producers		Users		Plots
	30 m	250 m	30 m	250 m	30 m	250 m	
<i>Abies concolor</i>	0.51	0.47	0.88	0.67	0.64	0.68	761
<i>Calocedrus decurrens</i>	0.51	0.52	0.86	0.69	0.61	0.67	643
<i>Pinus ponderosa</i>	0.56	0.52	0.88	0.70	0.63	0.66	632
<i>Quercus kelloggii</i>	0.56	0.48	0.86	0.61	0.59	0.61	510
<i>Pinus jeffreyi</i>	0.49	0.46	0.82	0.61	0.54	0.60	488
<i>Pinus lambertiana</i>	0.42	0.36	0.80	0.54	0.49	0.50	468
<i>Abies magnifica</i>	0.63	0.55	0.85	0.65	0.63	0.65	389
<i>Pseudotsuga menziesii</i>	0.59	0.52	0.86	0.61	0.58	0.62	370
<i>Pinus contorta</i>	0.60	0.42	0.84	0.59	0.58	0.48	340
<i>Quercus chrysolepis</i>	0.54	0.43	0.78	0.50	0.52	0.51	264
<i>Pinus monticola</i>	0.52	0.39	0.70	0.48	0.49	0.43	189
<i>Quercus wislizeni</i>	0.53	0.53	0.71	0.59	0.47	0.52	108
<i>Juniperus occidentalis</i>	0.40	0.42	0.64	0.50	0.34	0.42	102
<i>Quercus douglasii</i>	0.53	0.67	0.65	0.69	0.48	0.69	89
Average	0.53	0.48	0.79	0.60	0.54	0.57	

Table 3—Washington species presence/absence results, N = 763 plots for species of ≥ 5 percent frequency on plots

Species	Kappa		Producers		Users		Plots
	30 m	250 m	30 m	250 m	30 m	250 m	
<i>Pseudotsuga menziesii</i>	0.43	0.43	0.97	0.89	0.85	0.86	1,835
<i>Pinus ponderosa</i>	0.53	0.52	0.91	0.74	0.65	0.69	971
<i>Pinus contorta</i>	0.32	0.41	0.82	0.63	0.50	0.61	827
<i>Larix occidentalis</i>	0.54	0.51	0.89	0.70	0.61	0.67	809
<i>Abies lasiocarpa</i>	0.52	0.50	0.87	0.65	0.58	0.64	707
<i>Picea engelmannii</i>	0.41	0.35	0.81	0.54	0.49	0.52	643
<i>Abies grandis</i>	0.57	0.55	0.87	0.68	0.58	0.64	587
<i>Thuja plicata</i>	0.53	0.51	0.82	0.61	0.51	0.58	393
<i>Tsuga heterophylla</i>	0.49	0.44	0.78	0.53	0.45	0.48	277
<i>Pinus monticola</i>	0.35	0.27	0.68	0.37	0.33	0.33	240
<i>Abies amabilis</i>	0.67	0.62	0.86	0.66	0.60	0.66	219
<i>Tsuga mertensiana</i>	0.58	0.49	0.81	0.52	0.50	0.52	179
<i>Pinus albicaulis</i>	0.55	0.47	0.77	0.50	0.47	0.51	152
<i>Populus tremuloides</i>	0.15	0.16	0.40	0.23	0.16	0.20	146
<i>Betula papyrifera</i>	0.29	0.26	0.51	0.30	0.25	0.29	119
Average	0.46	0.43	0.78	0.57	0.50	0.55	

mean diameter compared to an almost equally random 0.05 for the 30-m analysis. In Oregon, where we had quite good results for structure with 30-m data, we mapped basal area with an r-square of 0.09 compared to 0.59 for the 30-m analysis, 0.03 for snags per hectare compared to 0.09, and 0.08 for quadratic mean diameter compared to 0.69.

Discussion

Accuracy of GNN Vegetation Maps

For the Washington and California study sites, species distributions were modeled equally well with the 250-m data and the 30-m data. Both the Kappa statistics and visual inspection of species maps indicated essentially the same

Table 4—Oregon species presence/absence results, N = 2,325 plots for species of ≥ 5 percent frequency on plots

Species	Kappa		Producers		Users		Plots
	30 m	250 m	30 m	250 m	30 m	250 m	
<i>Pseudotsuga menziesii</i>	0.22	0.19	0.97	0.95	0.91	0.91	684
<i>Alnus rubra</i>	0.32	0.18	0.89	0.70	0.67	0.66	440
<i>Tsuga heterophylla</i>	0.27	0.27	0.85	0.63	0.52	0.60	320
<i>Acer macrophyllum</i>	0.25	0.26	0.78	0.49	0.42	0.47	228
<i>Thuja plicata</i>	0.11	0.06	0.56	0.24	0.26	0.25	150
<i>Picea sitchensis</i>	0.52	0.42	0.83	0.50	0.50	0.53	128
<i>Abies grandis (concolor)</i>	0.39	0.29	0.63	0.34	0.34	0.33	52
<i>Quercus garryana</i>	0.47	0.33	0.62	0.32	0.43	0.38	40
Average	0.32	0.25	0.77	0.52	0.51	0.52	

pattern when moving from 30-m to 250-m data. Because the gradient models were largely composed of climate variables, there is actually little loss in predictor data information when using the 250-m data. This is because the climate data for both the 250-m and 30-m studies were interpolated from the same 1-km-resolution data. Species performance in Oregon was not as good, though it was also less accurate for the 30-m data. In both the 30-m and 250-m studies, we saw some definite differences between the results for Oregon and the results for Washington and California. Both California and Washington are precipitation limited, receive most of their precipitation during winter, and have large elevation gradients. The Coast Range in Oregon has much higher precipitation, milder temperatures, and lower overall topographic variation resulting in less orographic precipitation. Coastal Oregon has also had a long history of timber management and, therefore, has a large patchwork of even-aged stands.

Sources of Error in GNN—

The GNN and all nearest neighbor techniques, are particularly susceptible to errors introduced by natural variability at spectrally and environmentally similar sites. Whereas regression techniques model a trend and the departure from that trend, imputation retains the full range of variability within a data set. As such, for a certain location, a regression model with little predictive capability will predict the mean plus some small departure based on predictor variables and coefficients, whereas imputation will find the most environmentally similar site and select it. The

tendency for imputation to impute values similar to the actual target values is constrained by the strength of the relationship between available spatial predictor variables and the target response variables.

Other sources of error include (1) residual spatial error of predictor data sets and plot locations as well as plot registration, (2) temporal mismatches between inventory dates and imagery dates, and (3) the lack of adequate spatial data on disturbance and management history across large regions.

Advantages of GNN for Risk Assessment—

There are several advantages to GNN for risk assessment. The GNN technique retains the covariance structure for multiple attributes by imputing whole plots and provides mapped estimates of natural variability and sample sufficiency (Pierce and others 2009). Comparative risk assessment requires spatially explicit data with estimates of variability (Borchers 2005) in order to create probability surfaces for different management scenarios. For instance, what is the probability of the desired outcome given two different management choices, and are they statistically different? Without an estimation of uncertainty, this type of analysis can't be performed. By using a set of multiple potential neighbors, the variability in potential neighbors for a selected attribute can be mapped. In addition, by using the frequency distribution of all interplot distances in gradient space, thresholds for closeness in gradient space can be assigned and the number of candidate plots within a threshold calculated. This gives an indication as to whether

or not the inventory can provide adequate information for a certain pixel, and, as such, a map depicting the sampling support can be created.

Species Response Models in Multispecies Mapping—

One of the key areas of interest in natural resource risk assessment is the interactions among species. The location of invasive species and the presence of host species are two data surfaces of interest to managers. Mapping with single neighbor imputation ensures that the assemblages of tree species mapped are consistent with actual inventoried assemblages. This has both benefits and limitations. The benefit is robust assemblages of species as they currently exist. The limitation is that prediction for new interactions can not be inferred on the basis of these maps. Single-species models are probably best suited for predicting suitable habitat for an individual species, or rather the present distribution of habitat consistent with currently occupied habitat.

Risk Assessment Applications of GNN Predictions—

Single-neighbor imputation using GNN provides a very flexible wall-to-wall data set that includes any variable that can be derived from those measured on all inventory plots. This includes the ability to derive new variables or vegetation classifications after the initial modeling. A GNN imputation also provides a link to the full tree lists allowing for almost any kind of ecological modeling. The inclusion of multiple neighbors provides uncertainty data for Monte Carlo simulations or analyses seeking to show the uncertainty associated with different scenarios. As new risks or identification of new data needs arise, imputation maps are ready to adapt to new needs without the necessary production of a new model. However, at the 250-m scale, the variables that are correlated with broad climate patterns, specifically species distributions, will probably be characterized the best. Structure attributes, such as coarse woody debris and quadratic mean diameter, can be mapped, but the mapped variability will likely overwhelm the utility of such products.

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Integration of Population Genetic Structure and Plant Response to Climate Change: Sustaining Genetic Resources Through Evaluation of Projected Threats

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Abstract

To assess threats or predict responses to disturbances, or both, it is essential to recognize and characterize the population structures of forest species in relation to changing environments. Appropriate management of these genetic resources in the future will require (1) understanding the existing genetic diversity/variation and population structure of forest trees, (2) understanding climatic change and its potential impacts on forest species and populations, and (3) development and use of new tools to identify populations at risk and geographic areas that will provide suitable habitat in the future. Forest trees exist within distinct geographic populations created by climatic shifts, evolutionary processes, the availability of suitable habitats, and other environmental factors. These processes have occurred over millennia and continue to shape the biogeography and genetic structure of these species. Forest trees in Western North America are being defined on the basis of molecular markers and quantitative traits. For example, studies of whitebark pine (*Pinus albicaulis*) and western white pine (*P. monticola*) demonstrate the existence of several distinct populations that likely developed via the long-term processes described above. These studies and others have shown that distinct populations exist within western conifer species and further indicate that the biogeographies of forest species are quite dynamic over time and space. Here, we present a case study using genetic data from whitebark pine and western white pine coupled with landscape-based, plant-climate modeling. Suitable contemporary habitat is

accurately predicted for both species based on presence/absence of field observations. General circulation models were used to predict areas of suitable habitat for both species under the current climate and the projected climate around the year 2030. These models predict that these species will respond differently to projected climate change. Suitable habitat (i.e., climate space) for whitebark pine is predicted to decline dramatically by ca. 2030. Populations in the lower latitudes (below 45° N) and those persisting at the low elevation limits show the greatest threat to extinction from climate change. Predictions also indicate that suitable habitat for western white pine will be reduced in some southern latitudes, whereas the suitable habitat will be increased in the northern latitudes of its distribution. For whitebark and western white pine, both molecular markers and quantitative traits frequently reveal congruent genetic structure for species conservation. The combination of genetic studies with climate modeling can provide base-line tools that will enable managers to focus genetic conservation efforts on populations at highest risk while restoring areas that have the lowest risk for predicted climatic extirpation. The ability to define forest populations and predict landscape-level effects of climate change is critical for sustaining future forest health.

Keywords: Bioclimatic models, biogeography, genetic conservation, western white pine, whitebark pine.

Introduction

Climate change and associated glacial/interglacial cycles have had a profound impact on the biogeography of plant communities. Paleocological data from pollen sediment cores and packrat middens have been used to uncover many of the past plant-climate associations (e.g., Thompson and Anderson 2000, Thompson and others 1993, Whitlock 1993). These climatic oscillations have also affected intra-species genetic relationships that are recorded in changes in gene frequencies or adaptive traits, or both (Davis and Shaw 2001). In western North America, most forest trees have

broad, disjunct distributions that are associated with variable environmental conditions. The combination of disjunct intra-species distributions and environmental variables can contribute to barriers in gene flow and changes in adaptive or neutral gene frequencies, or both, caused by selection or genetic drift (Richardson and others 2005). Population genetics typically assess neutral genetic variation and provide estimates of genetic diversity and partitioning among a spatial hierarchy. Population genetic studies can estimate both contemporary and historical gene flow, and thereby aid in elucidating past intraspecies relationships and distributions. Such genetic data, coupled with paleoecological data, provide inferences to historical changes in plant biogeography and postglacial expansion/contraction (e.g., Godbout and others 2005, Jackson 2006, Magri and others 2006, Petit and others 2004, Richardson and others 2002, Thompson and Anderson 2000). Knowledge of the population genetic structure and adaptation is critical for successful forest health management and conservation/restoration efforts.

Five-needled pines are keystone species in numerous ecosystems in western North America. This paper focuses on two case studies: whitebark pine (*Pinus albicaulis* Engelm.) and western white pine (*P. monticola* Dougl. ex D. Don). Whitebark pine occupies subalpine habitats across the northern Rocky, Cascade, and Sierra Nevada Mountains and mountains of northern Nevada. Seed dispersal is dependent on a coevolved mutualism with Clark's nutcracker (*Nucifraga columbiana*) that deposits seeds in caches for later consumption (Tomback 2001). Flight distances with seeds vary but have been reported as far as 22 km (Richardson and others 2002, Vander Wall and Balda 1981). Whitebark pine has been recognized as critical for wildlife (Mattson and others 2001) and watershed stability (Arno and Hoff 1989).

Western white pine occupies a wider ecological niche than whitebark pine, occurring from sea level in Washington and British Columbia to a predominately subalpine distribution from Sierra Nevada northward to the Cascades. In the northern Rocky Mountains, this species occupies wet montane to subalpine habitats (Wellner 1962). It is valued as a timber species because of its rapid growth, straight

bole, and ability to regenerate as a seral species from wind-dispersed seeds. Since the early 1900s, the abundance of both five-needled pines has dropped precipitously, mainly from white pine blister rust caused by the fungus *Cronartium ribicola* (McDonald and Hoff 2001). Major efforts and resources have been directed toward restoration of five-needled pine ecosystems. Unfortunately, most restoration efforts have been operating on the assumption that the climate will remain stable. For successful management and restoration of five-needled pine ecosystems, future climate change must be considered and integrated into current efforts and plans for species management/conservation.

Research corroborated by diverse sciences provides strong evidence that the Earth is experiencing a warming process driven by increased greenhouse gas concentrations. Records indicate that mean global temperature increased 0.6 °C during the 20th century, and global temperature is expected to further increase 1.4 to 5.8 °C over the next 100 years (IPCC 2001). This predicted rate of climate change is unprecedented within the available historical records. Because of the extremely fast rate of predicted climate change, determining the responses of forest tree species and populations is one of the major challenges for future threat assessment (Jump and Peñuelas 2005). Recent studies have developed plant-climate models that are highly accurate in predicting a suitable contemporary climate envelope for several western North American species (Rehfeldt and others 2006). These models serve as means to project future climate scenarios using general circulation models (GCMs).

It is well known that historical climatic oscillations have dramatically shifted species distributions. However, these historical climate changes typically occurred at a relatively slow rate over the course of millennia. The predicted rate of climate change, due to the anthropogenic release of greenhouse gases, will greatly exceed historical rates (IPCC 2001). Studies based on several climate models have shown the predicted rate of climate warming could jeopardize plant taxa with limited seed dispersal distances or disjunct habitats (Malcolm and others 2002). In this analysis, we develop an approach to evaluate potential impacts of future climate change on genetic diversity and biogeography of whitebark pine and western white pine. Our objectives include:

1. Assess the existing genetic diversity and population structure of whitebark pine and western white pine.
2. Model each species climate space (i.e., suitable habitat) for contemporary and year 2030 climate.
3. Use the predicted biogeographical changes to prioritize genetic conservation efforts by identifying populations at risk and determining areas suitable for restoration.

Material and Methods

Population Genetic Analyses

Sampling of needle and bud tissue was conducted across the range of both whitebark pine and western white pine. For whitebark pine, 28 sites were sampled and were combined into 6 regions ($n = 21$ to 72, Richardson and others 2002). Western white pine was sampled from 15 sites ($n = \sim 30$) across the range (Kim and others unpublished). Plant tissue was used to extract genomic DNA using previously published protocols (Kim and others 2003, Richardson and others 2002).

For whitebark pine, two types of DNA markers were used: mitochondrial DNA (mtDNA) and chloroplast microsatellites (cpSSR) haplotypes. These markers are uniparentally inherited enabling separate estimates of gene flow via pollen by cpSSR and via seed dispersal by mtDNA. Protocols for DNA typing and analyses are described by Richardson and others (2002). For western white pine, an amplified fragment length polymorphism (AFLP) approach was used to provide high-resolution markers that are typically biparentally inherited (i.e., nuclear markers). Western white pine AFLP protocols and analyses are described in Kim and others (2003).

Plant-Climate Modeling

Whitebark pine and western white pine bioclimatic models used species presence/absence data for more than 118,000 point locations described geographically by latitude, longitude, and elevation (see “Acknowledgments”). Climate was estimated for each of these locations using a spline climate model (Rehfeldt 2006). This procedure yielded a

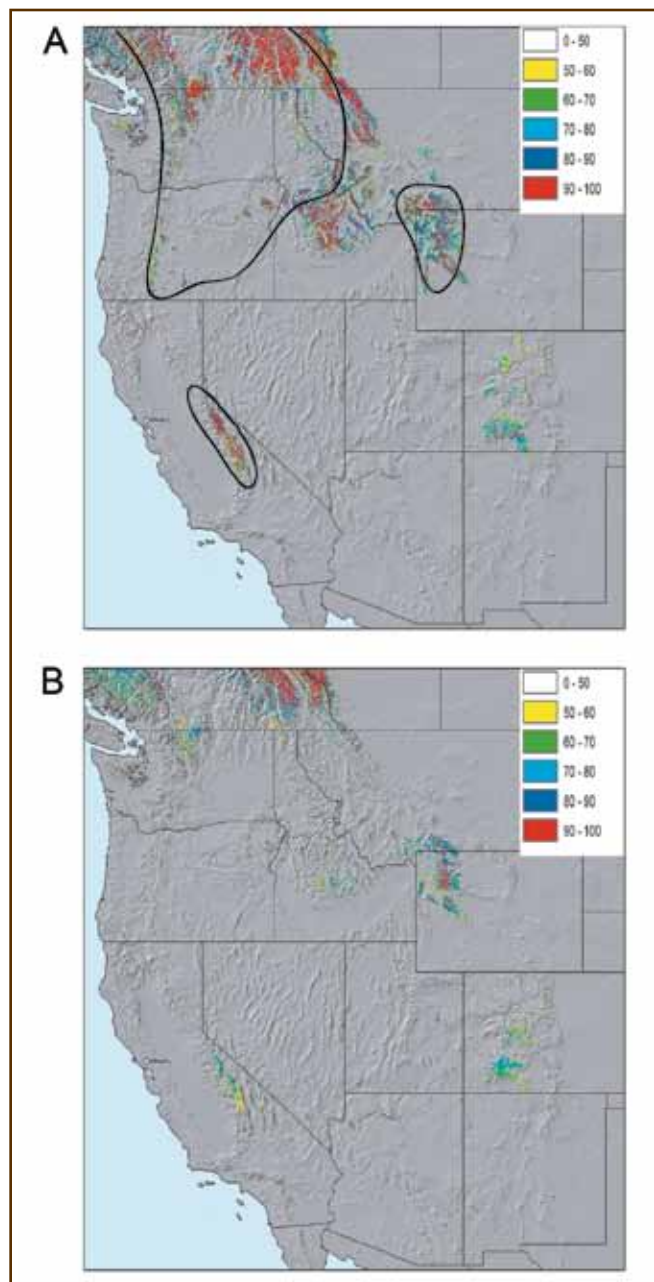


Figure 1—(A) The predicted contemporary distribution of whitebark pine (*Pinus albicaulis*) in the Western United States and southwestern Canada using a plant-climate model (Rehfeldt and others 2006). Pixel colors represent votes for whitebark pine occurrence; votes between 0 and 50 are not shown. The black drawn lines represent predicted metapopulations of whitebark pine based on chloroplast microsatellite molecular markers (Richardson and others 2002). (B) The predicted 2030 distribution of whitebark pine using the average of two general circulation models.

data set that described species presence or absence based on 33 climate variables representing simple interactions of

temperature or precipitation, or both, in relation to time, all of which have some relationship to plant responses. Random forests multiple classification tree analysis (Breiman 2001) was then used to identify climate variables important for predicting species presence and to build decision-tree-based bioclimatic models.

Arc map software (ESRI Inc.) was used to project the models predictions on a map. For this approach, the spline climate model was used to estimate climate in each 1-sq-km grid across the Western United States (more than 5.9 million pixels). Each pixel was then assessed using 100 independent decision trees. For each decision tree, a vote for or against species presence was cast. Pixels receiving vote tallies of 50 percent or greater were defined as having suitable climate space. These pixels are represented by the colored areas in Figures 1 and 2. These procedures are described in depth elsewhere (Rehfeldt and others 2006).

The response of a species contemporary suitable climate space under climate change was then assessed. The spline climate model was updated to the predicted climate of beginning in ca. 2030 using an average from the HadCM3GGa1 of the Hadley Centre (Gordon and others 2000) and CGCM2 of the Canadian Centre for Modeling and Analysis (Flato and Boer 2001) general circulation models. The procedure described in the previous paragraph was then repeated using the updated climate model to predict future climate for each pixel space.

Results

Whitebark Pine

Population Genetics—

Analysis of mtDNA reveals three haplotypes that are discretely partitioned among regions with two introgression zones. One introgression zone apparently occurs across a broad area in central Idaho, and another introgression zone occurs across a narrow area in the northern Cascades near Mount Rainier, Washington. The introgression of mtDNA haplotypes in the northern Cascades has been proposed to have resulted from the colonization of previously glaciated habitat during Holocene warming (Richardson and others 2002).

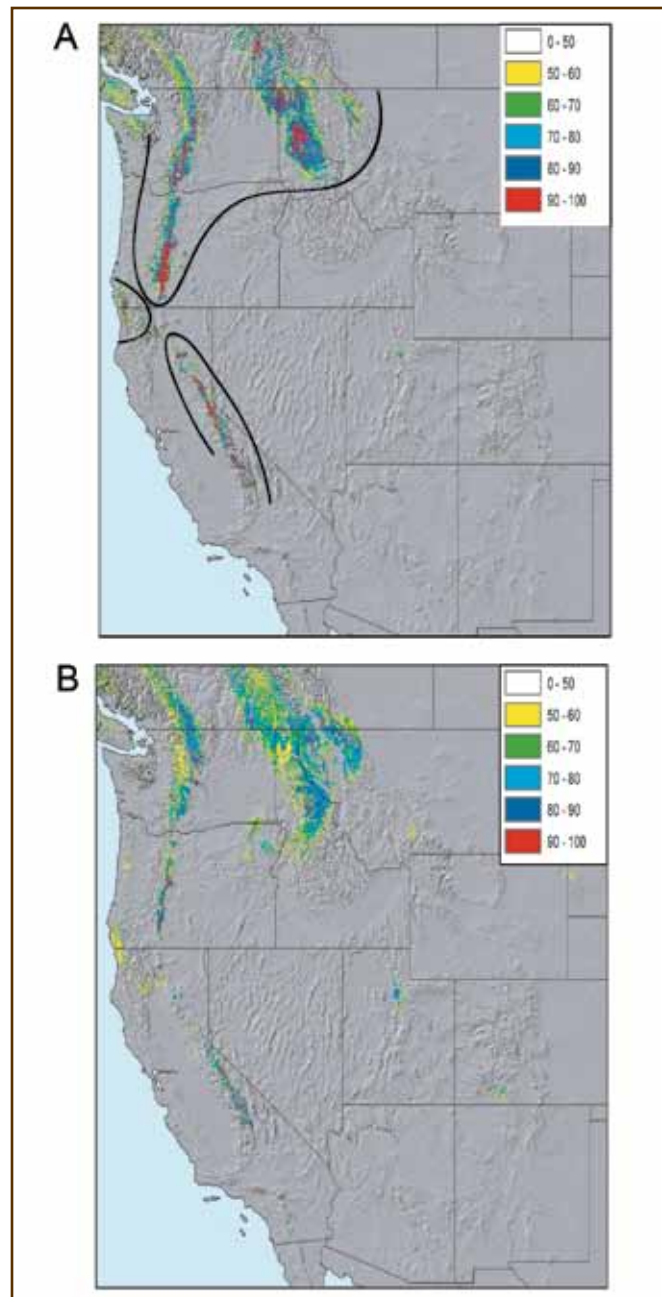


Figure 2—(A) The predicted contemporary distribution of western white pine (*Pinus moniticola*) in the Western United States and southwestern Canada using a plant-climate model (Rehfeldt and others 2006). Pixel colors represent votes for western white pine occurrence. The black drawn lines represent the predicted metapopulations of western white pine based on AFLP analyses. (B) The predicted 2030 distribution of western white pine using the average of two general circulation models.

Analysis of pollen gene flow using cpSSRs reveals broader population boundaries that include mtDNA

introgression zones. These data circumscribe three metapopulations: the Sierra Nevada Mountains, the Yellowstone region, and a large Pacific Northwest region that includes the northern Rocky Mountains and the Cascade (Figure 1A). Private alleles were typically found at low frequency within each region; however, one notable exception was the Sierra Nevada metapopulation that had a private allele at high frequency. Genetic diversity has been described as moderately high for cpSSR markers (Richardson and others 2002) to moderately low using allozyme analysis (reviewed in Bruederle and others 2001). Furthermore, genetic characterization of isolated populations in southern Oregon, northern California, and Nevada are still needed to assess genetic conservation efforts in these areas.

Plant-Climate Modeling—

The contemporary plant-climate model for whitebark pine fits its present distribution with remarkable precision. Comparisons between ground-truthed whitebark pine locations and predicted areas of suitable habitat showed an overall error rate of only 2.47 percent (Figure 1A). The mapped predictions for the occurrence of whitebark pine were more accurate than Little's (1971) published range maps (Warwell and others 2006). Based on two GCMs (Hadley and Canadian Centre for Climate Change), the projected climatic space for whitebark pine in ca. 2030 shows a dramatic change in comparison with the contemporary prediction. An estimated 70 percent of the current climate space for whitebark pine will be lost by 2030. Most of predicted loss occurs in the southern latitudes of the whitebark pine range, including the Oregon Cascades, Siskiyou, and northern Nevada mountain ranges, where current suitable habitats are limited to mountaintops in many locations (Figure 1B). The elevation of suitable climate space is also expected to rise 330 m, thereby leaving only the Sierra Nevada Mountains as a potential refuge with suitable climate space below 40° N latitude. In contrast, the model does predict suitable climate space will occur in central Colorado, which may represent a fundamental climate niche for whitebark pine. In the northern Rockies and Cascades, whitebark pine climate space is predicted to persist only among the highest elevations found in Yellowstone, south-central Idaho, Glacier National Park, and the northern Cascades. Areas north

of 50° N will also likely fit climatic space for whitebark pine, but these areas fell outside of the geographic window and are not included in the analyses.

Western White Pine

Population Genetics—

Analyses of AFLP loci discern three metapopulations in western white pine. These include a Pacific Northwest metapopulation that extends from the northern Rockies west to the Pacific and south along the Cascade Mountains to central Oregon. A distinct metapopulation exists in the Siskiyou Mountains of southern Oregon and northern California, and another metapopulation extends into the Sierra Nevada Mountains (Figure 2A). Like most other pines, western white pine contained moderate to high levels of heterozygosity in all three metapopulations (Kim and others unpublished).

Plant-Climate Modeling—

For western white pine, the contemporary climate space also fits with the current distribution with a small overall error of 3.85 percent. The projected current distribution of western white pine is shown in Figure 1A. The predicted 2030 climate space for western white pine provides a strong contrast with that of whitebark pine. Western white pine climate space is expected to increase 29 percent by 2030 (Figure 2B). Overall, suitable climate space for western white pine is expected to increase 157 m in elevation by 2030. This predicted expansion occurs in the northern latitudes, with notable expansion into northwestern Montana and northward into the Canadian Rocky Mountains. In the more southern latitudes of California and southern Oregon, populations are projected to persist at higher elevations. Predicted climate space is reduced in southern Oregon, thereby rendering the projected Sierra Nevada population more disjunct from the southern Cascades.

Discussion

Whitebark Pine

Past climate change has shaped the biogeography of whitebark pine and, hence, genetic relationships and potential adaptive traits. Since the last glacial maximum, whitebark

pine has apparently responded to the warming climate after the last glacial maximum by colonizing new habitat opened up by receding glaciers of the Canadian Rocky Mountains and the northern Cascades. Holocene expansion into the northern Cascade Mountains likely originated from source populations to the south and east of the north Cascades, based on the mtDNA at the contact zone (Richardson and others 2002). In contrast, a wider distribution of whitebark pine in more southern latitudes during the last glacial maximum was probably constricted to higher elevations by subsequent Holocene warming. This constriction in suitable habitat appears particularly dramatic in northern Nevada, where whitebark pine currently persists only on the highest mountaintops.

Population genetics of whitebark pine have identified distinct regional metapopulations occupying the Pacific Northwest (i.e., northern Rockies and Cascade Mountains), the greater Yellowstone region, and the Sierra Nevada. Other populations in the Great Basin and central Oregon have not been analyzed with mtDNA or cpSSR markers. However, allozyme analyses have shown distinct genetic structure among the Great Basin whitebark pine populations (Yandell 1992). Further studies to characterize genetic diversity and structure of whitebark pine in Washington and Oregon are in progress (Personal communication. C. Aubry. 2006. Area Geneticist, Olympia National Forest, 1835 Black Lake Blvd. Suite A, Olympia, WA 98512), and studies are assessing rangewide and regionwide adaptive traits in whitebark pine (Bower and Aitken 2006, Warwell and others unpublished). These studies and population genetic analyses are essential to determine appropriate genetic conservation efforts and proactive management that consider predicted climate change.

The contemporary climate space predicted using the plant-climate model is extremely precise for the distribution of whitebark pine. This precision has also been demonstrated with other western plant species (Rehfeldt and others 2006). The predicted suitable climate space for whitebark pine shows a dramatic reduction in the year 2030. At the highest risks are populations that currently only exist on mountaintops, where projected suitable climate space will be entirely lost from the region. Loss of local whitebark pine

populations is predicted to occur throughout the Oregon Cascades and mountaintops of northern Nevada, and further result in disjunct populations in the Sierra Nevada that will persist only at the highest elevations. These populations that face the highest risk of extirpation from predicted climate change and the white pine blister rust fungus should take priority for seed bank collections. Besides the Sierra Nevada, the plant-climate model predicts four major regions where climate space remains in the Western United States. These regions include the highest mountain ranges: the northern Cascades, Glacier National Park, south-central mountain ranges in Idaho, and the greater Yellowstone region. These regions that contain suitable climate space in 2030 predictions should be considered as priority regions for restoration efforts.

Western White Pine

Like whitebark pine, western white pine populations were also shaped by past climate change. This process created genetic structure with similarities to whitebark pine: a large Pacific Northwest metapopulation with partitions in southern Oregon and northern California (Kim and others unpublished). This population structure is also supported by an earlier study of isozyme markers (Steinhoff and others 1983). In addition, similar findings were reported in a study of adaptive traits for western white pine, showing a relationship between latitude and shoot elongation with a transition zone in southern Oregon (Rehfeldt and others 1984).

Again, the bioclimatic model predicted an excellent fit to the known distribution of western white pine. However, predicted 2030 climate space for western white pine shows a stark contrast to whitebark pine. The predicted increase in climate space is evident as an expansion in the northern Rocky Mountains that apparently relates to a slight increase in moisture. In the Sierra Nevada and Siskiyou Mountains, predicted climate space in 2030 does not show expansion. In these areas, moisture levels are predicted to increase more dramatically relative to the Pacific Northwest, and predicted climate becomes less suitable for western white pine, possibly owing to increased competition. Whereas losses of genetic diversity seem relatively small compared to whitebark pine, some areas of western white pine may

be at risk. Reductions in the abundance or extirpation of small isolated populations are conceivable for the Siskiyou Mountains and other areas of northern California where the western white pine populations exhibit a unique and diverse genetic background. Such populations should be considered as a priority for genetic conservation efforts.

Model Refinements and Uncertainties

Further refinements in the plant-climate model can be achieved by integrating knowledge from molecular and quantitative genetics. An important point is that this model currently utilizes ground-truthed locations for the entire distribution of each species for current and future predictions, ignoring genetic adaptation to local/regional climate. This model assumes that each climate-adapted population will be able to occupy its climate space in the future regardless of the geographic distance. For example, it is clear that the Sierra Nevada metapopulation of western white pine is genetically different and occupies a climate that is different from western white pine in the more northern latitudes. However, it is unclear where the populations occupying current climate space in Sierra Nevada may move in the future. Future plant-climate model improvements should be able to correlate climate variables and genetic data. Ongoing studies are focused on the delimiting population-based adaptive traits and refining molecular genetic data from western white pine populations to project suitable climates for these populations under current and future climate scenarios. Similar studies can be conducted with whitebark pine and other tree species, but this will require investing in efforts to conduct thorough, rangewide genetic studies.

Most uncertainty in the plant-climate model is associated with future climate scenarios and the GCMs. Precipitation is a major factor for predicting suitable climate space of plant species. For the Western United States and Canada, models display consistency for increased winter precipitation; however, summer precipitation predictions remain inconsistent (IPCC 2001). In addition, the time scale between loss of suitable climatic space and extirpation of local populations remains uncertain. Much of this uncertainty comes from site-specific biotic interactions. For example, two potential scenarios can be envisioned: (1) slow

attrition from succession where whitebark pine persists for decades or (2) stress from maladaptation leading to insect (e.g., bark beetle) outbreaks or disease (e.g., root rots) epidemics resulting in local extirpation within years. Projected scenarios are further complicated by mortality and loss of cone crops to white pine blister rust and insect attack. Continued studies are needed to address these uncertainties and improve predictions of climate-change impacts on distribution and population genetic structure of whitebark pine and western white pine.

Summary

Case studies were presented for whitebark pine and western white pine using plant-climate modeling of current and future climate predictions and genetic studies. These studies indicate dramatic differences in these two species' responses to climate change from the past and the future. These responses are largely dependent on the breadth of the ecological niches and life-history characteristics of each species. For whitebark pine, specialization for subalpine habitat is predicted to severely limit its distribution under future climate scenarios. Alternatively, coevolved relationships with Clark's nutcracker have enabled this species to colonize new habitat following Holocene warming. This characteristic may also aid its migration northward in the future, but successful migration to newly established suitable climate space is contingent upon the rate of climate change and the severity of impact from white pine blister rust and insect attack. Because western white pine has broader ecological range, it may be better suited to buffer effects of climate change. This ecological range is reflected in its predicted resilience under the future climate scenario. The creation of the plant-climate models and determining the impact from future climate scenarios for a particular species are dependent upon basic genetic research, both ecological and population genetics. Current research is focused on integration of genetic and climate data for these species. This approach represents a synthesis of multidisciplinary research to provide useful guidelines for forest management plans for genetic conservation and restoration that consider future climate change projections.

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- Donna Dekker (Lynn Wells and Ray Wiseman 1991 Report)
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- Jodie Krakowski, Master's Thesis, University of British Columbia
- RMRS Field Data
- RMRS Northern Idaho Habitat Type Data
- Ulla Geralyn Yandell, Master's Thesis, University of Nevada, Reno
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Threats to Private Forest Lands in the U.S.A.: A Forests on the Edge Study

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Abstract

The Forests on the Edge project, sponsored by the USDA Forest Service, uses geographic information systems to construct and analyze maps depicting threats to the contributions of America's private forest lands. For this study, watersheds across the conterminous United States are evaluated with respect to the amount of their private forest land. Watersheds with at least 10 percent forest land, of which 50 percent is privately owned, are then ranked relative to the contributions of their private forest lands to water quality, timber supply, at-risk species habitat, and interior forest. In addition, threats from housing development, fire, air pollution, and insect pests and disease to private forest land contributions are assessed. Results indicate that private forest lands contributions and threats are concentrated in the Eastern and Southeastern United States but are also distributed throughout the north-central, central hardwoods, and Pacific Northwest regions.

Keywords: Ecological services, forest contributions, geographic information systems, fourth-level watershed, land use change, private forest, sustainable forest management.

Introduction

America's forest lands contribute in a myriad of ways to the economic, ecological, and social well-being of the

Nation. Increasingly, however, forest lands are threatened from a variety of sources including urbanization, low-density housing development, climate change, invasive flora and fauna, wildfire, pollution, fragmentation, and parcelization. The increasing emphasis on sustainable forest management and loss of open space requires quantitative and spatial assessments of the impacts of threats to forest lands and their contributions. The Forests on the Edge (FOTE) project, sponsored by the Cooperative Forestry Staff, State and Private Forestry, USDA Forest Service, conducts map-based assessments of threats to the private forest lands of the United States using spatial data layers and geographic information systems. The objectives of the study described here are threefold: (1) to construct nationally consistent data layers depicting the spatial location of private forest lands and their contributions; (2) to construct similar layers depicting threats to the contributions of private forest land from sources such as conversion to urban and exurban uses, wildfire, and pollution; and (3) to identify watersheds whose private forest lands simultaneously make the most important contributions and face the greatest threats.

The Montreal process criteria and indicators provide an appropriate context for framing and conducting these assessments (McRoberts and others 2004). For example, criterion 2, maintenance of the productive capacity of forest ecosystems, includes indicators related to forest area and timber production; criterion 3, maintenance of forest ecosystem health and vitality, includes indicators related to fire, wind, disease, and insects; and criterion 4, conservation and maintenance of soil and water resources, includes indicators related to the contributions of forests to water quality.

Data Layers

All data layers were obtained as or constructed to be nationally consistent and were summarized at the spatial scale of fourth-level watersheds (Steeves and Nebert 1994). Watersheds were selected as the analytical units because they highlight the important connections between private forests and ecological processes.

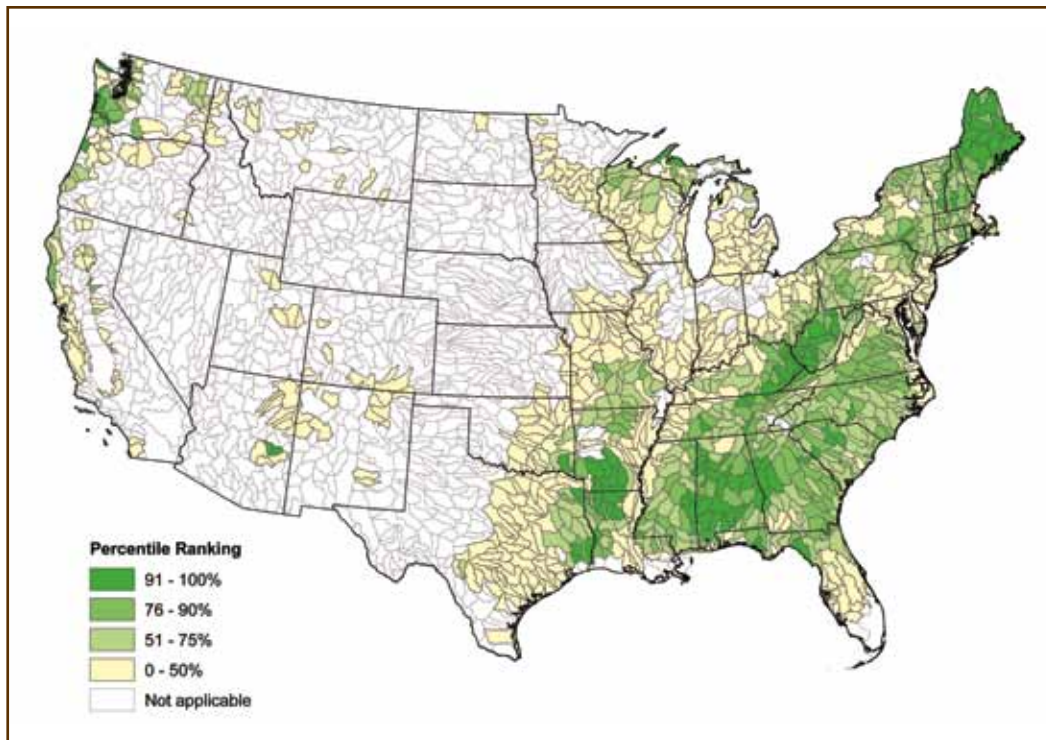


Figure 1—Percentile rankings of watersheds with respect to percentage of private forest land.

Private Forest Contributions

Area of Private Forest Land—

A 100-m-resolution forest ownership layer was constructed by aggregating the classes of the National Land Cover Dataset (NLCD) (Vogelmann and others 2001) into forest and nonforest classes and using the Protected Areas Database (PAD) (DellaSalla and others 2001) to distinguish ownership and protection categories. The emphasis for this study was private forest land, which includes tribal, forest industry, and nonindustrial ownerships. Stein and others 2005a, 2005b) provide detailed information on this layer. Only watersheds with at least 10 percent forest land, of which 50 percent is privately owned, qualified for subsequent analyses. Figure 1 depicts the percentile rankings of qualifying watersheds relative to the percentage of privately owned forest land; for example, a watershed in a 91 to 100-percentile category has a higher percentage of private forest land than at least 90 percent of the qualifying watersheds.

Water Quality—

Private forest lands provide nearly 60 percent of all water flow from U.S. forests and nearly 30 percent of the water

flow originating on land in the Lower 48 States (Personal communication. Thomas C. Brown. 2004. Hydrologist, Rocky Mountain Research Station, 2150 Centre Ave. Bldg. A, Suite 376, Fort Collins, CO 80526). Water flow from private forests is generally considered clean relative to water flow from other land uses and, therefore, makes a positive contribution to water quality. The water quality layer depicts the contribution of private forest land to the production of clean water and is based on three underlying assumptions: (1) water bodies near the heads of hydrologic networks are more sensitive to the loss of forest buffers than water bodies near the bases of the networks, (2) the presence or absence of upstream forest buffers influences water quality downstream in the networks, and (3) forest lands throughout watersheds, not just those in immediate proximity to water bodies, are important when considering the contributions of private forest land to water quality (FitzHugh 2001).

The water quality layer was constructed from two underlying layers, the forest ownership layer and the National Hydrography Dataset (NHD) (USGS 2000), which depicts water bodies in the 48 contiguous States. The

layer was constructed in four steps: (1) 30-m buffer was constructed around all water bodies, (2) the buffers were intersected with the private forest land class of the forest ownership layer to quantify the amount of private forest land in proximity to water bodies, (3) each buffer segment was assigned to one of four categories based on the relative position of the segment to the head of its hydrologic network, and (4) for each watershed, the percentage in each of the four categories was determined. Water quality index (WQI) was then calculated for each watershed as,

$$WQI = 0.6[A_1 + (A_1A_2)] + 0.4(0.48B_1 + 0.24B_2 + 0.16B_3 + 0.12B_4)$$

where A_1 = percentage of watershed in private forest land, A_2 = percentage of total forest land in watershed that is privately owned, B_1 = percentage of private forest land buffer in the first category (nearer head of hydrologic network headwater), B_2 = percentage of buffer in the second category, B_3 = percentage of buffer in third category, and B_4 = percentage of buffer in fourth category (farthest downstream from the head of hydrologic network). Variables A_1 and A_2 represent private forest coverage throughout the watershed, and variables B_1 through B_4 represent private forest coverage in the buffers. In WQI, A_1 and A_2 are collectively weighted 0.6, whereas variables B_1 through B_4 are collectively weighted 0.4 to reflect the third assumption above. Watershed boundaries for this and all other layers were determined using Steeves and Nebert (1994).

Timber Supply—

Private forest lands make a substantial contribution to America's timber resources, accounting for 92 percent of all timber harvested in the United States in 2001 (Smith and others 2004). The timber supply layer depicts the ranking of watersheds relative to an index of the importance of private timberland and is based on Forest Inventory and Analysis (FIA) plot data (<http://www.fia.fs.fed.us/tools-data/default.asp>). The private timberland importance index (TI) is based on three subindices of contributions of the timberland subset of private forest land. Timberland is defined by the FIA program as forest land that has not been withdrawn from production and that is capable of producing 20 cubic feet per year of industrial wood. For each watershed, the three subindices are calculated as follows: (1) growth index (GI)

is the average growing stock volume growth rate on private timberland in a watershed relative to the average across all private timberland, (2) volume index (VI) is the average net growing stock volume per acre on private timberland in a watershed relative to the average on all private timberland, and (3) area index (AI) is the ratio of private timberland area to total private land area in a watershed relative to the same ratio across all watersheds. TI was calculated for each watershed as, $TI = AI(GI + VI)$.

At-Risk Species—

Private forests provide the key to conservation for many species. In the Pacific Northwest, they provide significant habitat for the spotted owl (Holthausen and others 1995). NatureServe and its member Natural Heritage Programs and Conservation Data Centres, prepared a geographic data set depicting the number of at-risk species occurring on private forested lands within fourth-level watersheds in the Lower 48 States of the United States. At-risk species are defined as species with element occurrences (EO) that have been observed by an authoritative source within at least the last 50 years, and are either: (1) federally designated under the Endangered Species Act (Endangered, Threatened, Candidate, Proposed), or (2) designated as critically imperiled, imperiled, or vulnerable according to the NatureServe Conservation Status Ranking system (G1/T1- G3/T3) (<http://www.natureserve.org/explorer/ranking.htm>). An EO is an area of land or water, or both, in which a species or natural community is, or was, present.

NatureServe selected populations that only occur on private forested lands by conducting a geographic analysis comparing the location of at-risk populations with private forest locations (both protected and nonprotected). These species are labeled as forest-associated as opposed to forest-obligated because a separate analysis to refine this species list using knowledge of species habitat requirements and preferences was not conducted. Known data gaps include: (1) no at-risk species data available in Arizona, Massachusetts, and the District of Columbia, (2) no at-risk fish data available for Idaho, and (3) at-risk animal data in Washington are incomplete. Private forested lands were determined using the data layer described in the "Area of Private Forest Land" section.

