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Volume 1

Technical Editors

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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 precieve 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 *Phytophora 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. Lee[1] *and Jerome S. Beatty*[2]

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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Acknowledgments

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.forestencyclopedia.net. Funding for the conference, this publication, and its Web counterpart was provided by WWETAC, EFETAC, PNW, and SRS.

Contents

Volume 1

Land

Syntheses

Case Studies

Pests/Biota

Syntheses

Case Studies

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LAND

Syntheses

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 policymakers; 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	USDAForest 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 pastureland 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

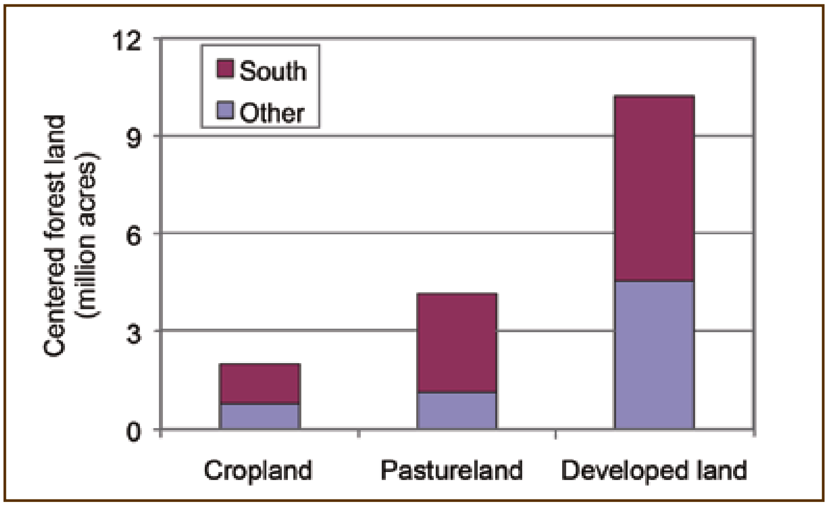

Figure 1—Conversion of nonfederal forest land by destination, and South vs. non-South, 1982-97 (USDA NRCS 2001) (note does not include rangeland and other miscellaneous uses).

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 inmigration 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 interspersion 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,

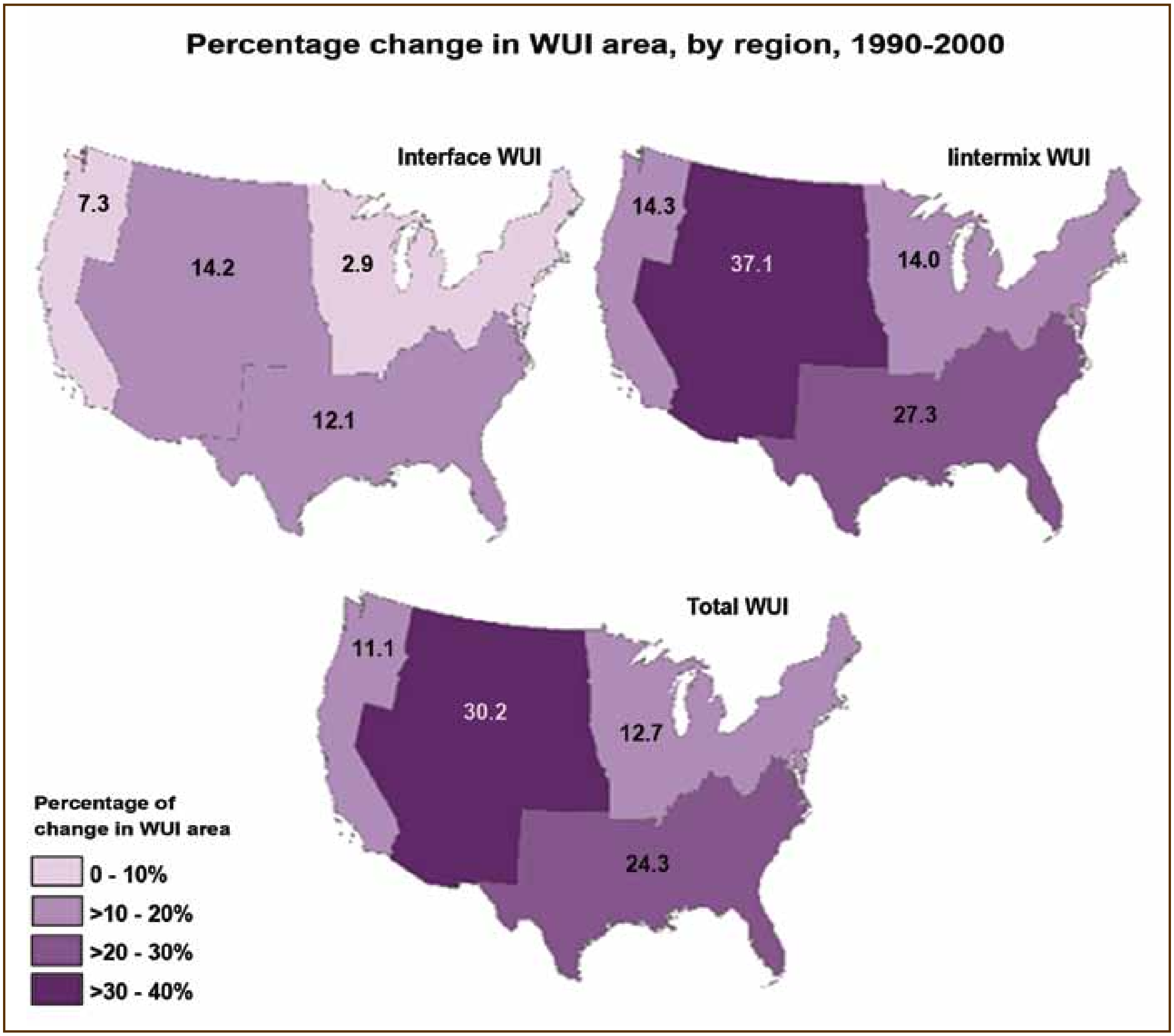

Figure 2—Wildland-urban interface in the United States and percentage change in area by region, 1990-2000.

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 km^2) 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 km^2 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

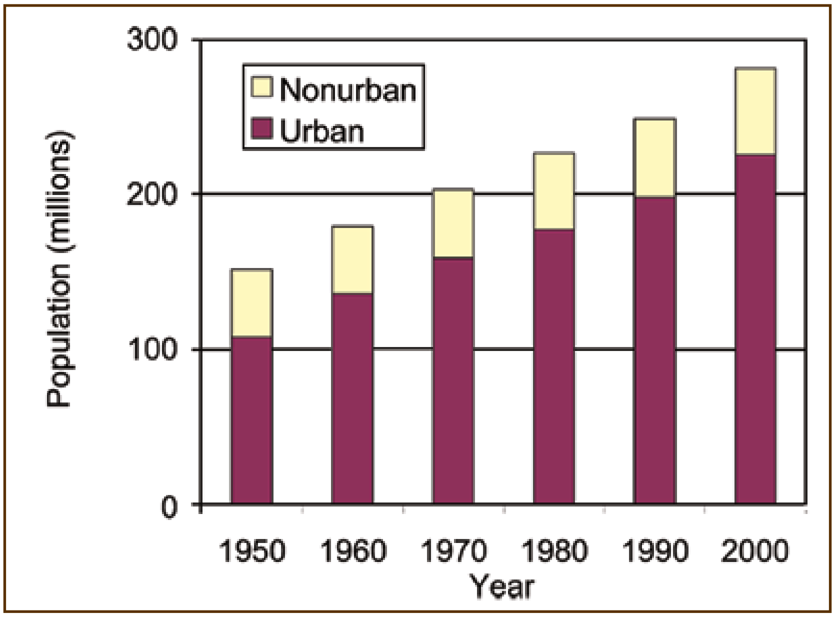

Figure 3–United States population by urban and nonurban components, 1950 to 2000.

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 multiobjective 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 Scholield 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.

