Forest Health Monitoring: National Status, Trends, and Analysis 2010

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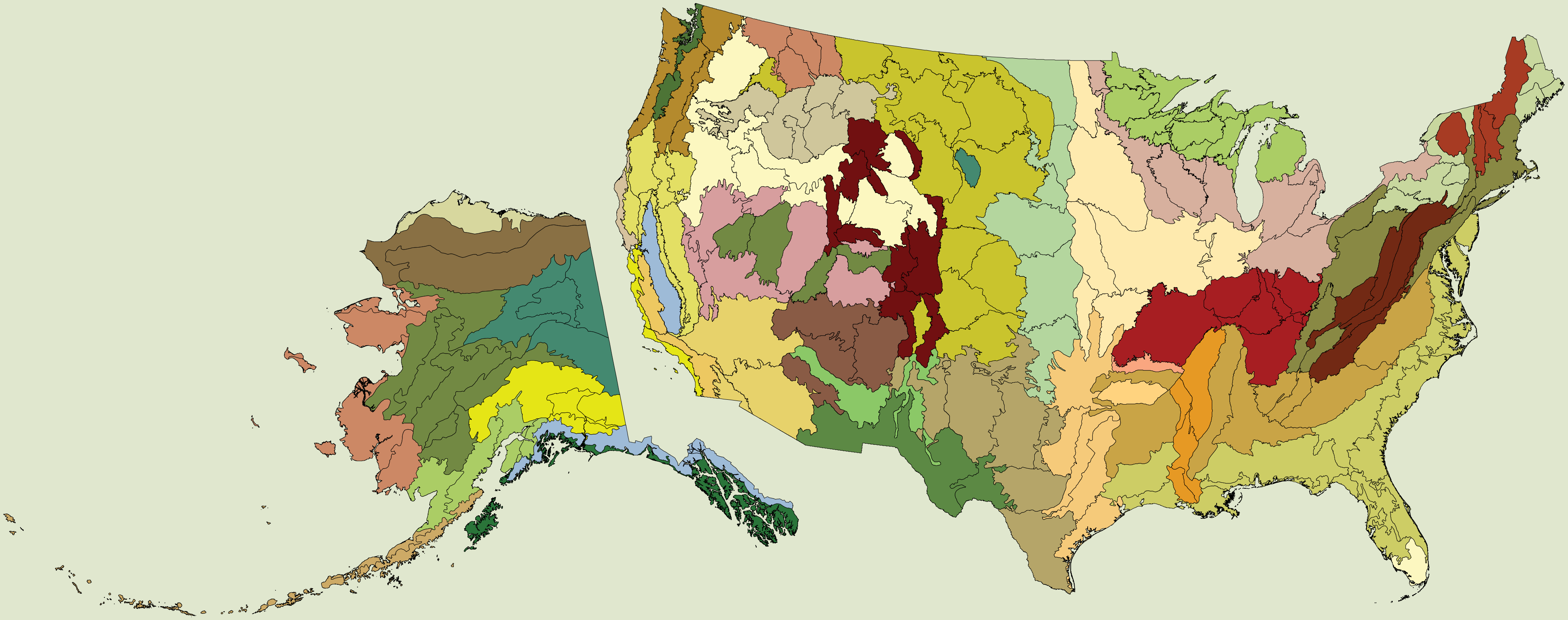

United States Department of Agriculture
Forest Service
Research & Development
Southern Research Station
General Technical Report SRS-176

Front cover map: Ecoregion provinces and ecoregion sections for the conterminous United States (Cleland and others 2007) and for Alaska (Nowacki and Brock 1995).

Back cover map: Forest cover (green) backdrop derived from Moderate Resolution Imaging Spectroradiometer (MODIS) satellite imagery by the U.S. Forest Service Remote Sensing Applications Center.

June 2013

Southern Research Station
200 W.T. Weaver Blvd.
Asheville, NC 28804

www.srs.fs.usda.gov

Forest Health Monitoring: National Status, Trends, and Analysis 2010

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ABSTRACT

The annual national report of the Forest Health Monitoring Program of the Forest Service, U.S. Department of Agriculture, presents forest health status and trends from a national or multi-State regional perspective using a variety of sources, introduces new techniques for analyzing forest health data, and summarizes results of recently completed Evaluation Monitoring projects funded through the national Forest Health Monitoring Program. In this 10th edition in the annual series of national reports, survey data are used to identify geographic patterns of insect and disease activity. Satellite data are employed to detect geographic clusters of forest fire occurrence. Data collected by the Forest Inventory and Analysis Program of the Forest Service are employed to detect regional differences in tree mortality. Established forest fragmentation assessment protocols are used to characterize and compare the fragmentation of landcover types nationally. A new methodology for the comparison of moisture conditions among different geographical areas and time periods is described. Forest Inventory and Analysis data are used to conduct an empirical assessment of the Nation's standing dead tree resources. The potential impacts of climate change on forest soil critical acid load limits are explored. Seven recently completed Evaluation Monitoring projects are summarized, addressing forest health concerns at smaller scales.

Keywords—Drought, fire, forest health, forest insects and disease, fragmentation, nonnative species, tree mortality.

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EXECUTIVE SUMMARY

Healthy ecosystems are those that are stable and sustainable, and are able to maintain their organization and autonomy over time while remaining resilient to stress (Costanza 1992). The Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture, with cooperating researchers within and outside the Forest Service and with State partners, quantifies the health of U.S. forests (chapter 1). The analyses and results outlined in sections 1 and 2 of this FHM annual national report offer a snapshot of the current condition of U.S. forests from a national or multi-State regional perspective, incorporating baseline investigations of forest ecosystem health, examinations of change over time in forest health metrics, and assessments of developing threats to forest stability and sustainability. For datasets collected on an annual basis, analyses are presented from 2009 data. For datasets collected over several years, analyses are presented at a longer temporal scale. Several chapters describe new techniques for analyzing forest health data as well as new applications of established techniques. Finally, section 3 of this report presents summaries of results from recently completed Evaluation Monitoring (EM) projects that have been funded through the FHM national program to determine the extent, severity, and/or causes of specific forest health problems (FHM 2010).

Monitoring the occurrence of forest pest and pathogen outbreaks is important at regional scales because of the significant impact insects and disease can have on forest health across landscapes (chapter 2). Survey data by the Forest Health Protection Program of the Forest Service from 2009 identified 62 different mortality-causing agents and complexes on nearly 4.69 million ha of forest in the conterminous United States. Similarly, defoliation damage attributed to 64 agents and complexes affected approximately 3.17 million ha. Significant large geographic hot spots of forest mortality were associated with mountain pine beetle in the West, while a smaller hot spot was associated with bronze birch borer in the northern Midwest. The largest hot spots of defoliation were also in the West, caused primarily by western spruce budworm. Forest tent caterpillar was associated with hot spots of defoliation in the East. The Northern spruce engraver beetle was the most important cause of mortality in Alaska, while the aspen leafminer was the most important defoliation agent there.

Forest fire occurrence outside the historic range of frequency and intensity can result in extensive economic and ecological impacts. The detection of regional patterns of fire occurrence can allow for the identification of areas at greatest risk of significant impact and for the selection of locations for more intensive analysis (chapter 3). In 2009, the Central California Coast experienced the most fires per 100 km^2 of forested area, while ecoregions in the southern plains of Oklahoma and Texas also had high densities of forest fire occurrence. In Alaska, an extremely high density of forest fires occurred in the Yukon Flats ecoregion. Several geographical hot spots of fire occurrence were located near the Pacific Coast and throughout the coastal plains of the South. The pattern and degree of clustering of these hot spots suggests that fires were more evenly distributed across the conterminous United States than in recent years, with slightly higher concentrations in a few areas.

Mortality is a natural process in all forested ecosystems, but high levels of mortality at large scales may indicate that the health of forests is declining. Phase 2 data collected by the Forest Inventory and Analysis (FIA) Program of the Forest Service offer tree mortality information at a more spatially intense sample than the FHM and FIA phase 3 data used in past forest health annual reports (chapter 4). An analysis of FIA plots from 27 States found that the highest ratios of annual mortality to gross growth occurred in ecoregion sections of the Plains States. Mortality was also high in parts of southeastern Texas and New York. Even in areas of relatively high mortality relative to growth, no mortality occurred on most sample plots. However, on the plots with mortality, larger-diameter trees exhibited higher mortality trends as compared to remaining smaller-diameter classes of surviving trees, suggesting the mortality is related to either senescence of older stands or some insect or disease issue.

The spatial arrangement of an environment affects human perceptions and ecological processes within that environment, but that usually happens in competing ways. A prerequisite for maintaining appropriate amounts and patterns of different landcover types at local, regional, and national scales is reliable information about landcover patterns at those scales. To that end, established protocols for assessing forest fragmentation were used to characterize and compare the fragmentation of forest, grassland, and shrubland landcover types from the 2001 National Land Cover Database (NLCD) national landcover map (chapter 5). This analysis forms a baseline for trend assessments when the comparable NLCD 2006 national landcover map becomes available. The analyses were conducted at six measurement scales, and the results summarized nationally and for five multi-State assessment regions.

Most U.S. forests experience droughts, with varying degrees of intensity and duration between and within forest ecosystems. Arguably, the duration of a drought event is more critical than its intensity, so it was important to revise a previously described methodology to examine moisture conditions in the United States across multi-year windows of time (chapter 6). Specifically, the moisture index difference (MID) methodology was standardized to allow for the comparison, for any given location, of drought status during different time windows regardless of their length, e.g., allowing comparison between 1-year and 5-year time windows, and during a single season rather than an entire year. Such analyses may have great relevance when estimating the risks associated with certain forest pests that are able to exploit acute drought stress in host trees, such as the oak splendor beetle (*Agrilus biguttatus*), a buprestid beetle found throughout Europe but considered a major threat to North American oak forests.

Given the importance of standing dead trees to numerous forest ecosystem attributes/

processes such as fuel loadings and wildlife habitat, the FIA program initiated a consistent nationwide inventory of standing dead trees in 1999. One of the first empirical assessments of the Nation's standing dead tree resources was conducted as the first cycle of standing dead tree inventories neared full national implementation (chapter 7). Results indicate that there are more than 10 billion standing dead trees in forests across the United States, most of them < 30 cm in diameter. Forests in the Rocky Mountains and Pacific Northwest have some of the largest mean biomass of standing dead trees per unit of forest land, while Plains States had the least. More than 130 species have greater than 1 million Mg of standing dead biomass nationwide. Given the emerging role of standing dead trees in biomass/bioenergy economies and carbon cycling, continued monitoring of this resource is highly warranted.

A forest ecosystem is considered to be at risk for health impairment when its soil critical acid load exceeds a level known to impair forest health. Higher exceedances, or excess loads, are expected to result in greater risk of ecosystem damage. A study explores which factors associated with establishing forest soil critical acid load limits will most likely be influenced by climate change, and how these changes might impact forest soil critical acid load limits across the United States (chapter 8). Base cation weathering could increase with global warming, along with nitrogen uptake as a function of increased forest growth across New England. Nationally, a moderate 20 percent increase in base cation weathering and nitrogen uptake would result in a 30.5 percent or greater decrease in the forest soil area that exceeds the critical acid load limit and a 64.4 percent or greater decrease in the area with high exceedance. While these results are encouraging, they do not account for other negative potential forest health risks associated with climate change.

Finally, seven recently completed EM projects address a wide variety of forest health concerns at a scale smaller than the national or multi-State regional analyses included in the first sections of the report. These EM projects, funded by the FHM program:

- Measured the amount of wildland forest that has been converted to other land uses in western Washington over the last 30 years, and quantified the degree to which the remaining forest has been fragmented (chapter 9)
- Compared the occurrence of yellow-cedar decline in Alaska as mapped by aerial survey and aerial photographs; determined the association between decline and elevation, aspect, and slope; and developed methods to detect healthy yellow-cedar populations (chapter 10)
- Quantified the amount of lodgepole pine mortality in stands infested by mountain pine beetle in Colorado, in addition to identifying differences in stand characteristics, tree species composition, and fire fuels before and 7 years after infestation (chapter 11)
- Measured the extent and severity of damage and mortality to subalpine fir and other true firs from balsam woolly adelgid throughout eastern Washington and Oregon, and assessed

management options for ensuring the long-term viability of subalpine fir (chapter 12)

- Evaluated and established baseline information on the health status of whitebark pine in Idaho, particularly with regard to the levels of mortality from white pine blister rust and mountain pine beetle (chapter 13)
- Determined the extent of larch mortality following larch sawfly infestation in Alaska, collected baseline forest health information to assess larch establishment and regeneration success, and identified risk factors associated with repeated infestation (chapter 14)
- Investigated whether fire damage could serve as an important indicator of bark beetle-related mortality in prescribed burn sites in ponderosa pine stands in northern Arizona (chapter 15).

The FHM program, in cooperation with forest health specialists and researchers inside and outside the Forest Service, continues to investigate a broad range of issues relating to forest health using a wide variety of data and techniques. This report presents some of the latest results from ongoing national-scale Detection Monitoring and smaller-scale EM efforts by FHM and its cooperators. For more information about efforts to determine the status, changes, and trends in indicators of the condition of U.S. forests, please visit the FHM Web site at www.fs.fed.us/foresthealth/fhm.

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CHAPTER 1

Introduction

Kevin M. Potter[1]

Forests cover a vast area of the United States, 304 million ha or approximately one-third of the Nation's land area (Smith and others 2009). These forests possess substantial ecological and socioeconomic importance. Both their ecological integrity and their continued capacity to provide goods and services are of concern in the face of a long list of threats, including insect and disease infestation, fragmentation, catastrophic fire, invasive species, and the effects of climate change.

Assessing and monitoring the health of these forests are critical and challenging tasks. While there is no universally accepted definition of forest health, the current understanding of ecosystem dynamics suggests that healthy ecosystems are those that are able to maintain their organization and autonomy over time while remaining resilient to stress (Costanza 1992), and that evaluations of forest health should emphasize factors that affect the inherent processes and resilience of forests (Kolb and others 1994, Raffa and others 2009). This national report, the 10th in an annual series produced by the Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture, attempts to quantify the status of, changes to, and trends in a wide variety of such indicators of forest health. These indicators encompass forest insect and disease activity, wildland fire occurrence, tree mortality, forest fragmentation, drought, standing dead tree resources, and forest soil critical acid loads.

This report has three specific objectives. The first is to present information about forest health from a national perspective, or from a multi-State regional perspective when appropriate, using data collected by the Forest Health Protection (FHP) and Forest Inventory and Analysis (FIA) programs of the Forest Service, as well as from other sources available at a wide extent. The chapters that present analyses at a national-scale, or multi-State regional scale, are divided between section 1 and section 2 of the report. Section 1 presents results from the analyses of forest health data that are available on an annual basis, allowing for the detection of trends over time and changes from one year to the next. Section 2 presents longer-term forest health trends, in addition to describing new techniques for analyzing forest health data at national or regional scales (the second objective of the report). While in-depth interpretation and analysis of specific geographic or ecological regions are beyond the scope of these parts of the report, the chapters in sections 1 and 2 present information that can be used to identify areas that may require investigation at a finer scale.

The second objective of the report is to present new techniques for analyzing forest health data as well as new applications of established techniques, presented in selected chapters of section 2. Examples in this report are chapter 6, which describes a newly developed drought index methodology that allows for the comparison of moisture conditions between geographical areas and across periods of time; and chapter 7, which is one of the first empirical assessments of the Nation's standing dead tree resources, using FIA phase 2 data.

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The third objective of the report is to present results of recently completed Evaluation Monitoring (EM) projects funded through the national FHM program. These project summaries, presented in section 3, determine the extent, severity, and causes of forest health problems (FHM 2010), generally at a finer scale than that addressed by the analyses in sections 1 and 2. Each chapter in section 3 contains an overview of an EM project, key results, and contacts for more information.

Where appropriate throughout this report, authors use Bailey's revised ecoregions (Cleland and others 2007, Nowacki and Brock 1995) as a common ecologically based spatial framework for their forest health assessments (fig. 1.1). Specifically, when the spatial scale of the data and the expectation of an identifiable pattern in the data are appropriate, authors use ecoregion sections or provinces as assessment units for their analyses. In Bailey's hierarchical system, the two broadest ecoregion scales, domains and divisions, are based on large ecological climate zones, while each division is broken into provinces based on vegetation macro features (Bailey 1995). Provinces are further divided into sections, which may be thousands of square kilometers in extent and are expected to encompass regions similar in their geology, climate, soils, potential natural vegetation, and potential natural communities (Cleland and others 1997).

Alaska ecoregion provinces

🟩 Alaska Mixed Forest (213)
🟨 Alaska Range Taiga (135)
🟫 Aleutian Meadow (271)
🟨 Arctic Tundra (121)
🟧 Bering Sea Tundra (129)
🟫 Brooks Range Tundra (125)
🟦 Pacific Coastal Icefields (244)
🟩 Pacific Gulf Coast Forest (245)
🟩 Upper Yukon Taiga (139)
🟩 Yukon Intermontaine Taiga (131)

Conterminous States ecoregion provinces

🟥 Adirondack-New England Mixed Forest - Coniferous Forest - Alpine Meadow (M211)
🟨 American Semi-Desert and Desert (322)
🟩 Arizona-New Mexico Mountains Semi-Desert - Open Woodland - Coniferous Forest - Alpine Meadow (M313)
🟩 Black Hills Coniferous Forest (M334)
🟨 California Coastal Chaparral Forest and Shrub (261)
🟨 California Coastal Range Open Woodland - Shrub - Coniferous Forest - Meadow (M262)
🟫 California Coastal Steppe - Mixed Forest - Redwood Forest (263)
🟦 California Dry Steppe (262)
🟫 Cascade Mixed Forest - Coniferous Forest - Alpine Meadow (M242)
🟫 Central Appalachian Broadleaf Forest-Coniferous Forest-Meadow (M221)
🟥 Central Interior Broadleaf Forest (223)
🟩 Chihuahuan Semi-Desert (321)
🟫 Colorado Plateau Semi-Desert (313)
🟩 Eastern Broadleaf Forest (221)
🟨 Everglades (411)
🟨 Great Plains - Palouse Dry Steppe (331)
🟩 Great Plains Steppe (332)
🟥 Intermountain Semi-Desert and Desert (341)
🟨 Intermountain Semi-Desert (342)
🟩 Laurentian Mixed Forest (212)
🟧 Lower Mississippi Riverine Forest (234)
🟫 Middle Rocky Mountain Steppe - Coniferous Forest - Alpine Meadow (M332)
🟫 Midwest Broadleaf Forest (222)
🟩 Nevada-Utah Mountains Semi-Desert - Coniferous Forest - Alpine Meadow (M341)
🟩 Northeastern Mixed Forest (211)
🟧 Northern Rocky Mountain Forest-Steppe - Coniferous Forest - Alpine Meadow (M333)
🟧 Ouachita Mixed Forest-Meadow (M231)
🟨 Outer Coastal Plain Mixed Forest (232)
🟧 Ozark Broadleaf Forest (M223)
🟩 Pacific Lowland Mixed Forest (242)
🟧 Prairie Parkland (Subtropical) (255)
🟨 Prairie Parkland (Temperate) (251)
🟨 Sierran Steppe - Mixed Forest - Coniferous Forest - Alpine Meadow (M261)
🟨 Southeastern Mixed Forest (231)
🟥 Southern Rocky Mountain Steppe - Open Woodland - Coniferous Forest - Alpine Meadow (M331)
🟫 Southwest Plateau and Plains Dry Steppe and Shrub (315)

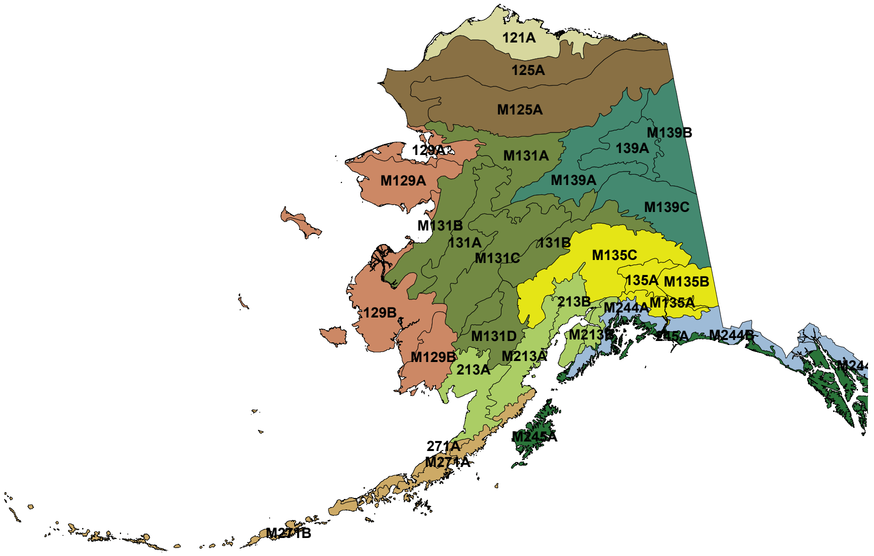
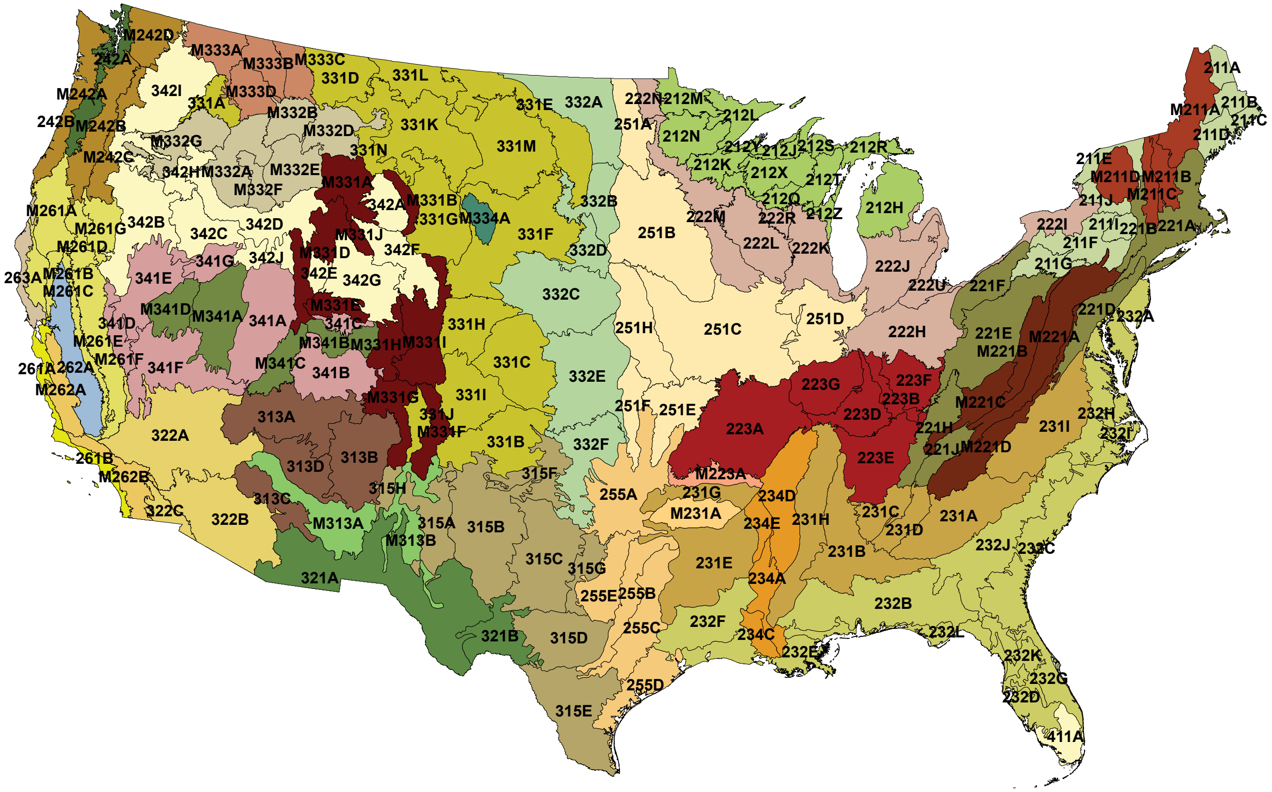


Figure 1.1—Ecoregion provinces and sections for the conterminous United States (Cleland and others 2007) and Alaska (Nowacki and Brock 1995). Ecoregion sections within each ecoregion province are shown in the same color.



DATA SOURCES

Forest Service data sources included in this edition of the FHM national report are FIA annualized phase 2 survey data (Bechtold and Patterson 2005), FHP national insect and disease detection survey forest mortality and defoliation data for 2009, Moderate Resolution Imaging Spectroradiometer (MODIS) Active Fire Detections for the United States database for 2009, and forest cover data developed from MODIS satellite imagery by the Forest Service Remote Sensing Applications Center. Other sources of data are daily weather station data from the National Climatic Data Center (NCDC) Parameter-elevation Regression on Independent Slopes (PRISM) climate mapping system data (PRISM Group 2009), and the 2001 National Land Cover Database (NLCD) map (Homer and others 2007).

A major source of data for FHM analyses has been the FIA program, which collects forest inventory information across all forest land ownerships in the United States. FIA maintains a network of more than 125,000 permanent forested ground plots across the conterminous United States and southeastern Alaska, with a sampling intensity of approximately one plot per 2 428.11 ha. The FIA program's phase 2 encompasses the annualized inventory measured on plots at regular intervals, with each plot surveyed every 5 to 7 years in Eastern States, but with plots in the Rocky Mountain and Pacific Northwest regions surveyed once every 10 years (Reams and others 2005). The standard one-sixth acre plot (fig. 1.2) consists of four 24-foot-radius subplots (approximately 0.0415 or 1/24 acre), on which field crews measure trees at least 5 inches in diameter. Within each of these subplots is nested a 6.8-foot-radius microplot (approximately 1/300th acre), on which crews measure trees smaller than 5 inches in diameter. A core-optional variant of the standard design includes four "macroplots," each with radius of 58.9 feet (approximately one-fourth acre) that originates at the center of each subplot (Woudenberg and others 2010).

FIA phase 3 plots represent a subset of these phase 2 plots, with one phase 3 plot for every 16 standard FIA phase 2 plots. In addition to traditional forest inventory measurements, data for a variety of important ecological indicators are collected from phase 3 plots, including tree crown condition, lichen communities, down woody material, soil condition, and vegetation structure and diversity. Additionally, data on ozone bioindicator plants are collected on a separate grid of plots. Most of these additional forest health indicators were measured as part of the FHM Detection Monitoring ground plot system prior to 2000[2] (Palmer and others 1991).

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[2] USDA Forest Service. 1998. Forest Health Monitoring 1998 field methods guide. Research Triangle Park, NC: U.S. Department of Agriculture Forest Service, National Forest Health Monitoring Program. 473 p. On file with: Forest Health Monitoring Program, 3041 Cornwallis Rd., Research Triangle Park, NC 27709.

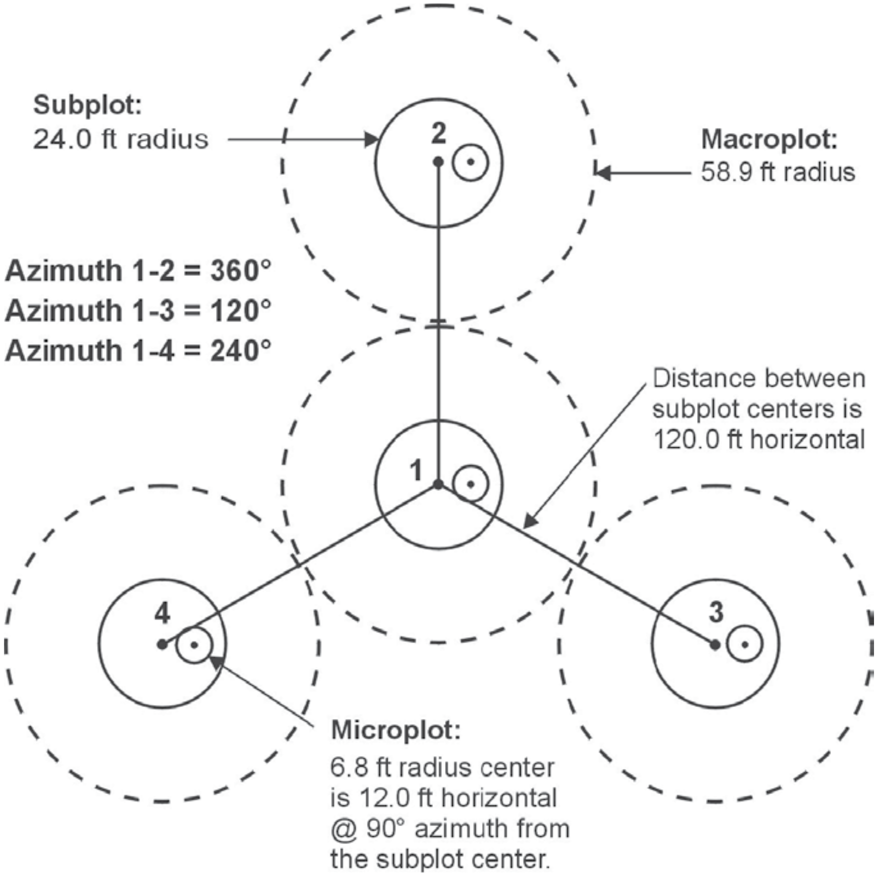

Figure 1.2—The Forest Inventory and Analysis mapped plot design. Subplot 1 is the center of the cluster with subplots 2, 3, and 4 located 120 feet away at azimuths of 360°, 120°, and 240°, respectively. (Source: Woudenberg and others 2010)

THE FOREST HEALTH MONITORING PROGRAM

The FHM program is a national program designed to determine the status, changes, and trends in indicators of forest condition on an annual basis, and covers all forested lands through a partnership encompassing the Forest Service, State foresters, and other State and Federal agencies and academic groups (FHM 2010). The FHM program utilizes data from a wide variety of data sources, both inside and outside the Forest Service, and develops analytical approaches for addressing forest health issues that affect the sustainability of forest ecosystems. It encompasses five major activities (fig. 1.3):

- Detection Monitoring—nationally standardized aerial and ground surveys to evaluate status and change in condition of forest ecosystems (sections 1 and 2 of this report).
- Evaluation Monitoring—projects to determine extent, severity, and causes of undesirable changes in forest health identified through Detection Monitoring (section 3 of this report).
- Intensive Site Monitoring—projects to enhance understanding of cause-effect relationships by linking Detection Monitoring to ecosystem process studies and to assess specific issues, such as calcium depletion and carbon sequestration, at multiple spatial scales (section 3 of this report).
- Research on Monitoring Techniques—work to develop or improve indicators, monitoring systems, and analytical techniques, such as urban and riparian forest health monitoring,

early detection of invasive species, multivariate analyses of forest health indicators, and spatial scan statistics (section 2 of this report).

- Analysis and Reporting—synthesis of information from various data sources within and external to the Forest Service to produce issue-driven reports on status and change in forest health at national, regional, and State levels (sections 1, 2, and 3 of this report).

In addition to its national reporting efforts, FHM generates regional and State reports. These reports may be produced with FHM partners, both within the Forest Service and in State forestry and agricultural departments representing a variety of analyses on disturbance and forest conditions (Steinman 2004), urban monitoring methods (Lake and others 2006), health conditions in national forests (Morin and others 2006), urban forest health monitoring (Cumming and others 2006, 2007), crown conditions (Randolph 2010, Randolph and Moser 2009), and ozone monitoring (Rose and Coulston 2009). Reports in the Forest Health Highlights series are annually produced for each State to profile current conditions, and are available on the FHM Web site at www.fs.fed.us/foresthealth/fhm. These highlights are produced by the FHM regions in cooperation with their respective State partners. FHM and its partners also produce reports and journal articles on monitoring techniques and analytical methods, including analyzing forest health data (Smith and Conkling 2004), soils as an indicator of

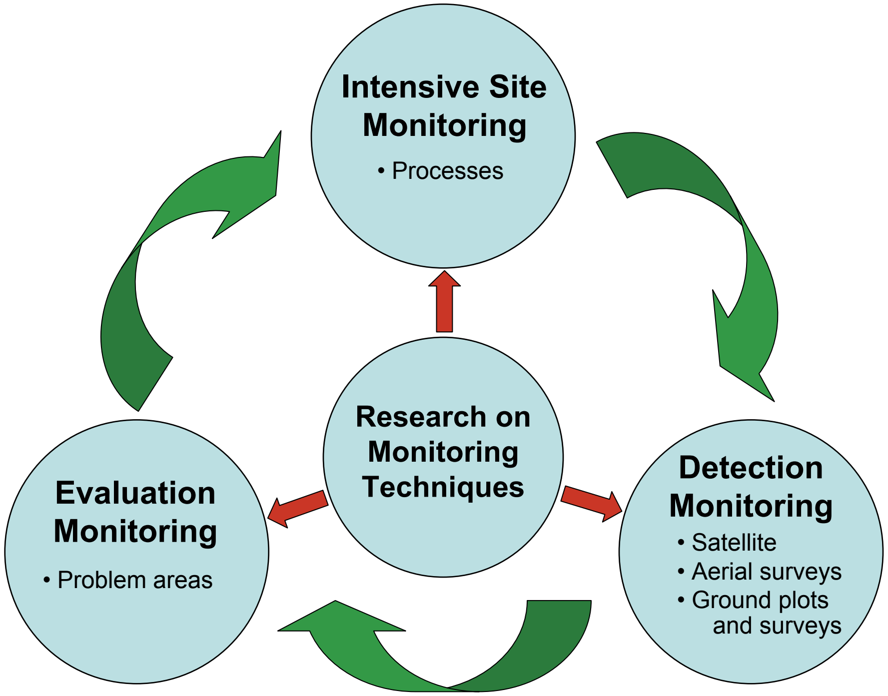

Figure 1.3—The design of the Forest Health Monitoring (FHM) Program (FHM 2003). A fifth component, Analysis and Reporting of Results, draws from the four FHM components shown here and provides information to help support land management policies and decisions.

forest health (O'Neill and others 2005), crown-condition classification (Schomaker and others 2007), sampling and estimation procedures for vegetation diversity and structure (Schulz and others 2009), and the overall forest health indicator program (Woodall and others 2010).

For more information about efforts to determine the status, changes, and trends in indicators of the condition of U.S. forests, visit the FHM Web site at www.fs.fed.us/foresthealth/fhm. This FHM national report is produced by national forest health monitoring researchers at the Eastern Forest Environmental Threat Assessment Center, which was established under the Healthy Forest Restoration Act to generate knowledge and tools needed to anticipate and respond to environmental threats. For more information about the research team, and about threats to U.S. forests, please visit www.forestthreats.org/about.

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SECTION 1.

Analyses of Short-Term Forest Health Data

CHAPTER 2.

Large-Scale Patterns of Insect and Disease Activity in the Conterminous United States and Alaska from the National Insect and Disease Detection Survey, 2009

Kevin M. Potter[1]

INTRODUCTION

Analyzing patterns of forest pest infestation, disease occurrences, forest declines, and related biotic stress factors is necessary to monitor the health of forested ecosystems and their potential impacts on forest structure, composition, biodiversity, and species distributions (Castello and others 1995). Introduced nonnative insects and diseases, in particular, can extensively damage the diversity, ecology, and economy of affected areas (Brockerhoff and others 2006, Mack and others 2000). Examining pest occurrences and related stress factors from a landscape-scale perspective is useful, given the regional extent of many infestations and the large-scale complexity of interactions between host distribution, stress factors, and the development of pest outbreaks (Holdenrieder and others 2004). The detection of geographic clusters of disturbance is one such landscape-scale approach, which allows for the identification of areas at greatest risk of significant impact and for the selection of locations for more intensive monitoring and analysis.

METHODS

Nationally compiled low-altitude aerial survey and ground survey data collected by the Forest Health Protection (FHP) Program of the Forest Service, U.S. Department of Agriculture, can be used to identify forest landscape-scale patterns associated with geographic hot spots of forest insect and disease activity in the conterminous United States, and to summarize insect and disease activity by ecoregion in Alaska (Potter and Koch 2012, Potter 2012). In 2009, FHP surveys covered approximately 156.8 million ha (61.5 percent) of the forested area in the conterminous United States, and 8.3 million ha (16.1 percent) of Alaska's forested area (fig. 2.1).

These surveys identify areas of mortality and defoliation caused by insect and pathogen activity, although some important forest insects (such as emerald ash borer and hemlock woolly adelgid), diseases (such as laurel wilt, Dutch elm disease, white pine blister rust, and thousand cankers disease), and mortality complexes (such as oak decline) are not easily detected or thoroughly quantified through aerial detection surveys. Such pests may attack hosts that are widely dispersed throughout diverse forests or may cause mortality or defoliation that is otherwise difficult to detect. A pathogen or insect might be considered a mortality-causing agent in one location and a defoliation-causing agent in another, depending on the level of damage to the forest in a given area and the convergence of stress factors such as drought. In some cases, the identified agents of mortality or defoliation are actually complexes of multiple agents summarized under an impact label related to a specific host tree species, e.g., subalpine fir mortality or aspen defoliation. Additionally, differences in data collection, attribute recognition, and coding procedures among States and regions can complicate the analysis of the data and the interpretation of the results.

The 2009 mortality and defoliation polygons were used to identify the mortality and

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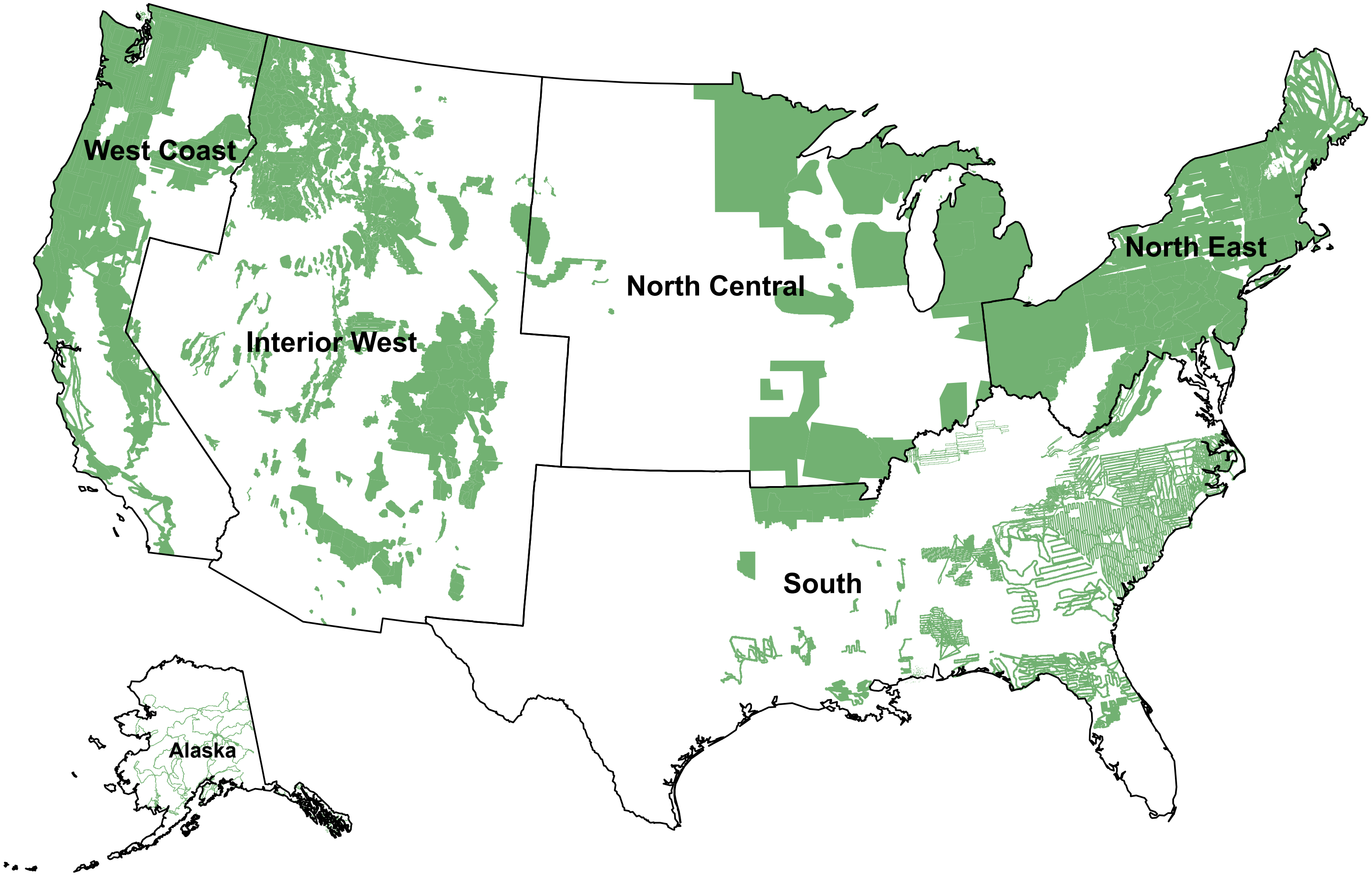

Figure 2.1—The extent of surveys for insect and disease activity conducted in the conterminous United States and Alaska in 2009. The black lines delineate Forest Health Monitoring regions. Note: Alaska is not shown to scale with map of the conterminous United States. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection)

defoliation agents and complexes found on more than 5000 ha in the conterminous United States in that year, and to identify and list the most widely detected defoliation and mortality agents for Alaska. All quantities are "footprint" areas for the agent or complex. The sum of agents and complexes is not equal to the total affected area as a result of reporting multiple agents per polygon in some situations.

A forest cover map (1-km^2 resolution), derived from Moderate Resolution Imaging Spectroradiometer (MODIS) satellite imagery by the Forest Service Remote Sensing Applications Center (USDA Forest Service 2008), was used to determine the amount and location of forest within survey defoliation and mortality polygons. A Getis-Ord hot spot analysis (Getis and Ord 1992) was then employed in ArcMap 9.2 (ESRI 2006) to identify forested areas with the greatest exposure to mortality-causing and defoliation-causing agents and complexes. The Environmental Monitoring and Assessment Program North American hexagon coordinates (White and others 1992) were intensified to develop a lattice of hexagonal cells, of approximately 2500 km^2 extent, for the conterminous United States. This cell size allows for analysis at a medium-scale resolution of approximately the same area as a typical county. The percent of forest area in each hexagon exposed to either mortality-causing or defoliation-causing agents or complexes was then calculated by dividing the forest-masked damage area by the forest-masked surveyed area.

The Getis-Ord G_i* statistic summed the differences between the mean values in a local sample, determined by a moving window consisting of each hexagon and its six adjacent hexagons, and the global mean of all the forested hexagonal cells in the conterminous United States. It was then standardized as a z score with a mean of 0 and a standard deviation of 1, with values greater than 1.96 representing significant ($p < 0.025$) local clustering of high values and values less than -1.96 representing significant clustering of low values ($p < 0.025$), since 95 percent of the observations under a normal distribution should be within approximately 2 standard deviations of the mean (Laffan 2006). In other words, a G_i* value of 1.96 indicates that the local mean of percent forest exposed to mortality-causing or defoliation-causing agents and complexes for a hexagon and its 6 neighbors is approximately 2 standard deviations greater than the mean expected in the absence of spatial clustering, while a G_i* value of -1.96 indicates that the local mortality or defoliation mean for a hexagon and its six neighbors is approximately 2 standard deviations less than the mean expected in the absence of spatial clustering. Values between -1.96 and 1.96 have no statistically significant concentration of high or low values. In other words, when a hexagon has a G_i* value between -1.96 and 1.96, it and its six neighbors have neither consistently high nor consistently low percentages of forest exposed to mortality- or defoliation-causing agents and complexes.

The threshold values are not exact because the correlation of spatial data violates the assumption of independence required for statistical significance (Laffan 2006). The Getis-Ord approach does not require that the input data be normally distributed because the local G_i* values are computed under a randomization assumption, with G_i* equating to a standardized *z* score that asymptotically tends to a normal distribution (Anselin 1992). The *z* scores are reliable, even with skewed data, as long as the distance band is large enough to include several neighbors for each feature (ESRI 2006).

The low density of survey data from Alaska in 2009 (fig. 2.1) precluded the use of hot spot analyses for the State. Instead, mortality and defoliation data were summarized by ecoregion section (Nowacki and Brock 1995), calculated as the percent of the forest within the surveyed areas affected by agents and complexes of mortality or defoliation. For reference purposes, ecoregion sections (Cleland and others 2007) were also displayed on the geographic hot spot maps of the conterminous United States.

RESULTS AND DISCUSSION

The FHP survey data identified 62 different mortality-causing agents and complexes on approximately 4.68 million ha of forest across the conterminous United States in 2009, an area slightly larger than the land area of Maryland and Massachusetts combined. Mountain pine beetle (*Dendroctonus ponderosae*) was the most widespread mortality agent, detected on 3.47 million ha (table 2.1). Other mortality agents and complexes detected across very large areas, each affecting more than 100 000 ha, were

Table 2.1—Mortality agents and complexes affecting more than 5 000 ha in the conterminous United States in 2009

2009 mortality agents/complexes	Area
	ha
Mountain pine beetle	3 467 925
Bronze birch borer	285 539
Fir engraver	172 004
Sudden aspen decline	144 353
Subalpine fir mortality	117 793
Spruce beetle	80 064
Douglas-fir beetle	72 445
Five-needle pine decline	56 217
Gypsy moth	46 797
Bark beetles	41 909
Pinyon *Ips*	34 789
Western pine beetle	28 171
Decline (unspecified)	27 621
Beech bark disease	17 778
Forest tent caterpillar	13 928
Western balsam bark beetle	10 562
White pine blister rust	9 415
Eastern larch beetle	7 694
Balsam woolly adelgid	6 984
Pine engraver	6 714
Winter moth	6 341
Emerald ash borer	5 198
Other mortality agents	25 509
Total, all mortality agents	**4 683 511**

Note: All values are "footprint" areas for each agent or complex. The sum of the individual agents is not equal to the total for all agents because of overlapping damage polygons.

bronze birch borer (*Agrilus anxius*), fir engraver (*Scolytus ventralis*), sudden aspen (*Populus tremuloides*) decline, and subalpine fir (*Abies lasiocarpa*) mortality.

Additionally, the survey identified 64 defoliation agents and complexes affecting approximately 3.17 million ha of forest across the conterminous United States in 2009, an area slightly smaller than the land area of New Hampshire, Delaware, and Rhode Island combined. The most widespread defoliator was western spruce budworm (*Choristoneura occidentalis*), affecting 1.81 million ha (table 2.2). Forest tent caterpillar (*Malacosoma disstria*), pinyon needle scale (*Matsucoccus acalyptus*), and gypsy moth (*Lymantria dispar*) also affected more than 100 000 ha.