Interior Forest and Habitat Contiguity—

Habitat contiguity is an index of the structural integrity of forests, an important conservation concern (Wear and others 2004). Habitat contiguity can be measured in terms of the amount of interior forest cover that is functionally distinct from forest edge. The interior forest layer was created using three steps. First, the forest cover layer described in the section “Area of Private Land” was used to identify forested pixels in each watershed. Second, forested pixels were labeled interior forest if 90 percent of the pixels in a surrounding 65-ha window were also forested. Third, the proportion of interior forest pixels in each watershed was determined, and all watersheds were assigned a percentile ranking based on this proportion. Note that a watershed could have very little forest land but a high proportion that satisfied the interior forest criteria.

Threats to Private Forest Lands**Development—**

The development layer depicts predicted threats to private forest lands resulting from conversion to urban or exurban uses. The layer is based on estimates of current population and housing density data obtained from the 2000 U.S. Census, and predictions of housing density increases. A spatially explicit model was used to predict the full urban-to-rural spectrum of housing densities (Theobald 2005). The model uses a supply-demand-allocation approach and is based on the assumption that future growth patterns will be similar to those in the past decade. Future patterns are forecast on a decadal basis in four steps:

1. The number of new housing units in the next decade is forced to meet the demands of the predicted populations.
2. A location-specific average population growth rate from the previous to current time step was computed for each of four density classes: urban, suburban, exurban, and rural.
3. The spatial distribution of predicted new housing units was adjusted with respect to accessibility to the nearest urban core area.
4. Predicted new housing density was added to the

current housing density under the assumption that housing densities do not decline over time.

For these analyses, predicted new housing was not permitted to occur on protected private land as indicated by PAD (DellaSalla and others 2001). The spatially explicit housing density predictions were combined with the forest ownership layer to identify watersheds with the greatest predicted conversion of private forest land to urban and exurban uses. Stein and others (2005b) provided detailed information on this layer.

Wildfire—

Although wildfire is one of the most compelling threats to forest land, particularly in the Western United States, predicting the threat of wildfire incidence is extremely complex and relies on a variety of regional models and regional variables. Further, even if the models could be readily implemented to construct a national layer, the geographic consistency of the layer would be questionable. Therefore, as a surrogate for wildfire risk, FOTE used the 1-km by 1-km-resolution current fire condition class (CFCC) data, which depict deviations of fire incidence from historical natural fire regimes and estimated efforts necessary to restore stands to historical regimes (Schmidt and others 2002). These data reflect the assignment of forest lands to one of three CFCC classes:

1. CFCC₁—forest lands with fire regimes that are within or near historical ranges and that can be maintained by treatments such as prescribed fire or fire use.
2. CFCC₂—forest lands with fire regimes that have been moderately altered from historical ranges and that may require moderate levels of prescribed fire, fire use, hand or mechanical treatment, or a combination to restore to historical fire regime.
3. CFCC₃—forest lands with fire regimes that have been substantially altered from historical ranges and that may need high levels of hand or mechanical treatment before fire is used to restore historical fire regimes.

The appeal of the CFCC classes is that they are objective, nationally consistent, and are assumed to correlate well

with the threat of wildfire incidence. Although these classes reflect the widely varying State-level commitments to wildfire mitigation efforts, they do not reflect ease of access to forest lands experiencing wildfire or the availability of resources to combat wildfires.

For each watershed, all private forest lands were assigned to one of the three CFCC classes, and a watershed-level index was calculated as,

$$CC = CC_1 + 2CC_2 + 4CC_3,$$

where CC_i is the area of private forest land in class $CFCC_i$. The wildfire layer used for this study depicts the percentile ranking of each watershed with respect to its CC index value.

Ozone

Ozone affects forest ecosystems by causing foliar lesions and rapid leaf aging, altering species compositions, and weakening pest resistance (Chappelka and others 1997, Miller and others 1996). It is the only gaseous air pollutant that has been measured at known phytotoxic levels at both remote and urbanized forest locations (US EPA 1996).

The ozone layer depicts private forest land threatened by ground-level ozone and was based on late summer observations by FIA field crews of ozone damage to bioindicator species known to be sensitive to ground-level ozone. Data for more than 2,500 FIA plots were available for the study. Each plot was assigned a biosite value based on a subjective assessment by trained observers of the quantity and severity of damages (Coulston and others 2003, Smith and others 2003). Inverse distance weighted interpolation was used to create a map of ozone damage. This map was then combined with the forest ownership layer to identify private forest land with elevated levels of ozone damage. For each watershed, the percentage of private forest land in moderate or high-damage categories was calculated.

Nitrate and Sulfate Deposition—

Acidic deposition, the transfer of strong acids and acid-forming substances from the atmosphere to the Earth, has become a critical stress affecting forested landscapes across the United States. Effects can include increased sulfate and nitrate levels in soils and waters, which, in

turn, can alter soil and water chemistry and affect trees and other living organisms that depend upon affected soils and waters (Driscoll and others 2001). The nitrate and sulfate layers were created from National Atmospheric Deposition Program data. The data were used to interpolate yearly wet sulfate and wet nitrate deposition maps using gradient plus inverse distance interpolation (Nalder and Wein 1998). This technique adjusts for elevational, longitudinal, and latitudinal gradients when present in the data based on local regression of the 20 nearest neighbors. The wet sulfate deposition maps (2000-2004) were then averaged to produce a map of average annual deposition 2000-2004 (kg/ha per year). The same was done for wet nitrate deposition. Cross-validation (Issaks and Srivastava 1989) was performed to estimate the bias and precision of the yearly map.

Insect Pests and Disease—

Forest insect pests and diseases can affect forest health, including the reduction of tree basal area. The Forest Health Monitoring Program of the USDA Forest Service formed a Risk Map Integration Team (RMIT) to coordinate the development of a nationally consistent database for mapping insect pest and disease risk. The RMIT developed a GIS-based multicriteria risk modeling framework based on Eastman's risk assessment process (Eastman and others 1997).

A five-step multicriteria process was used to construct a 1-km by 1-km-resolution map depicting risk (Krist and others 2006): (1) Identify a list of forest pests (risk agents) and their target host species; this is conducted at the regional level with certain models constrained to select geographic areas. (2) Identify, rank, and weight criteria (factors and constraints) that determine the susceptibility and vulnerability to each risk agent. (3) Standardize risk agent criteria values, and combine the resultant criteria maps in a final risk assessment using a series of weighted overlays. Users assign a level of potential to values within GIS layers that represent criteria. (4) Convert modeled values of potential risk of mortality for each pest to predicted basal area (BA) loss over a 15-year period. This is accomplished for each risk agent/forest host species pair included in the national risk assessment. (5) Compile the resultant values from step 4 and identify areas (1-km raster grid cells) on a national base

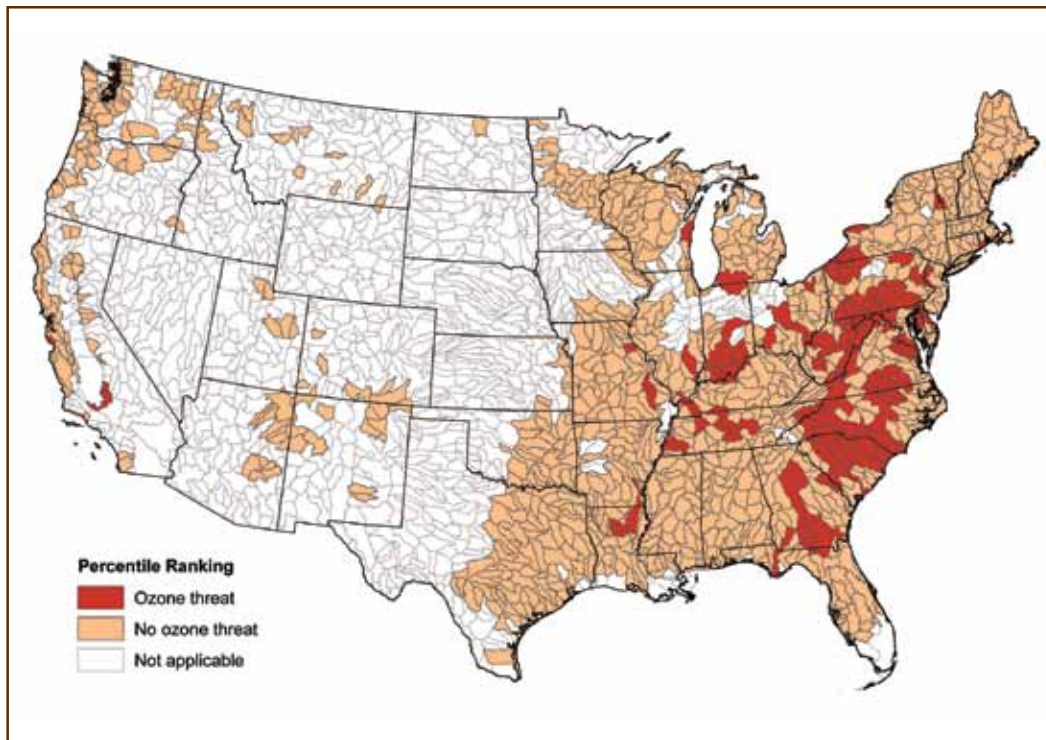


Figure 2—Percentile ranking of watersheds with respect to ozone threat.

map that are at risk of encountering a 25-percent or greater loss of total basal area in the next 15 years.

Methods

For each contribution and threat layer, the distribution of watershed index values was determined, and a percentile ranking was assigned to each watershed. Threats to particular contributions were evaluated in two steps. First, the averages of the contribution and threat percentile rankings were calculated on a watershed-by-watershed basis. Second, the distribution of the averages was determined and used to assign a new percentile ranking to each watershed. The results are depicted using percentile-based categories similar to those used for individual contributions and threats.

Results

The results are briefly discussed. Because of space limitations, only a few maps presenting data on individual layers of contributions or threats are displayed here. Instead, this paper focuses on some of the more interesting intersections of contributions and threats in the “Threats” section.

Contributions

Watersheds with the greatest percentages of private forest land are generally in New England, the Southeast, and the Pacific Northwest (Figure 1). The concentration in the East is not surprising because much of the forest land in the West is in public ownership. Watersheds whose private forests make the greatest contributions to water quality, timber supply, at-risk species habitat, and interior forest closely align with the watersheds with greatest amounts of private forest land.

Threats

Development threats to private forest land area are concentrated in southern New England and the Southeast, although some are also found in the Pacific Northwest; wildfire threats to private forest land (as indicated by the surrogate CC layer), are primarily in the Northeastern quadrant of the country. Data for these two layers are shown in conjunction with the “contributions” data in the next paragraphs. Threats to private forests from ozone are found scattered throughout the Eastern United States (Figure 2). Loss of basal area on

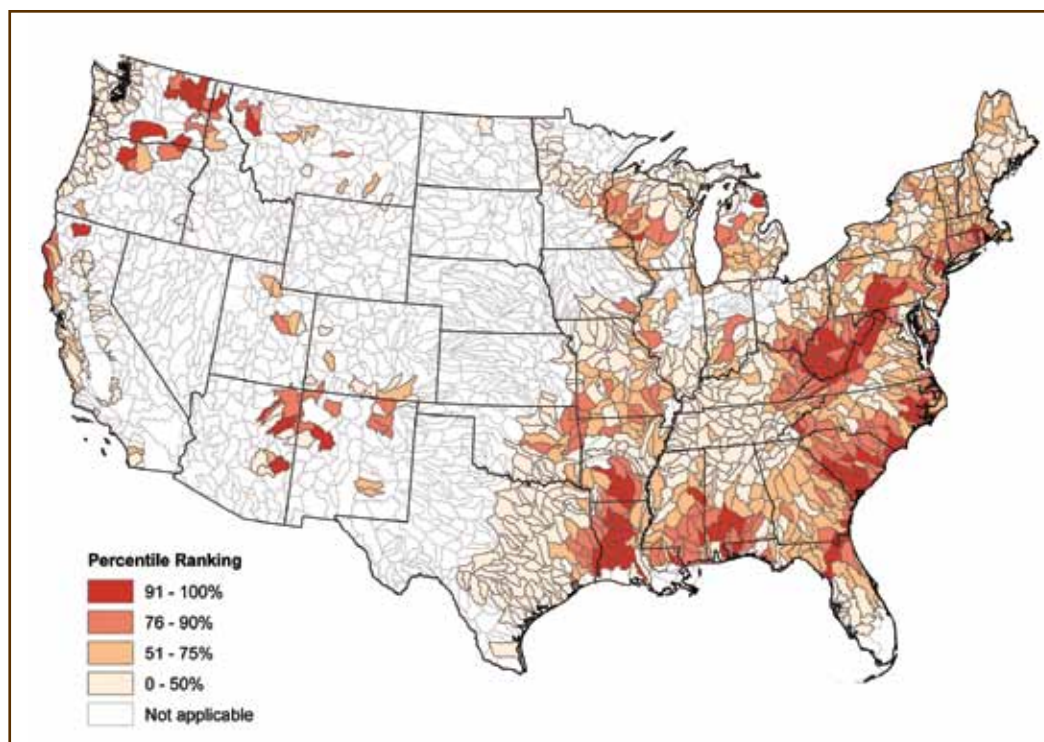


Figure 3—Percentile rankings of watersheds with respect to loss of basal area of private forests associated with insect pests and diseases.

private forest land from insect pests and diseases is most likely to occur in the East, but also in the lake States, the Southwest, California, and the Northwest (Figure 3).

Development threat to at-risk species associated with private forest is highest in the East, including Tennessee, North Carolina, and States immediately north and south of these States, as well as coastal California and the Pacific Northwest (Figure 4). Development threats to private interior forests are found throughout the Eastern United States and the Great Lakes area and are concentrated in the Southeast and Maine (Figure 5). Development threats to the contributions of private forest land to both water quality and timber supply are concentrated in southern New England and the Southeast (due to space limitations, these maps are not shown here).

Wildfire threats to both water quality and timber supply contributions are distributed throughout the East, the lake States, the central hardwoods region, and the Pacific Northwest (Figures 6 and 7).

Conclusions

Four primary conclusions may be drawn from this study:

1. The FOTE spatial approach to assessing threats to the contributions of private forest lands produces useful, visual information that is relatively easy to obtain. The only serious impediment is the difficulty in obtaining or constructing nationally consistent data layers that depict the contributions and threats of interest.
2. The watersheds making the greatest private forest contributions to water quality, timber supply, at-risk species, and interior forest are generally the watersheds with the greatest percentages of private forest land (i.e., those in the Eastern United States, particularly New England and the Southeast, and some watersheds in the Pacific Northwest). Two exceptions are noted. Some watersheds in western California and Florida do not have large amounts of private forest but do have large numbers of at-risk forest-associated species. In addition, some

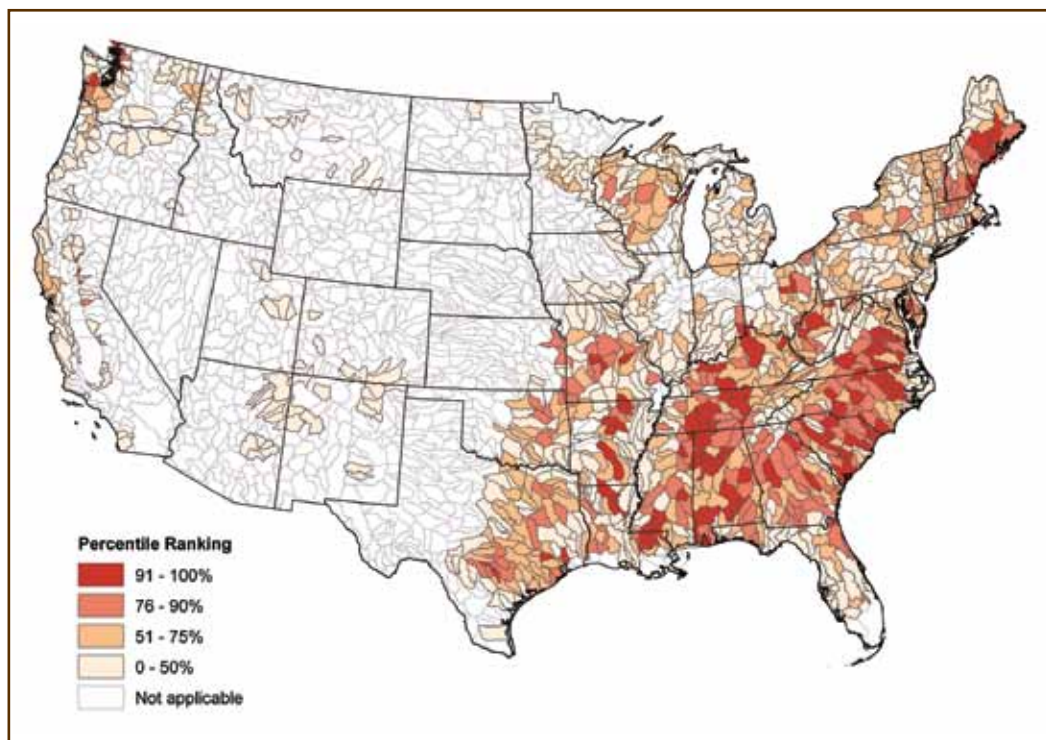


Figure 4—Percentile rankings of watersheds with respect to housing development threat to contribution of private forest land to at-risk species habitat.

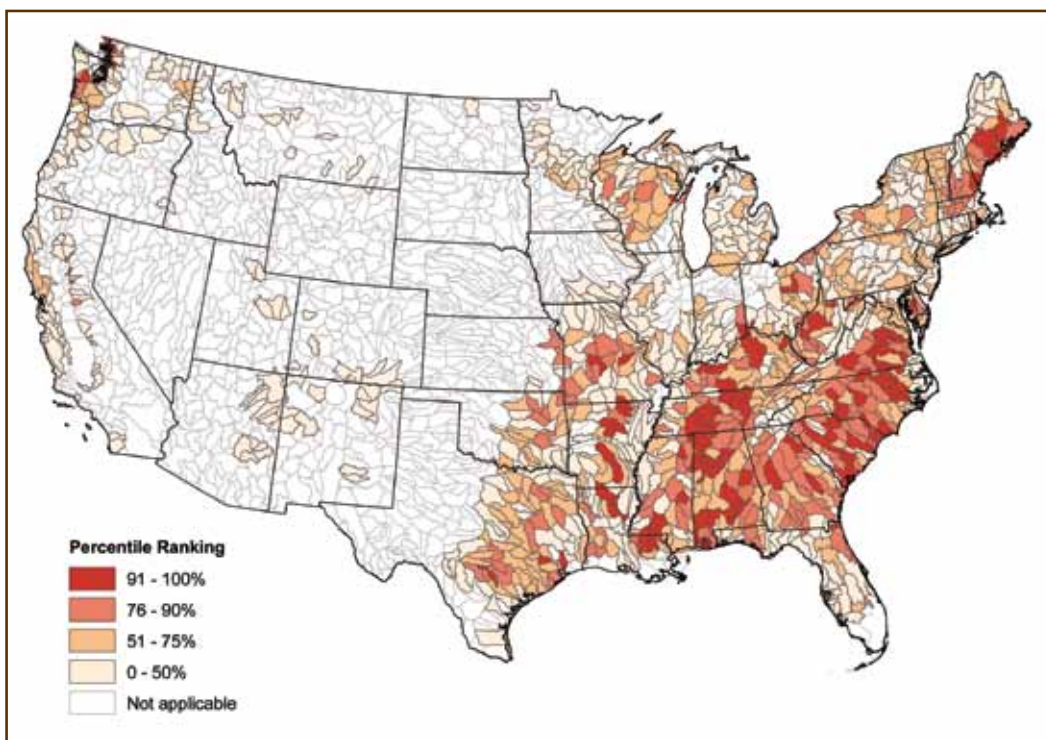


Figure 5—Percentile rankings of watersheds with respect to housing development threat to interior forest on private lands.

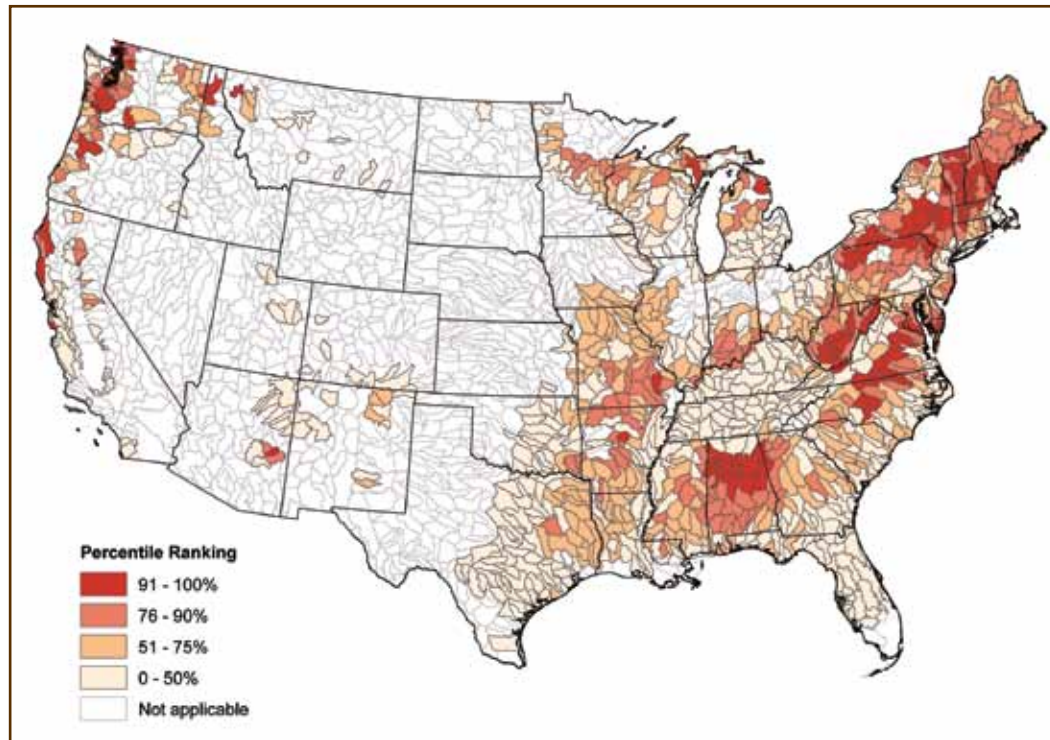


Figure 6—Percentile rankings of watersheds with respect to wildfire threat to contribution of private forest land to water quality.

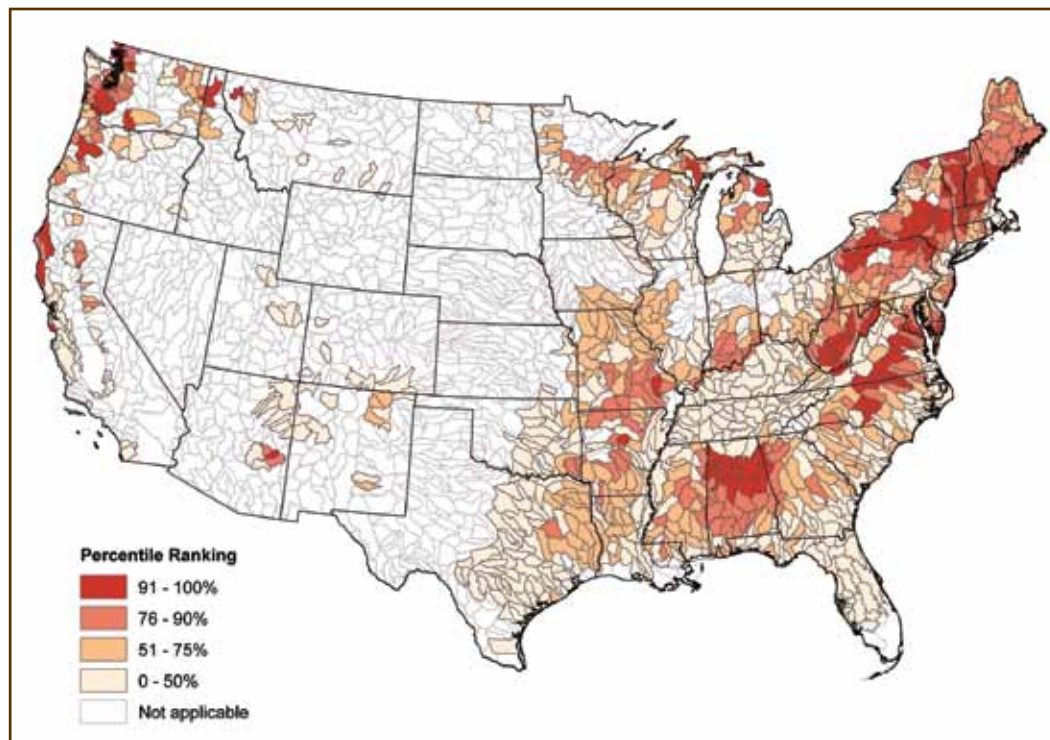


Figure 7—Percentile rankings of watersheds with respect to wildfire threat to contribution of private forest land to timber supply.

watersheds in eastern Texas, the Southwest, and Washington State also do not have large amounts of private forest but do have high proportions of private forest classified as interior forest.

3. Watersheds with the greatest development threat to the contributions to water quality, timber supply, at-risk species, and interior forest are also generally the watersheds with the greatest percentages of private forest land. The exceptions for at-risk species and interior forest noted in the previous paragraph apply here as well.
4. The CC surrogate for wildfire depicts the greatest threats to watersheds in the central part of the Eastern United States and the Pacific Northwest (although watersheds in the central part of the United States have relatively small percentages of private forest land). Percentile rankings of watersheds based on wildfire threat to private forest contribution to water quality and timber supply follow this pattern as well.

Future Forests on the Edge work will include assessment of additional contribution and risk intersections and the construction of an Internet-based system that permits users to select particular contribution and threat layers, options for combining them, and options for depicting the results. In addition, an assessment of national forests and grasslands most likely to experience increased pressures from housing development on adjacent lands is nearing completion. For more information on Forest on the Edge, go to our Web site at <http://www.fs.fed.us/openspace/fote>.

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A Spatial Model for Predicting Effects of Climate Change on Swiss Needle Cast Disease Severity in Pacific Northwest Forests

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Abstract

Swiss needle cast disease of Douglas-fir (*Pseudotsuga menziesii*) is caused by the ascomycete fungus *Phaeocryptopus gaeumannii*. Symptoms of the disease are foliage chlorosis and premature needle abscission due to occlusion of stomata by the ascocarps of the pathogen, resulting in impaired needle gas exchange. Severe defoliation and growth losses of 20 to 50 percent because of Swiss needle cast have been reported for about 150 000 ha of Douglas-fir plantations in western Oregon since 1996. Because the physiological effects of the disease (impaired CO₂ uptake and photosynthesis) are quantitatively related to the abundance of the pathogen (proportion of stomata occluded by ascocarps), pathogen abundance is directly related to disease and is a suitable response variable for assessing effects of climatic factors on disease. Climate factors most highly correlated with pathogen abundance are winter temperature and spring leaf wetness, and a model for prediction of disease severity based on these factors accounts for 77 percent and 78 percent of the variation in 1- and 2-year-old needles, respectively, for western Oregon sites. A trend of temperatures increasing by 0.2 to 0.4 °C during the winter months and spring precipitation increasing by 1.6 to 2.6 cm per decade since 1966 suggests that regional climate trends are influencing the current distribution and severity of Swiss needle cast disease. Forecasts of climate change in the Pacific Northwest region predict continued increases in temperatures during winter months of about 0.4 °C per decade through 2050, suggesting that the severity and distribution of Swiss needle cast is likely to increase in the coming decades as a result of climate change, with significant consequences for Pacific Northwest

forests. A climate-based disease prediction model is being developed as an online, interactive tool that can be used to guide further research, conduct extended model validations, perform climate change scenario analyses, and eventually to provide short- and long-term disease risk predictions and management cost/benefit analyses. The model will be useful for prediction of disease development trends under different climate change scenarios and temporal scales.

Keywords: Douglas-fir, foliage disease, forest pathology, *Phaeocryptopus gaeumannii*, *Pseudotsuga menziesii*.

History of the Problem

Swiss needle cast disease of Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) is caused by the ascomycete *Phaeocryptopus gaeumannii* (Rohde) Petrak. The disease, and the fungus that causes it, were first described from Douglas-fir plantations in Switzerland and Germany in 1925, and soon afterward reported from various locations in Europe, the British Isles, and Northeastern North America (Boyce 1940, Peace 1962). The causal agent, *P. gaeumannii*, was found to be abundant on foliage of diseased trees and was determined to be distinct from any previously described foliage fungi from coniferous hosts. Subsequent surveys of Douglas-fir in the Western United States found the pathogen was widespread throughout the Pacific Northwest region, where it had escaped notice because of its inconspicuous habit and negligible effect on its host. Boyce (1940) considered *P. gaeumannii* widespread but harmless on Douglas-fir in Western North America, and probably indigenous to the Pacific Northwest, where "...the fungus has been found at such widely separated localities in British Columbia, Washington, and Oregon that it must be considered generally distributed, although harmless, in the Douglas-fir region of the Pacific coast."

Douglas-fir is the only host of *P. gaeumannii*, which has accompanied cultivation of its host to various localities worldwide (Hood 1997, Temel and others 2003). Diseased trees have chlorotic foliage and may lose all but the current year's complement of needles (Figure 1). The earliest



Figure 1—Symptoms of Swiss needle cast, premature defoliation, loss of older needles.

confirmed records of *P. gaumannii* in western North America are herbarium specimens collected by J.S. Boyce from Oregon and California in 1916 (Boyce 1940). The fungus occurs on both coastal (*Pseudotsuga menziesii* var. *menziesii*) and interior (*P. menziesii* var. *glauca* (Beissn.) Franco) forms of Douglas-fir throughout the natural range of the host (Boyce 1940, Hood 1982). The fungus has also been reported from *P. macrocarpa* (Vasey) Mayr in New Zealand (Gadgil 2005), although it has not been reported on this host in North America.

Although Swiss needle cast disease in Europe first brought attention to *Phaeocryptopus gaumannii*, the fungus has long been considered a pathogen of negligible significance in western North American forests. Peace (1962) wrote of Swiss needle cast disease in Europe: “This is a classic case of a disease, of no importance in its native haunts, which has become damaging when transported to other areas.” With the growth of a commercial Douglas-fir Christmas tree industry in the Pacific Northwest in the 1970s, and the practice of shearing trees to promote a more compact growth form, Swiss needle cast first began to attract serious notice in its native region (Michaels and Chastagner 1984). Although episodic outbreaks of brief

duration in forest plantations in the Pacific Northwest were noted in the 1970s and 1980s (Russell 1981), Swiss needle cast was not considered a significant forest pathogen. The occurrence of Swiss needle cast disease primarily in forest plantations outside the native range of Douglas-fir, or on sheared Christmas tree plantations, has contributed to the perception of the disease as being primarily associated with inappropriate planting stock or stressed trees.

Current Extent and Impacts of Swiss Needle Cast in Pacific Northwest Forests

Since around 1990, unusually severe and persistent symptoms of Swiss needle cast have been observed in Douglas-fir forest plantations in western Oregon, particularly near the town of Tillamook (Hansen and others 2000). Annual aerial surveys conducted since 1996 by the Oregon Department of Forestry have documented the disease on about 150 000 ha of forest land in the Oregon Coast Range (Figure 2). Unlike Boyce’s (1940) characterization of the pathogen on native Douglas-fir as being inconspicuous and harmless, the fungus is abundant, trees frequently are defoliated of all but current-year needles, and attached foliage is often severely chlorotic (Figure 1). Growth reductions of 20 to 50 percent

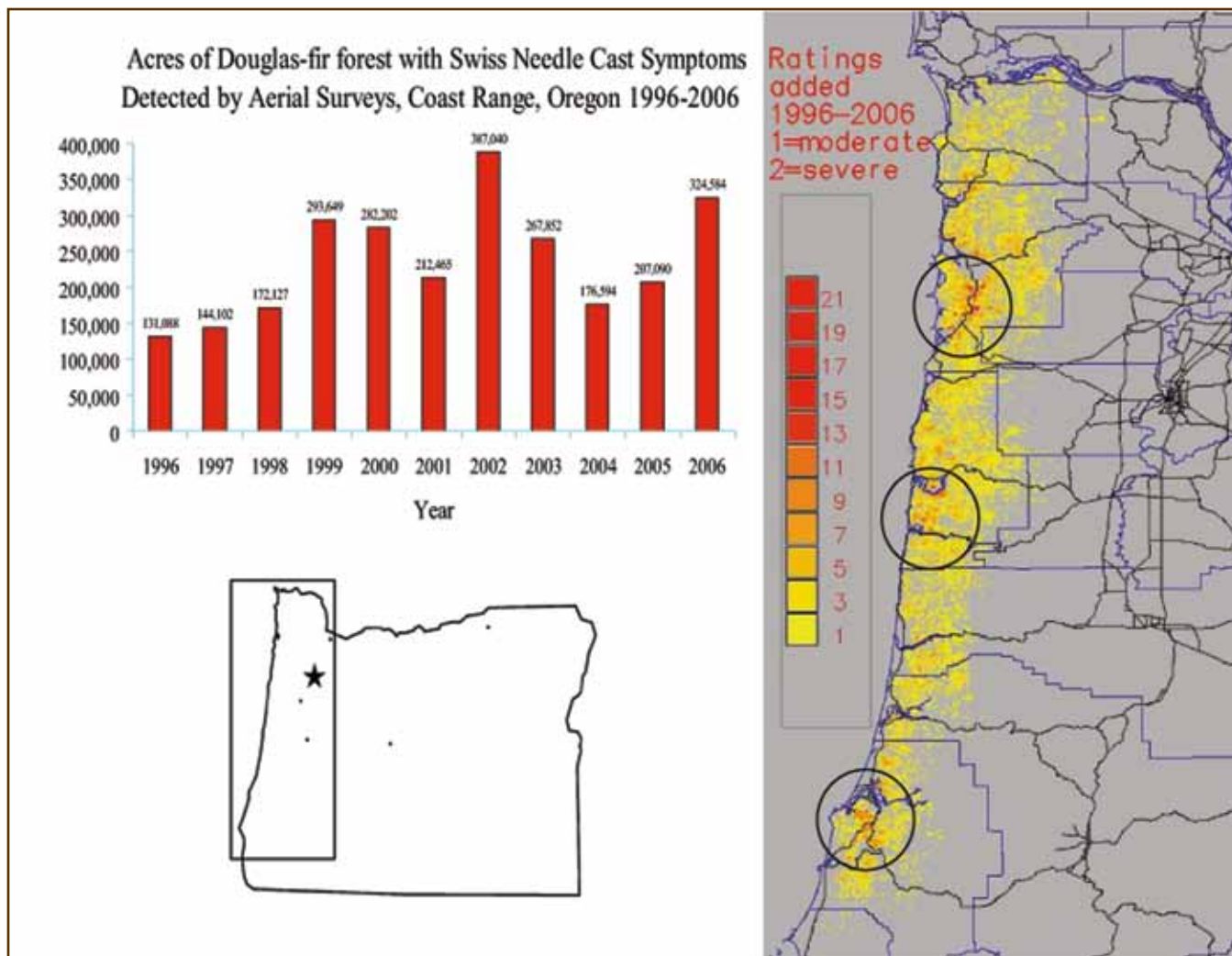


Figure 2—Annual variation in surveyed Swiss Needle Cast (SNC) area in western Oregon 1996–2006 (A) and combined disease severity survey scores for western Oregon (severity scores for all 11 years summed and superimposed on map B). Legend signifies 1 = disease moderate for single year to 22 = disease severe all 11 years of survey. Circled areas highlight regions where chronic SNC has been observed. Source data: Oregon Department of Forestry.

from Swiss needle cast have been measured in the affected area (Maguire and others 2002). The severity of the problem in Oregon has brought renewed interest in understanding the biology of the pathogen and epidemiology of Swiss needle cast disease. In particular, research has focused on understanding why an inconspicuous, insignificant native pathogen has become a significant forest health problem.

Aerial surveys for Swiss needle cast conducted by the Oregon Department of Forestry have classified patches of Swiss needle cast severity based on foliage discoloration, characterizing the discoloration as being severe or moderate. The affected area as determined by the aerial survey

lies along the entire length of the Oregon coast, extending inland about 40 km, with most symptoms occurring within 30 km of the coast. The crest of the Coast Range forms the approximate eastern edge of the affected area. The aerial survey covers about 1.2 million ha of coastal forest, with the symptomatic area comprising between 50 and 160 000 ha (Figure 2).

Disease severity (foliage retention, discoloration, crown sparseness, pathogen abundance) has also been monitored annually in permanent plots in the Oregon Coast Range (Hansen and others 2000) equipped with temperature and leaf wetness dataloggers. Foliage retention, and abundance

of *P. gaeumannii* ascocarps on 1- and 2-year-old needles have been monitored annually since 1996 in 9 to 12 Douglas-fir stands, initially aged 12 to 15 years. Study sites were selected to represent a range of elevations, distance from maritime influence, and disease severity. Within the area of severe disease, symptom severity is variable, but all Douglas-fir show some effects of the disease compared to healthy stands on the eastern slope of the Coast Range and in the Cascade Range. Normal needle retention in healthy coastal form Douglas-fir is about 4 years. Within the epidemic area, needle retention varies from about 1.5 to 2.6 years (Hansen and others 2000). Although symptom severity (e.g., foliage discoloration, needle retention) for all sites varies from year to year, relative disease severity (i.e., abundance of *P. gaeumannii* ascocarps on 1- and 2-year-old foliage) is fairly consistent. Disease tends to be more severe nearer the coast, at lower elevations, and on southern aspect slopes, gradually diminishing to the east (Hansen and others 2000, Manter and others 2003b, Rosso and Hansen 2003).

The Swiss needle cast epidemic area appears to correspond approximately to the Sitka spruce (*Picea sitchensis* (Bong.) Carr.) vegetation zone, a narrow strip of coastal forest characterized by elevations generally below 150 m, proximity to the ocean, a moderate climate, and a distinct forest type (Franklin and Dyrness 1973). Although Douglas-fir is considered the early seral dominant in the western hemlock vegetation zone, which borders the Sitka spruce zone to the east, its occurrence within the Sitka spruce zone is more sporadic. There Douglas-fir occurs mainly in mixtures with Sitka spruce and western hemlock (*Tsuga heterophylla* (Raf.) Sarg.), but normally not as pure stands as is typical of early postfire succession in the western hemlock zone (Franklin and Dyrness 1973).

The correspondence between the Sitka spruce zone and the main Swiss needle cast epidemic area suggests that environmental factors that influence the development of different plant associations in western Oregon may also influence the severity of Swiss needle cast. Irruption of the disease in the coastal zone may be owing changes in abundance of Douglas-fir in coastal forests owing to forest

management practices that favor the cultivation of Douglas-fir over other species (Hansen and others 2000). Swiss needle cast itself may be one such environmental determinant of the natural distribution of Douglas-fir in the western Coast Range, making it an inferior competitor where Swiss needle cast is severe (Hansen and Stone 2005).

But what factors affect the distribution and abundance of *P. gaeumannii*? It has long been suspected that local climate plays a key role in the pathogenicity of *P. gaeumannii*. Boyce (1940) suggested that seasonal patterns in local climate could differentially affect fungal growth and development, and this might explain the greater virulence of *P. gaeumannii* in Europe and the Eastern United States compared to the area where both *P. gaeumannii* and Douglas-fir are native. Within the Douglas-fir zone of the Pacific Northwest, there is also a relationship between disease severity and local climate. Hood (1982) found more *P. gaeumannii* in southern British Columbia and western Washington in coastal forests of Vancouver Island and the Olympic Peninsula, with lower levels in the rain shadow of eastern Vancouver Island and the interior, and attributed the difference mainly to precipitation patterns. More severe disease symptoms and greater fungal colonization are commonly observed on lower elevation sites near the coast, suggesting the possible involvement of maritime fog (Rosso and Hansen 2003).

Rosso and Hansen (2003) used a visual index of stand disease to investigate the relationship between several environmental variables and Swiss needle cast severity. The best model for predicting stand disease rating (a composite index of several factors) was a regression model that included July temperature, precipitation, and fog occurrence, which explained about 60 percent of the variation. Manter and others (2005) found a strong relationship between winter (Dec.–Feb.) mean daily temperature, spring (May–July) leaf wetness, and *P. gaeumannii* abundance.

Understanding the Disease

To better understand which environmental factors might be important regulators of *P. gaeumannii* abundance and how they affect the epidemiology of Swiss needle cast, it was essential to investigate the infection cycle of the pathogen

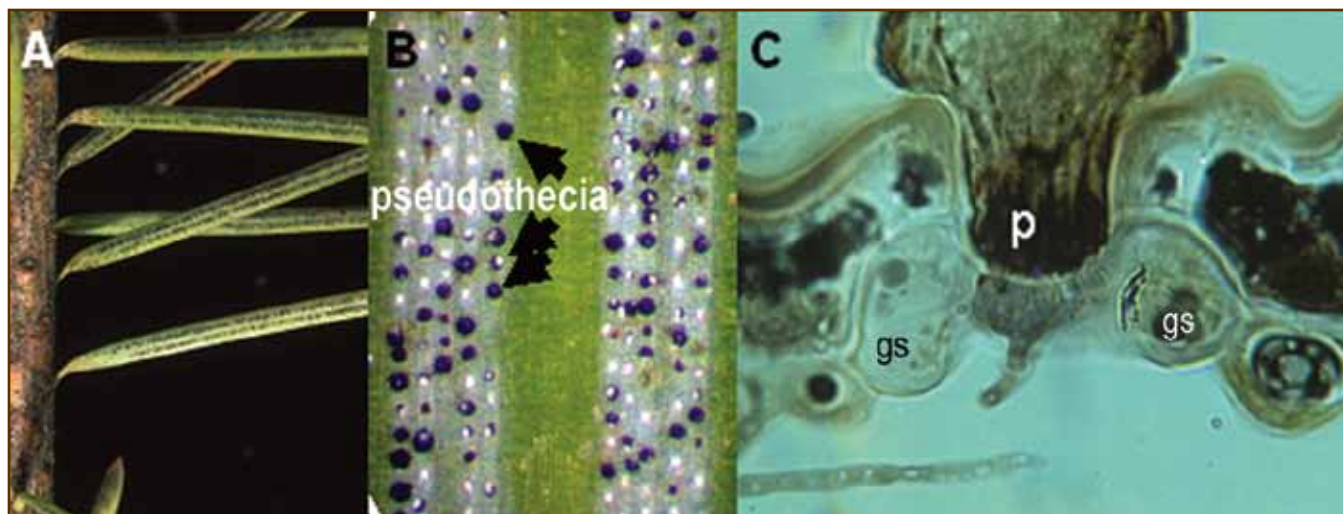


Figure 3—*Phaeocryptopus gaeumannii* on the underside of Douglas-fir needles with pseudothecia (fruiting bodies) emerging through stomata. A, B. Pseudothecia aligned along stomatal rows. C. Cross section through a stoma showing obstruction of the opening by the pseudothecium (p) between the guard cells (gc).

and its mechanism of pathogenicity. Ascospores of *P. gaeumannii* mature and are released during early May through late July, coinciding with bud break and shoot elongation of Douglas-fir. Ascospores are the only infective propagule; there is no conidial anamorph. Newly emerging foliage of Douglas-fir is most susceptible to infection. Infection of 1-year-old needles during the same period also occurs but is much less frequent. Germinating ascospores produce appressoria above stomata, and penetration pegs enter needles via stomata. Colonization within needles is exclusively intercellular; no intracellular hyphae or haustoria are produced. Internal colonization occurs gradually during the fall and winter. Ascocarp (pseudothecia) primordia begin to form in substomatal chambers at 4 to 9 months following infection. Concurrently with the formation of pseudothecial primordia, epiphytic hyphae emerge from the periphery of developing pseudothecia. These hyphae grow across the needle surface, form numerous anastomoses, and reenter the needle by producing appressoria above unoccupied stomata (Stone and others 2008). The importance of epiphytic hyphae in needle colonization and epidemiology of Swiss needle cast is not fully understood (Figure 3).

Internal colonization of needles continues as long as they remain attached, so numbers of ascocarps increase as needles age. Normally, fruiting bodies of the fungus are more abundant on needles aged 3 years or older and are

sparse or absent on younger foliage (Boyce 1940, Hood 1982). In recent years, however, trees having abundant fruiting bodies on current-year needles have been commonly observed in forest plantations along the Oregon coast, with older foliage being prematurely abscised due to the disease (Hansen and others 2000).

The ascocarp primordia completely occupy the substomatal space, thereby rendering the stoma nonfunctional. Occlusion of the stomata by pseudothecia of *P. gaeumannii* impedes gas exchange and regulation of transpiration, causing impaired photosynthetic activity and is considered the primary mechanism of pathogenicity (Manter and others 2000, 2003). Disruption of host cells by hyphal penetration has not been observed in infected needles examined in SEM and TEM preparations (Stone and others 2008), and attempts to find evidence of fungal phytotoxins to date have been inconclusive. Estimates of the effect of *P. gaeumannii* on CO₂ assimilation indicate that occlusion of 25 percent of stomata results in negative needle carbon budgets, i.e., respiration exceeds assimilation, on an annual basis (Figure 4) (Manter and others 2003a).