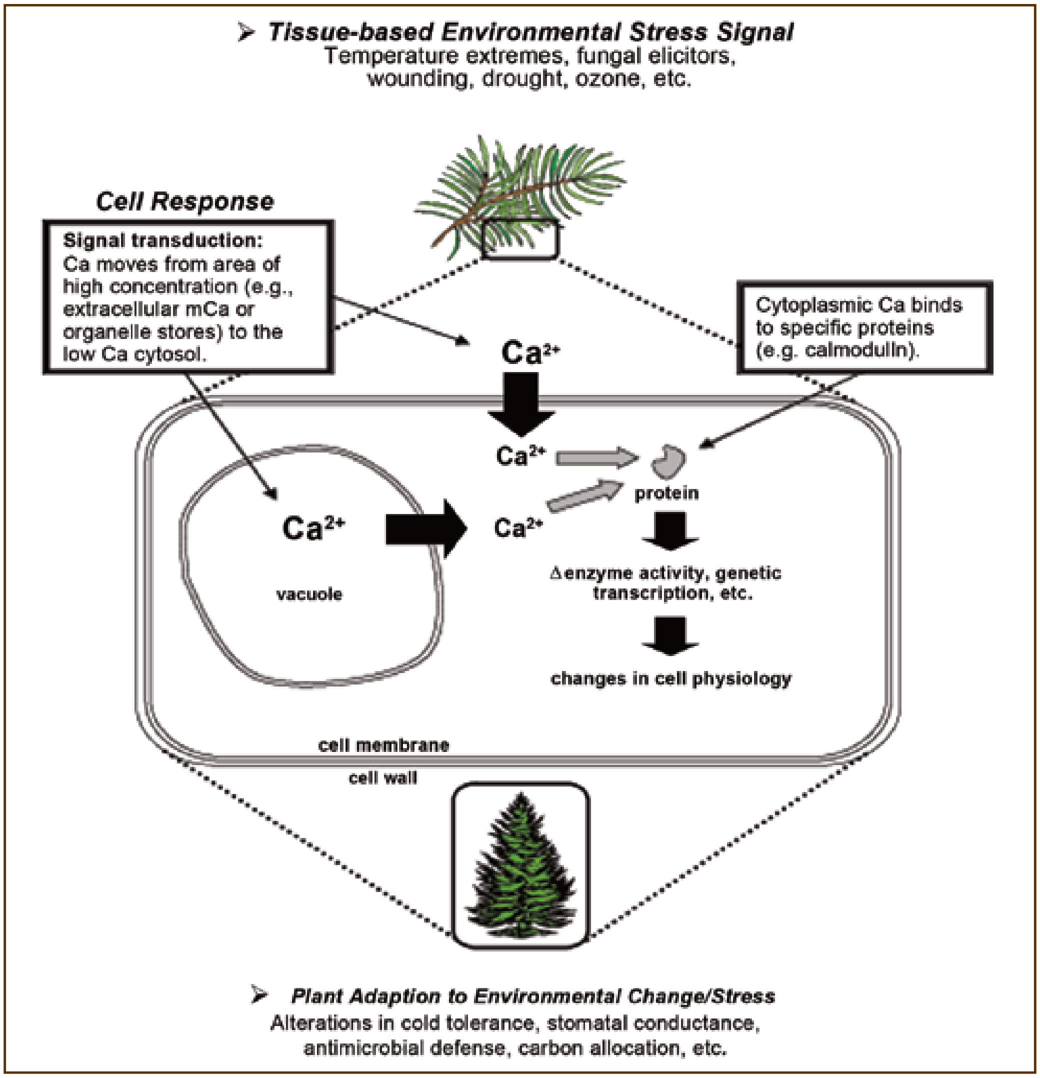

Figure 1—Schematic representation of how biological calcium (Ca) depletion may suppress plant stress response systems and predispose trees to decline. Environmental stimuli (e.g., a stress event) triggers the movement of Ca from areas of high concentration (i.e., extra-cellular and organelle pools) into 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 foliar 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

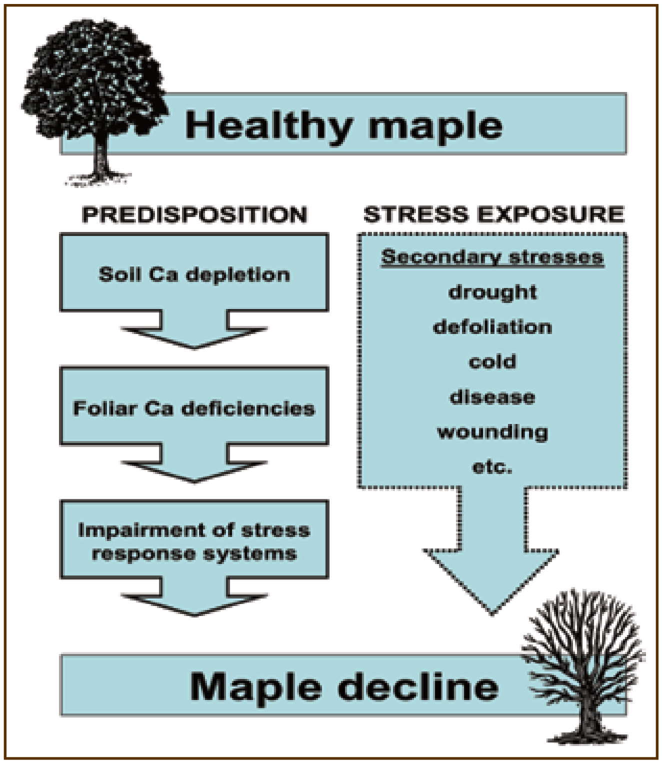

Figure 2—Proposed mechanism of sugar maple decline. This depicts how Ca deficiency may suppress stress response systems and predispose trees to decline following exposure to stress factors reported to contribute to sugar maple decline in the Eastern United States and Canada (Schaberg and others 2001).

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 destructiv*e 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 $\mu g\ g^{-1}$, 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 $\mu g\ g^{-1}$ 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 (Ouimet 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 keq ha^{-1} 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 keq ha^{-1} 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) kg ha^{-1} y^{-1} and N

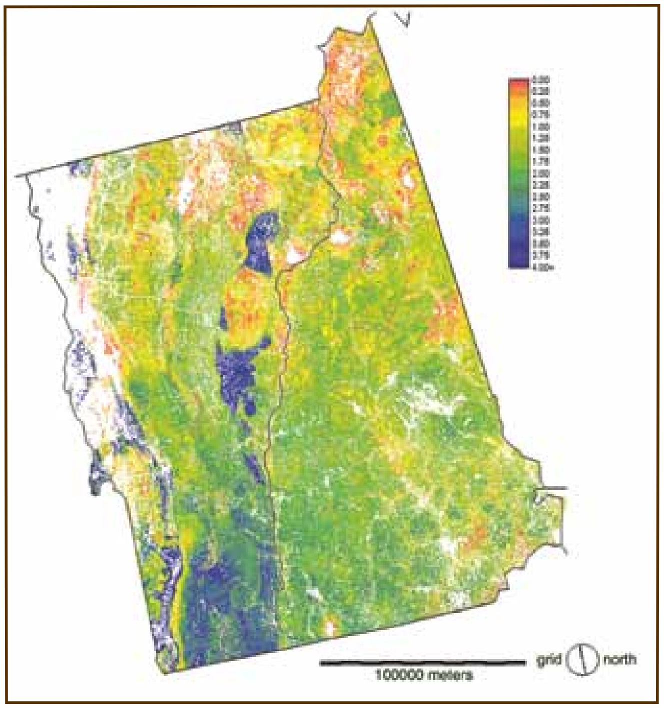

Figure 3—Critical loads of sulfur plus nitrogen to upland forests; S + N atmospheric deposition rates higher than the critical load result in greater exports of nutrient cations (Ca^{2+}, Mg^{2+}, K^{+}) than inputs of these nutrients, leading to the eventual deterioration of soil fertility, forest health, 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.

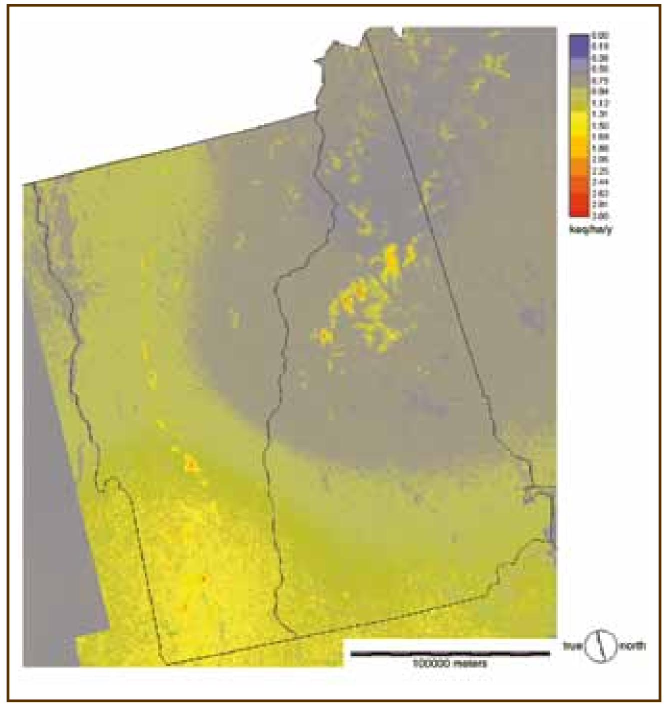

Figure 4—Average annual atmospheric deposition of sulfur and nitrogen (1999-2003). Total deposition (particle + SO_2 + precipitation + cloud water) expressed in terms of kilo-equivalents of charge per hectare per year in this map centered on New Hampshire and Vermont; N deposition includes both ammonium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

deposition (ammonium + nitrate) ranged between 3.3 and 25.2 (average 8.4) kg ha^{-1} y^{-1}, producing an aggregate acidifying and nutrient-leaching potential of 0.43 to 2.7 (average 0.92) keq ha^{-1} 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 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 keq ha^{-1} 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

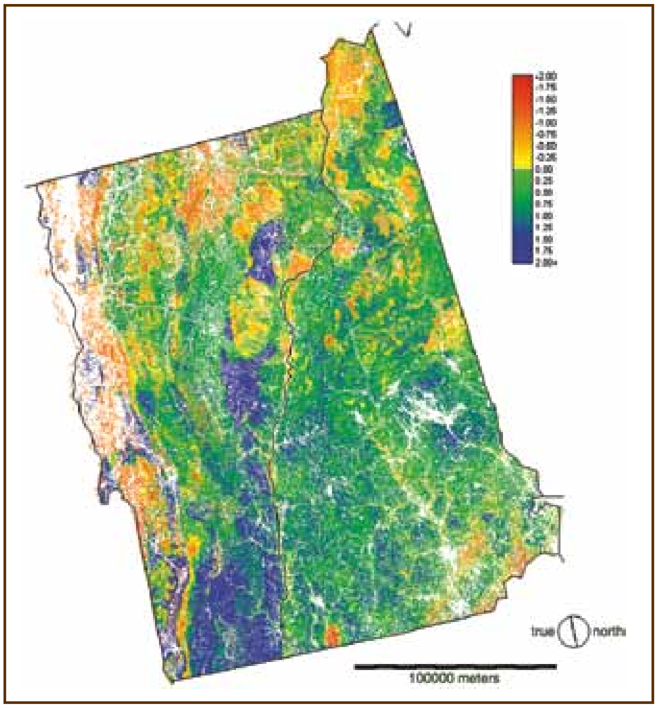

Figure 5—The deposition index for atmospheric sulfur and nitrogen deposition (1999-2003) with respect to Ca depletion. Positive values of the deposition index reflect the capacity of a forest ecosystem to tolerate additional acidic deposition. Negative values of the index correspond to the reduction 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.