The Interior West region (as defined by the Forest Health Monitoring [FHM] Program of the Forest Service) had, by far, the largest area on which mortality-causing agents and complexes were detected in 2009, approximately 3.67 million ha (table 2.3). Nearly all of the mortality was associated with mountain pine beetle. The hot spot analysis detected two major hot spots of insect and disease mortality in the region in which mountain pine beetle was by far the predominant mortality agent (fig. 2.2). A large and highly clustered hot spot was centered on the Montana ecoregion sections M332D-Belt Mountains, M332B-Northern Rockies and Bitterroot Valley, and M332E-Beaverhead Mountains. Another such hot spot was located in ecoregion section M331I-Northern Parks and Ranges of northern Colorado and

Table 2.2—Defoliation agents and complexes affecting more than 5 000 ha in the conterminous United States in 2009

2009 defoliation agents/complexes	Area
	ha
Western spruce budworm	1 812 242
Forest tent caterpillar	620 240
Pinyon needle scale	226 522
Gypsy moth	181 720
Spruce budworm	58 659
Pinyon sawfly	53 117
Unknown defoliator	52 196
Defoliators (unspecified)	23 590
Decline	19 063
Septoria leaf spot and canker	18 676
Larch casebearer	16 762
Western tent caterpillar	15 331
Aspen defoliation	8 228
Orangestriped oakworm	7 698
Larger elm leaf beetle	6 296
Jack pine budworm	5 140
Bruce spanworm	5 000
Other defoliation agents	36 146
Total, all defoliation agents	**3 165 733**

Note: All values are "footprint" areas for each agent or complex. The sum of the individual agents is not equal to the total for all agents because of overlapping damage polygons.

Table 2.3—The top five mortality agents and complexes detected in each Forest Health Monitoring region in 2009

2009 mortality agents/complexes	Area
	ha
Interior West	
Mountain pine beetle	3 281 941
Sudden aspen decline	144 275
Subalpine fir mortality	117 840
Spruce beetle	61 351
Five-needle pine decline	57 863
Total, all mortality agents	**3 670 065**
North Central	
Bronze birch borer	285 539
Beech bark disease	11 837
Mountain pine beetle	9 115
Eastern larch beetle	7 694
Emerald ash borer	394
Total, all mortality agents	**321 939**
North East	
Gypsy moth	46 797
Forest tent caterpillar	13 901
Winter moth	6 291
Beech bark disease	5 869
Emerald ash borer	3 006
Total, all mortality agents	**82 637**

2009 biotic mortality agents/complexes	Area
	ha
South	
Hemlock woolly adelgid	862
Southern pine beetle	77
Ips	32
Black turpentine beetle	2
Laurel wilt	2
Total, all mortality agents	**955**
West Coast	
Mountain pine beetle	342 796
Fir engraver	120 367
Bark beetles	40 311
Douglas-fir beetle	39 729
Western pine beetle	21 463
Total, all mortality agents	**607 434**
Alaska	
Spruce beetle	40 718
Northern spruce engraver	14 250
Yellow-cedar decline	6 458
Eastern larch beetle	43
Total, all mortality agents	**61 471**

Note: All values are “footprint” areas for each agent or complex.

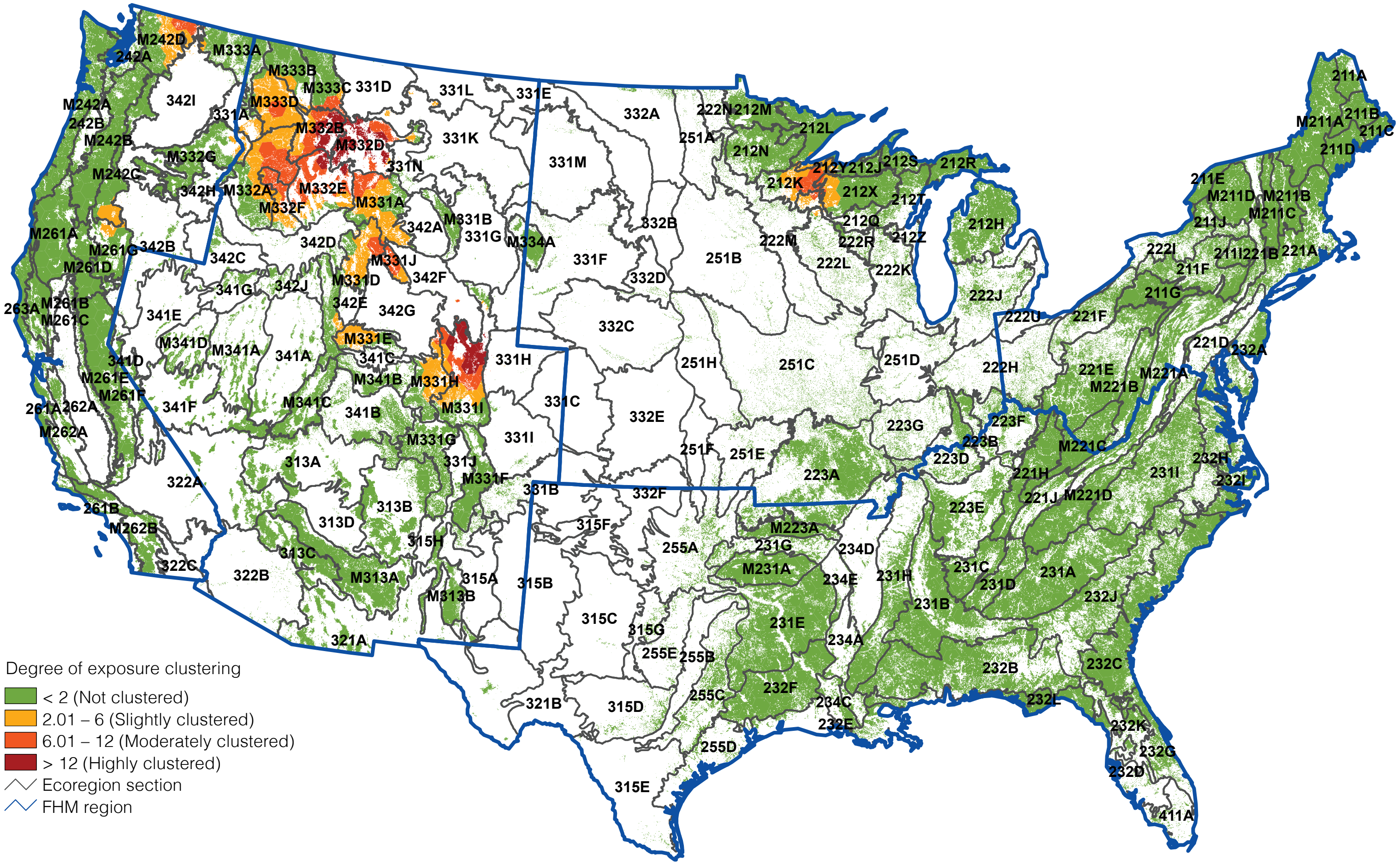

Figure 2.2—Hot spots of exposure to mortality-causing insects and diseases in 2009. Values are Getis-Ord G_i scores, with values greater than 2 representing significant clustering of high percentages of forest area exposed to mortality agents. (No areas of significant clustering of low percentages of exposure, -2, were detected.) The gray lines delineate ecoregion sections (Cleland and others 2007), and blue lines delineate Forest Health Monitoring regions. Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection)*

southern Wyoming. Four less concentrated hot spots of mountain pine beetle mortality were located in northern Idaho, centered in ecoregion section M333D-Bitterroot Mountains; in central Idaho, centered on ecoregion sections M332A-Idaho Batholith and M332F-Challis Volcanics; in western Wyoming, centered on ecoregion sections M331J-Wind River Mountains and M331D-Overthrust Mountains; and in northeast Utah, centered on ecoregion section M331E-Uinta Mountains.

Mountain pine beetle was also the leading cause of mortality in the West Coast region, where it was detected on approximately 343 000 ha (table 2.3). The region's two mortality hot spots were both associated with the beetle, the larger in ecoregion section M242D-Northern Cascades and the smaller in portions of ecoregion sections M242C-Eastern Cascades and M261G-Modoc Plateau (fig. 2.2). Fir engraver was another important agent of mortality in the West Coast region, affecting approximately 120 000 ha (table 2.3).

Bronze birch borer was by far the most important agent of mortality in the North Central FHM region, affecting approximately 286 000 ha (table 2.3). It was associated with the only mortality hot spot in the region, which occurred in the 212K-Western Superior Highlands, 212Q-North Central Wisconsin Highlands, and 212X-Northern Highlands (fig. 2.2).

No mortality hot spots were located in the other two FHM regions in the conterminous United States in 2009. Surveys detected forest mortality on approximately 83 000 ha in the North East region, where gypsy moth and forest tent caterpillar were the leading mortality agents (table 2.3). Surveys reported only 955 ha of mortality in the South region, where hemlock woolly adelgid (*Adelges tsugae*) was the leading mortality agent (table 2.3).

As with mortality, the Interior West FHM region encompassed the largest area on which defoliation agents and complexes were detected, at slightly more than 2 million ha (table 2.4). Western spruce budworm accounted for the largest area of detected defoliation, followed by pinyon needle scale. Several hot spots of defoliation were associated with western spruce budworm, including a major hot spot centered in ecoregion section M332D-Belt Mountains of Montana and extending into ecoregion sections M331A-Yellowstone Highlands, M332E-Beaverhead Mountains, and M332B-Northern Rockies and Bitterroot Valley (fig. 2.3). Nearby, in northern Idaho and northwestern Montana, another hot spot was located in ecoregion sections M333D-Bitterroot Mountains, M333B-Flathead Valley, and M333C-Northern Rockies. A third hot spot in central Idaho encompassed parts of ecoregion sections M332A-Idaho Batholith and M332F-Challis Volcanics. Western spruce budworm was also the causal agent relating to a hot spot of defoliation on the border between Colorado and New Mexico, in ecoregion sections M331F-Southern Parks and Rocky Mountain Range and M331G-South-Central Highlands. Finally, a defoliation hot spot in central Nevada was associated with pinyon needle scale, pinyon

Table 2.4—The top five defoliation agents and complexes detected in each Forest Health Monitoring region in 2009

2009 defoliation agents/complexes	Area
	ha
Interior West	
Western spruce budworm	1 639 697
Pinyon needle scale	226 522
Pinyon sawfly	53 117
Unknown defoliator	52 189
Decline	18 490
Total, all defoliation agents	**2 017 782**
North Central	
Forest tent caterpillar	160 661
Spruce budworm	58 527
Larch casebearer	11 221
Gypsy moth	5 238
Jack pine budworm	5 140
Total, all defoliation agents	**253 143**
North East	
Forest tent caterpillar	354 144
Gypsy moth	171 400
Defoliators (unspecified)	20 327
Septoria leaf spot and canker	18 676
Orangestriped oakworm	7 698
Total, all defoliation agents	**577 777**

2009 defoliation agents/complexes	Area
	ha
South	
Forest tent caterpillar	102 978
Gypsy moth	15 253
Larger elm leaf beetle	6 296
Baldcypress leafroller	2 223
Defoliators (unspecified)	2 128
Total, all defoliation agents	**126 004**
West Coast	
Western spruce budworm	176 149
Larch casebearer	5 532
Lodgepole needleminer	3 042
Douglas-fir tussock moth	1 746
Pine butterfly	1 561
Total, all defoliation agents	**190 690**
Alaska	
Aspen leafminer	125 696
Willow leaf blotchminer	56 515
Defoliators (unspecified)	5 973
Spear-marked black moth	5 791
Spruce bud moth	5 341
Total, all defoliation agents	**202 655**

Note: All values are "footprint" areas for each agent or complex.

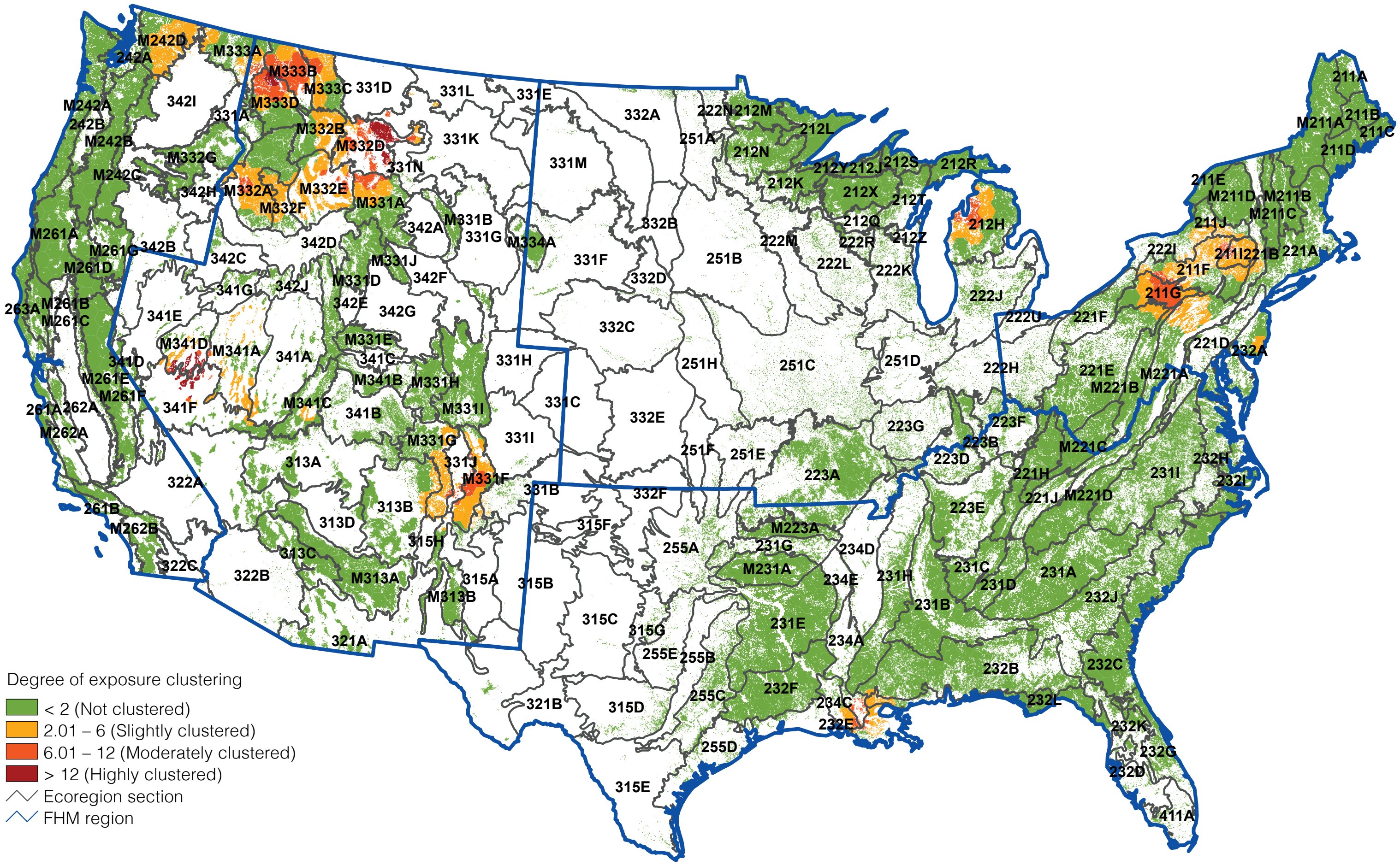

Figure 2.3—Hot spots of exposure to defoliation-causing insects and diseases in 2009. Values are Getis-Ord G_i** scores, with values greater than 2 representing significant clustering of high percentages of forest area exposed to defoliation agents. (No areas of significant clustering of low percentages of exposure, -2, were detected.) The gray lines delineate ecoregion sections (Cleland and others 2007), and blue lines delineate Forest Health Monitoring regions. Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection)*

sawfly, and aspen decline. This hot spot stretched across three ecoregion sections: M341D-West Great Basin and Mountains, M341A-East Great Basin and Mountains, and 341F-Southeastern Great Basin.

The western spruce budworm was also the leading cause of defoliation in the West Coast FHM region (table 2.4). This defoliation was most concentrated in a hot spot in central Washington, between ecoregion sections M242-Northern Cascades and M333A-Okanogan Highland (fig. 2.3).

Forest tent caterpillar was the leading defoliator in the three FHM regions of the Eastern United States (table 2.4), defoliating approximately 354 000 ha in the North East, approximately 161 000 ha in the North Central region, and approximately 103 000 ha in the South. At least one hot spot in each region was associated with this pest. In the North East, one hot spot was split between ecoregion sections 211F-Northern Glaciated Allegheny Plateau and 211I -Catskill Mountains in New York (fig. 2.3). A second hot spot, centered on ecoregion section 211G-Northern Unglaciated Allegheny Plateau, was caused by both forest tent caterpillar and gypsy moth. The forest tent caterpillar hot spot in the North Central region, meanwhile, was located in ecoregion section 212H-Northern Lower Peninsula of Michigan (fig. 2.3), while the hot spot in the South spanned 234C-Atchafalaya and Red River Alluvial Plains and 232E-Louisiana Coastal Prairie and Marshes ecoregion sections of southern Louisiana.

In 2009, four mortality-causing agents and complexes were reported for Alaska, affecting approximately 61 000 ha (table 2.3). Spruce beetle was the leading mortality agent, detected on about 41 000 ha, mostly in the south-central region of the State, including ecoregion sections M213A-Northern Aleutian Range and 213B-Cook Inlet Lowlands. As a result, these two ecoregions had the highest percent of exposure to mortality-causing agents and complexes in surveyed forest areas, 2.24 percent and 1.04 percent, respectively (fig. 2.4). Northern spruce engraver beetle (*Ips perturbatus*) was the second most widespread mortality agent, affecting about 14 000 ha of forest (table 2.3), mostly in the central and east-central parts of the State. Yellow-cedar (*Chamaecyparis nootkatensis*) decline was also an important mortality complex (6 458 ha) in the panhandle of the State.

Alaska forests were exposed to 12 defoliation agents and complexes recorded on approximately 202 000 ha (table 2.4). Aspen leafminer (*Phyllocnistis populiella*) had by far the largest extent, observed on approximately 126 000 ha across central Alaska. As a result of aspen leafminer, three ecoregion sections had relatively high percentages of defoliation exposure (fig. 2.5): M139C-Dawson Range, with 10.23 percent surveyed forest exposed; 139A-Yukon Flats, with 8.63 percent; and M139B-Olgivie Mountains, with 7.48 percent.

A second major defoliator was willow leaf blotchminer (*Micrurapteryx salicifoliella*), which was detected on approximately 57 000 ha. Like aspen leafminer, it occurred

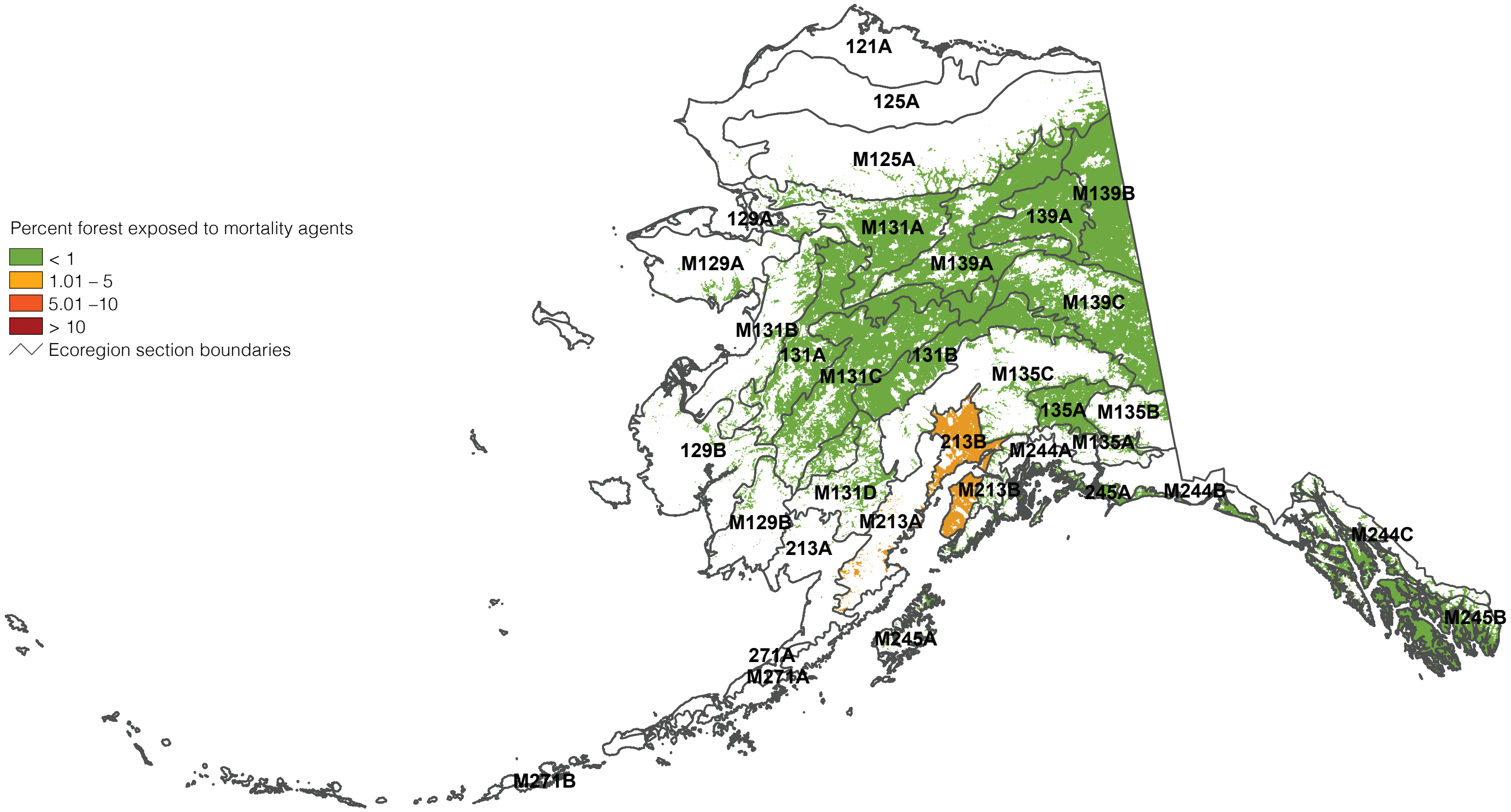

Figure 2.4—Percent of surveyed forest in Alaska ecoregion sections exposed to mortality-causing insects and diseases in 2009. The gray lines delineate ecoregion sections (Nowacki and Brock 1995). Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection)

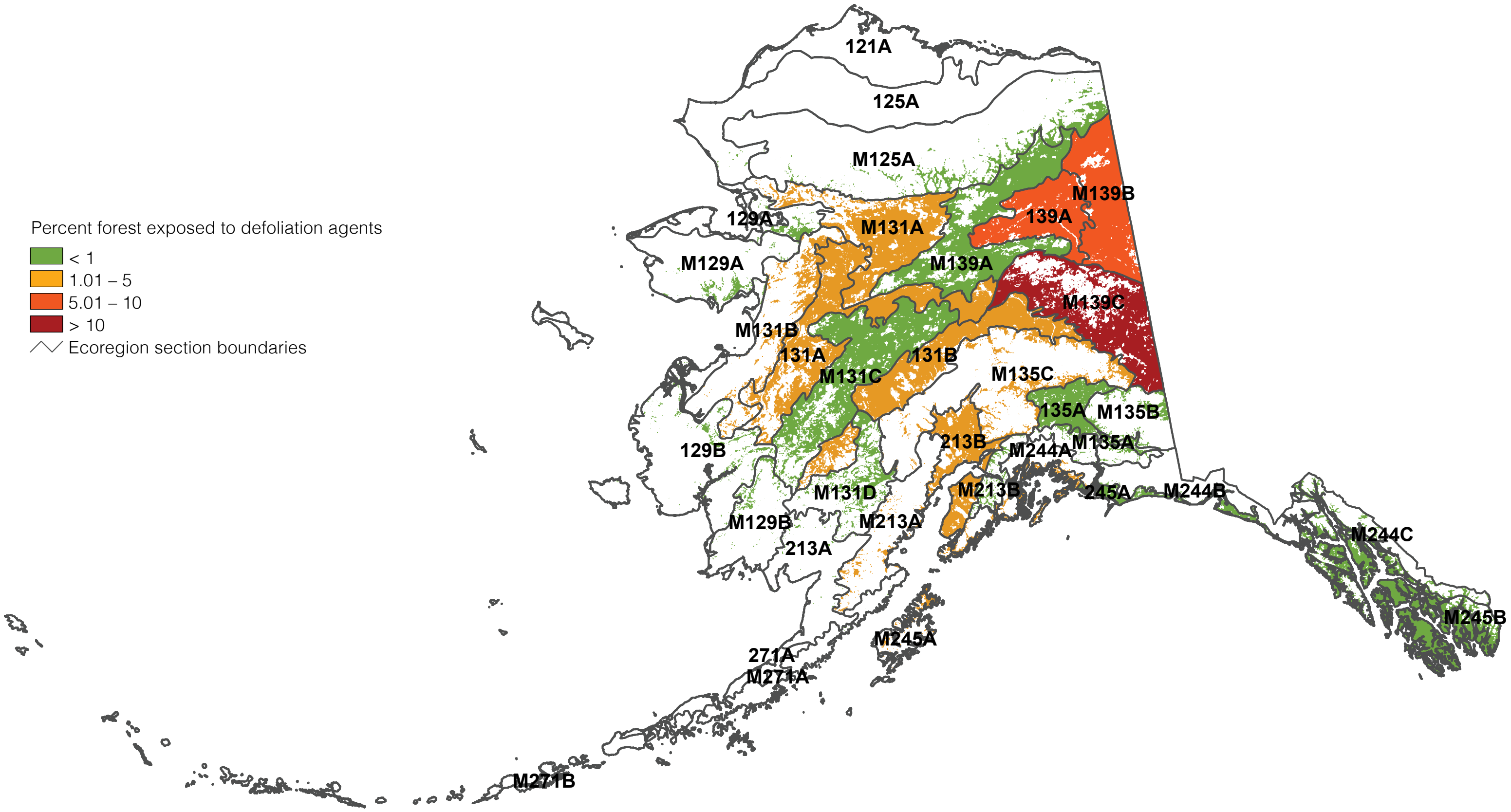

Figure 2.5—Percent of surveyed forest in Alaska ecoregion sections exposed to defoliation-causing insects and diseases in 2009. The gray lines delineate ecoregion sections (Nowacki and Brock 1995). Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection)

mainly in the east-central region of the State, including ecoregion sections 139A-Yukon Flats, M139B-Olgivie Mountains, 131A-Yukon Bottomlands, M131A-Upper Kobuk-Koyukuk, and 131B-Kuskokwin Colluvial Plain. Other important defoliators in 2009 were spear-marked black moth (*Rheumaptera hastata*), spruce bud moth (*Zeiraphera canadensis*), hemlock sawfly (*Neodiprion tsugae*) (1427 ha), and northern spruce engraver (1236 ha).

Continued monitoring of insect and disease outbreaks across the United States will be necessary for determining appropriate follow-up investigation and management activities. Because of the limitations of survey efforts to detect certain important forest insects and diseases, the pests and pathogens discussed in this chapter do not comprise all the forest health threats that should be considered when making management decisions and budget allocations. However, as these analyses demonstrate, large-scale assessments of mortality and defoliation exposure, including geographical hot spot detection analyses, offer one potentially useful approach for helping to prioritize geographic areas where the concentration of monitoring and management activities would be most effective.

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CHAPTER 3.

Large-Scale Patterns of Forest Fire Occurrence in the Conterminous United States and Alaska, 2009

Kevin M. Potter[1]

INTRODUCTION

Wildland fire represents an important ecological mechanism in many forest ecosystems. It shapes the distributions of species, maintains the structure and function of fire-prone communities, and is a significant evolutionary force (Bond and Keeley 2005). At the same time, fire outside the historic range of frequency and intensity can have extensive economic and ecological impacts. Current fire regimes on more than half the forested area in the conterminous United States have been either moderately or significantly altered from historical regimes, potentially altering key ecosystem components such as species composition, structural stage, stand age, canopy closure, and fuel loadings (Schmidt and others 2002). Fire suppression and the introduction of nonnative plants, in particular, have dramatically altered fire regimes (Barbour and others 1999). Additionally, fire regimes altered by global climate change could cause large-scale shifts in vegetation spatial patterns (McKenzie and others 1996).

METHODS

The Moderate Resolution Imaging Spectroradiometer (MODIS) Active Fire Detections for the United States database (USDA Forest Service 2010) allows analysts to spatially display and summarize fire occurrence on a yearly basis (Coulston and others 2005; Potter 2012a, Potter 2012b). Fire occurrences are defined as the satellite detection of wildland fire in a 1-km^2 pixel for one day. The data are derived using the MODIS Rapid Response System (Justice and others 2002) from the thermal infrared bands of imagery collected daily by two satellites at a resolution of 1 km^2, with the center of a pixel recorded as a fire occurrence when the satellites' MODIS sensors identify the presence of a fire at the time of image collection (USDA Forest Service 2010). The data represent only whether a fire was active, because the MODIS sensors do not differentiate between a hot fire in a relatively small area (0.01 km^2, for example) and a cooler fire over a larger area (1 km^2, for example). The MODIS Active Fire database does well at capturing large fires, but may underrepresent rapidly burning, small and low-intensity fires, as well as fires in areas with frequent cloud cover (Hawbaker and others 2008).

The number of fire occurrences per 100 km^2 (10 000 ha) of forested area was determined for each ecoregion section in the conterminous United States (Cleland and others 2007) and Alaska (Nowacki and Brock 1995) for 2009. This forest fire occurrence density measure was calculated after screening out wildland fires on non-forested pixels using a forest cover layer derived from MODIS imagery by the Forest Service Remote Sensing Applications Center (USDA Forest Service 2008). The total number of fire occurrences across the conterminous United States and in Alaska was also calculated.

Additionally, a Getis-Ord hot spot analysis (Getis and Ord 1992) in ArcMap 9.2 (ESRI 2006) was employed to identify forested areas in the conterminous United States with greater fire

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occurrence density than expected by chance in 2009. The spatial units of analysis were cells of approximately 2500 km^2 from a hexagonal lattice of the conterminous United States, intensified from Environmental Monitoring and Assessment Program (EMAP) North America hexagon coordinates (White and others 1992). This cell size allows for analysis at a medium-scale resolution of approximately the same area as a typical county. Fire occurrence density values for each hexagon were quantified as the number of forest fire occurrences per 100 km^2 (10 000 ha) of forested area within the hexagon. The Getis-Ord G_i* statistic was used to identify clusters of hexagonal cells with fire occurrence density values higher than expected by chance.

Briefly, G_i* sums the differences between the mean values in a local sample, determined in this case by a moving window of each hexagon and the six neighboring hexagons, and the global mean of all the forested hexagonal cells in the conterminous United States. G_i* is standardized as a *z* score with a mean of 0 and a standard deviation of 1, with values greater than 1.96 representing significant ($p < 0.025$) local clustering of higher fire occurrence densities and values less than -1.96 representing significant ($p < 0.025$) local clustering of lower fire occurrence densities, since 95 percent of the observations under a normal distribution should be within approximately 2 standard deviations of the mean (Laffan 2006). Values between -1.96 and 1.96 have no statistically significant concentration of high or low values; a hexagon and its six neighbors, in other words, have neither consistently high nor consistently low fire occurrence densities per 100 km^2 of forested area. The threshold values are not exact because the correlation of spatial data violates the assumption of independence required for statistical significance (Laffan 2006). The Getis-Ord approach does not require that the input data be normally distributed because the local G_i* values are computed under a randomization assumption, with G_i* equating to a standardized *z* score that asymptotically tends to a normal distribution (Anselin 1992). The *z* scores are reliable, even with skewed data, as long as the distance band is large enough to include several neighbors for each feature (ESRI 2006).

RESULTS AND DISCUSSION

The MODIS Active Fire database captured 76,611 wildland forest fire occurrences across the conterminous United States in 2009, the third most since the first full year of MODIS data collection in 2001, but fewer than in the 2 previous years (fig. 3.1). The annual mean number of forest fire occurrences since 2001 was 48,368. The database captured 33,331 fire occurrences in Alaska, also the third highest number since initial MODIS data collection. With a few exceptions, the conterminous United States and Alaska have seen opposite year-to-year trends in the number of forest fire

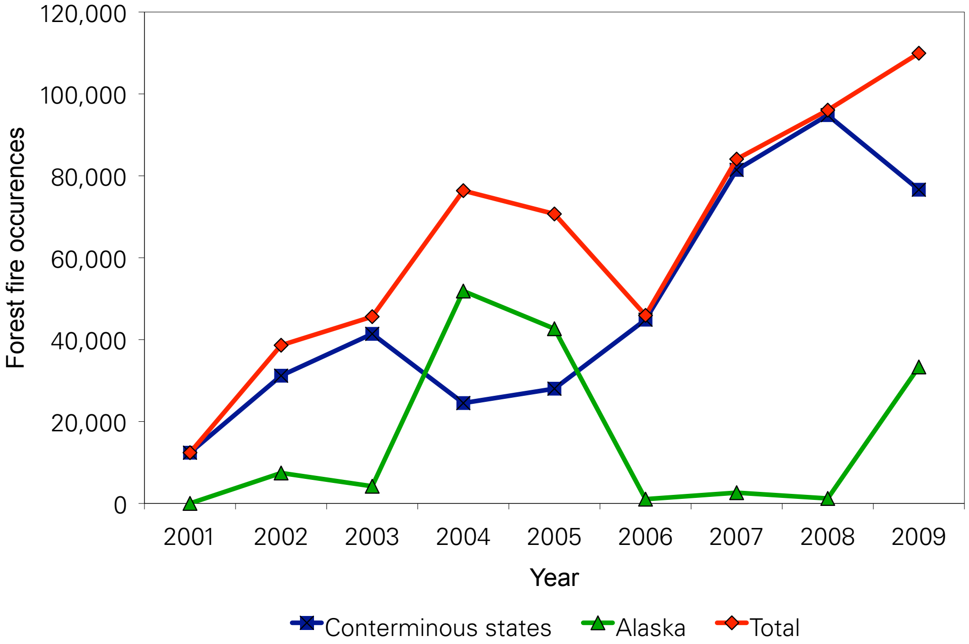


Figure 3.1—Forest fire occurrences detected by MODIS from 2001 to 2009, for the conterminous United States, Alaska, and the two regions combined. (Data source: U.S. Department of Agriculture Forest Service, Remote Sensing Application Center)

occurrences. This was the case between 2008 and 2009, with the number of fires decreasing in the conterminous United States and increasing in Alaska (fig. 3.1). The increase in total number of fire occurrences across both regions is consistent with the official wildland fire statistics, which show a 12-percent increase in the overall area burned nationally between 2008 (2 141 801 ha) and 2009 (2 397 484 ha) (NICC 2010).

In 2009, ecoregion section M262A-Central California Coast Ranges experienced the highest number of fire occurrences relative to its area of forest, with 38.5 fires per 100 km^2 of forested area (fig. 3.2). Two adjacent ecoregion sections also had high numbers of forest fire occurrences: M262B-Southern California Mountain and Valley and 261B-Southern California Coast, with 22.4 and 12.9 fires per 100 km^2 of forested area, respectively. The southern plains of Oklahoma and Texas also had high densities of forest fire occurrences: 15.8 in ecoregion section 315G-Eastern Rolling Plains and 15.0 in ecoregion section 255A-Cross Timbers and Prairie.

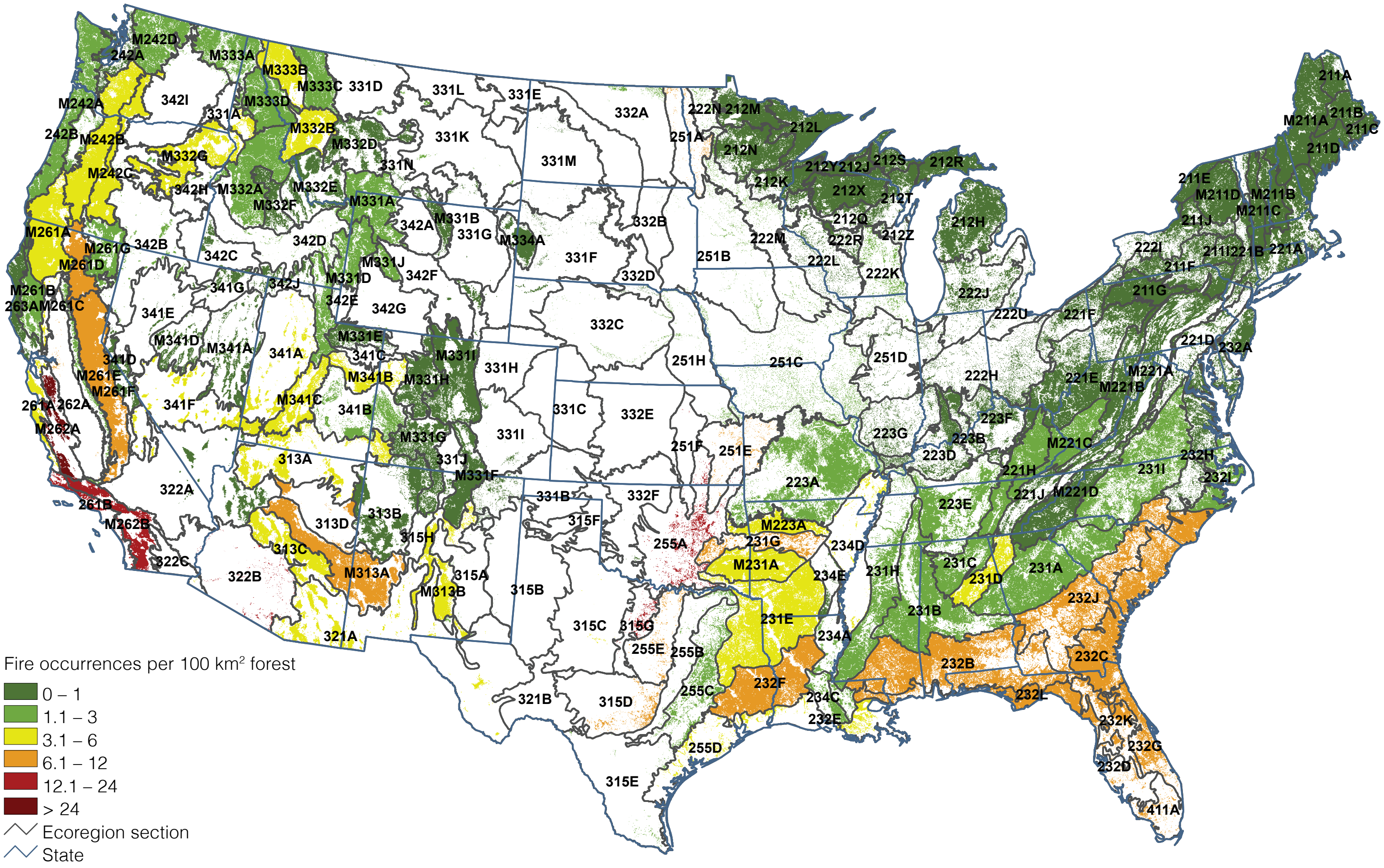


Figure 3.2—The number of forest fire occurrences, per 100 km^2 (10 000 ha) of forested area, by ecoregion section within the conterminous United States, for 2009. The gray lines delineate ecoregion sections (Cleland and others 2007). Forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: U.S. Department of Agriculture Forest Service, Remote Sensing Application Center)

Regions with moderately high forest fire density in 2009 included much of the Southeastern Coastal Plain and all of Florida as well as ecoregion sections 231G-Arkansas Valley of Oklahoma and Arkansas, 255E-Texas Cross Timbers and Prairie and 315D-Edwards Plateau of Texas, M313A-White Mountains and 313C-Painted Desert of Arizona and New Mexico, and M261E-Sierra Nevada and M261D-Southern Cascades of California (fig. 3.2).

In Alaska, the most fire occurrences in 2009 (28.2 per 100 km^2 of forest) were recorded in ecoregion section 139A-Yukon Flats (fig.3.3). Ecoregion sections 131A-Yukon Bottomlands and 131B-Kuskokwim Colluvial Plain also experienced a high density of forest fire occurrences, with 15.0 and 13.1 per 100 km^2 of forest, respectively. Three Alaska ecoregion sections had moderately high density of forest fire occurrences: M139B-Olgivie Mountains, 135A-Copper River Basin, and M131C-Kuskowkim Mountains.

While summarizing fire occurrence data at the ecoregion scale allows for the summary of fire density over time in a relatively large geographic area, a geographical hot spot analysis can offer insights into where fire occurrences are concentrated at a finer scale during a given length of time. Analyses of MODIS Active Fire data from previous years (Potter 2012a, Potter 2012b) indicated that geographical hot spots of fire occurrence density were limited almost entirely to the Pacific Coast, the Rocky Mountains, and the Southeastern Coastal Plain. In 2009, however, no hot spots existed in the Rocky Mountains, although hot spots were concentrated in Southern California and were scattered across the Southeastern Coastal Plain (fig. 3.4). Unlike previous years, which each encompassed at least one high-density geographic hot spot, the highest-density hot spots in 2009 were only moderately so. These hot spots were located in ecoregion sections 261B-Southern California Coast, M262B-Southern California Mountain and Valley, 261A-Central California Coast, and M262A-Central California Coastal Ranges. This pattern suggests that fires in 2009 were more evenly distributed across the conterminous United States, with slightly higher concentrations in a few areas.

These lower-density hot spots included these ecoregion sections:

- M261E-Sierra Nevada and M261F-Sierra Nevada Foothills of California
- M242B-Western Cascades and M242C-Eastern Cascades of southern Oregon
- M341C-Utah High Plateau and 341F-Southeastern Great Basin of southwestern Utah
- M313A-White Mountains-San Francisco Peaks-Mogollon Rim, 313C-Tonto Transition, and 313D-Painted Desert and 313A-Grand Canyon of Arizona
- 255A-Cross Timbers and Prairie, 231G-Arkansas Valley, and M231A-Ouachita Mountains of Oklahoma

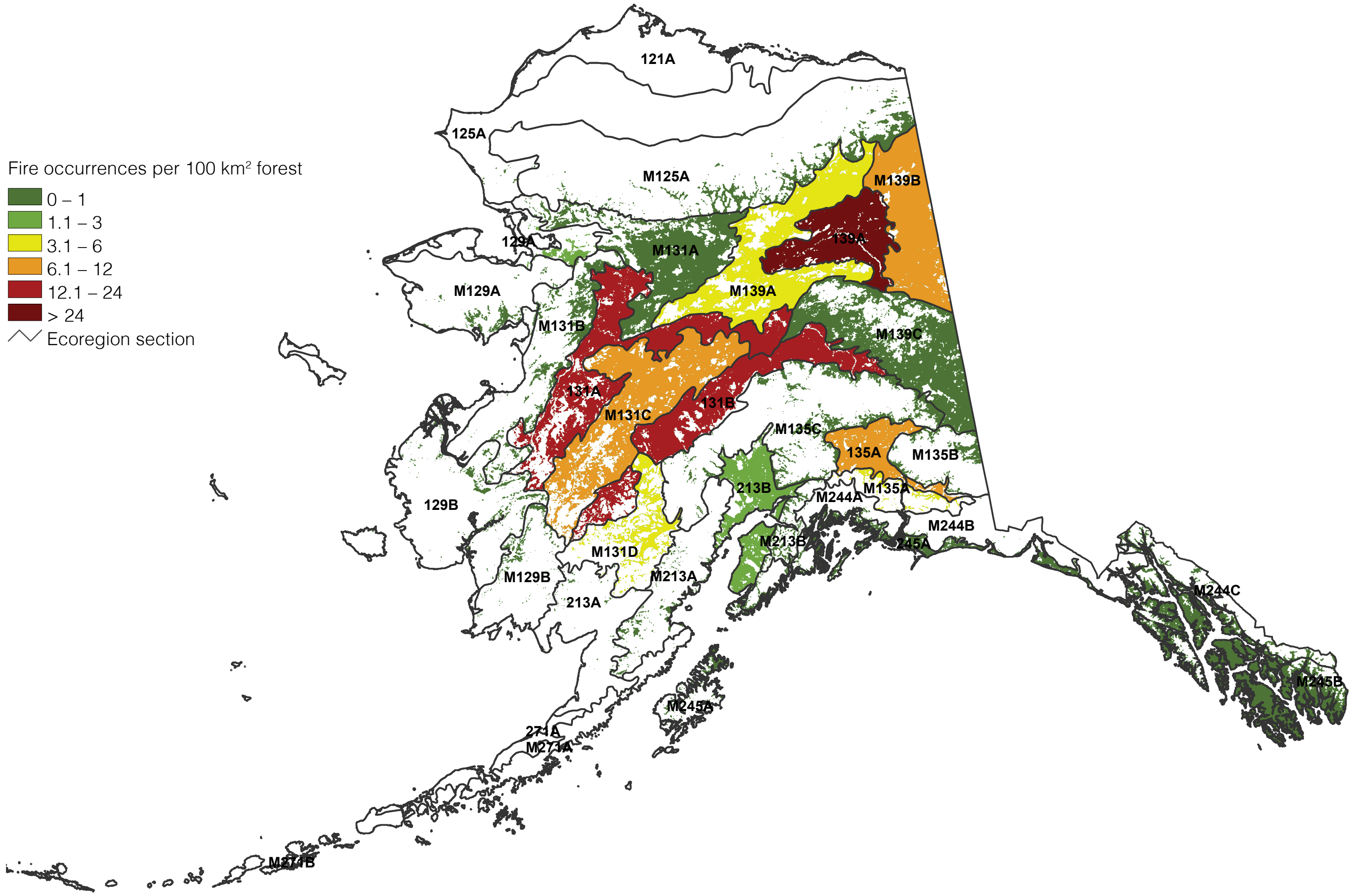
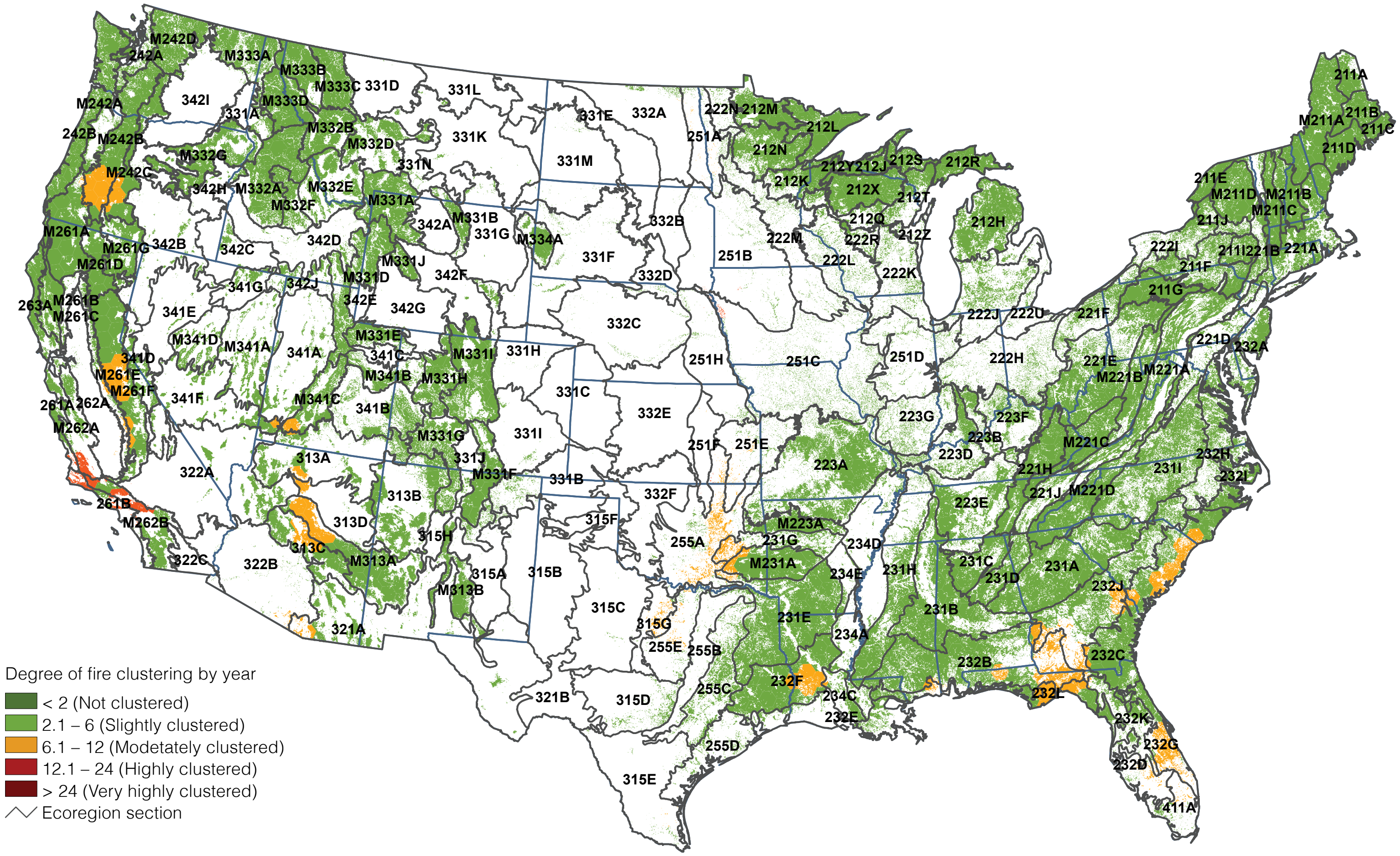


Figure 3.3—The number of forest fire occurrences, per 100 km² (10 000 ha) of forested area, by ecoregion section within Alaska, for 2009. The gray lines delineate ecoregion sections (Nowacki and Brock 1995). Forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: U.S. Department of Agriculture Forest Service, Remote Sensing Application Center)



Degree of fire clustering by year

- 🟩 < 2 (Not clustered)
- 🟩 2.1 – 6 (Slightly clustered)
- 🟧 6.1 – 12 (Modetately clustered)
- 🟥 12.1 – 24 (Highly clustered)
- 🟥 > 24 (Very highly clustered)
- ╱╲ Ecoregion section

Figure 3.4—Hot spots of fire occurrence across the conterminous United States for 2009. Values are Getis-Ord G_i** scores, with values greater than 2 representing significant clustering of high fire occurrence density values. (No areas of significant clustering of low fire density occurrence values, -2, were detected.) The gray lines delineate ecoregion sections (Cleland and others 2007). Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: U.S. Department of Agriculture Forest Service, Remote Sensing Application Center)*

- 315G-Eastern Rolling Plains and 255E-Texas Cross Timbers and Prairie of Texas
- 232F-Coastal-Plains and Flatwoods-Western Gulf of Louisiana
- 232B-Gulf Coastal Plains and Flatwoods and 232L-Gulf Coastal Lowlands of northern Florida and southern Georgia and Alabama
- 232C-Atlantic Coastal Flatwoods and 232J-Southern Atlantic Coastal Plains and Flatwoods of South Carolina and Georgia
- 232G-Florida Coastal Lowlands-Atlantic, 232D-Florida Coastal Lowlands-Gulf, and 411A-Everglades of southern Florida

The results of these geographic analyses are intended to offer insights into where fire occurrences have been concentrated, but are not intended to quantify the severity of a given fire season. Information about the concentration of fire occurrences may be useful for the identification of areas for management activities and for follow-up investigations related to the ecological and socioeconomic impacts of fires that may be outside the range of historic frequency.