The abundance of pseudothecia is also highly correlated with needle abscission because of the effect on CO₂ assimilation. It has been suggested that foliage abscission occurs when needles switch from being carbon sources to carbon sinks (Cannell and Morgan 1990). The mechanism of

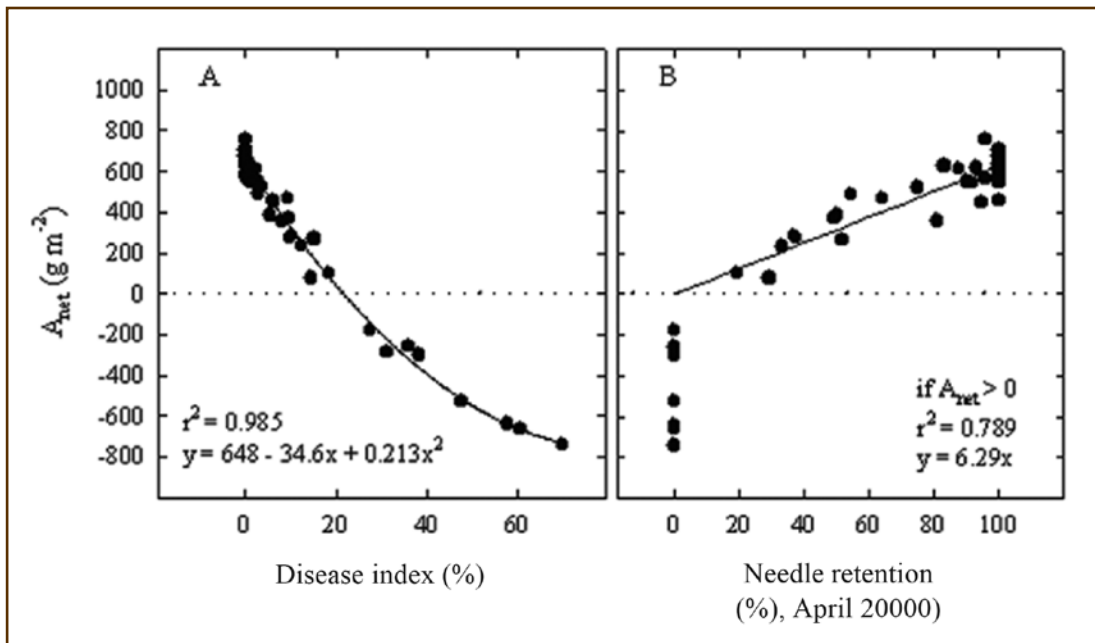


Figure 4—Relationship of net carbon dioxide uptake (A_{net}) per unit leaf area to [A] *Phaeocryptopus gaeumannii* pseudothecia abundance (disease index, percent) and [B] needle retention (percent). Taken from Manter and others (2003a).

pathogenicity of *P. gaeumannii*, therefore, can be accounted for by the physical blockage of the stomata and interference with photosynthetic gas exchange. The proportion of stomata occupied by pseudothecia on attached needles seldom exceeds 50 percent, suggesting that most needles are abscised before more than half the stomata are occluded by pseudothecia, regardless of needle age (Hansen and others 2000). Because the physiological effects of the disease (impaired CO₂ uptake and photosynthesis) are quantitatively related to the abundance of the pathogen (proportion of stomata occluded by ascomycetes), pathogen ascomycete abundance is a suitable response variable for assessing effects of climatic factors on disease.

Experimental Approaches to Understanding Climate-Disease Interactions

Because the most severe disease has been observed in sites within the low-elevation coastal fog zone (so called because of the frequency of summer maritime fog), the presence of free water on needle surfaces during the summer has been considered a possible factor affecting disease severity

(Hansen and others 2000, Rosso and Hansen 2003). Other investigators have noted a relationship between precipitation patterns and Swiss needle cast severity in the Pacific Northwest (Hood 1982, McDermott and Robinson 1989). Manter and others (2005) attempted to investigate the relative effects of individual climate factors on *P. gaeumannii* abundance experimentally. A factorial design was used to compare the effect of post-inoculation incubation conditions under two levels each of drip irrigation, shade, and intermittent mist on *P. gaeumannii* colonization. Seedling trees were exposed to inoculum in a diseased forest stand, then randomized and maintained under the different post-inoculation treatments. Abundance of *P. gaeumannii* ascomycetes on foliage was determined monthly. The different postinoculation conditions resulted in significant differences in *P. gaeumannii* development. Abundance of *P. gaeumannii* was negatively correlated with shade and mist—this unexpected outcome was interpreted as a result of indirect effects of shade and mist on temperature and not as a direct effect of shade or leaf wetness—and irrigation had no effect (Figure 5).

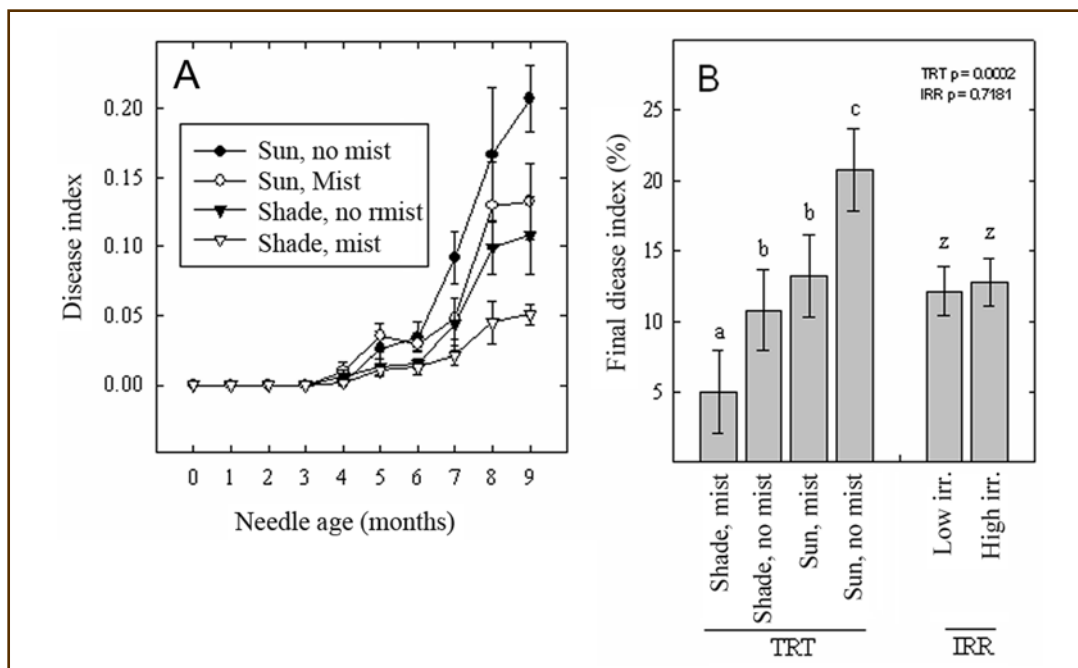


Figure 5—Final *Phaeocryptopus gaeumannii* disease index values for seedlings incubated under shade, mist, and irrigation treatments following inoculum exposure. Sun = 100 percent ambient, shade = 50 percent ambient, mist = three 2-hour overhead mist treatments per day, no mist = 0 overhead mist treatments per day, low irr. = 0.5 L day⁻¹ drip irrigation, and high irr. = 1.9 L day⁻¹ drip-irrigation. Bars are the arithmetic means and individual standard errors. Treatment effects were analyzed using a split-randomized complete block design, and bars with different letters are significant at $p < 0.05$. A. Monthly measurements of pseudothecia abundance by treatment. B. Final disease index by treatment. Taken from Manter and others (2005).

Modeling Swiss Needle Cast

The finding that small differences in temperature could affect rates of needle colonization and fungal development over the 11-month incubation period of *P. gaeumannii* prompted us to examine the effect of temperature in relation to infection data from field sites in western Oregon. Average daily temperature and cumulative leaf wetness hours were separated into 3-month groups, corresponding to major phases in the infection cycle, and subjected to stepwise regression against *P. gaeumannii* infection data to identify climate factors for use in a disease prediction model. Consistently strong correlations were found between winter (Dec.-Feb.) mean daily temperature and infection in both 1- and 2-year-old needles ($R^2 = 0.75$ to 0.92), and this was the only climate variable with R^2 values above 0.5. The best-fit climate model included winter mean daily temperature and cumulative spring leaf wetness hours, $R^2 = 0.78$ and 0.77 for 1- and 2-year-old needles, respectively. When this model was tested against infection data for different sites in

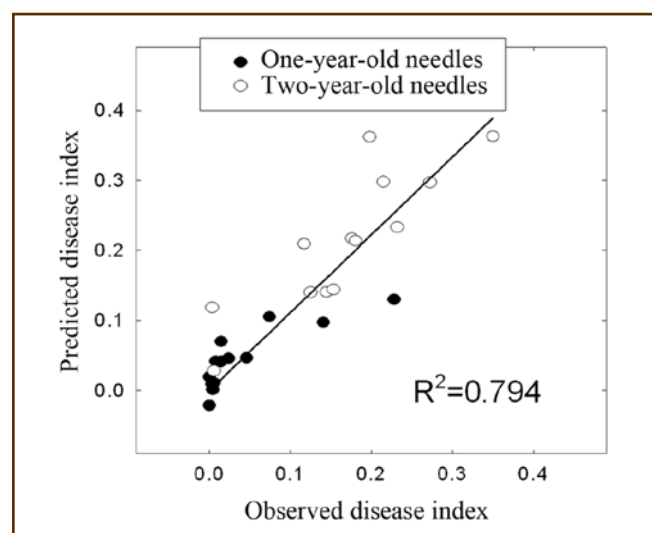


Figure 6—Best-fit model for predicting *Phaeocryptopus gaeumannii* abundance using only climate variables (winter temperature and spring leaf wetness), predicted vs. observed values. From Manter and others (2005).

different years, a significant 1:1 relationship was found ($R^2 = 0.79$, Figure 6).

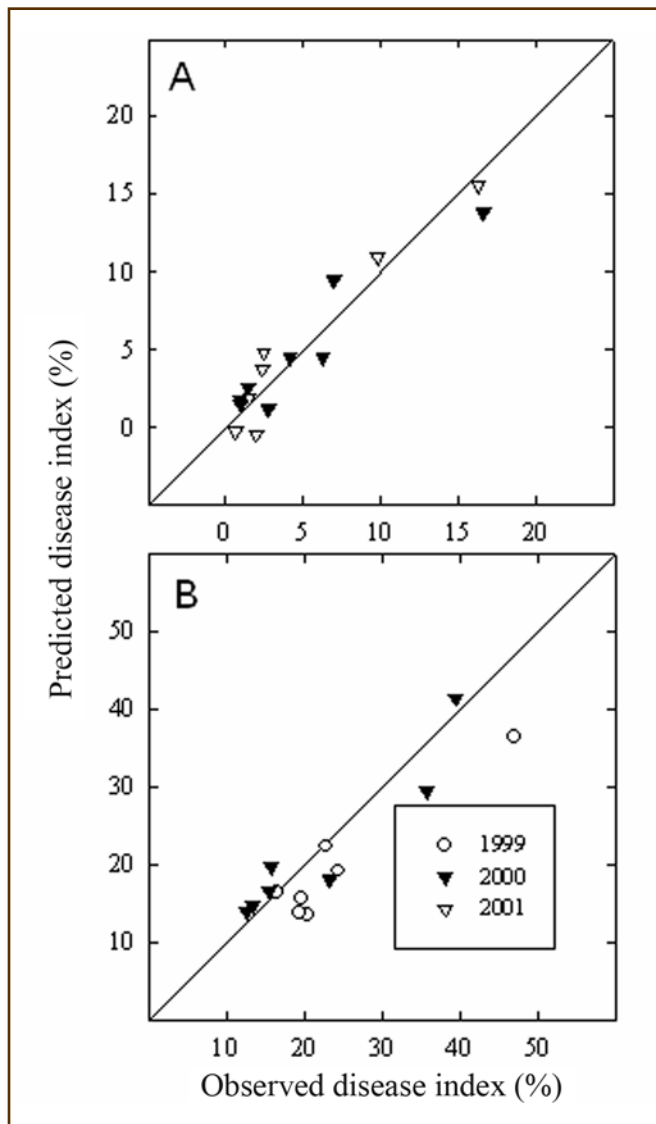


Figure 7—Plots of predicted vs. observed values for the best-fit model for predicting *Phaeocryptopus gaeumannii* abundance. Terms in the model were average daily winter temperature (Dec.-Feb.) and preceding year disease index for the corresponding needle cohort. From Manter and others (2005). Panel A, 1-year-old needles; panel B, 2-year-old needles.

A more general model was sought that could be used to predict geographic variation in Swiss needle cast severity with the aid of spatial climate models. However, because leaf wetness is not readily available in public meteorological databases or climate models, alternative models were tried. Winter temperature was the best single predictor of *P. gaeumannii* abundance, but its relationship with infection level varied by year. The abundance of *P. gaeumannii* and

severity of Swiss needle cast for a particular site in a given year are not independent of previous year disease severity for the site. Abundance of *P. gaeumannii* in 1-year-old needles is partly determined by the amount of inoculum present and the number of ascospores that initiate infection on a needle. Because ascospore infection of needles in their second year is negligible (Stone and others 2008, Hood and Kershaw 1975), abundance of *P. gaeumannii* on 2-year-old needles is partly determined by the degree of colonization in needles at the beginning of their second year. Therefore, two infection components, the amount of *P. gaeumannii* in 2-year-old and 1-year-old needles for the previous year, were added to the temperature model to predict infection in 1- and 2-year-old needles for the year of interest. Disease predictions generated by this model had a significant 1:1 relationship ($R^2 = 0.812$) when compared with observed values in the validation data set (Manter and others 2005), (Figure 7).

The temperature/infection model described above was then run over several iterations with temperature held constant over a range of values (2 to 10 °C). After five iterations, the infection level reaches a stable asymptote for any value of winter temperature. The maximum infection value is a function of winter temperature, regardless of initial infection level, and represents the point of equilibrium between colonization and inoculum production. A plot of the equilibrium infection level against temperature shows that between 3.77 and 8.90 °C, the range of average winter temperatures measured at coastal study sites, infection levels vary from zero to about 15 percent in 1-year-old needles and between 10 to about 40 percent in 2-year-old needles (Figure 8).

This model also was accurate for predicting disease levels for sites in the Western Coast Range ($R^2 = 0.85$) but was less accurate when infection data from sites from the east slope of the Coast Range and Willamette Valley were included ($R^2 = 0.70$). Observed infection levels for the inland sites were less than predicted by the equilibrium temperature model, and this is likely due to the comparative dryness of the inland sites. The model that included a term for spring leaf wetness was more accurate in predicting infection levels for the inland sites than the equilibrium

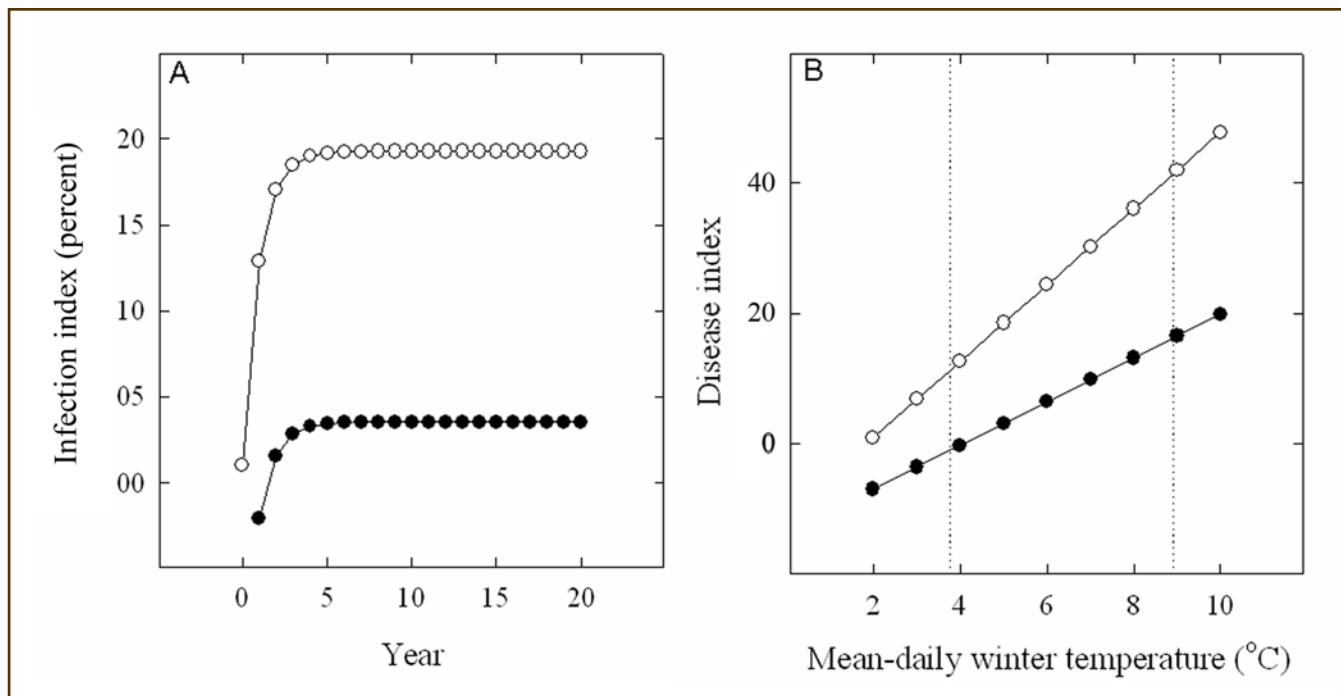


Figure 8—A. Simulation of *Phaeocryptopus gaeumannii* infection index over time as predicted by the combined winter temperature-infection model, with temperature held constant (5.13 °C) and initial infection index of 1 percent; B. Relationship between final equilibrium infection level and winter temperature for 1- (solid circles) and 2-year-old (open circles) needles.

temperature model, illustrating that the relative importance of different climate factors can vary spatially. The simple equilibrium temperature model appears suitable for predicting disease levels in the Western Coast Range, where spring surface moisture on foliage is probably rarely limiting, but additional parameters are necessary for disease prediction on more inland sites.

The equilibrium property of the model was then used to generate a disease prediction map for a portion of western Oregon based on the DAYMET climate model, a continuous surface at 1-km resolution base on 18-year (1980-1997) average temperatures (Figure 9). The model appears to correspond well with the distribution of disease symptoms as observed by aerial surveys and permanent monitoring plots. Research is in progress to compare disease distribution and severity predicted by different models with observed aerial survey data.

The sensitivity of *P. gaeumannii* to relatively small temperature differences helps to explain patterns of spatial variation in Swiss needle cast severity, and suggests that recent increases in Swiss needle cast severity

have been influenced by regional climate trends. Over the past century, average temperatures in the Pacific Northwest region have increased by about 0.8° C, with more warming occurring during winter months (Mote and others 2003). Average temperatures for the period January-March have increased by approximately 0.2 to 0.4 °C per decade since 1966 in coastal Oregon and Washington (USDC NOAA 2005). An increase in average winter temperature of 1 °C corresponds to an increase in infection index (proportion of stomata occluded) of 3 percent for 1-year-old needles and 6 percent for 2-year-old needles, based on the equilibrium temperature model. Spring precipitation has also increased on average by about 1.6 to 2.6 cm per decade since 1966 (USDC NOAA 2005). These climate trends suggest that over the past few decades, conditions have become more favorable for growth of *P. gaeumannii* and have contributed to increasing Swiss needle cast severity. Predictions for continued regional warming of about 0.4 °C per decade to 2050, together with increasing (+2 to 4 percent) spring precipitation (Mote and others 2003), suggest that conditions in the region will continue to be favorable for Swiss

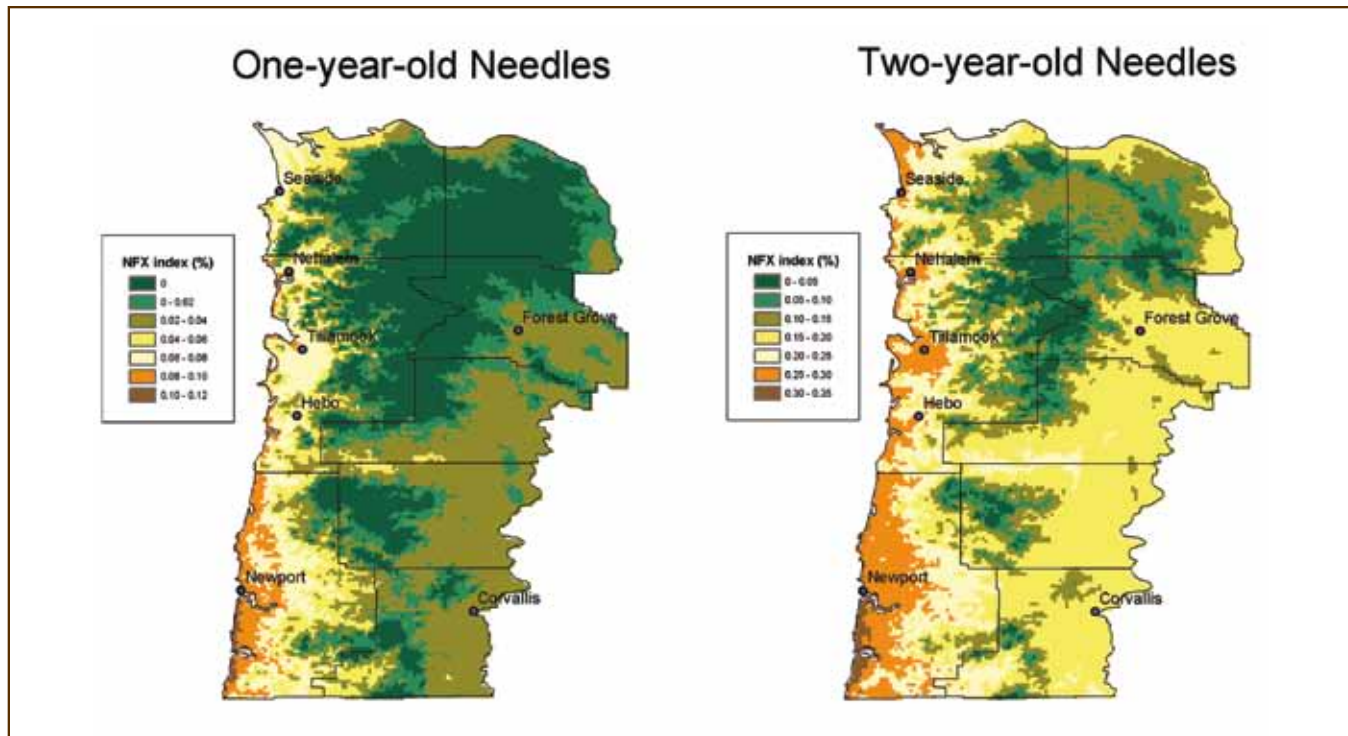


Figure 9—Disease prediction map for northwest Oregon as predicted by the equilibrium disease model. Infection indices were predicted from winter average daily temperatures from the DAYMET climate model. From Manter and others (2005).

needle cast development, and could result in expansion of the area affected by the disease beyond the Western Coast Range of Oregon. A goal of ongoing research is to develop an improved disease prediction model to investigate the interactions between climate and Swiss needle cast. An expanded disease prediction model will be designed to incorporate long-term climate trend forecasts to enable site-specific short- and long-term disease risk predictions, growth impact predictions, and incorporate climate change models to allow examination of disease development trends under different climate scenarios.

Conclusions

Improved understanding of the effects of climate factors on *P. gaumannii* abundance now helps to clarify the underlying causes for recently observed increases in Swiss needle cast in the Western Coast Range. Previous observations on the regional distribution of *P. gaumannii* in the Pacific Northwest have suggested a connection between *P. gaumannii* abundance and spring rainfall (Hood 1982). In

the Western Coast Range of Oregon, where spring precipitation is abundant, winter temperature has been found to be a highly reproducible predictor of the spatial variation in abundance of *P. gaumannii* and resulting Swiss needle cast severity, presumably because of its effect on fungal growth. Winter temperature alone is not a satisfactory predictor of *P. gaumannii* abundance regionwide or in areas where spring precipitation is not as abundant, such as the Willamette Valley, or Oregon Cascade Range, as shown by Manter and others (2005). The predictive disease model described here, therefore, is applicable for predicting spatial variation in *P. gaumannii* abundance and Swiss needle cast severity only for the western slope of the Oregon Coast Range. Within this area, there have been few historical reports of Swiss needle cast, and the disease has been considered an insignificant forest health issue. The natural distribution of Douglas-fir in the Western Coast Range has undoubtedly been influenced by *P. gaumannii* and Swiss needle cast, along with other disturbance agents. The effect of chronic, profuse *P. gaumannii* colonization of Douglas-fir foliage is to reduce growth rates of affected trees relative

to competing species, such as spruce and hemlock. Normally faster growing than western hemlock, Douglas-fir is an inferior competitor where Swiss needle cast disease pressure is high. In the coastal lowlands and interior valleys of the Western Coast Range, seasonal climatic conditions are the most favorable for *P. gaeumannii* growth and reproduction. In these areas, a distinct natural forest type has historically been dominated by western hemlock and Sitka spruce, with Douglas-fir occurring only sporadically. Douglas-fir gradually becomes more abundant in natural forests at higher elevations and further inland, as the Sitka spruce zone gradually merges into the western hemlock vegetation zone, where Douglas-fir is a successional dominant, and where climatic conditions are less favorable for *P. gaeumannii* growth. This leads to the conclusion that the Sitka spruce vegetation zone occurs as a consequence not only of favorable habitat for Sitka spruce and western hemlock, but also because of the inhibition of their main competitor, Douglas-fir, due to Swiss needle cast disease. This scenario also suggests that the severity of Swiss needle cast in the region may be the result of recent forest management in the Western Coast Range, where Douglas-fir has been strongly favored in forest plantations because of its greater economic value, increasing the abundance of the host species in the area most favorable for growth and reproduction of the pathogen. As noted above, however, recent climate trends also are likely to have contributed to current Swiss needle cast severity. Furthermore, forecasts of future climate trends for the Pacific Northwest suggest a probable expansion of the area affected by severe Swiss needle cast beyond the Western Coast Range as winter temperatures and spring precipitation continue to increase, resulting in greater disease pressure on Douglas-fir stands further inland.

Acknowledgments

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Analyzing Risks to Protected Areas Using the Human Modification Framework: A Colorado Case Study

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Abstract

A framework that organizes natural and protected areas is often used to help understand the potential risks to natural areas and aspects of their ecological and human dimensions. The spatial (or landscape) context of these dynamics is also a critical, but, rarely considered, factor. Common classification systems include the U.S. Geological (USGS) Gap Analysis Program (GAP) stewardship coding scheme, the International Union for the Conservation of Nature (IUCN) Protected Area Management Categories, and the American Planning Association (APA) Land-Based Classification Standards. The GAP and IUCN frameworks are coarse classifications (four to eight categories), whereas the APA focuses primarily on private land uses. To address these limitations, we develop here more refined implementation methods based on the human modification framework, which conceptually is rooted in characterizing the degree to which natural processes are free or controlled, and the degree to which landscape patterns are natural or artificial. To provide useful and tighter coupling of specific threats and spatial data surrogates, we refine the conceptual basis by identifying three primary types of human activities that cause modification of natural systems and patterns. These are land uses categorized as urban/built-up, recreation, and production/extraction. We detail specific metrics and common data used as surrogates that can provide a stronger basis for characterizing the degree of human modification.

We illustrate our methods by presenting analysis results for a Colorado case study.

Keywords: Human modification, protected areas, threat assessment, landscape pattern, land use.

Introduction

Overview of Conceptual Frameworks

A number of general wilderness frameworks have been established to provide a conceptual basis to understand protected areas. One of the earliest was provided by Brown and others (1978) who developed the Recreational Opportunity Spectrum (ROS), an approach that was based on characterizing resource, social, and managerial conditions for parks and recreation areas. The ROS has less well-developed guidance for characterizing threats and conservation on working and private lands. In Australia, Lesslie and Taylor (1985) developed the wilderness continuum concept that provided the basis for developing an inventory of high-quality wilderness areas. More recently, Cordell and others (2005) provided a concise review of the two most widely accepted attributes of wildlands: naturalness and wildness.

A number of efforts have built on these conceptual frameworks to provide detailed characterizations of wilderness through fine-grained mapping. Lesslie and others (1988) provided an early mapping technique based on overlaying distance classes from settlement and road infrastructure. Kliskey (1998) developed a method to better link perception of wilderness to the ROS in a spatially explicit mapping framework. Aplet and others (2000) developed a nationwide map of wilderness by distinguishing managed from natural landscapes and mapping a multidimensional index at 1-km resolution. Building on earlier work in Australia and the United States, Sanderson and others (2002) mapped the human footprint for the entire globe at 1-km resolution.

More recently, scientists have been moving from employing general indicators of wilderness (or degree of human modification) to recognizing specific indirect and direct threats (e.g., to wilderness ecosystems by Cole and Landres 1996) and the relative degree to which different

types of protected areas function to protect biodiversity. A common approach to categorizing or organizing different types of protections or threats has been to define categories based on whether land is managed for permanent biodiversity maintenance through some legal or institutional mechanism, or both. Both the World Conservation Union (IUCN) and the U.S. Geological Survey's (USGS) Gap Analysis Program (GAP) use this approach, though their protection categories differ somewhat (Davey 1998, Scott and others 1993). For example, the IUCN contains eight categories (IUCN 1994): I—scientific reserve/strict nature reserve; II—national park; III—natural monument/natural landmark; IV—managed nature reserve/wildlife sanctuary; V—protected landscapes; VI—resource reserve; VII—natural biotic area/anthropological reserve; and VIII—multiple-use management area/managed resource area.

The USGS GAP program found these too vague for their uses, and so they devised a stewardship classification with four status classes (Csuti and Crist 2000): (1) permanent protection from conversion of natural land cover, with natural disturbance events allowed; (2) permanent protection and some suppression of natural disturbance; (3) some extractive uses permitted; and (4) no protection from conversion of natural land cover.

The Land-Based Classification Standards (APA 2003) provide a multidimensional land use classification model based on five characteristics: activities (land use based on observable characteristics), functions (economic purpose), building types (structures), site development character (physical characteristics such as land cover types), and ownership constraints (private vs. public, or other legal devices to constrain use). They are richly developed for private land but are less directly applicable to public protected or conserved areas.

The IUCN categories have been revised substantially in the Conservation Measures Partnership, a collaborative effort mostly among nongovernmental organizations that includes The Nature Conservancy, World Wildlife Fund, and Conservation International, in addition to the IUCN. They recognize 11 classes of direct threats (IUCN-CMP 2006): residential and commercial development; agriculture and aquaculture; energy production and mining;

transportation and service corridors; biological resource use; human intrusions and disturbance (e.g., recreation, war, etc.); natural system modifications (e.g., fire suppression); invasive species and genes; pollution; geologic events; and climate change and severe weather.

Several alternative approaches have recently emerged. For example, the human footprint (Sanderson and others 2002) represents, in a general way, the total footprint of all the human population's influence as a continuum stretched across the land surface. Implicit in nearly all these approaches is that if an area is of high value, and it is unprotected, then it is vulnerable. However, incorporation of threat directly into these frameworks is underdeveloped, and these approaches have been critiqued because they neglect a means to assess the relative imminence of the threat or urgency of conservation action need—Are some gaps more important than others, and, if so, which ones (Margules and Pressey 2000, Theobald 2003)?

Objectives

We find four main limitations of these existing frameworks. First, they are often based on ad hoc categories or classes, with little or minimal attention to establishing explicit measures of threat. Second, they are based on discrete categories or classes, and so cannot resolve small differences. Also, because they are based on class (or nominal) data, they cannot be easily integrated at different spatial scales to examine landscape context. Third, the surrogate spatial variables that are used as a basis for classifying are generally too coarse grained to capture fine-grain patterns, ultimately limiting detailed characterization. Finally, and, especially, most approaches examine the in situ landscape content of a park or protected area (PPA) using levels of threat or protection. Typically, an individual PPA receives a single score, regardless of size or the types of adjacent land uses. For example, although national parks often have an intensely used, built-up portion (e.g., Yosemite Valley in Yosemite National Park) directly adjacent with designated wilderness, typical protocols and methods result in only a single value being applied to the entire PPA. In addition to content, we believe that the landscape context of a PPA is important as well. This aligns analytical methods with

Table 1—Example land uses along human modification gradient for each factor. Each row provides an example of a possible human use along the “wild” (0.0) to human-dominated (1.0) gradient for each of the three factors (columns)

Value	Convert (urban/built-up)	Extract (production/extraction)	Visit (recreation/work)
0.0	Wilderness	Wilderness	Municipal watershed (no access)
0.1	Wilderness with trails	Gathering	Private preserve (guided access)
0.2	Wilderness w/trails, bridges	Public grazing allotments	Wilderness area
0.3	Multiuse forest	-	Backcountry forest
0.4	Rural housing density	Timber selective cut	Multiuse forest
0.5	Campground	Hay meadow	Highuse trail/destination
0.6	Exurban housing density	Timber clearcut	Motorized recreation
0.7	Visitor center	Dryland crop	Cross-country ski resort
0.8	Suburban housing density	Oil/gas well	Downhill ski resort
0.9	Urban residential	Mine	Urban park
1.0	Commercial/industrial	Irrigated cropland	Sports stadium/complex

the growing concerns of land managers over these external threats. For example, two of the main threats to the National Forest System (loss of open space due to housing development and invasive species) are external (Bosworth 2004), and changes in surrounding land use are a leading threat to resources in United States national parks (GAO 1994).

We aim to address these limitations by building on the human modification framework (HMF) (Theobald 2004). The HMF is based on two dimensions: the degree to which natural processes are free or controlled, and the degree to which landscape patterns are natural or artificial. To provide useful and tighter coupling of specific threats and spatial data surrogates, we refine the conceptual framework by identifying three primary types of human activities that cause human modification of natural systems and patterns. These are land uses categorized as urban/built-up, recreation, and production/extraction. We detail specific metrics and common data used as surrogates that can be used to implement each of these factors to provide a stronger basis for characterizing the degree of human modification. Our overall goal in this paper is to develop a general yet simple method to characterize threats associated with human land use activities for managing landscapes and to provide metrics and more directly capture threats in a detailed, spatially explicit manner. Our objectives are to describe factors that can be used to refine the HMF, identify common surrogates of these factors that can be mapped at relatively fine grain, conduct a spatially explicit analysis of

the landscape context, and illustrate these methods using a case study example for the State of Colorado.

Methods

Human Modification Framework

Here we further refine the HMF to emphasize three primary factors that can be used to characterize land use based on what human activities occur at a given location. Principally, humans go to an area to produce or remove natural resources (production/extraction), visit but do not extract significant resources (recreation/tourism), or concentrate or intensify resources by reconfiguring and constructing buildings and other infrastructure (urban/built-up). Each of these three factors begins with a value of 0.0 denoting no human influence (i.e., a wild location), and 1.0 representing an area that is strongly influenced (developed) by human activities. We seek to fully characterize the spatial heterogeneity of threats by characterizing land uses at a relatively fine grain ($<1 \text{ km}^2$ cells or minimum mapping unit), rather than attributing an average or overall value across a broader spatial extent (e.g., a GAP stewardship value for an entire national park). We then can overlay our threat maps with a map of protected areas to assess gaps and opportunities.

For each factor, we explicitly place values for different levels of threat along a 0 to 1.0 axis (Table 1). Although these values are either assigned arbitrarily, based on expert opinion, mapped through surrogate variables, or quantified

Table 2—Possible surrogate spatial data for urban/built-up factor

Data	Source	Scale (extent/grain)	Notes
Housing density	Census Bureau; SERGoM v2 (Theobald 2005)	Nationwide/block Nationwide/1 ha	Better indicator of landscape change than population density
Population density	Census Bureau	Nationwide/block	Commonly used
Road density	Census TIGER; GDT; NavTec	Nationwide/1:100K	Need to identify the radius or size of the moving window in computing density. Major road types are often distinguished and weighted when computing road density. Need to extend to characterize road use (i.e., traffic volume) as well. Poor representation of rural, forest, park areas
Structures (houses, buildings, bridges)	Parcel-level National forest/park unit	County by county	Critical data for fine-grained analysis, but very limited availability etc.
Trail density	National forest/PPA	PPA/1:24K, GPS	Need to specify radius of moving window when computing density (~ 500 m) Can major differences in trail types, such as modes of use, use level be distinguished? Standardized, consistent data on trails are not available
Campground density	USFS	PPA/1:24K	Location of campgrounds, number of units/sites, and type of use
Dam density	National Inventory of Dams 1: 1,000,000	Nationwide/	For major dams (> 4 m in height); spatial precision issues; incomplete attributes such as year built, storage volume, etc.
Land cover	National Land Cover Data set (NLCD)	Nationwide/900 m ²	Urban/built-up classes (21, 22, 23)—update with NLCD 2001 when available

Each row provides an example of a possible spatial datum that is commonly used as a surrogate for this factor.

PPA = park or protected area.

using some combination of these, it is critical that we make our assumptions quantitatively explicit, at the very least to communicate how threats may affect a location and their relative importance to other threats. Also note that this approach is agnostic to land ownership—that is, land use activities for each of these factors can occur on both private and public lands (though some types may be more dominated than others). For example, although built-up is often associated with urban areas, some localized areas within protected areas (e.g., a national park visitor center) can be

highly built-up as well. Table 2 proceeds a short, preliminary listing of surrogate maps that are commonly used to estimate these activities for the three factors—yet much work needs to be done to fill these critical data gaps. Again, we emphasize that often these values are arbitrary, but providing explicit values is a useful and important exercise.

Urban/Built-Up—

This factor characterizes the intensity to which humans occupy a given location. Human activities often concentrate

or intensify resources by reconfiguring resources or constructing buildings and other infrastructure such as roads, dams, bridges, etc. Table 2 provides a list of useful data layers that can be used to measure the degree of the urban/built-up factor. Common surrogate spatial variables include population density and housing density. We prefer housing density because it more directly characterizes landscape and ecological changes on the ground, whereas population density is attributed typically only for primary residences. Data on residential land cover (e.g., from National Land Cover Data (NLCD)) are often used to represent built-up areas, though additional data from the Census Bureau (or parcel level) are needed to characterize lower density development beyond the urban fringe. Commercial, industrial, and transportation land uses also are associated with high levels of human modification (e.g., high levels of impervious surface), population density (less direct), etc. Data on transportation infrastructure, such as road density (e.g., Theobald 2003), are also strong measures of land uses associated with built-up areas. We believe that housing density and road density provide complementary information—high density of roads can occur not only in urban areas but also in remote rural areas (e.g., forest logging roads). Recently, researchers are beginning to distinguish major road types (e.g., interstate vs. secondary roads) when computing road density, though more work to characterize road use rather than the existence of roads is needed. Improved data sets are needed to better identify the location and intensity of structures in rural areas, particularly bridges, culverts, roadway fences, utility corridors (e.g., power lines, pipelines, etc.).

Production/Extraction—

This factor characterizes the intensity of human activities associated with the amount and intensity of extracting or removing resources. Agricultural production, especially cropland, but, also, grazing and mining and timber activities are common forms of this activity type (Table 3). It is common to use land cover data to map production land uses that have caused land cover conversion, such as agricultural croplands and orchards. The intense changes in pattern and the high degree of modification of ecological processes mean that this is a critical component to map. A variety of

also additional activities also contribute to human modification for which data are less commonly available. This would include spatially explicit information about grazing (intensity and animal density) through permits on public lands and stocking rates on private lands. Extraction of natural resources in the form of quarries and mines that cause significant land surface modification can be readily identified. However, activities with extensive but more diffuse disturbance (e.g., oil and gas wells or selective harvesting/thinning of forest products) are more difficult to identify and generate metrics from typical satellite imagery, though increasingly specialized databases (e.g., well locations) are being developed.

Recreation and Tourism—

This factor characterizes the intensity of human activities associated with use or visitation of an area. This is best represented by recreation and tourism activities, but can also include work-related activities. It includes the amount and type (mode) of recreational use (i.e., pedestrian only, passive, motorized, etc.). There are a large number of important data gaps here—particularly whether areas are publicly accessible, motorized vs. nonmotorized use, visitor use levels, etc. (Table 4).

Spatially explicit data on this aspect of human modification are particularly underdeveloped. Although general visitor use levels may be available for whole parks or national forests, more detailed data to map the visitation levels (and the timing) are particularly poor. Use has been estimated based on trail density (Schumacher and others 2000), though this suffers from the same limitation as road density methods—namely the assumption that areas close to trail heads receive the same amount of use as those more distant. Methods to estimate visitor use via spatial models of accessibility are increasingly common (Geertman and van Eck 1995).

Assessing Landscape Context—

To characterize risks within the landscape, we assign each factor a value along a 0 to 1.0 axis based on explicit criteria (Table 5). We use a fuzzy average approach to combine our individual surrogate layers (in GIS) into a composite value layer for each factor. This is calculated simply as

Table 3—Possible surrogate spatial data for production/extraction factor

Data	Source	Scale (extent/grain)	Notes
Grazing	U.S. Forest service; BLM; private	National/1:100K	Characterize by animal unit months, permit locations—but these data are mostly unavailable in electronic, spatial format
Oil & gas well density	Colorado Oil & Gas Conservation Commission	Statewide/1:100K	Point locations of wells, need to define “footprint”
Cropland, grazing	USGS National Land Cover Data set	National/0.09 ha	Cropland (NLCD classes 81-85) Grazing estimated from grassland, shrubland, and forested land
Mining (gravel, hard-rock, etc.)	Colorado Division of Minerals and Geology	Statewide/1:100k	-
Groundwater wells	State water engineer	Statewide/1:100K	Poor availability of data on recharge rate, spatial location often tied to center of Public Land Survey Section

Each row provides an example of a possible spatial datum that is commonly used as a surrogate for this factor.

Table 4—Possible surrogate spatial data for recreation/visitation factor

Data	Source	Scale (extent/grain)	Notes
Parks and protected areas	World Protected Areas Database Protected Areas Database; COMaP	Nationwide/block Nationwide/1 ha	Inconsistent; poor data for State, local (city and county), private (reserves, conservation easements)
Visitor use levels	National Forest Service survey?	Nationwide/?	Very coarse (State/forest) data, need data to be tied to trailheads
Recreational facilities (golf courses, ski areas, etc.)	Census TIGER; GDT; NavTec	Nationwide/1:100K	Use levels (visitor days per year) would be useful
Accessibility from urban populations	Cost distance weighting	Nationwide/1:100K	-

Each row provides an example of a possible spatial datum that is commonly used as a surrogate for this factor.

the maximum value (at every cell) of any single input. These ratio values are then spatially averaged over different neighborhood sizes and shapes to establish a measure of landscape context. A challenging part of developing estimates along the three factors is to be explicit about the spatial characterization. That is, for example, when a value for wilderness is prescribed, does this apply to an entire wilderness area (as a political designation), or within some defined neighborhood (e.g., within a 1-km circular window)? The spatial grain needs to be considered before one

provides or computes estimates of any given factor. Here we summarize the spatial context using circular moving windows with radii of 1 and 5 km. These distance values are arbitrary, but attempt to capture the relatively local and midscale spatial context pattern. Again, our goal here is to provide an illustration of the broader method, and users will likely want to employ different values that reflect the species, process, or patterning that they are concerned about, which likely varies across ecoregions.

Table 5—Surrogate variables used for human modification framework factors for Colorado

Surrogate variable	Factor		
	Convert urban/built-up	Extract Production/extraction	Visit Recreation/work
Land cover (NLCD) ¹	Comm/ind (23) = 1.0 High dens. res. (22) = 0.9 Low dens. res. (21) = 0.8 Barren (33) = 0.6 Grass/parks (85) = 0.5	Orchards (61) = 0.6 Pasture (81) = 0.5 Row crops (82) = 1.0 Nonrow crops (83) = 0.7 Other ag (84) = 0.8 Quarries/mines (32) = 0.9	-
2000 housing density ²	Urban (13-15) = 0.9 Suburban (12) = 0.8 Exurban (7-11) = 0.6 Rural (4-6) = 0.4	-	-
Roadedness ³	Interstates (CFC A1x) = 1.0, 60-m buffer Highways (CFCC A2x) = 1.0, 30-m buffer Secondary (CFCC A3x) = 1.0, no buffer Local (CFCC 4x) = 0.5, no buffer 4WD (CFCC 5x) = 0.2, no buffer	-	-
Accessibility (only public lands) ⁴	-	-	<0.5 hrs = 1.0 0.5-1 hr = 0.8 1-2 hrs = 0.5 2-4 hrs = 0.3 4-8 hrs = 0.1 >8 hrs = 0.0
Oil & gas wells ⁵	-	W/in 60-m radius = 1.0, w/in 100 m = 0.7, w/in 200 m = 0.3, w/in 300 m = 0.1	-
Mining operations ⁶	-	Coal exploration and unknown = 0.3 m, underground = 0.5, heap leach = 0.8, in-situ = 0.8, surface and underground = 0.9	-
COMaP (NF management types)	-	-	Ski resorts = 0.8 Motorized = 0.6 Multiuse = 0.4 Backcountry = 0.3 Wilderness = 0.2

¹ USGS National Land Cover Dataset, 0.09-ha resolution, ~1992-94.

² U.S. Census Bureau 2000 <http://www.census.gov>, Theobald 2005 www.asri.com/data, 1-ha resolution.

³ Created from ESRI Streetmap 2006 data.

⁴ Created from travel time along major roads, weighted by population size to nearest urban area, for urban areas with population > 100k and ≤ 100k, data from ESRI Streetmap 2006.

⁵ Colorado Oil and Gas Conservation Commission, Well and facility shapefiles, downloaded 16 July 2006 from <http://www.oil-gas.state.co.us>.

⁶ Mining data converted points to circles with area equal to permitted area ($\sqrt{[\text{Permit_acr}] * 4048 / 3.1415}$).

Note: CFCC = census feature class code; 4WD = four-wheel drive.