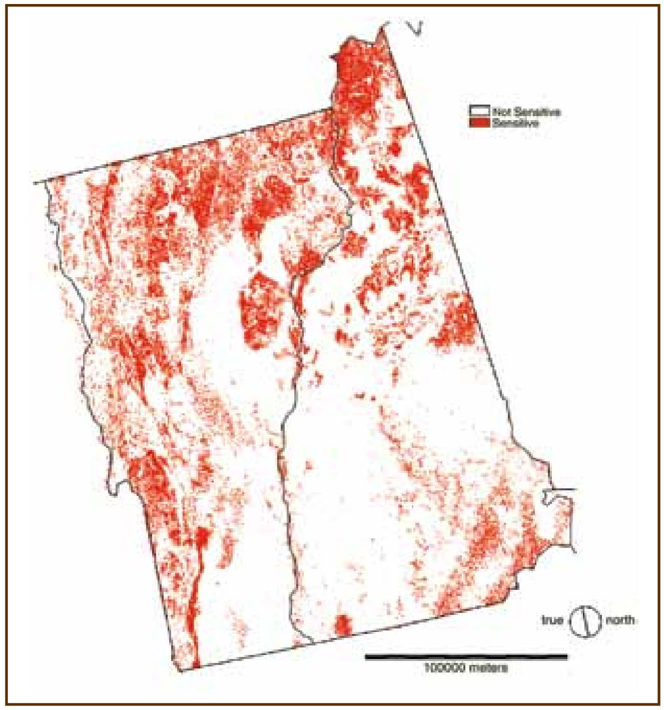

Figure 6—Areas with sulfur plus nitrogen deposition in excess of the critical load. Red-shade shows areas in New Hampshire and Vermont where S+N deposition exceed the critical load, and forests would be likely to experience Ca depletion under 1999–2003 deposition rates.

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).

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 (Nykvist 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

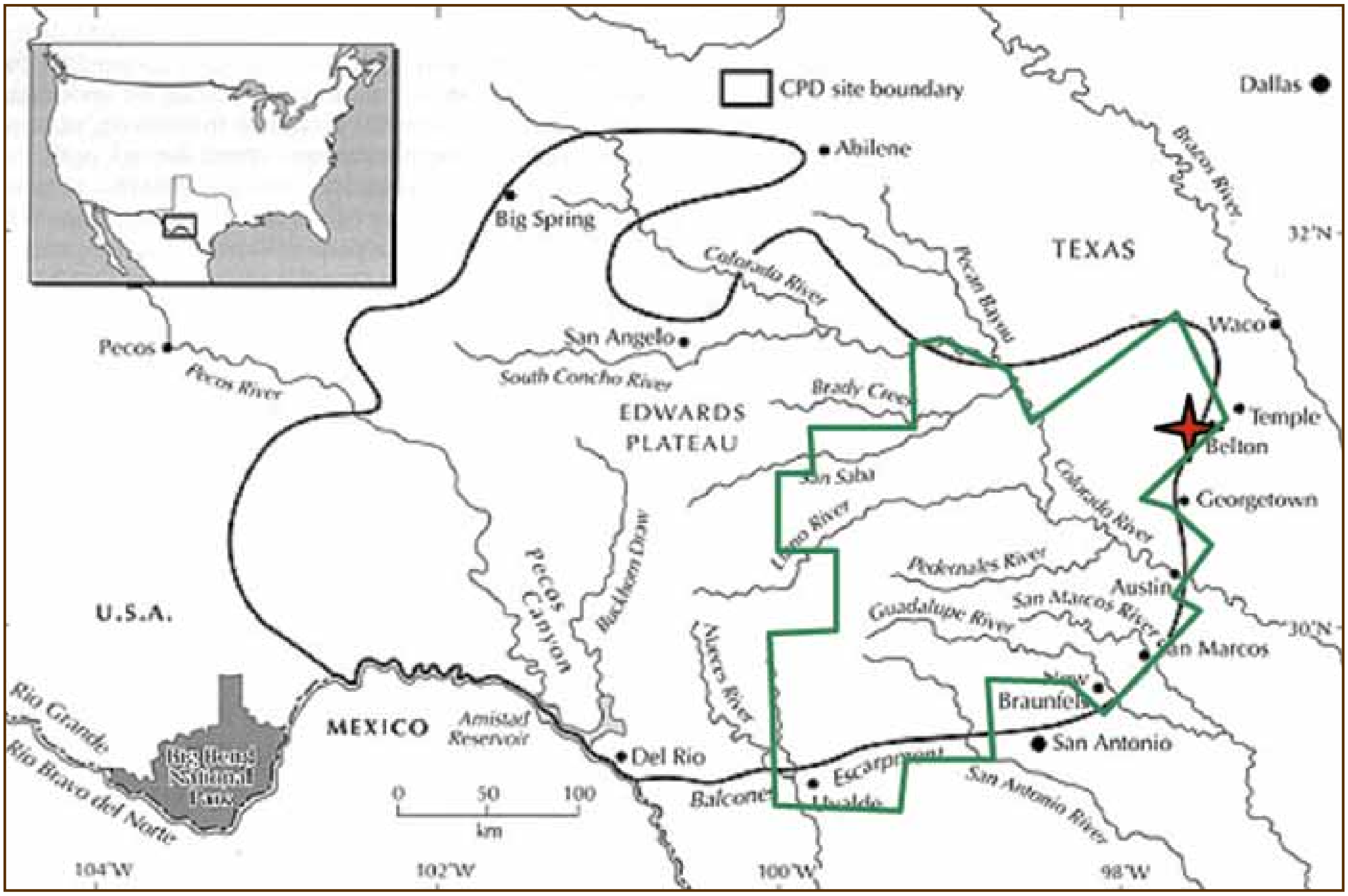

Figure 1—This map shows the region in Texas called the Edwards Plateau. Fort Hood is located with a red star, and the hill country is outlined in green.

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).

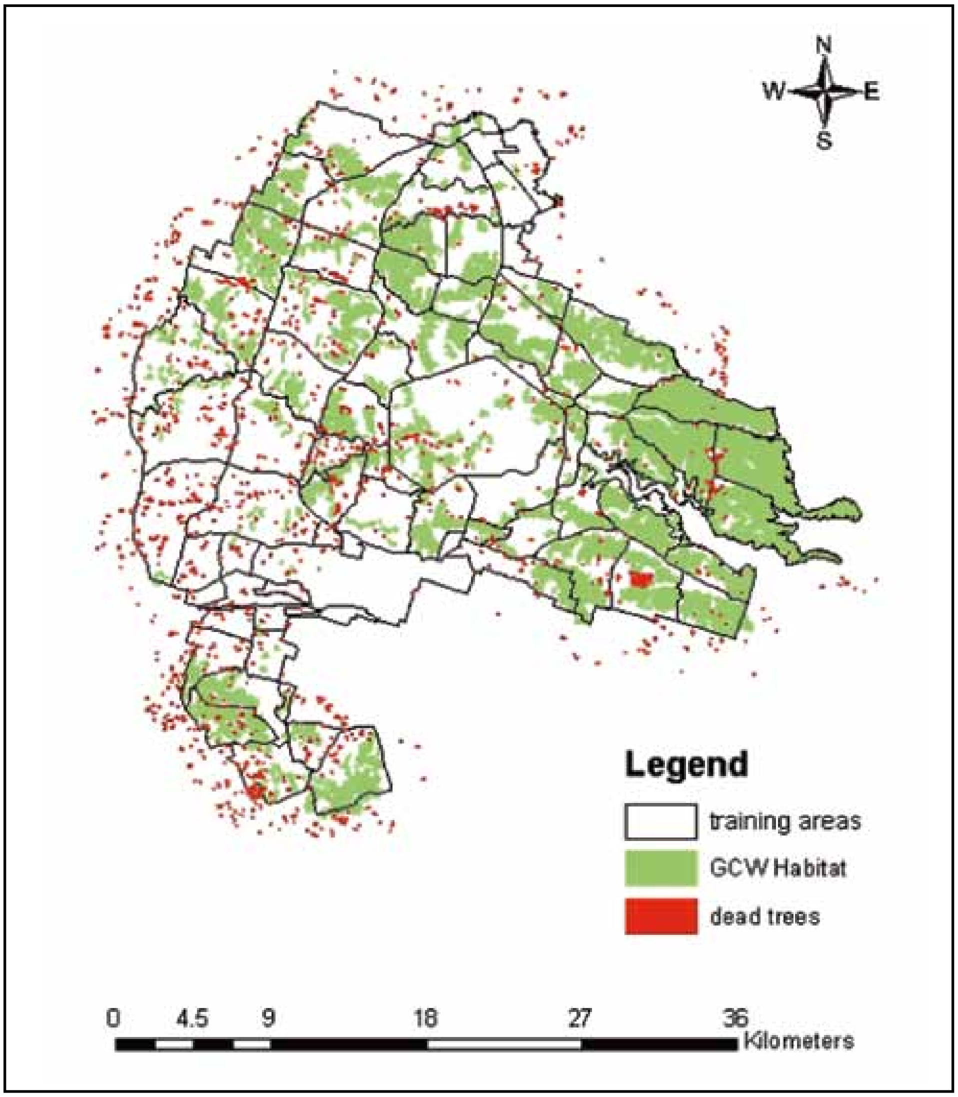

Figure 2—Photointerpreted polygon, training areas, and golden-cheeked warbler habitat at Fort Hood, Texas.