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CHAPTER 4.

Tree Mortality

Mark J. Ambrose[1]

INTRODUCTION

Tree mortality is a natural process in all forest ecosystems. However, extremely high mortality also can be an indicator of forest health issues. On a regional scale, high mortality levels may indicate widespread insect or disease problems. High mortality may also occur if a large proportion of the forests in a region is made up of older, senescent stands.

In early national reports by the Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture, mortality was analyzed using phase 3 data from the FHM and Forest Inventory and Analysis (FIA) programs of the Forest Service. Those data spanned a relatively long time period (nearly 10 years for some States), but the sample was not spatially intense (approximately one plot per 96,000 acres). In the 2008 and 2009 FHM national reports (Ambrose 2012a, Ambrose 2012b), the same method was applied to FIA phase 2 data from the relatively small number of States in the Eastern United States where repeated plot measurements had been taken. In this report, the method is applied to most of the Central and Eastern United States, using phase 2 data from repeated measurements in a much larger number of States.

The FHM mission to monitor, assess, and report on the status, changes, and long-term trends in forest ecosystem health in the United States (USDA Forest Service 1994). Thus, the aim of this mortality analysis contrasts with how mortality might be approached in other reports, such as FIA State reports or State Forest Health Highlights. The approach to mortality presented here seeks to detect nonspecific or multiple-host mortality patterns that might reflect subtle changes to fundamental ecosystem processes (due to such large-scale factors as air pollution, global climate change, or fire-regime change) that transcend individual tree species-pest/pathogen interactions or direct concern over forest resource production and availability.

At this point a mortality baseline is still being established for most of the United States using the first two cycles of annualized data (i.e., the first two measurements of each plot). To discern trends in mortality rates, a minimum of three cycles of FIA data are required.[2] With at most two cycles of data currently available, it is only possible to do a spatial comparison of ecoregions and identify regions of higher than average mortality (relative to growth) for further study.

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[2] In theory, one could estimate changes or trends in mortality rates using just two cycles of data by comparing, for example, plots measured in 2000 and 2005 with those measured in 2001 and 2006, those measured in 2002 and 2007, and so on. However, we choose not to do so because estimating mortality rates independently for each panel of data reduces the effective sample intensity by a factor of five and because an analysis of mortality rates using heavily overlapping time periods will be unlikely to detect subtle changes in mortality rates.

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DATA

FIA phase 2 inventory data are collected using a rotating panel sample design (Bechtold and Patterson 2005). Field plots are divided into spatially balanced panels, with one panel being measured each year. A single cycle of measurements consists of measuring all panels. This annualized method was phased in, State by State, beginning in 1999. Initially, a 5-year measurement cycle was instituted in the East and a 10-year cycle in the West. However, some Southeastern States later adopted a 7-year cycle. Any analysis of mortality requires data collected for at least two points in time from any given plot. Therefore, mortality analysis was possible for areas where data from repeated plot measurements using consistent sampling protocols were available (i.e., where one cycle of measurements had been completed and at least one panel of the next cycle had been measured, and where there had been no changes to the protocols affecting measurement of trees or saplings).

Once all phase 2 plots have been remeasured in a State, mortality estimates generally will be based on a sample intensity of approximately 1 plot for 6,000 acres of forest.[3] However, at this time not all plots have been remeasured in most of the States included in this analysis. When not all plots have been remeasured, mortality estimates are based on a lower effective sample intensity. Table 4.1 shows the 28 States from which consistent, repeated FIA phase 2 measurements

Table 4.1—States from which repeated Forest Inventory and Analysis (FIA) phase 2 measurements were available, the time period spanned by the data, and the number of panels of data available. Each panel represents approximately one-fifth of the plots in a State[a]

Time period	States	Number of phase 2 panels
1999-2008	ME	5
1999-2008	MN, MO, WI	4[b, c, d]
2000-2008	IA, IN, MI, PA	4
2000-2008	VA	3[e]
2001-2007	GA, TN	2
2001-2008	OH	2
2001-2008	AL, IL, KS, NE, ND, SD, TX[f]	3
2002-2007	AR, KY, SC	1
2002-2008	NY	1
2002-2008	NH	2
2003-2008	CT, MA, RI, VT	1

[a] States are listed by standard abbreviation.

[b] In Minnesota and Wisconsin, the phase 2 inventory was done at twice the standard FIA sample intensity, approximately 1 plot per 3,000 acres when the full five panels are measured.

[c] In Missouri, the phase 2 inventory was done at twice the standard FIA sample intensity, approximately one plot per 3,000 acres when the full five panels are measured, on national forest lands and at the standard intensity on all other lands.

[d] In Minnesota, Missouri, and Wisconsin, the field season often begins late in the calendar year, so while the earliest data are from 1999, they do not represent a separate panel but are part of the panel mostly measured in 2000.

[e] Only a small proportion of the plots measured in Virginia in 2000 used the current national standard plot design, so just slightly more than three full panels of remeasurement data were available for this analysis.

[f] Annualized growth and mortality data were only available for eastern Texas.

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[3] In some States, more intensive sampling has been implemented. See table 4.1 for details.

were available, the time period spanned by the data, and the effective sample intensity, based on the cycle length and the number of remeasured panels. The States included in this analysis, as well as the forest cover within those States, are shown in figure 4.1.

METHODS

FIA phase 2 tree and sapling data were used to estimate average annual tree mortality in terms of tons of biomass per acre. The biomass represented by each tree in tons was calculated by FIA and provided in the FIA database-version 4.0 (USDA Forest Service 2010). To compare mortality rates across forest types and climate zones, the ratio of annual mortality to gross growth (MRATIO) is used as a standardized mortality indicator (Coulston and others 2005a). Gross growth rate and mortality rate, in terms of tons of biomass per acre, were independently calculated for each ecoregion section (Cleland and others 2005) using a mixed modeling procedure where plot to plot variability is considered a random effect and time is a fixed effect. The mixed modeling approach has been shown to be particularly efficient for making estimates with data for which not all plots have been measured over identical time intervals (Gregoire and others 1995). MRATIOs were then calculated from the growth and mortality rates. For details on the method, see appendix A—Supplemental Methods in Coulston and others (2005b), and see appendix A—Supplemental Methods in Coulston and others (2005c).

The MRATIO can be large if an over-mature forest is senescing and losing a cohort of older trees. If forests are not naturally senescing, a high MRATIO (> 0.6) may indicate high mortality due to some acute cause (insects or pathogens) or due to generally deteriorating forest health conditions. An MRATIO value greater than 1 indicates that mortality exceeds growth and live standing biomass is actually decreasing.

In addition, the ratio of average dead tree diameter to average live tree diameter (DDLD ratio) was calculated for each plot where mortality occurred. Low DDLD ratios (much less than 1), i.e., small dead trees compared with the surviving trees, usually indicate competition-induced mortality typical of young, vigorous stands, while high ratios (much greater than 1), i.e., large dead trees compared with the surviving trees, indicate mortality associated with senescence or some external factors such as insects or disease (Smith and Conkling 2004). Intermediate DDLD ratios can be hard to interpret because a variety of stand conditions can produce such DDLD values. The DDLD ratio is most useful for analyzing mortality in regions that also have high MRATIOs. High (plot-level) DDLD values in regions with very low MRATIOs may indicate small areas experiencing high mortality of large trees or locations where the death of a single large tree (such as a remnant pine in a young hardwood stand) has produced a deceptively high DDLD.

To further analyze tree mortality, the number of stems and the total biomass of trees that

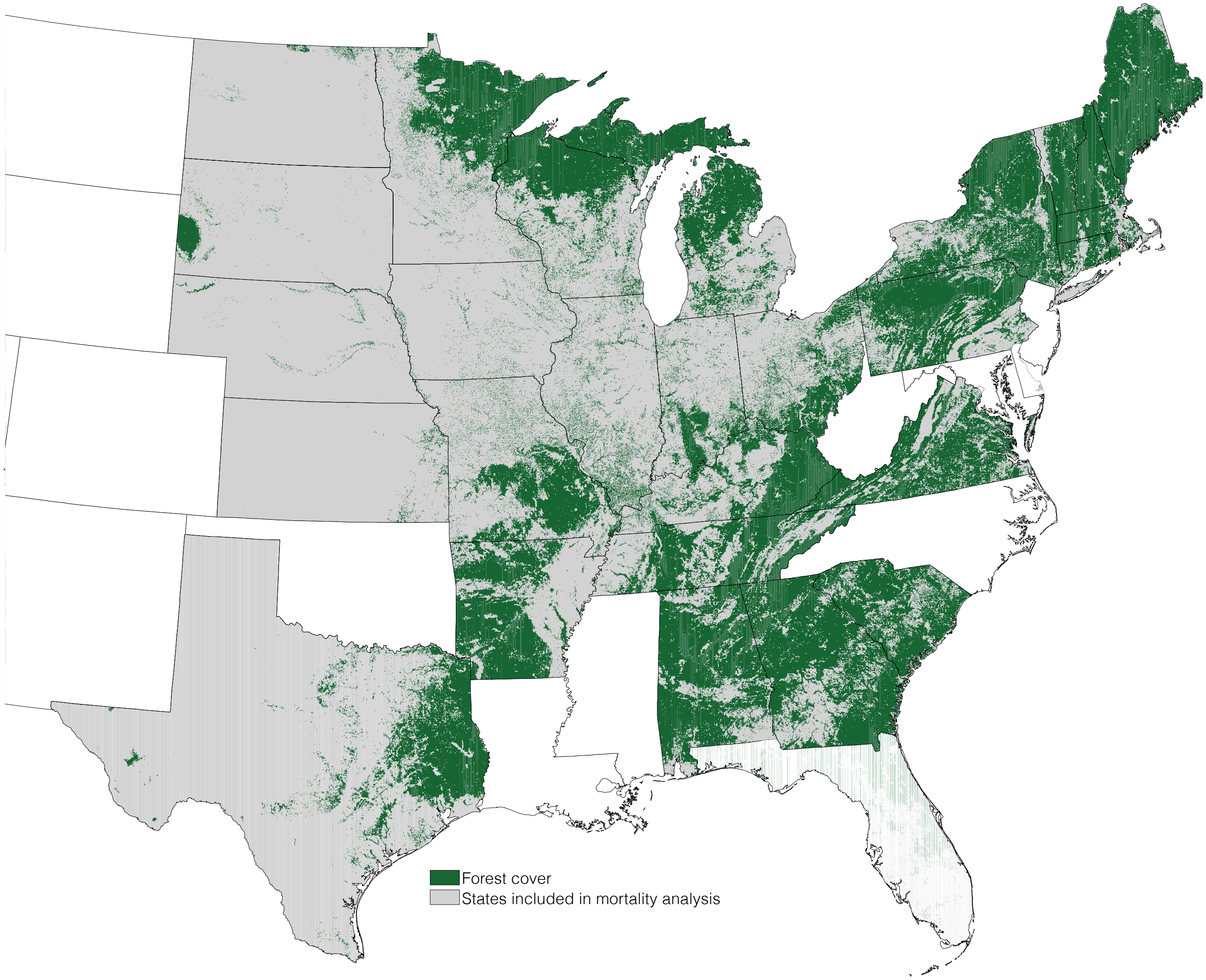

Figure 4.1—Forest cover in the States where mortality was analyzed. Forest cover was derived from Advanced Very High Resolution Radiometer satellite imagery (Zhu and Evans 1994).

died also were calculated by species within each ecoregion. Identifying the tree species experiencing high mortality in an ecoregion is a first step in identifying what forest health issue or issues may be affecting the forests. Although determining particular causal agents associated with all the observed mortality is beyond the scope of this report, often there are well-known insects and pathogens that are “likely suspects” once the affected tree species are identified.

RESULTS

The MRATIO values are shown in figure 4.2. The highest MRATIOs occurred in ecoregion sections 332C-Nebraska Sand Hills (MRATIO = 1.38) and 331F-Western Great Plains (MRATIO = 1.36), where mortality exceeded growth. Other areas of extremely high mortality relative to growth occurred in 332A-Northeastern Glaciated Plains (MRATIO = 0.98), 251B-North Central Glaciated Plains (MRATIO = 0.85), 251C-Central Dissected Till Plains (MRATIO = 0.84), and 332E-South Central Great Plains (MRATIO = 0.78). Mortality was also very high in 255D-Central Gulf Prairies and Marshes in southeastern Texas (MRATIO = 0.64) and M211D-Adirondack Highlands in New York (MRATIO = 0.61). In interpreting these MRATIOs, one must remember that the MRATIO is an ecoregion-level indicator. The mortality which produces a high MRATIO may be spatially concentrated within a region.

The results of the analysis of the relative sizes of trees that died, the DDLD ratio, are shown in figure 4.3. The DDLD ratio is a plot-level indicator and is so represented in the figure. However, with the density of FIA phase 2 plots, overlap of plot values represented on a national-scale map can give a misleading impression, so close-up views of the Upper Midwest, the Northeast, and the Southeast are also provided.

These figures show that even in areas of high mortality relative to growth, there was no mortality on most sample plots. However, on the plots where mortality occurred, the trees were large compared with surviving trees, suggesting the mortality is related to either senescence of older stands or some insect or disease issue.

In the three ecoregion sections exhibiting highest mortality relative to growth [332C-Nebraska Sand Hills, 331F-Western Great Plains (South Dakota and Nebraska), and 332A-Northeastern Glaciated Plains (North Dakota)], the predominant vegetation is grassland, and there were very few forested plots measured. Tree growth rates in these regions (especially in 331F) are quite low, so the high MRATIOSs are due to a combination of low growth and high mortality. Most of the forest in these sections is riparian forest, and, indeed, many of the species experiencing greatest mortality (table 4.2) are commonly found in riparian areas.

In ecoregion section 332C-Nebraska Sand Hills, where the MRATIO was highest, by far the largest amount of biomass that died was eastern cottonwood (table 4.2); more than half of the cottonwood biomass and more than one-third of the cottonwood stems had died by the end of the analysis period.

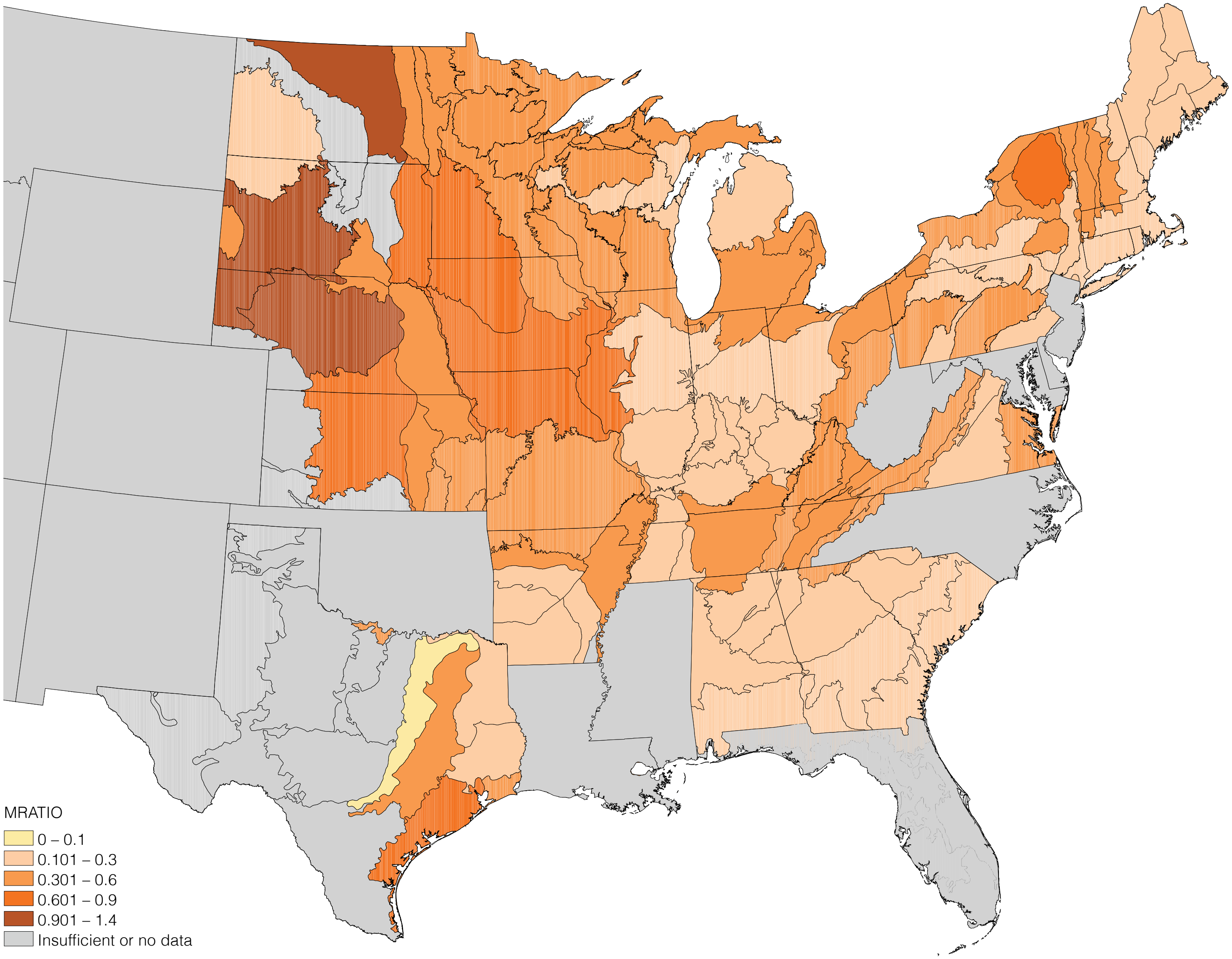

Figure 4.2—Tree mortality expressed as the ratio of annual mortality of woody biomass to gross annual growth in woody biomass (MRATIO) by ecoregion section (Cleland and others 2005). (Data source: U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis Program)

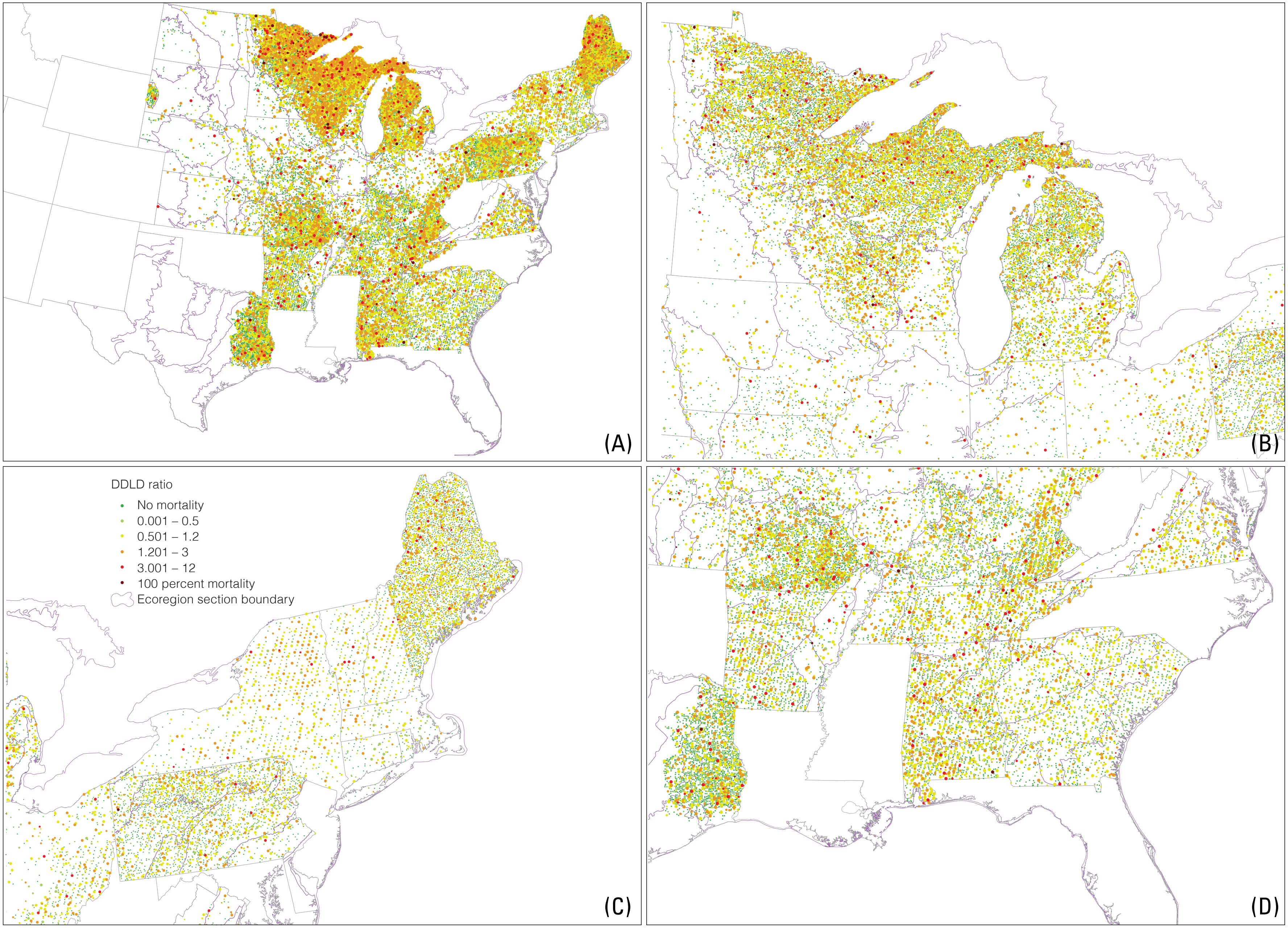

Figure 4.3—The ratio of average dead tree diameter to average surviving tree diameter (DDLD) on each plot at the time of its last measurement: (A) Eastern United States, (B) upper Midwest, (C) Northeast, (D) Southeast. Dot sizes are scaled relative to the biomass that died on each plot. Plot locations are approximate. (Data source: U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis Program)

Table 4.2—Tree species responsible for at least 10 percent of the mortality (in terms of biomass) for ecoregions where the MRATIO was 0.60 or greater

Ecoregion section	MRATIO	Tree species	Percent of total ecoregion mortality biomass	Mean age of dead trees[a]	Species percent mortality	
					Biomass	Stems
332C-Nebraska Sand Hills	1.38	Eastern cottonwood (*Populus deltoides*)	56.33	54	55.59	35.46
		Green ash (*Fraxinus pennsylvanica*)	10.22	67	14.65	13.67
331F-Western Great Plains	1.36	Ponderosa pine (*Pinus ponderosa*)	37.73	80	4.90	8.81
		Green ash (*F. pennsylvanica*)	32.77	42	21.80	22.48
332A-Northeastern Glaciated Great Plains	0.98	Quaking aspen (*Populus tremuloides*)	28.97	52	14.53	12.42
		American elm (*Ulmus americana*)	20.08	55	71.29	68.11
		Bur oak (*Quercus macrocarpa*)	19.88	101	7.15	2.25
		Green ash (*F. pennsylvanica*)	17.43	93	11.95	12.48
		Balsam poplar (*Populus balsamifera*)	10.86	45	53.03	74.00
251B-North Central Glaciated Plains	0.85	American elm (*U. americana*)	34.18	56	27.08	30.09
251C-Central Dissected Till Plains	0.84	American elm (*U. americana*)	13.98	52	18.82	22.39
332E-South Central Great Plains	0.78	Hackberry (*Celtis occidentalis*)	27.68	59	15.77	1.68
		Box elder (*Acer negundo*)	11.91	32	22.69	15.40
		Eastern cottonwood (*P. deltoides*)	11.64	62	7.25	17.65
		Green ash (*F. pennsylvanica*)	10.25	49	4.91	4.54
255D-Central Gulf Prairies and Marshes	0.64	Loblolly pine (*Pinus taeda*)	43.77	33	10.53	9.03
		Water oak (*Q. nigra*)	27.14	–	16.67	21.13
M211D-Adirondack Highlands	0.61	American beech (*Fagus americana*)	12.62	83	6.22	11.68
		Red maple (*A. rubrum*)	12.51	77	3.79	13.47
		Sugar maple (*A. saccharum*)	11.49	81	3.13	13.27
		Yellow birch (*Betula alleghaniensis*)	10.99	81	5.01	7.27

– = Data not available

[a] Ages are estimated from the stand age as determined by the Forest Inventory and Analysis (FIA) field crew. It is possible, especially in mixed-species stands, that the age of individual trees that died differed significantly from the stand age. Value may be missing if no stand age was given in the FIA data for most of the plots on which the mortality occurred.

In ecoregion section 331F-Western Great Plains, most of the mortality (37.73 percent in terms of biomass) was ponderosa pine. However this represented only 4.9 percent of the total ponderosa pine biomass. In contrast, green ash, which was 32.77 of total mortality (by biomass) in the ecoregion, suffered about 22 percent mortality, both in terms of biomass and number of stems. This suggests that there may be a more serious forest health issue affecting green ash than pine in that region.

Green ash also represented a large portion of the mortality in ecoregions 332A-Northeastern Glaciated Great Plains (17.43 percent), 332C-Nebraska Sand Hills (10.22 percent), and 332E-South Central Great Plains (12.25 percent). The cause of this mortality is not immediately apparent. One might be tempted to suspect the invasive insect, the emerald ash borer. However, this pest has not yet been reported in or near these regions (USDA Forest Service and others, N.d.). In ecoregion 332A-Northeastern Glaciated Great Plains, the age of the dead trees (table 4.2) suggests that older, senescent stands may be dying.

American elm was the only species that represented more than 10 percent of the mortality (by biomass) in ecoregions 251B-North Central Glaciated Plains (34.18 percent) and 251C-Central Dissected Till Plains (13.98 percent), which together stretch from southeastern North Dakota to western Illinois. American elm was also 20.08 percent of the mortality in ecoregion 332A-Northeastern Glaciated Great Plains. Dutch elm disease is the suspected cause. The pathogen which causes it is known to occur throughout the Midwest, including every county of Iowa since 2002 (Feeley 2010). Dutch elm disease has severely affected riparian forests in North Dakota (North Dakota Forest Service 2007). The disease is also reported to be a problem in Illinois (Illinois Department of Natural Resources 2009) and Minnesota (Minnesota Department of Natural Resources 2009).

The mortality pattern shown in these analyses does not immediately suggest large-scale forest health issues. Mortality is rather low in most of the areas for which data are available. The areas of highest mortality occur in the mostly riparian forests of several plains ecoregions. The causes of the mortality of several of the tree species experiencing high mortality in these regions (including eastern cottonwood in ecoregion 332C-Nebraska Sand Hills and balsam poplar in ecoregion 332A-Northeastern Glaciated Great Plains) are not immediately apparent. Further study of the health of these forests is probably warranted.

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SECTION 2.

Analyses of Long-Term Forest Health Trends and Presentations of New Techniques

CHAPTER 5.

Fragmentation of Forest, Grassland, and Shrubland

Kurt H. Riitters[1]

INTRODUCTION

As humans introduce competing land uses into natural landscapes, the public concerns regarding landcover patterns are expressed through headline issues such as urban sprawl, forest fragmentation, water quality, and wilderness preservation. The spatial arrangement of an environment affects all human perceptions and ecological processes within that environment, but this usually happens in competing ways, so the task for resource managers is to maintain appropriate amounts and patterns of different landcover types to provide the desired balance of social and ecological benefits. A prerequisite for informed management actions at local, regional, and national scales is reliable information about landcover patterns at those scales. National assessments of landcover patterns make it possible to identify national strategies to achieve particular objectives. To the extent that national data are also able to capture local details, the same information can be used for local planning as well.

Previous reports by the Forest Health Monitoring Program of the Forest Service, U.S. Department of Agriculture, (e.g., Ambrose and others 2008; Conkling 2011; Coulston and others 2005; Potter and Conkling 2012a, Potter and Conkling 2012b) have addressed different aspects of forest, grassland, and landscape spatial patterns. The objective of this chapter is to characterize and compare the fragmentation of forest, grassland, and shrubland landcover types by using assessment protocols which have been used before in national assessments of forest fragmentation (Riitters and others 2002, EPA 2008, USDA Forest Service 2004, USDA Forest Service 2011, Wickham and others 2008). The measurements were taken on the 2001 National Land Cover Database (NLCD) national landcover map.

METHODS

Briefly, a landcover pattern metric known as “area density” was applied separately to the forest, grassland, and shrubland components of the national landcover map. For a given location, area density was the proportion of a surrounding neighborhood that was a given landcover type, i.e., forest, grassland, or shrubland. Six measurement scales were defined as the sizes of the neighborhoods within which the measurements of area density were taken. The results were mapped at the same spatial resolution as the input map, permitting summaries by landcover type and by assessment region.

Landcover Maps

The landcover map from the 2001 NLCD (Homer and others 2004, 2007) covers the 50 States plus the District of Columbia and Puerto Rico (fig. 5.1). It has a spatial resolution of 0.09 ha per pixel (i.e., each pixel is 30 m x 30 m). For this analysis, the landcover classification was condensed from 16 to 8 landcover types (table 5.1) including the forest, grassland, and shrubland types of interest. The distribution of

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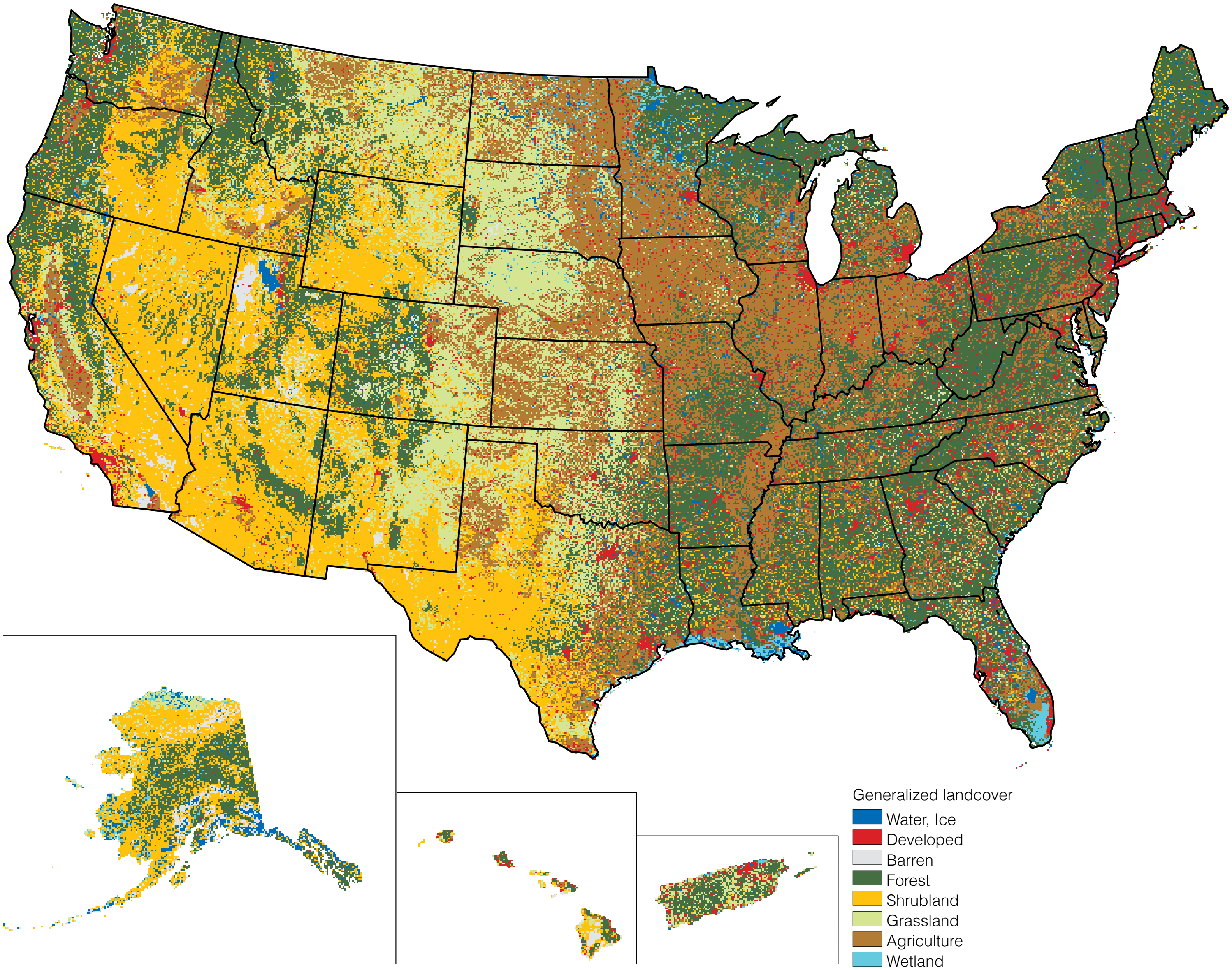

Figure 5.1—The input data was an eight-class version of the 2001 National Land Cover Database (NLCD) landcover map. Note the scale differences for Alaska, Hawaii, and Puerto Rico (insets) in comparison to the conterminous United States.

Table 5.1—The original National Land Cover Database (NLCD) landcover legend as condensed to an eight-class legend

NLCD landcover legend	Eight-class legend
Water Perennial ice/snow	Water
Developed, open space Developed, low intensity Developed, medium intensity Developed, high intensity	Developed
Barren land (rock/sand/clay)	Barren
Deciduous forest Evergreen forest Mixed forest Woody wetlands	Forest
Shrub/scrub	Shrubland
Grassland/herbaceous	Grassland
Pasture/hay Cultivated crops	Agriculture
Emergent herbaceous wetlands	Wetland

total area among those eight classes is shown in table 5.2. Ocean area adjacent to land was included in the measurements, but data summarization was limited to the boundaries of detailed State maps (fig. 5.1) (ESRI 2005).

Area Density

Area density is a measure of landcover dominance that describes a given location on a landcover map by the proportion of a surrounding neighborhood that is a specified landcover type. For this analysis, three specific types of area density were defined by looking separately at forest pixels (forest area density), grassland pixels (grassland area density), and shrubland pixels (shrubland area density). Six measurement scales were defined by six neighborhood sizes[2] of 10.9 acres (4.41 ha; 7 pixels x 7 pixels), 37.6 acres (15.21 ha; 13 x 13), 162 acres (65.61 ha; 27 x 27), 1,460 acres (590.49 ha; 81 x 81), 13,100 acres (5 314.41 ha; 243 x 243), and 118,000 acres (47 829.69 ha; 729 x 729). Six measurement scales were used because fragmentation naturally is scale-dependent, because the effects of fragmentation may be scale-dependent, and because knowledge of fragmentation as manifested at different scales is required to inform resource management as practiced at those different scales.

For a given landcover type and measurement scale, a measurement was taken separately for each location defined by the 10.4 billion subject

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[2] The neighborhood sizes are hereafter shown in acres with three significant digits.

Table 5.2—Percent of total area covered by eight generalized landcover types, national and by Resource Planning Act (RPA) region, according to the 2001 National Land Cover Database landcover map

	Land cover type							
	Water	Developed	Barren	Forest	Shrubland	Grassland	Agriculture	Wetland
RPA regions[a]	Percent of total area							
Alaska	8.92	0.09	8.41	28.97	43.05	7.27	0.02	3.26
North	2.57	8.90	0.24	42.63	1.15	1.95	40.90	1.64
Pacific Coast	1.26	5.23	3.32	32.32	37.33	9.13	10.78	0.62
Rocky Mountain	1.13	2.11	1.86	16.26	33.86	27.25	16.84	0.70
South	2.17	7.07	0.38	39.17	15.19	10.96	23.22	1.84
National	2.91	4.50	2.40	30.08	25.21	13.84	19.51	1.55

[a] RPA regions are illustrated in figure 5.3.

pixels on the landcover map. The result of a given measurement was stored in a new map at the location of the subject pixel, with the same pixel size. A pixel value on such a map describes the area density within the surrounding neighborhood. Eighteen new maps were created by repeating the process for each combination of six measurement scales and three focal landcover types. All of the measurements for a given landcover type were then converted from a continuous variable to a categorical variable by using a classification model (table 5.3) that identified seven classes called intact, interior, dominant, transitional, patchy, rare, and none. For example, on a map of grassland area density measured at the 10.9-acre scale, a transitional

Table 5.3—All measurements for a given landcover type were converted from a continuous area density proportion to an area density class categorical variable using a classification model

Area density (p) measurement	Area density class
$p = 1.0$	Intact
$0.9 \le p < 1.0$	Interior
$0.6 \le p < 0.9$	Dominant
$0.4 \le p < 0.6$	Transitional
$0.1 \le p < 0.4$	Patchy
$0.0 \le p < 0.1$	Rare
$p = 0.0$	None

pixel was surrounded by a 10.9-acre neighborhood that contained 40 percent to 60 percent grassland landcover. This example emphasizes that area density is a measure of the neighborhood context of a given location.

The 18 maps of area density were then post-stratified by geographic overlays upon the original landcover map, to extract 6 maps of forest area density values for the pixels that were forest on the landcover map, 6 maps of grassland area density values for the grassland pixels, and 6 maps of shrubland area density for the shrubland pixels. The area density class called “none” never appears on those extracted maps because there is always at least one pixel in a neighborhood (i.e., the subject pixel itself) which is the focal landcover type.

Figure 5.2 illustrates the classification and post-stratification of forest area density. For clarity, a portion of the NLCD landcover map was converted to a legend showing forest and nonforest pixels (fig. 5.2A). For each pixel on that map, the proportions of forest pixels in surrounding neighborhoods of size 37.6 acres and 1,460 acres were calculated and converted to forest area density class values (figs. 5.2B and 5.2C, respectively). In comparison to larger neighborhoods, smaller neighborhoods portray more local detail of area density (or, equivalently, are more sensitive to higher-frequency variation in the spatial domain). In figures 5.2B and 5.2C, every pixel has a forest area density class, including pixels that were nonforest on the input map (fig. 5.2A). The results of post-stratifying figures 5.2B and 5.2C by geographic overlays with figure 5.2A are shown as figures 5.2D and 5.2E, respectively. The post-stratification retained only the area density class values for pixels that were forest landcover, and necessarily did not include any of the area density class called “none.”

For data summaries, the Resources Planning Act (RPA) assessment regions (fig. 5.3) were selected for consistency and comparability with other Forest Service national resource assessments. The area density class values for the maps illustrated in figures 5.2D and 5.2E were summarized within RPA regions by the percentages of all forest (or grassland or shrubland) pixels in the six remaining area density classes. Note that a percentage was based on the total area of forest (or grassland or shrubland) that was actually present in a given region, not the total area of the region itself. The process of post-stratification followed by geographic aggregation focuses interpretations on the relative fragmentation of the existing landcovers, as distinguished from differences in the absolute amounts or historic losses of landcover in different regions. Comparisons may be made across geographic units even though the units are different sizes, and across landcover types even though there are different absolute amounts of those landcover types. Selected statistics were summarized by county to illustrate geographic trends nationally in map format.

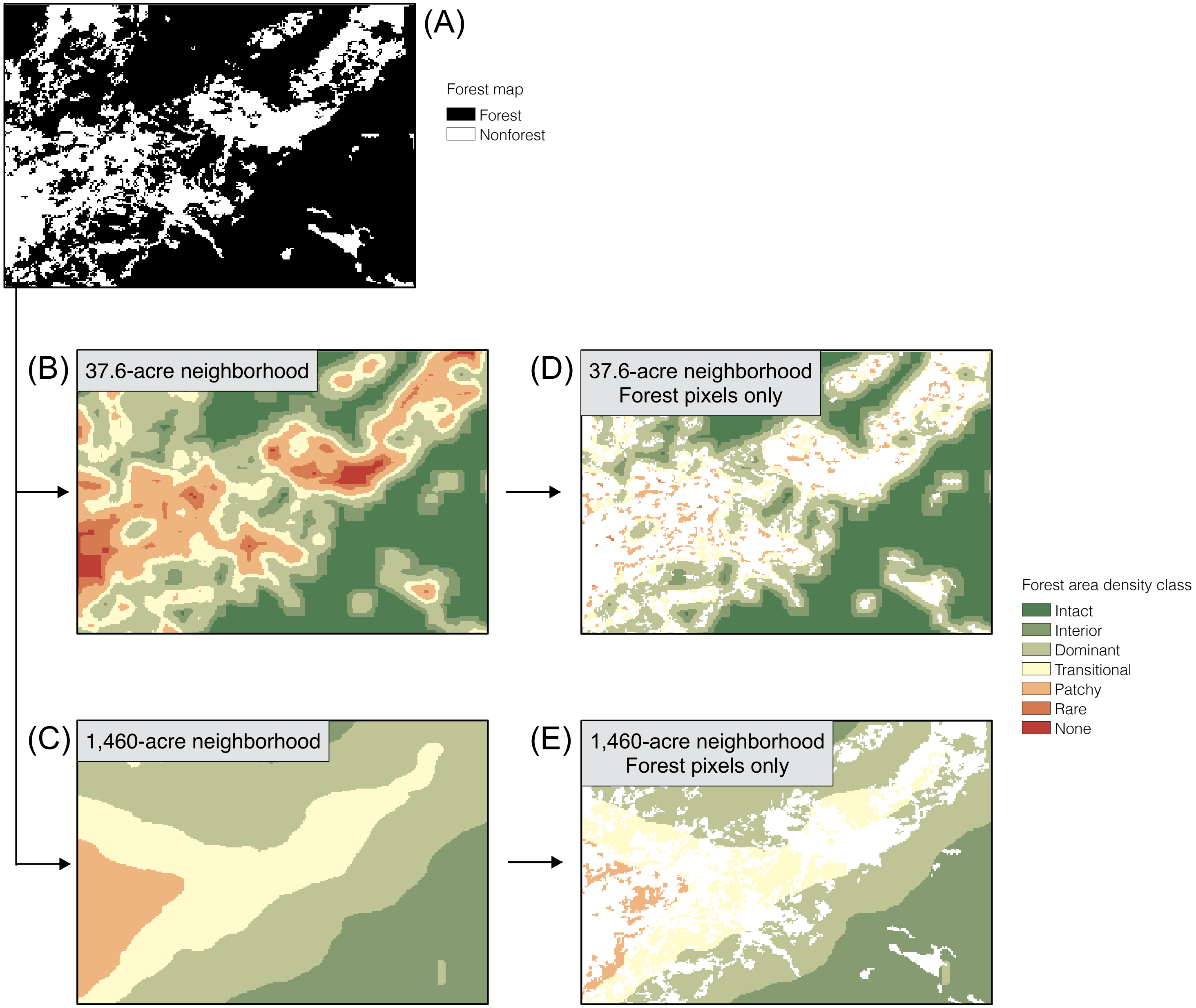

Figure 5.2—Illustration of input and output maps for forest area density mapping. See text for explanation.