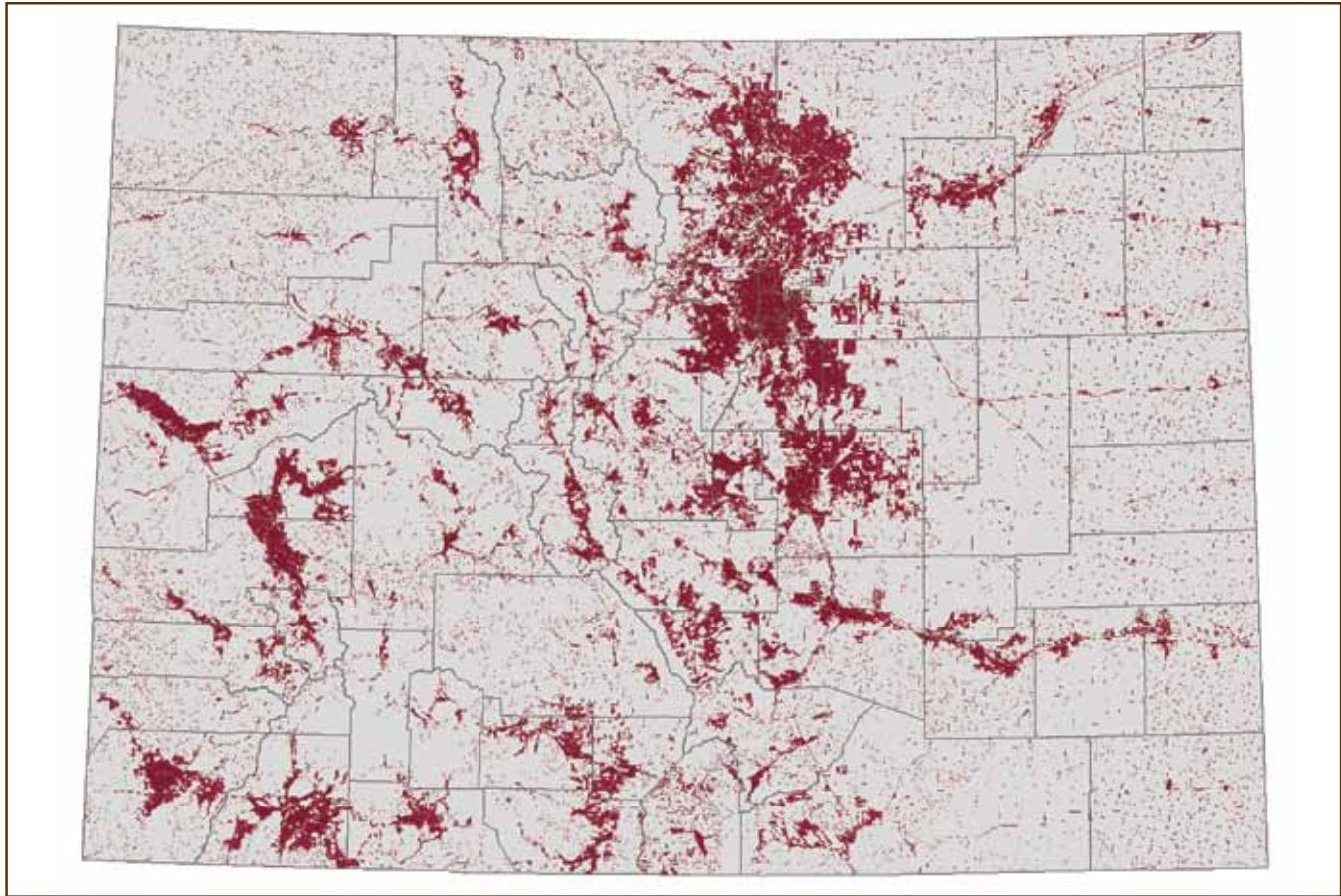


Figure 1—This composite map for urban/built-up areas in Colorado shows the degree to which locations have been converted, or built-up. This is a composite of commercial, industrial, and residential development (from National Land Cover Data Set); a “roadedness” layer that depicts the footprint of interstates, highways, and roads; and housing density (including exurban and rural densities) from SERGoM v2 (Theobald 2005). Darker shades show a higher level of built-up, grey shows absence, and grey lines show county boundaries. Note that the Front Range (Fort Collins, Denver, Colorado Springs) is clearly visible, as are many of the mountain valleys.

Results

The composite map of the degree to which an area is urban/built-up shows a relatively familiar pattern of urban areas, particularly the cities along the Colorado Front Range (Figure 1). Including information on slightly lower density of houses provides a fairly large expansion of locations that are developed beyond those locations identified in NLCD, particularly along mountain valleys.

The composite map of production and extraction-oriented land use provides a fairly different picture of Colorado (Figure 2). Extensive lands have been converted to cropland agriculture in the eastern plains and San Luis Valley. Another factor included is the extensive distribution of

oil and gas wells, particularly in the northern Front Range area and western slope.

Perhaps the most unique contribution of this map is to portray the distribution of the extent and types of recreational uses found in Colorado (Figure 3). Compared with the typical map of public land ownership, portraying national forests, grasslands, and parks as being largely the same, this map clearly differentiates the heavy influence that accessibility to large populations (like the Front Range) can have. Accessibility here is computed as the one-way travel time (along roads) from urban areas. Also differentiated are the types of land management activities on public lands, distinguishing the heavily-recreated resort areas such as Aspen, Vail, and Breckenridge. This map also reflects

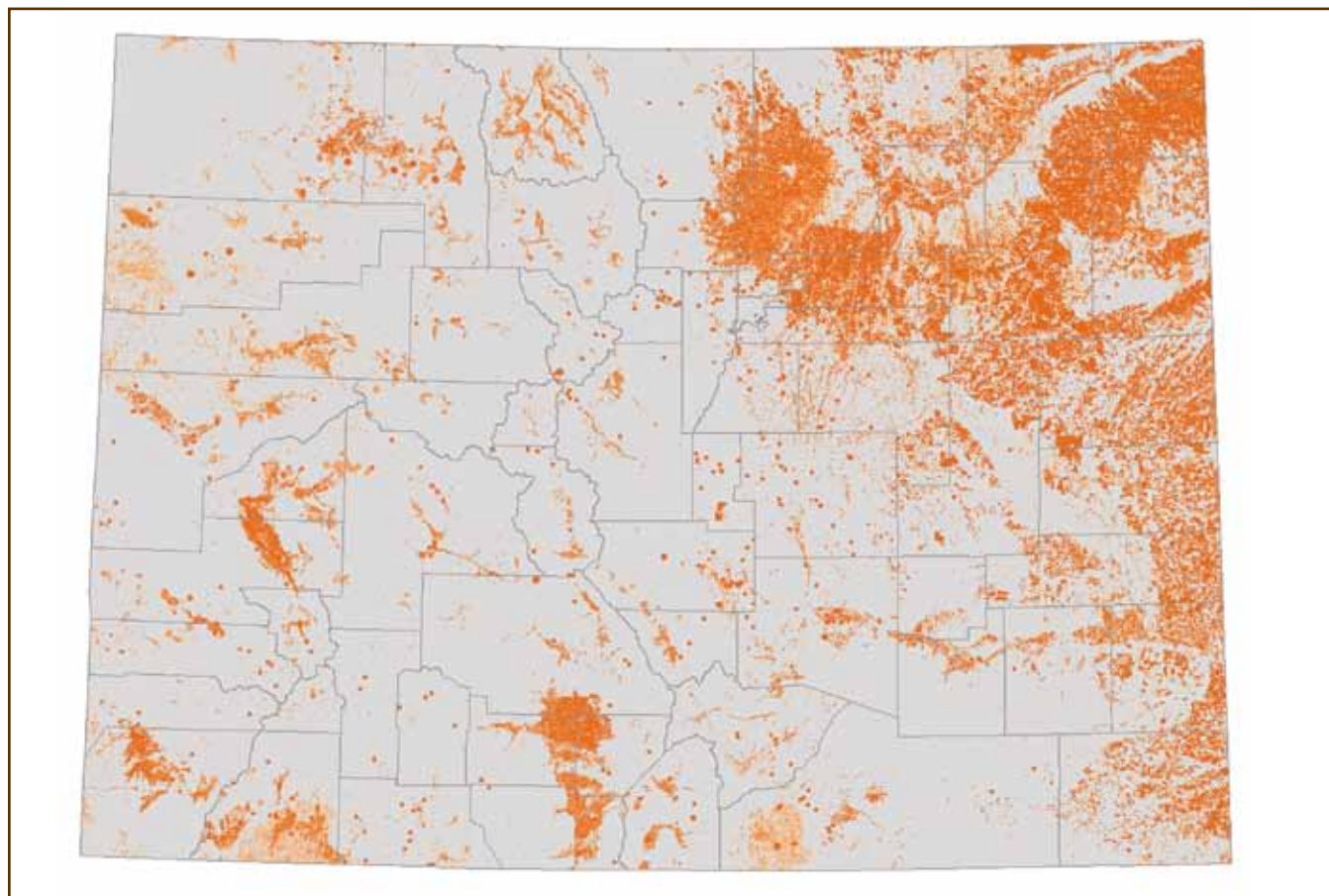


Figure 2—This map of Colorado shows the degree to which locations have some production or extraction-oriented land use. This is a composite of agricultural lands (from National Land Cover Data Set); mining activities; and oil and gas wells. Darker shades of orange show a higher level of activity, grey shows absence, and grey lines show county boundaries. Note that the productive agricultural lands of Weld and Adams County are clearly visible, as are the northeastern plains counties, whereas the San Luis Valley is visible in the south-central part of Colorado.

locations that are dominated by motorized vs. non-motorized recreation (future improvements should incorporate specific travel management plans).

The degree of human modification values vary across the State, with fine-grained differences particularly noticeable in riparian zones (Figure 4). The average HMF value is 0.392 (SD = 0.231). The range and distribution of HMF values can also be summarized by various ecological system types, such as lower montane, mixed, and upper montane zone (Table 6). Our results show that the midmontane zone has the largest average HMF score (0.428), indicating the highest degree of human activities that may modify or affect ecological systems in this zone. The upper montane zone (0.329) is the lowest of the conifer-dominated zones

(lower montane = 0.405), but aspen forests have the overall lowest degree of human modification (0.313). In addition to summarizing by counties, different analytical units such as watersheds, ecoregions, ecosystems, etc., can be supported. It is important to note that although one can summarize values over fairly large areas, as we have done by major ecological system type, the fine-grained data will allow differentiation of more localized areas that are particularly impacted or have particularly low HMF values.

Commonly, population density is used as a surrogate indicator of the degree of impact on ecological systems. Although we found a loose relationship between population density and HMF score (Figure 5), there is a fair amount of variation as well, showing the additional information that

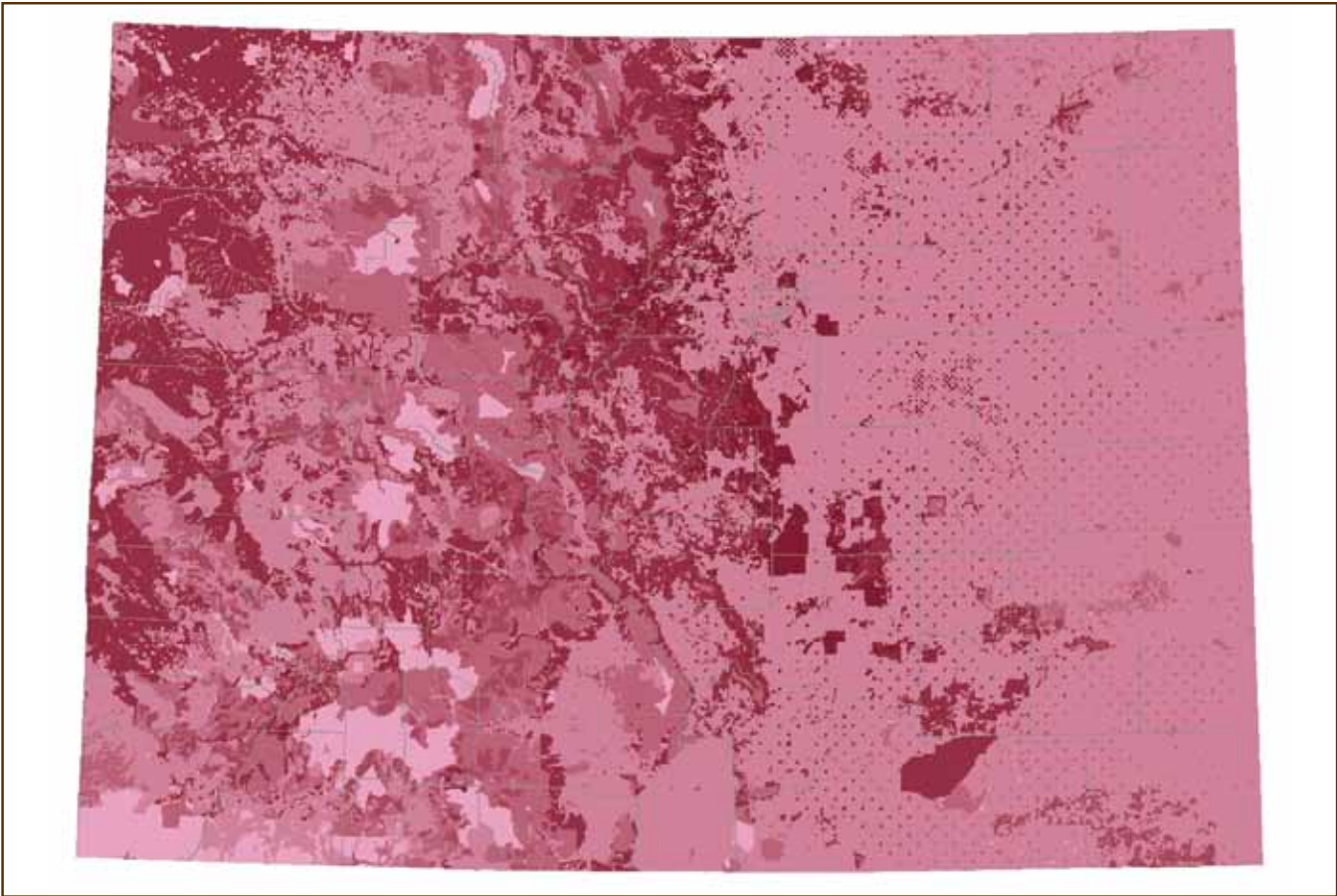


Figure 3—This composite map of Colorado for recreation/visitation shows the degree to which locations are likely to have recreation or tourism-oriented land use. This is a composite of an accessibility map that depicts the one-way travel time (along roads) from urban areas, and the type of public recreation and visitation targeted for different land management categories (especially for national parks and forests—data from the Colorado Ownership Management & Protection project). Darker shades of maroon show a higher level of activity and grey lines show county boundaries. Note that the public lands directly adjacent (mostly west of) the Front Range are highly affected, as are downhill ski resorts, and motorized recreation areas.

Table 6—Summary statistics for forested ecosystems in Colorado

Ecological systems ^a	Mean HMF (SD)
Upper montane (spruce-fir; 24, 26, 28)	0.329 (0.159)
Midmontane (Douglas-fir, lodgepole pine; 29, 30, 32)	0.428 (0.192)
Lower montane (ponderosa pine, pinyon-juniper; 34, 35, 36)	0.405 (0.203)
Aspen (22, 38)	0.313 (0.135)

^a Data from the Southwest ReGap program.

The mean and standard deviation (SD) of the final human modification (HMF) score for upper, mid, and lower montane, and aspen forests.

Table 7—Summary statistics of human modification framework (HMF) scores as summarized by GAP status level in Colorado

GAP status ^a	Mean HMF (SD)	Minimum and maximum HMF
1	0.244 (0.149)	0.1, 0.893
2	0.295 (0.166)	0.098, 1.0
3	0.421 (0.133)	0.1, 1.0
4	0.397 (0.192)	0.1, 1.0

^a Data from the Southwest ReGap program.

The mean, standard deviation (SD), minimum, and maximum of the final human modification score summarized for GAP status levels.

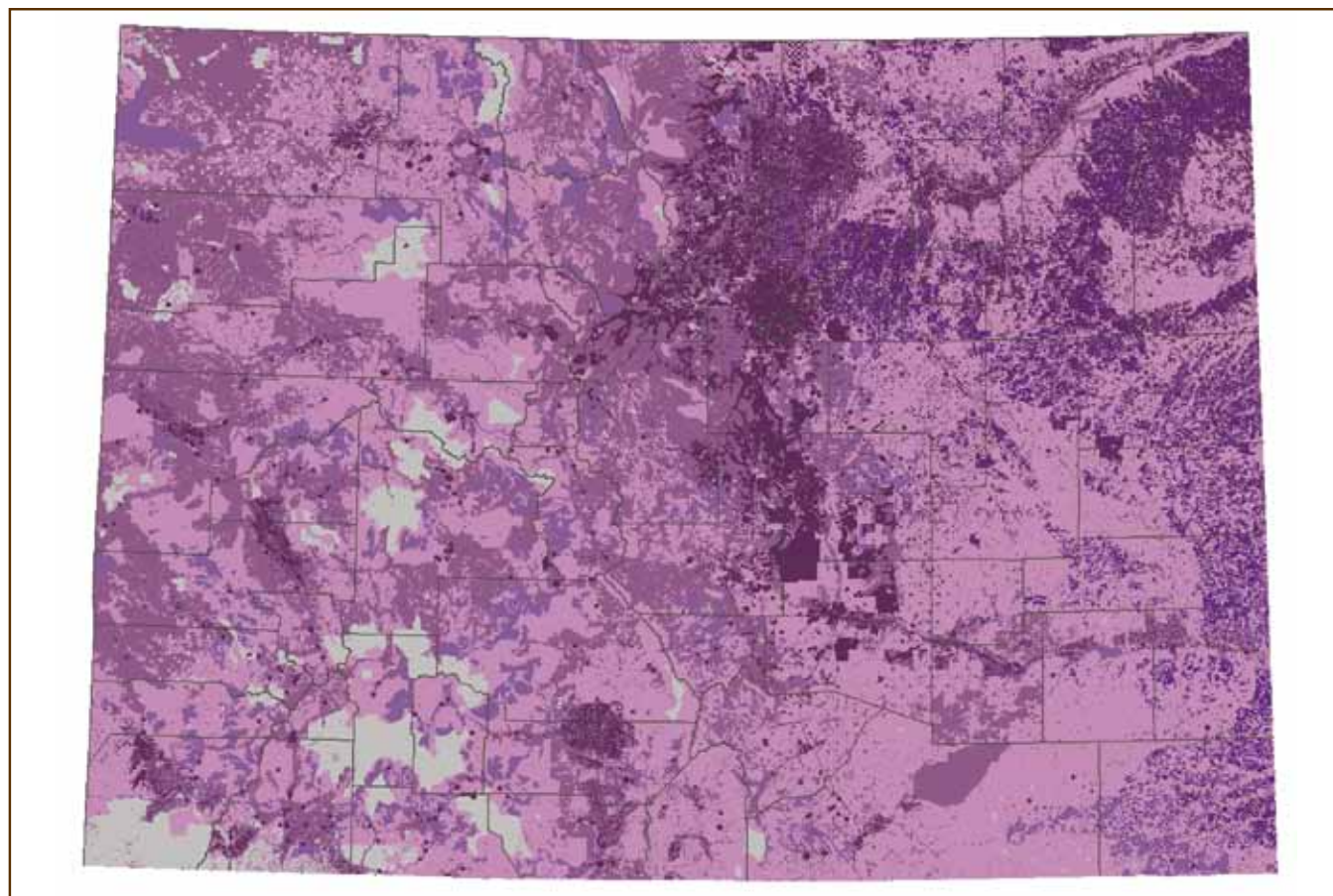


Figure 4—This map of Colorado shows the final map of degree of human modification (darker shade depicts higher degree of modification). This is a composite map of the urban/built-up, production/extraction, and recreation factor maps.

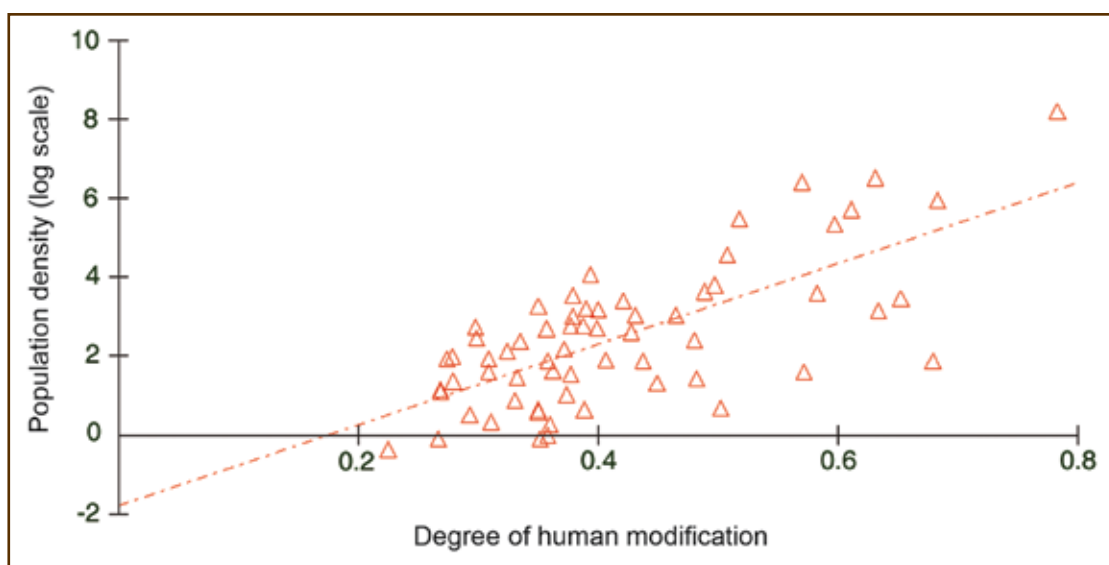


Figure 5—Population and degree of modification are compared in this graph of the county population for 2000 against the mean degree of human modification score. Although some counties fall along the linear relationship line, supporting the notion that larger populations lead to larger effects, there is a fair amount of variation as well.

this approach provides. By the same token, HMF provides a finer degree of detail than housing density—although housing density had a strong relationship.

We also compared HMF to GAP stewardship categories from the Southwest Regional GAP study (Table 7). We found that status 1 lands had lower average HMF values than status 2 (0.244 vs. 0.295), which were less than status 3 and 4. Interestingly, we found that status 4 had a lower mean HMF value than level 3 (0.397 vs. 0.421). Also, all status levels had very large maximum values (0.89 for level 1 and 1.0 for 2, 3, and 4).

Conclusion

In summary, we offer the human modification framework as an explicit approach to better quantify the spatial patterning and degree to which locations have been altered by human activities. We found that in our case study area, HMF showed markedly different results, particularly in the spatial patterns and distribution of disturbed areas, as compared to GAP status levels. An important, nonintuitive result is that we found that level 4 had less evidence of alteration than did level 3. This can occur because the HMF relies on surrogate variables that characterize current land use activities that have been shown to have effects on biodiversity. In contrast, GAP status levels characterize the legal protection from land use conversion (or management plans)—not the current condition *per se*. It is important to note that the HMF, as employed in this paper, shows existing (or at least currently known) modifications to the landscape, whereas the GAP and IUCN approach estimate long-term protection (in perpetuity). Although we did not document these trends here, the HMF methods could be easily applied to future growth or potential land use scenarios to examine the differences between their effects. The fine-grained approach and spatial context analysis in particular would be sensitive to differences in patterns of potential future land uses.

We believe that current approaches to estimating the degree of human modification can be improved in three ways. First, as understanding and elicitation of threats to protected areas matures, we believe that approaches will

need to move from using general surrogates to factors that are more closely linked to the threats, ideally in a mechanistic way. In addition, we believe that finer-grained data are increasingly required to address concerns about fragmentation and other pattern issues. Finally, the spatial (or perhaps landscape) context is critical to incorporate. Further work is needed to develop methods for generating landscape context that more directly reflect ecological processes. For example, neighborhoods for freshwater should use a series of flow-connected watersheds. Similarly, the flow or movement of processes through other transport vectors such as air, soil transport, animal movement, seed dispersal, etc., should be explicitly represented as well.

There are a number of data gaps that need to be filled. Some of these gaps could be filled by collecting data from field offices, standardizing the data and entering them into an electronic database (with spatial coordinates recorded). For example, grazing permit data have been collected for many decades, but much of it remains in an unstandardized, unmapped, and nonelectronic format. The location of timber harvests (year, harvest method) data are available, but not in a consistent, easy-to-use format. Another type of data gap would simply require acquisition and compilation of existing spatial databases (e.g., highway structures such as fencing, culverts, bridges, etc., or hiking trails that would include structures, tread type, width, use, etc.). Spatial data about water infrastructure, in addition to large dams, would be useful as well for structures such as ditches, levees, rip-rap, and channelized segments.

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Modeling Species' Realized Climatic Niche Space and Predicting Their Response to Global Warming for Several Western Forest Species With Small Geographic Distributions

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Abstract

The Random Forests multiple regression tree was used to develop an empirically based bioclimatic model of the presence-absence of species occupying small geographic distributions in western North America. The species assessed were subalpine larch (*Larix lyallii*), smooth Arizona cypress (*Cupressus arizonica* ssp. *glabra*), Paiute cypress (syn. Piute cypress) (*Cupressus arizonica* ssp. *nevadensis*), and Macfarlane's four-o'clock (*Mirabilis macfarlanei*). Independent variables included 33 simple expressions of temperature and precipitation and their interactions. These climate variables were derived from a spline climate model for the Western United States that provides point estimates (latitude, longitude, and altitude). Analyses used presence-absence data largely from the Forest Inventory and Analysis, USDA Forest Service database. Overall errors of classification ranged from 1.39 percent for Macfarlane's four-o'clock to 3.55 percent for smooth Arizona cypress. The mapped predictions of species occurrence using the estimated realized climatic niche space were more accurate than published range maps. The Hadley and Canadian general circulation models (scenario IS92a for 1 percent increase GGa/year) were then used to illustrate the potential response of the species' contemporary realized climatic niche space to climate change. Predictions were mapped at a 1-km² resolution. Concurrence between species' geographic distribution and their contemporary realized climatic niche rapidly disassociates through the century. These models demonstrate the heightened risk for species occupying small geographic ranges of displacement into climatic

disequilibrium from rapid climate change and provide tools to assist decisionmakers in mitigating the threat.

Keywords: Bioclimatic models, climatic distributions, climatic niche, global warming, Random Forests multiple-regression tree, response to climate change, narrow endemic.

Introduction

Climate is a principle factor that controls where species occur in nature (Woodward 1987). As the climate changes so then does the distribution of species. Long-lived plant species have adapted repeatedly to past climate change (see Ackerly 2003). When climate change exceeds species' tolerance limits, continued survival is dependent on the species ability to genetically adapt or migrate, or both, to suitable climate. These processes have contributed successfully to the persistence of long-lived plant species in response to past climate change (Davis and Shaw 2001). Their effectiveness under future climate change may be exacerbated by the increased rate of change predicted to occur over the present century. Projections for change in global climate rival historical periods of climate change at an accelerated rate (Houghton and others 2001). The accelerated rate of change threatens to displace current plant species distributions into climatic disequilibrium resulting in an increased potential for extinction of all or portions of species' ranges (Thomas and others 2001).

The objective for these analyses was to develop bioclimatic models that predict the occurrence of species with small natural distributions in the Western United States and project where suitable climates for the natural occurrence of these species may occur in the future in response to global warming. Three species were selected for the purposes of this analysis. The first was subalpine larch (*Larix lyallii* Parl.), a high-elevation, deciduous conifer inhabiting the Pacific Northwest. The second included two subspecies of Arizona cypress (*Cupressus arizonica* Greene), the smooth Arizona cypress (*C. arizonica* var. *glabra* (Sudw.) Little), which is endemic to central Arizona, and the Paiute cypress

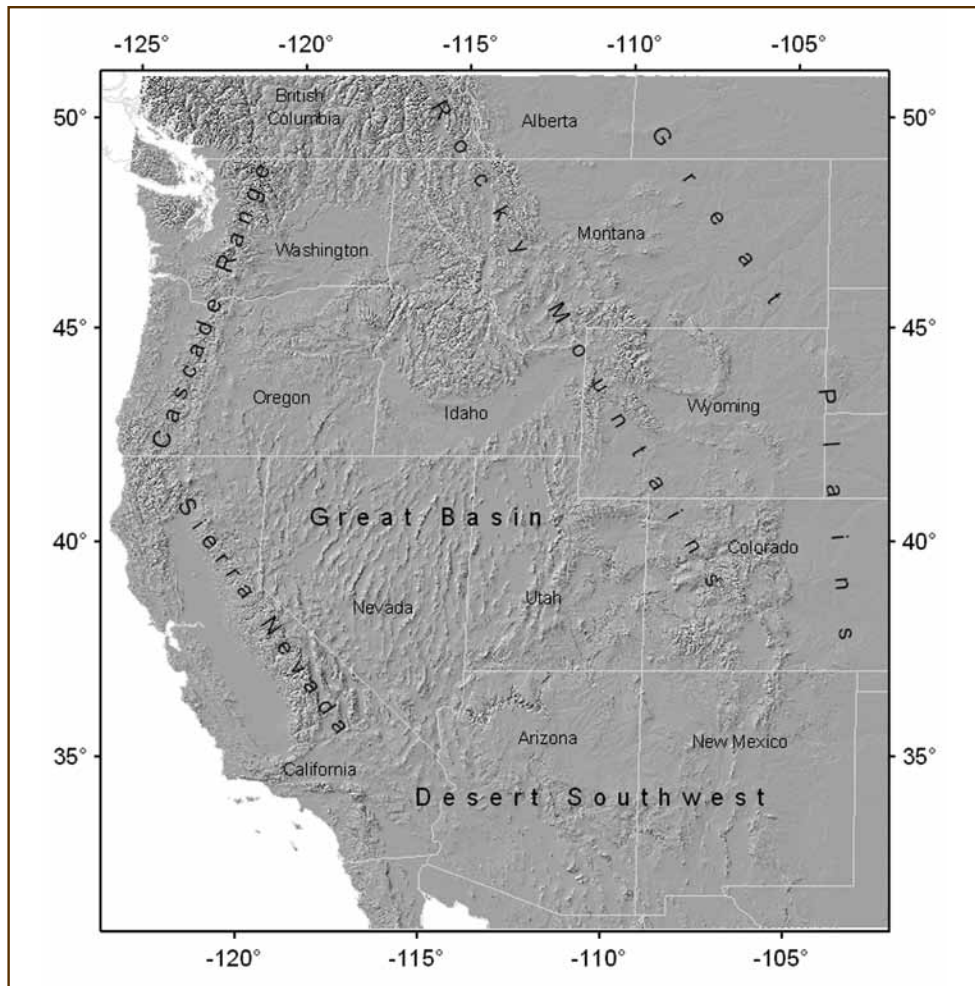


Figure 1—This shows the Western United States and southwestern Canada (from Rehfeldt and others 2006).

(syn. Piute cypress) (*C. arizonica* ssp. *nevadensis* Abrams E. Murray), which is endemic to the southern tip of the Sierra Nevada. The third was Macfarlane's four-o'clock (*Mirabilis macfarlanei* Constance & Rollins), a long-lived, deep-rooted perennial species that inhabits mid-to low-level canyon grasslands near the Oregon-Idaho northern border and is listed as endangered by the Bureau of Land Management (BLM) (USFWS 1996).

Methods

Study Area

The area of study consisted of the region supported by the climate surfaces of Rehfeldt (2006). This area encompasses

the Western United States and southwestern Canada, latitudes 31° to 51° N and longitudes 102° to 125° W (Figure 1).

Climate Estimates

Spline climate model (Rehfeldt 2006) was used to estimate climate at point locations (latitude, longitude, and altitude). The model is based on monthly minimum, maximum, and average temperatures and precipitation normalized for the period of 1961-90. This data was obtained from more than 3,000 weather stations dispersed throughout the conterminous Western United States and southwestern Canada. Hutchinson's (1991, 2000) thin plate splines were used to develop geographic surfaces for monthly data. The result produced 48 surfaces representing minimum, maximum, and average temperature and precipitation for each month

Table 1—Climate variables and their acronyms used as independent variables in regression analyses (from Rehfeldt and others 2006)

Acronym	Definition
MAT	Mean annual temperature
MTCM	Average temperature in the coldest month
MMIN	Minimum temperature in the coldest month
MTWM	Mean temperature in the warmest month
MMAX	Maximum temperature in the warmest month
MAP	Mean annual precipitation
GSP	Growing season precipitation, April through September
TDIFF	Summer-winter temperature differential, MTWM-MTCM
DD5	Degree-days > 5 °C
DD0	Degree-days < 0 °C
GSDD5	Accumulated growing-season degree days > 5 °C, April through September
MINDD0	Minimum degree-days < 0 °C
SDAY	Julian date of the last freezing date of spring
FDAY	Julian date of the first freezing date of autumn
FFP	Length of the frost-free period
D100	Julian date the sum of degree-days > 5 °C reaches 100
AMI	Annual moisture index, DD5/MAP
SMI	Summer moisture index, GSDD5/GSP
PRATIO	Ratio of summer precipitation to total precipitation, GSP/MAP
Interactions: MAP x DD5, MAP x MTCM, GSP x MTCM, GSP x DD5, DD5 x MTCM, MAP x TDIFF, GSP x TDIFF, MTCM/MAP, MTCM/GSP, AMI x MTCM, SMI x MTCM, TDIFF/MAP, TDIFF/GSP, PRATIO x MTCM, PRATIO x DD5	

of the year. For our analyses, these surfaces were used to estimate 33 climate variables that represent a range in climate variables from simple sums to complex interactions of temperature and precipitation (Table 1).

An updated version of the spline climate model was used to predict the effects of global warming for decades beginning in 2030, 2060, and 2090. The model used a regional summary of the IS92a scenario (1 percent per year increase in greenhouse gases after 1990) of the International Panel on Climate Change (Houghton and others 2001) from General Circulation Models (GCMs) produced by the Hadley Centre (HadCM3GGa1) (Gordon and others 2000) and

the Canadian Centre for Climate Modeling and Analysis (CGCM2_ghga) (Flato and Boer 2001). These two models are generally well respected, and averages of their predictions should provide an illustration of the potential effects of global warming.

Vegetation Data

Species were selected for this analysis based on their small geographic range and the availability of point location data to the authors. Species presence data were acquired from Barnes (1996) for Macfarlane's four-o'clock, Arno (1970), Khasa and others (2006), and Forest Inventory and Analysis, USDA Forest Service (FIA) database for subalpine larch, and unpublished field data from Forest Service RMRS, Moscow Forestry Sciences Laboratory, for smooth Arizona cypress and Paiute cypress (see Rehfeldt 1997). The FIA plot data were used to identify locations where the species did not occur. Their database was developed from a systematic sampling of woody vegetation permanent plots on forested and nonforested lands (Alerich and others 2004, Bechtold and Patterson 2005) within the United States. In accordance with FIA proprietary restrictions, plot locations are not available for publication. However, we were given access to their database to generate climate estimates for each plot directly from uncompromised field data. The resulting data sets include a tabulation of the presence-absence of species at approximately 117,000 locations.

Statistical Procedures

The Random Forests classification and regression tree package (Breiman 2001, Liaw and Wiener 2002, R Development Core Team 2004) was used to model species presence and absence. This tree-based method of regression uses a nonparametric approach and is resistant to overfitting, as multicollinearity and spatial correlation of residuals are not issues (Breiman 2001). Consequently, the algorithm was well suited for our analyses, which used variables among which intercorrelations could be pronounced.

An analysis data set was constructed for each species that initially included all predictor variables. Observations in this data set include all the observations with presence = yes weighted by a factor of 3. These observations made up

Table 2—Number of field plots in which each species was present, the number of locations contained within the 33-variable climatic envelope, the number of standard deviations for each climate variable in which the envelope was expanded, and the number of locations within the expanded envelope

Species	Number of plots			
	Present	Envelope	Standard deviations	Expanded envelope
Subalpine larch (<i>Larix lyallii</i>)	168	683	± 4.2	12,809
Smooth Arizona cypress (<i>Cupressus arizonica</i> ssp. <i>glabra</i>)	22	156	± 3.5	1,782
Paiute cypress (<i>Cupressus arizonica</i> ssp. <i>Nevadensis</i>)	49	537	± 11.6	3,811
Macfarlane's four-o'clock (<i>Mirabilis macfarlanei</i>)	27	107	± 3.0	1,910

40 percent of the total for a species (Table 2). The remaining observations (60 percent) of presence = no were selected by a stratified random sample of locations from two strata constructed using threshold values of the 33 predictor variables that define a climatic envelope for the species. The first stratum is the space formed by an expanded climatic envelope, where the expansion is defined by increasing the range for the threshold values of each climatic variable. The climate envelopes were expanded by factors large enough to produce about 20 times the number of locations sampled. The second stratum included locations outside the expanded envelope.

Regression analysis used 10 independent forests of 100 independent regression trees. Random Forests builds each tree using a separate boot-strap sample of the analysis data resulting in about 36 percent of the observations being used to compute classification error.

A set of regressions were run. The first regression used 33 climate predictors. Random Forest produces indices of variable importance (mean decrease in accuracy). The 12 least important variables were dropped after the first run. The regression procedure was then rerun nine more times with the remaining predictors, whereby the least important 1 to 3 predictors were dropped at each run until classification errors began to increase. The Random Forests run with the fewest variables selected prior to detecting an increase in classification error was considered the most parsimonious bioclimatic model for the species.

Mapping Procedure

Rehfeldt's climate surfaces (2006) and those updated to convey global warming were used to estimate the climate for nearly 5.9 million pixels (1 km² resolution) representing the terrestrial portion of the study area. The average altitude was made available from Globe (1999). The estimated climate and projected climates of each pixel were run down the 100 regression trees in the final set and the number of trees that predict the species is present and the number predicting absence were tabulated. A single-tree prediction is termed a vote. The votes were grouped into six categories: < 50, 50-60, 60-70, 70-80, 80-90, 90-100 percent. We consider any pixel in the first group as not having suitable climate for the species and define pixels in the other five categories as the species' realized climatic niche space. The fit of the mapped projections were assessed visually by comparing them with locations where the species were observed or Little's range maps (1971, 1976) that are available as digitized files (USGS 2005), or both.

Results

We found that reasonably parsimonious bioclimatic models are driven by either three or four climate variables (Table 3). Overall out-of-bag classification errors from fitting of the Random Forests algorithm ranged from 1.39 percent for Macfarlane's four-o'clock to 2.2 percent for subalpine larch. The models predicted species occurrence where species were known to occur with perfect accuracy (0 errors of omission); the error therefore is due to commission,

Table 3—Statistical output from bioclimatic models: classification errors from the confusion matrix, and the important predictors according to mean decrease in accuracy; variable acronyms are keyed to Table 1

Species	Classification errors (%)			Important variables
	Omission ^a	Commission ^b	Over all ^c	
Subalpine larch (<i>Larix lyallii</i>)	0	3.94	2.2	GSP × TDIFF, MAP × MTCM, MMIN
Smooth Arizona cypress (<i>Cupressus arizonica</i> ssp. <i>glabra</i>)	0	6.37	3.55	PRATIO, GSP × TDIFF, TDIFF
Paiute cypress (<i>Cupressus arizonica</i> ssp. <i>Nevadensis</i>)	0	3.09	1.73	MMIN, MAP × DD5, MTCM × GSP, PRATIO
Macfarlane's four-o'clock (<i>Mirabilis macfarlanei</i>)	0	2.55	1.39	MINDD0, MAP × TDIFF, TDIFF

^a Errors of omission (species presence = yes but model prediction presence = no).

^b Error of commission (species presence = no but model prediction = yes).

^c Combined overall error from omission and commission statistics.

predicting the presence of a species when, in fact, it was absent. The use of digital elevation model (DEM) and GIS to make predictions on a 1-km grid introduces additional error and uncertainty. The use of Little's range maps to validate the models' mapped projections also introduced the possibility of error. Range maps are known to contain errors (see Rehfeldt and others 2006) but perhaps more importantly, they only provide two-dimensional limits of the species distribution and do not indicate where species actually occur within the species boundaries. Even so, a visual comparison between these range maps and our predictions can be used for general validation of the models.

Subalpine Larch

A visual comparison of Little's range map shows that the projection of the bioclimatic model overestimates the distribution of subalpine larch in the western portion of southwestern British Columbia and portions of the central Rocky Mountains (Figure 2A). Error associated with Canadian projections may have been exacerbated by the lack of species presence-absence data for that region. Nonetheless, the model does an excellent job of predicting the United States distribution. Here the majority of the pixels receiving the greatest proportion of votes occur well within the range limits.

Mapped projections for the effect of global warming on distribution of subalpine larch's realized climatic niche space in decades 2030 (Figure 2B), 2060 (Figure 2C) and

2090 (Figure 2D) indicate a rapid decline in distribution (Table 4). For area studied, about 62 percent of the contemporary realized climatic niche space is projected to disappear by the end of decade 2030 (Table 4). Only about one-third of the remaining proportion is expected to occur in pixels suitable for the species today. By decade 2030, the mean elevation of the realized climatic niche space is projected to occur 100 m higher in altitude, and the minimum elevation is expected to rise almost 300 m by the end of the century. Also, by the end of the century, only 3 percent of the contemporary realized climatic niche space is expected to remain and should be relegated to a few high mountain peaks in south-central Idaho, southwestern Montana, the northern Cascade Range in Washington and to the Grand Teton Range in northwestern Wyoming. Note that suitable climate space in the Grand Teton Range is projected to first occur in decade 2030 and persist in decades 2060 and 2090.

Smooth Arizona Cypress and Paiute Cypress

The predicted contemporary realized climatic niche space for smooth Arizona cypress and Paiute cypress better described actual distributions than Little's range map (Figures 3A, 4A). Although a few populations were not mapped in the realized climatic niche, we suspect that this omission was due to either a mapping resolution that was too coarse or to field locations that were not accurate enough.

In response to global warming under the IS92a scenario, the realized climatic niche space of Arizona smooth

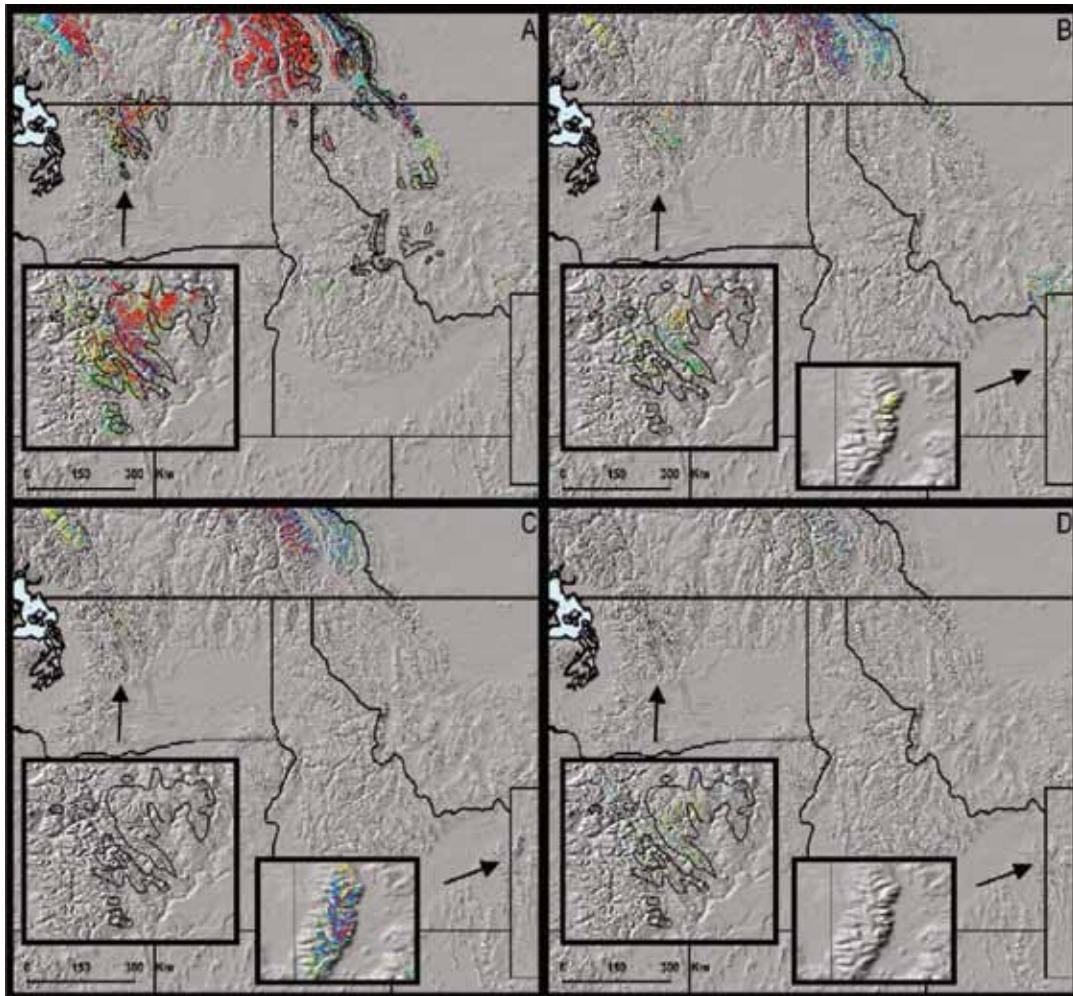


Figure 2—Modeled realized climatic niche for subalpine larch (*Larix lyallii*) and its predicted response to climate change for decades beginning in (B) 2030, (C) 2060, and (D) 2090. Areas in (A) outlined in black depict species boundaries defined by Little (1971). Colors code the proportions of votes received by a pixel in favor of being within the climate profile: no color 0%–50%; yellow 50%–60%; green 60%–70%; light blue 70%–80%; dark blue 80%–90%; and red 90%–100%.

cypress should shift about 200 to 350 km northwest of its contemporary location (Figures 3B, 3C, 3D). The area occupied should increase by about 1.5 and 2 times its contemporary size in decades 2030 and 2060, respectively (Table 4). In decade 2090, the area decreases to 1.2 times the contemporary size as the distribution shifts to northern Nevada and southwestern Colorado. In all three future decades, the contemporary realized climatic niche space is expected to be prominent in valleys where the Arizona, Nevada, and Utah borders meet. This includes the Virgin Mountains in Nevada, an area where naturalized populations of the subspecies have been observed (Charlet 1996).