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.

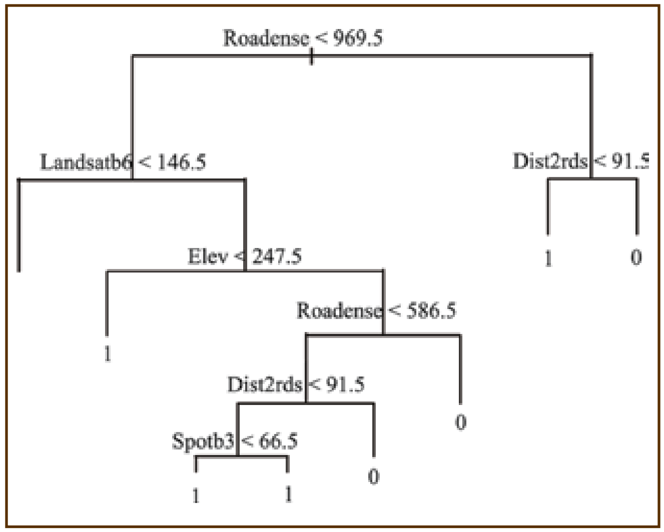

Figure 3—Classification tree model of golden-cheeked warbler (GCW) nesting-site characteristics.

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 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
		Percent
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]
						Inches
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 mi^2 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 mi^2), 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).

R. Billings

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://texasforestservice.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

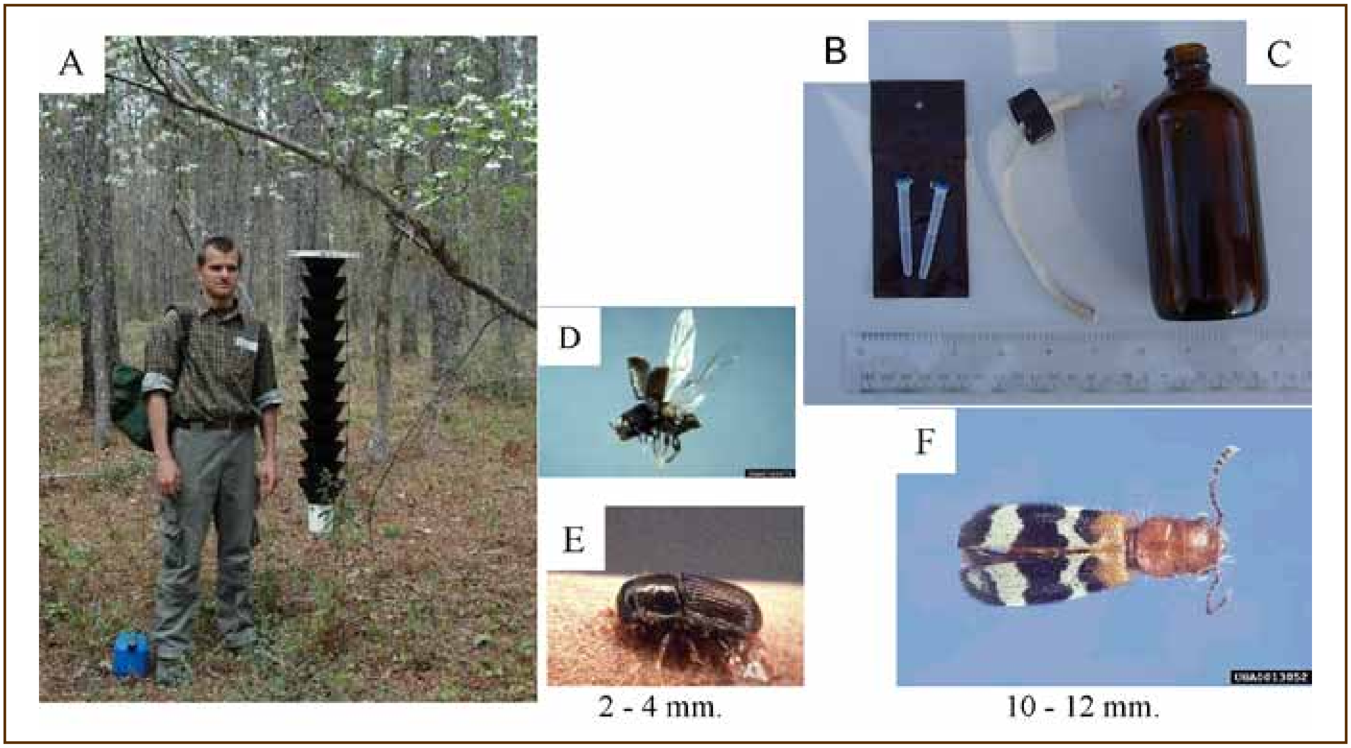

Figure 2—The Southern Pine Beetle Prediction System uses 12-funnel survey traps. The traps are (A) placed in the field for 4 weeks in the spring starting when dogwoods bloom. Each trap is baited with the SPB pheromone frontalin (B) and a bottle of steam-distilled pine turpentine (C), an attractive bait 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 frontalin (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

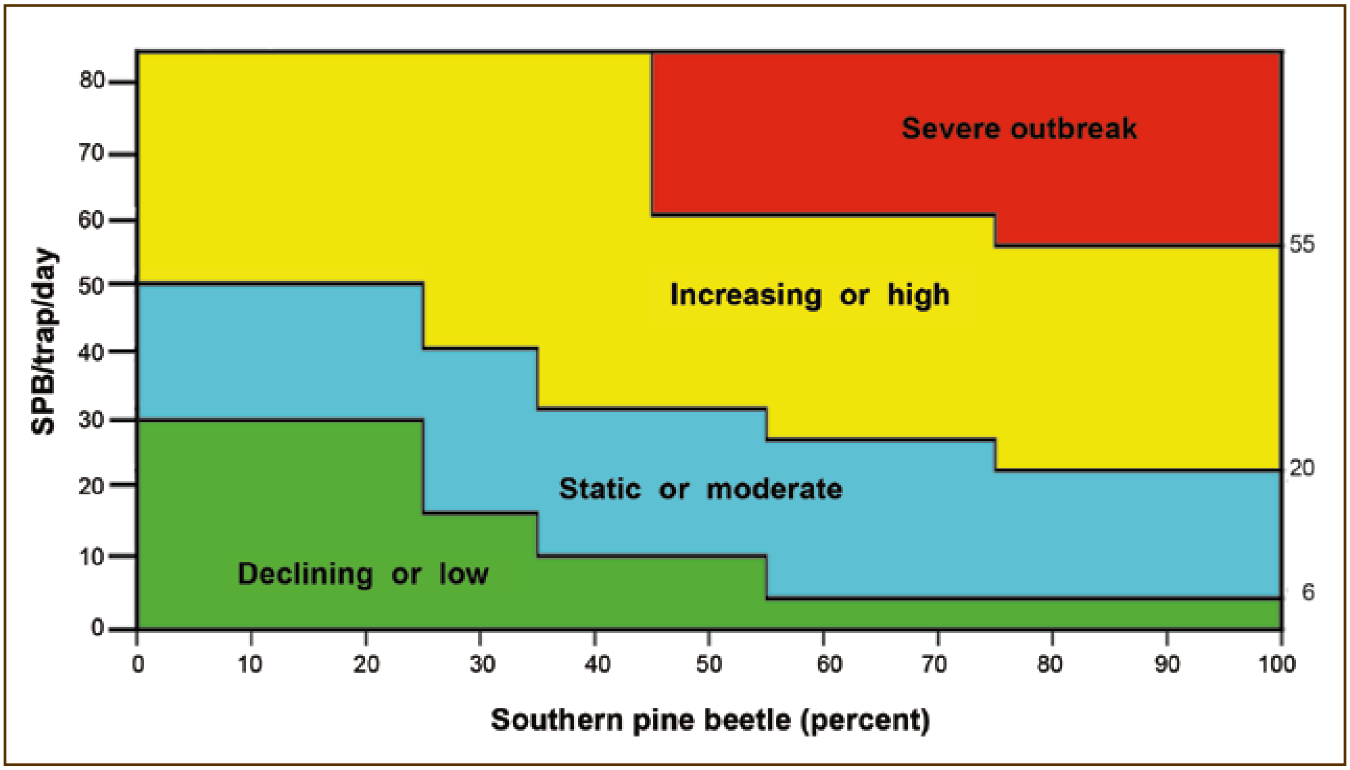

Figure 3—Southern pine beetle prediction chart. The prediction of SPB infestation trend and level for the current year is derived by plotting the mean number of SPB per trap per day and percent SPB for the current year and comparing these data to that for the previous year (if available) for a given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