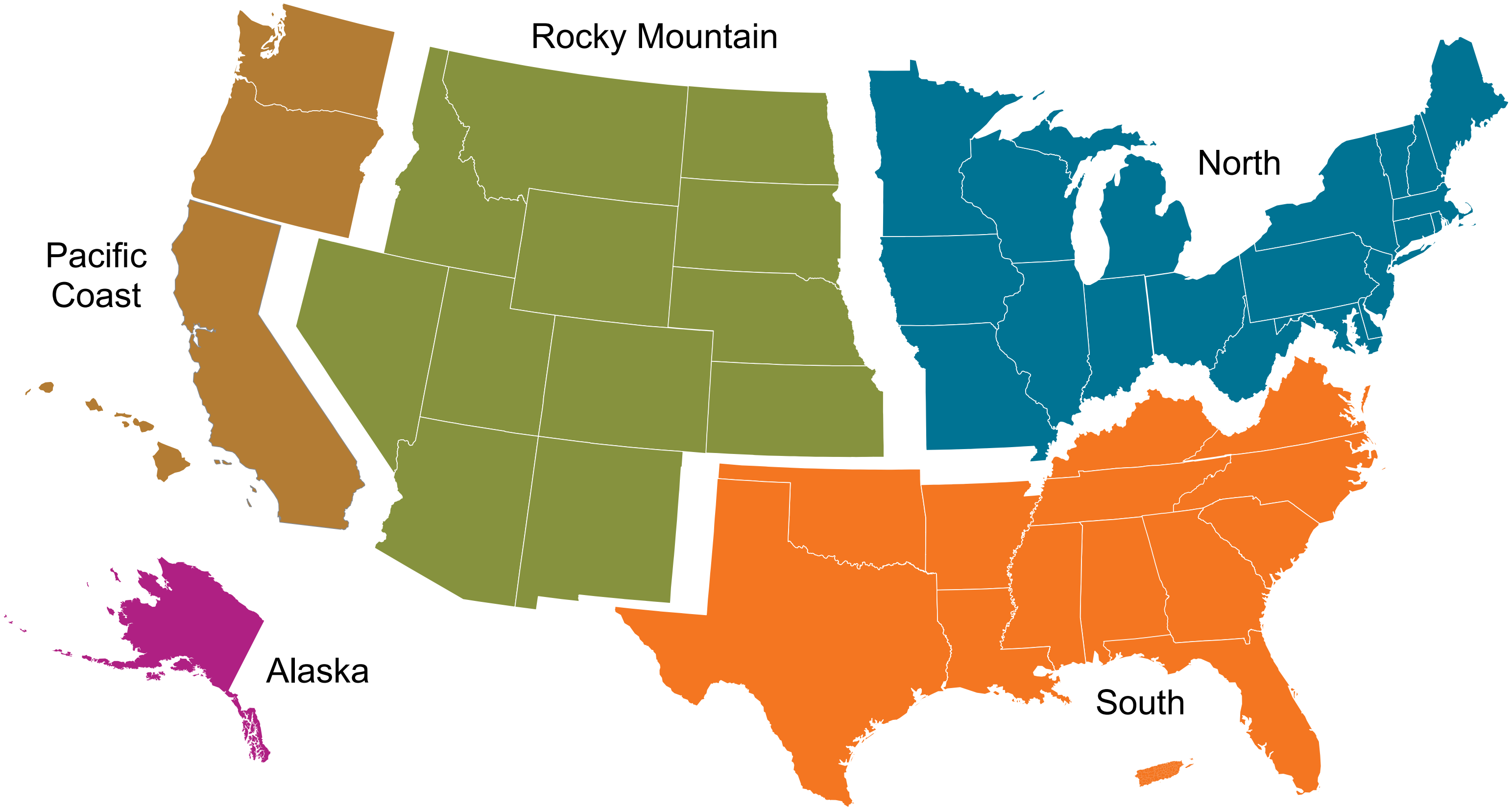

Figure 5.3—Illustration of Resources Planning Act (RPA) assessment regions. Note: Alaska is not shown to scale with map of the conterminous United States.

RESULTS AND DISCUSSION

The percentages of total forest, grassland, and shrubland in each of six area density classes, nationally and by region, are shown in figure 5.4 for each of six measurement scales. In the following discussion, the six area density classes are interpreted as a gradient of fragmentation from low fragmentation (intact class) to high fragmentation (rare class). The results for grassland and shrubland in the North region are often quite different from other regions only because the North region contains very small percentages (< 2 percent) of those two landcover types (table 5.2).

Considering first the general trends of intact and interior landcover with increasing neighborhood size, there is more intact landcover in smaller neighborhoods and less intact landcover in larger neighborhoods, for all three types of landcover considered. That is because of a high degree of spatial autocorrelation of each landcover type, which results in locally intact forest, grassland, or shrubland. At the national scale, approximately one-half of all forest, one-half of all grassland, and one-half of all shrubland are labeled as either interior or intact at 10.9-acre scale, and the smaller percentages observed for grassland and shrubland in the North and South regions may be attributed to the relative scarcity of grassland or shrubland in those regions (table 5.2). However, over larger neighborhoods the pervasiveness of human land uses (e.g., roads) prevents the occurrence of large intact regions of natural landcover types (Riitters and Wickham 2003). As a result, the percentages of interior plus intact forest and grassland decrease rapidly with increasing neighborhood size; decreases are also observed for shrubland but they are less dramatic in comparison to forest and grassland. Nationally, within neighborhoods 1,460 acres and larger, < 1 percent of total forest and total grassland are characterized as intact, and < 25 percent are characterized as interior. The comparable percentages for shrubland are approximately twice the values obtained for forest and grassland.

In comparison to interior plus intact landcover, the percentages of intact plus interior plus dominant landcover exhibit smaller decreases with increasing neighborhood sizes. That is because each landcover type tends to dominate in the areas where it occurs, even if it is fragmented. That can be seen, for example, by noting on the original landcover map (fig. 5.1) the existence of large geographic regions that appear to be mostly-forested, mostly-grassland, or mostly-shrubland, as driven by regional climate differences that favor one or another of those landcover types. While competing human land uses remove natural landcover and introduce some degree of fragmentation almost everywhere, most of the remnant forest, grassland, and shrubland still exists in landscapes where that same natural landcover still dominates the landscape (Riitters and others 2002). Human land uses tend to occur either as inclusions on a background of natural landcover, or they have removed so much of the natural landcover in a given area, e.g., a city, that the remnant natural landcover has a minor influence on aggregate regional and

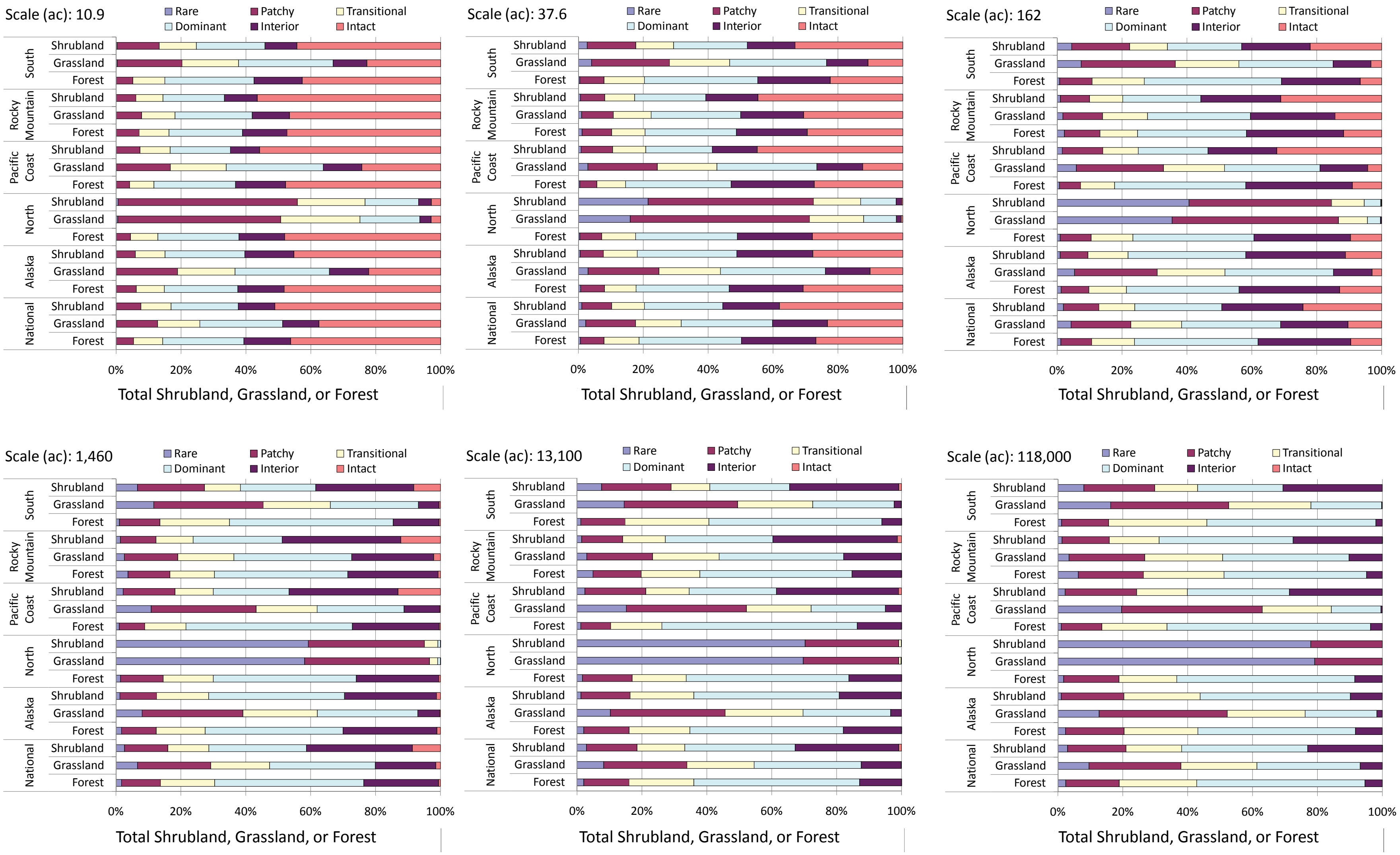

Figure 5.4—The percentage of total forest, grassland, and shrubland in each of six area density classes, for six measurement scales, nationally and by region ("ac" in the figure means acre).

national statistics (Riitters and others 2009). Thus, grassland is more heavily fragmented than forest or shrubland landcover over the largest neighborhood size; approximately 40 percent of grassland is contained in 118,000-acre neighborhoods labeled as intact plus interior plus dominant, in comparison to approximately 60 percent of forest and shrubland.

Another way to interpret the summary statistics is in terms of the likely mechanisms by which different types of fragmentation effects may be caused. For example, a wildlife habitat quality model may distinguish the effects of "edge" (Murcia 1995), "matrix" (Ricketts 2001), or "isolation" (MacArthur and Wilson 1967). In terms of the summary statistics in figure 5.4, edge effects may be expected anywhere the landcover is not in the intact class, and the influence of different "edge widths" can be gauged because of the correspondence between neighborhood size and minimum edge width in an intact neighborhood (Riitters and others 2002). For an edge width of approximately 550 feet (an intact 10.9-acre neighborhood), at least half of all area of the existing forest, grassland, and shrubland in the United States is habitat edge and is therefore subject to potential edge effects. In contrast, matrix effects are likely when habitat is not dominant within a neighborhood. In comparison to potential edge effects in 10.9-acre neighborhoods, potential matrix effects likely impact less than one-quarter of the total forest, grassland, or shrubland area because those landcover types tend to be dominant if they occur at all in a neighborhood. Finally, isolation effects occur when habitat is physically separated from other habitat, which is most common when landcover is in the rare and patchy area density class. On that basis, isolation effects are likely relevant for even smaller proportions of overall habitat area, with the exception of those places where overall habitat area is itself low.

Whether or not landcover patterns can be interpreted in specific habitat terms, public attention is often focused on identifying the locations with relatively unfragmented forest, grassland, and shrubland. Those locations may be considered priority areas for the conservation of relatively intact landcover patterns, or as demonstration regions where the existing landcover is arranged into relatively compact patterns even if there is not much of that landcover. To illustrate these locations, the maps of area density classes were re-aggregated by county and the statistics were summarized as maps showing the percentage of existing forest, grassland, or shrubland that were in the intact plus interior area density classes. Figures 5.5 and 5.6 show the results for the 37.6-acre and 1,460-acre neighborhood sizes, respectively. These figures illustrate trends noted earlier; the proportions of intact plus interior forest and grassland are much more sensitive to neighborhood size than the same proportions for shrubland.[2]

As might be expected, counties containing a high proportion of a given landcover type must contain relatively high proportions of intact plus

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[2] A complementary analysis of area density statistics without post-stratification by landcover type is presented in Riitters (2011).

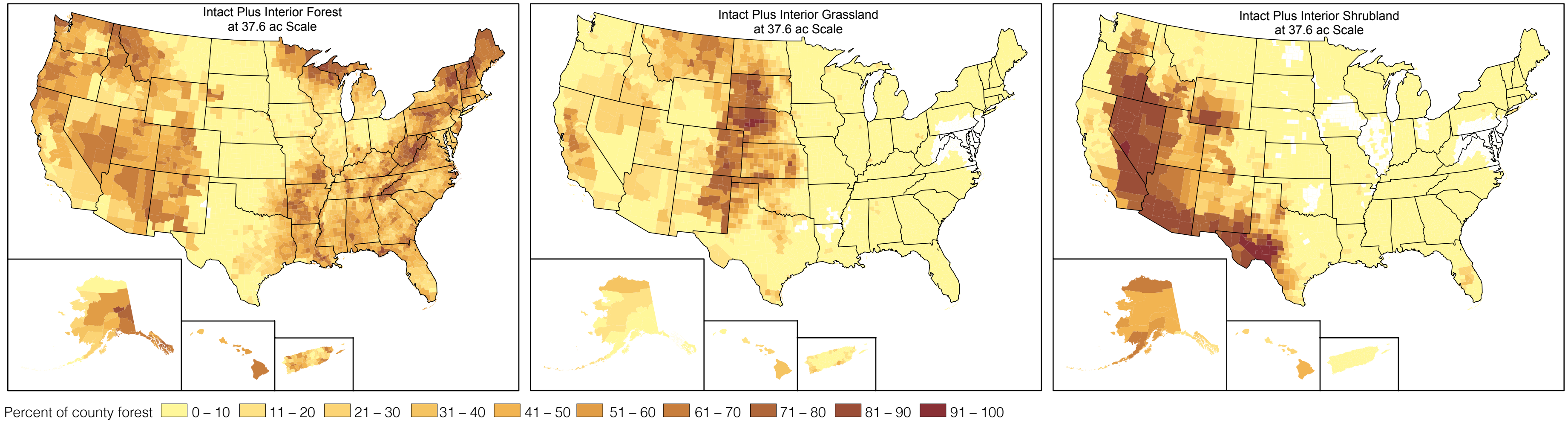

Figure 5.5—The percentage of total county forest, grassland, and shrubland that was labeled as intact or interior, for the 37.6-acre measurement scale. Counties lacking a given landcover type are not shaded.

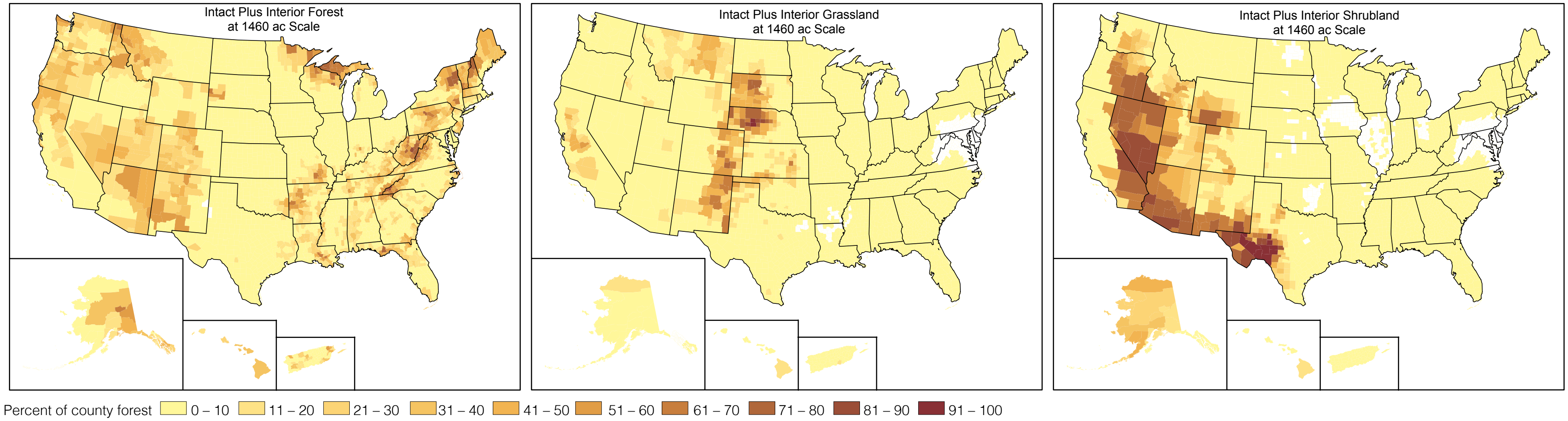

Figure 5.6—The percentage of total county forest, grassland, and shrubland that was labeled as intact or interior, for the 1,460-acre measurement scale. Counties lacking a given landcover type are not shaded.

interior of that landcover type (compare to fig. 5.1). However, it is not obvious from looking at the landcover map (fig. 5.1) that some counties exhibit high proportions of intact and interior landcover even if they contain relatively small amounts of that landcover. For example, many counties in the Intermountain West, e.g., in Nevada and Utah, contain relatively high proportions of intact plus interior forest because the existing forest, while relatively less abundant, is spatially concentrated in the higher-elevation parts of those counties.

In summary, landcover patterns affect many social perceptions and ecological processes within a landscape. Assessments of area density at multiple scales from national landcover maps provide insights about the fragmentation of forest, grassland, and shrubland that are not evident by just looking at a landcover map. Most of the existing forest, grassland, and shrubland in the United States is relatively intact at fine spatial scales and highly fragmented at coarser scales. By all measures the grassland landcover type is more fragmented than the other two, and the shrubland landcover type is least fragmented at coarser scales.

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CHAPTER 6.

An Improved Method for Standardized Mapping of Drought Conditions

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William D. Smith,
and John W. Coulston

INTRODUCTION

Virtually all U.S. forests experience droughts, although the intensity and frequency of the droughts vary widely between, as well as, within forest ecosystems (Hanson and Weltzin 2000). Generally, forests throughout the Western United States are subject to annual seasonal droughts, while forests in the Eastern United States can be characterized by one of two predominant patterns: random, occasional droughts (in the Appalachian Mountain region and the Northeast) and common late-summer droughts (in the Southeastern Coastal Plain and near the eastern edge of the Great Plains) (Hanson and Weltzin 2000). In terms of impacts, a reduction in basic growth processes, i.e., cell division and enlargement, is the most immediate plant response to drought; photosynthesis, which is less sensitive than these basic processes, decreases slowly at low levels of drought stress, but begins to decrease more sharply when the stress becomes moderate to severe (Kareiva and others 1993, Mattson and Haack 1987). Drought stress also makes some forests more susceptible to infestations of tree-damaging insects and diseases (Clinton and others 1993, Mattson and Haack 1987). Furthermore, by impeding decomposition of organic matter and reducing the moisture content of downed woody materials and other potential fuels, drought may substantially increase wildland fire risk (Clark 1989, Keetch and Byram 1968, Schoennagel and others 2004).

In the 2008 national report by the Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture, we outlined an approach for mapping drought stress using historical, high-spatial-resolution climate data (Koch and others 2012a). We proposed this methodology as a means to generate outputs that would offer forest managers and researchers a finer-scale alternative to spatial data products currently available from such sources as the National Climatic Data Center (NCDC) (NCDC 2007) or the U.S. Drought Monitor program (Svoboda and others 2002). We began by developing annual moisture index maps covering a 100-year period (1907–2006) for the conterminous United States using gridded climate data (approximately 4-km² spatial resolution) created with the Parameter-elevation Regression on Independent Slopes (PRISM) climate mapping system (Daly and others 2002). We then calculated per-map-cell differences between each year's moisture index map and a corresponding long-term "normal" moisture index map, which represented the mean of the 100 annual maps. Based on the resulting difference values as well as characteristics of the values' statistical distribution through time, we assigned each map cell to one of nine categories ranging from extreme wetness to extreme drought, thus allowing us to create national maps of drought conditions for each year in our 100-year study period. Maps demonstrating the methodology can be found in the 2008 FHM National Technical Report (Koch and others 2012a).

Evidence suggests that forests are relatively resistant to short-term drought events (Archaux

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and Wolters 2006), although individual tree species differ in their responses to drought (Hinckley and others 1979, McDowell and others 2008). Arguably, the duration of a drought event is more critical than its intensity (Archaux and Wolters 2006); for instance, multiple consecutive years of drought (2 to 5 years) are more likely to result in high tree mortality than a single dry year (Guarín and Taylor 2005, Millar and others 2007). Therefore, to provide a more realistic characterization of drought impact in forested areas, we expanded our methodology to examine moisture conditions in the United States over longer (i.e., multi-year) time windows. Historical and recent examples illustrating our multi-year methodology, again focusing on a 100-year study period (1908–2007 in this case), can be found in the 2009 FHM National Technical Report (Koch and others 2012b).

In the current chapter, we present a revised drought mapping methodology that expands upon our previous work in two key ways. Primarily, we have implemented a standardized drought indexing method, such that we can more easily compare, for any given location, its drought status during different time windows, regardless of their length (e.g., allowing comparison between 1-year, 3-year, and 5-year time windows). Moreover, this improved standardization permits analysis of specific and relatively short time windows (i.e., a single season rather than an entire year). Such analyses may have great relevance when estimating the risks associated with certain forest pests that are able to exploit acute drought stress in host trees.

We highlight the potential utility of short-term drought analysis using the example of the oak splendor beetle (*Agrilus biguttatus*), a buprestid beetle found throughout Europe but considered a major threat to North American oak forests if it were introduced and successfully able to establish.

METHODS

When we performed these analyses, monthly PRISM grids for total precipitation, mean daily minimum temperature, and mean daily maximum temperature were available from the PRISM group Web site (PRISM Group 2010) for all years from 1895 to 2009. Each gridded dataset covered the entire conterminous United States.

Potential Evapotranspiration Maps

As in our previous analyses (Koch and others 2012a,b), we adopted an approach in which a moisture index value for each location of interest, i.e., each grid cell, was calculated based on both precipitation and potential evapotranspiration values for that location during the time period of interest. Potential evapotranspiration is a measure of the loss of soil moisture through plant uptake and transpiration (Akin 1991). It does not represent actual moisture loss, but rather the loss that would occur under ideal conditions, i.e., if there was no lack of moisture for plants to transpire (Akin 1991, Thornthwaite 1948). The inclusion of both precipitation and potential evapotranspiration provides a fuller accounting of a location's water balance than precipitation alone.

So, to complement the available PRISM monthly precipitation grids, we computed corresponding monthly potential evapotranspiration (PET) grids using the Thornthwaite formula (Akin 1991, Thornthwaite 1948):

$$PET_m = 1.6 L_{lm} (10 \frac{T_m}{I})^a \qquad (1)$$

where

PET_m = the potential evapotranspiration for a given month *m* in cm

L_{lm} = a correction factor for the mean possible duration of sunlight during month *m* for all locations, i.e., grid cells, at a particular latitude *l* [see Table V in Thornthwaite (1948) for a list of *L* correction factors by month and latitude]

T_m = the mean temperature for month *m* in °C

I = an annual heat index, calculated as,

$$I = \sum_{m=1}^{12} \left(\frac{T_m}{5} \right)^{1.514}$$

where

T_m = the mean temperature for each month *i* of the year

a = an exponent calculated as $a = 6.75 \times 10^{-7} I^3 - 7.71 \times 10^{-5} I^2 + 1.792 \times 10^{-2} I + 0.49239$ [see appendix I in Thornthwaite (1948) regarding the empirical derivation of *a*]

To implement equation 1 spatially, we created a grid of latitude values for determining the *L* adjustment for any given 4-km^2 grid cell in the conterminous United States [see Thornthwaite (1948) for a table of *L* correction factors]. We calculated the mean temperature grids for each month by averaging the corresponding PRISM monthly mean minimum and maximum temperature grids.

Moisture Index Maps

We used the precipitation (*P*) and *PET* grids to generate baseline moisture index grids for the past 100 years, i.e., 1910–2009, for the conterminous United States. We used a moisture index, *MI'*, proposed by Willmott and Feddema (1992), which has the following form:

$$MI' = \begin{cases} P/PET - 1 & , \quad P < PET \\ 1 - PET/P & , \quad P \geq PET \\ 0 & , \quad P = PET = 0 \end{cases} \qquad (2)$$

where

P = precipitation

PET = potential evapotranspiration

(*P* and *PET* must be in equivalent measurement units, e.g., mm).

This set of equations yields a dimensionless index scaled between -1 and 1. *MI'* can be calculated for any time period, but is commonly calculated on an annual basis using summed *P* and *PET* values (Willmott and Feddema 1992). An alternative to this summation approach is to calculate MI' from monthly precipitation and potential evapotranspiration values and then, for a given time window of interest, calculate the moisture index as the mean of the MI' values for all months in the window. This "mean-of-months" approach limits the ability of short-term peaks in either precipitation or potential evapotranspiration to negate corresponding short-term deficits, as would happen under a summation approach.

For each year in our study period (1910–2009), we used the mean-of-months approach to calculate moisture index grids for three different time windows: one year (MI_1'), 3 years (MI_3'), and 5 years (MI_5'). Briefly, the MI_1' grids are the mean of the 12 monthly MI' grids for each year in the study period, the MI_3' grids are the mean of the 36 monthly grids from January 2 years prior through December of each year, and the MI_5' grids are the mean of the 60 consecutive monthly MI' grids from January 4 years prior to December of each year. For example, the MI_1' grid for the year 2010 is the mean of the monthly MI' grids from January to December 2010, the MI_3' grid is the mean of grids from January 2008 to December 2010, and the MI_5' grid is the mean of the grids from January 2006 to December 2010.

Annual and Multi-Year Drought Maps

To determine degree of departure from typical moisture conditions, we first created a normal grid, MI'_{norm}, for each of our three time windows, representing the mean of the 100 corresponding moisture index grids (i.e., the MI_1', MI_3', or MI_5' grids, depending on the window; see figure 6.1). We also created a standard deviation grid, MI'_{SD}, for each time window, calculated from the window's 100 individual moisture index grids as well as its MI'_{norm} grid. We subsequently calculated moisture difference z-scores, MDZ_i, for each time window using these gridded data sets:

$$MDZ_i = \frac{MI'_i - MI'_{norm}}{MI'_{SD}} \tag{3}$$

where

i = a particular target year in our 100-year study period, i.e., 1910–2009.

MDZ scores may be classified in terms of degree of moisture deficit or surplus (table 6.1). The classification scheme includes categories (e.g., severe drought, extreme drought) like those associated with the Palmer Drought Severity Index, or PDSI (Palmer 1965). Importantly, because of the standardization in equation 3, the breakpoints between categories remain the same regardless of the size of the time window of interest. To highlight the potential for comparative analysis, we generated classified *MDZ* maps, based on all three time windows, for the target year 2009 (figs. 6.2 and 6.3).

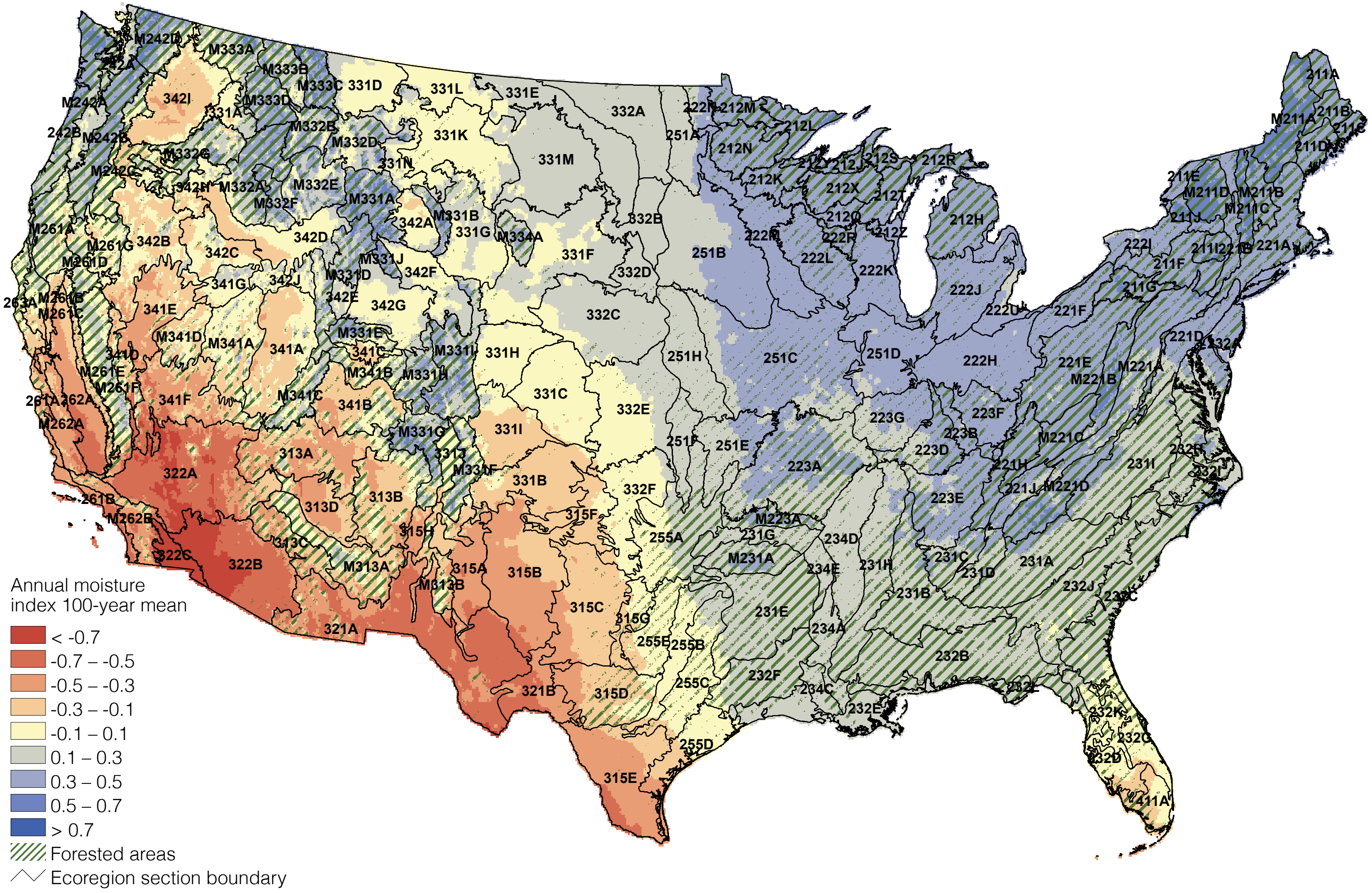

Figure 6.1—The 100-year (1910–2009) mean annual moisture index, or MI_1'*, for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries and labels are included for reference. Forest cover data (overlaid green hatching) derived from MODIS imagery by the U.S. Department of Agriculture Forest Service, Remote Sensing Applications Center. (Data source: PRISM Group, Oregon State University)*

Table 6.1—Moisture difference z-score (MDZ) value ranges for nine wetness and drought categories, along with each category's approximate theoretical frequency of occurrence

MDZ score	Category	Frequency
		percent
< -2	Extreme drought	2.3
-2 to -1.5	Severe drought	4.4
-1.5 to -1	Moderate drought	9.2
-1 to -0.5	Mild drought	15
-0.5 to 0.5	Near normal conditions	38.2
0.5 to 1	Mild moisture surplus	15
1 to 1.5	Moderate moisture surplus	9.2
1.5 to 2	Severe moisture surplus	4.4
> 2	Extreme moisture surplus	2.3

Late Spring-Early Summer Drought Maps for Pest Risk Analysis

In its Invasive Species Information Program Area, the Forest Health Technology Enterprise Team (FHTET) of the Forest Service develops national-scale products for the detection, prevention, and control of nonnative insect and disease species with the potential to significantly impact U.S. forests (USDA Forest Service 2010). Among the products developed by FHTET are risk maps that depict the introduction and establishment potential of the species of interest. Typically, these are species that have either recently been discovered in the United States or that are considered highly likely to be introduced. One pest of interest is the oak splendor beetle (*Agrilus biguttatus*), which is found throughout Europe as well as in Northern Africa, the Middle East, and Russian Asia (Davis and others 2005, Moraal and Hilszczanski 2000). The oak splendor beetle has been implicated in oak decline in Europe (Moraal and Hilszczanski 2000), and while it has not been discovered in the United States, most of the country is believed to be climatically suitable for its establishment (Davis and others 2005).

The primary concern about possible U.S. invasion by the oak splendor beetle is that it could cause extensive mortality in the Nation's oak forests, particularly those already stressed by defoliating insects or drought (Ciesla 2003). A committee convened by FHTET (including the authors of this chapter) determined that two factors were most critical to the establishment potential of the beetle in the United States:

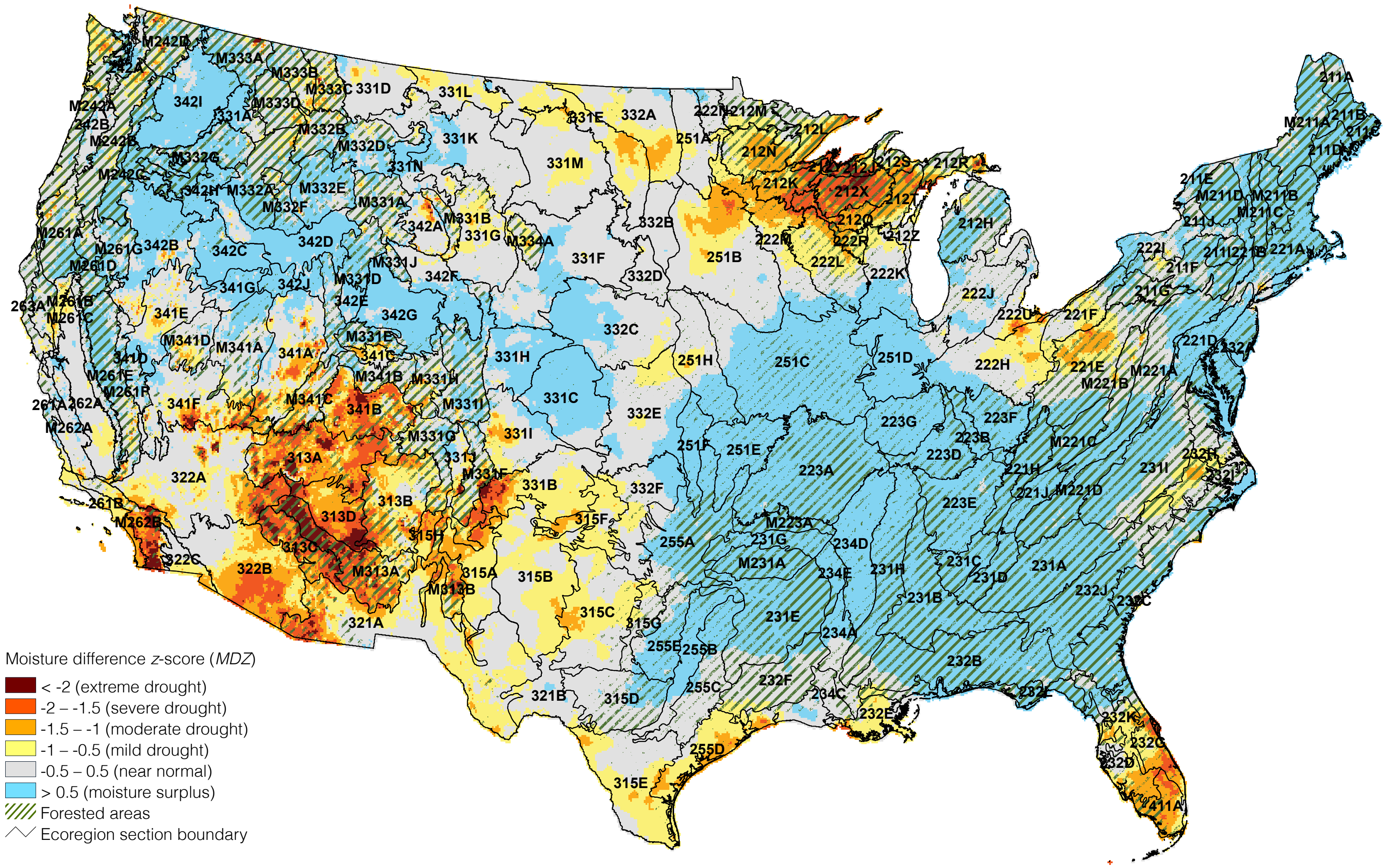

Figure 6.2—The 2009 annual (i.e., 1-year) moisture difference z-score, or MDZ, *for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries and labels are included for reference. Forest cover data (overlaid green hatching) derived from MODIS imagery by the U.S. Department of Agriculture Forest Service, Remote Sensing Applications Center. (Data source: PRISM Group, Oregon State University)*

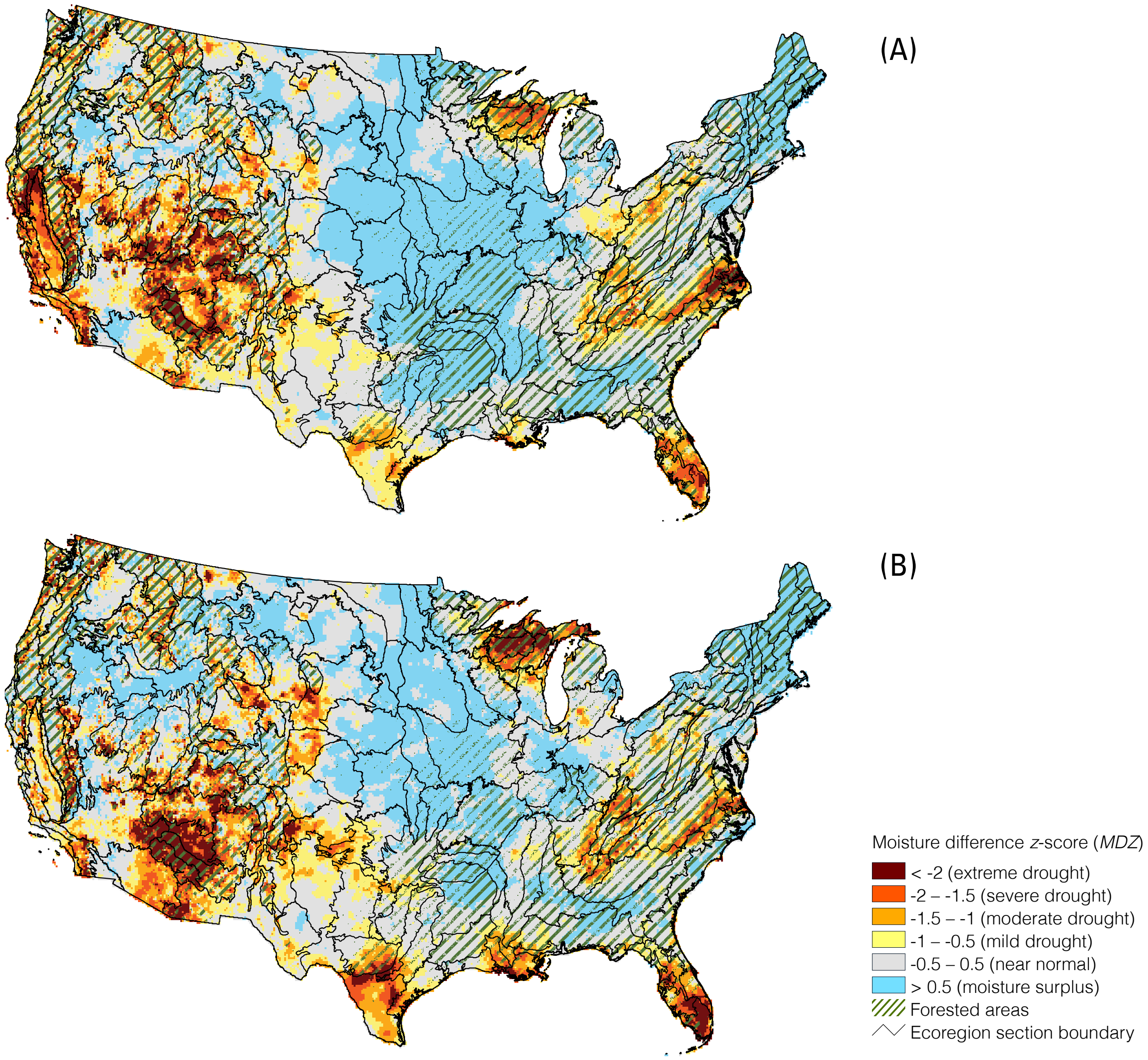

Figure 6.3—(A) The 2007–09 (i.e., 3-year) moisture difference z-score, or MDZ, *for the conterminous United States; (B) the 2005–09 (i.e., 5-year)* MDZ *for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries are included for reference. Forest cover data (overlaid green hatching) derived from MODIS imagery by the U.S. Department of Agriculture Forest Service, Remote Sensing Applications Center. (Data source: PRISM Group, Oregon State University)*

(1) abundance of host, i.e., oak forests, both natural and urban, and (2) the level of drought stress on host trees during late spring-early summer, an approximately 3-month "season" coinciding with beetle emergence from the host trees (Ciesla 2003). With respect to the first factor, phase 2 plot data collected by the Forest Inventory and Analysis (FIA) Program of the Forest Service were spatially interpolated to create a national-scale map of oak host distribution, which was supplemented by the use of National Land Cover Data to estimate forest cover in urban areas. To characterize and represent drought stress for risk mapping purposes, we employed our new, standardized drought mapping methodology to identify areas of the United States that recently (i.e., in the last few years, 2007–09) experienced severe or extreme drought during the late spring-early summer period.

As opposed to the mean-of-months approach used in the previously described analyses, i.e., for 1-year, 3-year, and 5-year time windows, in this case we calculated MI' (Eq. 2) based on the total P and PET values summed over the 3-month period. Notably, late spring-early summer represents a different time window depending on geographic location within the conterminous United States (i.e., depending on latitude, elevation, and climatic regime). For this reason, we actually calculated nationwide MI' grids for three different 3-month windows during each year of our 1910–2009 study period: March–May, April–June, and May–July.

For each of these 3-month windows, we calculated distinct MI'_{norm} and MI'_{SD} grids based on the window's 100 individual MI' grids. We then applied equation 3 to generate distinct MDZ grids for each window in each year of our study period. To combine the March–May, April–June, and May–July MDZ grids for each year into a single nationwide grid depicting late spring-early summer moisture conditions, we first subset them using spatial data related to frost-free period. Basically, these data served to represent the approximate beginning of spring and the growing season. In summary, we divided the conterminous United States into three geographic regions (called zones) (fig. 6.4), based on the 30-year mean Julian date of the last spring freeze: Zone 1, including all areas with a mean Julian date ≤ 90 (i.e., last freeze prior to April 1); Zone 2, all areas with a mean Julian date between 90 and 120 (i.e., last freeze between April 1 and April 30); and Zone 3, all areas with a mean Julian date > 120 (i.e., last freeze after April 30). Next, we matched each 3-month window to the most appropriate zone (fig. 6.4), and then clipped the corresponding MDZ grid to the zonal boundaries. Finally, we mosaicked these clipped grids into a single grid covering the conterminous United States.

For the FHTET model for potential establishment of the oak splendor beetle, we analyzed the late spring-early summer MDZ grids for 2007–09 (fig. 6.5). For each of these 3 years, we identified all U.S. areas that exhibited severe or extreme drought stress during this season. The resulting binary grids (where 1 = severe or extreme drought stress during late spring-early summer and 0 = moderate to no drought stress during this period) were added together using

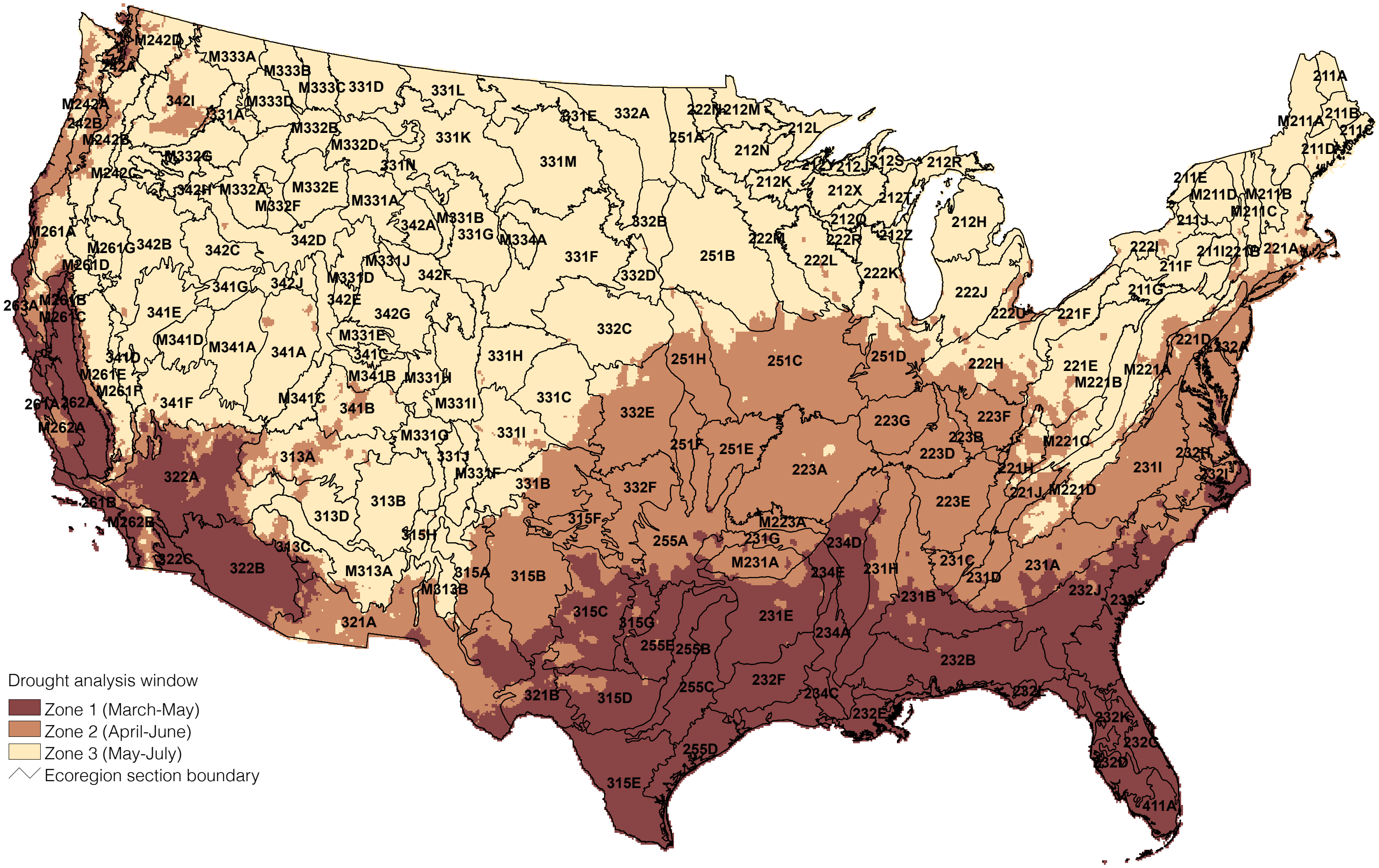

Figure 6.4—Three analysis zones, each corresponding to a particular 3-month time window used when calculating late spring-early summer drought conditions for the associated areas of the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries and labels are included for reference. Zones were developed from data describing frost-free period. (Data source: The Climate Source, LLC, Corvallis, OR)

map algebra, generating a frequency map with values from 0 (no late spring-early summer drought in 2007–09) to 3 (drought during all 3 years). Details about how the drought frequency map and host distribution data were combined to create a final establishment risk map are provided in Downing and others (2010). For comparison, both the 3-year drought frequency map and the final establishment risk map are included here (fig. 6.6).