Only 14 percent of the climate space is expected to remain in place in its contemporary location through the decade of 2030, and none is to remain in place for decades 2060 and 2090. By the end of the century, average elevation is projected to increase 611 m.

By decade 2030, the Paiute cypress realized climatic niche space will lie outside its contemporary distribution. The realized climatic niche space is projected to shift nearly 40 km to the east along the southeastern slopes of the Sierra Nevada (Figure 4B). Overall, the area of its distribution should shrink about 86 percent (Table 4). Altitudes, however, should increase by about 100 m. The realized

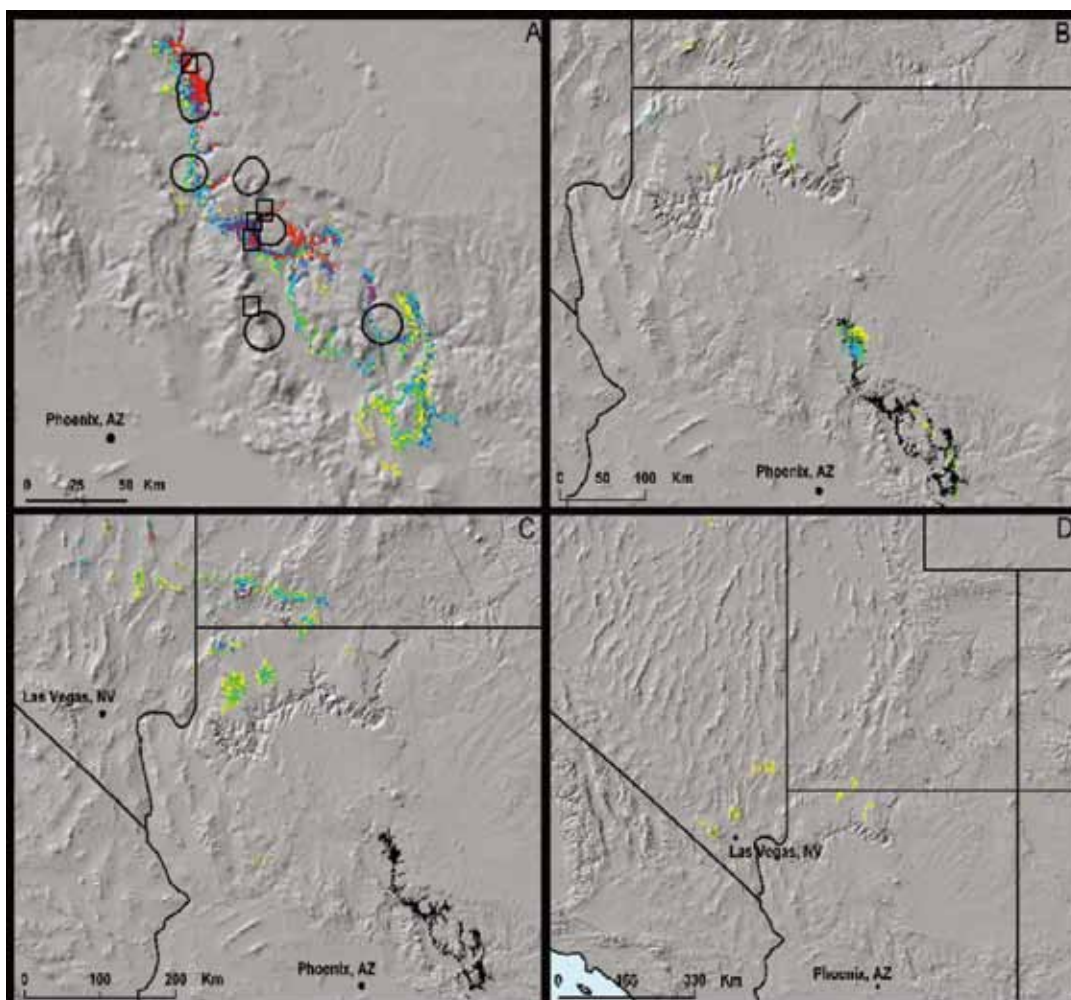


Figure 3—(A) Modeled realized climatic niche for smooth Arizona cypress (*Cupressus arizonica* ssp. *glabra*) and its predicted response to climate change for decades beginning in (B) 2030, (C) 2060, and (D) 2090. Areas in (A) outlined in black depict species boundaries defined by Little (1971) and squares indicate sites known to be inhabited. Colors code the proportions of votes received by a pixel in favor of being within the climate profile: no color 0%–50%; yellow 50%–60%; green 60%–70%; light blue 70%–80%; dark blue 80%–90%; and red 90%–100%.

Table 4—Projected change in area of species realized climatic niche space in response to climate change during three decades, and, in parenthesis, the percentage realized climatic niche space expected to match their contemporary distribution

Species	Year of projected change		
	2030	2060	2090
	Percent		
Subalpine larch (<i>Larix lyallii</i>)	-62 (26)	-76 (21)	-97 (3)
Smooth Arizona cypress (<i>Cupressus arizonica</i> ssp. <i>glabra</i>)	+158 (14)	+210 (0)	+115.7 (0)
Paiute cypress (<i>Cupressus arizonica</i> ssp. <i>nevadensis</i>)	-86 (0)	-91 (0)	-95 (0)
Macfarlane's four-o'clock (<i>Mirabilis macfarlanei</i>)	+807 (23)	+1912 (2)	+3343 (0)

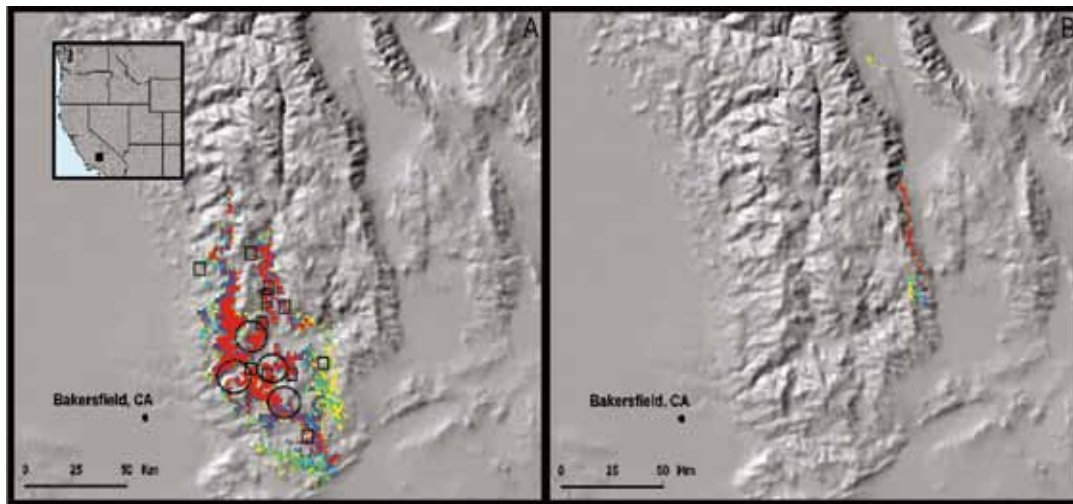


Figure 4—(A) Modeled realized climatic niche for Paiute cypress (*Cupressus arizonica* ssp. *nevadensis*) and its predicted response to climate change for the decade beginning in (B) 2030, in California. Areas in (A) outlined in black depict species boundaries defined by Little (1971) and squares indicate sites known to be inhabited. Colors code the proportions of votes received by a pixel in favor of being within the climate profile: no color 0%–50%; yellow 50%–60%; green 60%–70%; light blue 70%–80%; dark blue 80%–90%; and red 90%–100%.

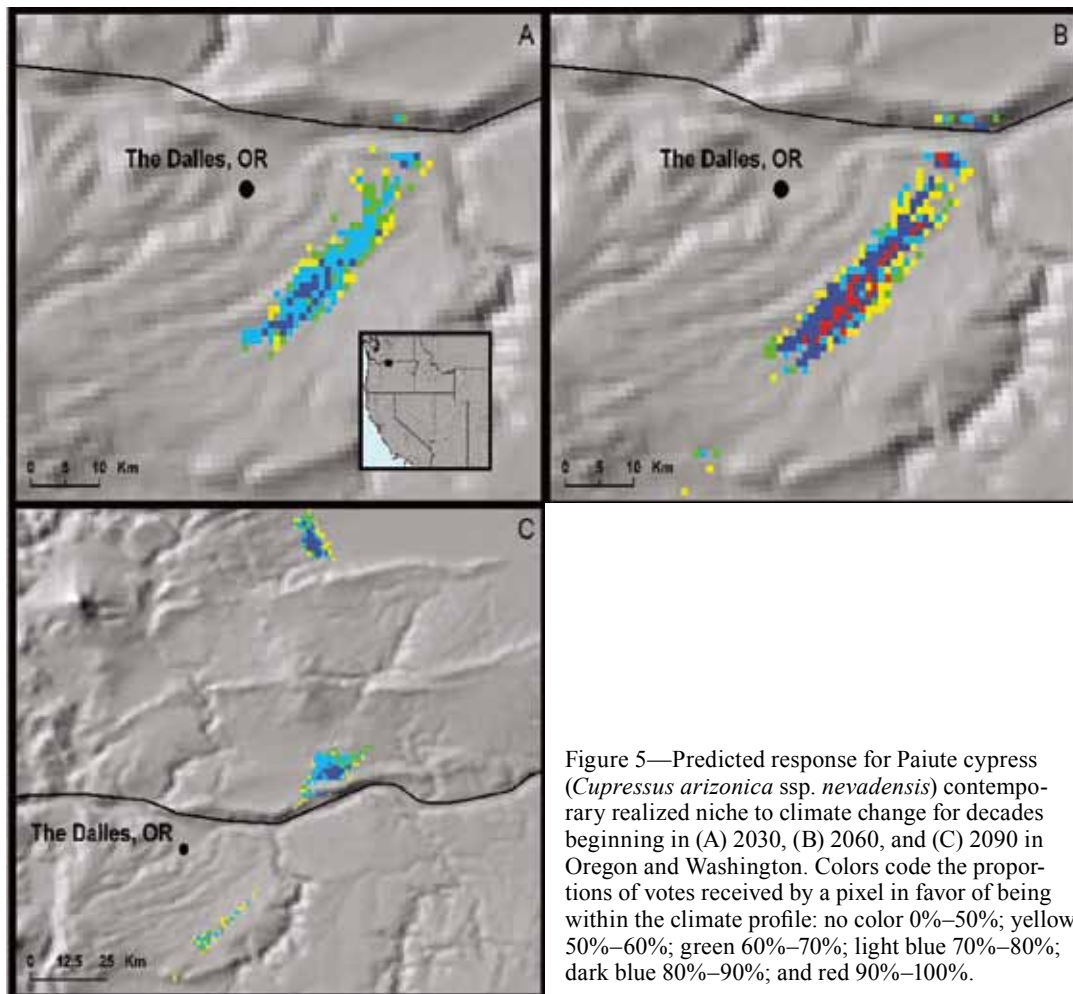


Figure 5—Predicted response for Paiute cypress (*Cupressus arizonica* ssp. *nevadensis*) contemporary realized niche to climate change for decades beginning in (A) 2030, (B) 2060, and (C) 2090 in Oregon and Washington. Colors code the proportions of votes received by a pixel in favor of being within the climate profile: no color 0%–50%; yellow 50%–60%; green 60%–70%; light blue 70%–80%; dark blue 80%–90%; and red 90%–100%.

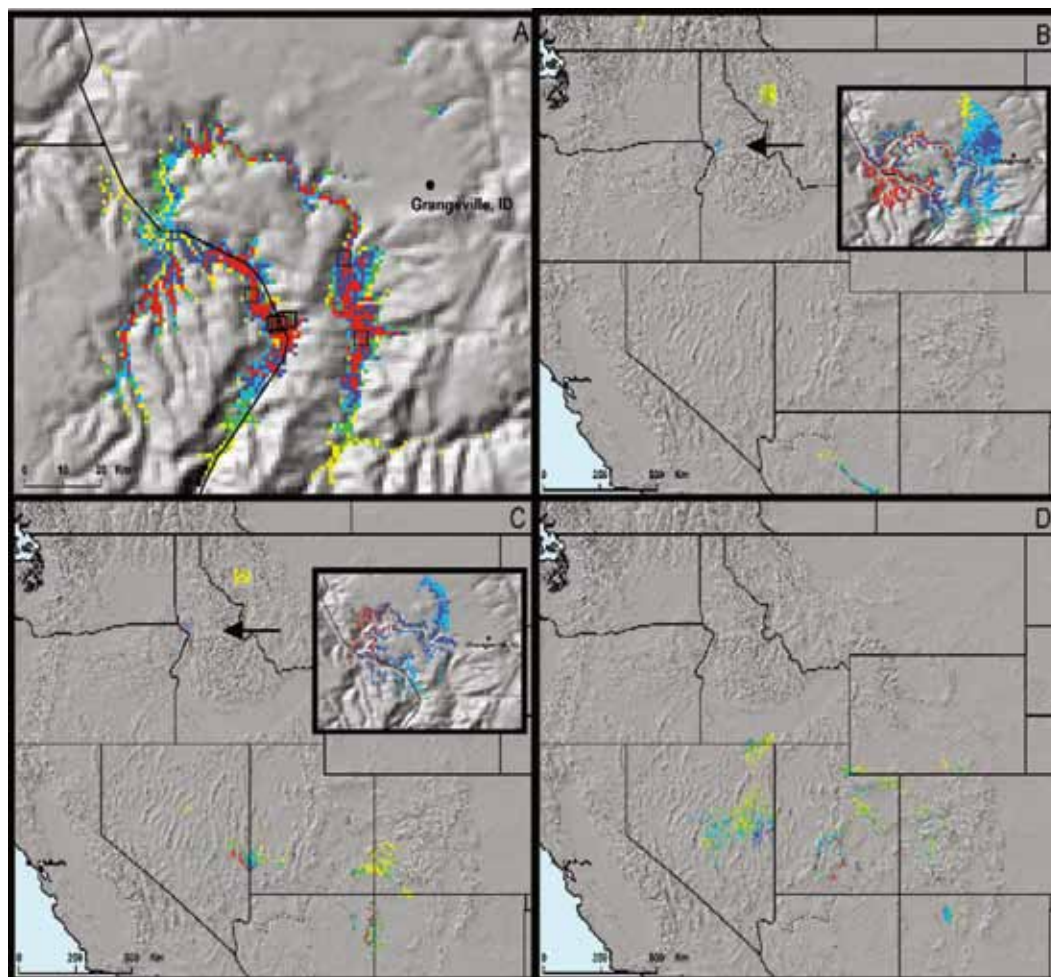


Figure 6—(A) Modeled realized climatic niche for Macfarlane's four-o'clock (*Mirabilis macfarlanei*) and its predicted response to climate change for decades beginning in (B) 2030, (C) 2060, and (D) 2090. Squares in (A) indicate sites known to be inhabited. Colors code the proportions of votes received by a pixel in favor of being within the climate profile: no color 0%–50%; yellow 50%–60%; green 60%–70%; light blue 70%–80%; dark blue 80%–90%; and red 90%–100%.

climatic niche space should also appear nearly 1100 km to the north near The Dalles, Oregon, by the decade of 2030 (Figure 5A) where it is projected to persist throughout the century (Figures 5B, 5C). In fact, by 2060, the realized climatic niche space of this subspecies occurs exclusively in Oregon and should occupy an area less than 90 percent of its contemporary distribution. The mean elevation for these new sites in Oregon and Washington are about 200 to 300 m lower in elevation than the contemporary distribution.

Macfarlane's Four-O'clock

Through most of the century, climate suitable to Macfarlane's four-o'clock should remain near its contemporary

location (Figures 6B, 6C). The realized climatic niche, however, is expected to climb the canyon's slopes along the Snake, Imnaha, and lower Salmon Rivers in decade 2030 (Figure 6B). In the decade of 2060, it should occur in these canyons at altitudes nearly 142 m higher than today (Figure 6C), but, by the end of the century, it should disappear in these particular canyons but reappear within the Snake River watershed in southern Idaho (Figure 6D). Elsewhere in the West, however, the profile of this species should expand rapidly and increase in altitude (Table 4, Figures 6B, 6C, 6D), increasing by a factor of 8, 19, and 34 times the contemporary realized climatic niche space in decades 2030, 2060, and 2090, respectively (Figure 6, Table 4).

Discussion

Our bioclimatic models predict the occurrence of species and attempt to identify suitable habitats. As shown, these models can also be updated with predicted future climates to project the redistribution of species' contemporary realized climatic niche. The validity of these projections are dependent on the accuracy of the bioclimatic models as well as how closely the predicted future climate using the IS92a scenario from the Canadian and Hadley GCMs matches the actual climate of the future.

The climate variables identified as important for predicting species occurrence differed among the species and subspecies analyzed. These differences suggest that they will respond uniquely to climate change. Despite these differences, it appears that their small geographic distributions predispose them to a shared threat of more rapid onset of climatic disequilibrium in comparison with species occupying large-scale distributions. Analyses of species occupying larger scale distributions tend to report greater areas of unaffected distribution and show less separation between existing distributions and the geographic occurrence of suitable climate (Bakkenes and others 2002, Iverson and Prasad 1998, Rehfeldt and others 2006). The extent of disequilibrium predicted by these studies may, however, be an underestimate owing to the potential for maladaptation to climate within species (Rehfeldt 2004, Rehfeldt and others 1999, Rice and Emery 2003).

As natural systems attempt to regain equilibrium with the novel distribution of climates, distributions will shift. Redistribution rates, however, will be influenced by genetic structure, autecology, life history, reproductive capabilities, and ecophysiology (see, for instance, Ackerly 2003). Consequently, all of these factors should be taken into consideration in interpreting projections from bioclimatic models. The physiology of subalpine larch, for instance, appears to be consistent with the climatic profile described by our models. Its realized niche seems dependent on a superior hardiness and resistance to winter desiccation (Burns and Honkala 1990). Subalpine larch, however, does not compete well with other evergreen species (Arno and Hammerly 1984). Hence, the continued warming trend will

likely eliminate subalpine larches from the Western United States through competitive exclusion. Common garden studies with smooth Arizona cypress and Paiute cypress by Rehfeldt (1997) revealed genetic variability equivalent to that conveyed by broadly distributed conifer species. This indicates populations within the species may only be adapted to a portion of its present or future realized climate space. In addition, both smooth Arizona cypress and Paiute cypress are fire dependent. Fire management practices may have had a substantial influence on limiting their distribution (Marshall 1963). Consequently, for these subspecies of *Cupressus arizonica*, our estimate of the realized climatic niche space may underestimate the breadth of climatically suitable area. Finally, Macfarlane's small distribution is attributed to reduced compatibility between its floral biology and pollinators (Barnes 1996). The rates that the range may expand would depend, therefore, on the presence of compatible pollinators.

Bioclimatic models represent tools that can be used to assist decisionmakers in managing threats associated with global warming. These models can be updated with predicted future climate to identify locations where suitable climate should occur. These predictions can be used by managers to assist the natural processes by transferring species to the future location of climate suitable for their survival. Ideally the application of bioclimate models for this purpose should use multiple general circulation models and climate change scenarios to reduce uncertainty associated with predicted future climate (see Rehfeldt and Jaquish 2010). To be sure, additional modeling is needed that integrates a general understanding of climate, climate change, and plant-climate relationships. Interpretation of results by resource managers also requires integration of additional layers of information such as land use (see Hannah 2006).

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AIR and WATER

Syntheses

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The Influence of Forest Management on Vulnerability of Forests to Severe Weather

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Abstract

Excessive wind, ice, and snow regularly cause major disturbances to forests in many parts of the world, significantly impacting both ecological conditions and economic returns to forest landowners. These events cause immediate losses for landowners, and the broken and uprooted trees left in the wake of a storm increase the risk that wildfires, disease, and pest outbreaks will cause secondary damage to the surviving trees. Although weather severity (e.g., wind-speed and duration, or form and amount of precipitation) is clearly an important factor in the occurrence and severity of forest damage, site conditions, tree characteristics, and stand characteristics play a major role in determining resistance of a forest stand to wind, ice, and snow loading. However, the relationships between site, tree, and stand characteristics and weather damage are complex and vary spatially and temporally. In this article, we review and synthesize the literature on the risk of forest damages from severe weather—focusing on wind, ice, and snow—and the factors that influence vulnerability. Forest management decisions are found to play an important role in influencing risk associated with severe weather events. The risk of damages can be managed through strategies such as selection of planting site and species, stocking, and selection and timing of silvicultural treatments. Optimal management strategies under endogenous risk vary based on the probability of damage and management objectives.

Keywords: Damage mitigation, ice, natural disturbances, forest policy, risk management, wind.

Introduction

Natural disturbances play an important role in forest stand dynamics and ecology, with effects that are highly dependent on characteristics of the specific disturbance (Foster and Boose 1995). Severe weather events that bring high winds or heavy precipitation or both (e.g., hurricanes and ice, snow, and hailstorms), drought, flooding, lightning strikes, wildfires, and disease and pest outbreaks are a regular feature of forest landscapes. Although these disturbances may serve important ecological functions, they also significantly—and usually negatively—affect economic returns to forest landowners. Wind, ice, and snowstorms affect large areas of forest relatively frequently in some important timber production regions. These storms impact forest landowners by causing the loss of merchantable timber, increased risk of secondary damages to wildlife and from disease and pests in damaged stands, and depressed timber prices in the immediate aftermath of events that cause widespread damage. These risks have led to considerable interest in identifying factors that influence damage levels as well as ways to mitigate damages.

To optimally manage risk, it is necessary to identify and characterize the hazards faced, assess the risk associated with these hazards, identify options to mitigate risk, choose the optimal combination of options given the manager's objectives (e.g., producing amenities or maximizing profitability of timber production), and implement the strategy selected. In the case of forested landscape management, these tasks are complicated by long planning horizons, which lead to substantial uncertainty concerning future prices, production costs, forest policies and regulations, and objectives, in addition to uncertainty regarding severe weather impacts (Wilson and Baker 2001). Spatial and temporal heterogeneity combined with the infrequency of major storms impacting a given area (compared with a human lifespan) make it relatively difficult to learn from previous storms. Damage severity depends on the complex

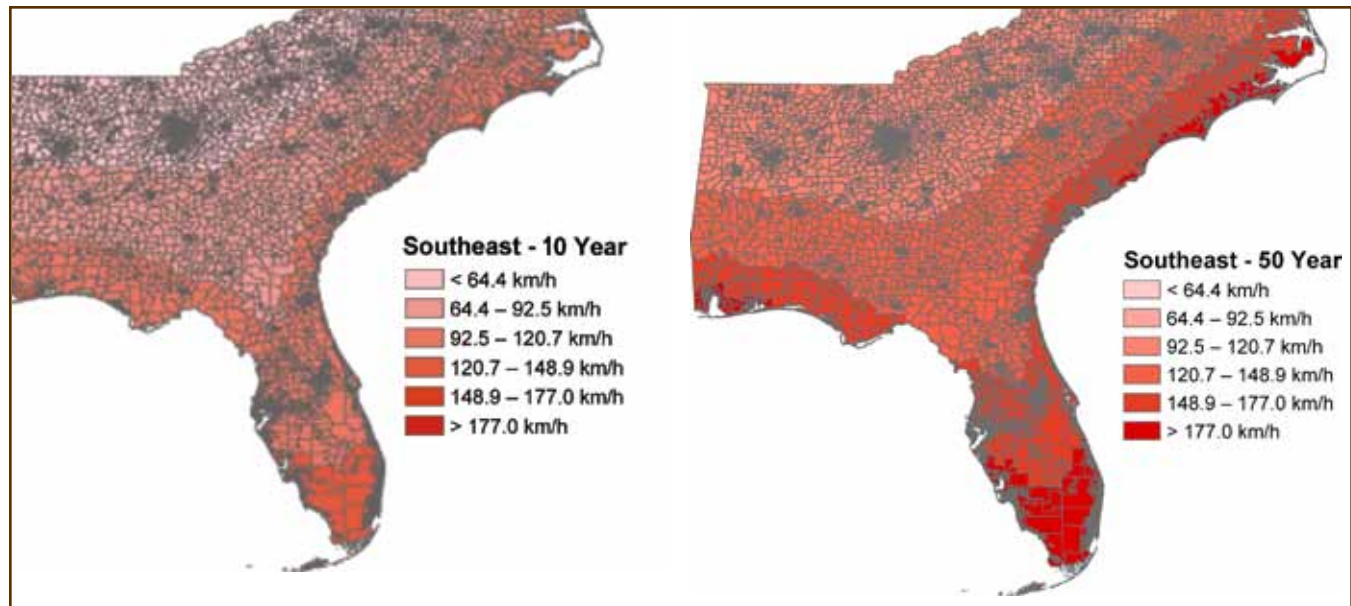


Figure 1—Simulated peak hurricane windspeed associated with hurricanes in the Southeast United States over 10- and 50-year periods using FEMA's HAZUS-MH model. In (a), which is for a 10-year period, only southern Florida and a few coastal census tracts in Alabama, South Carolina, and North Carolina are expected to experience hurricane-force winds (greater than 119 km per hour) within this time-frame. Over a 50-year period, however, the entire region is expected to have winds of at least tropical storm strength, with much higher winds in some coastal areas, as shown in (b). Source: FEMA HAZUS-MH model.

interaction of numerous tree, stand, and site characteristics. For instance, although windspeed is likely to be the most important determinant of timber losses, topography, soil conditions, planting density, species, stand age, the presence of gaps and edges, and other factors determine a stand's resistance to wind loading.

Many studies focus almost exclusively on biophysical properties of forest plots or individual trees to explain differences in damages without explicitly examining the role of landowners' forest management decisions in influencing those properties. However, forest management decisions can change susceptibility to wind damage through effects on tree and stand characteristics such as tree species, tree height, tree diameter, crown area, rooting depth and width, and stand density (Dunham and Cameron 2000, Kerzenmacher and Gardiner 1998, Peltola and others 1999, Peltola and others 2000). Stand age and forest structure may also contribute to vulnerability of forests to high winds (Everham and Brokaw 1996, Francis 2000, Mitchell 1995, Ruel 1995). Silvicultural treatments such as thinning and clearcutting adjacent stands are important as well (Foster 1988, Lohmander and Helles 1987, Mitchell 2000, Wilson

and Baker 2001). Similar factors have been related to the degree of ice damage to forests (Bragg and others 2003, Van Dyke 1999), though with different magnitudes and, in some cases, opposite direction of effect. Overlooking the impacts of management decisions on the vulnerability of a stand to weather damages may lead to inefficient decisionmaking by policymakers and private landowners.

In this article, we review and synthesize the literature on the risk of forest damages from severe wind, snow, and ice, the factors influencing the extent of damages in the event of severe weather, alternatives for mitigating weather damages, and policy implications. This type of information can inform forest management decisions and increase the efficiency of policy for managing public forest land as well as public policy that encourages timber stand management and compensates landowners for weather damages.

Severe Weather and Forest Impacts

This section describes characteristics of severe weather storms and selected major events that have taken place in recent years as well as resulting impacts on forests. These impacts include a variety of damages to trees because of

excessive loadings by high winds, ice storms, and snow accumulation. In addition to immediate uprooting and stem breakage, heightened tree mortality can continue over time as a result of substantial crown loss. Even trees that recover may have suffered permanent internal damage that reduces wood quality. In addition, damaged stands are more prone to secondary damages from fire, disease, insects, and competing plants.

High Winds

Hurricane-strength winds can cause severe defoliation and directly damage and kill trees through uprooting, breakage and loss of minor and major branches, and stem breakage. Biodiversity is affected through an increase in gaps of broken and uprooted trees and reallocation of light, water, nutrients, and other resources (Nowacki and Kramer 1998). The amenity value of a forest can decrease because of the higher proportion of treefall gaps. Wind damage imposes costs on commercial forestry owing to reduction in timber yields, unscheduled and costly thinnings, and added uncertainty for forestry planning (Quine and others 1995). In addition, broken and uprooted trees left unharvested can lead to additional costs by increasing the probability of disease outbreaks, insect infestations, and wildfires in the remaining growing stock as well as increasing the costs of containment.

Between 1900 and 2005, an average of 1.6 hurricanes per year made landfall between Texas and Maine, with an average of 0.6 major hurricanes (category 3 or above on the Saffir-Simpson scale) annually. In 2004 and 2005 alone, seven major hurricanes had an impact on the United States (Blake and others 2006). Figure 1 shows peak windspeeds associated with hurricanes in the Southeastern United States that are expected in 10-year and 50-year periods based on estimates from the Federal Emergency Management Agency's HAZUS-MH model. Within a 50-year period, the entire Southeast is expected to have winds of at least tropical storm strength.

Hurricanes resulting in the greatest loss of timber from 1900 to 2000 were the Hurricane of 1938 (in New England) and Hurricanes Camille (1969), Frederic (1979),

Hugo (1989), Opal (1995), and Fran (1996) (Lutz 2005). For example, Hurricane Hugo swept through South Carolina on September 21, 1989, damaging more than one-third of the State's timberland. The damaged volume of timber was estimated to be 36.8 million m³ (Remion 1990). More recently, Hurricanes Katrina and Rita damaged more than 2 million ha of forest in the gulf coast States, leading to timber losses of almost \$6 billion for affected landowners (though an estimated one-third of these losses may be recovered through salvage efforts) (SAF 2005).

Other regions are also affected by high winds. On average, there are almost 1,000 tornadoes per year in the United States, mostly in the continental interior. Although many are relatively small and weak and cause little damage, a small proportion do considerable damage to forests (see Peterson 2000). Dale and others (2001) estimate that tornado damage in the United States averages \$154 million annually. Other windstorms also occasionally result in major damages, such as the July 4, 1999, windstorm that severely impacted approximately 193 000 ha of Minnesota forest (Mattson and Shriner 2001). The hurricane-force winds that affect the coastal Pacific Northwest have a return interval of about 20 years (Wilson 2004).

Ice and Snow Storms

Recurring ice and snow storms are a significant hazard to forests in temperate climates. Snow or ice causes damage when the weight of frozen precipitation exceeds the buckling load of the portion of the tree bearing the load, causing bending or breakage. Bending can result in permanent internal damage without external signs of damage. There are important interactions between winter precipitation and wind because windspeeds required to cause damage are lower for trees loaded with snow and ice, all else equal. Every winter, ice and snow storms affect portions of eastern North America and Europe to some degree, with occasional major events covering millions of hectares and causing millions of dollars in damages. The majority of research on ice and snow damage has been conducted in these regions. Below, we focus on ice storms, recognizing that heavy snow can produce similar damages, particularly in the case of unusually wet snow.

Ice storms require a layer of warm air, with temperatures above freezing, between layers of air that are below freezing. As frozen precipitation falls through the layer of warm air, it melts and reaches the cold lower layer as rain. As the water droplets fall through this cold surface layer, they become supercooled, which means that the water in the droplets is cooled to temperatures below 0 °C without freezing. When this supercooled liquid strikes a cold surface, the impact triggers an almost instant transformation to a smooth thin layer of ice. Although ice storms can result in significant accumulation and major damages when conditions for ice formation are favorable, ice accumulation depends critically on characteristics of the layers of air and the water droplets. Small temporal or spatial differences in air temperature and droplet size frequently result in freezing rain mixed with sleet, snow, or nonfreezing rain, which substantially complicates area-specific forecasting of ice formation and accumulation (Heidorn 2001).

Ice accumulation is one of the most frequent forest disturbances in temperate regions (Irland 2000). However, most ice storms produce minimal ice accumulation and do not result in substantial losses. Some locations in North America experience at least some ice accumulation every 1 to 2 years on average, though most areas have return intervals of 5 years or more (Bragg and others 2003). Ice storms are estimated to occur about every 5 years in Northern New England States (Smith and Musser 1998), while ice storms have been estimated to occur every 6 years in the natural range of loblolly pine (*Pinus taeda* L.) (Shultz 1997) and every 5 to 12 years inland from the Gulf of Mexico (Bragg and others 2003).

Storms resulting in major damages are estimated to have a return time of 20 to 100 years in Northern North America (Van Dyke 1999). Several recent ice storms have caused catastrophic damages over very large areas. For instance, a 1998 ice storm in southeastern Canada and the Northeastern United States damaged over 10 million forested hectares (Irland 1998, Miller-Weeks and others 1999). The Southern United States is less prone to major ice storms, but the region has experienced a number of major events such as the December 2000 ice storms in Arkansas



Figure 2—Bending and windthrow due to a major windstorm. Illustrated in this picture is wind damage in quaking aspen (*Populus tremuloides*) in Chequamegon National Forest, northern Wisconsin, resulting from a windstorm in July 1999. The majority of damage is in the form of bending and windthrow rather than stem breakage.

that damaged an estimated 40 percent of the 7.4 million ha of forest in the State (Bragg and others 2003).

Tree Damage and Mortality

Wind can cause severe defoliation and can directly injure and kill trees through obvious damage such as uprooting (see Figure 2), breakage and loss of minor and major branches, and stem breakage (see Figure 3). Trees can be permanently damaged by the bending caused by wind, ice, and snow (see Figures 2 and 4), and compression injuries may be present even in seemingly undamaged trees. Severe bole injuries can lead to sap rot, slower growth, formation of compression wood, unusual growth rings, and intercellular voids between rings (Bragg and others 2003). Winds often carry saltwater inland, and foliage may die on trees that receive salt spray, although most affected trees will grow new leaves and recover (Gresham 2004).

The type of damage suffered tends to differ by tree size. Younger trees are more likely to bend with the wind without suffering serious damage, whereas older trees are more likely to uproot or break. Similarly, saplings and small-diameter trees can become seriously bent by ice accumulation, although these trees often recover. As diameter increases, the proportion of bending to breakage decreases. The majority of ice damage in larger trees is loss of branches and breakage of the main stem within the crown

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Figure 3—Stem breakage from a tornado. Widespread stem breakage is pictured in loblolly pine (*Pinus taeda*) from tornado winds in Texas. The damage appears very similar to that caused by major hurricanes.

(Van Dyke 1999). Understory trees may be damaged by windthrown trees or broken portions of trees. In addition, tree species differ in their ability to withstand high winds, ice, and snow. Key factors include shape and size of the crown, rooting depth, and wood strength.

Whether a tree recovers from damage depends significantly on crown loss. Hardwood trees are not usually killed by breakage, but those with greater than 75 percent crown loss are likely to die. Conifers broken below the live crown or that have lost a majority of the crown will generally not survive (Van Dyke 1999).

Increased Risk of Secondary Damages

The immediate loss of timber is not the only damage imposed by weather events. Uprooting of trees also causes significant soil turnover and increases risk of soil erosion (Lutz 1940, Lyford and MacLean 1966, Meyers and McSweeney 1995). In addition, the increased number of broken and uprooted trees raises the risk of wildfires as well as disease and pest outbreaks for the surviving trees, whereas newly created canopy gaps provide opportunities for plant invasions. The presence of damage and decay may also increase vulnerability to future wind, ice, and snow (Bragg and others 2003).

Fire hazard may be increased following major storms that result in increased dead tree biomass and low-growing



Figure 4—Tree bending from ice accumulation. The bending that can result from the weight of ice accumulation brought by a significant ice storm is shown in this picture. Although many of these trees will survive and may appear undamaged after they have straightened themselves, permanent internal damage may be present.

plants (Myers and Van Lear 1998). Federal and State forest resource departments frequently mention the heightened fire risk following major storm events as a key consideration for short-term forest recovery and management. In addition to increased fuel loading, access may be limited by downed trees, which further increases risk (Irland 2000). Myers and Van Lear (1998) examined the historical interaction between hurricanes and fires and concluded that these interactions had a major impact on the composition and structure of ecosystems in the Southern United States, although modern fire suppression efforts may have obscured this relationship in recent years.

Young trees can suffer bark damage following severe bending, which may increase susceptibility to disease. Any wounds permit disease fungi to infect damaged trees. Stain fungi may infect exposed wood within weeks, leading to wood quality degradation. Also, sap rot has been linked to sudden exposure of the stem to intense sunlight following canopy damage (Bragg and others 2003).

Broken and uprooted trees left unharvested also can lead to detrimental insect attacks on the remaining stand (Bakke 1989, Ravn 1985, Schroeder and Eidmann 1993). For example, severe outbreaks of spruce bark beetle (*Ips typographus*) occurred in Norway in 1971–1981, damaging trees equivalent to 5 million cubic meters of timber as

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an indirect consequence of extensive wind damage (Bakke 1989). Cain and Shelton (1996) reported that an ice storm in southern Arkansas contributed to a southern pine beetle (*Dendroctonus frontalis*) outbreak. Hurricane Hugo caused large areas of loblolly pine (*Pinus taeda*) to be broken or uprooted, and these trees were susceptible to insect infestations (Yates and Miller 1996). Within months of Hurricane Katrina, virtually all of the downed material in De Soto National Forest in Mississippi had been infested by bark beetles and woodborers, and bark beetles (*Ips* engravers) had begun to infest and kill root-sprung trees (Meeker and others 2006).

Gaps resulting from tree fall and defoliation present opportunities for establishment of invasive plants (Burke and Grime 1996, D'Antonio 1993, Davis and others 2000, Hobbs 1989, Hobbs and Huenneke 1992, Réjmanek 1989), which may change forest structure and decrease populations of indigenous species. Increased light levels from large canopy gaps may also encourage the growth of invasive plant species, which could influence forest succession and alter ecosystem function. Snitzer and other (2005) found that mean cover percentage of invasive plants increased by 47.8 percent in high-light canopy gaps created by Hurricane Isabel and by only 4.2 percent in less-damaged forest. Invasive species such as cogongrass (*Imperata cylindrica* (L.) P. Beauv.) can also contribute to heightened wildfire risk (SAF 2005).

Economic Impacts

Because major severe weather events can cause the direct loss of great quantities of timber, large numbers of landowners can suffer revenue losses and the entire forest products sector can be affected. Revenues from salvage operations are generally lower than those from typical harvests because saleable volume is reduced, particularly in cases where there is extensive stem breakage, but also because fungal decay reduces wood quality. At the same time, costs of salvaging damaged stands are usually higher than costs of conventional harvesting. Manley and Wakelin (1989), who took into account the effects of increased costs and reduced revenues, calculated that the discounted net present value of an impacted forest decreases by as much

as 11 percent for a 1-percent annual level of damage. Forest landowners also may be affected by depressed timber market prices (at least in the short run) because of the large volume of salvaged timber on the market. Other things being equal, greater production risk unambiguously reduces optimal rotation length and decreases expected returns to forest land and land values, although the effects of price risk are ambiguous (Prestemon and Holmes 2000, Prestemon and others 2001). Prestemon and Holmes (this volume) provide more information on the economic impacts of hurricanes on landowners.

Factors That Influence Vulnerability

The severity of wind, ice, and snow damage depends on the interaction of numerous biological, topographic, stand, tree, and management factors. Before reviewing forest management activities to mitigate damage, we need to know what factors most strongly influence the severity of damage caused by wind, ice, and snow. Many empirical studies have indicated that the susceptibility of forests to damage depends on tree and stand characteristics such as species, height, diameter, crown area, rooting depth and width, stand density, and presence of forest gaps or edges (Dunham and Cameron 2000; Kerzenmacher and Gardiner 1998; Nykänen and others 1997; Peltola and others 1999, 2000; Quine 1995; Solantie 1994; Valinger and others 1993; Zeng and others 2004). However, the statistical significance and even the direction of the effect of individual variables differ across studies. The natural systems involved and the interactions among factors are complex, so these apparent inconsistencies may be a result of nonlinearities in response and the omission of individual variables in some studies.

Weather Severity

The intensity and duration of severe weather events are among the most important determinants of forest damage associated with severe weather. The greater the intensity and duration of loading placed on trees, the higher the likelihood this loading will exceed critical thresholds and cause damage. However, few empirical studies have explicitly incorporated spatially differentiated, quantitative, physical measures of severity or duration or both (e.g., peak

wind gusts or ice accumulation). Storm conditions can vary considerably within a relatively small region, so local storm severity must be gauged accurately if researchers are to properly evaluate the impact of local management activities on damages.

Greater windspeed and duration mean that more force is exerted on trees. Thus, the probability of uprooting, stem breakage, or other damages is expected to increase with windspeed and duration. However, forests in areas where strong winds are common may be less susceptible to catastrophic damage because they have developed higher wind resistance in response to greater normal loading. For related reasons, wind direction also influences damage. Storm winds that come from a direction other than the typical prevailing winds may cause greater damage (Ruel 1995). For winds of a given velocity, wind direction affects the loading placed on objects.

Similarly, the greater the accumulation of ice or snow on trees, the greater the likelihood that the trees will suffer structural failure. The National Weather Service declares an ice storm when there is at least 0.6 centimeters of ice accumulation, but minor glazing (0.6 to 1.3 centimeters) may cause little damage (Lemon 1961). Major events may result in accumulations as high as 10 to 20 centimeters (Bragg and others 2003). Depending on weather conditions, ice may remain on trees for some time before melting or being shed. As ice is retained on the trees for longer periods, damages increase (Schultz 1997).

Numerous mechanistic simulation models have been used to simulate the probability of structural failure in response to increased wind and ice, or snow loading or both (Blennow and Sallnäs 2004, Dunham and others 2000, Miller and others 2000, Peltola and others 1999, Talkkari and others 2000). These models generally predict critical windspeeds and ice or snow loading or both required for uprooting and breakage as a function of site, tree, and stand characteristics based on biomechanical properties. Several models incorporate both sources of risk to account for interactions between ice and snow loadings and wind risk (e.g., Miller and others 2000, Talkkari and others 2000).

A few empirical studies have included windspeed or ice and snow accumulation as explanatory variables in their

regression models. Ramsey and others (2001) incorporated windspeed and duration to explain the variation in hurricane damage among forest types along the Atchafalaya River basin of Louisiana. Using forest-type distribution from Landsat Thematic Mapper image data, hurricane-impact data from very high-resolution radiometer images, and windspeed and duration calculated from a wind field model, they found that the estimated impact for each forest type was strongly related to the duration and speed of extreme winds from Hurricane Andrew. Proulx and Greene (2001) found higher damage in northern hardwoods associated with greater ice accumulation, but Olthof and others (2004) did not find a direct relationship between ice storm precipitation and forest damage.

Site Conditions

Site conditions are commonly identified as influencing severe weather damage. These are physical properties of a location, such as topography and soil conditions, that cannot typically be directly modified in a particular stand through management actions.

Numerous studies have shown that local topography is a key determinant of risk of wind damage (e.g., Boose and others 2004, Foster and Boose 1992, Kulakowski and Veblen 2002, Quine 1995, Ruel 1995), but the relationship is complex, and specific topographic factors identified as important differ from study to study. Impacts may be increased when there is a barrier behind the stand (Talkkari and others 2000). Stands at higher elevations, on easterly slopes, and near ridges were more likely to suffer wind damage in Colorado (Kulakowski and Veblen 2002). The key issue for topography seems to be its influence on the probability of local windspeeds reaching critical speeds (Quine 1995). In many cases, the topography variables are serving as proxies of local wind intensity in the absence of direct measures of windspeed. Links between topography and ice and snow damages are less studied, but ice damages are increased at greater elevations (Olthof and others 2004, Rhoads and others 2002, Van Dyke 1999) as well as on steep slopes (Proulx and Greene 2001, Van Dyke 1999).

Findings about relationships between soil conditions and wind damage are relatively consistent (Wilson 2004).

Soil saturation is an important factor affecting the probability of windthrow. When hurricanes or other wind storms pass through an area following periods of heavy rainfall, blowdown is more likely (Ruel 1995). Deeper, well-drained soils lower the risk of wind damage. Beese (2001) found wet soil to be one of the largest contributing factors to wind damage. Similarly, damages from ice and snow are greater for wet soils (Van Dyke 1999). Other conditions that prevent establishment of deep roots, such as the presence of boulders, bedrock, or other barriers close to the surface, also increase the probability of uprooting (Rhoads 1999).

Tree Characteristics

A number of studies have focused on relationships between wind, ice, and snow damage and various tree characteristics. Tree characteristics of interest include species, health, age, size, height-to-diameter (H/D) ratio, crown shape, and canopy position.

Conifers tend to be more susceptible than hardwoods to hurricane damage (Foster 1988, Foster and Boose 1992, Jalkanen and Mattila 2000), though there are differences in susceptibility between taxa related to several species characteristics, including crown form, branching, rooting depth, center of gravity, and wood strength. Species susceptibility to hurricane-related damages also varies across types of damage, including breakage, uprooting, exposure to saltwater, and secondary insect and disease damage (Barry and others 1998). In addition, trees planted outside their natural range may be more susceptible to damage.

Differences in susceptibility to ice storms are related to species characteristics such as crown form, crown size, branch thickness, branch angle, root depth, and wood strength. Trees that have an excurrent form (apical dominance) are better able to shed ice and snow than those with decurrent form, and the vase-shaped form of many elms and oaks makes these trees especially vulnerable to ice and snow accumulation (Van Dyke 1999). Trees with branches that are pliable may better withstand ice and snow accumulation because they can shed ice or transfer the force to other parts of the tree, the ground, or neighboring trees. However, wood that is cold, “green,” or less dense has lower

resistance to breakage than warm seasoned wood of the same species (Bragg and others 2003).

Root and stem rots, cankers, insect infestation, or any form of prior injury increases the probability of wind damage (Wilson and Baker 2001) and ice damage (Van Dyke 1999) (with the possible exception of defoliating insects or other agents that reduce crown surface area). Trees are likely to break at the point of decay when exposed to excessive loading by wind, ice, and snow. For instance, trees with advanced beech bark disease experienced more ice damage (Rhoads and others 2002). Vine coverage may also increase susceptibility to glaze damage by increasing the surface area available for ice accumulation (Bragg and others 2003). Boose and others (2001) examined hurricanes affecting New England since European settlement in 1620 and found that forest damage from an event was strongly dependent on natural disturbance history. High proportions of compression wood, which can form in response to tree bending in past weather events, may increase storm damage (Dunham and Cameron 2000).