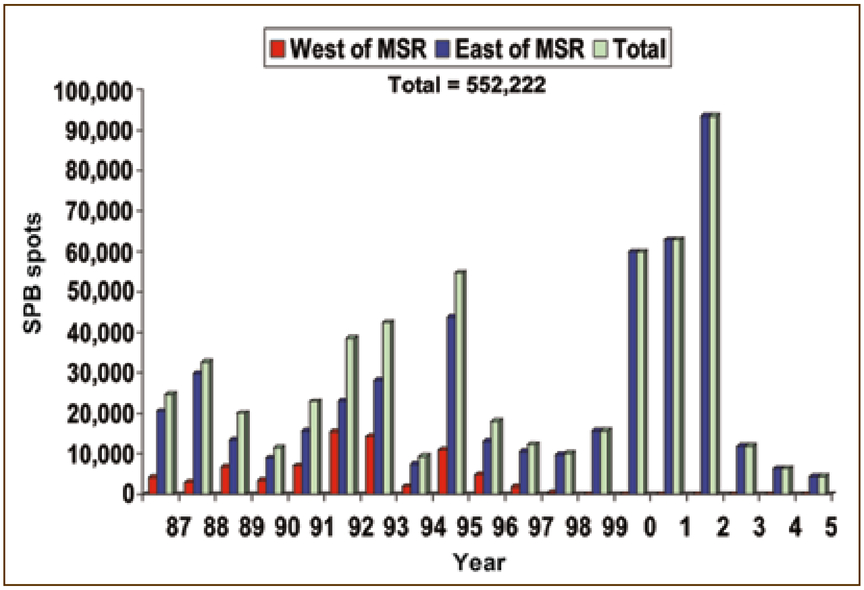

Figure 4—Total numbers of southern pine beetle infestations (spots) detected in 16 States for the period 1987-2005. This figure shows marked differences in infestation levels between those States located east and those located west of the Mississippi River (MSR).

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

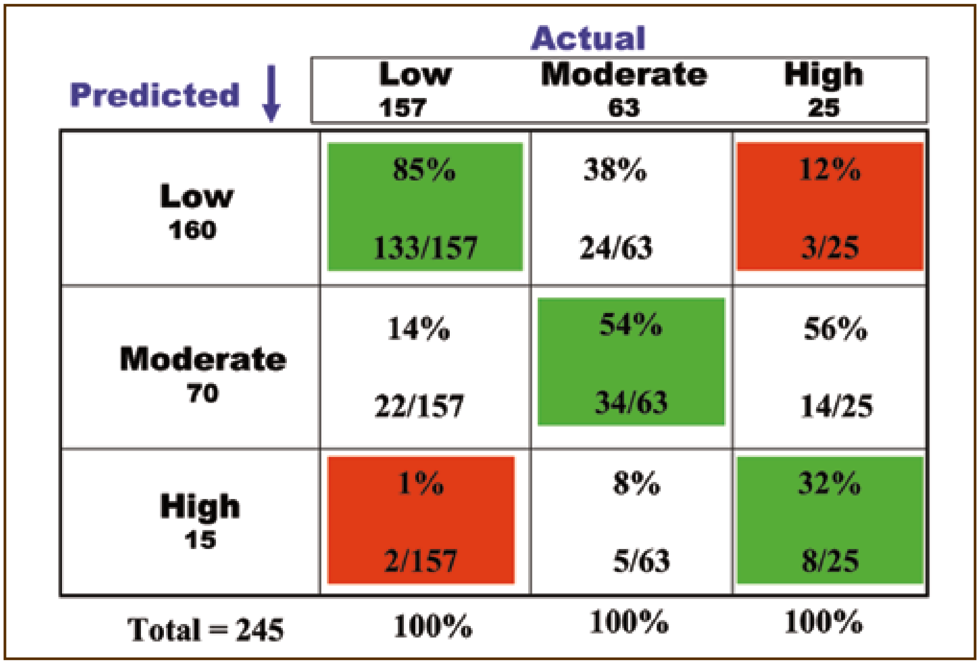

Figure 6—Matrix showing accuracy of SPB population level predictions at the State level, based on 245 predictions over the 19-year period 1987–2005 in 16 States.

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 orthocorrection 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, orthocorrection 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 Information 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 orthocorrection—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. Photography'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 orthocorrect 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 orthocorrection 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 orthocorrection software tools and the online data and camera reports. Also of value are the simple techniques for creating pseudocamera reports.

Although image orthocorrection is not required to detect changes, orthocorrection 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.

Orthocorrection Software Tools

Commercially available orthocorrection 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 orthocorrect large blocks (or sets) of overlapping photos.

Online Data (DOQs, DEMs, and Camera Reports)

In addition to the scanned photography, orthocorrection 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 orthocorrection.

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 orthocorrected 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 orthocorrected, 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 orthocorrected using the orthocorrection tools in ArcMap 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 orthocorrect photography from 1967, 1976, 1988, and 1995 over a small portion of the Fishlake National Forest. In addition, we added orthocorrected NAIP imagery from 2004. The

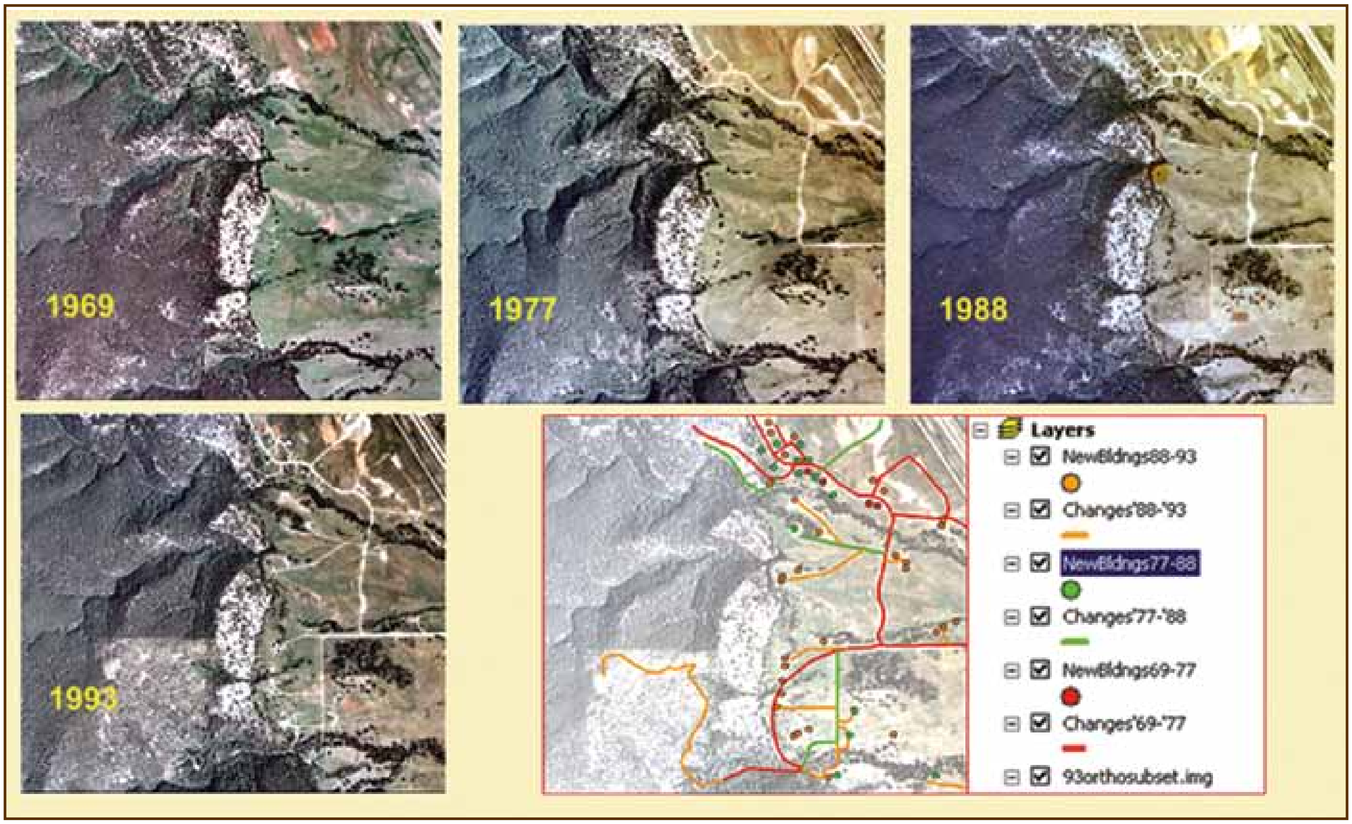

Figure 1—Aerial photography from 1969, 1977, 1988, and 1993 over a portion of the Black Hills National Forest was orthocorrected. New roads and buildings were identified and digitized for each cycle of photography. The result (bottom-right of graphic) was a compelling graphical documentation of 24 years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

1967 photography did not have a camera report, so we created one using methods described in above. Once all of the photography was orthocorrected, 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