RESULTS AND DISCUSSION

The 100-year mean annual moisture index, or MI_1', grid (fig. 6.1) serves as a general representation of the climatic regimes for the conterminous United States. (The 100-year mean MI_3' and MI_5' grids differed negligibly from the mean MI_1' grid, and so are not shown here.) In general, wet climates ($MI' > 0$) are typical through the Eastern United States, especially the Northeast. Notably, it appears that southern Florida—in particular, ecoregion sections 232C-Florida Coastal Lowlands-Atlantic, 232D-Florida Coastal Lowlands-Gulf, and 411A-Everglades—is the driest region of the Eastern United States. Although the region typically has a high level of precipitation, the precipitation is more than offset by a high level of potential evapotranspiration, which results in negative MI' values. This pattern, i.e., high P offset by high PET, greatly contrasts with the pattern observable in the driest regions of the Western United States, particularly the Southwest, e.g., ecoregion sections 322A-Mojave Desert, 322B-Sonoran Desert, and 322C-Colorado Desert, where potential evapotranspiration is very high but precipitation levels are typically very low. In fact, dry climates ($MI' < 0$) are common across much of the Western United States because of generally lower precipitation than in the Eastern United States. However, mountainous areas in the central and northern Rocky Mountains as well as the Pacific Northwest are relatively wet, e.g., ecoregion sections M242A-Oregon and Washington Coast Ranges, M242B-Western Cascades, M331G-South-Central Highlands, and M333C-Northern Rockies. This wet climate is likely influenced, at least in part, by winter snowfall.

Figure 6.2 shows the annual (i.e., 1-year) *MDZ* map for 2009 for the conterminous United States. Much of the country exhibited a moisture surplus for 2009, particularly in the East. There were pockets of drought scattered throughout the United States, such as an area of mild to moderate drought in the eastern portion of ecoregion section 232E-Louisiana Coastal Prairie and Marshes, as well as moderate to severe drought in the aforementioned ecoregion sections in southern Florida. Regarding the latter region, the observed conditions for 2009 partially reflect lingering effects from the previous year, with drought peaking in April before shifting back towards near-normal conditions (NCDC 2010). In addition to these and other drought pockets, there were a few U.S. regions with sizeable areas of severe to extreme drought during 2009: the Upper Midwest, especially ecoregion sections 212J-Southern Superior Uplands, 212X-Northern Highlands, and 212Y-Southwest Lake Superior Clay Plain;

the Desert Southwest, especially the forested portions of ecoregion sections M313A-White Mountains-San Francisco Peaks-Mogollon Rim, 313A-Grand Canyon, and 313C-Tonto Transition; and southern California in the forested areas of ecoregion section M262B-Southern California Mountain and Valley.

In fact, the Upper Midwestern United States experienced persistent drought conditions during most of the 7-year period from 2003 to 2009 (NCDC 2010). Likewise, much of the Western United States, particularly the Southwest region, has been regularly subjected to some level of drought for the last 10 to 15 years (Groisman and Knight 2008, Mueller and others 2005, NCDC 2010, O'Driscoll 2007). These prolonged drought conditions are partially captured by the 3-year and 5-year *MDZ* maps for the conterminous United States (fig. 6.3). When combined with the annual *MDZ* map in figure 6.2, these multi-year maps provide an overview of the recent chronology of moisture conditions. For instance, the 5-year *MDZ* map (fig. 6.3B) appears to show more extensive and/or severe drought conditions than the 3-year *MDZ* map (fig. 6.3A) in nearly all of the aforementioned geographic regions: southern Florida, the Desert Southwest, the Upper Midwest, as well as southern Texas, i.e., forested areas in section 315D-Edwards Plateau. This discrepancy may reflect longer-term persistence of drought in the regions of interest; however, it may also mean that the historically worst drought years in these regions are simply less recent than in, for instance, a region of the Southeastern United States, i.e., in parts of ecoregion sections 231I-Central Appalachian Piedmont, 232H-Middle Atlantic Coastal Plain and Flatwoods, and 232I-Northern Atlantic Coastal Plain and Flatwoods, where the 3-year *MDZ* map shows worse drought conditions than the 5-year map. In the latter case, a historically exceptional drought that occurred during 2007 (O'Driscoll 2007) is likely the major factor behind the difference in the two maps. Note also that the 1-year *MDZ* map for 2009 (fig. 6.2) shows only mild to moderate drought conditions in this region, suggesting that the 2007 exceptional drought was a relatively short-term event.

Similarly, the 3-year *MDZ* values (fig. 6.3A) in much of northern California (e.g., ecoregion sections M261C-Northern California Interior Coast Ranges, M261F-Sierra Nevada Foothills) and M262A-Central California Coast Ranges, are lower than the corresponding 5-year *MDZ* values (fig. 6.3B). Since this region experienced persistent drought starting in 2007 (NCDC 2010), the difference between the two multi-year maps serves to highlight the fact that the region must have also experienced near normal or even moisture surplus conditions during 2005 and 2006 (NCDC 2009a, b), offsetting somewhat the apparently severe or extreme drought conditions during subsequent years. The 2009 annual *MDZ* map (fig. 6.2) suggests a recent improvement of moisture conditions in northern California. This apparent fluctuation of conditions over the course of the last several years suggests a need for future monitoring of the region.

The late spring-early summer *MDZ* maps (fig. 6.5) depict a fairly dramatic shift in seasonal moisture conditions during the 3-year

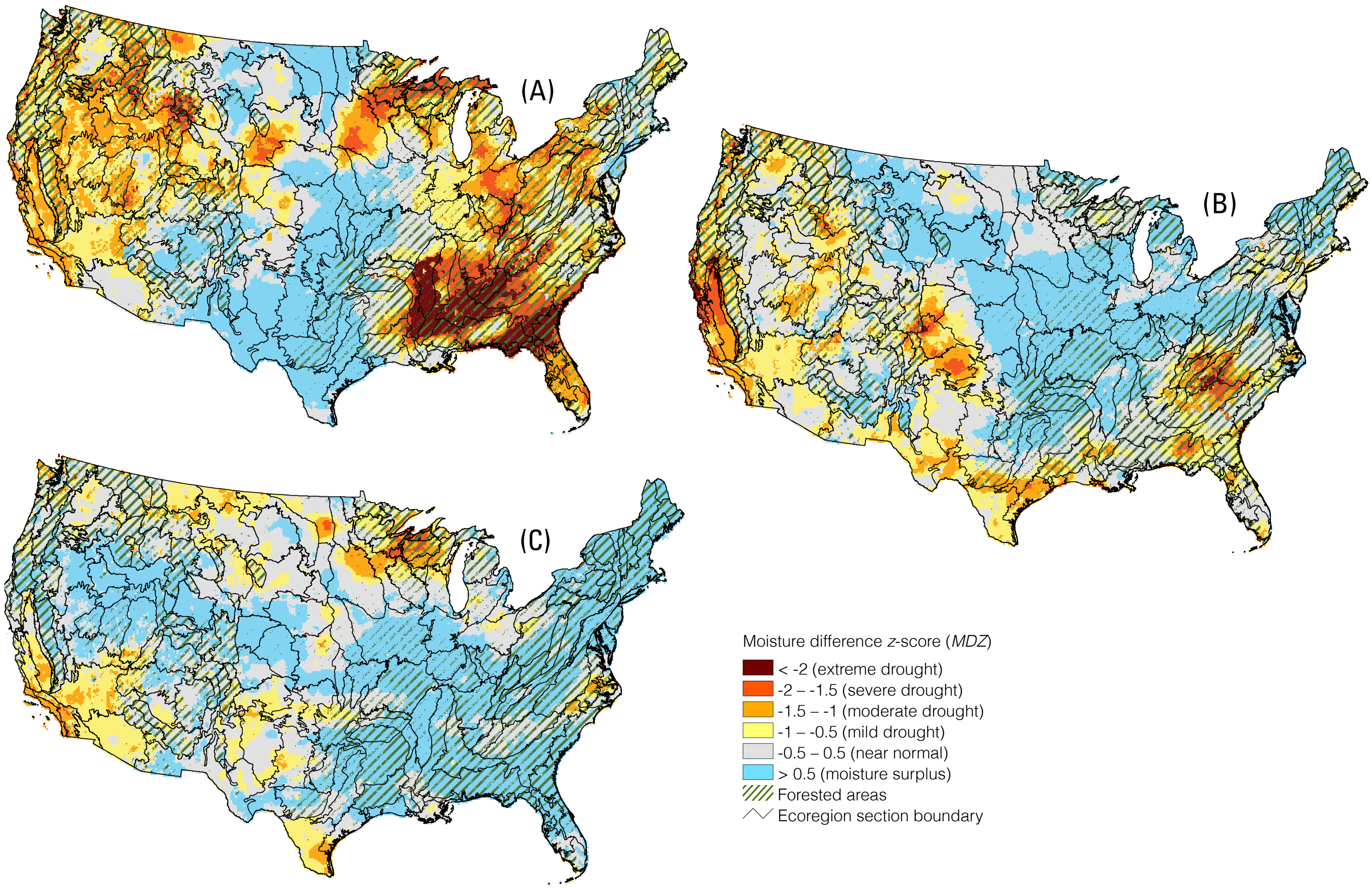

Figure 6.5—(A) The 2007 late spring-early summer moisture index z-score, or MDZ, *for the conterminous United States; (B) the 2008 late spring-early summer* MDZ; *(C) the 2009 late spring-early summer* MDZ. *Ecoregion section (Cleland and others 2007) boundaries are included for reference. Forest cover data (overlaid green hatching) derived from MODIS imagery by the U.S. Department of Agriculture Forest Service, Remote Sensing Applications Center. (Data source: PRISM Group, Oregon State University)*

period from 2007 to 2009. Foremost, a large portion of the Southeastern United States was subjected to extreme drought during late spring-early summer 2007 (fig. 6.5A). At the same time, much of the remainder of the East experienced mild to severe drought, as did most areas west of the Rocky Mountains. In contrast, most of the Great Plains region experienced a moisture surplus. In late spring-early summer 2008 (fig. 6.5B), most of the country faced no worse than mild drought conditions, although there were hot spots of severe to extreme drought in northern California, especially ecoregion section M263-Northern California Coast; east of the Central Rocky Mountains reaching into forested areas of ecoregion section M331I-Northern Parks and Ranges; and in the Southern Appalachian Mountains, especially the southern portion of ecoregion section M221D-Blue Ridge Mountains. In late spring-early summer 2009, most of the country experienced near normal or moisture surplus conditions, with the most notable exceptions being two geographic regions previously noted in the annual and multi-year *MDZ* maps (figs. 6.2 and 6.3): southern California and the Upper Midwest.

In the Southern Appalachian Mountains, the Upper Midwest, and northern California, there were sizeable areas where severe or extreme seasonal drought occurred in 2 out of 3 years between 2007 and 2009 (fig. 6.6A). In turn, these areas have been labeled as having extreme establishment potential for the oak splendor beetle (fig. 6.6B), despite a fair amount of variation between the regions in terms of the distribution and abundance of oak hosts. There were a handful of small patches where severe or extreme drought occurred during late spring-early summer of all 3 years (2007–09), in the previously noted ecoregion sections M262A and M262B, as well as M242B-Western Cascades, M242D-Northern Cascades, and M333A-Okanogan Highland (fig. 6.6A). However, none of these locations showed an elevated establishment potential for the beetle, presumably because of a lack of suitable host.

In summary, these results demonstrate the kinds of analyses that are possible with our newly standardized moisture difference index. If the most recent spatial data, i.e., the high-resolution maps of precipitation and temperature, underlying these analyses continue to be made available for public use, then the approaches described here—or similar approaches—could be installed as standard components of national-scale forest health reporting. Nevertheless, it is important for users to interpret and compare the *MDZ* drought maps cautiously. Although the maps use a standardized index scale that applies regardless of the window, it should also be understood that, for example, an extreme drought (i.e., where $MDZ < -2$) that persists over a 5-year period has substantially different forest health implications than an extreme drought over a 1-year period. Because this is a new methodology, we are still in the process of determining what analyses are most appropriate and possible. This will be a focus of future work.

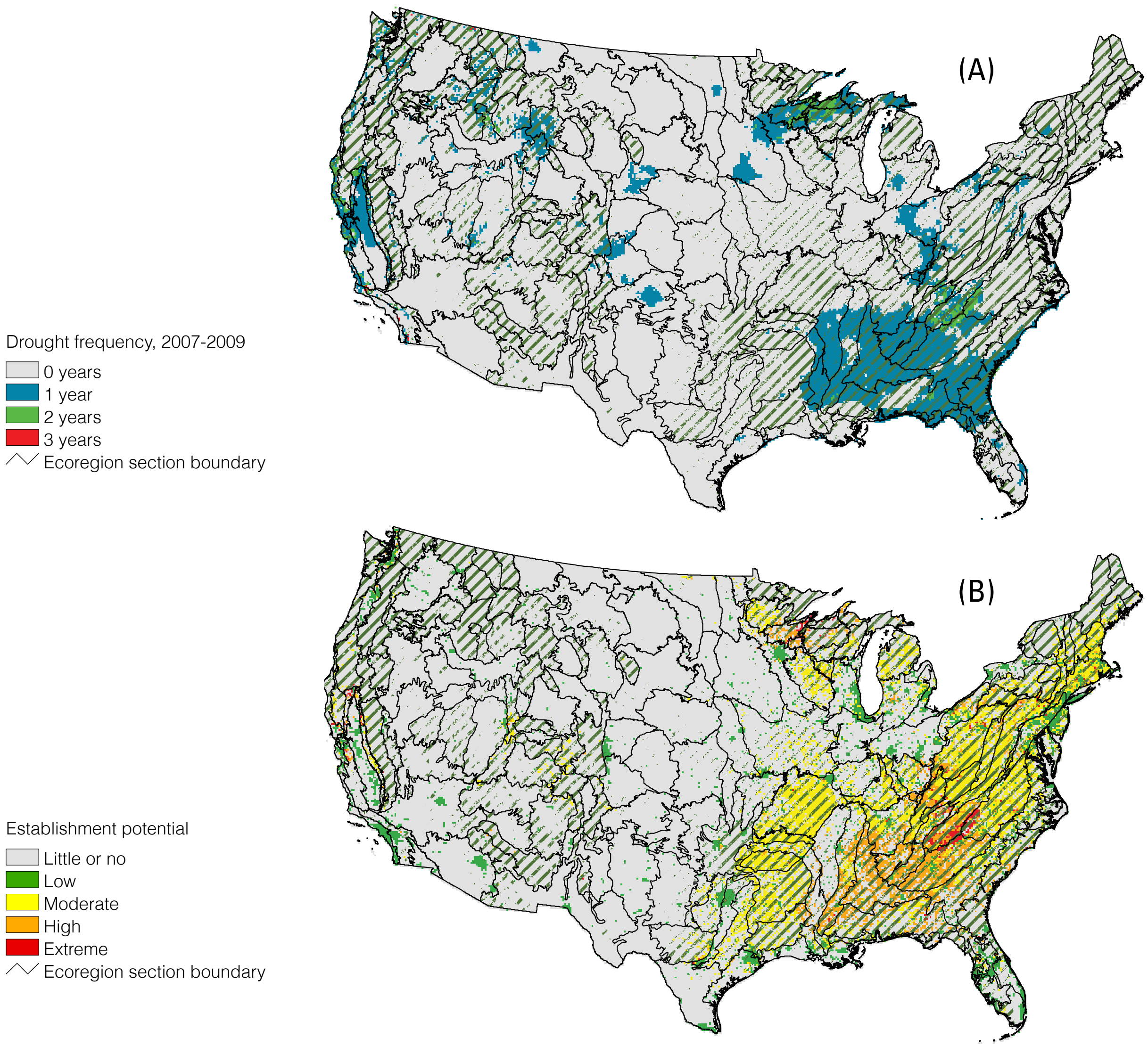

*Figure 6.6—(A) The annual frequency of severe or extreme drought during late spring-early summer over a 3-year period, 2007–09; (B) establishment potential for oak splendor beetle (*Agrilus biguttatus*) in the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries are included for reference. [Data sources: PRISM Group, Oregon State University (drought frequency); U.S. Department of Agriculture Forest Service, Forest Health Technology Enterprise Team (establishment potential)]*

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CHAPTER 7.

Standing Dead Tree Resources in Forests of the United States

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and James E. Smith

INTRODUCTION

Given the importance of standing dead trees to numerous forest ecosystem attributes/processes such as fuel loadings and wildlife habitat, the Forest Inventory and Analysis (FIA) Program of the Forest Service, U.S. Department of Agriculture, initiated a consistent nationwide inventory of standing dead trees in 1999. As the first cycle of annual standing dead tree inventories nears full national implementation, the goal of this study was to conduct one of the first empirical assessments of the Nation's standing dead tree resources. Results indicate that there are a substantial number of standing dead trees in forests across the United States, exceeding 10 billion nationwide and consisting of mostly small-sized trees (< 30 cm d.b.h.). Forests in the Rocky Mountains and Pacific Northwest have some of the largest mean biomass of standing dead trees per unit of forest land (+3 Mg/ha), whereas Plains States had the least. The species composition of standing dead trees is quite diverse with over 130 species having more than 1 million Mg each nationwide, but is dominated by western tree species (e.g., Douglas-fir, +200 million Mg). Given the emerging role of standing dead trees in biomass/bioenergy economies and carbon cycling, continued monitoring of this resource is highly warranted.

Standing dead trees, sometimes referred to as snags, may be defined as remnants of once living trees that are still upright, self-supported, and lean less than 45° from vertical (USDA Forest Service 2006). Standing dead trees are an integral component of forest ecosystems, enhancing the structural diversity of forests of all ages. They have many roles such as providing wildlife habitat, storing carbon, and contributing to the overall fire hazard in a stand. Numerous wildlife species depend on standing dead trees for shelter, nesting sites, and food, including a variety of avian (Raphael and White 1984) and forest invertebrate species (Harmon and others 1986). Analysis of forest carbon pools have become an important component of national resource assessments. The United Nations 1992 Framework Convention on Climate Change (United Nations 1992) called for yearly reporting of the carbon mass stored as dead wood, of which standing dead trees are a considerable component, in forests among all signatory nations (EPA 2004).

When the total biomass of standing dead trees in a forest becomes excessive (Kirby and others 1998), the standing dead trees themselves may constitute a substantial fire hazard. Dead trees of different heights can potentially act as a fuel ladder to live tree crowns (Stephens 1998) and may help predict the amount of down woody debris through fuel succession models (Schimmel and Granstrom 1997). Overall, dead tree information has been used to assess a variety of forest stand attributes/processes such as growing stock mortality, wildlife habitat, wildfire hazards, or biomass/carbon.

There has been a dearth of information regarding standing dead wood resources across the United States. In the past, most standing dead tree analyses were at local/regional scales (Cline and others 1980, Goodburn and Lorimer 1998,

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Healy and others 1989, Ohmann and Waddell 2002), while national-scale forest resource analyses omitted dead tree attributes entirely (Smith and others 2004). The lack of published national standing dead tree resource estimates can be attributed to the absence of a nationally consistent standing dead tree inventory. Standing dead trees were not consistently inventoried in all States during FIA periodic forest inventories (prior to 2000). The extent of sampling ranged from a minimum necessary to determine rates of mortality to measurement of all standing dead trees sufficient for population level estimates (similar to current annual inventories). To address the growing need for consistent and timely standing dead tree resource information, the FIA program initiated annual inventories of standing dead trees at the onset of the 21st century. Woodall and others (2009) provided an initial examination of a partial inventory of the U.S. forests with data measured from 2000 to 2005. Since then, a more complete and vetted national standing dead wood database with new volume models (Heath and others 2009) has become available. Subsequently, estimates of standing dead tree attributes have been generated and national-scale analysis is now possible. The goal of this study was to summarize the current standing dead tree resource in forests of the United States with emphasis on the following attributes: size distribution, species composition, State-level biomass, and change estimates.

METHODS

The FIA program conducts a three-phase inventory of forest attributes in the United States (Bechtold and Patterson 2005). The FIA sampling design is based on a tessellation of the United States into hexagons that are approximately 2428-ha in size and have at least one permanent plot established inside each hexagon. In phase 1, the population of interest is stratified and plots are assigned to individual stratum, such as forest, nonforest, and forest-edge, to increase the precision of estimates. In phase 2, tree and site attributes are measured in forested conditions for field plots established in the 2428-ha hexagons. Phase 2 plots consist of four 7.32-m fixed-radius subplots or 17.95-m macroplots on which standing dead trees ≥ 12.7 cm d.b.h. are inventoried. Individual tree variables include species, d.b.h., and total height (Bechtold and Patterson 2005, USDA Forest Service 2006).

All standing dead tree estimates were based on empirically sampled forest inventory data from the most current, publicly available inventory, i.e., within the FIA database, for each State (excluding interior Alaska and Hawaii). Inventory data were from annual inventories collected between 1999 and 2008, except for Wyoming (2000) and New Mexico (1999), where periodic inventories were used. The number of FIA plots used in this study where at least one forested condition was observed totaled 87,401.

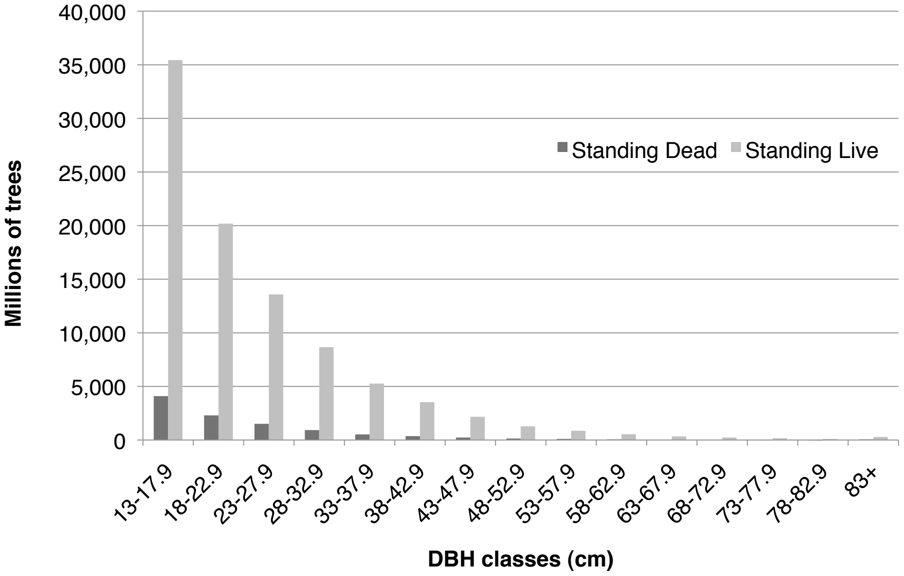

Figure 7.1—Diameter distribution for standing live and dead trees, United States (Note: Sampling error below 100 million trees for all estimates).

The total number of standing dead and live trees was estimated nationwide for 5 cm d.b.h. classes. Nationwide total aboveground dry biomass (Biomass; Mg) was determined for all tree species. Only the top 10 tree species with respect to this population estimate are reported in this analysis. The top and bottom 10 forest types in terms of mean standing dead tree biomass per unit forest land area (Mg/ha) was determined. For each State, the total standing dead tree biomass (Mg), biomass per unit forest land area (Mg/ha), and the ratio of standing dead and live biomass were determined. Finally, in order to estimate change for one contiguous region of the United States, where re-measured data were available, the change in the ratios of standing dead to standing live tree biomass between two points in time (time 1: 1999–2004, time 2: 2004–08) was examined for a selection of North Central States. General FIA population estimation procedures are detailed by Bechtold and Patterson (2005).

RESULTS AND DISCUSSION

There is 1 standing dead tree for every 10 standing live trees across the United States with a total of 10.6 billion standing dead trees nationwide (fig. 7.1). The diameter distribution is similarly shaped for standing live and dead trees. When tree counts were viewed as a percentage of the total distribution, standing live trees had only slightly higher percentages than standing dead trees for smaller diameter classes (d.b.h. < 53.0 cm) and vice versa for larger-sized trees (table 7.1). Because large-

Table 7.1—Diameter distribution (diameter class count divided by total) for standing live and dead trees and factor (difference between standing live and standing dead trees population counts), United States

D.b.h.[a] (cm)	Live trees (percent)	Dead trees (percent)	Factor
13-17.9	38.3	38.3	8.65
18-22.9	21.8	21.5	8.78
23-27.9	14.7	14.1	8.98
28-32.9	9.3	8.7	9.34
33-37.9	5.7	4.9	10.01
38-42.9	3.8	3.4	9.69
43-47.9	2.3	2.1	9.46
48-52.9	1.4	1.3	9.14
53-57.9	0.9	1.0	8.24
58-62.9	0.6	0.6	8.01
63-67.9	0.4	0.4	7.27
68-72.9	0.3	0.3	6.68
73-77.9	0.2	0.3	5.89
78-82.9	0.1	0.2	5.66
83+	0.3	0.6	4.19

Note: Inventory data were from annual inventories collected between 1999 and 2008, except for Wyoming (2000) and New Mexico (1999), where periodic inventories were used.

[a] d.b.h. = diameter at breast height.

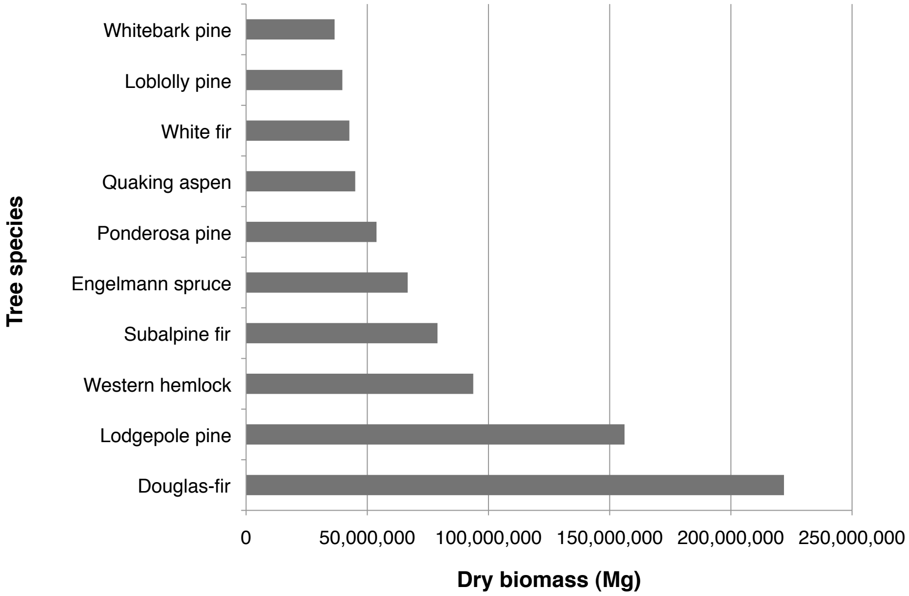

Figure 7.2—Top 10 tree species in terms of total aboveground dry biomass (Mg), United States.

sized standing dead trees are resident in forest ecosystems for longer time periods than rapidly decaying smaller-sized trees, the relative frequency of tree counts is greater in large-sized standing dead trees when compared to standing live trees. Whereas live trees outnumber standing dead trees by a factor of almost 9 to 1 for the smallest trees (d.b.h. 13.0–17.9 cm), this factor was only approximately 4 to 1 for the largest trees (d.b.h. > 83.0 cm) (table 7.1).

The species composition of standing dead trees is quite diverse with more than 130 species having over 1 million Mg each nationwide, but is dominated by tree species of the Western United States, e.g., Douglas-fir, > 200 million Mg. In terms of total standing dead tree biomass across the United States, western tree species account for nine of the top 10 species with loblolly pine the sole exception (fig.7.2). The combined total nationwide standing dead tree biomass of Douglas-fir and lodgepole pine across the United States is almost equal to the next six species combined. Although western forests possess vast acreage of highly productive monocultures, e.g., Douglas-fir, these estimates may point to a potential forest health and fire hazard challenge facing forest ecosystems in the Western United States. On a per-unit-area basis, western forest types dominate all the top 10 forest types across the United States in terms of biomass per unit forest land area (fig. 7.3A). Western hemlock forest types have over 30 Mg/ha, while the remaining top nine forest types have biomass averaging over 15 Mg/ha. The bottom 10 forest types in terms of mean standing dead biomass per unit forest land area, are represented by woodland species, eastern hardwoods, and southern yellow pines (fig. 7.3B). Woodland tree species are often multi-stemmed and may not attain the same large-diameter sizes of forest land tree species. Because the FIA program only inventories standing dead trees with a d.b.h. in excess of 12.7 cm, many of the smaller-sized woodland tree species stems might be excluded from this analysis where woodland species do not attain large diameters. One reason the southern yellow pine forests have low levels of dead tree biomass may be due to active land managers intentionally capturing anticipated tree mortality through commercial thinning treatments, thereby incurring relatively low levels of tree mortality.

Mirroring the species- and forest type-specific standing dead tree results, western States have a tremendous amount of standing dead tree biomass (fig. 7.4A). The northwestern States of Washington, Oregon, and Idaho have the highest amounts of standing dead tree biomass per hectare in the country, typically in excess of 4 Mg/ha (fig. 7.4B). When ratios of standing dead to standing live biomass are examined, almost all Rocky Mountain States have the highest ratios for the United States (fig. 7.5), possibly due to long-term drought and insect/disease effects (van Mantgem and others 2009).

Due to gradual implementation and remeasurement of standing dead tree plots across the United States, only North Central States were examined for changes in standing dead to live tree biomass ratios (fig. 7.6). Almost all examined States had higher ratios of dead to live tree

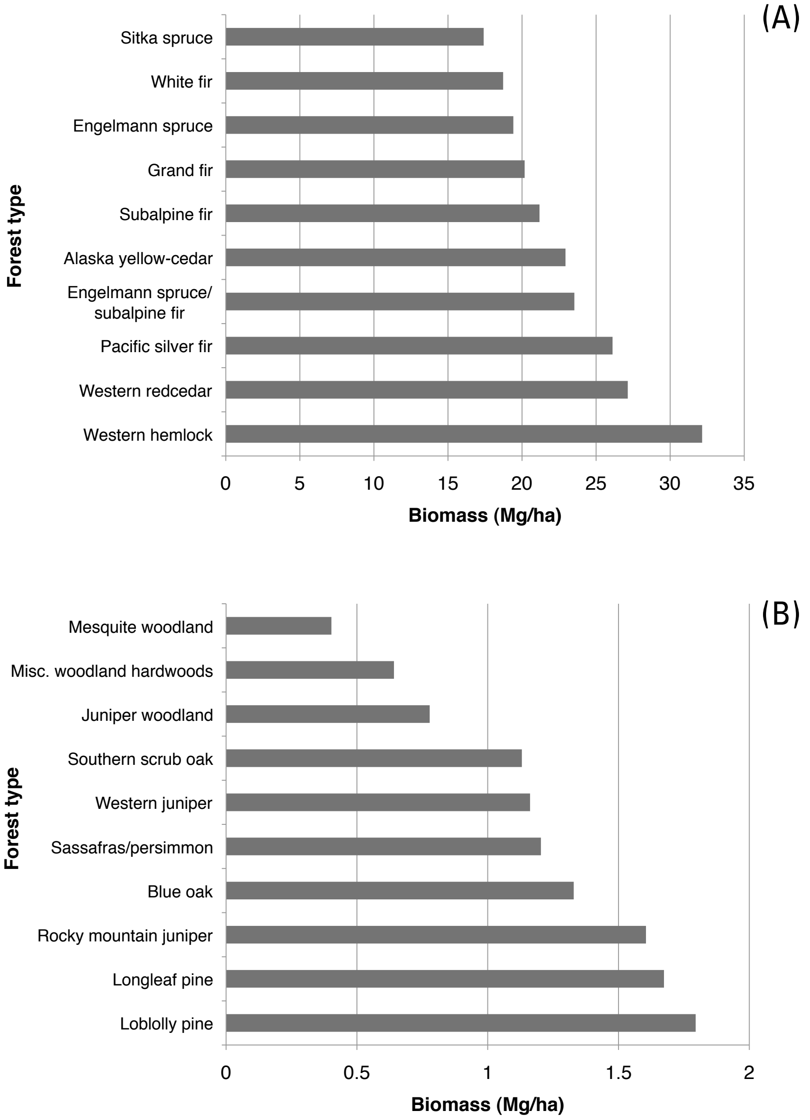

Figure 7.3—Estimates of standing dead tree biomass per unit of forest land area (Mg/ha) for the (A) top 10 and (B) bottom 10 forest types, United States.

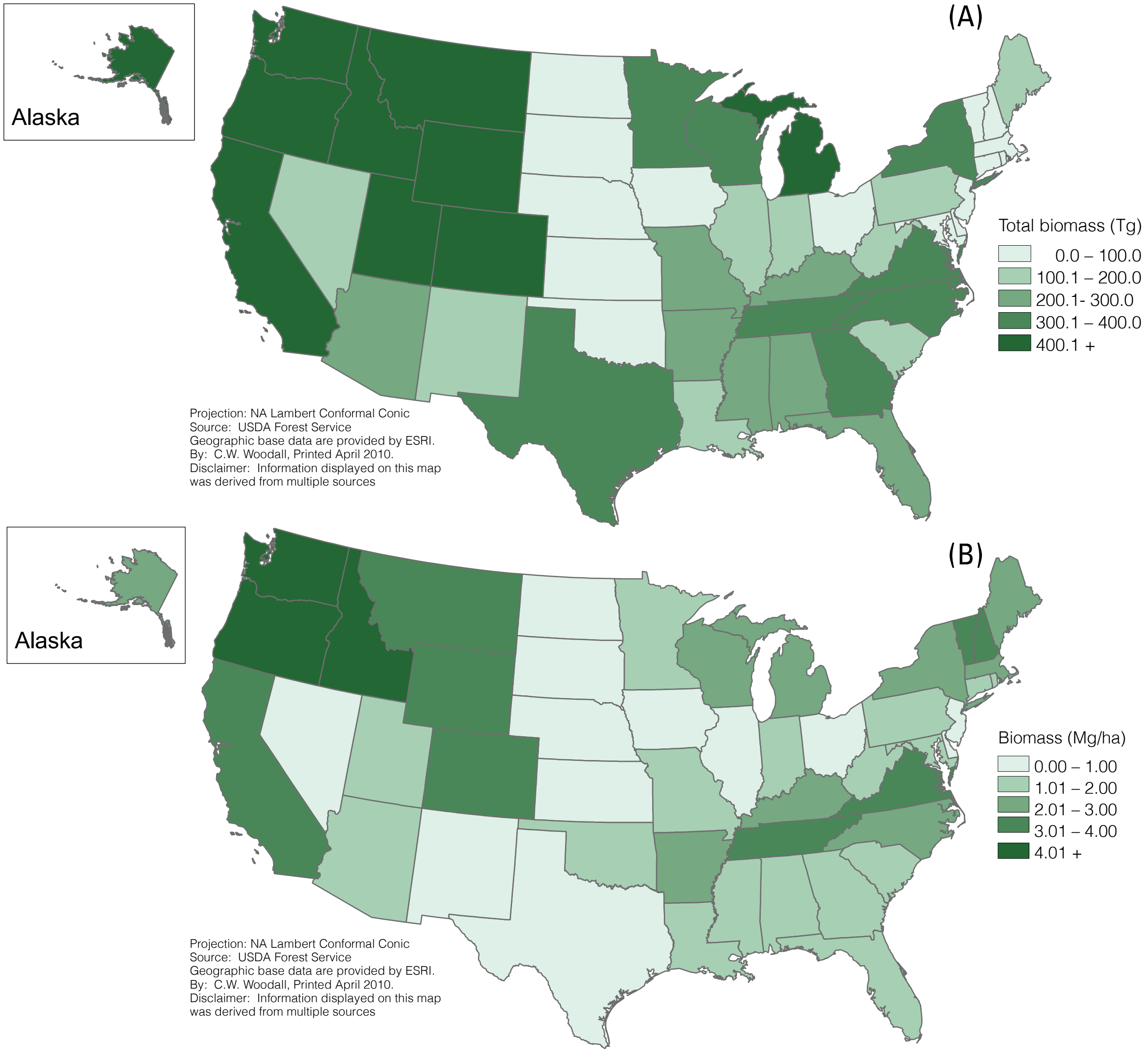

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Figure 7.4—Estimates of (A) total biomass (Tg) and (B) biomass per unit forest land area (Mg/ha) of standing dead trees in States (excluding Hawaii and interior Alaska). Note: Alaska is not shown to scale with map of the conterminous United States.

biomass at remeasurement, although differences were within associated sampling errors indicating a lack of statistical difference across this relatively short remeasurement period. Changes in this ratio may serve as an indicator of advanced stages of stand development, e.g., stem-exclusion, or forest health concerns, e.g., such mortality events as pest outbreaks.

Overall, standing dead trees are a sizeable component of forest ecosystems across the United States, but still pale in comparison to live tree resources (in terms of both frequency and biomass). The highly productive West Coast forests retain the greatest amount of biomass in standing dead trees compared to other States across the Nation. In particular, Rocky Mountain States have a high ratio of standing dead to standing live biomass of 0.12 or larger, while most other States in the Nation have ratios below 0.09. Standing dead trees play divergent roles in forest ecosystems of the United States. While they may serve as an indicator of imminent forest health threats such as wildfire, they also serve as critical habitat, a potential bioenergy source, and an important element of the carbon storage capacity of forests across the country. The continued monitoring of this forest resource will ensure the varied roles of standing dead trees are elucidated during forest management and policy decisions.

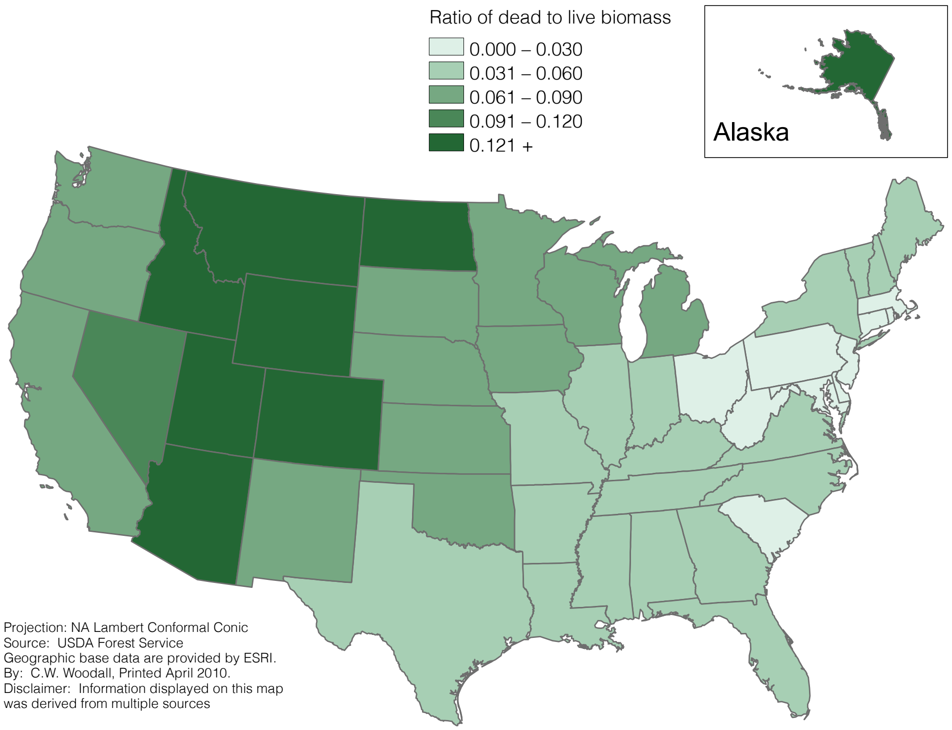

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Figure 7.5—Ratio of standing dead tree biomass to standing live tree biomass for all States (excluding Hawaii and interior Alaska). Note: Alaska is not shown to scale with map of the conterminous United States.

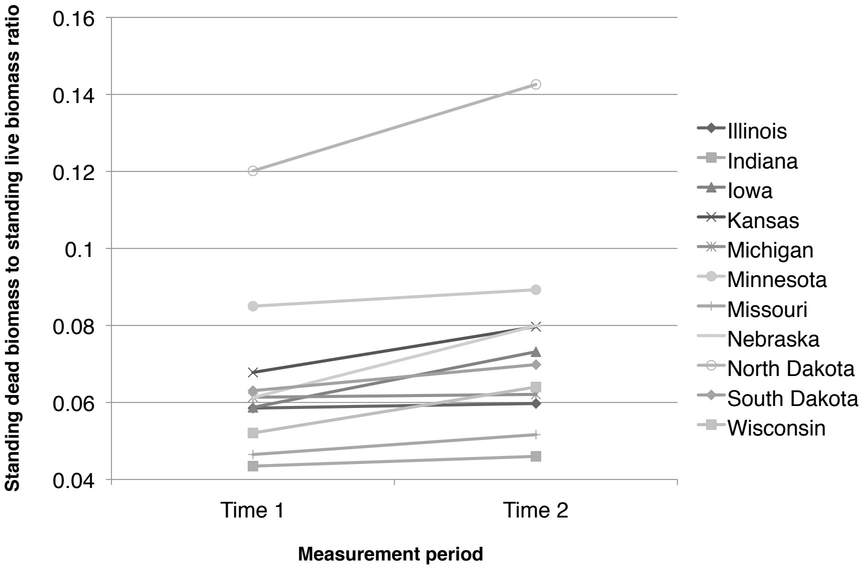

Figure 7.6—Changes in ratio of standing dead tree biomass to standing live tree biomass for North Central States between two measurement periods (1999–2004 to 2004–2008).

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CHAPTER 8.

Climate Change Impacts on Forest Soil Critical Acid Loads and Exceedances at a National Scale

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INTRODUCTION

Federal agencies are currently developing guidelines for forest soil critical acid loads across the United States. A critical acid load is defined as the amount of acid deposition (usually expressed on an annual basis) that an ecosystem can absorb. Traditionally, an ecosystem is considered to be at risk for health impairment when the critical acid load exceeds a level known to impair forest health. The excess over the critical acid load is termed the *exceedance*, and the larger the exceedance, the greater the risk of ecosystem damage. This definition of critical acid load applies to a single, long-term pollutant exposure. These guidelines are often used to establish regulations designed to maintain acidic deposition, e.g., nitrogen and sulfur, inputs below the level shown to exceed an ecosystem's critical acid load. The traditional definition for a critical acid load generally assumes that the ecosystem is in a steady state condition, i.e., no major changes in the factors that regulate the ecosystem's ability to absorb acids. Unfortunately, climate change is altering weather patterns and, thus, impacting the factors that regulate critical acid load limits. This chapter explores which factors associated with establishing forest soil critical acid load limits will most likely be influenced by climate change, and how these changes might impact forest soil critical acid load limits across the United States. In New England, for example, base cation weathering could increase with global warming, along with nitrogen uptake as a function of increased forest growth. Nationally, a moderate 20-percent increase in base cation weathering and nitrogen uptake would result in at least a 30.5-percent decrease in the amount of forest soil area that exceeded the critical acid load limit and at least a 64.4-percent decrease in the amount of high exceedance area. While these results are encouraging, they do not account for other negative potential forest health risks associated with climate change such as elevated fire, insect, or disease risk. Additional study is needed before the full impact of climate change on forest health can be assessed.

Airborne nitrogen (N) and sulfur (S) from industrial pollution and automobile exhausts have been deposited across Europe and the Northeastern United States for over 70 years in the form of acid rain. Heavily polluted areas can receive over 50 kg N ha^{-1} each year (Holland and others 2005). The environmental impacts of air pollutants have been studied since N and S were first suspected to cause forest damage and decline across the region in the mid-1980s. High pollutant levels and forest mortality can lead to mobilization of soil aluminum (Al^{3+}) and nitrate (NO_3^-) (Berg 1986, Cronan and Schofield 1979, Johnson and others 1994) and subsequent increases in stream Al and NO_3 concentrations. Increased Al and NO_3 stream concentrations can have negative health impacts on fish populations and human water supplies (Baker and others 1996). High forest soil acidity can also cause aluminum toxicity in roots (Shortle and Smith

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1988), foliar nutrient imbalances (Cronan and Grigal 1995, Zoettl and Huettl 1986), reduced tree cold tolerance (Sheppard 1994), and tree freezing injury (DeHayes and others 1992). Each of these stressors can lead to tree mortality (Aber and others 1989, McNulty and others 2007).

Traditionally, an ecosystem is considered to be at risk for health impairment when its critical acid load from deposition (and rarely from rocks) exceeds the amount of acid that the ecosystem can absorb. Often, 200 eq [equivalents or moles of charge (Aherne 2008)] ha^{-1} yr^{-1} is used as a standard known to impair forest health. Acidic deposition in excess of the critical acid load is termed the acidic exceedance, and the larger the acidic exceedance, the greater the risk of ecosystem damage. This definition of critical acid load applies to a single, long-term pollutant exposure. However, a static critical acid load level may not accurately assess ecosystem risk to damage when an ecosystem is subjected to multiple, episodic environmental stresses. If multiple stress impacts (drought, insects, wildfire) are included in critical acid load assessments, the critical acid load may need to be lowered in many areas to maintain long-term ecosystem health.