Stand age and tree size may also contribute to forest vulnerability to high winds. Although age and size are generally correlated, different studies alternately focus on age, height, diameter, or H/D ratio. Many studies indicate that susceptibility to wind damage increases with increasing stand age during the initial decades of forest development (Everham and Brokaw 1996, Francis 2000, Jalkanen and Mattila 2000). Foster (1988) found that conifers' susceptibility to wind damage increases rapidly after age 15, but hardwood damage increases more gradually with age. Foster and Boose (1992) found wind damage increased approximately linearly with stand height. Ameteis and Burkhart (1996), on the other hand, argued that older, larger trees are stronger and have greater resistance to damage.

Francis (2000) and Greenberg and McNab (1998) also reported increasing uprooting with increasing tree height. Francis (2000) showed that tree height strongly influenced wind damage in San Juan during Hurricane Georges. Other authors suggested that height relative to the height of surrounding trees is the factor that most strongly determines susceptibility to wind damage (Asner and Goldstein 1997, Francis and Gillespie 1993, Hedden and

others 1995). Jalkanen and Mattila (2000) and Francis and Gillespie (1993) both showed that the susceptibility of a stand to wind damage was increased by larger mean diameter. Achim and others (2005) found that stem mass was the factor most highly correlated with wind damage. Francis and Gillespie (1993) related gust speed to tree damage in Hurricane Hugo, and their results indicate that the probability of a tree suffering some form of damage increases with increasing tree diameter.

Findings were mixed for ice damage. Bragg and others (2004) found the lowest levels of damage from ice in the oldest plantations. They found that the smallest trees were much more likely to bend severely in response to ice loading, that intermediate-sized trees had both bending and stem breakage, and that the largest trees had little of either but had more crown damage and branch loss. Rhoads and others (2002) found heavy ice damage in intermediate-sized trees that were 24 to 28 years old but little ice damage in younger stands in the affected forest. Proulx and Greene (2001) found that very small trees (3.2 to 9.5 cm in diameter) were more likely to suffer damage, that small and intermediate trees (<17.8 cm in diameter) were more likely to bend or snap, and that large trees were more likely to lose branches. Van Dyke (1999) indicated that older trees may be more susceptible to injury from ice because they have larger crowns, greater likelihood of internal decay, and decreased branch flexibility.

A number of studies have examined H/D ratios rather than either height or diameter individually and have typically found that larger H/D ratios tend to increase the risk of damage from high-wind events (Wilson and Baker 2001), snow (Kato and Nakatani 2000), and ice accumulation (Van Dyke 1999). However, Valinger and Fridman (1997) found that the ratio of the H/D ratio to the upper diameter (measured at 3 or 5 meters) was the best predictor of the probability of snow and wind damage, and that for a given upper diameter, the probability of damage was higher for a site with trees of low H/D.

In addition to looking just at tree height, diameter, or H/D ratio, some studies examine the influence of crown shape and canopy position. Mechanistic models of critical

wind, ice, and snow loading generally include crown size and shape because greater crown surface area increases the drag coefficient and potential for loading from accumulation (Peltola and others 1999, Talkkari and others 2000). Empirical observations and models have generally supported this assumption. Foliated trees and trees with large crowns have more surface area for ice and snow accumulation and for wind to act on, increasing vulnerability to damage; however, damage is lowered by greater crown symmetry (Bragg and others 2003, Kato and Nakatani 2000, Van Dyke 1999). Beese (2001) found greater wind damages in trees with larger crowns in montane coastal forests in British Columbia. Hardwoods are less susceptible than conifers to damage from ice or snow storms that occur during the winter months, when the hardwoods do not have foliage. Mitchell (1995) and Ruel (1995) reported that wind damage depends on canopy density and tree position in the canopy. Van Dyke (1999) indicated that dominant trees are more likely to suffer ice damage.

Stand Characteristics

Disturbances that open the canopy may increase the severity of damage from subsequent windstorms (Everham and Brokaw 1996). Stand density, thinning regime, and the presence of edges and gaps each have been found to influence forest damage from severe weather. However, the effects of density on forest damage depend on the interplay of a complex set of factors. Higher densities may lead to greater H/D ratios and trees that are less wind resistant. However, higher stand density may increase the probability of stem bending but decrease the probability of stem breakage because neighboring trees provide support for bending stems (Bragg and others 2004). Wind may move through low-density stands more freely, so that energy is dissipated and wind loadings are lowered. In addition, trees in the interior areas of low-density stands may normally receive higher winds and develop greater wind resistance than those in high-density stands. However, ice damage may be expected to increase at lower stand densities because trees in lower density stands develop larger crowns, and these tend to accumulate more ice and snow (Gardiner and Quine 2000).

It is also possible that bending or breakage of trees results in damage to other trees. When windthrow or breakage occurs in lower density stands, there is less chance that neighboring trees will be damaged (Baker and others 2002). Beese (2001) found that striking of trees by other trees was one of the most common categories of wind damage in their study region.

Recent thinning can increase vulnerability to wind, ice, and snow. Thinning removes structural support of proximate trees, and recently exposed pines tend to have relatively weak stems and ice-accumulating surfaces more concentrated in the crown (Bragg and others 2003). Burner and Ares (2004) found that thinning increases susceptibility to ice damage. Jalkanen and Mattila (2000) suggested that thinning will increase wind damage but decrease snow damage. Peltola and others (1999) simulated critical snow loading for both uprooting and stem breakage for managed and unmanaged stands of Scots pine (*Pinus sylvestris* L.) in southern Finland and concluded that critical snow loading for both uprooting and breakage is lower for unthinned stands than thinned stands.

Proximity to forest edge is highly related to wind (Wilson 2004) and ice storm damage (Olthof and others 2004). Zeng and others (2004) integrated a mechanistic wind damage model and an airflow model with a forest database containing information at the tree, stand, and regional levels and simulated damages for current forest edges and situations where new edges might be created through clearcutting. They found that clearcuttings increased windspeeds at forest edges and increased risk of damage for cases where vulnerable stands remained at the newly created edges. Others have found reduced damage at stand edges, however, especially when the edge has been established for a long time. This is likely due to trees on the edges facing greater wind loading on a normal basis and responding by increasing strength to resist wind loading (Foster 1988).

Management under Endogenous Risk

The probability of a severe weather event occurring is exogenous to the landowner; that is, landowners are unable to influence this probability through their own actions. However, although these events cannot be prevented, their

potential impact can be mitigated through forest management, so the extent of severe weather damages to a particular forest area is partially under the landowner's control. Recognizing this endogeneity of risk and understanding the factors that influence damage patterns are of key importance to efficient forest management. Although there is substantial variation in the empirical findings on the most important factors influencing weather damages, there are some common management recommendations. Landowners can incorporate expectations of severe weather damages into their decisions regarding site selection, species selection, silvicultural treatments, and planning for damage recovery to decrease expected impacts (Bragg and others 2003, Lohmander and Helles 1987, Olofsson and Blenow 2005, Persson 1975, Zeng and others 2004). Another strategy for the forest industry is to diversify spatially to manage risk across total forest holdings thereby avoiding the loss of a large share of their standing timber to a single weather event.

Moore and Quine (2000) presented one example of the ability of forest management to mitigate weather risk. They compared wind risk in Sitka spruce (*Picea sitchensis* (Bong.) Carr.) plantations in Great Britain with that in radiata pine (*Pinus radiata* D. Don) plantations in New Zealand using the FORESTGALES simulation model. They reported that the plantations in New Zealand were at greater risk of damage than those in Great Britain because of differences in management even though the New Zealand plantations were subjected to a less severe wind regime. In Great Britain, silvicultural practices have been adopted to reduce density, conduct more careful site preparation, and reduce thinning. This comparison demonstrates the influence of forest management on the risk of weather damage, but it also demonstrates that mitigation options adopted should reflect the level of risk faced. Managers of plantations in New Zealand have been able to focus on maximizing profit under a less constraining wind risk, and plantations in New Zealand are more profitable than those in Great Britain.

Of course, appropriate response depends on the degree of risk. In cases where the exogenous risk of damages is low, incorporation of weather risk into strategy may result in minimal or no change in optimal forest management.

In contrast, there may be no feasible strategy for reducing damages when the exogenous risk of damages is high other than simply limiting investments and therefore the expected cost of damages. In the intermediate case, where the exogenous risk of damages is significant but there are cost-effective strategies for mitigation, the incorporation of weather risk into decisionmaking may substantially alter optimal forest management practices (Gardiner and Quine 2000).

This implies that any evaluation comparing weather damages on managed and unmanaged sites will be inaccurate if it does not properly account for selection bias. Selection bias arises because the forest management strategies observed reflect decisions made by forest managers based in part on the perceived vulnerability of their stands to weather-related damages and the perceived effectiveness of potential silvicultural actions in those stands. Thus, the value of managing weather risk could be understated if these actions are selectively being undertaken in stands with high expected damages and the outcomes following a storm are compared with those in areas with low expected damages where managers chose to adopt fewer mitigation options. On the other hand, the effectiveness of damage reduction strategies may be overstated if little or no action to reduce impacts is being taken in certain stands because the expected damages in those locations are so large that there are few feasible mitigation options. Thus, careful and systematic program evaluation is needed to account for endogenous selection and to ensure that the effectiveness of damage mitigation options is being assessed after controlling for selection of mitigation options based on perceived baseline risk (Butry 2006).

Damage Mitigation Options

The primary categories of damage mitigation options are site and stand selection, selection and timing of silvicultural treatments, and stand recovery activities. Forest management is unlikely to significantly alter climate, topography, or soils in a given site. However, forest managers establishing a new stand can affect these conditions for forests they manage through site selection. In addition, owners with multiple sites can manage overall risk by selecting a portfolio of sites

that offers the desired risk-return profile. Typically, weather risk is not the overriding consideration in the selection of sites for forest investments, but taking the risk of forest damages into account when selecting a site may improve the ability to meet management objectives.

Similarly, among numerous other factors affecting choice of species (e.g., productivity, market forces, objectives of the forest manager), the differential weather risk associated with different species should be taken into account. Some species are more vulnerable to weather damage than others based on crown configuration, rooting, wood strength, and many other factors. In addition, planting trees outside of their natural range may increase their vulnerability to damage (Bragg and others 2003). Selecting a tree species that is less productive or has a lower market price but also provides a reduced risk of severe weather damage may result in a higher expected present value of the stand if the baseline risk of severe weather and the species-to-species difference in expected damages in the event of severe weather are sufficiently large.

The effects of management decisions about stocking, thinning, and pruning on storm damage risk have been identified as key issues in a number of studies. A number of trade-offs are associated with these activities, partly because short-term and long-term effects on risk may differ substantially. For instance, low-density stocking, thinning, or pruning may increase the risk of damage in the short term, but expected damages are reduced farther into the future as trees respond by growing stronger and more resistant to damages. Limited tree-size variation increases susceptibility to developing high H/D ratios in the dominant trees (Wilson 1998). There is typically less variation in tree size in plantations than in naturally regenerated stands, and this may lead to greater H/D ratios of dominant trees, reducing the stability of these trees. To keep H/D ratios down, it is necessary to conduct thinning operations before competition that leads to increased H/D (Wilson and Baker 2001). Similarly, Wilson and Oliver (2000) found that plantation H/D ratios can be lowered by reducing planting density or early thinning to encourage development of strong stems and crowns; later thinning did not appear to be as effective. Also, if the rotation length is reduced

sufficiently, the reduction of time at risk of damage may more than offset the increase in expected damages if an event does occur. One of the best defenses against weather damages may be to grow the trees out of vulnerable intermediate sizes as quickly as possible through the application of silvicultural practices such as reduced density and proper thinning and pruning to increase site productivity.

The standard Faustmann equation maximizes discounted net revenue from an infinite series of rotations, accounting for the costs of maintaining land in forest production. The optimal rotation under certainty occurs where the increase in stand value from one additional period of growth is equal to the costs of delaying current and future harvests by one period. Incorporating an exogenous positive probability of damages into the equation unambiguously yields a reduction in optimal rotation length (Martell 1980, Reed 1984, Reed and Errico 1985, Routledge 1980). Under some conditions, increased production risk may also lead to the use of lower intensity production methods such as naturally regenerated stands (Haight and others 1986). Accounting for endogeneity of expected forest damages alters optimal management strategies by explicitly recognizing effects of management decisions on the expected present value of the stand through accounting for changes in expected damages.

Stocking influences, in several ways, damages that result from weather events. Less densely stocked stands may be more vulnerable in the short term because they will have less support from their neighbors. However, less dense stocking increases the growth of individual trees and reduces the time trees require to reach marketable size. In addition, low stand density will result in trees that are more resistant to loading because they are larger and subject to greater loading on a normal basis. One caveat is that low stand density commonly results in trees with larger, less tapered crowns that will tend to accumulate more ice and snow, although their increased strength may offset this effect.

Thinning may result in stronger trees that are more able to withstand wind, ice, and snow (Kato and Nakatani 2000) a few years after thinning but increase the likelihood of damage in the interim. Coates (1997) found that partial

cutting doubled the percentage of trees with wind damage in the 2 years after thinning, from 1.1 percent to 2.2 percent but concluded that this increase in damage was too small to warrant changes in management (this increased damage may also be offset by increased tree growth rates that reduce the time spent at risk). Similarly, pruning can increase wind damage if a storm passes through the area soon after the treatment but can reduce damage in the future after trees have strengthened. Pruning was found to result in an increased resistance to snow damage in the short term, though, because the reduced surface area of the crown decreases accumulation. However, to the extent that heavy pruning reduces diameter growth more than height growth, resistance to snow loading may decline in the years after pruning because of increased H/D ratios (Kato and Nakatani 2000).

Thus, in areas that receive severe weather very frequently, managers may be less willing to conduct thinning or pruning because of the high probability that a storm will occur during the period when the stand is more vulnerable. In addition to removing support for remaining trees, thinning operations often cause minor damage to individual trees, and the damage sites can serve as entry points for insects and disease that then become established and infest the stand (Lohmander and Helles 1987). However, there is a reduced probability of trees damaging their neighbors following thinning because the spacing between trees is increased (Bragg and others 2003). In addition, identifying and removing trees most vulnerable to damage during thinning operations may be advantageous. Cameron and Dunham (1999) found that trees that were damaged by wind and snow had significantly more compression wood than did undamaged trees, and Dunham and Cameron (2000) therefore recommended preferentially removing trees likely to contain compression wood.

Because of the complexity of the interactions between various factors affecting expected forest damages, the relative costs and benefits of a particular action will vary spatially and temporally. Thus, optimal management strategies regarding planting density and whether, when, and how intensely thinning and pruning should be conducted need to be determined on a site-specific basis.

If a plan for stand recovery is formulated before a damaging event takes place, this can help mitigate the damage. Salvage efforts after ice storms should concentrate on minimizing economic losses from damage to timber. Timber subject to losing the greatest value to degradation in the short term should be salvaged first, although timber of all ages and sizes may require some remediation to lessen fire danger and insect outbreaks. Salvage of severely damaged pine is more time sensitive than salvage of severely damaged hardwoods because pine is more susceptible to infestation by bark beetles and wood borers, but downed hardwoods should also be salvaged as soon as possible to avoid degradation (Van Dyke 1999). Remedial actions following hurricanes, such as staking saplings upright, are not clearly effective and are feasible only on small areas. In one study, saplings left in a leaning position did not lose measurably more growth than those staked upright, and many righted themselves (Gresham 2004), although it is important to recognize that bent trees may have permanent internal damage. In some cases, severely bent trees will not fully recover and may die or grow at greatly reduced rates. Managers may wish to explore selective harvesting of damaged trees.

Optimal Thinning and Rotation Decisions

As an illustrative example of the implications of treating weather damage risk as endogenous, we examine optimal thinning and rotation decisions. The implications of production risk for optimal rotations have been examined in several previous studies, including Martell (1980), Routledge (1980), Reed (1984), and Reed and Errico (1985), all of which evaluated changes in optimal rotation length under the exogenous risk of forest fires. Each found that optimal rotation age unambiguously declined where production risk was assumed to exist. Thorsen and Helles (1998) modified these earlier models to treat risk as endogenous, focusing on the timing and intensity of thinning activities to influence the risk of damage. Thinning is one of the silvicultural practices that is most frequently mentioned in the literature as playing an important role in damage risk and as being readily applied.

Thinning offers the benefit of immediate income, and it promotes faster diameter growth of remaining trees, which increases net price received and reduces the risk of windthrow. Thinning also reduces the remaining volume that could potentially be damaged by storms. However, thinnings are costly because of fixed costs of thinning operations and increased risk of storm damage for several years after thinning. These dynamics are complex, but important to consider in determining optimal forest management strategies.

Thorsen and Helles (1998) examined an even-aged forest stand managed to maximize expected present value. They modeled the selection of an optimal thinning strategy and optimal rotation age (at which time the stand is clear-cut), taking into account the stand growth function, net timber price, and risk of severe weather damage. Stand density, age, site, tree, and stand characteristics, including time since thinning and thinning intensity, all affect the risk of damage. The damage risk functions describe how the stand's stability is impacted by current and past stand management decisions. The less the risk functions are affected by management decisions, the more resistant the stand is to disturbances such as thinning, and the less important it is to consider the implications of management decisions for weather damage risk. Thorsen and Helles (1998) used numerical optimization procedures to determine optimal management in the case of Norway spruce (*Picea abies* L. Karst.), which they identified as one of the species most often suffering from health or windthrow problems that necessitate salvage harvest of entire stands. Based on their assumptions regarding growth, risk dynamics, storm probability, and other variables, they found that optimal rotation age is reduced from 74 years under no risk to 69 years if storm risk is treated as exogenous, but 71 years when risk is assumed to be endogenous. In addition, the optimal number of thinnings increases from 10 to 11 between the case of no risk and exogenous risk. When risk is considered to be endogenous, the optimal number of thinnings increases to 18, but the intensity of each thinning is reduced. In addition, they found that incorporating endogeneity in determining optimal management increases expected present value of the stand.

The findings of this study are dependent on the growth, risk, and other dynamic relationships assumed. Optimal management will differ across stands depending on baseline damage risk, effects of thinning on growth rates and damage risk, prices and costs, and other variables (Straka and Baker 1991). In addition, this example assumes profit-maximizing behavior, which may be appropriate for forest industry but not for many nonindustrial private forest (NIPF) landowners or public forest managers. NIPF landowners and public forest managers are expected to consider the amenity value of standing forests, which would add an additional term measuring amenity value to the optimal control problem discussed above. This term would depend on site, tree, and stand characteristics. Nevertheless, treating severe weather risk as endogenous may enable both private and public managers to reduce damage risk while increasing the expected present value of their stands.

Policy Implications

To the extent that weather damage risk is endogenous, private and public forest managers may be able to improve ecological and economic returns by allowing for the effects of their decisions on risk. However, this does not tell us whether any particular strategy should be adopted in the case of any specific stand. Options to be adopted depend on managers' objectives and the cost of the option relative to the reduction in risk provided. Many key factors interact in a very complex manner to influence weather risk, making it vital to consider case-specific conditions in making decisions regarding mitigation of weather risk. It is important to consider joint risk of different types of weather events, which may reinforce or contradict each other.

Policy implications are expected to differ among forest industry, NIPF landowners, and public forest managers because these groups have different objectives. Forest industry managers are expected to be profit maximizing, whereas NIPF landowners likely have more varied objectives and value nontimber amenities of standing forests more highly. Damage mitigation options that would be optimal for forest industry may be less suitable for NIPF lands or vice versa, depending on the joint effects of the strategies on damage risk, expected present value, and stand amenity

values. Similarly, public land managers are expected to take various nontimber objectives into account. The presence of endogenous storm risk may be an opportunity for the public sector to provide incentives for private landowners to take actions to mitigate storm damage, especially where private inaction could result in secondary insect and disease infestations or other conditions that could have negative effects on neighboring properties.

Private Forest Management Decisions

For forest industry, management of weather risk should be selected to maximize expected profitability, assuming risk neutrality. Under risk neutrality, firms are interested in maximizing the expected value of profits without regard for variability, whereas risk-averse firms would be willing to accept lower expected returns in exchange for reduced variability of profits. Decisionmaking under risk aversion could be analyzed using similar methods but would have to account for risk tolerance. One strategy when operating under risk is to reduce rotation age. Accounting for damage risk reduces the optimal rotation age relative to that obtained by solution of the standard Faustmann equation under certainty (Gardiner and Quine 2000, Haight and others 1996). However, if risk is truly endogenous, then managers may be able to alter practices to mitigate risk and potentially even increase expected present value of a stand, as described in "Optimal Thinning and Rotation Decisions." Selective removal of specific high-risk trees when thinning, as well as adjustment of thinning timing and intensity could reduce risk of storm damage. However, it must be possible to identify and remove these trees cost effectively. Heavily damaged or dead trees are obvious candidates for removal, but it may be worth selectively removing those trees with features that increase their risk, such as large, asymmetric crowns, and trees that have been heavily bent.

Numerous other management strategies, including site selection, species selection, and planting density, could be modified to mitigate risk and should be adopted if they offer expected benefits exceeding their costs. Each of these strategies and combinations could be examined through empirical simulations using a model similar to that described in "Optional Thinning and Rotation Decisions."

To optimally incorporate these strategies into forest management practices, it is necessary to understand the implications of each for the dynamic risk of storm damage in the event of a storm at different points in time, forest growth rates, and stand value.

For NIPF landowners, behavior may be more complex because the perceived impacts on aesthetics, wildlife, and other nontimber amenities provided by standing forests are a more important part of their management objectives. These landowners are likely to have substantially different optimal management policies as a result, probably tending toward fewer thinnings and increased rotation age relative to land managed by the forest industry, although this may vary depending on the primary objectives of particular NIPF landowners.

Public Policy and Forest Stand Management—

Management of public lands to mitigate weather risk also is likely to differ from the profit-maximizing forest industry case. Wilson and Baker (2001) argued that managing stands to allow greater future flexibility may be more important in the case of public lands than in that of private lands because objectives of public land management are more likely to change over time with shifting political pressures. It is important to develop stands that are capable of producing multiple outputs, from timber to older forest habitat, when future objectives are uncertain. Like NIPF landowners, public land managers are likely to consider aesthetics, wildlife, recreation, and other nontimber products in determining optimal forest management strategies.

In addition, the presence of endogenous forest risk and the extensive losses that could result from a major event could lead public agencies to become involved in providing incentives for improved private forest management, especially to avoid secondary impacts such as insect and disease infestations that may negatively affect neighbors. Private landowners will not fully account for the damages that may result on neighbors' lands and may underinvest in storm damage mitigation relative to the socially optimal level. This negative externality may be an argument for government intervention to correct this market failure through subsidies of mitigation practices, by establishing penalties

for not following recommended practices, or by establishing regulations requiring specific practices. However, it is important to construct such programs with care to ensure that they result in net increases in forest investment and improvements in forest health and do not simply offset private investment that would occur in the absence of the policy.

In Switzerland, short-term damage resulting from storm Lothar was similar across cantons (states) even though different cantons employed significantly different forest protection measures (Bisang and Zimmerman 2006). More expensive measures adopted in some cantons were not much more effective in preserving forests and preventing forest damages than much less expensive options. In addition, forest owners' salvage decisions appeared to be independent of the financial incentives provided by cantons. Forest owners spent more of their private resources in cantons that provided minimal financial support, and their marketing of salvaged timber was less successful in these cantons than in those that provided greater public resources. Thus, the varying strategies for public involvement in providing financial incentives for forest management had important distributional effects for forest owners and government spending but not for forest health (Bisang and Zimmerman 2006).

Conclusions

Our review characterizes primary influences on damage risk as site, tree, or stand characteristics, each of which is influenced by forest management decisions. Findings about key factors differ from study to study, and this indicates both the complexity of storm damage and the substantial variation in data and methods used to conduct analyses. A few studies simultaneously incorporate many of the factors identified in the literature as having an important influence on damages; most focus on only a few variables and do not necessarily report statistical significance. Thus, it is unclear just how broadly applicable the findings of many studies are. This review reveals a need for additional empirical research to better quantify the effects of various factors influencing the risk of forest damages, particularly in the case of complex stands that are unlikely to be

captured adequately through standard mechanistic models. Future studies should place greater emphasis on empirical analyses that examine the effects of variables believed to affect damages and their interactions simultaneously. Another important issue is the need for more systematic program evaluation that accounts for endogenous selection of damage mitigation options to better identify the effects of management actions while controlling for differences in underlying conditions.

Our review indicates that some general management practices are likely to affect risk and that forest managers must consider the influence of their management decisions on the extent of weather damages. Site and species selection, as well as choice and timing of silvicultural treatments, affect expected damages. In the event of damage, the effectiveness of management plans for storm recovery is another key determinant of economic impacts and secondary damages, although management planning for storm recovery is not a focus of this chapter. Thus, optimal stand management where damage risk is treated as endogenous is likely to differ from management where risk is treated as completely exogenous. Treating risk as endogenous results in more frequent, less intense thinnings and a longer rotation age than when risk is treated as exogenous, though both cases have shorter optimal rotation ages than when there is assumed to be no weather risk (Thorsen and Helles 1998). This finding depends on a number of assumptions regarding the change in expected damages associated with thinning, however, and may differ across stands depending on their baseline risk of damage, effects of thinning on tree growth rates, timber price, thinning costs, and other variables. Developing better parameterizations of these relationships is an important area for additional research to better understand optimal decisionmaking under endogenous risk for a variety of climate and forest characteristics.

Management practices should be selected on the basis of the risk faced and management objectives. In the case of the forest industry, risk-neutral firms should undertake damage mitigation measures only if the increase in the expected present value of the stand exceeds the cost of the mitigation option. For NIPF landowners, management decisions under endogenous risk are generally expected to be more complex

because of objectives other than profit maximization. Thinning regime and rotation age have implications for nontimber amenities provided by forests. Therefore, to the extent that NIPF landowners have objectives that include the production of such amenities, management by NIPF landowners will differ from management where the goal is profit maximization. Incorporating the nontimber amenities provided by standing forest will probably tend to reduce the optimal number of thinnings and increase rotation age, although this may vary depending on the primary objectives of particular NIPF landowners (Foster and Orwig 2006). Similarly, the implications for public land management under endogenous risk are expected to differ from the profit-maximizing case based on the varied objectives of public agencies managing forest land.

Recognition of the endogenous nature of forest damages and the large potential losses could induce public agencies to provide incentives for private landowners to engage in improved management practices. This is particularly true when damages may lead to secondary impacts such as insect infestation or forest fires that could spread to neighbors' land. Landowners do not fully account for impacts on neighbors or invest at socially optimal levels in damage mitigation and stand recovery measures, and this provides a rationale for potential government intervention. However, it is important to construct such programs with care to ensure that they result in net increases in forest investment and improvements in forest health rather than simply offsetting private investment.

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Economic Impacts of Hurricanes on Forest Owners

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Abstract

We present a conceptual model of the economic impacts of hurricanes on timber producers and consumers, offer a framework indicating how welfare impacts can be estimated using econometric estimates of timber price dynamics, and illustrate the advantages of using a welfare theoretic model, which includes (1) welfare estimates that are consistent with neo-classical economic theory, and (2) wealth transfers among various market participants that can be evaluated. Timber producers in the Southern United States are faced with the regular risk of damages from intense hurricanes. Individual events can kill several million cubic feet of standing timber, with attendant losses for forest owners. One result of using a welfare theoretic model that is not apparent using simpler models is that timber producers with undamaged timber suffer economic losses in the short term because of a price depression, and they may be compensated in the long term by an enhancement of market prices owing to the loss of standing inventory. Catastrophic storms induce losses to timber producers holding damaged timber owing to quality degrade, price depression, and the inability to salvage all of the damaged timber. To minimize decay-related losses, owners of damaged timber should salvage as quickly as possible and favor salvage of higher value trees. Owners of undamaged timber should delay harvesting until salvage wood is exhausted from the market. Evidence suggests that timberland investors under hurricane threat could benefit by diversifying their holdings geographically, favoring areas far enough from coastal counties to minimize catastrophic losses from such storms but close enough to benefit from market-level price enhancements resulting from regional inventory losses.

Keywords: Economic welfare, Hurricane Hugo, Hurricane Katrina, timber salvage, tropical cyclone.

Introduction

This paper provides a description of the hurricane process in the United States in the context of economic impacts on forest land owners, presents the welfare theoretic model used to quantify timber market impacts, describes and compares the impacts of a few large and recent hurricanes on timber markets in the South, and concludes with some recommendations for landowners based on the research. This section focuses on measurement of catastrophic storm impacts on landowners.

More than 150 hurricanes have struck the U.S. Atlantic and Gulf of Mexico coasts during the past century, and these storms have caused economic damages totaling billions of dollars (Blake and others 2007). Human loss of life and suffering, destruction of housing, and disruptions in economic activities are the most obvious impacts of catastrophic storms, and recent, large hurricanes are proof of their ability to economically and politically transform affected regions. States vulnerable to the most intense storms include Florida, Texas, North Carolina, and South Carolina, but all States in the South and some States in the mid-Atlantic region and the Northeastern United States are also vulnerable to hurricanes.

Less appreciated, perhaps, are the effects that catastrophic storms have on forests and the people who own them. The effects of hurricanes on landowners are a complex function of many factors, including storm severity (windspeeds), prior rainfall activity (whose effect in turn is modified partially by soils), the strength of the timber market, tree species affected, and timber decay rates. Overall effects on landowners from hurricanes depend on hurricane frequencies and intensities. Given these variables affecting damages, the long-term effect of hurricanes will depend on long-term trends in climate and the timber market.

Research has shown that individual storms can create forest damages into the billions of dollars, and those damages create widespread wealth redistributions. The timing, scale, and implications of these losses and benefits have been the subject of research by the authors of this paper and by others. We have learned that measuring the impacts

Table 1—Hurricane landfalls, 1851–2006, and timber growing-stock volumes and production, 2002

State	All categories (1-5)	Major hurricanes (3-5)	Standing growing-stock volume, 2002 <i>Million ft³</i>	Timber, production 2002 <i>Thousand ft³</i>
Florida	113	37	15,366	560,475
Texas	60	19	12,939	769,501
Louisiana	52	20	18,844	958,981
North Carolina	50	12	32,742	957,675
South Carolina	30	6	17,702	682,901
Alabama	26	6	27,847	1,298,533
Georgia	23	3	31,704	1,447,941
Mississippi	16	9	20,611	1,149,880
Virginia	10	1	26,487	654,829
New York	12	5	21,831	141,068
Connecticut	11	3	3,192	11,691
Massachusetts	11	3	5,732	26,437
Rhode Island	9	4	496	1,742
Maine	6	0	20,891	441,729
Maryland	2	0	5,092	40,507
Delaware	2	0	695	7,654
New Jersey	2	0	2,819	10,549
New Hampshire	2	0	9,015	140,282
Pennsylvania	1	0	24,903	215,912

Note: State-by-State information about the number and intensity ranges of hurricanes, where they made landfall (1851–2006), and the total current (nondamaged) amount of growing stock and timber production by State (for 2002) are given. Hurricane intensities are represented using Saffir-Simpson categories.

Sources: Blake and others (2007, p. 18); Smith and others (2004).

on landowners can be accomplished using standard economic concepts of supply and demand in combination with econometric analysis of timber price dynamics following a catastrophic event. This approach is consistent with welfare economic methods described by Just and others (1982) and as applied to forest impacts by Holmes (1991). These concepts have been used by the authors to measure the effects of Hurricane Hugo on prices, producers, and consumers (Prestemon and Holmes 2000, 2004). These techniques have also been used to quantify the timber market impacts of large-scale fires (e.g., Butry and others 2001, Prestemon and others 2006). The implications of these findings suggest that landowners can take steps before and after such events to reduce exposure to the risk process and to mitigate damages when subjected to it.

The Biophysical Risk Process of Atlantic Basin Hurricanes in the United States

This section provides a discussion of biophysical risk processes associated with timberland ownership in the Eastern and Southern States of the United States. Atlantic hurricane arrival rates are available from the National Oceanic and Atmospheric Administration, which are reported here to provide the scope of the issue for forest landowners. Table 1 and Figure 1 show the number of hurricanes, by Saffir-Simpson scale category, striking each State in the Atlantic Basin of the United States, 1851–2004. Table 1 can be used to infer the probability of a hurricane striking a specific State by dividing the number of years represented by the number of landfalls (although a spatially weighted probability listing would look different). This table shows that there is a coincidence of high hurricane frequency and high timber volumes. This coincidence is apparent when the

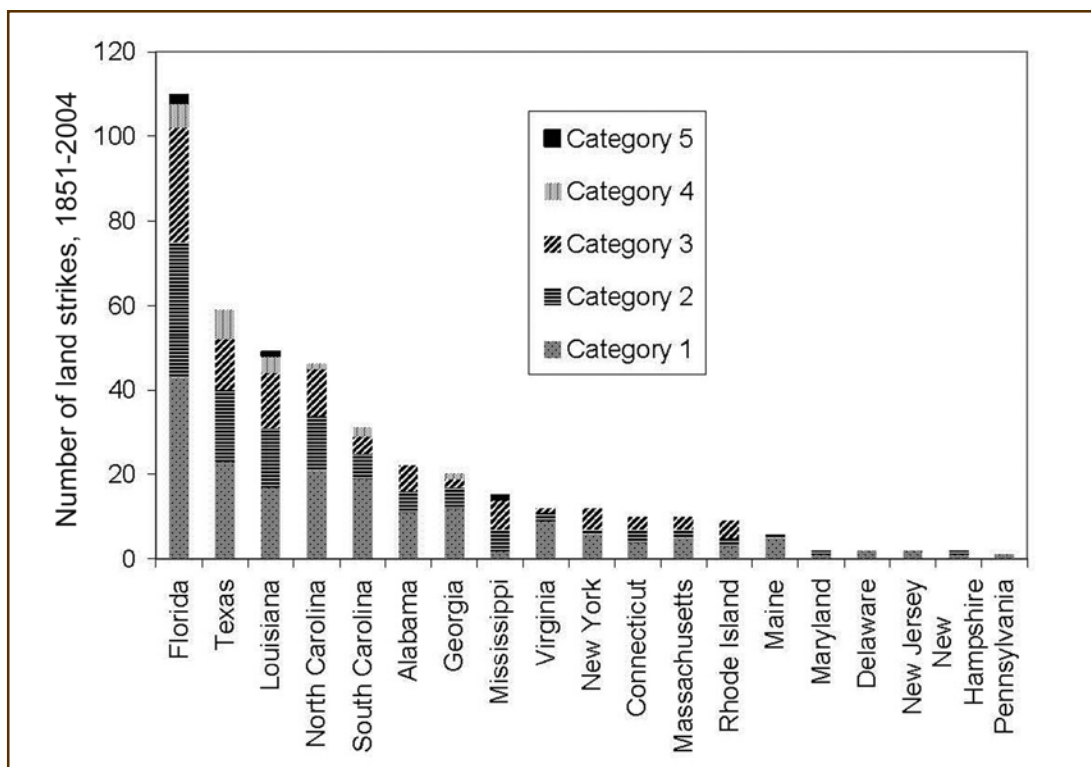


Figure 1—Atlantic hurricane strikes, 1851–2004, by State by Saffir-Simpson Category. Source: National Oceanic and Atmospheric Administration (2006).

States are ranked by hurricane landfall. The States with the greatest number of hurricanes are Florida, Texas, Louisiana, and, then, North and South Carolina. The strongest timber markets are found along the gulf coast, so the southern timber resource is particularly threatened by severe losses from large-scale wind events. In other words, timber owners in these high-frequency States are at particularly high risk of timber loss. See Stanturf and others (2007) for additional details about timber and hurricane frequencies and the United States Department of Commerce, National Hurricane Center (2007) for information on hurricane return intervals.

Data from the National Oceanic and Atmospheric Administration (2006) show that the number of hurricanes in the United States has recently demonstrated an upward trend. These data are consistent with Emanuel (2005) who indicated that hurricane intensities in many parts of the world appear to have risen over the past several decades. Both of these trends are consistent with some predictions of anticipated effects of climate change (e.g., Knutson and

Tuleya 2004) and with recent global ocean temperature rises (Barnett and others 2001). Whether or not the recent increase in hurricane activity is due to climate change (Gray 1984a, 1984b; Gray and others 1996) or primarily associated with multidecadal Atlantic Ocean sea surface temperature oscillations is still being debated. What is clear is that the Southern United States is facing a period in which substantial damage to timber resources from hurricanes can be anticipated, along with the associated consequences for forest landowners.

Timber Market Dynamics Following Hurricanes

This section provides information on (1) timber market shifts following natural disturbances, (2) resulting timber price movements through time, (3) welfare movements through time, and (4) salvage recovery values following natural disturbances. This will provide the reader with an overall picture of timber market behaviors and offer intuition about expected price movements and production

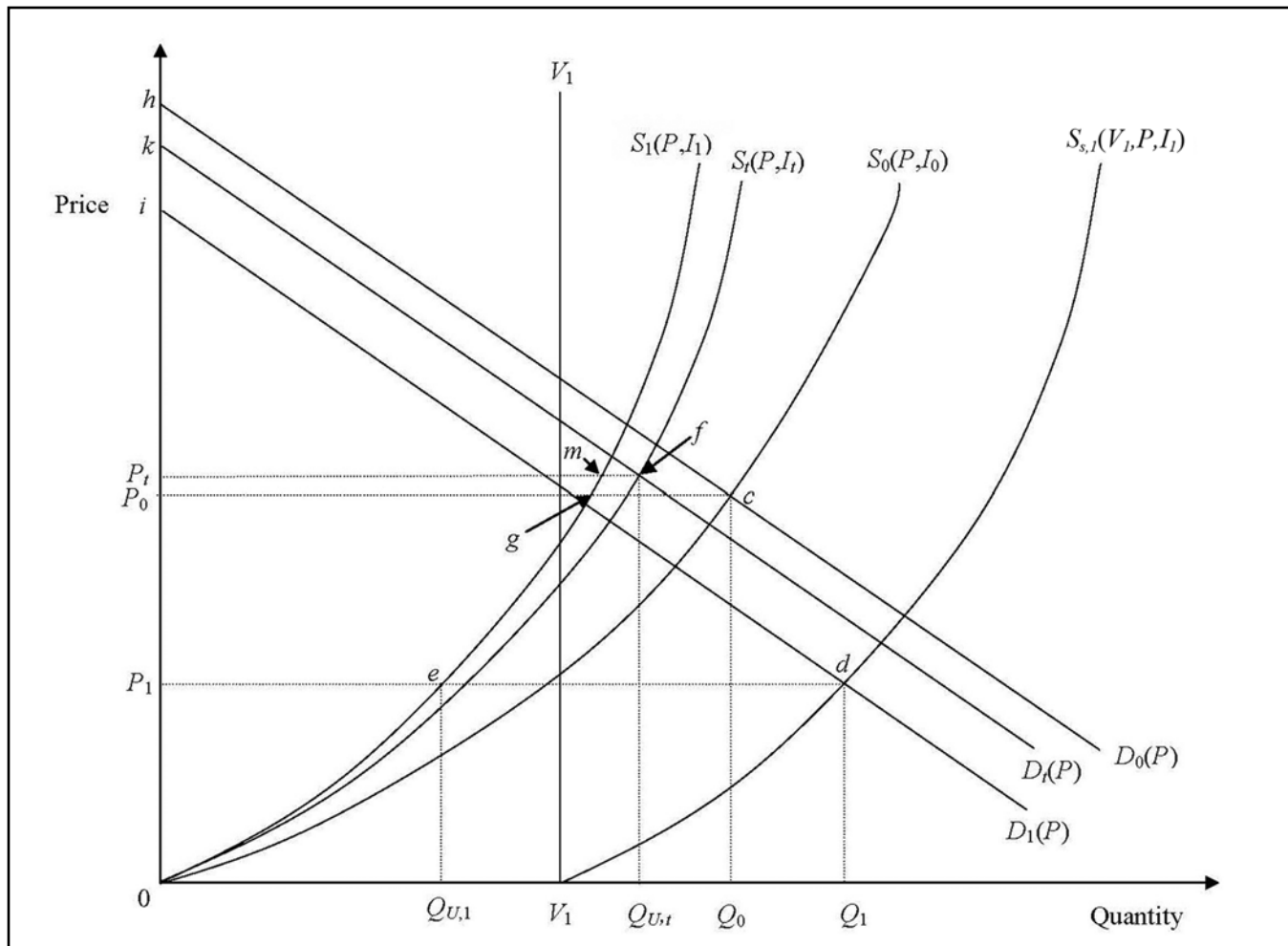


Figure 2—A representation of the timber market following a natural disaster such as a hurricane. The primary effect of hurricane damage to timber is to temporarily shift supply outward (from S_0 to $S_{s,1}$) during salvage and then shift back to S_1 due to salvage exhaustion and inventory loss. As inventory (I) regrows, supply shifts outward toward S_0 , and an intermediate position may be S_t . Demand, initially at D_0 , may shift backward to D_1 because of mill closures and higher market prices; intermediate demand level is at D_t during inventory regrowth. Price paths over time can be traced to these market shifts. Lower-case letters have been added at intersection points to facilitate detailed discussion in the text.

shifts in the aftermath of large-scale catastrophes affecting timber.

Graphical Overview of Market Shifts

Following a large hurricane (or any natural disaster), a timber market may undergo substantial changes in supply and demand. In Figure 2, supply of timber is represented as S_0 initially and is a function of price (P) and available inventory (I_0). Demand, also a function of price, is the curve sloping down from the left, and initial levels are represented by the curve D_0 . The initial (prestorm) price is designated as P_0 , and quantity is Q_0 . Following the storm, the timber

supply curve shifts inward to S_1 due to a reduction in available standing live inventory of undamaged timber (I_1). However, in the immediate aftermath of the storm, a pulse of timber salvage, V_1 , enters the market. This supply is zero when price is zero—i.e., it is a single point—but jumps to a large positive amount when price exceeds zero (Holmes 1991; Prestemon and Holmes 2000, 2004). In other words, the supply curve parallels the horizontal axis between 0 and the quantity V_1 . Any stand with fatally damaged timber with nonzero stumpage value is part of salvage supply. The combination of the inventory effect and the salvage effect is

to create a total market supply of $S_{s,1}$. Again, as in the case of the salvage supply, $S_{s,1}$ consists of a single point at (0,0) but jumps to V_1 when price slightly exceeds zero. During this salvage period, the equilibrium price is P_1 , lower than the prestorm price. When the pulse of salvaged timber is exhausted, the supply curve reverts to S_1 , with a higher price (P_1) than prior to the storm. As inventory grows over time, the supply curve gradually shifts back out to the original position of S_0 , achieving the original price and quantity at equilibrium. At that point, assuming no investment shifts owing to the storm, the effects of the hurricane disappear from the market.

In addition to impacts on timber supply, hurricanes might also impact timber demand. Many mills were closed because of either hurricane damage to mill facilities or anticipated long-term timber shortages following salvage operations after the passage of Hurricane Hugo. Syme and Saucier (1992) uncovered some evidence of short-term mill closures. In that case, demand might shift back (to D_1), as capital is reallocated to other, undamaged regions or out of the wood processing sector. The degree of backward shift in demand identified by Prestemon and Holmes (2004) was slight. Such a backward demand shift tended to dampen any price swings following Hugo. However, it could be that, with time, demand expands in response to timber inventory regrowth, allowing for prices that are not as low (e.g., P_1) as they would be if demand did not respond.

Timber Price Movements Through Time

Timber prices following a hurricane proceed through three phases: (1) price depression, (2) price enhancement, and (3) the gradual return of timber prices to prestorm levels, as implied by the discussion of the graphical representation of the hurricane's impact on a timber market. First, immediately following the storm, a price depression occurs, as was identified by Yin and Newman (1999) and Prestemon and Holmes (2000) following Hurricane Hugo. The finding of a price depression is consistent with what Holmes (1991) found in the case of southern pine beetle (*Dendroctonus frontalis* Zimmermann), and in all cases it is due to timber salvage activity, which drives down the market price owing to the supply expansion.

Prestemon and Holmes (2000) also found that there was a second stage involving a higher set of prices for timber products owing to the shortage of timber inventory following Hugo's destruction. This is the enhancement effect identified by them statistically, consistent with theory and with local markets where timber is expensive to move. Such an effect has been observed in other circumstances, as well (Berck and Bentley 1997, Olson and others 1988). In the third stage, timber returns to the prestorm level, when the inventory returns to normal levels.

Figure 3 illustrates timber price dynamics following a large-scale natural catastrophe. This figure shows a brief, negative price spike for timber because of salvage, a longer term price enhancement owing to inventory shortages, and then a gradual price reduction to precatastrophe levels as inventory regrows. Both a timber price index (where the precatastrophe timber is normalized at an index value of 100) and a timber inventory index (where the precatastrophe inventory volume is normalized at an index value of 1.00) are shown, indicating that 25 percent of timber inventory is killed. The figure is built upon assumptions of a constant demand elasticity with respect to timber price of -0.5 and constant supply elasticities with respect to price and inventory of 0.3 and 1.0, respectively. (These elasticities represent the percentage change in timber supplied to the market, given a 1-percent change in price or inventory volume, respectively). It also assumes that (1) 30 percent of timber killed is salvaged, (2) it takes 4 quarters (a quarter is 3 consecutive months) to exhaust the salvage, (3) the salvage quantity is highest immediately following the event and tapers off linearly following the event, and (4) the salvage quality discount factor is 5 (i.e., a unit of salvage timber has a quality discount of 80 percent).