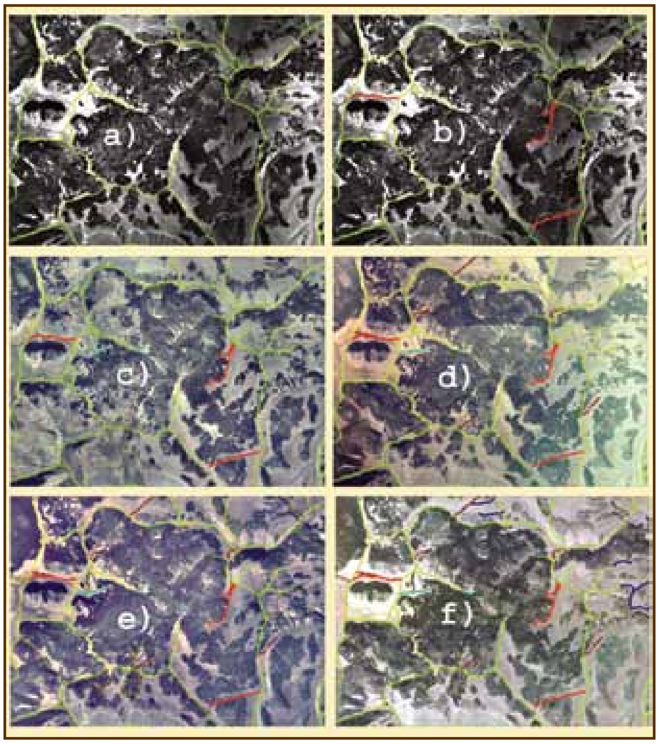

Figure 2—Aerial photography from 1967 (Figures 2a and 2b), 1976 (Figure 2c), 1988 (Figure 2d), and 1995 (Figure 2e) was orthocorrected, and 2004 NAIP photography (Figure 2f) was obtained to assess unmanaged recreation over a portion of the Fishlake National Forest. The official managed trails were obtained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

Forest Service and provided behind the Forest Service firewall.

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.

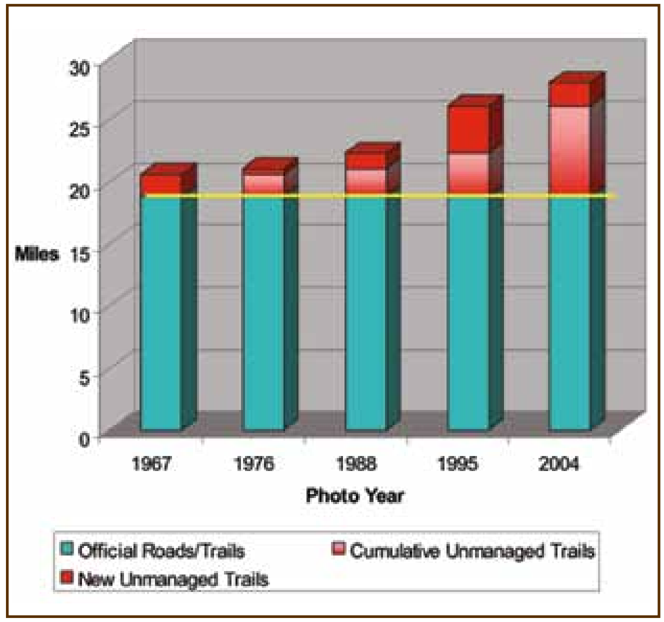

Figure 3—This bar graph documents the increase in unmanaged OHV trails for our project area on the Fishlake National Forest, 1967-2004. Official managed trails are graphed in cyan (and highlighted with the yellow line), newly interpreted unmanaged trails are shown in solid red, and the cumulative unmanaged 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

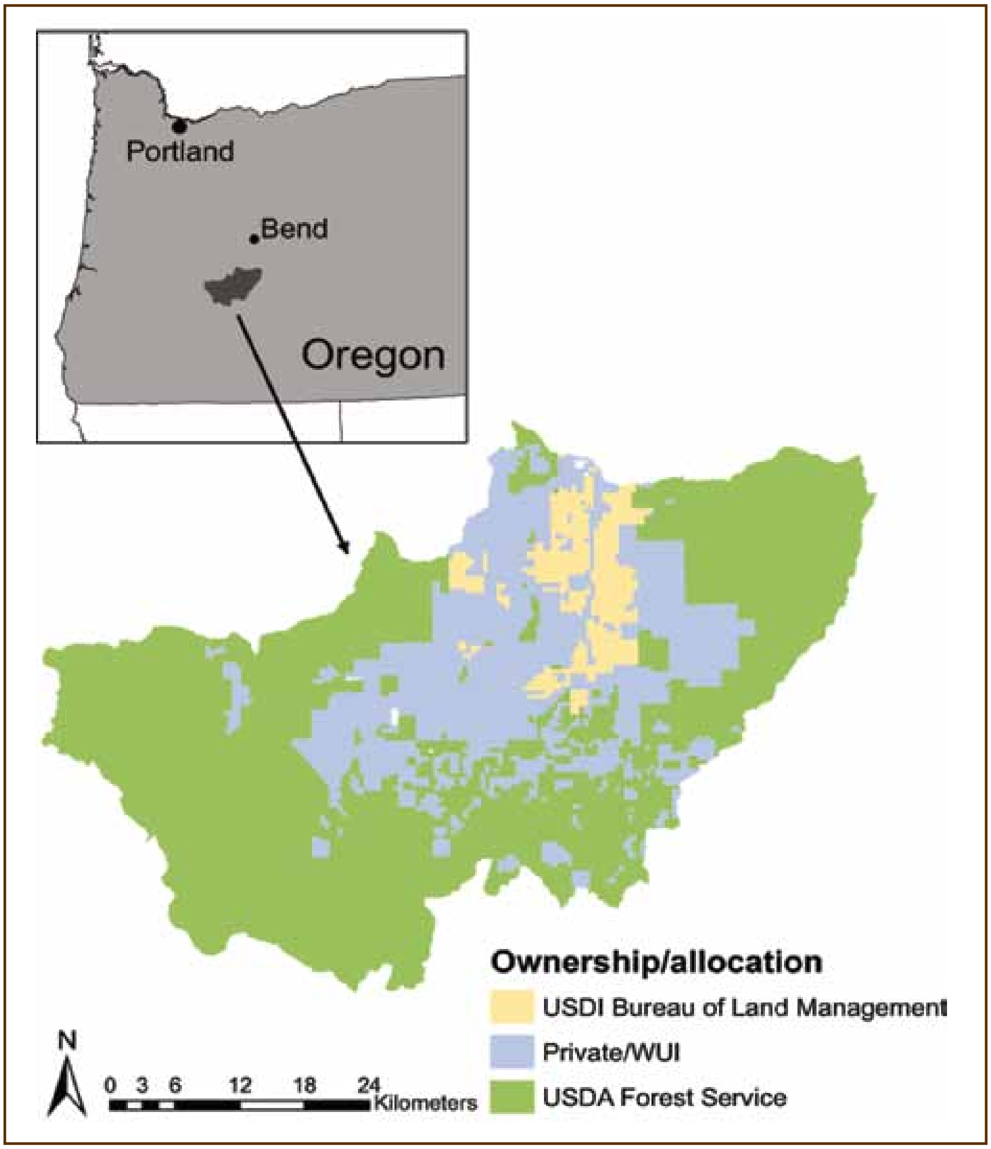

Figure 1—The Five Buttes study area in central Oregon, U.S.A. WUI = wildland-urban interface.

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.
		Percentage	*cm*
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	≥ 1 3 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 (*Juniperus occidentalis* Hook.)
Ponderosa pine	Ponderosa pine (*Pinus ponderosa* P. & C. Lawson)
Douglas-fir/white fir	Douglas-fir (*Pseudotsuga menziesii* (Mirbel) Franco) and white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.)
Grand fir	Grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.), Douglas-fir, and other conifers
Lodgepole pine	Lodgepole pine (*Pinus contorta* Dougl. ex Loud.)
Pacific silver fir	Pacific silver fir (*Abies amabilis* (Dougl. ex Loud.) Dougl. ex Forbes), noble fir (*Abies procera* Rehd.), and Douglas-fir
Mixed conifer	Variable mixtures of white fir, Douglas-fir, Engelmann spruce (*Picea engelmannii* Parry ex Engelm.), mountain hemlock (*Tsuga mertensiana* (Bong.) Carr.), and other conifers at upper elevations
Subalpine parkland	Mosaic of subalpine fir (*Abies lasiocarpa* (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 (Cattelino 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
		Years		
Juniper	Replacement	1000	0.001	R#JUPIse 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 multistoried 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.

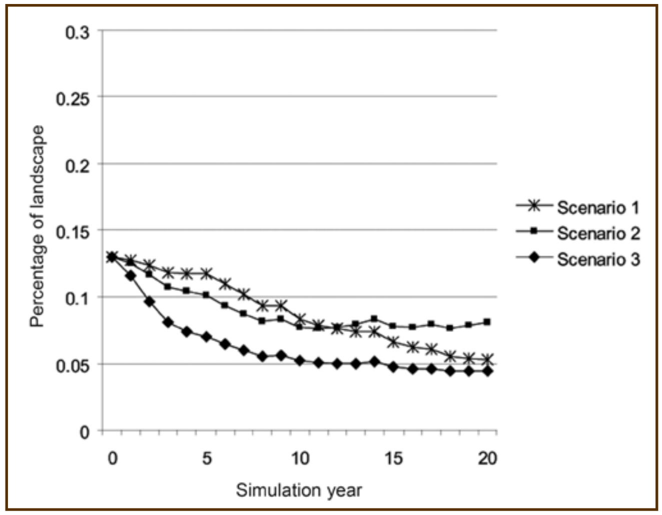

Figure 2—Percentage of the landscape in multistory large and very large tree structure classes by scenario in the Five Buttes area, central Oregon, U.S.A.