Various methods have been developed to assess the ecosystem soil's critical acid load. One of the most common methods for determining an ecosystem's critical acid load is the use of a simple mass balance equation (SMBE) that uses static soil, climate, vegetation, and pollutant deposition data to estimate a soil's critical acid load. Previous SMBEs have been used to estimate forest soil critical acid loads and exceedances at a 1-km^2 resolution across the conterminous United States (McNulty and others 2007). In this chapter, an SMBE is modified to examine how climate change could alter an ecosystem's critical acid load and potential for acid soil exceedance.

Climate Change Impacts on Critical Acid Loads

Climate change is a generic term used to define a host of changing environmental conditions associated with the atmospheric increase of greenhouse gases and global warming. Climate change is characterized by both climatic shifts and increased climate variability. Both inter-decadal shifts in climate and inter-annual climate variability can influence the critical acid load of forest soils. Each of these impacts is examined below.

Short- and Long-Term Droughts

Water is one of the principle determinants of ecosystem type. Average annual precipitation in temperate forests ranges from 50 to 250 cm per year. Deserts, scrubland, and woodlands receive between 0 and 125 cm of precipitation per year (Whittaker 1970). Millennia of plant competition have favored vegetative species that best adapt to limited resources (including water). Short-term, i.e., < 2 years, drought can reduce ecosystem productivity (Hanson and Weltzin 2000), leaf longevity in deciduous species (Jonasson and others 1997), and leaf area (Gholz and others 1990). These factors reduce biological demand for nitrogen which can alter soil critical acid loads. Under short-term, i.e., < 2 years, extreme drought, reduced soil moisture can cause reduced nitrogen mineralization and nitrification that then result in reduced ammonium and nitrate availability. These conditions would not impact critical acid load in the short term if both nitrogen demand and supply are reduced. However, nitrogen will continue to accumulate in the ecosystem during the drought. A nitrate pulse could occur following a drought if nitrogen mineralization and nitrification rates respond to available water before plant demand for nitrogen increases.

Long-term, i.e., > 2 years, droughts can cause additional ecosystem disruptions and therefore have the potential to significantly lower forest soil critical acid load levels. Long-term droughts have all of the characteristics of short-term drought plus the potential for tree mortality due to water stress (Kloeppel and others 2003), increased insect outbreak potential (Mattson and Haack 1987), and increased fire risk (Flannigan and Wotton 2001). As with short-term drought, long-term drought may reduce biological nitrogen demand and supply. Additionally, the potential for terrestrial vegetation mortality could lead to a significant decrease in biological nitrogen uptake. If tree mortality is severe, a large nitrate pulse could occur following the drought, similar to the nitrate pulse observed following forest harvesting (Vitousek and Matson 1985).

The forest soil critical acid load may be significantly reduced for several years after drought-induced forest mortality, because new growth cannot fully utilize existing water, light, and nutrients. For example, around Mt. Mitchell in the Southern Appalachian Mountains a combination of drought, increased air temperature, and insects likely caused the mortality of mature high elevation red spruce (*Picea rubens*) trees in 2001 (McNulty and Boggs 2010). The forest soil critical acid load for this area was reduced until new growth could fill in gap openings and increase biological nitrogen uptake.

Climate Change Shifts in Precipitation

Both short- and long-term droughts are transient weather events. However, climate change is a permanent shift in the amount and timing of precipitation for a region. Changes in tree species distributions, nutrient cycling, and water flow are all likely with climatic shifts. Reductions in precipitation would cause a shift toward more open, drought-tolerant woodlands (Hansen and others 2001). As tree density decreases, nitrogen demand and uptake by vegetation decreases. Therefore, a forest soil critical acid load for an ecosystem receiving less precipitation could decrease. Conversely, the forest soil critical acid load could increase if climate change results in an increase in precipitation, along with a shift toward more dense forests with higher nitrogen demands.

Permanent precipitation change-induced forest species shifts can also change the nitrogen cycle. Mesic tree species tend to be more nitrogen demanding (Watmough and others 2004). Therefore, increased precipitation could gradually shift a forest toward a higher forest soil critical acid load, while reductions in precipitation could have a negative impact on nitrogen uptake and soil critical acid loads.

Climate Change Shifts in Air Temperature

During the next century, substantial changes are expected to occur in a variety of environmental variables including temperature. The magnitude of these changes is expected to vary temporally and spatially. The Intergovernmental Panel on Climate Change (IPCC) concluded that average global surface temperature is projected to increase by 1.8 C° to 3.6 C° above 2000 levels by 2100 (IPCC 2007).

Biological processes accelerate as air temperature increases. Increases in tree respiration and metabolism can shorten leaf retention time as temperature increases. Litter decomposition, soil nitrogen mineralization, and soil nitrification also increase with increasing temperature. Increases in both nitrogen demand and supply can offset each other, so the forest soil critical acid load may not change. In cooler regions, increases in air temperature may increase forest productivity and therefore nitrogen uptake. However, in warmer climates air temperature may be at (or above) the optimal levels for forest growth. Additional warming would decrease tree growth and reduce nitrogen uptake. If tree nitrogen demand does not keep pace with nitrogen availability, then the forest soil critical acid load could decrease with increasing air temperature.

METHODS

Assessing Climate Change Impact on Critical Acid Loads Using a Simple Mass Balance Equation

Climate change can impact many aspects of ecosystem function, e.g., insect outbreaks, wildfire occurrence, and susceptibility. While some of these potential impacts cannot be represented in an SMBE, some important aspects of a critical acid load that may be impacted can be assessed using the SMBE approach. The complete methodology and databases used to produce the assessment of historic acidic deposition impacts on forest soil critical acid loading are available in McNulty and others (2007). Several databases (table 8.1) were used to run the SMBE model, and all operations occurred in a geographic information system (GIS). The soil database was used as the base layer for the GIS operations, and it had a spatial resolution of 1 km². All other databases used in the SMBE model were aligned to the soil database and rescaled to 1 km². Critical acid loads are calculated using the following SMBE:

$$CAL(S+N) = BC_{dep} - Cl_{dep} + BC_{w} - BC_{u} + N_{i} + N_{u} + N_{de} - ANC_{le,crit} \qquad (1)$$

where

$CAL(S+N)$ = the forest soil critical acid load for S and N

BC_{dep} = base cation [i.e., calcium (Ca) + potassium (K) + magnesium (Mg) + sodium (Na)] deposition

Table 8.1—Descriptions of input datasets used to run the simple mass balance equation (SMBE) model. All operations occurred in a geographic information system

Data set	Source	Temporal scale	Original display scale	Original spatial scale
Dry deposition	U.S. EPA (2007)	1994-2000	NA	NA
Forest type	USDA Forest Service (unpublished)	2002-2003	< 1:2,000,000	250-m
Runoff	Gebert and others (1987)	1951-1980	1:7,500,000	1-m
Soils	State Soil Geographic (STATSGO) database (USDA NRCS 1995)	NA	1:250,000	6.25-km
Wet deposition	Grimm and Lynch (2004)	1994-2000	NA	330-m or 1-km
	National Atmospheric Deposition Program (NADP) annual isopleths maps (NADP 2005)	1994-2000	NA	2.5-km
Wilderness area	National Atlas (National Atlas of the United States 2005)	NA	1:2,000,000	NA

NA = Not applicable.

Cl_{dep} = chloride deposition

BC_w = base cation weathering

BC_u = uptake of base cations (i.e., Ca + K + Mg) in trees

N_i = nitrogen immobilization

N_u = uptake of nitrogen in trees

N_{de} = denitrification

$ANC_{le,crit}$ = forest soil acid neutralizing capacity of CAL leaching (Gregor and others 2004).

Each parameter in the SMBE was represented by a GIS layer. Critical acid loads for the conterminous United States are shown in figure 8.1.

Critical acid load exceedance (eq ha^{-1} yr^{-1}) was calculated using the following equation:

$$Ex(S{+}N_{dep}) = S{+}N_{dep} - CAL(S{+}N) \tag{2}$$

where

Ex = exceedance of the forest soil critical N and S load

$S{+}N_{dep}$ = the deposition of S plus N

$CAL(S{+}N)$ = the forest soil critical load of S plus N (Gregor and others 2004).

Higher *Ex* values reflect greater exceedance of acidic deposition above the level associated with an increased likelihood of environmental harm (fig. 8.2).

Base cation weathering has the largest influence on SMBE estimates of a critical acid load (Li and McNulty 2007). The most recent IPCC report suggests that a global warming may increase the earth's surface temperature by almost 4 C° by the end of the 21st century (IPCC 2007). For this examination of critical acid load sensitivity to changes in climate, we re-ran the equations that estimated base cation weathering by adding 4 C° to the average annual air temperature values across the conterminous United States.

As previously discussed, forest productivity may increase (cooler regions) or decrease (warmer regions) with a 4 C° increase in air temperature. To simulate this variability, we developed another scenario in which forest productivity (and therefore nitrogen uptake) was increased and decreased by 20 percent from historic levels. The combinations of increased base cation weathering and nitrogen uptake and decreased nitrogen uptake were entered into the SMBE to examine climate change impacts on critical acid loading and exceedances across the conterminous United States at a 1-km^2 scale.

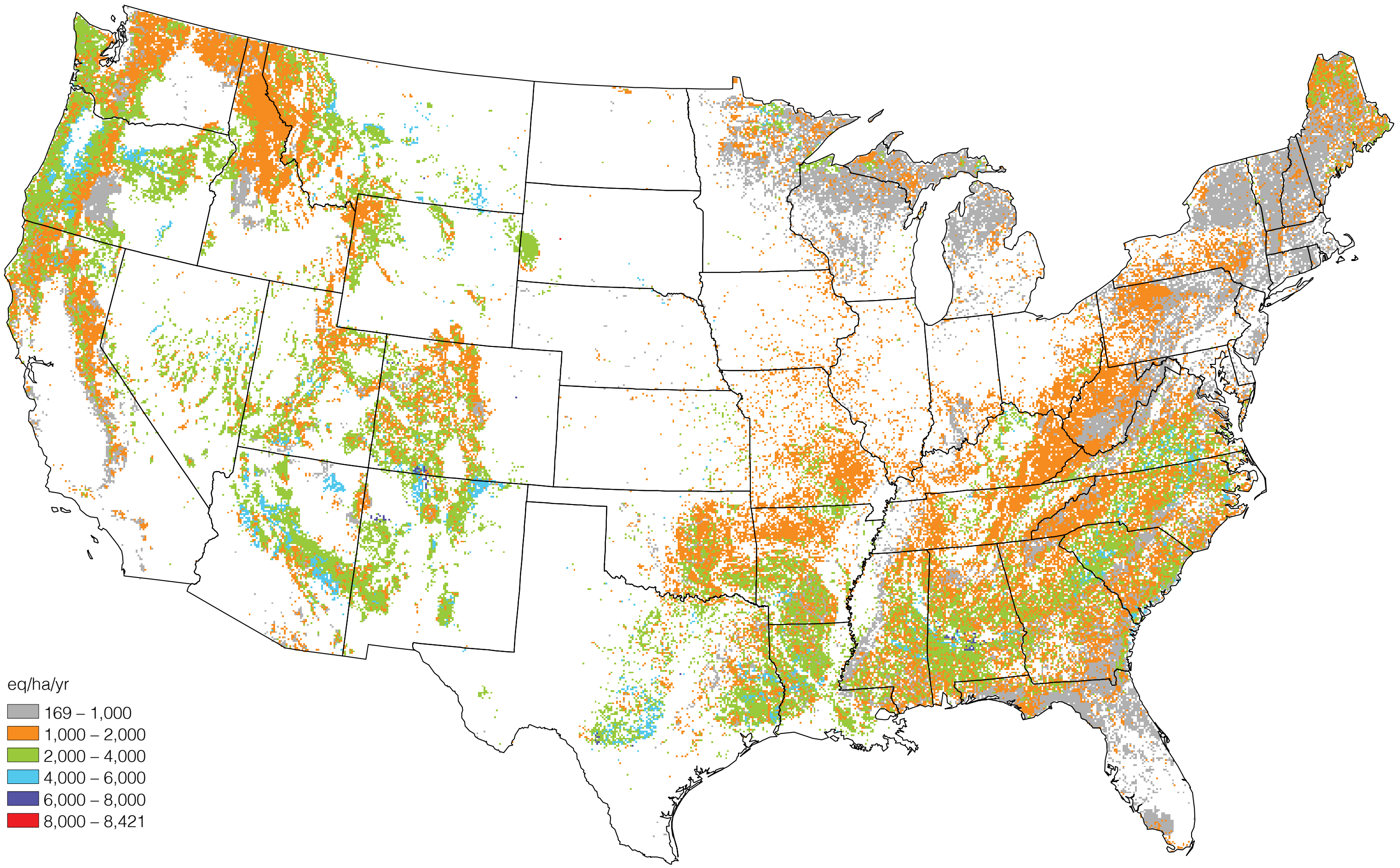

Figure 8.1—Critical loads of acidity in forest soils, baseline. (Data sources: EPA 2007, Gebert and others 1987, Grimm and Lynch 2004, National Atlas of the United States 2005, NADP 2005, and USDA NRCS 1995)

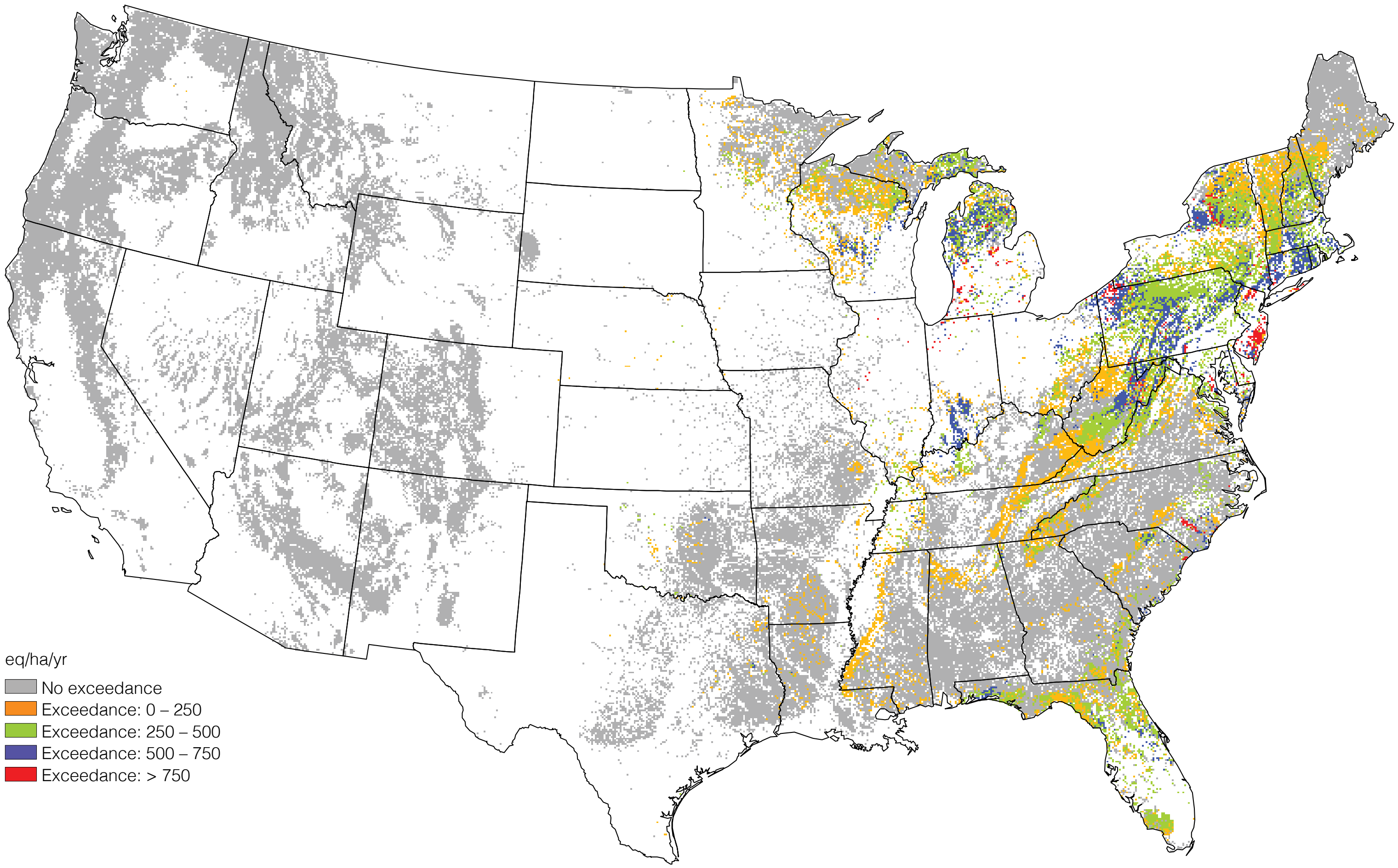

Figure 8.2—Exceedance of critical loads of acidity in forest soils, baseline. (Data sources: EPA 2007, Gebert and others 1987, Grimm and Lynch 2004, National Atlas of the United States 2005, NADP 2005, and USDA NRCS 1995)

RESULTS

A 4 C° increase in air temperature resulted in a 20-percent increase in base cation weathering. As expected based on the SMBE sensitivity analysis (Li and McNulty 2007), a 20-percent increase in base cation weathering and nitrogen uptake had a significant impact on both the forest soil critical acid loads level (fig. 8.3) and the amount of exceedance across the conterminous United States (fig. 8.4). The total percentage of forest area that was in exceedance of the critical acid load decreased from 22 percent in the baseline scenario to 16.6 percent in the increased base cation weathering scenario. This represents a 24.5-percent decrease in forest area impacted by acidic deposition. More significantly, the forest areas that were most impacted (≥ 500 eq acid l^{-1}) experienced an even larger reduction in impacted area, dropping from 4.5 percent to 2.0 percent of the total forest area (a 55.6-percent reduction) under the baseline and climate change scenarios, respectively (tables 8.2 and 8.3).

Conversely, increases and decreases in nitrogen uptake associated with changes in forest growth had a relatively small impact on the total percentage of forest area with soils that exceeded their critical load limits. A 20-percent increase in N uptake reduced the total area in exceedance of the forest soil critical acid load by 0.5 percent (22.0 percent to 21.9 percent of total area) and had no impact on the most impacted forest area. In comparison, 20-percent and 40-percent decreases in nitrogen uptake that would be associated with reductions in forest growth increased the amount of total and most impacted forest area with soils in exceedance of the critical acid load (tables 8.2 and 8.3).

The combination of increased nitrogen uptake and increased base cation weathering did not significantly reduce the amount of forest area in exceedance of the critical acid load beyond the reduction due to increased base cation weathering alone. When compared to the baseline, the increased weathering and 20-percent decreased nitrogen uptake scenario reduced the total forest area impacted by acid loading by 24.1 percent and reduced highly impacted forest area by 53.3 percent. Without the addition of increased nitrogen uptake, the increased weathering scenario alone reduced the total forest area impacted by 24.5 percent and reduced highly impacted forest area by 55.6 percent (table 8.3).

Although reductions in forest growth and nitrogen uptake are likely under a changing climate in some areas of the United States, these areas are more likely to be limited to the warmer, i.e., southern, regions of the country. The Northern United States could experience increased forest growth and nitrogen uptake associated with longer growing seasons. The majority of forest soil areas currently in exceedance of their critical acid load are located in the Northern United States (fig. 8.2). Therefore it is unlikely that increased weathering and reduced N uptake (associated with reductions in productivity) will be occurring in areas where most of the exceedances occur, i.e., Northeastern United States.

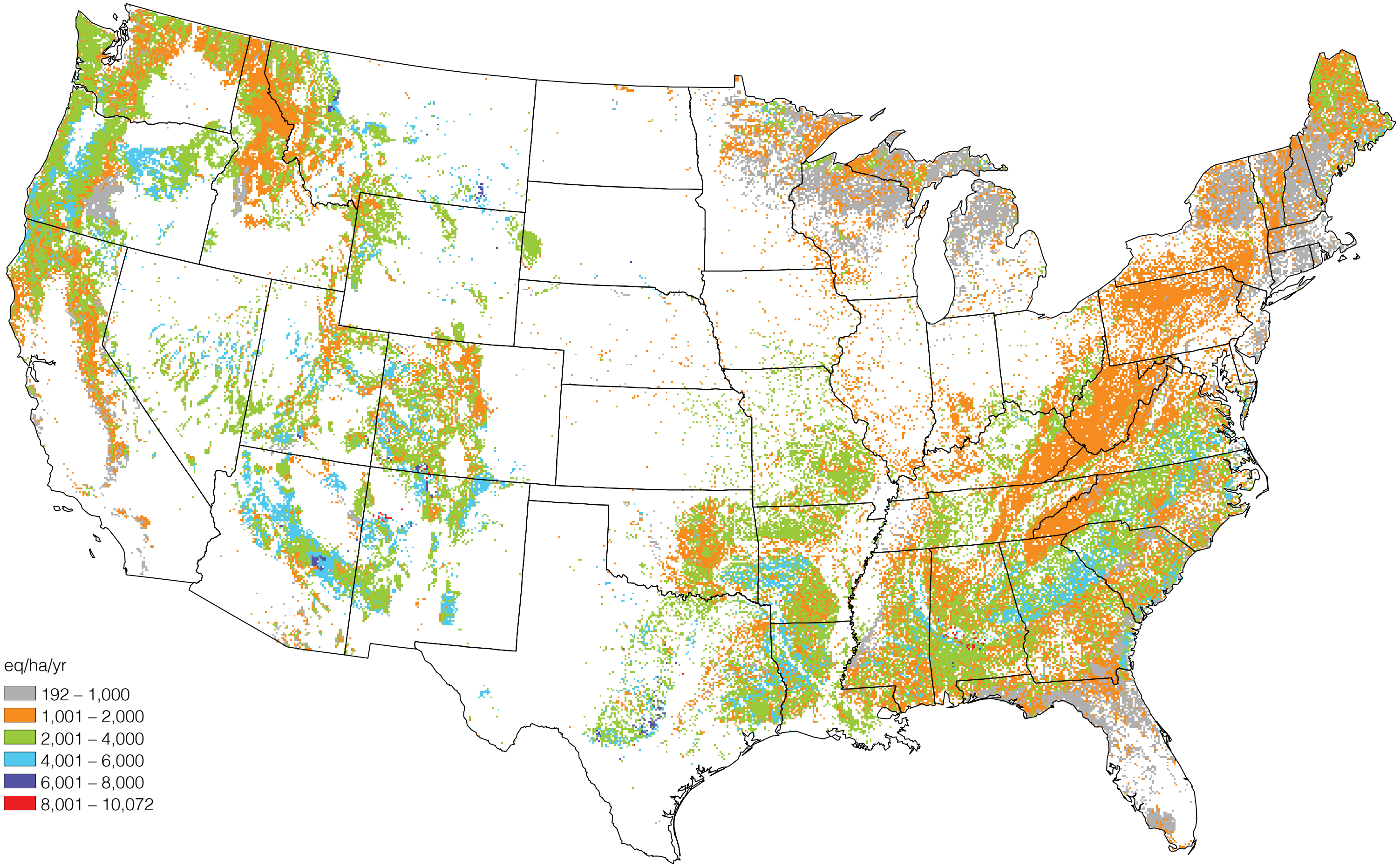

Figure 8.3—Critical loads of acidity in forest soils, base cation weathering 20-percent increase and nitrogen uptake 20-percent increase. (Data sources: EPA 2007, Gebert and others 1987, Grimm and Lynch 2004, National Atlas of the United States 2005, NADP 2005, and USDA NRCS 1995)

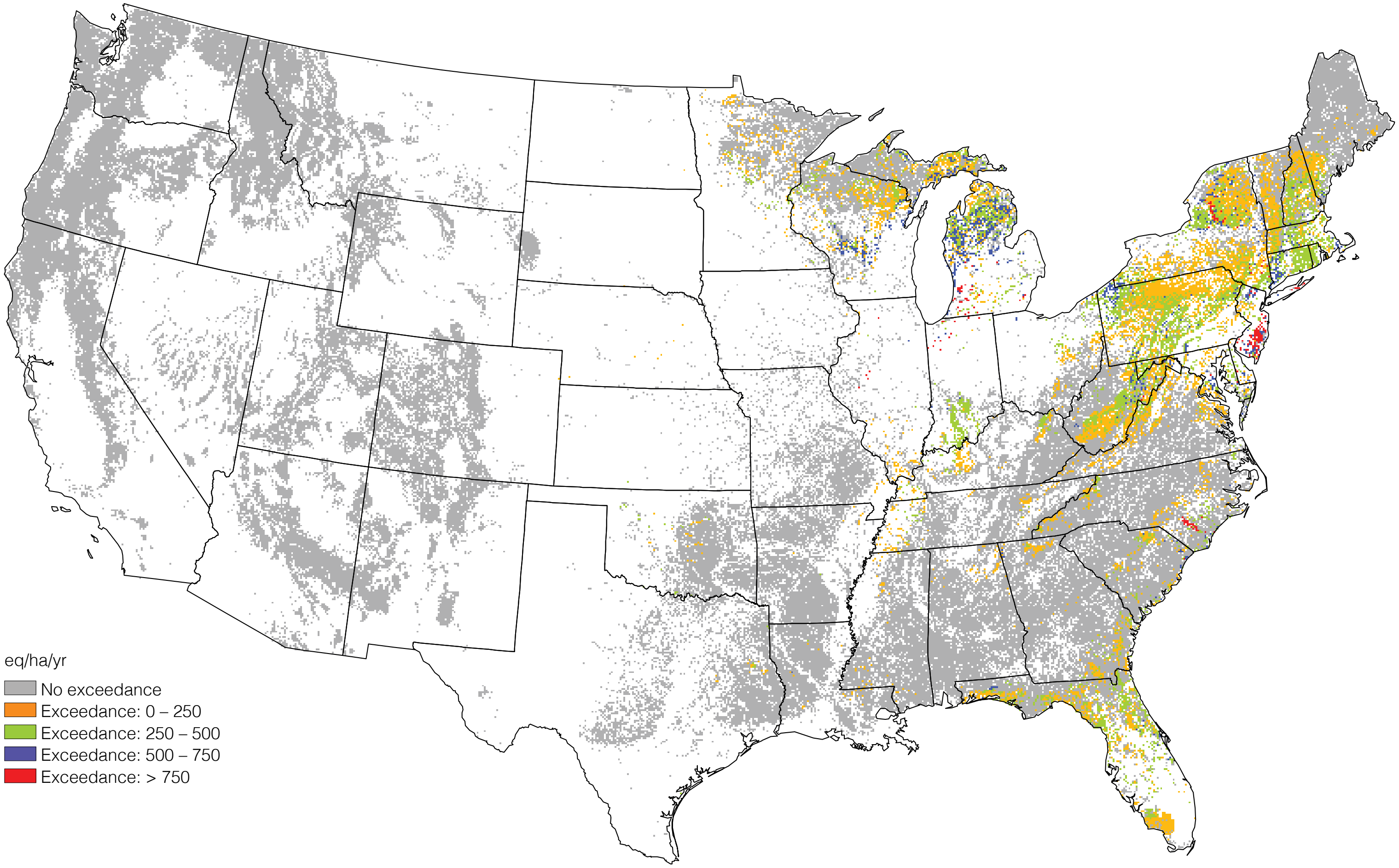

Figure 8.4—Exceedance of critical loads of acidity in forest soils, base cation weathering 20-percent increase and nitrogen uptake 20-percent increase. (Data sources: EPA 2007, Gebert and others 1987, Grimm and Lynch 2004, National Atlas of the United States 2005, NADP 2005, and USDA NRCS 1995)

Table 8.2—Percent of entire conterminous United States forest area in exceedance, and percent of United States forest area in exceedance of 500 eq ha^{-1} yr^{-1}

Scenario	Baseline	BCW20%i	N20%i	N20%d	N40%d	BCW20%i N20%i	BCW20%i N20%d	BCW20%i N40%d
Percent of forested area impacted by acidic deposition	22.0	16.6	21.9	23.2	24.4	15.3	16.7	18.2
Percent of forest area in exceedance ≥ 500 eq/ha/yr	4.5	2.0	4.5	5.8	7.1	1.6	2.1	2.9

BCW = base cation weathering; N = nitrogen uptake; d = decrease; i = increase.

Table 8.3—Percent change of conterminous United States forest area in exceedance and percent change of United States forest area in exceedance of 500 eq ha^{-1} yr^{-1} when comparing simple mass balance equation (SMBE) scenarios to baseline conditions

Scenario	BCW20%i	Na20%i	N20%d	N40%d	BCW20%i N20%i	BCW20%i N20%d	BCW20%i N40%d
Percent change, forest area impacted by acidic deposition	-24.5	-0.5	5.5	10.9	-30.5	-24.1	-17.3
Percent change, forest area in exceedance ≥ 500 eq/ha/yr	-55.6	0.0	28.9	57.8%	-64.4%	-53.3	-25.6
Forest area impacted	Reduction	Reduction	Increase	Increase	Reduction	Reduction	Reduction

BCW = base cation weathering; N = nitrogen uptake; d = decrease; i = increase.

CONCLUSIONS

Climate change is expected to negatively impact forest ecosystems in several ways during this century. However, the exacerbation of forest soils affected by acidic deposition may not be among those negative impacts. The results of this SMBE suggest that those soils most likely to be in exceedance of their critical acid load levels, e.g., in New England forests, may also be the most likely to benefit from the warming associated with climate change. Increases in both nitrogen uptake and base cation weathering are predicted in this region of the country.

The increase in weathering rates would be particularly important for increasing the region's critical acid load capacity. This study suggests that a major reduction in the area representing the most seriously impacted forest soils could occur. While this is potentially good news for ecosystem management, other aspects of climate change are not accounted for by the simple mass balance equation approach to critical acid load determination. Potential increases in insect outbreaks, hurricane intensity, wildfires, and changing patterns of ice storms could all reduce forest productivity and therefore nitrogen uptake. These conditions would reduce the buffering capacity of the ecosystem and make it more likely that the ecosystem would be in exceedance of its critical acid load, so caution should be used when interpreting these results.

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SECTION 3.

Evaluation Monitoring Project Summaries

Each year the Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture, funds a variety of Evaluation Monitoring (EM) projects, which are "designed to determine the extent, severity, and causes of undesirable changes in forest health identified through Detection Monitoring and other means" (FHM 2009). In addition, EM projects can produce information about forest health improvements. Evaluation Monitoring projects are submitted, reviewed, and selected in two main divisions: 1) base EM projects and 2) fire plan EM projects. More detailed information about how EM projects are selected, the most recent call letter, lists of EM projects awarded by year, and EM project poster presentations can all be found on the FHM Web site: www.fs.fed.us/foresthealth/fhm.

Beginning in 2008, each annual FHM national report contains summaries of recently completed EM projects. Each summary provides an overview of the project and results, citations for products and other relevant information, and a contact for questions or further information. The summaries provide an introduction to the kinds of monitoring projects supported by the FHM program and include enough information for readers to pursue specific interests. Nine project summaries are included in this report.

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CHAPTER 9.

Changes in Fragmentation of Western Washington Forest Land (Project WC-EM-08-01)

Andrew Gray[1]

INTRODUCTION

Land conversion and forest fragmentation is an important issue across the country. Forest fragmentation is one of the international Montréal criteria and indicators that the Forest Service, U.S. Department of Agriculture, must assess periodically (Riitters and others 2004). Characteristics of increased fragmentation include increases in the amount of forest-nonforest edge, and decreases in the size of forest patches. Edges are associated with greater drying of forest fuels and the spread of weedy species (both plants and animals). As forest patches become smaller and distances between them increase, the number of native species that can survive in them declines. Smaller forest parcels are also less likely to be managed for timber production (Kline and others 2004, Wear and others 1999).

Forest monitoring—by the Forest Inventory and Analysis (FIA) Program of the Forest Service—in western Washington (defined as the 19 counties west of the crest of the Cascade Mountains) has detected rapid conversion of forest land to nonforest land uses in the previous decade, with 5 percent of private timberland (270,000 acres) being converted between 1990 and 2000, following a similar rate of loss in the previous decade (Gray and others 2005). Human population increase in the State is no doubt a contributing factor; population increased from 3.4 million to 6.5 million people between 1970 and 2007 (96 percent), with most of the increase occurring in the western portion of the State. Not surprisingly, homes are often associated with forest fragmentation in Washington. Homes mixed in forest land results in increased cost of fighting wildfire as well as increases in the economic damage caused by fire. The State legislature has prioritized the acquisition of valuable timberlands in danger of conversion, and researchers are developing strategies for identifying and retaining those forest lands.

METHODS

Although some aspects of fragmentation focusing on tree cover are readily assessed with satellite imagery (Riitters and others 2004), a prior study in western Washington indicated that classifying land use and land use change with computer algorithms and low-resolution satellite imagery was not very accurate (Kline and others 2009). This study adopted an approach based on manual image interpretation that has been successfully conducted in Oregon, with the results being built into State sustainable management indicators (Lettman and others 2009). The imagery available for this study was digital photography from 2006 and the early 1990s, and hardcopy photos from 1976 used by FIA for double-sample stratification. Photos from 1976 were scanned and registered with the later imagery, with all analyses conducted in GIS.

Images were classified into the following land-use zones, with polygons mapped across all of western Washington:

Wildland Forest: Polygon ≥ 640 acres, < 5 structures per 640 acres, > 80 percent of area is forest land.

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Intensive Agriculture: Polygon ≥ 640 acres, < 9 structures per 640 acres, > 80 percent of area is agricultural.

Mixed Forest/Agriculture: Polygon ≥ 640 acres, < 9 structures per 640 acres, 20 to 80 percent of area is agricultural and forest land is ≥ 50 percent of the remainder. Improved roads are typically one-half of a mile apart or more.

Low Density Residential/Commercial: Polygon of any size, but must have ≥ 9 structures within it. On average, housing developments are less than 80 acres. Improved road patterns are typically one-quarter of a mile apart or less. House lots are usually > 1 acre. May meet urban qualifications, but is < 40 acres.

Urban: Polygon ≥ 40 acres, commercial, service or residential subdivided area with city road patterns and closely spaced buildings. House lots are usually < 1 acre.

Structures were counted in 80-acre circles around a grid of points outside national forests and national parks; point spacing was approximately one per 470 acres, representing a 17-percent sample of the land area. "Structures" were meant to represent individual dwellings, so multiple associated buildings were only counted once, e.g., an adjoining rural house, barn, shed and pump-house would all count as a single structure. To date, changes in area in forest land use have been calculated and are presented by county, and simple indices of fragmentation have been calculated from mean forest zone polygon size and forest edge-to-area ratio, and are presented by ecoregion section (Cleland and others 2007). An in-depth analysis of land use change is in progress for publication within the next year.

RESULTS AND DISCUSSION

The area in the non-Federal wildland forest land use class in western Washington was 7.82 million acres in 1976 and declined by 490,000 acres (6.3 percent) between 1976 and 2006. The area in the mixed forest/agriculture and intensive agriculture classes declined as well, by 93,000 acres and 160,000 acres, respectively. Although less acreage than for the wildland forest class, these declines represent a 26-percent and 19-percent reduction in the area mixed forest/agriculture and intensive agriculture classes, respectively. Of this 743,000-acre loss of resource lands, two-thirds ended up in the low-density residential class and one-third in the urban class by the end of the 30-year period. The proportional reduction in area in the wildland forest class varied by county, with the highest rates around the Puget Sound and in the southernmost county, Clark County (fig. 9.1).

On average, the smallest wildland forest polygons, and those with the lowest area-perimeter ratios, were found in the Puget Trough and Willamette Lowland ecoregions (table 9.1). The North Cascades had the largest mean polygon area, whereas the Southern Cascades had the largest area-perimeter ratios. Although the amount of intact forest is high in both ecoregions, the higher elevations and extensive alpine areas in the North Cascades is

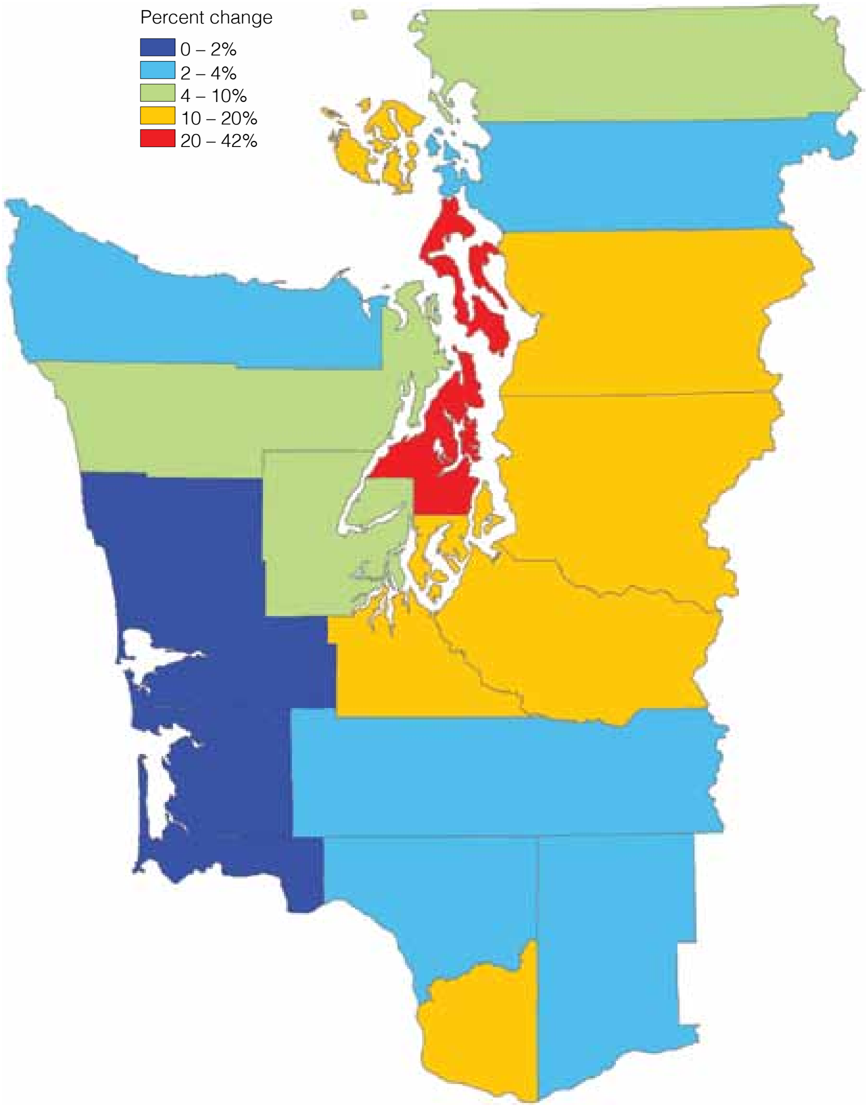

Figure 9.1—Loss of area in the wildland forest land use class between 1976 and 2006 in western Washington.

a likely source of natural fragmentation. Mean polygon areas are substantially smaller in the Coast Ranges than in the Cascades, but area-perimeter ratios are slightly higher than the North Cascades, suggesting a substantial portion of interior forest despite the relatively small size. The forest class area-perimeter ratio decreased for all ecoregions between 1976 and 2006, indicating an increase in edge conditions. Mean polygon area decreased over time for the Puget Trough and Willamette Lowlands, and southern Cascades ecoregions, but were relatively stable in the other ecoregions.

CONCLUSIONS

The area of non-Federal land in the wildland forest land class in western Washington has declined over the last 30 years. The same has occurred for agricultural lands, with resource lands being converted to residential and urban land uses. The remaining forest land class areas have become more fragmented over time, with greater edge to interior proportions, and generally smaller patch sizes. Ongoing work will complete the analysis for the rest of the State and investigate additional measures of forest fragmentation.

Table 9.1—Wildland forest land use fragmentation metrics by ecoregion and year for western Washington

		Year		
Ecoregion section	Metric	1976	1994	2006
Coast Ranges	Total area (ha)	1 731 361	1 724 978	1 721 180
Coast Ranges	Total perimeter (km)	5 473	5 732	5 866
Coast Ranges	Area/Perimeter (ha/km)	316	301	293
Coast Ranges	Mean area (ha)	21 114	20 294	20 490
Northern Cascades	Total area (ha)	1 203 997	1 201 562	1 200 651
Northern Cascades	Total perimeter (km)	3 748	3 852	3 881
Northern Cascades	Area/Perimeter (ha/km)	321	312	309
Northern Cascades	Mean area (ha)	401 332	400 521	400 217
Puget + Willamette lowlands	Total area (ha)	882 813	781 461	727 166
Puget + Willamette lowlands	Total perimeter (km)	15 658	16 090	16 319
Puget + Willamette lowlands	Area/Perimeter (ha/km)	56	49	45
Puget + Willamette lowlands	Mean area (ha)	4 184	3 869	3 513
Southern Cascades	Total area (ha)	1 383 101	1 370 864	1 363 656
Southern Cascades	Total perimeter (km)	2 953	3 374	3 486
Southern Cascades	Area/Perimeter (ha/km)	468	406	391
Southern Cascades	Mean area (ha)	138 310	97 919	85 228

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See also fhm.fs.fed.us/posters/posters09/changes_fragmentation_wa_forest_land.pdf.

ACKNOWLEDGMENTS

We thank four anonymous reviewers for reviews of an earlier version of this manuscript. Beth Willhite and Gary Lettman assisted with project development. Tyler Robinson and Joel Thompson assisted with data collection. Funding was provided by the Pacific Northwest Research Station of the Forest Service, U.S. Department of Agriculture, and by the national Forest Health Monitoring Program of the Forest Service.

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CHAPTER 10.

Evaluating Key Landscape Features of a Climate-Induced Forest Decline (Project WC-EM-07-01)

Paul Hennon and Dustin Wittwer[1]

INTRODUCTION

Yellow-cedar is a culturally, economically, and ecologically important tree in coastal Alaska that has been experiencing a widespread mortality known as yellow-cedar decline for about 100 years. Mapping during annual aerial detection surveys has identified nearly the entire geographical distribution of the problem, which totals over 500,000 acres in Alaska (Lamb and Wurtz 2009) and additional acreage in adjacent British Columbia (Hennon and others 2005). This broad-scale approach to detection has been useful in determining the general occurrence of this forest decline, but mapping from aircraft produces large polygons that are too coarse to evaluate some associated landscape features. Examining yellow-cedar decline at a finer spatial scale is needed to test associations of factors such as slope, elevation, and aspect.

We conducted a mid-scale analysis of yellow-cedar decline on Mount Edgecumbe on Kruzof Island near Sitka, AK (fig. 10.1). Mount Edgecumbe is a dormant volcano with radial symmetry and wet plant communities that likely support abundant yellow-cedar at a range of elevations. Thus, it is a model landscape for this project. Our objectives were to compare the occurrence of yellow-cedar decline as mapped by aerial survey and from aerial photographs, determine the association of elevation, aspect, and slope on the presence of yellow-cedar

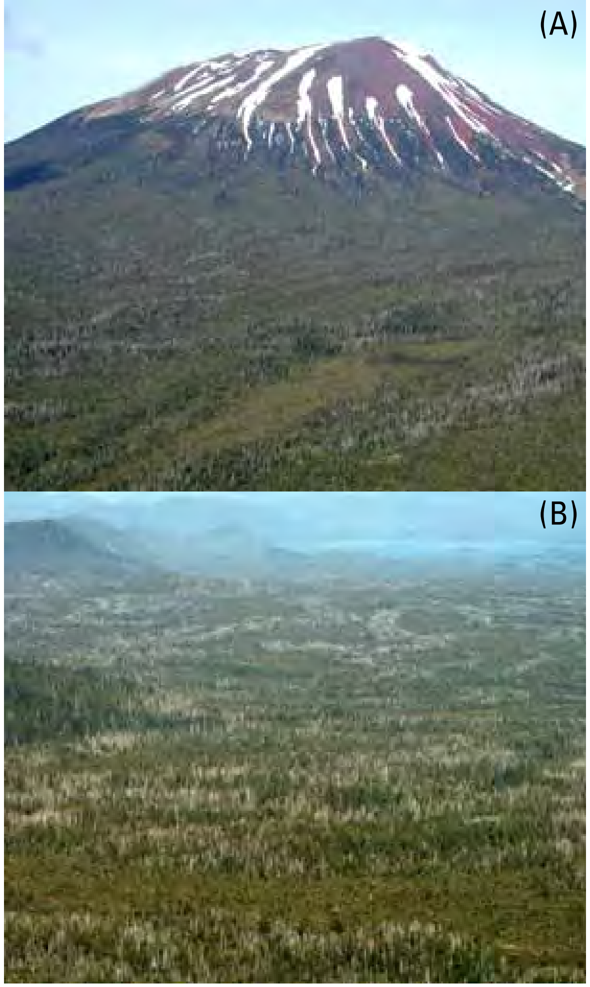

Figure 10.1—Photographs of (A) Mount Edgecumbe and (B) dead yellow-cedar forests at lower elevation. (Photos by Dustin Wittwer, U.S. Department of Agriculture Forest Service)

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decline, and develop methods to detect healthy yellow-cedar populations. New information indicates a tight relationship between snow and the health of yellow-cedar forests (Beier and others 2008, Hennon and others 2010, Schaberg and others 2008); thus, a second phase of this project was to model snow accumulations to identify future suitable habitat on Mount Edgecumbe for yellow-cedar.

METHODS

Aerial survey methods and photo interpretation documented yellow-cedar decline in similar locations (fig. 10.2), but with widely different spatial results. Photo interpretation produced about 25 times the number of polygons of yellow-cedar decline, but only about 25 percent of the acreage compared to aerial survey. Thus, the two methods differ in their spatial resolution of the problem with aerial photography producing a more resolute map, which we used in the analyses below. We are unaware of similar efforts to validate results from aerial detection surveys using other forms of remote sensing such as aerial photograph interpretation.

Using raster analysis, the occurrence of decline was classified and quantified according to the terrain derivatives of elevation, aspect and slope, each calculated from a 30-m digital elevation model (DEM). The decline area was classified in equal intervals and normalized based on the total land area in the study area corresponding to the terrain derivative class categories.

RESULTS AND DISCUSSION

Histograms of the normalized data revealed elevation, aspect, and slope all were associated with the occurrence of yellow-cedar decline. The decline problem is restricted to low elevations, with considerably less decline above about 600 feet. We detected an elevation-aspect interaction, where yellow-cedar decline occurred higher in elevation on the warmer southerly aspects than northerly aspects. Also, yellow-cedar decline was associated with gentle slopes compared to steep slopes. These results are consistent with results from other evaluations of elevation (Hennon and others 2010, Lamb and Wurtz 2009) and slope-drainage (D'Amore and Hennon 2006, D'Amore and others 2009).