Prestemon and Holmes (2000) conducted an intervention analysis of southern pine sawtimber and pulpwood stumpage (timber) prices, modeled after a similar analysis conducted by Holmes (1991). The approach was to model timber prices as autoregressive-integrated-moving average processes, including an intervention component that captured the effect of the hurricane on South Carolina timber prices. Comparison of the South Carolina price processes to price processes for the same product in other locations

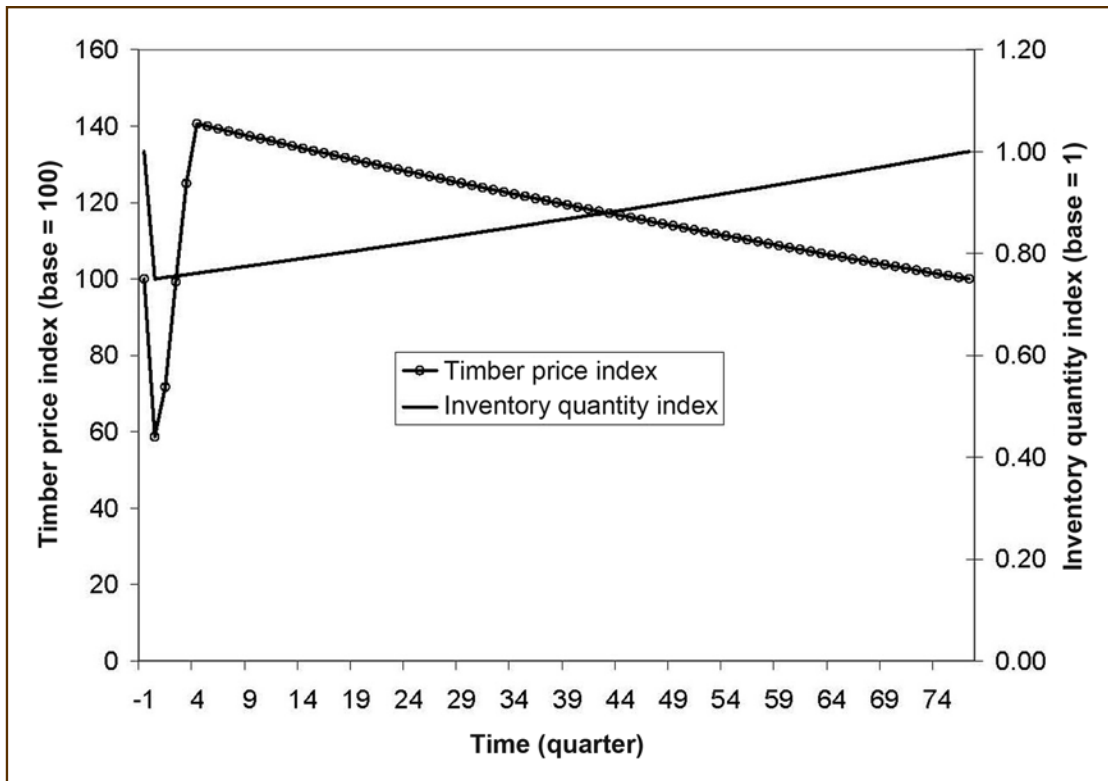


Figure 3—A hypothetical price path over time for timber following a large-scale natural catastrophe. In this figure, prices in the quarter immediately preceding the catastrophe (quarter -1) are at 100 and inventory is at 1.00. In the quarter of the catastrophe, inventory drops to about 0.75 and prices fall to 60 as salvage volume moves into the timber market. With salvage exhaustion, the inventory shortage effect is felt in timber prices, as they are temporarily elevated to as high as about 140. With time, prices recover to their original level as inventory returns to its initial volume.

in the South allowed identification of the full price effect of the storm, in both the short and long terms. As Holmes (1991) pointed out, statistical identification of the market price effects of such an event requires a sufficiently long series before and after the modeled event in order to identify derivative patterns in the prices owing to the event. Prestemon and Holmes (2000) found in their intervention analysis that salvage induced an average 30-percent decline in green (undamaged) pine pulpwood and pine sawtimber stumpage prices during the salvage period and that the inventory loss induced a persistent 10- to 30-percent enhancement (increase) in those prices after salvage was exhausted. This finding relied on tests that confirmed the nonstationarity of timber prices in South Carolina and identification of significant co-integration relations between South Carolina prices and those of other States. Nonstationarity of the State's timber prices has been also confirmed using longer

time series, inclusion and exclusion of time trends, alternative deflation methods, and alternative tests of stationarity by Prestemon and others (2004). Using a different approach that relied on an assumption of trend-stationary timber prices in South Carolina, Yin and Newman (1999) identified no long-run price enhancement in South Carolina.

Welfare Movements through Time

This section describes how timber damages, salvage, and price shifts result in effects on different segments of the timber market. Results from published research are used to illustrate these shifts. Prestemon and Holmes (2004) traced out the economic equity (welfare, or economic surplus) impacts of hurricanes and similar large-scale catastrophes in an analysis that is based on methods applied by Holmes (1991) and those of Just and others (1982). Thurman and Easley (1992) showed how, when an event occurs in a

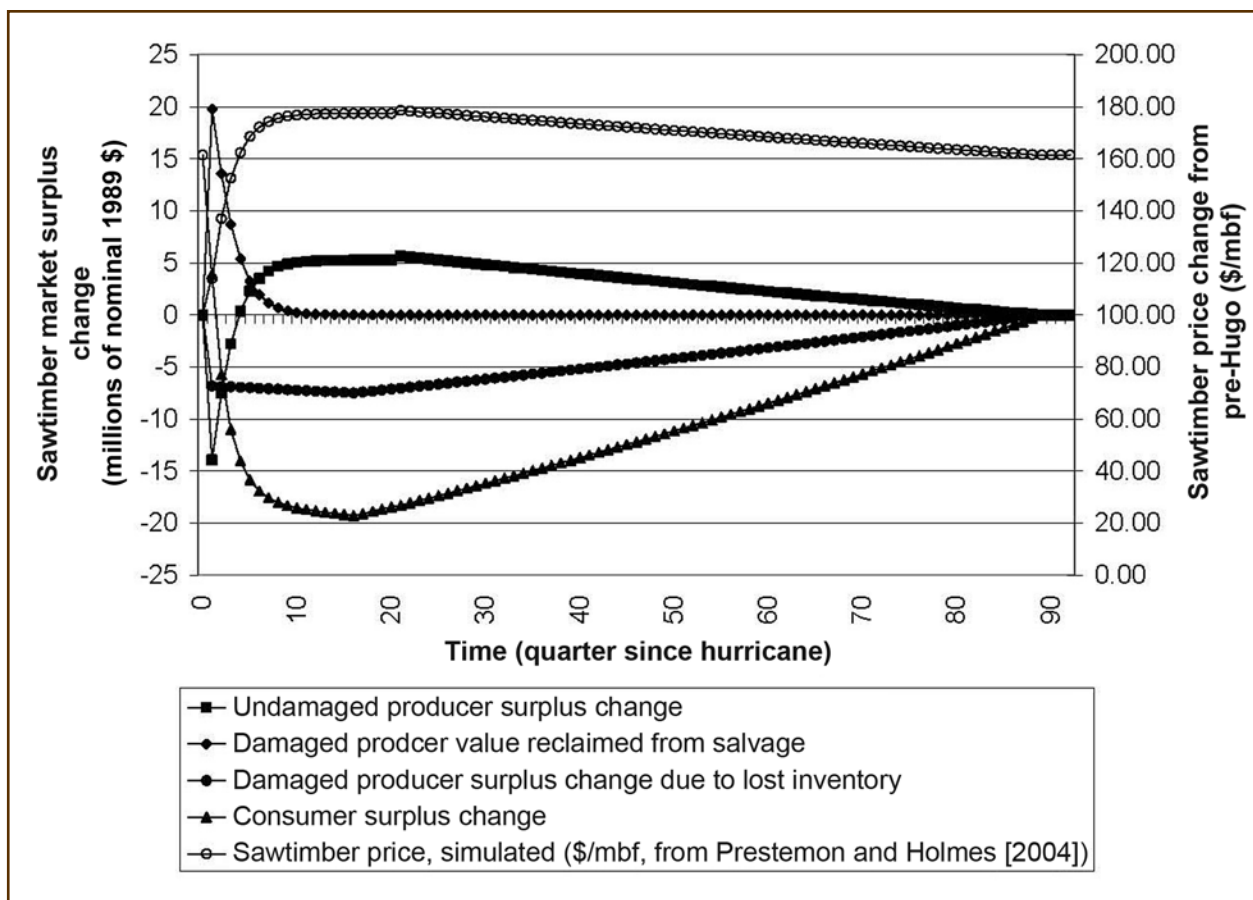


Figure 4—Identified price and welfare shifts in the Timber Mart-South (Norris Foundation, 2004–2006) Region 2 market, South Carolina, immediately before and after Hurricane Hugo, for southern pine sawtimber, by quarter. Figure based on the base case scenario data from Prestemon and Holmes (2004). [mbf = million board feet].

primary product market (such as timber), a welfare analysis of the economic impacts of the event on the primary market will capture the effects on all higher levels of processing deriving from that market. Given this, in our study, the full timber market economic impacts can be partitioned across owners of damaged timber, owners of undamaged timber, and timber consumers. This requires definition, however. Consumer surplus is defined, in Figure 2, as the area below the demand curve, above the equilibrium market price, and to the left of the equilibrium quantity supplied to the market. For example, before the hurricane, the consumer surplus is area P_0ch , while in the immediate aftermath of the storm, it was area P_1di . Producer surplus of undamaged producers is the area above and to the left of the supply curve and below price. Before the storm, producer surplus

was P_0c0 . In the immediate aftermath of the storm, producers of undamaged timber receive surplus represented by the area P_1e0 , but as timber salvage is exhausted, say in period t , this subset of producers actually gains, with their surplus valued as $0fmP_t$. The producer surplus gained by damaged producers upon salvage of their timber by the storm is equivalent to salvage volume (e.g., V_1) times the price received during the salvage period (P_1). This is the area represented by the rectangle P_1dQ_10 . In the immediate aftermath of the storm, then, these undamaged producers, because they cannot produce undamaged timber, also lose an area represented by a curved shape bounded by 0 , S_1 , S_0 , and demand curve, D_1 , the shape represented by the shape $0cgt$.

Figure 4 illustrates how changes in southern pine timber market economic surplus, salvage revenues, and timber

price varied over time following hurricane Hugo, as identified by Prestemon and Holmes (2004). In their simulations, inventory regrowth is projected, based on Forest Inventory and Analysis surveys in 1993 and 1999 in South Carolina. The rate of regrowth is a key determinant of the quantity and timing of post-hurricane welfare impacts. The figure shows that consumer welfare increased briefly following the hurricane because of lower prices and greater quantity consumed. Undamaged producers were briefly harmed from this salvage, but damaged producers captured some value for a few quarters. Consumers were harmed over the long term, undamaged producers gained over the long term, and damaged producers lost in the long term. In this figure, prices were determined to return to prestorm levels in the sawtimber market by 2012, about 90 quarters (23 years) after the storm. A similar price and welfare path was also traced out by Prestemon and Holmes (2004) for the southern pine pulpwood market there.

As shown, all of these losses and gains proceed into the future until the timber inventory returns to the prestorm inventory volume and timber production and price levels. The long-term discounted economic surplus lost and gained by the various market participants is tracked by tallying these changes for all periods into the future, discounting the values using an economic discount rate to the present. In a later section, we describe those changes for Hurricane Hugo in a table.

Salvage

This section describes how the value of salvage is calculated and some of the factors influencing the value recovered by salvaging timber. Readers interested in more detail about the economics of the salvage decision should review Haight and others (1996), whereas those interested in operational aspects of salvage should see Stanturf and others (2007). Our research and research conducted by others (e.g., Lowell and others 1992), indicate that the value of salvage depends on several factors: (1) the severity of initial damage (function of wind, prestorm rain); (2) the amount of timber decay, which is a function of tree species, the post-event ambient air temperature, the post-event ambient humidity, the dimensions of the materials, and time; (3) the aggregate

volume of salvage entering the timber market; and (4) market supply and demand sensitivities with respect to price and market supply sensitivity with respect to inventory. The species, temperature, humidity, log dimension, and time all govern the current value of the salvage discount factor, K . K is used to translate the salvage volume into a value-equivalent “green” timber volume for market valuation purposes (see Holmes 1991):

$$Q^s = Q^D K^{-1},$$

where Q^D is the raw quantity damaged or killed and Q^s is the value-equivalent of the timber in “green” timber volume. As K increases, the value-equivalent reduction in timber volume increases.

What the research implies is that hurricanes (which occur generally in warm, humid areas of the United States and cause a great amount of physical damage to the internal log structure from wind) produce values of K that are large. Prestemon and Holmes (2004) found that K likely averaged about 4.5 for southern pine sawtimber following Hugo and 6.6 for southern pine pulpwood. Events such as fires (e.g., Lowell and others 1992) and southern pine beetle outbreaks (e.g., de Steiguer and others 1987) produce smaller values for K (in the range of 1.1 to 1.5 in the months immediately following the event).

Forest and Timber Impacts of Recent Hurricanes

The need for landowners and policymakers to plan for effects of increased hurricane activity is discussed in this section, and comparative information about damage levels of recent vs. older hurricanes is provided. Recent hurricanes, including Ivan and Frances in 2004, and Katrina and Rita in 2005, provide evidence of the timber market impacts of these events, including their impacts on landowners. These can also be compared with the impacts of older hurricanes, such as Hugo in 1989 and Camille in 1969, to help us understand how changes in timber prices as well as volumes affected may be leading to rising economic impacts. Below, we focus on the kinds of losses experienced by owners of timber in terms of hardwood and softwood, pulpwood, and sawtimber-product categories.

Table 2—Timber damage volume and dollar impacts from six severe hurricanes affecting selected States in the United States Atlantic Basin

	Units	Camille, 1969 ¹	Hugo, 1989 ¹	Frances, 2004 ²	Ivan, 2004 ²	Katrina, 2005 ¹	Rita, 2005 ²
Softwood timber mortality							
South Carolina	million ft ³		1,008				
Florida	million ft ³			87	208		
Mississippi	million ft ³	216				619	
Louisiana	million ft ³					287	296
Alabama	million ft ³				603	126	
Texas	million ft ³						239
Hardwood timber mortality							
South Carolina	million ft ³		319				
Florida	million ft ³			58	94		
Mississippi	million ft ³	74				426	
Louisiana	million ft ³					193	177
Alabama	million ft ³				414	91	
Texas	million ft ³						293
Total timber mortality							
South Carolina	million ft ³		1,327				
Florida	million ft ³			145	302		
Mississippi	million ft ³	290				1,044	
Louisiana	million ft ³					480	473
Alabama	million ft ³				1,017	216	
Texas	million ft ³						532
Total, measured affected States, all species	million ft ³	290	1,327	145	1,319	1,740	1,005
Softwood portion of damaged volume	Percentage	74	76	60	61	54	53
Timber value damaged (price × quantity)							
Sawtimber value lost	Millions of 2005 dollars	177	494 ³	153	2,094	1,600	504
Pulpwood value lost	Millions of 2005 dollars	25	136 ³	9	83	113	32
All products	Millions of 2005 dollars	202	630 ³	162	2,177	1,713	536
Timber economic surplus							
Producer surplus lost, all owners and periods	Millions of 2005 dollars		121 ³				
Consumer surplus lost, all periods	Millions of 2005 dollars		1,402 ³				
All market groups	Millions of 2005 dollars		1,523 ³				

¹ Final USDA Forest Service Forest Inventory and Analysis estimates of volumes of timber mortality. Economic impacts in dollars based on these final estimates.

² Initial USDA Forest Service Forest Inventory and Analysis estimates of volumes of timber mortality. Economic impacts in dollars based on these initial estimates.

³ Dollar values shown for Hurricane Hugo are for softwood timber only.

Sources: Surplus values for Hurricane Hugo are from Prestemon and Holmes (2004); volumes for Camille are from Van Hooser and Hedlund (1969); pine values are from Mississippi Forestry and Agricultural Experiment Station (2006); baldcypress and hardwood values are from and Louisiana Department of Agriculture (1969); volumes for Frances, Ivan, Katrina, and Rita are from USDA Forest Service Forest Inventory and Analysis (2004a, 2004b, 2005a, 2005b), respectively; and values are from Norris Foundation (2005–2006).

The 2005 hurricane season was the worst, in terms of timber damages, in a generation—much bigger than the largest in the last two decades, Hurricane Hugo (Table 2) (USDA Forest Service, Forest Inventory and Analysis 2005a, 2005b). Hurricane Katrina damaged 1.74 billion cubic feet of timber, and about 41 percent of the volume was hardwood. Other estimates, not reported here, put long-term damages much higher than this. Hurricane Rita added to those damages, so when combined, these two storms damaged at least twice as much timber in the affected States of Alabama, Louisiana, Mississippi, and Texas as was damaged by Hugo in South Carolina in 1989 (although those South Carolina damages exclude damages in North Carolina, another affected State).

The market value of these timber inventory losses for Katrina and Rita was quantified in a special report provided to the Forest Service, expressed in terms of 2005 dollars to account for inflation. For this review, we also quantified the same values for Hurricane Frances (which affected Florida) and Hurricane Ivan (which affected both Florida and Alabama) in 2004. These economic effects are reported at the bottom of Table 2, principally in market-value terms. In Table 2, we also include the economic effects of two other large Hurricanes, Camille (1969) and Hugo (1989). At the bottom of Table 2, we include information for Hurricane Hugo, addressing both market value and economic surplus, as calculated by Prestemon and Holmes (2004). Market values can be termed timber value losses, whereas economic surpluses can be termed economic losses. We note that the welfare impacts for softwood calculated by Prestemon and Holmes (2004) depend on assumptions regarding market supply and demand parameters and timber regrowth rates, and they depend on the validity of the price paths identified by Prestemon and Holmes (2000). Prestemon and Holmes (2004) conducted Monte Carlo and sensitivity analyses on these damages, which report ranges of possible impacts. The welfare amounts shown in Table 2 for Hugo are average estimates.

Hurricane Frances is the smallest hurricane impact shown in the table, with 145 million cubic feet of damage. Camille damaged twice as much timber as Frances, but it still is small compared to the rest of the hurricanes shown

in the table. For example, Hurricane Ivan created timber damages that are comparable to those measured for Hugo. Rita created damages that totaled a billion cubic feet, about 75 percent the size of Hugo. The amount of softwood lost in these storms varied, mainly according to the amount of pre-existing softwood in the timber inventory. In the South, the amount generally exceeds 50 percent.

The values of timber damaged or killed in the storm are variable, as documented in Table 2. One surprising finding from this comparison is that Ivan's timber damages exceed the value of those found for Katrina. This is explained by the substantially higher timber prices in effect in Florida and Alabama prior to the hurricane, in comparison with the timber mainly damaged in Mississippi and Louisiana. It is also partially explained by the former hurricane's higher proportion of damage to softwood and sawtimber-sized trees. Still, when Rita's damages, which totaled over \$536 million, are added to those of Katrina, those two storms damaged volumes that are more than twice those of Hugo or Ivan.

Hugo's timber value lost in the southern pine market, measured as price times quantity, was \$630 million. The analysis by Prestemon and Holmes (2004) showed, however, that the market impacts in welfare terms were 2.4 times larger. This magnitude of difference between market value and welfare impact should be viewed with caution when attempting to translate between market-value losses to welfare losses for other storms or catastrophic events. Even for Hugo, this translation depended on many assumptions, especially regarding (1) timber salvage rates, market supply and demand sensitivities; (2) rates of timber inventory regrowth; (3) the reality of a long-term inventory-induced price enhancement; and (4) the scale of timber market demand and supply. Nonetheless, a rough calculation could be attempted that would suggest the economic surplus impacts of those other storms are much larger—maybe two to three times larger—than the timber value impacts shown. In short, recent hurricanes have created market losses totaling into the many billions of dollars. As Prestemon and Holmes (2004) and Holmes (1991) pointed out, these losses represent a net redistribution of wealth in the market,

in favor of undamaged producers and to the detriment of consumers and damaged producers.

There are obviously other potential outcomes of hurricanes on the broader economy as they relate to the timber market. Structure (e.g., housing) losses are quite high when large events such as Hurricane Katrina sweep through heavily populated areas. The National Oceanic and Atmospheric Administration (2006) indicated that hurricanes Katrina, Rita, and Wilma in 2005 combined to create nearly 2.8 million insurance claims, totaling \$40 billion in insured losses. Data provided by the American Red Cross, as cited by the National Association of Home Builders (2005), compiled preliminary estimates indicating that Hurricane Katrina damaged 275,000 homes in the Gulf States of Louisiana, Mississippi, and Alabama, 10 times the second-largest loss in homes, which was caused by Andrew in 1992 in south Florida. The National Association of Home Builders (2005) report stated that Katrina, coupled with Rita and Wilma (in Florida), would drive up prices for southern pine framing lumber, plywood, and other sheathing products. This price effect on forest products should filter back to the timberland, helping to support prices in the Southern United States during the reconstruction phase. The implication here is that landowners whose timber was unaffected by the storms should enjoy some price support, increasing the chances for a post-hurricane price enhancement, even if that enhancement does not emanate directly from lost timber inventory in the region.

Management Implications

We now discuss the future of hurricane activity in the United States and the implications of the existence of hurricane risk on land management strategies, with a focus on timber salvage. What we can glean from the previous discussion is that the impacts of hurricanes on landowners will be partitioned according to the location of the timberland. Hence, owners closer to the impact zone—typically, closer to the ocean—will face a higher likelihood of timber damages and long-term negative impacts. Those farther from the coast may face lower impacts and a greater chance that they could enjoy a long-term price enhancement owing to timber inventory losses and, potentially, output effects on

the construction market. Due at least partly to the species mixes and particular species vulnerabilities, it appears also that owners of softwood timber may face greater economic risks than owners of hardwood timber, but this is a hypothesis that remains to be tested in the literature. Hardwood damages frequently occur as degrade, rendering the stand lower in value in the long term and potentially vulnerable to a higher rate of mortality (Sheffield and Thompson 1992).

As we appear to be in a period (or riding a trend) of higher hurricane activity, we might expect that the frequency and magnitude of large-scale catastrophic hurricanes will increase or remain high for several years or decades. In response to damage, timberland owners would do well to prepare for such storms by assessing the value of the timber before the storm. Creditable casualty losses require that the current basis of affected timber be assessed. Following such storms, timberland owners whose timber is unaffected would benefit from withholding their mature timber from the timber market until after salvage material has been recovered from damaged forests and then consumed at area mills. As well, owners of damaged timber should endeavor to quickly arrange for the salvage of their timber, owing to the high likelihood of timber decay for killed timber. Here, some colloquial evidence suggests that hardwoods should receive lower priority, as they may remain alive and hence valuable for a longer period, compared to softwoods, which are more likely to die following a storm (Sheffield and Thompson 1992). Because of the rapid decay rate of killed timber, then, it is especially beneficial to quickly salvage such timber following the event, especially timber stocks with the least amount of damage, to justify salvage.

Government actions following catastrophic windstorms may have an effect on the value of salvaged timber. For example, timber salvage on publicly owned lands has a negative impact on private landowners but a positive impact on consumer welfare (Prestemon and others 2006). There are, however, government actions that can produce broader benefits. Governments can facilitate salvage activities by (1) rapidly clearing roads, (2) relaxing weight limits on roads to allow logging trucks to carry heavier loads, (3) permitting larger than normal log storage volumes at mills,

(4) subsidizing or providing tax breaks for mills to produce water storage facilities for logs, (5) aiding in post-hurricane damage assessment, and (6) assisting private landowners and nongovernmental organizations with salvage planning (Freeman 1996, Lupold 1996). However, we would caution that research is currently not available to help evaluate either the form or the scale of public interventions that would be economically or socially optimal.

Research Needs

We conclude our synthesis of the impacts of hurricanes on forest landowners with a discussion of the research needed to further advance our understanding of these impacts and to identify alternative strategies for coping with these storms. This assessment reveals a long list of potential research needs. A primary need in all timber market assessments of hurricanes is up-to-date and precise information on green timber supply, salvage supply, and demand function parameters. Our results for Hurricane Hugo depend critically on assumed market sensitivities, and conclusions about the ultimate economic (especially welfare) impacts of other events depend on similar assumptions that need to be evaluated. We would note that for published salvage studies (starting with Holmes 1991), assumptions were made about the perfect inelasticity of the salvage supply function. This function is unlikely to be perfectly inelastic (unresponsive to prices). Although sensitivity analysis is useful for establishing plausible bounds on potential economic impacts, up-to-date studies of southern pine supply and demand functions would reduce the need for making assumptions regarding market conditions. Additionally, new studies are needed to identify how hurricanes affect markets for products other than southern pine. A large component of total timber impact results from damages to hardwoods, yet hardwood markets are notoriously under-researched. Such studies could reveal not only the hardwood market economic impacts but also potentially identify new and innovative strategies that owners of hardwoods could apply before and after hurricanes and other catastrophic events.

Timber salvage is a means of recovering value following such storms, yet little is known about how various

factors affect the size of the timber salvage discount, particularly how it varies over time and among species. Such studies could provide valuable information regarding optimal salvage strategies following major hurricanes. Our economic calculations—and, hence, recommended salvage strategies—could be improved and changed with better information on this segment of the post-hurricane timber market.

Other analysts in these proceedings provide evaluations of the timber management consequences of holding timber assets that are subject to large-scale natural disturbances. However, research to date has not addressed whether long-term price enhancements emanating from a hurricane or other large natural disturbance can generate expected profits that are, in aggregate, greater than those expected in regions without such events. Timber losses occur in a subset of the timber market—consumers and damaged producers—but the gains could be significant for some landowners. Identifying the circumstances under which timberland owners gain, in aggregate, from these storms is an area worthy of further study.

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AIR and WATER

Case Studies

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Using Remotely Sensed Data and Elementary Analytical Techniques in Post-Katrina Mississippi to Examine Storm Damage Modeling

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Abstract

Hurricane Katrina's passage through south Mississippi on August 29, 2005, which damaged or destroyed thousands of hectares of forest land, was followed by massive salvage, cleanup, and assessment efforts. An initial assessment by the Mississippi Forestry Commission estimated that over \$1 billion in raw wood material was downed by the storm, with county-level damage percentages ranging from 50 percent to 60 percent across Mississippi's three coastal counties. Remotely sensed data were used to provide a more complete picture of the damage inflicted by Katrina. Moderate (56- to 29-m) and high (1- to 0.3-m)-resolution data were acquired from spaceborne and airborne platforms in natural color and MultiSpectral (MS) formats. Transformed data such as Normalized Difference Vegetation Index (NDVI) and Normalized Difference Moisture Index (NDMI), along with damage estimates obtained by interpreting aerial photography, were also used as variables in a linear modeling process. This continuous damage prediction process demonstrated the effect of incorporating forest condition thematic information, prestorm moderate-resolution imagery with transforms, and poststorm moderate-resolution imagery with transforms. The resulting models, all of which used a large number of regressors, had overall fit values of $R^2_{\text{adj}} = 0.708$ and $\text{RMSE} = 0.130$ with all variable types used, $R^2_{\text{adj}} = 0.492$ and $\text{RMSE} = 0.172$ with all variables except the forest condition data, and $R^2_{\text{adj}} = 0.599$ and $\text{RMSE} = 0.153$ with all variables except the poststorm imagery data.

Keywords: AWiFS, damage modeling, forest damage, hurricane damage, Katrina damage, timber damage.

Introduction

Overview

Hurricane Katrina made landfall in Mississippi near the outlet of the Pearl River on August 29, 2005, as a category 3 storm on the Saffir-Simpson scale (Knabb and others 2005). Loss of life and damage to property were catastrophic, as New Orleans was flooded, and many towns and cities along the Louisiana and Mississippi Gulf Coasts were destroyed or severely affected. Rural areas whose economy depends on agriculture and forest industry were devastated also. Accordingly, preliminary damage estimates obtained through aerial surveys of the affected region by the Mississippi Forestry Commission (MFC) exceeded \$1 billion in damaged wood and timber stumpage. These estimates underscore the need for more continuous damage estimates that can be developed when remotely sensed, storm, and pre-existing thematic data are employed in the modeling process.

Moderate-resolution remotely sensed data, from sources such as Landsat, have been used in modeling various forest parameters related to timber harvesting (Healey and others 2005), canopy closure (Butera, 1986, Cohen and others 2001, Larsson 1993), and other forest attributes (Cohen and others 2001, 2003). Cohen and others (2001) modeled percentage green canopy cover in a predominantly evergreen softwood region of the Pacific Northwest, similar to south Mississippi, with a coefficient of determination (R^2) of 0.74 and a root mean squared error (RMSE) of 12 percent. Similarly, Healey and others (2005) used a series of transformations on independent variables as well as a natural logarithm transformation on percentage cover, the dependent variable, in a series of simple linear regressions to determine adequate univariate models. The results of their work were promising with regard to using single Short Wave InfraRed (SWIR) bands, as well as the Normalized

Difference Vegetation Index (NDVI) and Normalized Difference Moisture Index (NDMI) transformations described in the methods section of this work.

Damage assessments of past catastrophic tropical storms in the Southeast United States were not able to use the large number of image sources and processing techniques that are presently available. The use of geographic information systems (GISs) and remote sensing techniques in these assessment activities were, for the most part, restricted to a minor role in the wake of Hurricane Hugo in South Carolina (Nix and others 1996) with an expanded role noted for Hurricane Andrew (Jacobs and Eggen-McIntosh 1993, Ramsey and others 1997, 2001). With these two storms and the studies mentioned, a progression of technology and techniques can be noted. In Nix and others (1996), remote Hugo forest damage assessments were made in a GIS through aerial photointerpretation and digitization. Jacobs and Eggen-McIntosh (1993) also used visual image interpretation to perform assessments of Hurricane Andrew-induced damage with the imagery taking the form of airborne digital video frames. The two works led by Ramsey (Ramsey and others 1997, 2001) show a final evolution to satellite acquired moderate (Landsat)- and coarse (Advanced Very High Resolution Radiometer or AVHRR)-resolution imagery, along with storm data, in identifying Andrew's damage in a largely hardwood area in south Louisiana.

Objectives

Because the MFC assessment was performed rapidly through aerial viewing using expert approximation, a more definitive and continuous damage assessment model was sought. In our work here, we studied the viability and possible methods needed to develop predictive storm damage assessment models. The acquisition and analysis of remotely sensed data acquired before and after hurricane Katrina, along with various storm data and pre-existing thematic data created for the Mississippi Institute for Forest Inventory (MIFI), were used to determine the feasibility of mapping storm impacts in a more accurate and continuous form.

Beyond characterization of Katrina-induced forest resource damage, we explored model development to predict the likely scope and severity of damage from future hurricanes. This procedure involved use of MIFI forest thematic data, storm data, prestorm imagery (which can be simulated to note the effects from different size and intensity storms), and poststorm imagery to determine their relative importance in producing predictive damage models. The implications here involve two aspects: (1) model performance without poststorm data so that predictive equations can be used to forecast future hurricane damage, and (2) level of predictive model improvement afforded by use of forest type and age thematic layers that accompany inventory protocol employed by MIFI.

Methodology

Remotely Sensed Data

Acquiring Remotely Sensed Data—

Moderate-Resolution Data

Indian Remote Sensing (IRS) Advanced Wide Field Sensor (AWiFS) data were acquired for use in pre- versus post-storm damage assessment as a moderate spatial resolution (56 m at nadir) data source. These data were selected for several reasons:

- Relatively high visibility, although minor cloud coverage was noted in both pre- and poststorm images.
- Compatibility of spatial resolution with Landsat Thematic Mapper (TM) data, which were used in the creation of thematic data also used in this study.
- Acquisition dates of June 19 (prestorm) and September 4 (poststorm), less than one week after Katrina's landfall in Mississippi.

Spectral attributes associated with the AWiFS sensor included four bands representing the following reflected energy wavelengths, respectively: green (520-590 nm), red (620-680 nm), near-infrared (NIR) (770-860 nm), and shortwave infrared (SWIR) (1550-1700 nm).

High-Resolution Data

We used prestorm digital imagery taken throughout the summer of 2004 (before Katrina's landfall) that was

acquired through the United States Department of Agriculture's (USDA) National Agriculture Inventory Program (NAIP) and made available via the internet by the Mississippi Automated Resource Information System (MARIS). These data were acquired in natural color, sampled at a spatial resolution of 1 m, and presented as mosaiced county-level images. Within 2 months of Katrina's landfall, a private contractor using a Leica ADS40 sensor provided post-Katrina digital imagery for the U.S. Army Corps of Engineers (USACE) over south Mississippi from 31° N to the Gulf Coast. These poststorm data were made available by the United States Geological Survey (USGS) via a disaster-support Web site. They were acquired in natural color with an approximate 0.3-m spatial resolution.

Preprocessing Remotely Sensed Data—

Using the existing thematic MIFI data as a georegistration base, the AWiFS data were georegistered in Leica's ERDAS Imagine 8.7 using first- or second-order polynomial models (ERDAS 2003) in order to achieve subpixel spatial root mean squared (RMS) values. The resulting products were thus projected into the Mississippi Transverse Mercator (MSTM) (MARIS 2005), as this was the native projection of the MIFI base data. This procedure, as recommended by Lillesand and Kiefer (2000) and Lu and others (2004) to analyze multitemporal imagery, ensured highly aligned overlapping pixel registration so that data extraction for later modeling purposes would use correctly sampled reflectance and thematic values. In contrast to AWiFS data, visual inspection of the spatial orientation of high-resolution data sets appeared to match the MIFI base data, so no georegistration was required.

Cloud cover and corresponding shade, although minimal, was present in the AWiFS data sets and required removal to reduce the possibility of sampling erroneous reflectance data. To achieve this removal, both pre- and poststorm rectified AWiFS imagery was clustered using the Iterative Self-Organizing Data Analysis Techniques (ISODATA) algorithm in Imagine (ERDAS 2003) with 250 clusters, 12 maximum iterations, and a convergence threshold of 0.95. The resulting two thematic layers were next interpreted, in a heads-up fashion, coded, and recoded in order to create two cloud and cloud shadow (code = 1)

versus noncloud (code = 0) masks. These masks were then used to remove all cloud-tainted spectral information from the rectified AWiFS images by recoding the eight image layers to zero (four per pre- and poststorm imagery) in these problem areas.

Transforming Remotely Sensed Data—

In exploring various simple band differences, general trends were noted to be unique but subtle for the green and red, NIR, and SWIR bands across the anticipated storm-damaged region. Visually, the red and green bands appeared highly correlated with each other. Because the red appeared more contrasting moving orthogonally from the anticipated center of damage just east of the eye's track (Boose and others 1994), it was chosen for further examination along with NIR and SWIR bands.

Previous studies confirmed the choice in directly using the red, NIR, and SWIR bands (Hame 1991), along with their use in two transformations. These transformations employed two band ratios proven to work in forest change detection: Normalized Difference Vegetation Index (NDVI) (Healey and others 2005; Jin and Sader 2005; Mukai and Hasegawa 2000; Ramsey and others 1997, 2001; Sader and others 2003) and Normalized Difference Moisture Index (NDMI) (Healey and others 2005; Jin and Sader 2005; Sader and others 2003). These indices not only use the visible and infrared bands of interest through proven functions, they also serve to reduce the dimensionality of the data to be analyzed. The formulae for NDVI and NDMI are:

$$NDVI = \frac{NIR\ band - Red\ band}{NIR\ band + Red\ band} \quad (1)$$

and

$$NDVI = \frac{NIR\ band - SWIR\ band}{NIR\ band + SWIR\ band} \quad (2)$$

Land Cover and Type Thematic Data

In Collins and others (2005), the creation of thematic data for use by MIFI in a statewide forest inventory, per the inventory's procedural pilot study (Parker and others 2005), was outlined and resulted in forest age and type thematic layers. The forest-age layer used an approximate 5-year temporal resolution back to the genesis of the Landsat

program in the early 1970s and covered the entire State. In other words, this data set attempted to identify the year of regeneration for areas harvested between 1972 and 2003 in 5-year increments. The data were created using Landsat Multi-Spectral Scanner (MSS) and TM data with the finished products' resolution taking on the finer TM resolution (29 m). The forest-type layer mapped the entire State into water, other nonforest, regenerating forest, softwood, mixed softwood-hardwood, and hardwood classes using 2002–03 TM imagery. Data were georegistered to USGS Digital Ortho Quarter Quadrangles (DOQQs) county-level mosaics, making this the base resolution and orientation for all subsequent analyses involved in this study.

Ancillary Storm Data

Four data set types were obtained from the Internet for use in this study as storm attribute layers. The first two types were acquired from the Atlantic Oceanographic and Meteorological Laboratory (AOML), a subunit of the National Oceanic and Atmospheric Administration's (NOAA's) Hurricane Research Division (HRD), in 14, 3-hour-interval gridded surface wind data sets depicting conditions from 21:00 CDT, August 28, 2005, to 12:00 CDT, August 30, 2005. The grid spacing of these data as the storm passed through south Mississippi was approximately 0.054 degrees in latitude and longitude resulting in a linear distance of approximately 5.25 km in easting and 6 km in northing near the city of Bay St. Louis, Mississippi. These data included sustained surface windspeed (mph) and direction (azimuth degrees), both of which are believed to be influential in structural (Powell and Houston 1996) and forest damage (Ramsey and others 2001) over time. The third data type also came from the AOML and was another gridded surface wind product demonstrating the maximum sustained windspeeds (mph) inflicted by Katrina as it moved through the entire State. The fourth and final storm attribute data obtained for this project included a storm surge extent vector layer acquired from Federal Emergency Management Agency's (FEMA's) Katrina recovery GIS Web site (http://www.fema.gov/hazard/flood/recoverydata/katrina/katrina_ms_gis.shtm).

Model Creation

Independent Variable Assignment—

Remotely Sensed Data

Using the two described band transforms, NDVI and NDMI, and the four original bands from both pre- and poststorm imagery, 18 remotely sensed variables were defined. These variables were created using bands one through four for both pre- and poststorm data sets as well as a delta variable whereby the six prestorm layers, comprising bands and transforms, were subtracted by their poststorm counterparts.

Land Cover and Type Thematic Data

The MIFI thematic data demonstrating statewide age and forest types were used as continuous variables and model strata, respectively. In this construct, age, which could at best be determined back to 1972, was used as a continuous variable with detected year of regeneration being reduced by 2, to account for nearly half the temporal resolution of the age-creation process, and then subtracted from 2005, the year of Katrina's landfall. If the year of regeneration was not found for a forested area, we assumed it was older than the timeframe afforded by the Landsat program, and it was coded with age 40 (2005 minus 1965). This represented a reduction of one temporal resolution interval of 5 minus the interval midpoint correction of 2 years for 1972, the last year of detection. Our assumption was that within the three forest types beyond this age, stand-stocking levels and size were probably more alike than not. As a discriminant, the types could or could not be used as classes for stratifying three different models for softwood, mixed, and hardwood areas.

Ancillary Storm Data

Among the four data set types downloaded for use in this study as storm attributes, one was removed from consideration, one was left alone, and two were compressed into a series of time- and location-dependent storm variables. The surge variable was removed owing to low sampling intensity because the data set indicated that 21 hardwood, 4 mixed, and 0 softwood plots were located within the area of mapped surge inundation. Unlike the 3-hour intervals of

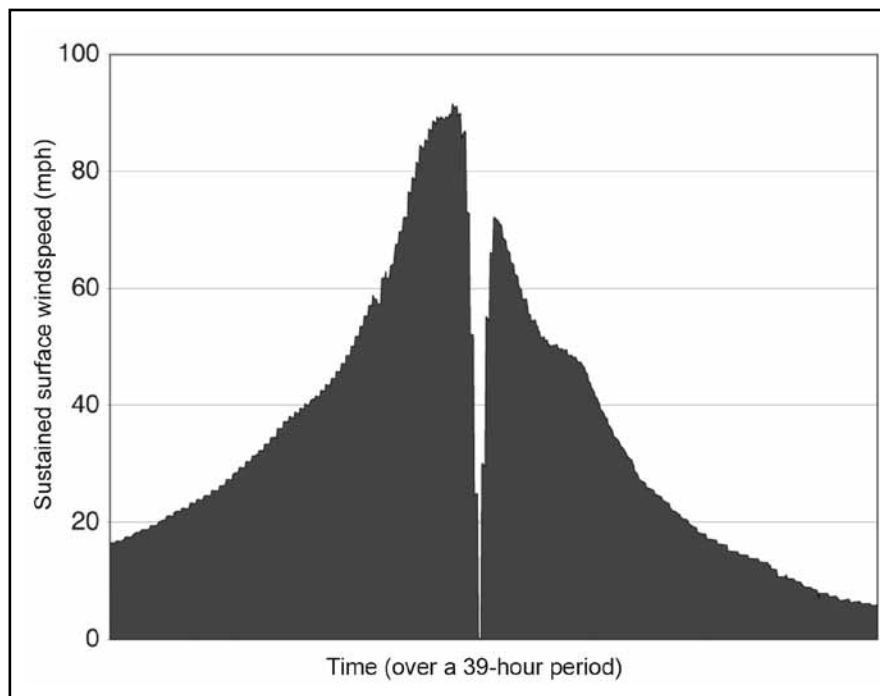


Figure 1—This is a plot-level graph of the resulting 3-minute-interval continuous wind-speeds over a 39-hour period from 21:00 CDT, August 28, 2005, to 12:00 CDT August 30, 2005, for a plot located along Katrina's eye track (note the drop to near zero in the graph's middle), illustrating the converted data used in determining wind duration and stability variables.

windspeed and direction data, the maximum sustained wind variable required no manipulation for use in the model, but it did require interpolation. Using Imagine 8.7, these gridded data were surfaced to the same resolution and orientation as the MIFI base data by using a linear rubber sheeting method (ERDAS 2003).

The first step in transforming the direction and speed attributes from the gridded 3-hour-interval surface wind field data into more usable variables was to create a Microsoft Excel spreadsheet. This helped interpolate the wind data at each plot into more continuous values with regard to speed; direction, as azimuth drastically steps between 360° and 0°; and time, which, in this case, was interpreted into 3-minute (0.05-hour) intervals. Azimuth values were converted to sine and cosine trigonometric function values, and the plot nearest a respective grid point at that point's designated time was allowed to assign its sine, cosine, and wind-speed values to that plot at that time. Time intervals located between fixed 3-hour periods were next assigned weighted x- and y-coordinate locations away from a weighted location

of the storm's eye. Weighting was defined so that proximity to the upper or lower 3-hour time bound for an interpolation time was used to create proportional weights, with the two weights summing to one, for calculating the location and variable weighted averages at an interpolation point. These weights were next used in weighted variable averaging by taking an interpolation time point's weighted location, with respect to storm eye, and calculating windspeed and directional values from the corresponding above and below bounding time points at the same relative weighted location. In Figure 1, a graph of the resulting 3-minute windspeeds for a plot located near Katrina's eye track, illustrates an example of this process's result.

The resulting 3-minute windspeed, sine, and cosine values were next used to create wind duration and stability variables per the anticipated applicability (Powell and Houston 1996, Ramsey and others 2001) of these variables in hurricane damage modeling. The duration variables were calculated over given windspeed thresholds in 10 mph intervals from 30 to 100 mph. For example, the variable for

wind duration at or above 40 mph at a specific plot would count the number of 3-minute time intervals attributed to that plot that were at or above 40 mph and multiply that count by 0.05 hours to get duration in hours. The stability values were also calculated over given windspeed thresholds in 10 mph intervals from 30 to 100 mph. They were done such that the sine and cosine values, isolated for interpolated time points that met the windspeed threshold criteria, were used to calculate two variances, one each for sine and cosine, and then combined into a pooled variance.

Dependent Variable Assignment—

To replace the lack of field data due to ongoing storm damage field sampling, we performed interpretations of high-resolution imagery, pre- and poststorm, with the expectation that aerial-viewed canopy damages were highly correlated with field-measured forest damage. The sample area included Mississippi's six southernmost counties, corresponding to that portion of south Mississippi from 31° N to the Gulf Coast. The USACE imagery was acquired and stratified into 54 interaction classes based on combined forest type, maximum sustained wind (max mph windspeed of >93.5, 93.5-76.5, and <76.5), and age (year constraints of >1993, 1993-88, 1987-83, 1982-78, 1977-70, and <1970). Then we randomly allocated 5 plots into each interaction class, yielding 270 total plots.

Crown closure interpretations and resulting pre- and poststorm differences began with the creation of GIS-generated 0.084-ha rectangular plots (29 by 29 m). Interpretation of these plots involved use of GIS-generated plot boundaries and regular dot grids to employ a systematic method for determining green canopy coverage. The grids were created in a 5 by 5 construct, allowing each dot to represent 4 percent of plot canopy, with outer rows and columns being spaced 2.9 m from their immediate plot bounds and inner rows being spaced 5.8 m in sequence from each other (Figure 2). Using this grid, interpretation was reduced from estimating plot-level green canopy percentages to counting the dots in each plot that fell on interpreted green canopy pixels. The purpose in creating this data set was to develop a bank of prestorm, poststorm, delta (i.e., pre- minus poststorm), or all, canopy data for use in model construction.

Resulting interpretation data were next edited to remove plots that fell within the cloud-classified areas in the AWiFS-derived cloud mask. This masking reduced the plot count from 270 to 252 with an additional 7 plots being removed in the photointerpretation phase owing to lack of forest canopy (this either indicated error in the MIFI forest type layer or canopy removal since acquisition of the 2003 imagery used to generate the MIFI product). The resulting 245 plots were situated across the mapped softwood, mixed, and hardwood forest types in counts of 82, 83, and 80, respectively.

Upon reviewing these edited prestorm, poststorm, and delta canopy measures, the intuitive dependent variable choice appeared to be the delta canopy variable (defined as pre- minus poststorm green canopy cover percentage). However, because these data were largely dependent on prestorm canopy data, this variable was added to the list of independent variables and incorporated into the list of prestorm variables. Also, we believe that prestorm canopy can be created for the study area with moderate to good success (Cohen and others 2001).

Model Definition and Fitting—

The model creation stage of this study was focused on creating multiple linear regression models using ordinary least squares. The sought-after final models were hoped to be parsimonious with optimistic fit values, which, in this case, were high adjusted coefficient of determination or R^2_{adj} values and low RMSE values, for the variable types of interest in the study's objectives. Initial attempts to use the 37 previously described variables (Table 1) in single linear terms produced unsatisfactory fit values. Immediate improvements, however, were noted with the introduction of interaction and quadratic terms (Rawlings and others 1998). The result of creating all these new variables from the above single variables increased the number of possible variables from 37 to 740.