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

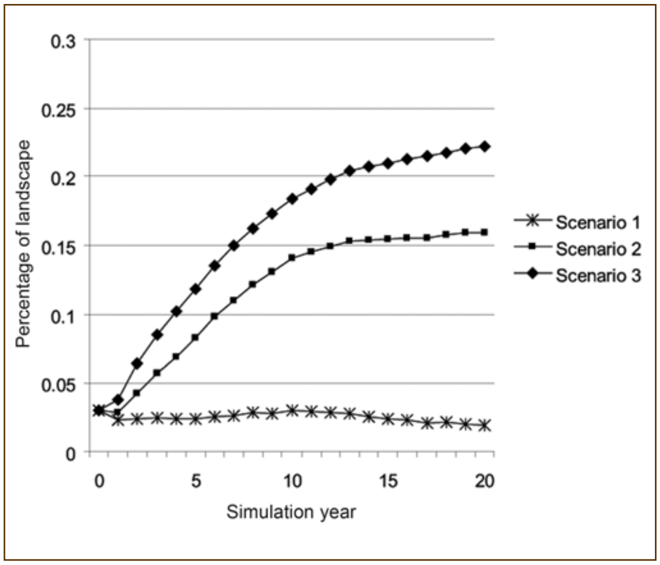

Figure 3—Percentage of the landscape in single-story very large and large tree structure classes by scenario in the Five Buttes area, central Oregon, U.S.A.

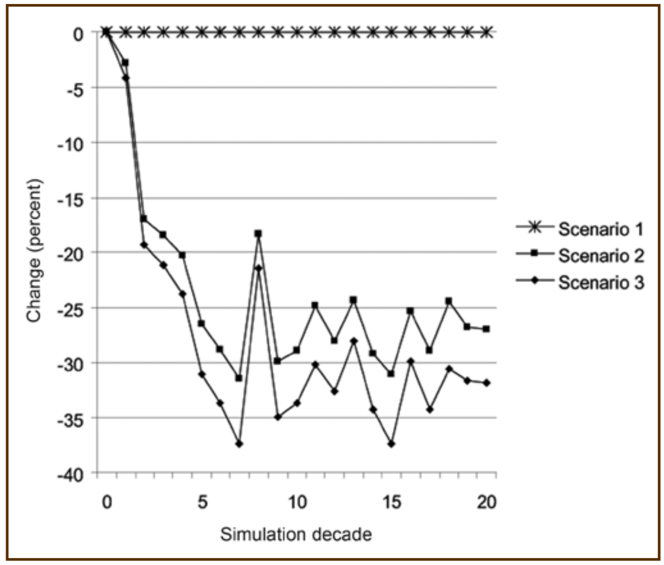

Figure 4—Percentage change in average decadal area affected by high-severity wildfire under scenarios 2 and 3 compared to scenario 1 in the Five Buttes study area, central Oregon, U.S.A.

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).

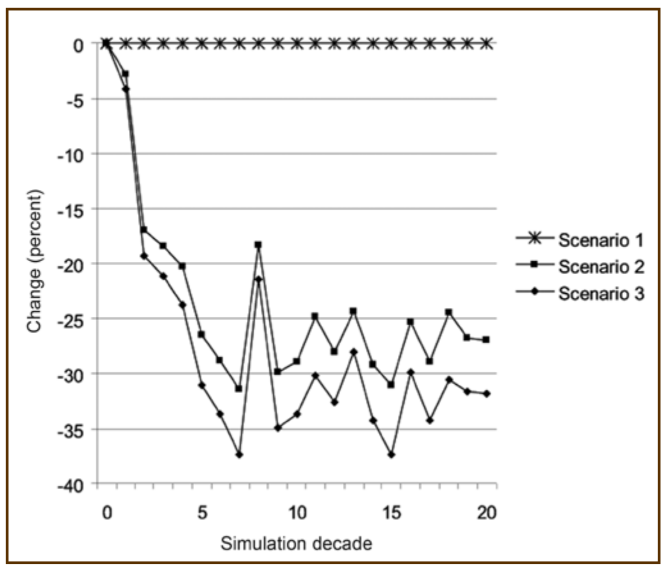

Figure 5—Average decadal area treated with mechanical fuel treatments and thinnings by scenario in the Five Buttes area, central Oregon, U.S.A.

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

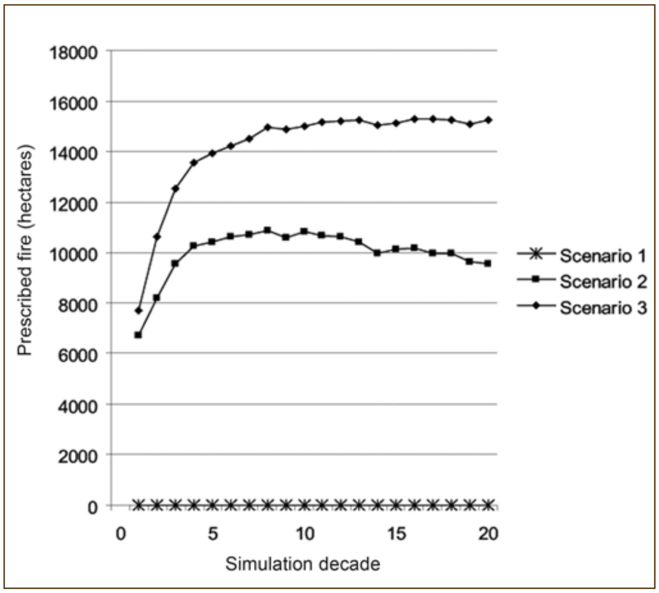

Figure 6—Average decadal area treated with prescribed fire by scenario in the Five Buttes area, central Oregon, U.S.A.

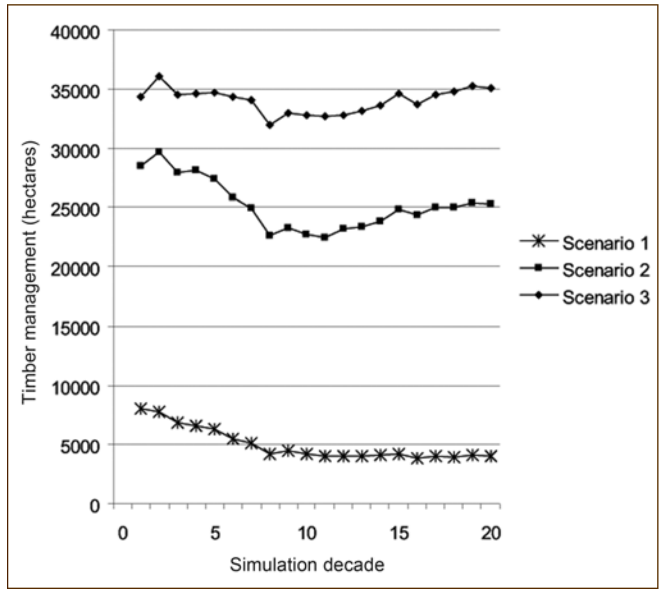

Figure 7—Average decadal area treated with management activities that may produce commercial timber products by scenario in the Five Buttes study area, central Oregon, U.S.A.

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

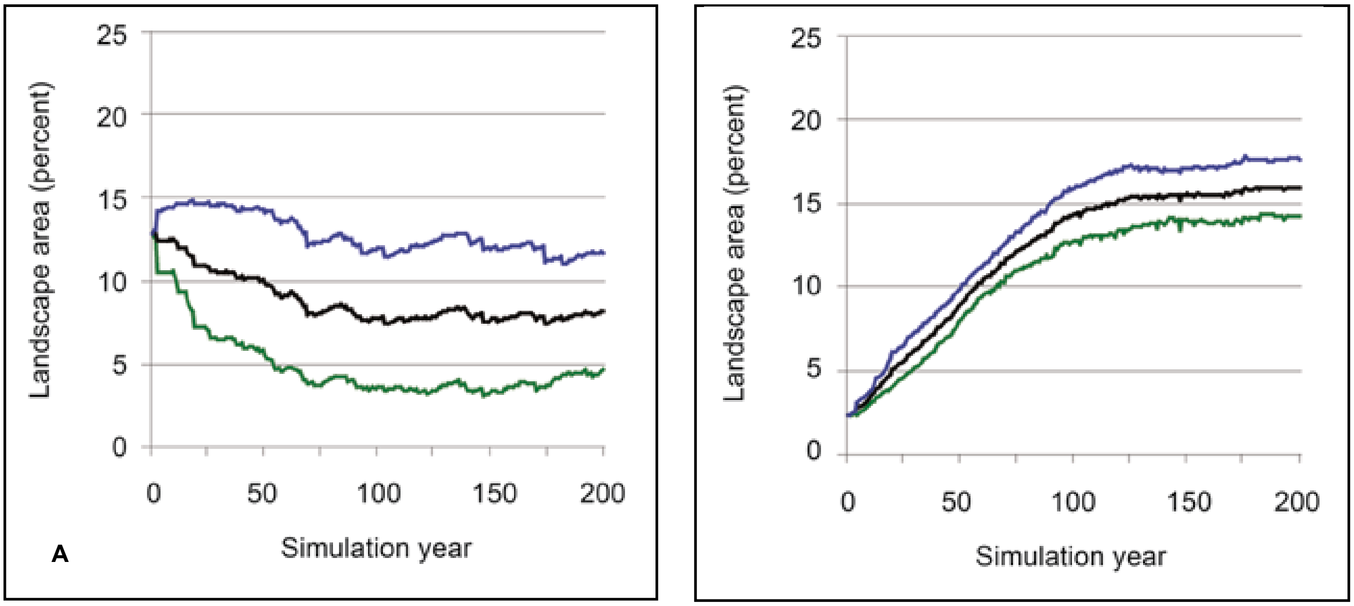

Figure 8—Variation in amounts of multi- and single-story large- and very-large-tree forests under scenario 2 for 30 Monte Carlo simulations in the Five Buttes area, central Oregon, U.S.A. Upper and lower lines are plus and minus one standard deviation from the mean.