Helicopter surveys produced a reliable method for detecting yellow-cedar in the apparently healthy forests at higher elevations. This is the first successful remote sensing approach for verifying the occurrence and amount of yellow-cedar in healthy forests. A kriged map produced from the helicopter survey points illustrates the substantial population of healthy yellow-cedar above the decline zones (fig. 10.3). Ground checks verified the results from helicopter surveys. Our field observations above the dead zones indicate differences in the "quality" of yellow-cedar crowns of live trees where yellow-cedars on the north side of the mountain had full, green crowns and those on the south side often had thin crowns. Thus, our sampling approach to detect live and dead yellow-cedar forests included the increasingly detailed methods of aerial detection survey from an airplane, aerial photographs, helicopter surveys, and ground plots.

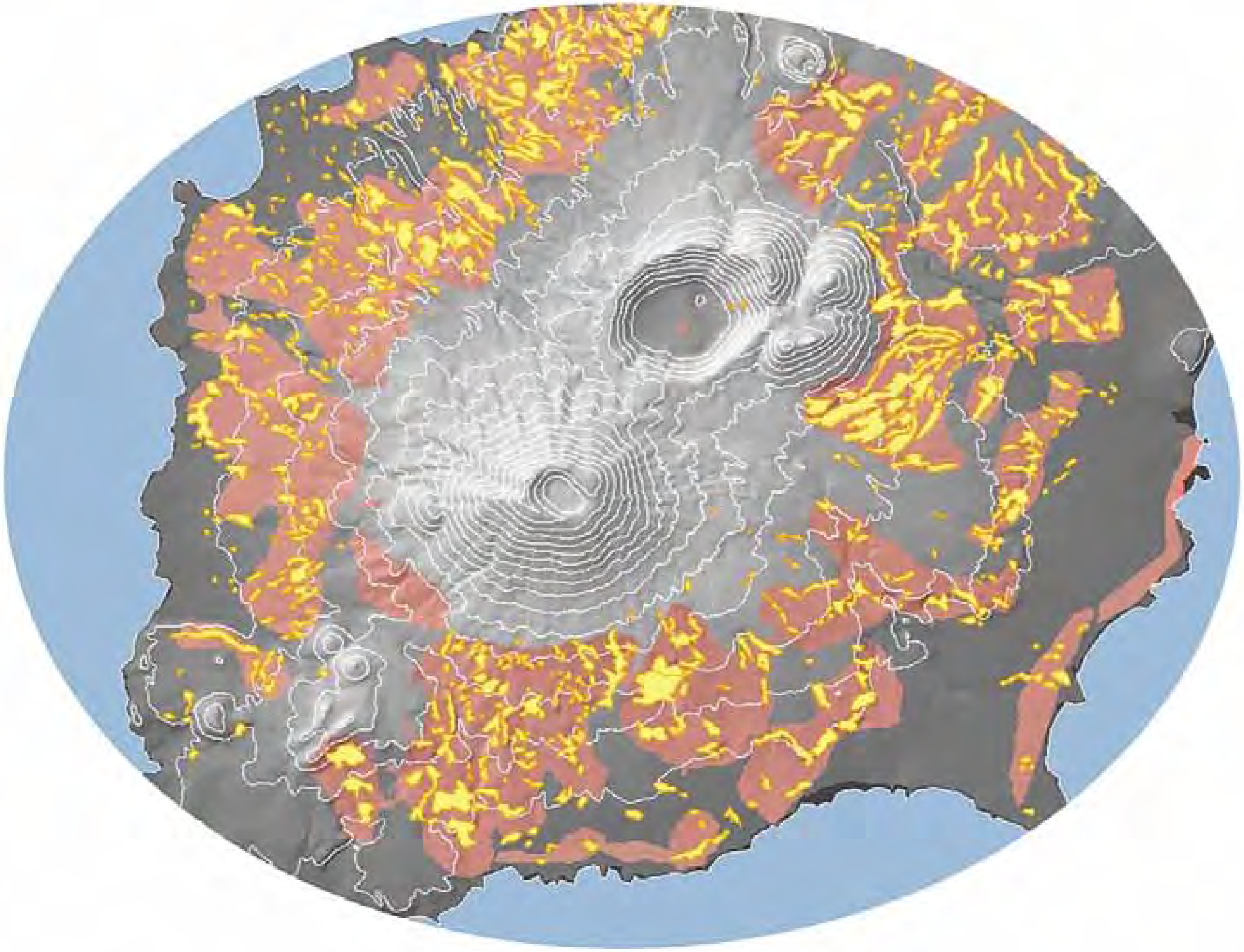

Figure 10.2—Map of Mount Edgecumbe depicting the occurrence of yellow-cedar decline as detected by aerial survey (orange) and analysis of aerial photography (yellow).

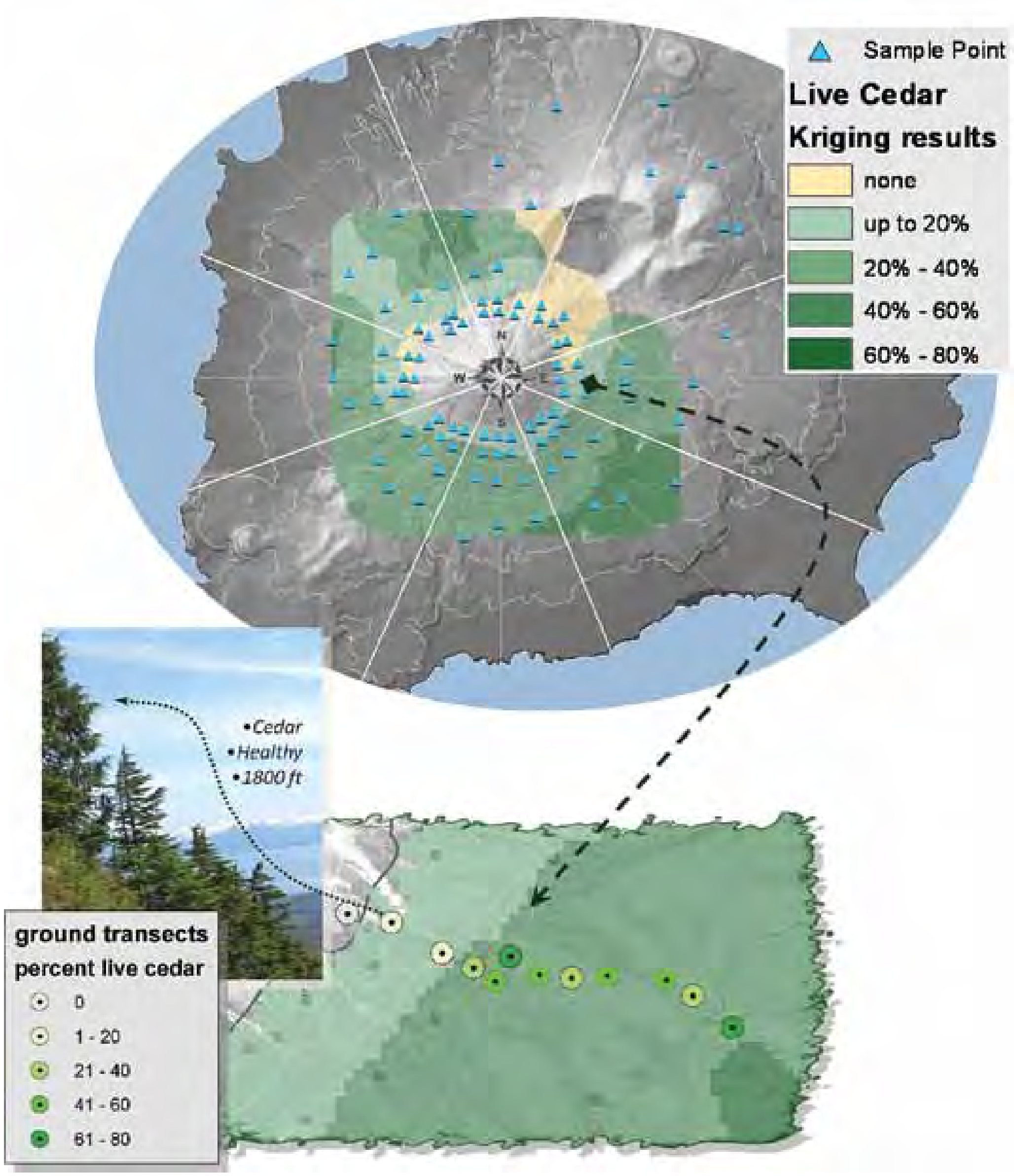

Figure 10.3—A helicopter survey was used to construct forest tree composition in the healthy higher elevation forests. The map above displays the percent occurrence of yellow-cedar by kriging the helicopter survey data. Ground surveys confirmed the abundance of cedar there.

These results support the contention that patterns of snow accumulation dictate the health of yellow-cedar forests (Hennon and others 2008). In the second phase of our project, we used GIS PRISM (Parameter-elevation Regressions on Independent Slopes Model) tools (Daly and others 1994), a new elevational adjustment technique (Wang and others 2006) for downscaling, and global circulation models to produce snow accumulation models for Mount Edgecumbe. Output maps show estimated snow patterns in the early 1900s to the present using weather station data in PRISM. Future predictions of snow accumulation to 2080 were made using PRISM and a conservative global circulation model (we used Coupled Global Circulation Model, Second Generation, B2 scenario) (IPCC 2001). The amount of annual snow accumulation sufficient to protect yellow-cedar is displayed as shades of blue on the maps (fig. 10.4). We used the current distribution of yellow-cedar decline (red) as a benchmark to determine the critical snow accumulation threshold. The maps show substantially more snow in the early 1900s, but shrinking zones of adequate snow to the present and into the future. By 2080, only a small area near the cone of the volcano is predicted to have sufficient snow to protect yellow-cedar from the root freezing injury.

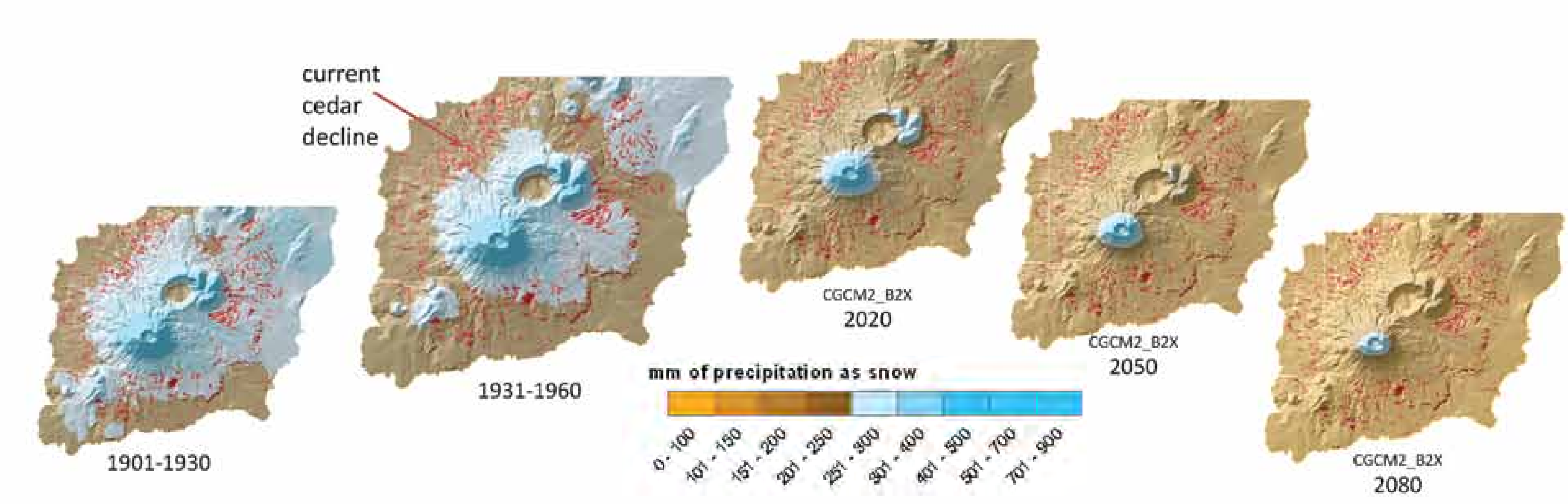

Figure 10.4—Past and projected (CGMC2 B2 scenario shown here) annual snow accumulations using PRISM data, with downscaling by an elevational adjustment (Wang and others 2006). Light blue zones represent sufficient snow to protect cedar from spring freezing injury (annual precipitation as snow = 2500 mm); current areas of cedar decline mapped from aerial photographs are shown in red. Note the abundance of habitat protected by snow (shades of blue) in the early 1900s and progressive shrinking of this habitat through this sequence, to being nearly absent by 2080. (Data sources: PRISM Group, Oregon State University; IPCC 2001)

CONCLUSIONS

We hope to extend these approaches to the entire geographical distribution of yellow-cedar in coastal Alaska. In another project, extensive use of forest inventories are being used to build a general distribution map and habitat models of healthy yellow-cedar, because we still do not currently have adequate knowledge of where healthy cedar populations exist. Snow modeling will be expanded geographically in an attempt to identify current and future suitable habitat for yellow-cedar throughout the entire region. This search for suitable habitat is a key component to developing a conservation and management strategy (Hennon and others 2008) for promoting the valuable yellow-cedar tree into the future.

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CHAPTER 11.

Mountain Pine Beetle in Lodgepole Pine: Mortality and Fire Implications (Project INT-F-07-03)

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INTRODUCTION

Mountain pine beetle (*Dendroctonus ponderosae* Hopkins) has infested over 2 million acres of lodgepole pine (*Pinus contorta* Dougl. ex Loud.) forest since an outbreak began approximately in 2000 in north central Colorado. The tree mortality from mountain pine beetle outbreaks has the potential to alter stand composition and stand characteristics, along with fuel complexes. In general, it is assumed that these changes in stand structure from mountain pine beetle outbreaks in lodgepole pine forests increase fire hazard (Arno 1980, Jenkins and others 2008), though lodgepole pine fire regimes are characterized as having stand replacing high-severity fires, with nonlethal surface fires generally playing a lesser role in lodgepole pine ecosystems (Arno 1980, Kipfmueller and Baker 2000). To quantify the amount of mortality in infested lodgepole pine stands, along with identifying differences in stand characteristics and tree species composition before and after infestation, a study was conducted in Colorado lodgepole pine 7 years after mountain pine beetle outbreak initiation. Furthermore, litter, duff, and fuel bed depth along with downed woody debris loads and vegetation characteristics were examined in infested and uninfested stands. We also compared potential fire behavior and first order effects modeled with the Fire and Fuels Extension to the Forest Vegetation Simulator (FFE-FVS) in uninfested stands, stands 7 years after mountain pine beetle outbreak initiation, and infested stands with projected fuel and stand characteristics that represent 10-percent and 80-percent tree fall.

METHODS

The study was conducted in the Sulphur Ranger District, Arapaho National Forest, in Colorado (40° 4′ N, 106° 0′ W). During 2006 and 2007, a geographic information system was used to randomly select potential plot locations within the lodgepole pine forest type from Forest Service, U.S. Department of Agriculture, vegetation cover maps. We established a total of 221 0.05-acre (26.3-foot radius) plots that were in uninfested areas (51 plots) and infested areas (170 plots). Plots in areas with infested lodgepole pine were either initially infested between 2000 and 2003 (68 plots), hereafter referred to as plots 4 to 7 years after infestation, or between 2004 and 2007 (102 plots), hereafter referred to as plots 0 to 3 years after infestation (0 represents current infestation in 2007). In a plot, the tree with the earliest year of infestation by mountain pine beetle determined the time since infestation category for the plot. The year a tree was infested by mountain pine beetle was estimated by degradation status of the crown. For plots 4 to 7 years after infestation, all needles had been shed from the earliest infested lodgepole pine, whereas for the plots 0 to 3 years after infestation, most needles remained on the infested lodgepole pines.

In each plot, diameter at breast height (d.b.h.) (4.5 feet above ground), tree species, condition (live, infested or killed by mountain pine beetle, or dead from other causes), and year of mountain pine beetle infestation were recorded. Downed woody fuel loads and fuel depth were recorded on modified Brown's fuel transects (Brown 1974) and vegetation cover was visually estimated in

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each plot. Fuel loads and stand characteristics were used as input into a fire behavior model to evaluate potential fire behavior, fire type, tree mortality, and smoke production. The Central Rockies variant (Suppose 2.02) (Edminster and others 1991, Reinhardt and Crookston 2003) of FFE-FVS was used to model potential fire behavior and effects for uninfested plots and for infested plots under high fire weather conditions at three points: 7 years after outbreak initiation, at projected 10-percent infested tree fall, and at projected 80-percent infested tree fall. Mitchell and Preisler (1998) indicated that about 10 percent of mountain pine beetle-killed lodgepole pine in Oregon had fallen 6 years after infestation and 80 percent after 12 years. For infested plots in this study, 10-percent mountain pine beetle-killed tree fall was chosen to represent about 6 years after infestation and 80 percent to represent about 12 years. Surface fuel loads under projected 10-percent and 80-percent tree fall were calculated from infested trees using allometric equations (Brown 1978, Pearson and others 1984).

RESULTS

Stand Conditions

We found that stand characteristics and surface fuel loads have been significantly altered due to this recent mountain pine beetle outbreak (Klutsch and others 2009). Lodgepole pine density was subsequently reduced by 42 percent in stands infested with mountain pine beetle, resulting in an average density of 240 live lodgepole pine per acre (standard error of the mean [SEM] = 18). The distribution of trees per diameter class killed by mountain pine beetle was significantly different than the distribution of live trees (chi-square = 291.5, $p < 0.0001$) in infested plots (fig. 11.1—live and infested lodgepole pine bars). Mountain pine beetle selectively infested larger lodgepole pine resulting in a reduction of live quadratic mean diameter of 34 percent in affected areas. There was still a component of other tree species, such as Engelmann spruce, subalpine fir, and quaking aspen, in infested plots, though lodgepole pine was the most abundant live tree species in the d.b.h. classes of 1 to 12 inches (fig. 11.1).

Logistic regression modeling identified lodgepole pine basal area (plot level) as a predictor of the probability of attack. Other variables modeled included, lodgepole pine/ha but did not demonstrate a strong relationship to probability of attack. There was a positive coefficient of lodgepole pine basal area which indicates an increase in the likelihood of attack as lodgepole pine basal area increases. The estimated probability of infestation for a plot was 0.64 with a lodgepole pine basal area of 9 square feet per acre, the lowest observed in our study.

Downed fine and coarse woody debris loads were not different between infested plots and uninfested plots 7 years after outbreak initiation. However, the median litter depth was significantly greater in plots 4 to 7 years after initial infestation compared to plots in 0 to 3 years after initial infestation and uninfested plots. The average amount of downed woody

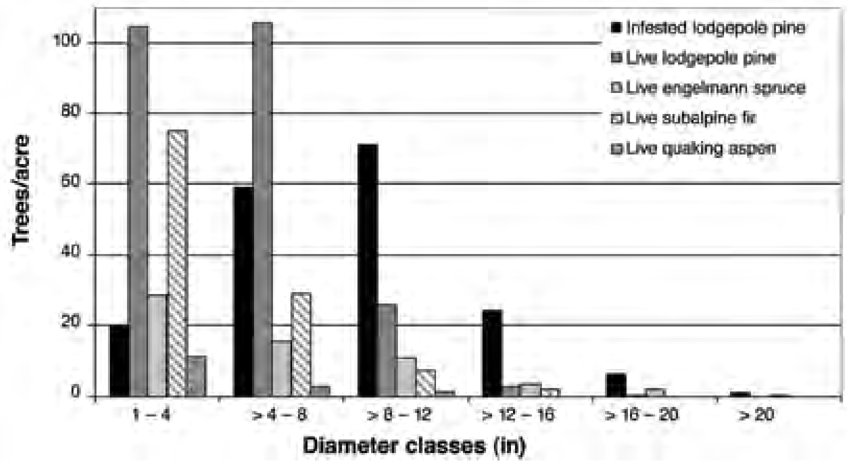

Figure 11.1—Diameter distribution of mountain pine beetle-killed lodgepole pine, live lodgepole pine, live Engelmann spruce, live subalpine fir, and live quaking aspen in stands with mountain pine beetle-caused mortality, Arapaho National Forest, Colorado, 2006–07.

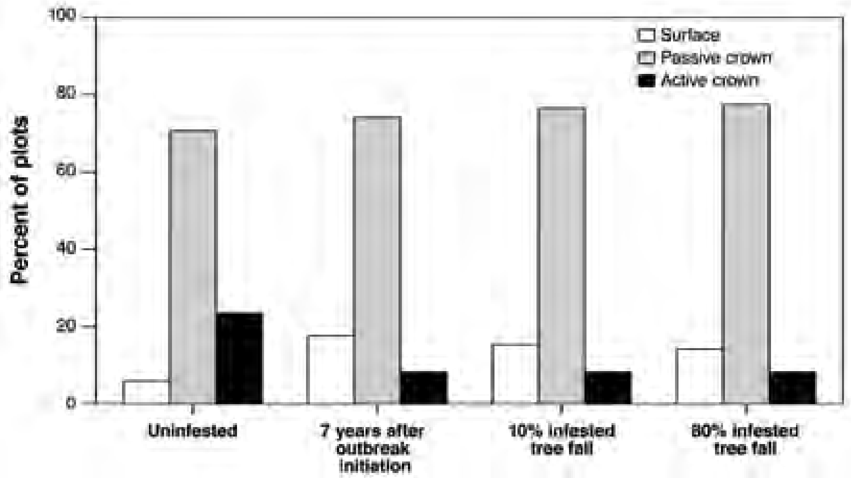

Figure 11.2—Percent of plots within infestation class with surface, passive, conditional or active fire type with 90th percentile fire weather. Significantly different distribution (p-value< 0.0001) tested with a Pearson's chi-square test (test statistic = 79.5, df = 9).

debris in infested stands was not different from uninfested stands, with total surface fuel loads of 8.7 tons per acre (SEM = 8.0) and 9.8 tons per acre (SEM = 8.3), respectively. Simulations in which 80 percent of affected trees were converted to surface fuels showed that projected downed woody fuel loads (total, fine, and coarse) were significantly greater than with no tree fall, resulting in an estimated 22.8 tons per acre increase in total fuel load.

Fire Behavior

There were significant differences in potential fire behavior between uninfested and infested plots under high fire weather conditions. Infested plots had a greater potential for fire to move from the surface to the crown as a passive fire but had less potential for fire to move between crowns as an active crown fire as compared to uninfested plots (fig. 11.2). The potential smoke production from a fire was modeled to be greater with 80-percent infested tree fall than in uninfested plots and plots with less tree fall. Tree mortality from a fire was predicted to be almost 100 percent in both uninfested and infested plots.

DISCUSSION

Stand Conditions

Mountain pine beetle populations in our study sites have initially shown a preference for densely stocked lodgepole pine stands. Mountain pine beetle has been more active in stands with higher lodgepole pine basal area, and lower tree density and basal area of non-host species. This is consistent with literature that indicates mountain pine beetle prefer densely stocked stands dominated by suitable host tree species (Fettig and others 2007). Within infested stands, mountain pine beetle exhibited preference for larger diameter trees, as killed trees were significantly larger than the residual live trees. This could be due to beetles preferring trees with thicker phloem for brood development and/or larger trees being older, less vigorous, and less able to defend themselves than younger trees. In addition, the distribution of live lodgepole pine and mountain pine beetle-killed trees by diameter classes in infested plots indicates preference for large diameter trees, which is also consistent with the literature (Amman 1977, Cole and Amman 1969).

The depth of litter increased in plots 3 to 7 years after infestation compared to plots 0 to 3 years after infestation and uninfested plots, as enough time had elapsed for most of the needles and some of the twigs to drop from these trees. Mortality that had occurred within 3 years after infestation includes trees that still had a large component of needles attached at the time of measurement. This is consistent with studies on mountain pine beetle epidemics in lodgepole pine forests in Utah and Idaho, where Page and Jenkins (2007) showed a greater amount of litter in current epidemic stands compared to stands with endemic levels of mountain pine beetle.

Quantification of downed woody debris showed no change in mountain pine beetle-affected stands within 7 years after outbreak initiation, which was unexpected as we anticipated the initiation of tree fall. The lack of tree fall in our plots suggests that tree fall rates will be slower in north-central Colorado than those reported by Mitchell and Preisler (1998) for Oregon. Nevertheless, when 10 percent of killed trees fall, the amount of downed woody debris in infested plots was projected to not be different from that of uninfested plots. With 80 percent of infested trees down, we anticipate a significant increase in surface fuel loads. The median amount of coarse woody debris > 3 inches that is expected to be on the ground is 29 tons per acre [the median and median absolute deviation (MAD) = 13]. Coarse woody debris amounts are highly variable (Brown and others 2003); therefore, the amount of downed woody debris projected to accumulate after tree fall in our study sites is not likely to be different from that reported for historical and disturbed stands (Brown and See 1981, Tinker and Knight 2000). Coarse woody debris accumulation from the mountain pine beetle outbreak will persist for many decades due to slow degradation rates. Brown and others (1998) found that windthrown lodgepole pine logs in north-central Colorado took more than 150 years to completely decompose, though degradation rates may be more rapid due to bark beetle infestation (Busse 1994).

Fire Behavior

The different fire types and potential fire behavior predicted for infested and uninfested plots is due to a combination of the surface fuel loads and stand characteristics. The extensive mortality in infested stands reduced the canopy fuel continuity. The greater potential for passive fire to occur in infested plots is due to less fuel in the crown with the passive fire occurring in the lodgepole pine and other tree species remaining. Similarly, in Engelmann spruce forests, DeRose and Long (2009) modeled potential fire behavior 10 to 29 years after an outbreak of spruce beetle (*Dendroctonus rufipennis* Kirby) and found active crown fires to occur in areas unaffected and areas with low densities of spruce beetle-killed trees but not in areas with high densities of mortality.

Mountain pine beetle-caused mortality after an outbreak in lodgepole pine-dominated forest will affect fire behavior through accumulations of surface fuels and changes in stand characteristics. Plots without mountain pine beetle-caused mortality with intact and continuous canopy fuel components were expected to have greater potential for active crown fire than infested plots, though potential mortality from a fire was extremely high for both uninfested and infested plots. Although bark beetle-caused mortality changes fuel complexes, the occurrence of a wildfire is also dependent on the timing of fire conducive weather with a sustainable ignition event.

IMPLICATIONS

Effects of the current mountain pine beetle outbreak on ecological processes such as downed wood accumulation and decomposition, hydrology, windthrow events, fire, and others in lodgepole pine forests of north-central Colorado will be long lasting. Native bark beetles are important disturbance agents in forest ecosystems and have co-evolved with their hosts for millennia. Under a climate change scenario it is expected that disturbance processes may change in frequency and become more intense (Dale and others 2001) with eruptive forest insects having the potential to create large-scale ecological changes (Logan and others 2003). The distribution, range, and abundance of different conifer species will depend on their response to increasing temperatures (Nielson and others 2005). The high intensity, large-scale, and infrequent fires of lodgepole pine forests have been associated with region-wide and multi-year drought conditions (Bessie and Johnson 1995, Kipfmueller and Baker 2000). The occurrence of a fire during high fire weather could be stand replacing in areas affected and not affected by mountain pine beetle. As demonstrated, the speed at which the fire moves may be different, with passive fires in mountain pine beetle-affected areas potentially moving slower than active crown fires in areas not affected. It will be imperative to continue long-term monitoring of ecological processes after these extensive bark beetle outbreaks as lessons may be of benefit to understanding these novel interactions and to mitigate, as appropriate, future bark beetle outbreaks if they continue to be exacerbated under a changing climate scenario.

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CHAPTER 12.

Assessment of Balsam Woolly Adelgid Damage to Eastern Washington and Oregon Subalpine Fir (Project WC-F-07-01)

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INTRODUCTION

Balsam woolly adelgid (BWA) (*Adelges piceae*) is an exotic, aphid-like sucking insect originally from Europe that feeds only on *Abies* species. North American species are particularly susceptible, with the apparently most susceptible being subalpine (*A. lasiocarpa*), Frasier (*A. fraseri*), and balsam fir (*A. balsamea*) (Newton and Hain 2005). The BWA feeds directly through the bark on stems, branches, and buds, causing swelling at the buds and branch nodes ("gouting"), dieback, and tree death. During feeding, the insect injects a salivary substance into the host tree, causing branch calluses and abnormal wood formation. Heavy bole infestations usually kill the tree. Branch and twig infestations cause gouting, which progressively weakens the tree. Infestations can occur at any point in a tree. This results in topkill, top curl, dead branches in the middle, or random dead branches throughout the crown (fig. 12.1).

The BWA was first found infesting balsam fir in North America around 1900 in New England and northeastern Canada (Balch 1952). It was documented west of the Cascades in Oregon in about 1930 and Washington in 1952 (Livingston and others 2000). It was first found east of the Cascades in 1972 near Walla Walla in southeastern Washington (Curtis and Johnson 1975) and by 1999 could be found throughout Idaho on subalpine fir (Livingston and others 2000).

Figure 12.1—Subalpine fir on left has thinning crown due to infestation by balsam woolly adelgid. Subalpine fir on right appears healthy. (Photo by Lia Speigel, U.S. Department of Agriculture Forest Service).

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The adelgid has a complex, two-host lifecycle that includes a sexual stage on spruce and a parthenogenetic stage (reproduction without male fertilization) on true firs. In North America and much of Europe, it is known only from its fir hosts. The females can produce more than one generation a year of 30 to 100 eggs each (Mitchell 1966), and, because it takes just one female to initiate a population, populations can quickly establish and expand.

The BWA has been studied intensely in the Eastern United States, where it has decimated Frasier fir, and in the Western United States, west of the Cascades, where it has nearly eliminated old grand firs in low elevations. But its impact on high elevation subalpine fir is not well understood. In recent years, much of the BWA mortality documented in aerial surveys has occurred east of the Cascades in subalpine firs. The pattern of mortality to larger trees seen in both Frasier and grand fir (Ragenovich and Mitchell 2006) appears to hold for subalpine firs as well; however, this has not been well documented.

Surveys conducted in 1998 to 2000 in Oregon and Washington revealed several areas in northeastern Oregon and central Washington, where there were no BWA symptoms and the adelgids could not be found.

This project established long-term monitoring plots in subalpine fir stands throughout eastern Oregon and Washington. These plots are in a range of elevations, in areas that encompass both long-established adelgid populations and areas that had not had adelgids 10 years prior. The extent and severity of damage and mortality to subalpine fir and other true firs from BWA were measured and will be monitored in the future. In addition, we assessed management options to ensure the long-term viability of subalpine fir on the landscape.

METHODS

Permanent plots were installed in 26 sites on the Okanogan, Wenatchee, Deschutes, Ochoco, Malheur, Wallowa-Whitman, and Umatilla national forests east of the Cascades in Oregon and Washington, and in Mt. Rainier National Park in Washington. We established 8 to 12 plots at each location. Data on trees 5 inches in diameter and larger were recorded on 0.1-acre plots, data on trees down to 1 foot tall were recorded on nested 0.02-acre plots. All green trees including blown down trees were measured. Dead standing trees were also measured to facilitate remeasurement in 5 to 10 years.

Three measures of BWA damage and abundance were developed to characterize the infestation in east-side true firs. Bole infestation was recorded, gouting severity was measured, and a BWA rating (BWAR) system captured crown damage in the form of dieback and dead branches.

The BWAR system is an adaptation of the white pine blister rust severity rapid rating system developed by Six and Newcomb (2005) and based on the Hawksworth (1977) dwarf mistletoe rating system. As in the Hawksworth system, tree crowns are divided into equal thirds. Each crown third received a rating of 0-4, with 0 meaning no visible damage, 1 meaning < 25 percent of crown third infested/affected or dead, 2 meaning 25-49 percent of crown third infested/affected or dead, 3 meaning ≥ 50 percent of crown third infested/affected or dead, and 4 meaning no live branches. These numbers are summed for an individual whole tree rating. This also allows tracking of crown thirds over time since BWA damage can manifest in sudden dead tops, individual dead branches, dead sections, or overall decline, depending on where trees are attacked and the severity of attack.

Gouting severity is a measure of infestation evident on branches. Feeding by the adelgid causes branch nodes and terminal and lateral buds to swell. This reduces branch elongation and new foliage and cone production. We counted the maximum number of gouted branch nodes visible on each tree to characterize gouting damage.

Bole infestation was recorded as an estimated density of BWA individuals visible on the bole (up to about 6 feet in height), assigning a rating of 0-4, where 0 means no adelgids present, 1 means a light infestation of fewer than one adelgid per square foot, 2 means a moderate infestation with 1 to 10 adelgids per square foot, 3 means a heavy infestation of 11 to 100 adelgids per square foot, and 4 means a very heavy infestation at more than 100 adelgids per square foot. Bole infestations lead to branch dieback and whole tree mortality.

RESULTS AND DISCUSSION

The BWA is not yet present in north-central Washington. Recent increased mortality there was caused by a combination of Cytospora canker and *Pityokteines minutus*, which seemed to be acting as a tree killer in drought-affected stands.

All areas of northeastern Oregon previously without BWA in the 1998 to 2000 surveys are now infested, with some of the more recently infested areas experiencing some of the higher current mortality.

Because BWA symptoms are often difficult to distinguish, stem infestations are corroboration that the insect is present on a site. Two sites in central Oregon did not have stem infestations. However, gouting and crown decline (BWAR) were fairly severe. We suspect the woolly material was not detected on these sites due to low densities or infestations higher than the 6 feet we could inspect on the bole. BWA insects on branches are very difficult to detect and in fact we saw them very rarely there. Two sites in Washington also did not have stem infestations. These sites had very little gouting and crown decline and thus presence of BWA on these sites is not confirmed. Further monitoring at these sites in particular is needed.

Cause of subalpine fir mortality was difficult to determine. We did not confirm the presence of rotholtz (compression-like wood caused by adelgid feeding), few trees had evidence of gouting, no old adelgid wool was found and so we cannot definitively attribute cause of death to

BWA. A few trees had evidence of fir engraver, *Scolytus ventralis*, but most had no signs of bark beetles. However, the widespread presence of BWA in most of the plots indicates BWA is likely involved in much of the subalpine fir mortality.

Mortality at many sites was much higher in subalpine fir than other species. Exceptions were plots where bark beetles had been active recently killing lodgepole pine or Engelmann spruce or a few sites where overall mortality was fairly low.

Mortality at all sites of subalpine firs larger than 1 inch diameter at breast height (d.b.h.) varied from 52 percent at one site in Central Oregon to 4 percent in northeastern Oregon. Overall mean mortality was 23 percent. Trees smaller than 1 inch d.b.h. had very little gouting or other evidence of BWA and almost no mortality (fig. 12.2). All but four sites had higher subalpine fir mortality in the 5-inch d.b.h. and larger trees than the trees 1-inch to 4.9-inch d.b.h. Mortality of subalpine fir over 5 inches in d.b.h. was over 45 percent on three sites in central Oregon and the Blue Mountains (northeastern Oregon and southeastern Washington). These findings agree with those in other areas that indicate older, mature trees are more susceptible to BWA (Ragenovich and Mitchell 2006).

BWAR and gouting varied together between plots with generally more gouting coinciding with more heavily impacted crowns (fig. 12.3, Kendall's $\tau = 0.4545$, $\alpha < 0.01$). This agrees with previous reports that crown dieback and thinning occur where branch infestations are

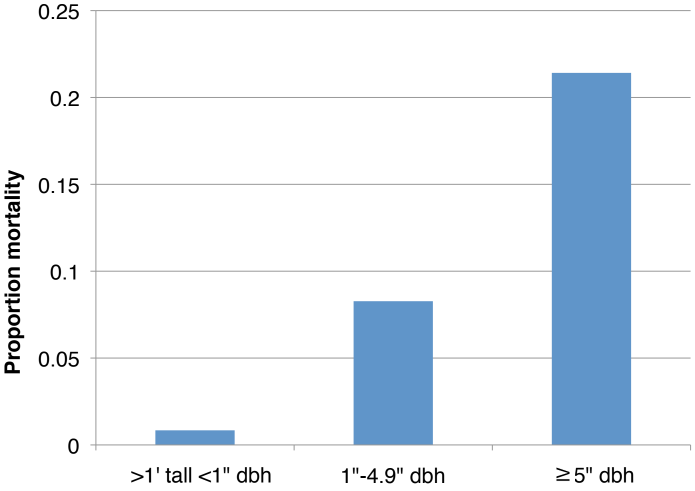

Figure 12.2—Proportion mortality in subalpine fir > 1 foot tall but < 1 inch d.b.h., 1–4.9 inches d.b.h. and ≥ 5 inches d.b.h., averaged across all sites. Larger trees experienced much greater mortality at some sites.

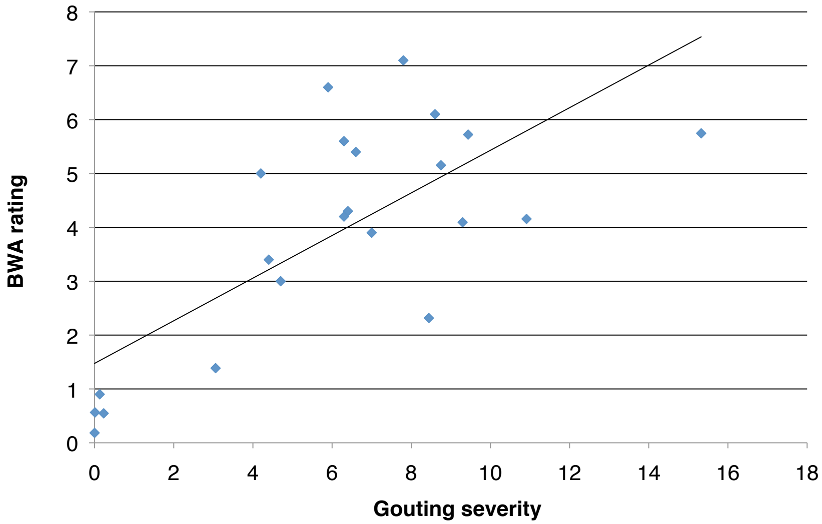

Figure 12.3—Gouting severity varied with balsam woolly adelgid rating (BWAR) crown rating. Gouting severity is the mean of the maximum number of branch nodes exhibiting gouting at each site.

heavy enough to impact growth but not so heavy as to cause outright branch death (Balch 1952).

Stem infestation measures correlated with BWAR (fig. 12.4, Kendall's $\tau = 0.4632$, $\alpha < 0.01$) but not with gouting (Kendall's $\tau = 0.134$, $\alpha > 0.1$). This is as expected as stem infestations affect crown growth and form but are somewhat independent of gouting as they are two different sites of attack. Stem infestations frequently are associated with sudden tree mortality while branch infestations result more commonly in gouting and stunted growth that results in the slow decline of the tree over many years (Ragenovich and Mitchell 2006).

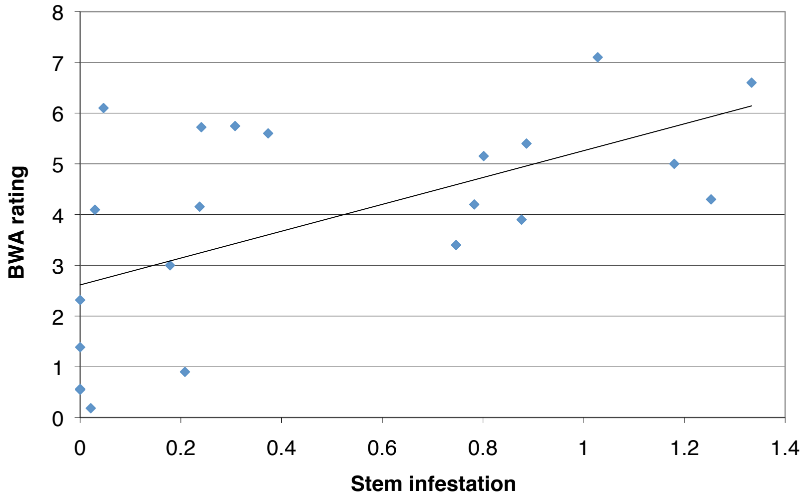

Figure 12.4—The balsam woolly adelgid rating (BWAR) crown rating varied with the stem infestation measurements.

Both BWAR and gouting were low to absent in Washington. Gouting was more severe in central Oregon than the Blue Mountains, while the BWAR's were similar. Very few trees had any crown thirds with no live branches (a BWAR of 4). The poorest crowns occurred at one plot in northeastern Oregon on the Umatilla National Forest, where nearly every crown third of every tree had damage and where BWA has been present for at least 10 years.

CONCLUSIONS

Occurrence and effects of BWA were more widespread and severe in central Oregon and the Blue Mountains than anticipated. The opposite was true in central Washington with some locations having no adelgids present where they were expected.

Once present at a site, BWA remains, with populations apparently fluctuating with the weather (Balch 1952, Mitchell 2001). Previous research has indicated that environmental factors appear to be the prime regulators in the abundance of this insect, with warmer conditions favoring population increases (Mitchell 2001). If a long-term trend of warmer summer temperatures occurs, this insect will likely expand to higher subalpine elevations and new habitats. A landscape analysis of BWA occurrence and severity by site factors such as elevation and precipitation is planned.

When BWA first colonizes a stand, the more vigorous, open grown or edge trees are the first to be attacked (Mitchell 1966). In subalpine fir, attacks typically begin in the upper crown and move down, while in grand fir they typically begin in the lower crown and move up (Mitchell 1966). The smaller trees in stands are frequently suppressed, growing under overstory infested trees. As the overstory trees are killed, the understory trees are released and grow into susceptibility. How an individual tree's susceptibility changes with stand changes is not understood but appears to be related to the nutrients available to the adelgid at specific feeding sites. Continued monitoring of these sites will shed some light on these relationships.

Effects of BWA on subalpine stand dynamics are still not clear, now 30 to 40 years after establishment. Much subalpine fir east of the Cascades in Oregon and Washington occurs in mixed stands. These will probably become more dominated by overstory lodgepole pine and Engelmann spruce, depending on site conditions, if the older subalpine firs die and are not replaced. Currently understory firs are abundant and declines in recruitment were not yet detected.

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CHAPTER 13.

Health, Reproduction, and Fuels in Whitebark Pine in the Frank Church River of No Return Wilderness Area in Central Idaho (Project INT-F-05-02)

Lauren Fins[1] and Ben Hoppus[2]

INTRODUCTION

Whitebark pine (*Pinus albicaulis* Engelm.) is in serious decline across its range, largely due to the combined effects of *Cronartium ribicola* J. C. Fisch (an introduced fungal pathogen that causes white pine blister rust), replacement by late successional species, and widespread infestation of mountain pine beetle (*Dendroctonus ponderosae* Hopkins) (Gibson and others 2008, Hoff and others 1980).

C. ribicola was introduced into North America from Europe, possibly as early as 1898 (Benedict 1981, Fins and others 2001). From two initial points of introduction, blister rust has spread to all five-needle pine species in North America, including nearly the full range of whitebark pine populations (McDonald and Hoff 2001). Whitebark pine trees of all sizes are susceptible to infection, but mortality from blister rust is usually < 100 percent. While uninfected trees may be escapes, an alternative hypothesis is that some of them are genetically resistant, as several genetically controlled defense mechanisms against blister rust have been observed in whitebark pine (Hoff and others 2001, Mahalovich and others 2006). But, because the level of native rust resistance is estimated to be low, only a small percentage of trees are predicted to survive to maturity when rust infection is epidemic (Hoff 1994, Kendall and Keane 2001). Selection and breeding for specific defense mechanisms is likely to improve resistance.

Mountain pine beetle, the most destructive insect pest of pine species in western North America (Gibson 2003), is a native species that ranges from British Columbia and Alberta to northern Mexico. The beetle's recorded hosts include many pine species of western North America. Beetles have killed nearly 6 million high elevation five-needle pines in the last 5 years, with mortality highest in lodgepole pine (Gibson and others 2008). Whitebark pine mortality from mountain pine beetle was reported across almost 500,000 acres in 2007 (Gibson and others 2008). The recent expansion of mountain pine beetle, both northward and upward in elevation, has been attributed largely to warmer than normal temperatures (Bentz and Schen-Langenheim 2007, Carroll and others 2003).

The steep, rocky, and mountainous terrain of the Frank Church River of No Return Wilderness Area (hereafter referred to as the Frank Church) covers approximately 2.3 million acres and includes much of the Salmon River drainage in central Idaho. It is the second largest wilderness area in the conterminous United States.

Whitebark pine is broadly distributed in the Frank Church from approximately 2300 m to timberline, but to date, no extensive studies have been published on the health status of these populations. As a keystone species in these ecosystems, the general condition of whitebark pine populations is of major importance and any significant decline could have broad ecological ramifications. The purpose of our study was to provide baseline information on the health status of whitebark pine populations in the Frank Church. This information will be useful to managers in assessing population trends and

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developing management strategies, especially if any remedial action is anticipated. Our objectives were to:

1. Evaluate and establish baseline information on levels of infection and mortality from white pine blister rust and mountain pine beetle in whitebark pine populations in the Frank Church.
2. Establish baseline information on existing stand structures and occurrence of whitebark pine regeneration.
3. Estimate loadings of downed woody material in these populations.

MATERIALS AND METHODS

Between the summers of 2005 and 2008, we established and assessed the conditions of trees in 119 permanent study plots in six whitebark pine populations in the Frank Church. Populations were selected based on landscape level GAP analysis cover type data (Scott and others 2002), as well as on fire history data from the Forest Service, U.S. Department of Agriculture (Gibson and Morgan 2009), but selection was also influenced by accessibility and logistical considerations. The initial study design consisted of a 3 x 3 factorial of plot types (3 habitat classes x 3 burn classes) replicated three times each, i.e., 27 study plots in each study population. We were able to locate sites and establish the full factorial design (nine habitat/burn factor combinations) in three of the six populations, two of which included at least three replications per habitat/burn combination. In the other three populations, where some of the habitat/burn combinations were either limited or missing, plots were established in the combinations that were available.

Plot locations were selected based on availability of areas with appropriate habitat/burn combinations and accessibility, with a minimum distance of 30 m between plots. However, most of the plots (> 80 percent) were separated from the nearest plot by at least 100 m. Rectangular plots (150 feet x 30 feet = 45.7 m x 9.1 m = 0.04 ha) were established as per Whitebark Pine Ecosystem Foundation guidelines (Tomback and others 2005). Plot elevations ranged between 2290 m and 2930 m, primarily on south aspects, with slopes from 0 to 80 percent, averaging 32 percent. Data were analyzed using a nested model, with stand composition and burn history nested within population. Population, habitat type, and burn history were treated as fixed independent variables.

Data collection generally followed protocols recommended by the Whitebark Pine Ecosystem Foundation, with some additions and minor modifications (Tomback and others 2005). Data on whitebark pines > 1.4 m tall included d.b.h., distance along and away from the transect (midline of the plot), overall health, canopy condition, blister rust status, and mortality. Types and locations of blister rust cankers were recorded, as well as whether the cankers were active or dead. Also recorded were counts of whitebark pines shorter than 1.4 m (considered as the regeneration class), occurrence of blister rust in this group, and whether trees were shorter or taller than 0.5 m. Plot data also included location, aspect, elevation, and year plots were established, percent composition of other tree species with stems > 1.4 m tall, mountain pine beetle infestation on live whitebark pine trees, and likely agent of mortality on dead mature whitebark pines. Frank Church RONRWA research protocols were followed in establishing and monumenting plots.