With 740 possible variables available for use in this modeling exercise, a two-step process was organized to reduce the possible number of variables to 40 or fewer, which represented about half (or fewer) the number of plots in each forest type. The first reduction took place

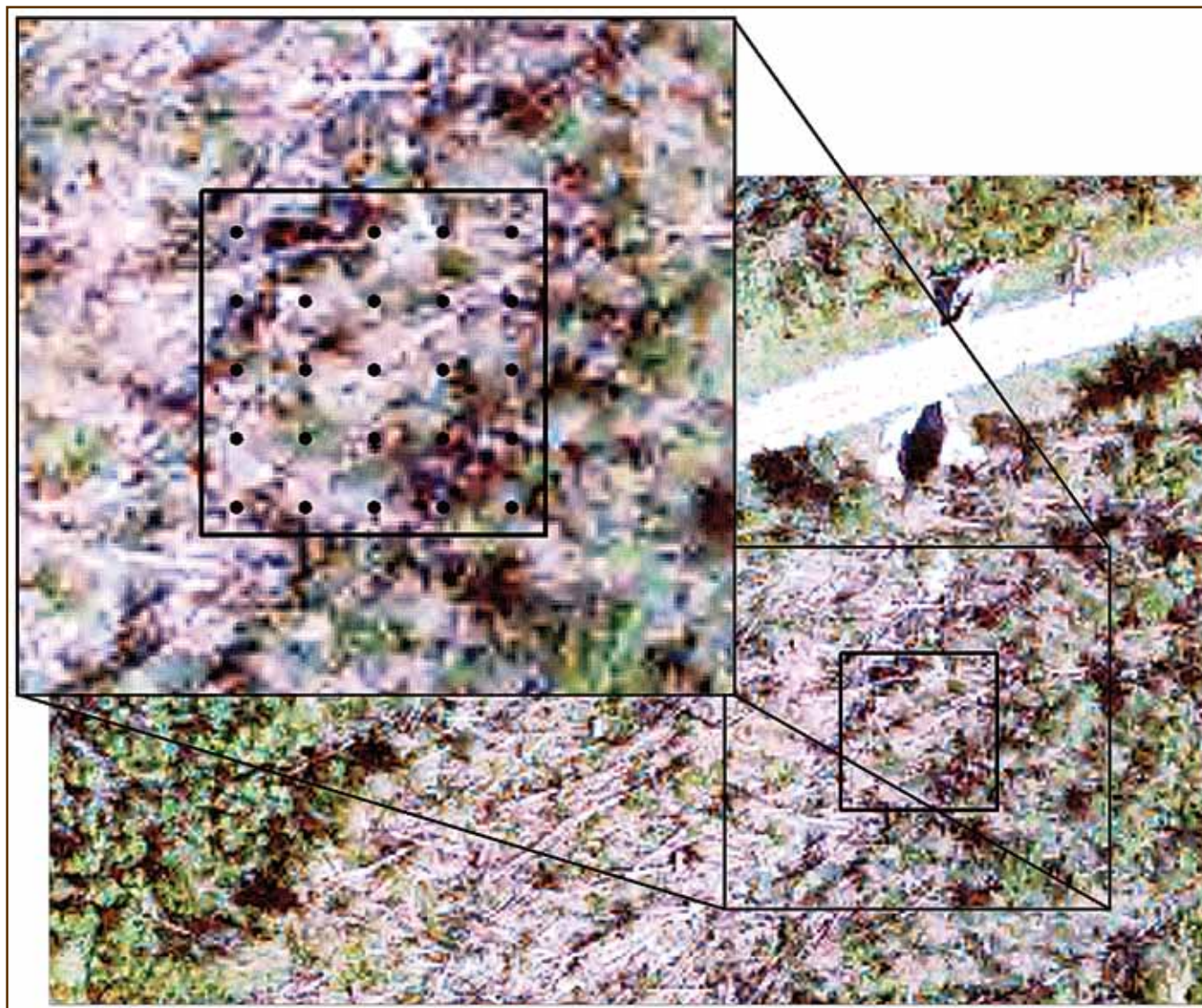


Figure 2—A plot-level view of a photointerpretation plot bound and grid. This image illustrates a highly storm-damaged plot from Pearl River County, Mississippi (note the southeast to northwest oriented downed stems). In the inset, the 5- by 5-point grid used in determining green canopy percentages along with the plot's boundary can be seen.

using stepwise regression methods in SAS's PROC REG procedure (SAS 1999). Using this method, eight model types were reduced using some or all available variables: an overall (regardless of forest type) pre- (including ancillary storm variables) and poststorm data model; an overall prestorm data model; three (for each forest type) prestorm, poststorm, and MIFI variable (forest age) models; and three prestorm and MIFI variable models. In order to regulate that the output stepwise models identify 40 or fewer independent variables for phase two reduction, the significance entry level (SLENTRY in SAS) was set at 0.5, whereas the

significance stay level (SLSTAY in SAS) was accordingly and incrementally adjusted up or down from an initial setting of 0.25 in 0.01 increments until 40 or fewer variables were isolated in the final step.

After isolating eight reduced but still cumbersome models, the leaps-and-bounds algorithm (Furnival and Wilson 1974) in PROC REG (SAS 1999) was employed to hone in on the eight best remaining variables from the field of 40 or fewer with respect to R^2_{adj} and RMSE. The use of R^2_{adj} was deemed advantageous at this point because this value tends to be more comparable than R^2 over models

Table 1—Individual (main effects) model variables and their types

Variable	MIFI	Pre-storm	Post-storm	Variable	MIFI	Pre-storm	Post-storm
Forest age	X			Delta AWiFS NDVI			X
Prestorm green canopy		X		Delta AWiFS NDMI			X
Max sustained wind		X		Wind duration > 30 mph		X	
Prestorm AWiFS band 1		X		Wind duration > 40 mph		X	
Prestorm AWiFS band 2		X		Wind duration > 50 mph		X	
Prestorm AWiFS band 3		X		Wind duration > 60 mph		X	
Prestorm AWiFS band 4		X		Wind duration > 70 mph		X	
Prestorm AWiFS NDVI		X		Wind duration > 80 mph		X	
Prestorm AWiFS NDMI		X		Wind duration > 90 mph		X	
Poststorm AWiFS band 1			X	Wind duration > 100 mph		X	
Poststorm AWiFS band 2			X	Wind stability > 30 mph		X	
Poststorm AWiFS band 3			X	Wind stability > 40 mph		X	
Poststorm AWiFS band 4			X	Wind stability > 50 mph		X	
Poststorm AWiFS NDVI			X	Wind stability > 60 mph		X	
Poststorm AWiFS NDMI			X	Wind stability > 70 mph		X	
Delta AWiFS band 1			X	Wind stability > 80 mph		X	
Delta AWiFS band 2			X	Wind stability > 90 mph		X	
Delta AWiFS band 3			X	Wind stability > 100 mph		X	
Delta AWiFS band 4			X				

Note: A listing of all 37 single variable or main effects used in the study's modeling exercise along with pertinent variable type classifications used in model comparisons. Even though MIFI forest type data were used, they were employed defining strata, not as a model variable.

involving different numbers of parameters (Rawlings and others 1998). This best-eight rule followed the rule of thumb to have 10 observations for each variable while being mindful that the hardwood strata had only 80 observations. To make a more theoretically sound set of model decisions, we also used Hocking's (1976) prediction criteria of using the smallest model, with regard to variable count, that had a Mallows's C_p (Rawlings and others 1998) value of less than or equal to 1 plus the particular model's variable count.

Results

In Table 2, the best eight variable models illustrate moderate fits with RMSE values below 0.15 in only one model and R^2_{adj} values above 0.55 in two models. Between the no MIFI variables pre- and poststorm models, there was a dramatic increase in R^2_{adj} from 0.176 in the prestorm model to 0.439 in the pre- and poststorm situation along

with a matching magnitude reduction in RMSE from 0.219 to 0.181, respectively. In order to compare these no MIFI models to MIFI models, which were created in multiples of three matching the three MIFI forest type designations, a set of pooled RMSE and R^2_{adj} values were created. This creation occurred by combining the three models' error sum of squares, in the case of RMSE values, and by combining the three error and total corrected mean sum of squares considering all three models' degrees of freedom, in the case of the R^2_{adj} values. Comparing these fit values for the eight variable models again demonstrated the drastic improvement in model fit. With the use of poststorm imagery, we observed R^2_{adj} values increasing from 0.381 to 0.506 and RMSE decreasing from 0.190 to 0.170. Additional gains were made in using the MIFI data in models by increasing prestorm no MIFI versus pooled R^2_{adj} values from 0.176 to 0.381 and reducing RMSE from 0.219 to

Table 2—Model fit results by variable types and variable selection criteria

Variables type(s) used	Forest type(s)	Best eight variable models		Hocking's criteria models			Other models of interest		
		R^2_{adj}	RMSE	No. of var.	R^2_{adj}	RMSE	No. of var.	R^2_{adj}	RMSE
Prestorm, post- storm ^a	All	0.4392	0.1805	23	0.4923	0.1717			
Prestorm, post- storm, MIFI	Softwood	0.6796	0.1168	24	0.7893	0.0947			
Prestorm, post- storm, MIFI	Mixed	0.5701	0.1800	25	0.8588	0.0860			
Prestorm, post- storm, MIFI	Hardwood	0.4897	0.2009	15	0.5881	0.1805			
Prestorm, post- storm, MIFI ^a	Pooled	0.5058	0.1694		0.7081	0.1302			
Prestorm ^a	All	0.1758	0.2188	21	0.2765	0.2050			
Prestorm, MIFI	Softwood	0.3366	0.1680	26	0.6405	0.1237			
Prestorm, MIFI	Mixed	0.4300	0.1727	37	0.7735	0.1088	25	0.7142	0.1223
Prestorm, MIFI	Hardwood	0.3647	0.2242	37	0.7542	0.1394	15	0.5258	0.1937
Prestorm, MIFI ^a	Pooled	0.3812	0.1896		0.7345	0.1242		0.5994	0.1525

Note: The modeling exercise results indicating variable types used and MIFI forest type usage along with fit values (adjusted R^2 or R^2_{adj} and RMSE) and number of variables used, when not fixed by the variable selection method.

^a All types or pooled model rows used in numerical data type comparisons.

0.190. Similarly, by incorporating MIFI thematic data, poststorm models were affected with an R^2_{adj} increase from 0.439 to 0.506 and an RMSE reduction from 0.181 to 0.170.

The models derived using Hocking's (1976) method (Table 2) illustrate the same general trend as the eight variable models. Again, improvements with the addition of MIFI and poststorm data were noted in overall (with no MIFI data) and pooled (with MIFI data) R^2_{adj} and RMSE values, except in cases where the application of Hocking's method yielded models with very large variable counts (2 instances used 37 variables). In an attempt to rectify this problem, the other models in Table 2 were created for models that still used a large number of variables. These further reduced models corresponded to the Hocking's identified MIFI, prestorm, and poststorm variable models in the number of employed variables for mixed and hardwood, MIFI, and prestorm models. Examination of these modified models indicates one difference from the eight variable comparisons. The pooled fit values for the prestorm and MIFI models (R^2_{adj} = 0.599 and RMSE = 0.153), in tandem

with the prestorm, poststorm, and MIFI models (R^2_{adj} = 0.708 and RMSE = 0.130), when compared to the pre- and poststorm model (R^2_{adj} = 0.492 and RMSE = 0.172) and prestorm model (R^2_{adj} = 0.277 and RMSE = 0.205) indicate an increased advantage from the eight variable models with regard to using MIFI data as opposed to poststorm image data.

Discussion

Model and Variable Characteristics

The prediction-minded evaluation of data/variable types reported in the previous section states the obvious—that more independent variables tend to improve model predictability of dependent variables. This situation says nothing about the direct applicability of all the work involved with this study and the possible creation of predicted damage values across or outside of the study area, or both, with the various models produced. What is of importance in this work, however, is where, with respect to the variables and variable types used, the gains in model fit occur, although

no statistical inference can be associated with these gains. We expect that similar models can yield predicted canopy changes with RMSE values at or near 0.13 (13 percent) for situations like the passage of a strong hurricane over a mostly undisturbed Southern forested area, such as south Mississippi before Katrina. These estimates can be improved from the forest industry perspective. Industry is often focused on the softwood resource in the South, which is where the best of the stratified models we developed demonstrated a RMSE 0.09 (9 percent). Model fit was comparable in mixed and softwood stands but was poor in hardwoods. This could be the result of a variety of issues from some unknown data bias that was unintentionally introduced into the modeling process or some natural occurrence unknown and unaccounted for in these analyses. These poor results could also illustrate the inherent difficulty and complexity in modeling conditions in hardwood areas.

Potential model flexibility to create comparable predictive models, regardless of the use of poststorm imagery, was a much sought-after finding in this study with mixed results. The reason for this exploration was to display the applicability of modeling anticipated storm damage prior to a weather event. This focus was best explored in comparing the other models (or adjusted) pooled fit values for the MIFI and prestorm variables models and Hocking method pooled MIFI, prestorm, and poststorm variables models where RMSE values were 0.153 versus 0.130, respectively. This comparison is somewhat indecisive as MIFI, prestorm, poststorm variables models outperformed the MIFI and prestorm variables models but only by a small amount (difference in RMSE of < 0.03). Similarly, in the corresponding eight variable models, there was a difference of 0.02 with respect to RMSE. This difference is of a smaller magnitude, however, than the lack of MIFI data comparisons of overall pre- and poststorm variables ($R^2_{\text{adj}} = 0.492$ and RMSE = 0.172) versus prestorm only variables ($R^2_{\text{adj}} = 0.277$ and RMSE = 0.205). In comparing the Hocking pre- and poststorm model (RMSE = 0.172) versus the adjusted pooled MIFI and prestorm model (RMSE = 0.153) and the corresponding eight variable models (RMSE = 0.181 versus

RMSE = 0.190), it does appear that use of the MIFI data in model development at least offsets, maybe even improves, model performance in using prestorm data with the absence of poststorm data.

Possible Model/Variable Improvements

Actual field damage values are being collected in MIFI's Southeast region, which includes Jefferson Davis, Covington, Jones, Wayne, Marion, Lamar, Forrest, Perry, Greene, Pearl River, Stone, George, Hancock, Harrison, and Jackson Counties. These data are the direct metric of interest in this series of work, as opposed to the photointerpreted canopy metric used here. Incorporation of these data is expected to improve development, although model fits may worsen, of any hurricane damage assessment model subsequently created owing to the dependent variable's added meaning. The data could also help address a noted problem of hardwood defoliation versus damage. Poststorm high-resolution imagery indicated that many hardwood areas, particularly in the Pearl River bottom, were defoliated with only minor damage to tree crowns and boles. Differences in this defoliation versus damage aspect of hardwood areas may also be more sensitive to individual hardwood species, which is one of the field metrics, as opposed to the whole hardwood type.

Along with the analysis of field data, future work will also incorporate statistically inferential results, such as variable significance, as opposed to the simple fit comparisons made in this work. These analyses will provide more meaningful results with potential adaptations for collinearity and validation of model assumptions. Model development for repeated application may also be achieved in order to create a more robust and possibly automated product.

Conclusions

Clutter and others (1983) defined risk in the statement: "the inability to estimate future cash flows with certainty is the basic cause of risk in an investment." At play in this analysis are other issues that effect the probability of acquiring an expected return. Examples include rotation lengths (with which MIFI type information may be of further use), intermediate weather conditions (i.e., droughts, floods, etc.), and market fluctuations. Whereas this study is not a risk

assessment in totality, it does place a foundation, albeit not a large one, for development in this direction.

The implications developed in this work with regard to variable creation and the data types utilized are promising for future meaningful region-level continuous damage assessment model creation. The thrust will continue to locate additional ancillary data that may serve to further supplant the advantages of poststorm imagery incorporation in the development of these models. Field data will soon replace the photointerpreted data so heavily relied upon here, and with it a new set of obstacles are expected. In all, however, it does appear possible to create a meaningful damage model that will aid in both economic recovery and assessment of risk associated with storms similar to Hurricane Katrina, possibly before said storms occur.

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Evaluating the Vulnerability of Maine Forests to Wind Damage

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Abstract

Numerous factors, some of which cannot be controlled, are continually interacting with the forest resource, introducing risk to management, and making consistent predictable management outcomes uncertain. Included in these factors are threats or hazards such as windstorms and wildfire. Factors influencing the probability (risk) of windthrow or windsnap occurring can be grouped into four broad categories: regional climate, topographic exposure, soil properties, and stand characteristics. Of these categories, stand characteristics are most commonly and easily modified through forest management. To augment our understanding of the interaction between forest management and wind damage vulnerability in Maine, we developed a wind damage model that reflects site and stand characteristics. Model calibration used information from published literature and experiences of regional managers. The model was evaluated using spatially explicit wind damage records from a 40 800-ha managed forest area in northern Maine. A comparison of means analysis identified significant differences in vulnerability index values between categorical populations of stands that have either recorded blowdown or no recorded blowdown during the last 15 years.

Keywords: GIS, natural disturbance, vulnerability assessment, wind modeling, windthrow risk.

Introduction

“Windthrow is a complex process resulting from interactions between natural and anthropogenic factors” (Ruel 1995). Understanding the interactions between these factors and the inherent risk to stands from this disturbance has numerous benefits to natural resource managers. Windthrow is affected by several factors; climate, topography, physical and biological stand attributes, soil characteristics, and silviculture all play a role in the dynamics of wind disturbance (Ruel 1995).

Understanding windthrow risk throughout the landscape can provide insights into natural vegetation patterns and habitat types. Risk evaluation can also be used to help predict how current forests may change without harvesting and subsequent impacts to forest health. Risk evaluation can help managers (1) evaluate where to locate plantations, (2) decide which regeneration strategies are appropriate, (3) determine where thinning and other partial harvests are acceptable, and (4) determine what species composition or rotation length is desirable for individual sites. Predicting damage, or potential for damage, provides the opportunity for impacts to be considered during prescription development, allowing for revision of management objectives or incorporation of mitigative actions into management plans (Mitchell 1998).

Currently, little is known about the extent of both catastrophic and endemic wind damage in Maine. As a result, this project’s aim was to explore the nature of wind disturbance throughout the State’s large forest ownerships. To augment our understanding of the interaction between forest management and wind damage vulnerability, this project developed a generalized wind damage model that reflects topographic exposure (distance-limited Topex) (Ruel and others 1997), soil conditions (rooting depth), and stand characteristics (density, edge, height, species composition, and treatment history). Results from similar modeling projects in British Columbia suggest that these risk factors are consistent in varied locations, and this indicates that general models may be applicable to landscapes other than the ones for which they were developed (Lanquaye-Opoku and Mitchell 2005). This model was calibrated using information from published literature and experiences of regional managers. We then evaluated it using spatially explicit wind damage records from a 40 800-ha area of managed forest in northern Maine.

Methods

Windthrow risk in many parts of the world has been modeled and assessed. Empirical models are best suited for areas with complex, heterogeneous stand structure and composition (Lanquaye-Opoku and Mitchell 2005, Mitchell

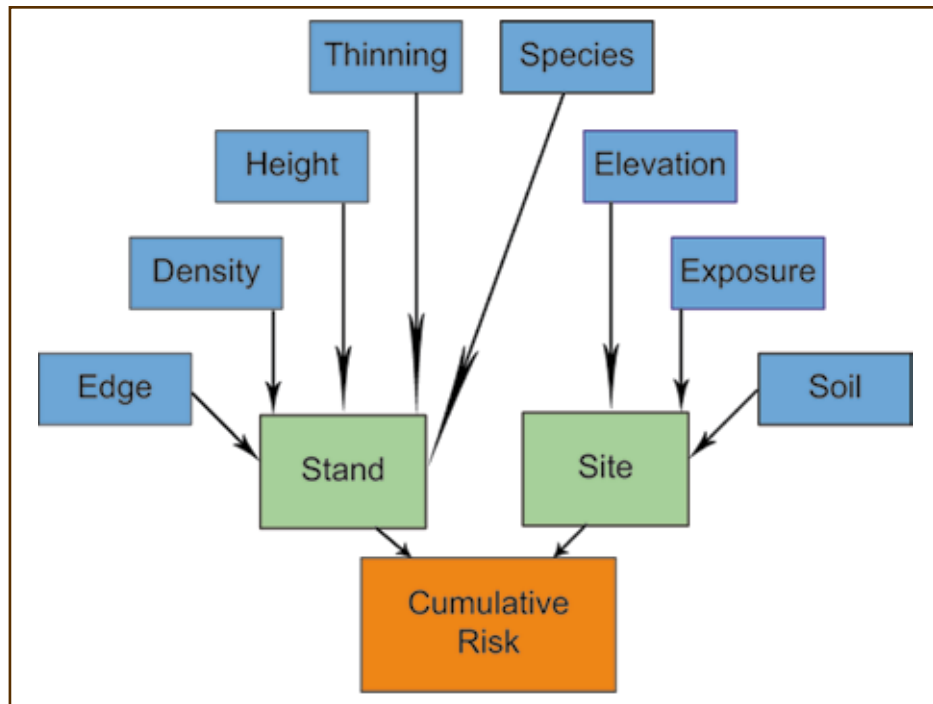


Figure 1—Flow chart of cumulative risk grid development shows the grouping and integration of individual variables into composite risk variables. Composite stand and site risk variables are then combined into a cumulative windthrow risk variable. All variables are indexed between 0 and 1 (high to low risk) and are combined additively.

and others 2001) like the forests of Maine. This empirical approach typically utilizes regression models relating wind damage to physical stand components. Generally, the models produce a probability value rating or index of damage potential based on the stand's suite of environmental conditions. Index modeling of spatial phenomena is enhanced with GIS, which allows for the integration of spatially explicit model parameters.

Logistic regression is commonly used for evaluating these models and isolating highly correlated component variables (Lanquaye-Opoku and Mitchell 2005, Mitchell and others 2001). Rather than using logistic regression, this project produced a generalized model, retaining variables that would not be statistically significant in a logistic regression analysis. This approach is unique because it attempts to create a model that may be applicable regionally and not be limited to the landscape where it was developed.

Eight environmental parameters (elevation, soil rooting depth, topographic exposure, stand species type, stand height, stand density, stand history, and exposed edges)

were used to generate a spatially explicit vulnerability index value. Mitchell (1995, 1998) advocated grouping the factors into three broad categories—exposure, soils, and stand characteristics—to form a windthrow triangle, a conceptual model of the relationships among these interacting factors. For this model, factors were broken down further into site and stand parameters and combined to generate the cumulative windthrow risk index (Figure 1). Data for model variables, including a spatially referenced database of windthrow history, was acquired from various sources and covered an area of private landholdings in the northern portion of the State.

Site: Exposure

Topographic exposure is a critical variable in assessing stand vulnerability. Several indices have been created to describe relative topographic exposure or topographic protection. Topex wind exposure index has been used for some time to assess windthrow risk in Great Britain (Miller 1985). The importance of topographic exposure in modeling windthrow

risk has been demonstrated in other areas with forest-based economies. According to Ruel and others (2002), this variable accounts for over 77 percent of the British wind hazard rating system's total score. Distance-limited Topex was chosen for this project because of its relatively easy calculation and strong correlation to wind tunnel simulation (Ruel and others 1997). Topographic exposure rasters were generated using a model developed and provided by The Windthrow Research Group, University of British Columbia, Vancouver, Canada. The scripts calculate an index of exposure that is the summation of the maximum and minimum angles to the skyline within a user-specified distance. The index can be calculated in the eight cardinal directions and weighted according to user preferences. Ten exposure grids were produced for this project, simulating unweighted exposure at two limiting distances (1000 m and 1500 m) and exposure in the eight cardinal directions (limiting distance of 1000 m).

Site: Soils

Forest soils are also a major component in understanding susceptibility of a forest stand to wind damage. Soil aeration, ease penetration by roots (rooting depth), and moisture-holding capacity all affect the pattern of root development. Generally, loose dry soils facilitate deeper rooting and spreading than do shallow clayey soils (Mergen 1954). Shallow soils, which limit rooting depth and saturate easily like those commonly found in the spruce flat forest type, are increasingly prone to windthrow when saturated. The mass of soil that roots adhere to for anchorage becomes so wet it no longer adheres to itself, and the tree loses a substantial portion of its basal mass, crucial for resistance to windthrow (Day 1950). To compound the problem on wet soils, the rocking of the root plate can pump mud out from under the tree, further reducing the tree's stability (Maccurach 1991).

Depth to groundwater was consistently cited by Maine forest-land managers as crucial to predicting the likelihood of blowdown in stands. The University of Maine's Cooperative Forestry Research Unit provided a continuous depth-to-water-table raster for this project. This variable was combined with coarse-scale soil depth data to create

a raster selecting the minimum depth of the two available data sources and indexed between 0 and 1.

Site: Elevation

Elevation was also incorporated into the site component of the model. Elevation values from a 30-m resolution digital elevation model (DEM) of the study were recalculated into an index between 0 and 1. Elevation had statistically significant correlation with wind damage in cut-block edge vulnerability modeling by Mitchell and others (2001). Studies by Worrall and Harrington (1988) in Crawford Notch, New Hampshire, found that gap size from chronic wind stress and windsnap or windthrow increased strongly with elevation from over 60 percent of the gap area at 764 m (2,521 feet) to almost 85 percent of the gap area at 1130 m (3,729 feet). Gap formation led to subsequent mortality from chronic wind stress and windthrow in gap edge trees. This trend was confirmed by Perkins and others (1992) on Camel's Hump in Vermont. These trends are driven by surface friction acting counter to the force of the wind. Windspeed will increase locally with elevation because surface friction will decrease (Bair 1992).

Stand: Composition and Characteristics

Variables describing stand composition and characteristics were extracted from the forest landowner GIS database, which contains stand-level information to a minimum size of 1 acre. Stand composition is recorded under a three-variable scheme (Table 1). The database also reports stand history from the present to the mid-1980s including stand damage by wind storms and previous harvest entries. The history records include the year of the event and the event type or silvicultural prescription. An iterative network of Microsoft Access™ queries was developed and used to isolate prior stand entry and wind damage events and to create vulnerability indices for individual stands based on stand type and the presence or absence of balsam fir, stand height, and stand density.

The species risk index assigns ranks for the four potential forest types ($H = 0.3$; $S = 0.7$; $HS = 0.45$; $SH = 0.55$). The forest type rank is combined additively with an adjustment factor for the presence of balsam fir (*Abies*

Table 1—Three-variable stand type scheme displays 64 unique stand type combinations possible

Species type code	Height code	Density code
H: > 75% hardwoods	1: seedlings	A: 100-75% crown closure
S: > 75% softwoods	2: saplings	B: 75-50% crown closure
HS: > 50% hardwoods	3: pole-size timber	C: 50-25% crown closure
SH: > 50% softwoods	4: sawlog timber	D: 25-0% crown closure

Each stand is assigned only one code per category; for example, SH2B, representing a softwood-dominated mixed-wood stand of saplings with crown closure between 75 and 50 percent. All stands in the GIS database are characterized by this three-variable code.

balsamea (L.) Mill.) in the overstory because high rates of root rot predispose fir to wind damage (Whitney 1989). The balsam fir adjustment considers the relative abundance of balsam fir in the overstory as the primary, secondary, or tertiary overstory species.

The stand height risk grid was built directly from the landowner database. The stand type height code was rasterized, creating a raster with five potential data values (0-nonforest; and 1 through 4 representing the height classes found in Table 1). The values were divided by 4, the maximum value, to create the desired index range between 0 and 1.

The density grid captures the overstory density of the stands in the study area. Access queries determined the height and density of the most dominant or two most dominant species in each stand, if more than one species were present. Queries assigned values corresponding to the original alphabetical density codes of the most dominant overstory species. Density of the primary overstory species was collected and modified if the secondary species was also in the same canopy strata.

Stand: Thinning

Stands are more vulnerable to windthrow following thinning for two reasons. First, the increase in spacing resulting from thinning creates more canopy roughness, which, in turn, increases turbulence of the wind at the canopy level. Increased turbulence and wind penetration result in reduced tree stability (Blackburn and others 1988, Maccurach 1991). Second, high initial stand density produces unfavorable height-to-diameter (H:D) ratios (Wilson and Oliver 2000). This is less of a problem if stand density remains high; however, thinning removes the support of neighboring

stems, dramatically increasing stand vulnerability. This period of vulnerability may last from a few years to over a decade.

Owing to the prominence of partial removal harvests in the State (McWilliams and others 2005), it was determined that a binary variable would best capture risk associated with stand entry from thinning and partial harvesting. A binary index raster of stand treatments was created from the records of stand entry in the landowner database. All prior entries that involved incomplete removal of the overstory and occurred in the decade preceding the most recent wind event (2001) were classified as thinned. Clearcuts and uncut stands were classified as unthinned. Thinned stands were assigned a value of 1, and unthinned stands a value of 0.

Stand: Edge

The edge raster variable represents the percentage of the stand classified as edge. For this project, edge is defined as a two height-class difference between stands. Production of this data layer was a multistep procedure. First a raster of stand height was produced. Topex scripts were run on this raster with a limiting distance of 30 m, or 1 pixel. This identified all height-class differences between stands across the landscape. Positive values indicated edges of shorter stands, and negative values indicated edges of taller stands. The original height grid was also analyzed with Arc9's zonal range statistics tool. Range statistics defined all edges classified by the height differences between the two adjacent stands. This raster was recoded to display only edges greater than two height-class differences. The Topex-generated edge raster and range statistics raster were combined to identify the edges of the taller stands (negative Topex scores) when the height difference between adjacent

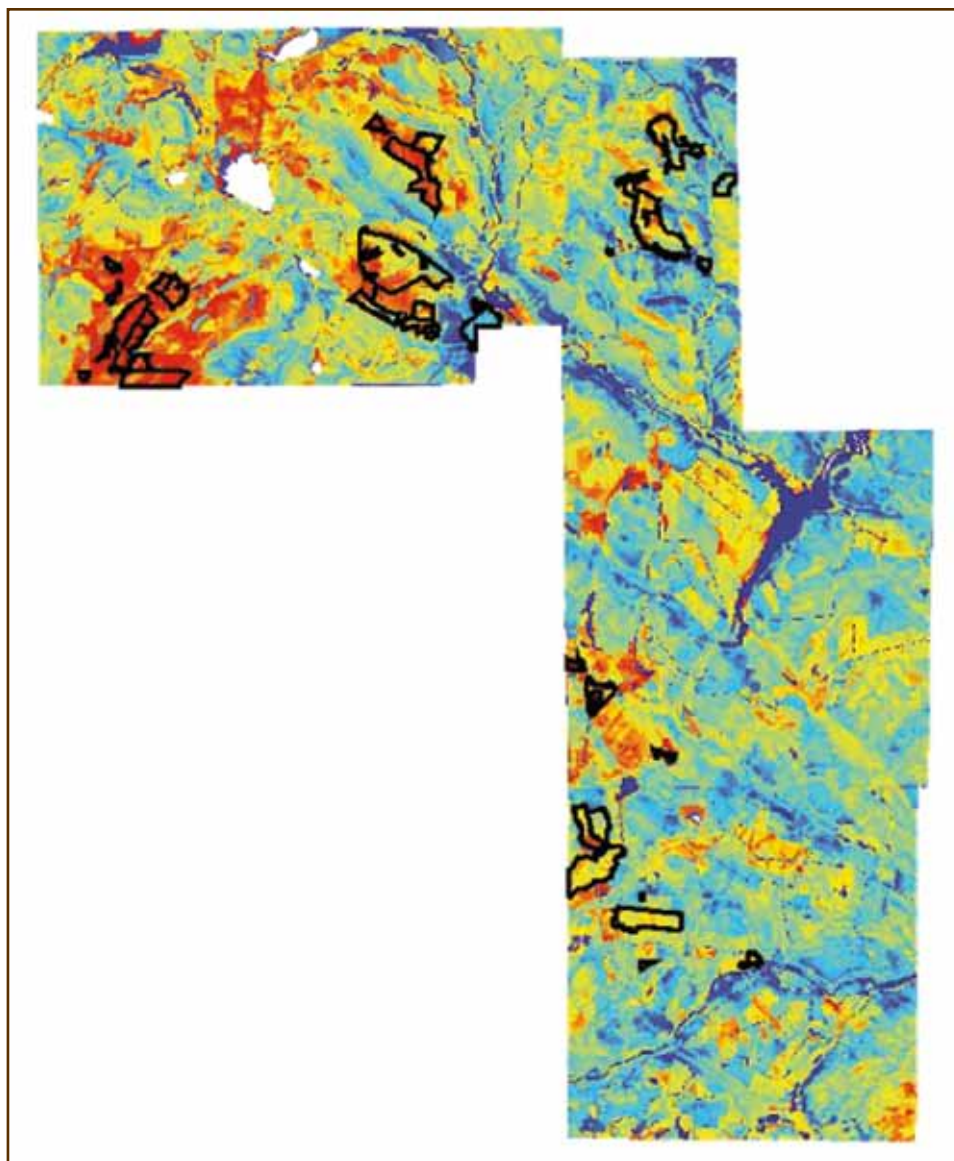


Figure 2—The cumulative wind damage risk grid represents the integration of the two composite model variables representing site and stand risk, generated from eight unique variables. Risk values are displayed along a color ramp from red to blue. Red represents the highest risk and blue the lowest. Actual areas of recorded wind damage are shown in black.

stands was greater than two height classes (range statistic greater than 2). Stands delineated in the GIS database were used as zones to calculate the percentage of edge within the individual stands, and the index was generated directly from these values.

Cumulative Risk

The cumulative risk grid is composed in two stages. In the first stage, the three individual site and five individual

stand components are combined additively to form separate composite site and composite stand grids. The second stage combines the composite site risk grid and composite stand risk grid additively to form a cumulative windthrow risk grid. Figure 2 displays the cumulative windthrow risk grid. Ten separate cumulative windthrow risk grids were produced, one for each unique exposure input grid (described in “Site: Exposure”). All grid combinations were performed

Table 2—Results for individual component variables

Variable	Minimum	Maximum	Direction	Consistent
Density	0.141	0.945	NS	—
Edge	0.001	0.969	Negative	No
Height	0.000	0.137	Positive	Yes
Species	0.060	0.960	NS	—
Thinning	0.000	0.000	Positive	Yes
Elevation	0.000	0.007	Positive	Yes
Soil depth	0.000	0.101	Negative	Yes
Expos_1000	0.050	0.929	Positive	No
Expos_1500	0.060	0.873	NS	—
Expos_north	0.002	0.805	Negative	No
Expos_ne	0.040	0.922	Negative	No
Expos_east	0.164	0.924	NS	—
Expos_se	0.247	0.926	NS	—
Expos_south	0.119	0.911	NS	—
Expos_sw	0.014	0.791	Positive	No
Expos_w	0.143	0.891	NS	—
Expos_nw	0.001	0.647	Negative	No

The table displays the minimum and maximum p-values obtained from the 10 samples in the comparison of means analysis. Exposure variables are listed by their corresponding directional weights. The column titled Direction indicates variable agreement with model assumptions, whereas NS indicates a nonsignificant response. The column titled Consistent indicates whether the variable's response was consistently statistically significant.

with the single output map algebra tool in Arc9's spatial analyst toolbox.

Model Evaluation

To avoid problems associated with spatial autocorrelation, the wind damage vulnerability model was analyzed with a comparison of means from a random sample of polygons within the study area. Hawth's analysis tools (Beyer 2006) were used to maintain a minimum distance between sampled points. To ensure consistent results, 10 separate random samples of polygons were drawn from the study area. These 10 samples are analyzed individually and the results pooled to measure consistency between the samples. Approximately 560 polygons are sampled in each iteration of random sampling, accounting for roughly 14 percent of the study area in each sample.

The analysis used either a two-sample t-test or a Mann-Whitney test to detect differences between the population means. Mann-Whitney was chosen as the default test because this nonparametric test is justified in all situations

where the t-test is applicable and in situations where the assumptions of the two-sample t-test are not met (Zar 1984).

The two-sample t-test was used for Mann-Whitney results indicating near statistical significance when variables met the assumptions of this test. Tests used an alpha of 0.05 to test the null hypothesis, which is that the two population means are equal: $H_0: \mu_1 = \mu_2$. Results from the analysis of the 10 samples were tested for consistency with a t-test. Significant results from the Mann-Whitney and two-sample t-tests were coded either 1, positive correlation with the model, or -1, negative correlation with the model. Non-significant results were coded as a 0.

The t-test procedure tested for statistically significant differences between the responses of the individual model variables across the 10 samples. A mean that was statistically not equal to 0 indicated consistent significance, reflecting either positive or negative correlation with the model. Tests used an alpha of 0.05 to test the null hypothesis that the means for the two populations are equal:

$H_0: \mu_1 = \mu_2$.

Table 3—Results for composite risk variables with minimum and maximum p-values obtained from the 10 samples in the comparison of means analysis

Variable	Minimum	Maximum	Direction	Consistent
STAND	0.000	0.000	Positive	Yes
SITE_1000	0.045	0.818	Positive	Yes
SITE_1500	0.046	0.793	Positive	Yes
SITE_north	0.214	0.968	NS	—
SITE_ne	0.101	0.968	NS	—
SITE_east	0.037	0.941	NS	—
SITE_se	0.002	0.348	Positive	Yes
SITE_south	0.127	0.190	NS	—
SITE_sw	0.002	0.343	NS	—
SITE_w	0.005	0.425	NS	—
SITE_nw	0.089	0.919	NS	—
CMLTV_1000	0.000	0.000	Positive	Yes
CMLTV_1500	0.000	0.000	Positive	Yes
CMLTV_north	0.000	0.000	Positive	Yes
CMLTV_ne	0.000	0.000	Positive	Yes
CMLTV_east	0.000	0.000	Positive	Yes
CMLTV_se	0.000	0.000	Positive	Yes
CMLTV_south	0.000	0.000	Positive	Yes
CMLTV_sw	0.000	0.000	Positive	Yes
CMLTV_w	0.000	0.000	Positive	Yes
CMLTV_nw	0.000	0.000	Positive	Yes

Site risk and cumulative risk variables are listed by their corresponding directional weights. The column titled Direction indicates variable agreement with model assumptions, whereas NS indicates a non-significant response. The column titled Consistent indicates whether the variable's response was consistently statistically significant.

Results and Discussion

Several model variables were found to have statistically significant differences between the two populations (blowdown and nonblowdown). However, not all statistically significant differences were in the direction expected. Positive differences indicate statistically significant population-to-population differences that are in the direction that is expected on the basis of preliminary model research. Negative differences indicate statistically significant population-to-population differences that are in a direction that is opposite the direction expected on the basis of preliminary model research.

The consistency analysis utilized a one-sample t-test. Variables that displayed the same relationship in all 10 iterations cannot be tested for consistency in this manner, but variables that displayed the same relationship in all 10

iterations are considered inherently consistent. The thinning, elevation, composite stand, and all 10 cumulative risk variables displayed a positive difference between the population means through all 10 iterations and are considered statistically consistent. Results are displayed in Tables 2 and 3.

For both tables, positive difference indicates that the mean risk value for the population of stands with recorded wind damage was higher than the mean risk value for the population of stands without recorded wind damage. Positive differences agree with the assumptions used during model construction. Negative difference indicates that the mean risk value for the population of stands with recorded wind damage was lower than the mean risk value for the population of stands without recorded wind damage. Negative differences do not agree with the assumptions used during model construction.

The density variable (Table 2) did not produce significant differences between the population means in any of the iterations. The assumption based on the wind vulnerability literature (Gardiner and others 1997, Lohmander and Helles 1987) was that the less dense stands would be more susceptible. This was thought to be the case in an area with a long management history of natural regeneration and frequent stand entry.

The edge variable (Table 2) had a negative difference between population means 30 percent of the time, not frequently enough to be considered statistically consistent. Most of the stands in the landscape being evaluated are in the two tallest height classes. This trend results in a landscape with very little edge in general. The edge that is present may be in areas at lower risk to wind or edge may not be a critical factor in this landscape.

The height variable (Table 2) had a positive difference between population means 90 percent of the time. This statistically consistent difference likely reflects the increased vulnerability to wind damage with increased tree size (Lohmander and Helles 1987, Peltola and others 1997, Smith and others 1997). The species variable (Table 2) did not have significant differences between the population means in any of the iterations. This was surprising considering managers all cited softwoods and, most notably, balsam fir, as being the most sensitive to wind disturbance. It may indicate homogeneity within the landscape or an insensitivity of the index to differences in composition.

The thinning variable (Table 2) had statistically significant positive differences between the population means 100 percent of the time. This agrees with conventional wisdom of the land managers—that windthrow is much more common in previously thinned stands. An evaluation of the landscape shows that 99.72 percent of the recorded blowdown occurred in thinned stands.

The composite stand grid, STAND, (Table 3) also had significant positive differences between the population means 100 percent of the time. This may be primarily driven by the combination of the height and thinning components that compose this composite variable. The thinning variable's binary property makes it a relatively powerful component of the composite grid.

Differences between population means for the topographic exposure variables (Table 2) were never statistically consistent. This trend was noticed by Mitchell and others (2001) when an analysis of their model revealed a level of contribution from topographic variables to the model lower than expected. Simple terrain variables may not adequately describe airflow phenomena induced by complex terrain (Mitchell and others 2001).

The elevation variable (Table 2) had a positive difference between population means in the model 100 percent of the time. This agrees with the assumption that susceptibility increases in higher areas of the landscape, where exposure and windspeed are greater (Bair 1992).

The soil variable (Table 2) had a negative difference between population means. This trend was statistically consistent, occurring 70 percent of the time. This is counter to the original assumption of the model that forests growing in areas with more restricted rooting depths would be more vulnerable to wind disturbance (Day 1950, Mergen 1954). An analysis of the correlation between soil depth and elevation in this landscape yields a mean Pearson correlation of 0.344 for the 10 iterations. This is substantially larger than the test statistic (0.088) for an alpha of 0.05 and a sample size greater than 100. This result indicates a statistically significant positive correlation between the two variables (Zar 1984); deeper soils are correlated with higher elevations in the landscape being evaluated.

Differences between population means for the composite site variables (Table 3) were either positive or non-significant. Differences between population means were statistically consistent for three exposure variants, both site grids with nondirectionally weighted exposure input variables (SITE_1500 and SITE_1000) and SITE_se, the site variant modeling topographic exposure to the southeast.

The positive difference between the population means of all site variables (with the exception of the north, northeast, and northwest exposure variants) suggests that topographic exposure is important, even though significant differences between the population means were not consistently detected for exposure as an individual variable.

Although the data within the GIS database are fairly coarse, the general spatial model we developed associates

moderate to high vulnerability ratings with reported wind damage in the landscape. All of the final risk assessment variables, CMLTV_direction (Table 3), have a positive difference between the population means. This validates the model's ability to differentiate between vulnerability of damaged stands and vulnerability of undamaged stands. The difference between the population means in the cumulative risk variables is highly significant with p-values of 0.000 recorded in all 10 iterations for all 10 exposure variants.

Conclusion

Wind damage to forests in Maine is a continual consideration for forest managers across the region. The importance of wind damage in the State is likely to increase as large areas of forest that were regenerated during the spruce budworm outbreak of the 1970s and 80s continue to mature. In addition, the vast majority of harvesting in the State utilizes partial harvesting techniques that increase stand susceptibility to windthrow. If these trends in stand height and area thinned continue, managers will need tools and techniques to help them manage the growing wind damage threat. Spatial risk index modeling with GIS provides an alternative view of the landscape, allowing for threat assessment and more informed decisionmaking. The wind vulnerability model developed for this project can be used as a tool to assist in forest planning and provide insight into historical trends in forest dynamics and habitat associations. This tool should be portable to other regions because it contains variables that are frequently identified as critical in predicting windthrow vulnerability. The stand-level variables are general enough to adapt to similar forest typing schemes used by other managers in the State.

There are multiple complexities associated with modeling vulnerability to wind damage in forests. Foremost among these is modeling the interaction of rare regional wind events, chaotic local wind behavior, changing soil conditions (saturation and freezing), and dynamic stand characteristics (growth and manipulation). One approach for managing the uncertainty surrounding wind damage is to develop relatively simple models of vulnerability based

on past observations of factors influencing damage. These more general models, like the one developed for this project, would not be expected to predict past wind damage as well as models developed directly from damage information collected after a particular storm or in a specific landscape. However, they may prove less biased towards particular sites, stand conditions, or individual wind events and, therefore, be more useful for guiding forest management across a large region or as stand conditions change through time.

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English Equivalents

When you know:	Multiply by:	To find:
Nanometers (nm)	3.94×10^{-8}	Inches (in)
Millimeters (mm)	0.0394	Inches
Centimeters (cm)	.394	Inches
Meters (m)	3.28	Feet (ft)
Cubic meters (m ³)	35.3	Cubic feet (ft ³)
Meter per second (m/s)	2.24	Miles per hour (mph)
Kilometers (km)	.621	Miles (mi)
Kilometers per hour (kph)	.621	Miles per hour
Hectares (ha)	2.47	Acres (ac)
Microliters (μL)	.0000338	Ounces (fluid)
Milliliters (mL)	.0338	Ounces (fluid)
Grams (g)	.0352	Ounces
Kilograms (kg)	2.205	Pounds
Kilograms per cubic meter (kg/m ³)	.0624	Pounds per cubic foot (lb/ft ³)
Kilograms per hectare (kg/ha)	.893	Pounds per acre (lb/ac)
Megagrams per hectare (Mg/ha)	.446	Tons per acre
Kiloequivalents per hectare (keq/ha)	.405	Kiloequivalents per acre
Trees per hectare	.405	Trees per acre
Micrograms per gram (μg/g)	1	Parts per million
Degrees Celcius (C)	$1.8C + 32$	Fahrenheit (F)

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