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

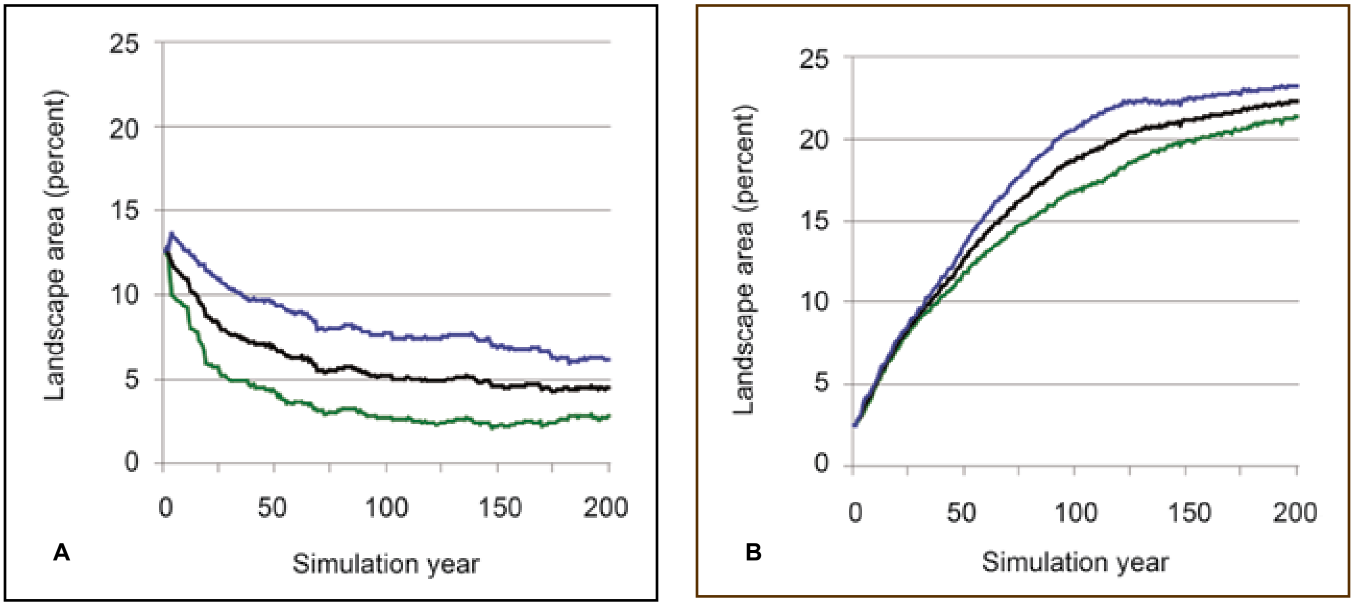

Figure 9—Variation in amounts of multi- and single-story large- and very-large-tree forests under scenario 3 for 30 Monte Carlo simulations in the Five Buttes area, central Oregon, U.S.A. Upper and lower lines are plus and minus one standard deviation from the mean.

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 thinnings 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

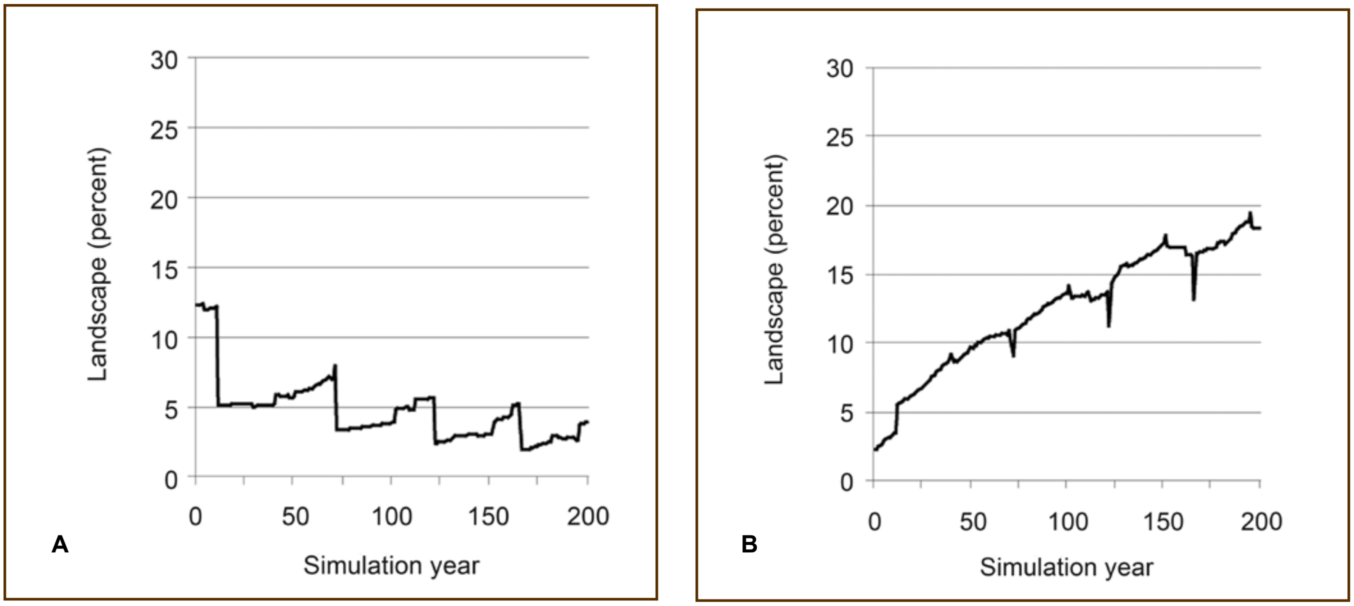

Figure 10—One randomly selected example simulation run showing amounts of multi- and single-story large- and very-large-tree forests under scenario 2 in the Five Buttes area, central Oregon, U.S.A.

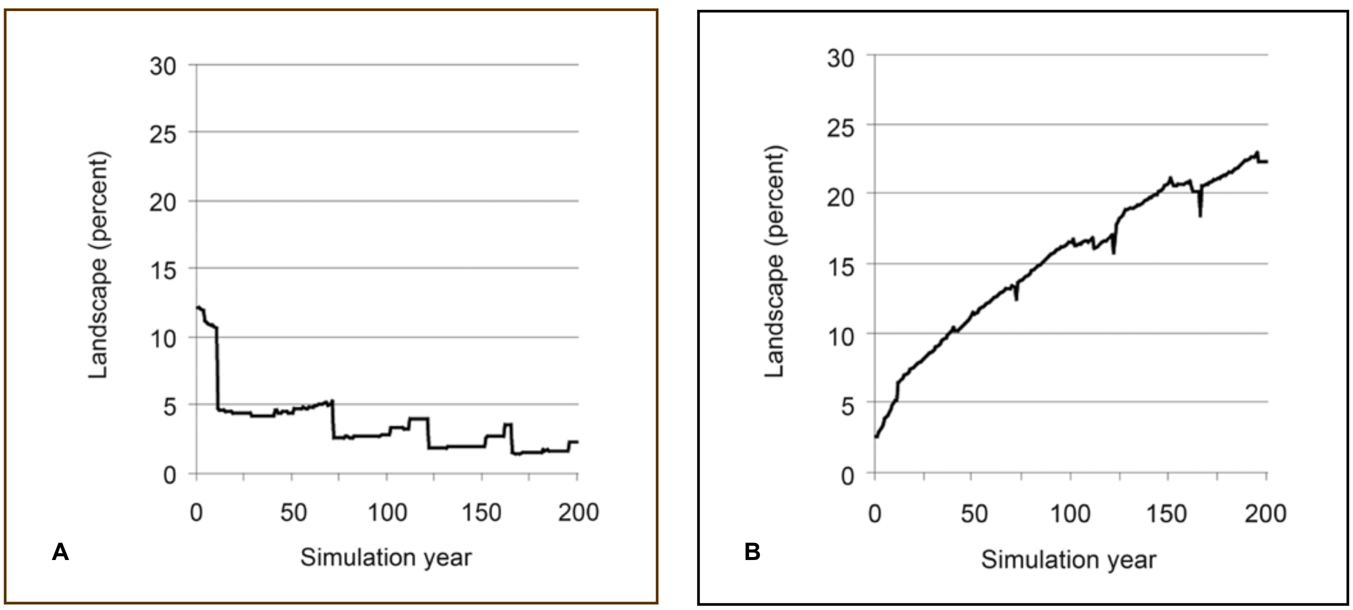

Figure 11—Variation in amounts of multistory large- and very-large-tree forests under Scenario 3 in the Five Buttes area, central Oregon, U.S.A.

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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