RESULTS

Within the 119 permanent plots in the study were 3,529 whitebark pine trees ≥ 1.4 m and 3,950 whitebark pine seedlings/saplings < 1.4 m. Trees in the latter category (the regeneration class) were found in 110 of the 119 plots (92 percent), their numbers ranging from 1 to 160, and averaging 36 stems (< 1.4 m tall) per plot. Of 3,529 larger whitebark pines in our plots, 661 (19 percent) were dead. Of the 661 dead trees, 23 (3 percent) were clearly killed by blister rust, 252 (38 percent) killed by mountain pine beetle, 262 (40 percent) by fire, and 124 (19 percent) by unknown causes. Our values for blister rust and mountain pine beetle mortality are likely conservative since some of the 124 trees killed by unknown causes are likely to have been killed by one of these two factors.

Blister rust infection was present in all six populations. Plots (of trees ≥ 1.4 m tall) averaged 18.6 percent of live trees infected (range 0–64.5 percent). This average is based on 2008 assessments in five populations and the 2006 assessment in one population. Using data only from the three populations in which all plot types were present, we found a statistically significant difference among the populations in blister rust infection ($p < 0.001$). Blister rust was observed in the regeneration class in only seven of the plots. Differences in blister rust infection levels were not associated with either composition or burn class.

Active mountain pine beetle infestation was observed in 22 of 119 plots (19 percent of plots), ranging from 1.5 percent to 52 percent active infestation within plots. Affected plots averaged 15.7 percent of trees showing active infestations, but across all plots, the overall mean of live trees under attack was only 2.9 percent. Levels of mountain pine attack were significantly different among populations ($p < 0.001$) and among burn classes nested within population ($p < 0.001$). Recently burned plots in the Sleeping Deer Mountain population were attacked at higher levels than other plot types. Habitat type was not associated with differences in levels of mountain pine beetle attack.

The mean litter loading was 4.94 tons/ha (2 tons per acre), ranging from 0 to 22.23 tons/ha (0 to 9 tons per acre); mean duff loading was 38.8 tons per ha (15.7 tons per acre), ranging from 0 to 150.4 tons/ha (0 to 60.9 tons per acre); mean loading of the 0 to 7.62 cm class (0 to 3 inch) fuels was 5.18 tons/ha (2.1 tons per acre), ranging from 0 to 587.86 tons/ha (0 to 23.8 tons per acre); and mean > 7.62-cm class (> 3 inch) fuels was 19.02 tons/ha (7.7 tons per acre), ranging from 0 to 148.94 tons/ha (0 to 60.3 tons per acre).

DISCUSSION

The mean level of white pine blister rust infection on live whitebark pine trees in the Frank Church River of No Return Wilderness Area ($\bar{x}$ = 18.6 percent) is lower than has been reported in other parts of the species' range, but comparable to blister rust levels assessed between 1995 and 1996 near the perimeter of the southern half of the wilderness area, which averaged 17.6 percent infection (Smith and Hoffman 2000). The low level of blister rust infection in regeneration is encouraging, but infection levels are likely to increase over time as the trees live longer, with more opportunities for infection and increased target area (Fins and others 2002).

The mean level of mountain pine beetle attack in live whitebark pine across the six populations was quite low (2.9 percent), and the trees under attack were in small widely scattered clusters. However, since only whitebark pine was inspected for mountain pine beetle in our study, and lodgepole pine was not inspected, the actual numbers of trees attacked per hectare is likely to be much higher. The plots with the highest levels of attack (up to 39 percent of live whitebark pine) were in a recently burned area in the Sleeping Deer Mountain population where as many as 36 whitebark pine trees per hectare were infested and will likely die in the near future.

The mean fuel loads measured on our plots were slightly lower, but comparable to those found in Fire Group 10 habitats described by Crane and Fischer (1986). Fuels were discontinuous and were influenced by the topography and heterogeneity of the landscape. This lack of uniformity was observed among neighboring plots and across populations.

Before modern fire suppression, fires of mixed severity occurred at intervals ranging from 30 to 300 years (Arno 2001). Stand replacing fires created sites for Clark's nutcracker to cache seeds, and light intensity fires killed understory spruce and fir (Arno and Hoff 1990). But fire exclusion in the last century has altered natural fire cycles where whitebark pine is seral, resulting in successional replacement by more shade tolerant species such as subalpine fir (Arno 1986). Restoring natural fire regimes may be possible and useful in maintaining whitebark pine populations, because the fires would create cache sites for the Clark's nutcracker, and at least some of the cached seeds will have come from trees that survived blister rust infection and which may be genetically resistant (Keane and Arno 2001). Further discussion of results and more detailed descriptions of analyses can be found in Hoppus (2009). Photographs of the study area, a description of habitat types and burn classes, and tables and graphs of results by populations and/or by plot type can be found at www.fs.fed.us/foresthealth/fhm/posters/posters09/posters09.shtml (Fins and others 2009).

CONCLUSION

Although population recruitment was evident in most of these populations (92 percent), the persistence of blister rust, in conjunction with losses to mountain pine beetle and potential habitat shifts due to climate change, suggest the possible future loss of some whitebark pine populations. As a hedge against future losses and a strategy to maintain a broader array of future options, we argue for continued monitoring for changes in health in these populations and the collection and archiving of genetically representative samples, including seed and scion from trees in each population.

Our work provides baseline information on the health and regeneration potential of whitebark pine populations in the Frank Church, and the 119 permanent plots we established can be used to help understand and predict the trajectory of these populations in the future. The levels of blister rust infection in the Frank Church were relatively low in 2008 compared to other areas in the range of whitebark pine, such as in Glacier National Park. However, changes in climate or blister rust wave years could sharply increase infection levels over just a few years.

Given the ecological importance of whitebark pine as a keystone species in these ecosystems, it is imperative that genetic materials, such as seed and/or scion, are collected from these populations and placed in genetic archives for their potential use in ecosystem restoration in the future. Furthermore, the populations should be monitored regularly to determine the trajectories of their health status over time and action taken as appropriate.

All options intended to reverse the decline of these populations should be considered, including the dissemination of genetically resistant materials, restoration of natural fire regimes, and maintaining and archiving the gene pools of current populations.

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ACKNOWLEDGMENTS

We thank the Forest Service, U.S. Department of Agriculture, for financial support of this project and for the contributions of the many individuals who helped us in accomplishing the work. We also thank Jim and Holly Akenson for their inspiration on the project, and for their help in finding student workers and in getting the field work accomplished.

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CHAPTER 14.

Assessing Mortality and Regeneration of Larch[1] (*Larix laricina*) after a 1999-2004 Landscape Level Outbreak of the Larch Sawfly (*Pristiphora erichsonii*) in Alaska (Project WC-EM-08-03)

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Mark Schultz,[3]
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and James Kruse[4]

INTRODUCTION

Larch sawfly (LSF) (*Pristiphora erichsonii*) is an invasive defoliator in Alaska. Based on aerial survey data, this insect has impacted an estimated 600,000 to 700,000 acres of eastern larch (*Larix laricina*) stands in Alaska during a 6-year period between 1999 and 2004. Mortality of larch within the sawfly-defoliated area was 80 percent or more (aerial survey data) in the majority of affected larch stands. Consecutive years of larch defoliation on the poorest sites resulted in 100 percent mortality.

A majority of interior Alaska's larch stands are only accessible by float plane, an expensive mode of transport, so very little historical larch stand information has been collected across the species' Alaskan range. Forest health specialists in Alaska used historical National Insect and Disease Detection Survey data, archived geospatial datasets for slope and elevation across interior Alaska, and data from a previous "healthy larch" Forest Health Monitoring Evaluation Monitoring (EM) project (FHM project #WC-EM-05-02) of the Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture, to identify the most likely areas with moderate to high densities of larch for ground sampling. Project goals included evaluation of regeneration potential in the post-outbreak stands and evaluation of the biotic and abiotic factors of past and present larch mortality.

This project also addressed the following EM project selection criteria: the project was initially identified from FHM Aerial Detection Surveys (ADS); the proposed project is significant in geographic scale because it addresses the statewide distribution of larch in Alaska; and the project assessed the biological impacts of the recent extensive larch sawfly outbreak (1999–2004) and effects of the catastrophic 2004 and 2005 fire season in interior Alaska over the entire statewide distribution of larch utilizing previously acquired geospatial data in a geographic information system (GIS) from several sources. Due to limited access to the remote eastern larch distribution in Alaska's interior, GIS greatly aided site selection and cost efficiencies for small plane charter and travel time to complete this EM project.

At the time of this investigation, the regeneration potential of most stands with healthy larch remained unknown because the surviving residual trees may have been too small (young) to produce cones. Based on previous ADS estimates, larch mortality appeared to be concentrated in the largest diameter trees. It is believed that heavy sawfly infestation was a factor in significant mortality over extensive areas, with or without evidence of eastern larch beetle (LB) (*Dendroctonus simplex*) activity in the same stands, based primarily on aerial survey estimates and observations (Seybold and others 2002). We wanted to assess, on the ground, any evidence of LB as a primary mortality agent. Also, larch is a shade-intolerant, early-succession species that requires a significant component of mature, cone bearing trees for recolonizing a site after significant stand disturbances. Many stands sustained close to 100 percent mortality 3 to 4 years after this outbreak was first documented

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[1] Other names: Alaska larch, eastern larch, hackmatack, tamarack, *Larix alaskensis* W. F. Wight, *L. laricina* var. *alaskensis* (W. F. Wight) Raup; reference: Viereck, L.A.; Little, E.L., Jr., 2007. Alaska trees and shrubs, 2d ed., University of AK Press. 273 p.

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and aerially mapped. Without a cost-effective way to conduct ground checks of aerial survey observations within most of the areas, e.g., slack water swampy areas associated with black spruce, insect mortality causal agents are difficult to assess or distinguish from other potential mortality causes, e.g., disease, abiotic agents, and the like.

OBJECTIVES

Fourteen stands were evaluated in 2008 and 2009 to:

1. Determine the primary source of mortality in larch stands between 1999 and 2004.
2. Determine the extent of larch mortality since 2004 that could be attributed to bark beetles (e.g., LB).
3. Collect base forest health information and stand level detail to better assess larch stand establishment and regeneration success in interior Alaska.
4. Pin-point risk factors associated with repeated LSF defoliation and mortality impacts from that defoliation.

METHODS

GIS was used to analyze previously acquired data layers (primarily insect damage and vegetation cover from archived aerial survey data) to target and select a subset of stands containing a significant component of larch for ground survey and sampling. During July–August 2008 ground sampling was completed in seven road-accessible stands along the Chena and Tanana River floodplains in Fairbanks, north of Fairbanks (Chena Hot Springs), and within the Bonanza Creek Experimental Forest along the Tanana River west of Fairbanks, AK (fig. 14.1). During August 2009, we visited seven additional larch stands by floatplane access into lakes near the study sites (fig. 14.2). The GIS analysis helped to concentrate sampling in areas with a significant component of larch to maximize sampling time on the ground.

In each stand, tree species, including the diameter at breast height (d.b.h.) of all trees down to 1 inch was recorded within a 16.5-foot wide (1/4 chain) transect through the site. A regeneration plot was established every fifth chain for all tree species and significant shrubs, i.e., willow and alder, under 1-inch in diameter (table 14.1). Presence of cones was noted for live larch. Dead larch and spruce and LB mortality evidence was also tallied (table 14.1) (figs. 14.1 and 14.2). Dead larch killed by fire scorching, blown or fallen over, were not recorded, i.e., were assumed to predate the recent sawfly disturbance event. Other mortality causal agents, e.g., root disease, causal agents other than LB, were not recorded due to time and cost constraints. Recent evidence of fire or LSF defoliation was noted as well as general stand and site characteristics (species composition, drainage, soils, predominant ground cover, and the like). Basal stem discs from dominant or codominant spruce (healthy) and larch (dead and healthy) were taken on all sites in 2008 and 2009 to estimate stand age. For analysis, the 14 sites evaluated in this study were divided into two distinct groups (table 14.1) (fig. 14.3) based upon overstory species composition.

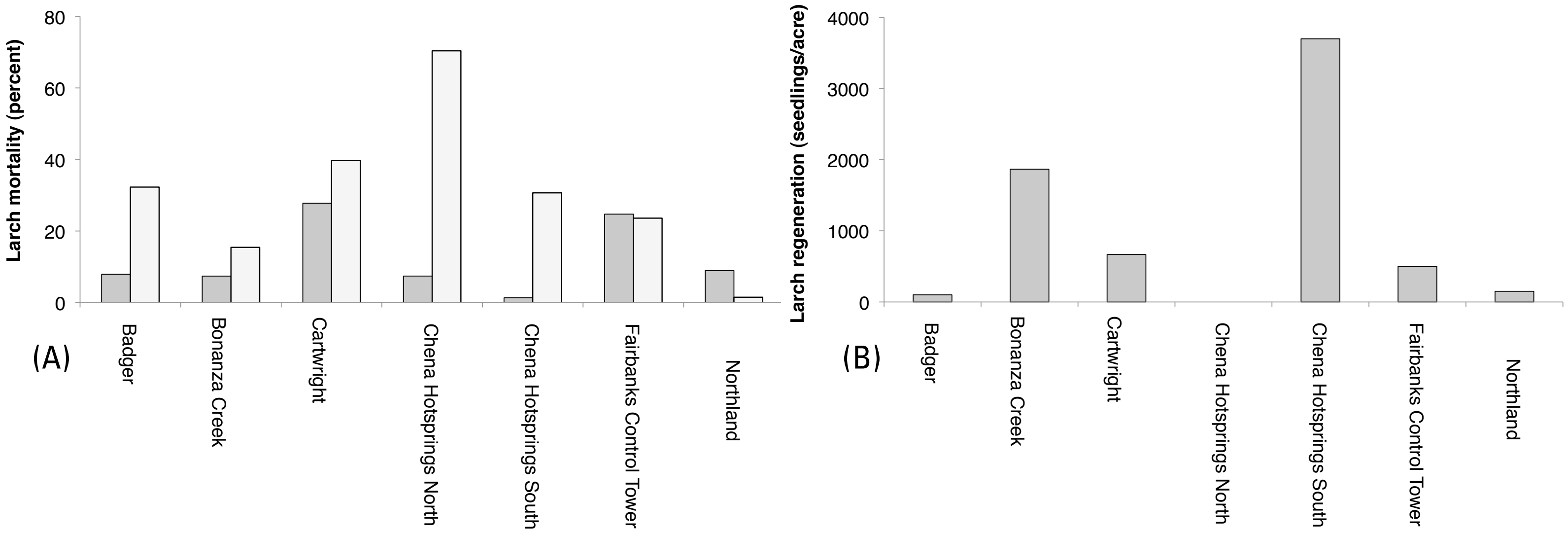

Figure 14.1—Data collected at seven road-accessible sites between Fairbanks and North Pole (interior Alaska) from July to August 2008: (A) percent mortality associated with larch beetle attacks (light gray bars) and "other" mortality which include disease, fire, etc. (dark gray bars); (B) larch regeneration estimates.

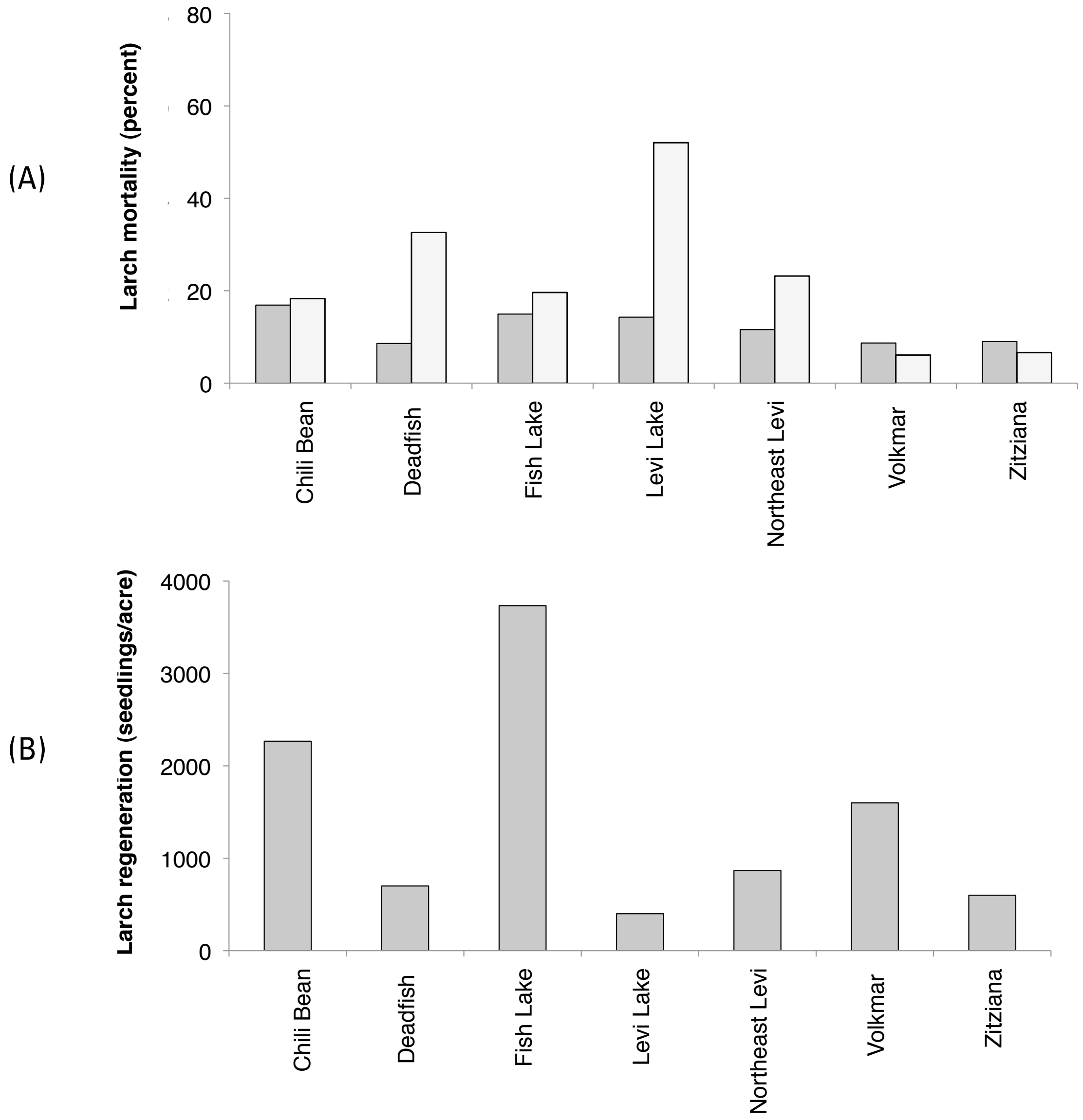

Figure 14.2—Data collected at seven remote sites in interior Alaska during August 2009: (A) mortality associated with larch beetle attacks (light gray bars) and "other" mortality which include disease, fire, etc. (dark gray bars); (B) larch regeneration estimates.

Table 14.1—Larch regeneration estimates, larch beetle mortality, and general stand characteristics averaged along site transects within each productivity class

Group 1 hydric sites	Average size class	Total stems/ acre	Larch stems/ acre	Spruce stems/ acre	Birch stems/ acre	Cottonwood stems/acre	Willow/ alder stems/acre	Dead larch/ acre	Larch dead from beetles/ acre	Live larch age	Larch seedlings/ acre
Chili Bean Lake	1.4	2,987	379	2,608	0	0	0	67	32	48	2,267
NE of Levi Lake	1.5	5,269	368	4,901	0	0	0	64	21	45	867
Bonanza Creek (Tanana R. floodplain)	1.8	1,973	1,659	219	11	0	85	189	61	39	1,867
Deadfish Lake	1.5	3,896	2,232	1,616	16	0	32	460	96	32	700
Fish Lake	1.4	768	571	176	21	0	0	99	43	26	3,733
Chena Hot Springs Road 2 (Fbks)	1.5	3,744	600	3,120	24	0	0	96	4	70	3,700
Northland Wood (Fbks)	1.5	3,488	918	2,122	36	0	412	14	12	35	150
Average	**1.5**	3,160.8	960.9	2,108.9	**15.4**	**0**	75.6	141.2	38.5	42.1	**1,897.6**

Group 2 mesic sites	Average size class	Total stems/ acre	Larch stems/ acre	Spruce stems/ acre	Birch stems/ acre	Cottonwood stems/acre	Willow/ alder stems/acre	Dead larch/ acre	Larch dead from beetles/ acre	Live larch age	Larch seedlings/ acre
Badger Road (North Pole, AK)	1.5	4,688	2,528	1,144	80	48	888	508	100	66	100
Levi Lake	1.3	4,896	784	4,016	96	0	0	260	56	28	400
Airport 2 Cartwright Road (Fbks)	1.8	2,891	672	2,064	133	0	21	227	93	57	667
Zitziana River	1.8	2,128	885	1,109	133	0	0	91	40	28	600
Chena Hot Springs Road 1 (Fbks)	2.3	3,867	240	3,476	151	0	0	93	9	61	0
Volkmar Lake	1.9	848	613	59	176	0	0	45	27	79	1,600
Airport 1 (near Fbks Int'l Airport Control Tower)	2.0	2,952	712	1,920	224	48	48	172	88	25	500
Average	**1.8**	3,181.3	919.2	1,969.7	**142.0**	**13.7**	136.8	199.4	59.0	49.1	**552.4**

Note: Averages shown in bold represent differences between the two groups greater than one standard error.

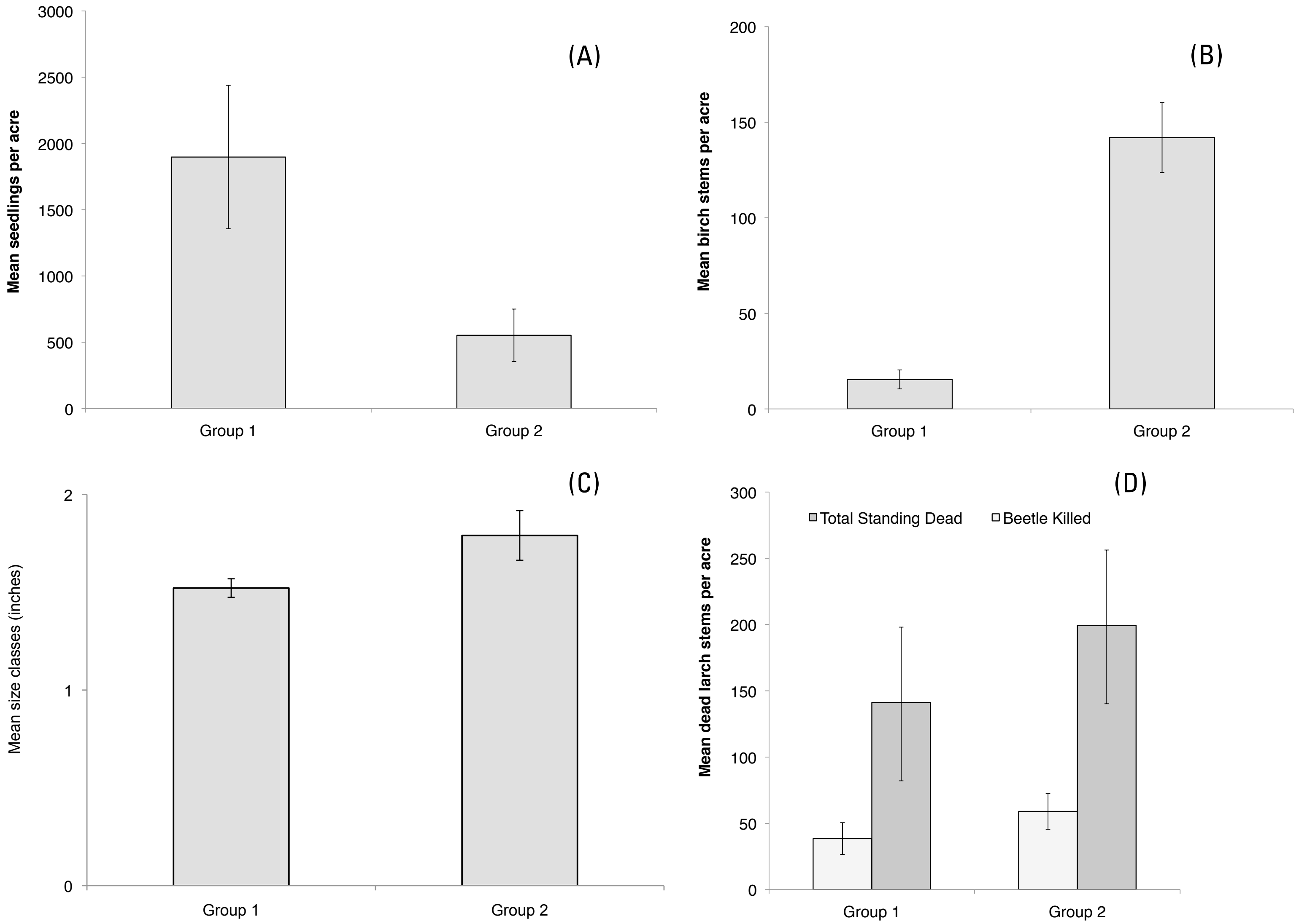

Figure 14.3—Stand characteristics of study sites grouped in table 14.1. (A) larch regeneration (B) number of birch trees (C) average size class, and (D) number of dead larch trees. Group 1 represents lower productive (i.e. more hydric) sites, while Group 2 represents the higher productive (i.e. more mesic) sites (+/- SE).

RESULTS AND DISCUSSION

The first group of larch stands was on more hydric sites with lower productivity and few, if any, hardwoods. The second group of sites are characterized as more mesic, with higher productivity and a substantial component of birch (*Betula neoalaskana*), and to a significantly lesser degree, balsam poplar (*Populus balsamifera*) (also known as "cottonwood"). Mesic sites contained 10 times more birch per acre on average than hydric sites. Both groups had about 3,000 stems per acre, and the average stem diameter on mesic sites was 20 percent larger than on hydric sites. Data from the 14 study sites suggest a strong, inverse relationship between site productivity (i.e., relative productivity of competing hardwood species such as birch) and larch regeneration (< 1 inch d.b.h.) (table 14.1) (fig. 14.3).

The occurrence of larch mortality between the two groups, and mortality from LB on the sites and component of dead larch examined was substantially greater on the mesic sites supporting birch trees (although this difference was statistically marginal due to large natural variation observed). These more mesic sites also had significantly fewer larch seedlings on a per-acre basis than the more hydric sites. Higher productivity of competitor species could make larch, a poor competitor for light and nutrients, more susceptible to disturbance agents (LSF and LB) on better drained sites. This could be attributed to larch's inability to compete with other species on undisturbed sites (Girardin and others 2002, Jardon and others 1994, Johnston 1990) and the increased likelihood of mortality observed in larch growing on mineral soil following insect attacks (Beckwith and Drooz 1956, Girardin and others 2002). Mortality of larch over the 14 sites (and transects sampled) attributable to the LB is also higher on mesic sites, i.e., 5 percent or more birch composition, compared to hydric sites, i.e., < 5 percent birch composition (table 14.1) (fig. 14.3).

While the number of sampling sites is likely not large enough to be able to make any definitive conclusions about the primary cause of larch mortality or regeneration success, comparisons can be made among the various sites. These results should provide forest health professionals with some initial clues on the productivity of the Alaska larch sites.

Larch regeneration, establishment, growth, and mortality are both directly and indirectly related to larch stand disturbances, including insect outbreaks. The current project was designed to conduct a point-in-time sampling of larch stocking levels, estimate general stand characteristics of larch stands, determine larch regeneration potential (existing stocking, presence/absence of cones), and collect data on its primary insect mortality agent, i.e., larch beetle, in the area of a recent landscape-level outbreak of LSF.

The recent landscape-level LSF outbreak from 1999 to 2004 (associated with equally dramatic mortality in 80 percent of LSF-impacted stands) as well as dramatic evidence of landscape-level changes in vegetation cover and diversity, changes in continuity of permafrost layers, and other landform changes resulting from climate change, illustrate potential threats to the sustainability of the forests in interior Alaska. This has also led to concerns about the genetic conservation of native larch in Alaska. The current project successfully determined larch stocking levels, larch regeneration potential (existing stocking, presence/absence of cones), and primarily insect mortality agents, e.g., larch beetle, in the area of a recent landscape-level LSF outbreak.

ACKNOWLEDGMENTS

Thanks to Hans Buchholdt, GIS Specialist from the Alaska Department of Natural Resources, Division of Forestry, Forest Health Protection Program, Anchorage, and Chuck Frank from the Forest Service, U.S. Department of Agriculture (detailer filling in for Interior Forest Health Unit in Fairbanks) for assistance with this project. Also, thanks to Andy Greenblatt, Shadow Aviation Inc., in Fairbanks, for floatplane transport and assisting with data collection and other plot work to the remote interior sites in 2009. This project was funded by an Evaluation Monitoring grant from the Forest Health Monitoring Program of the Forest Service, U.S. Department of Agriculture.

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CHAPTER 15.

Bugs and Burns: Effects of Fire on Ponderosa Pine Bark Beetle (Project INT-F-07-02)

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INTRODUCTION

Fire-damaged trees that otherwise would have survived can be killed by bark beetles (McCullough and others 1998, McHugh and others 2003). Wallin and others (2008) found that fire weakens a tree's defense against bark beetles. An unacceptable level of tree mortality may occur after a controlled burn as a result of weakened tree defenses (Sullivan and others 2003). Breece (2006) monitored tree mortality in the Birds and Burns Network sites (coordinated by the U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station) for the first three growing seasons (2004–06) after experimental implementation of prescribed fire treatments. Part 1 of this study continues to monitor these sites for an additional 3 years. In part 2, we set up additional sites in the Kaibab National Forest to monitor pre-burn bark beetle populations and stand conditions. This portion of the study provides us with data prior to the burn cycle to identify potential indicators of future high levels of bark beetle-caused mortality.

METHODS AND RESULTS

Part 1

Study sites are part of the Birds and Burns Network (www.rmrs.nau.edu/lab/4251/birdsnburns) and are in the Kaibab, Coconino, and Apache-Sitgreaves national forests in Arizona, and in the Gila National Forest in New Mexico. Sites were treated with prescribed fire in 2004 and measured in 2004, 2005, and 2006, and will be measured in 2007, 2008, and 2009, as follows:

Stand characteristics—Stands range from ponderosa pine dominated to mixed conifer. Each 250- to 400-ha stand is paired with an unburned control site of similar size and stand structure. Burns were conducted in the fall of 2003 or spring of 2004. Each treatment and control stand has a permanent sampling grid of 25 to 40 sampling stations.

Standard tree measurements—A total of 994 ponderosa pines on burned sites and 1,097 ponderosa pines on unburned control sites were sampled. Other non-dominant species, such as Douglas-fir, Gambel oak, and alligator juniper, were also sampled. At each sampling station, a 10-m radius circular permanent plot (0.03 ha) was established. Within each plot, all trees > 13 cm diameter at breast height (d.b.h.) were tagged and measured for tree species, d.b.h., tree height, length of live crown, and live crown ratio.

Assessment of fire damage to trees—All trees at prescribed fire sites were measured for bole char severity, char height and direction, percent of the bole circumference charred, percent of crown scorched by fire, percent of crown consumed by fire, length of the pre-fire live crown and the percent of the crown volume with green needles, and needles that were black or consumed by the fire (McHugh and Kolb 2003).

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Bark beetle sampling—Each spring and fall of 2004–06, each sample tree was examined for mortality and bark beetle activity. We assigned each tree an insect attack rating (IAR). If a tree had an IAR of 1 or 2 (partial or mass attack), and 75 percent or more of the crown was fading, a 30 x 30 cm section of the bark was cut out on the north side of the tree at heights of 1 m, 3 m, 5 m, and 7 m. Each bark sample was examined for bark beetles and galleries and, if possible, we identified the species of bark beetle making the galleries.

Statistical analysis—We transformed the mortality data using the arc-sine transformation ($2*\arcsin\sqrt{Pi}$) to account for differences in variance between sites. We performed a repeated measures MANOVA to look at the interaction between time, treatment, and the number of trees killed from bark beetles. We also ran t-tests to determine the significance of tree characteristics and fire damage characteristics on bark beetle mortality rates.

Surveys for bark beetle-related mortality were completed in the summer of 2009. Significantly higher rates of bark beetle mortality were observed in the burned sites (fig. 15.1, $F = 73.3130$, $df = 1$, $p = 0.0033$), with little to no mortality observed in the unburned control sites. Starting in 2007, however, mortality from bark beetle attacks dropped off and mortality rates in burned plots became similar to those observed in unburned plots. By 2009, bark beetle mortality in plots for both burned and unburned plots were not significantly different ($F = 0.372$, $df = 1$, $p = 0.5850$). These results suggest that bark beetle-related mortality in burned plots does drop off to background rates a few years following burning. The most common species that remained in these experimental plots were the roundheaded pine beetle (*Dendroctonus adjunctus* Blandford), western pine beetle (*D. brevicomis*), and *Ips knausi*.

When examining tree fire damage, mortality was highly related to higher percentages of crown scorch ($t = -8.4024$, $p < 0.0001$), crown consumption ($t = -4.3626$, $p < 0.0001$), total crown damage ($t = -11.1188$, $p < 0.0001$), and higher bole char ratings ($t = -10.3813$, $p < 0.0001$). These results suggest that trees experiencing higher levels of fire damage may be more susceptible to bark beetle attack following prescribed burning.

Part 2

Four stands were located in the Kaibab and Coconino national forests of Arizona as areas scheduled to be burned along with unburned paired stands. The “standard tree measurements,” as described above, were recorded prior to burning. We compared the tree measurements and fire damage seen at these new stands to the stands used in part 1 to determine any significant differences between stands that may contribute to differences in results using a one-way ANOVA. Two sets of three Lindgren funnel traps were placed within the control and treatment at the four stands. Each trap was baited with a different combination of lures targeting *Ips pini*, *Ips lecontei*, and *Dendroctonus brevicomis*. The lures deployed will catch several species of predators of the targeted bark species. We will analyze trap catches to determine if there was a significant change in ratio of predators to bark beetles before and after burning. Traps were deployed July 9, 2007, to August 20, 2007, prior to burning, and then again during the same time period in 2008, after burning. Trap data was log transformed [$\log_{10}(N + 1)$] to correct for heteroscedasticity. We ran a repeated measures MANOVA to look at interactions between treatments and species caught between 2007 and 2008.

There were no significant differences in tree basal area, live crown, or tree diameter between control and treatment plots in the new study stands. The new study stands differed from those used in Breece (2006) by having lower live crown ratios (F = 160.36, df = 1, $p < 0.0001$). Unfortunately, due to budget and time constraints, only two of the stands were burned during the study: Sitgreaves and Tusayan. Compared to the stands used in the Breece (2006) study, the new study stands had moderate fire damage with crown damage and bole char.

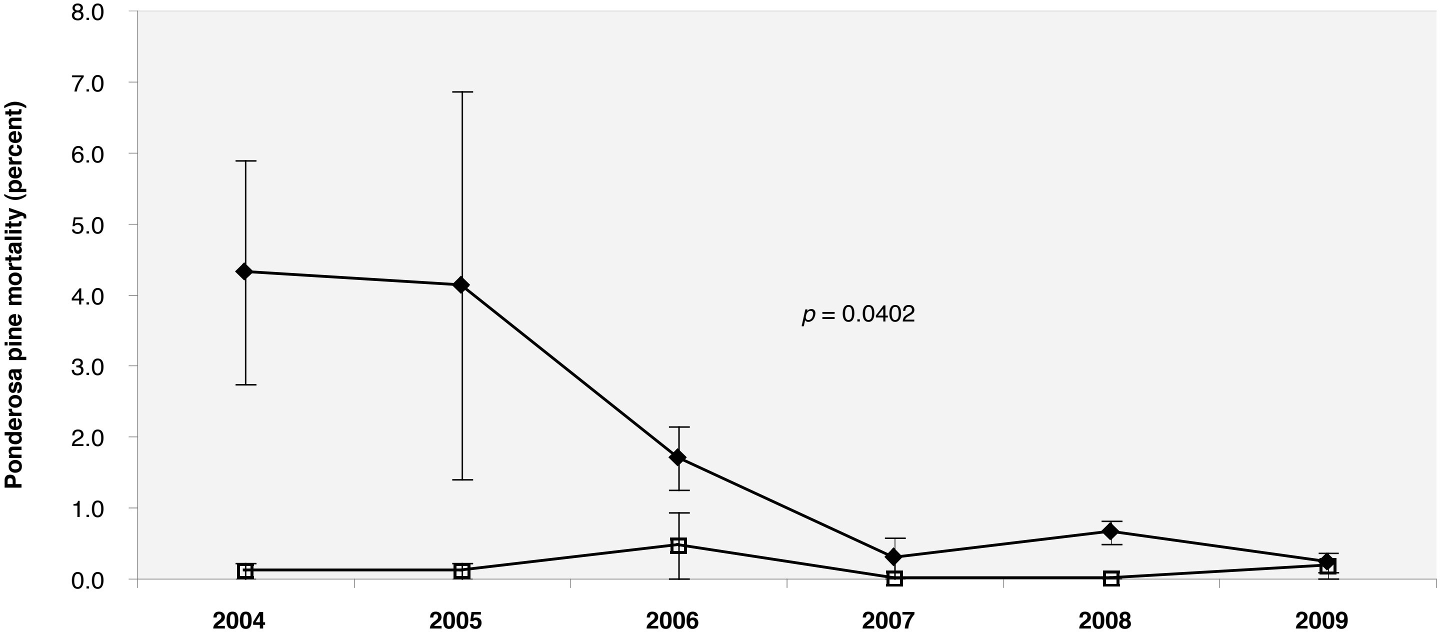

Figure 15.1—The proportion of ponderosa pine trees killed by bark beetles in the experimental plots used in Breece (2006). The p*-value compares trends in bark beetle mortality in burned (diamonds) and unburned stands (open squares).*

However, the difference in fire damage was not significantly different to the fire damage found in the stands used in part 1.

Over the 6-week trapping period prior to burning, relatively few beetles were caught (see table 15.1, burned = 80 beetles, unburned = 66 beetles). Prescribed burning was performed at Sitgreaves and Tusayan in the fall of 2007. In 2008, there were significant increases in the number of Ips beetles regardless of treatment, especially for *I. pini* beetles, while the number of all *Dendroctonus* species trapped declined. However, the number of *D. valens* increased in control and burned stands. The total numbers of beetles caught increased significantly in both burned stands between 2007 and 2008 (F = 8.4022, df = 1, $p = 0.0096$). However, there were no significant increases in the populations of any specific species including predator to prey ratios (table 15.1) that resulted from prescribed burning (F = 2.4726, df = 8, $p = 0.0528$).

Very little mortality from bark beetle attacks was observed at the new study sites following the burning treatments. Seven trees died during the course of the study, with five of those trees dying as a result of bark beetle activity. Those five trees were located in the Sitgreaves burned site. Upon running a one-way ANOVA, this level of

Table 15.1—Trap catch totals by species at two trapping locations in Arizona, for the trapping period from July to August in 2007 (prior to burning) and 2008 (after burning) to determine beetle populations in response to prescribed burning treatments

	Treatment	Year	*I. pini*	*I. lecontei*	*I. calligraphus*	*I. latidens*	*D. frontalis*	*D. brevicomis*	*D. valens*	*D. approximatus*	*Temnochila*	*E. sphegus*	*E. lecontei*	*Cerembycidae*	*Buprestidae*	*Predator total*	*Beetle total*	*Predator to prey*
Sitgreaves	Burned	2007	14	1	6	0	7	2	8	17	164	8	16	20	12	188	55	3.42
		2008	58	4	5	0	2	0	15	4	225	75	153	47	84	453	88	5.15
	Control	2007	11	4	4	1	1	0	10	7	307	10	26	17	14	343	38	9.03
		2008	12	1	0	2	5	0	13	4	173	45	4	26	58	222	37	6.00
Tusayan	Burned	2007	12	0	4	0	1	7	1	0	11	0	0	4	1	11	25	0.44
		2008	208	0	6	0	1	0	10	0	9	2	3	3	11	14	225	0.06
	Control	2007	17	0	4	0	1	1	4	1	26	0	5	5	1	31	28	1.11
		2008	38	0	1	0	3	0	1	0	10	0	0	0	8	10	43	0.23

mortality does not suggest a significant increase in bark beetle-related mortality in the burned stands (fig. 15.2, F = 1, df = 1, $p = 0.4226$). Unlike bark beetle-attacked trees at the Bugs and Burns sites, trees that were attacked at the newer sites were lower in basal area, d.b.h., and height. However, the low mortality rate makes it hard to definitively say that such tree characteristics are indicators of bark beetle mortality for areas treated with prescribed burning. Similar to the trees in the Bugs and Burns study, attacked trees had higher crown scorch and bole char. Higher than average precipitation during the summer months, lower bark beetle population sizes prior to burning, and the seasonal timing of burning treatments in the area might also have attributed to the lower levels of bark beetle mortality observed at these sites during the course of the study.

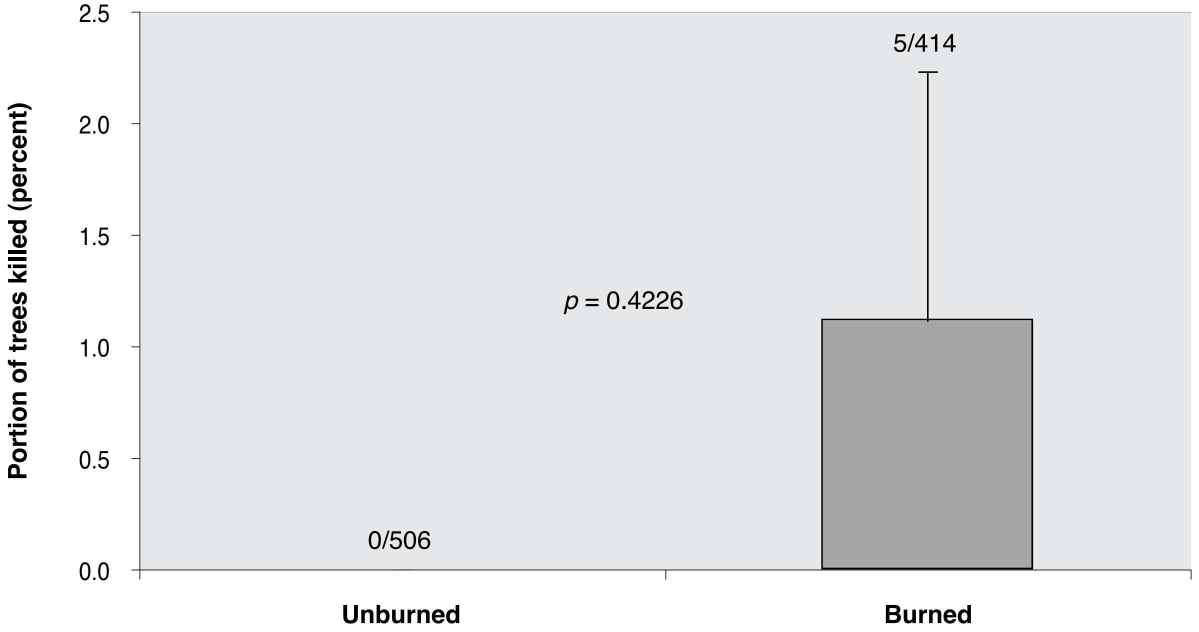

*Figure 15.2—The proportion of ponderosa pine trees killed by bark beetles (*Dendroctonus *and* Ips *spp. pooled) in burned and unburned stands in the Coconino National Forest in Arizona over two post-fire growing seasons (2008–09). Numbers above the bars are the total number of trees killed out of the total number of surveyed trees. The* p*-value compares the proportion of bark beetle-killed trees on burned and unburned stands.*

DISCUSSION AND RECOMMENDATIONS

The results from both the Bugs and Burns study sites and those located in the Kaibab National Forest indicate that fire damage could serve as an important indicator of bark beetle-related mortality in prescribed burn sites. Trees that died from bark beetle attacks in both studies had high levels of crown scorch (87 percent in the Coconino plots and 43.6 percent in the Bugs and Burns plots) and higher bole char ratings (1.6 for both studies). Although few trees died during the course of the study in the Kaibab National Forest, those that were killed as a result of bark beetle attacks showed higher levels of fire damage. Despite the sample size constraints of this study, the results support the findings of previous work, which found that fire damage was positively related to bark beetle mortality in fire-impacted stands (Breece and others 2008, McHugh and Kolb 2003, McHugh and others 2003, Sieg and others 2006).

Bark beetle population size may also be an important indicator of the magnitude of bark beetle attacks after a fire. The low mortality in the Kaibab experimental sites could be due to having smaller beetle populations in the area prior to burning. Previous studies have suggested similar conclusions (Bentz and others 1993, DeNitto and others 2000, Jenkins and others 2008, Santoro and others 2001). While there was a rise in the number of bark beetles present within the treatment stands following burning, the damage was minimal and not significantly higher than mortality witnessed in unburned stands. However, further studies that investigate bark beetle responses with varying sizes of bark beetle populations are needed.

FUTURE RECOMMENDATIONS

- Long-term monitoring on bark beetle populations for a forest are needed to determine if population abundance is at low or high levels.
- Because only two of the four experimental sites were treated with prescribed burning, it is hard to investigate the connection between the burning treatments and bark beetle mortality within the Kaibab experimental plots. A repeat of the study would be beneficial to continue to explore the connection between bark beetle mortality and fire damage.
- Further studies are needed to investigate the relationship between the size of pre-burn bark beetle populations and resulting mortality after a burn. The methods used in this study can provide a good model for future investigations, but should include more study sites and more variation in bark beetle population abundance and diversity.

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ACKNOWLEDGMENTS

The research described in this report was supported in part through the project "Forest Health Monitoring, Assessment and Analysis" of Research Joint Venture Agreement 09-JV-11330146-087 (July 22, 2009, through March 21, 2011), the project "Forest Health Monitoring, Analysis and Assessment" of Research Joint Venture Agreement 10-JV-11330146-064 (June 10, 2010 through February 28, 2012), and the project "Forest Health Monitoring, Assessment, and Analysis" of Research Joint Venture Agreement 11-JV-11330146-090 (July 12, 2011, through May 31, 2013) between North Carolina State University (this institution is an equal opportunity provider) and the U.S. Department of Agriculture, Forest Service, Southern Research Station, Asheville, NC. This research was supported by funds provided by the U.S. Department of Agriculture Forest Service, Southern Research Station, Asheville, NC.

The editors and authors of this report thank the following for their reviews, constructive comments, and advice: Chris Asaro, Mark Ambrose, Tom Hall, Frank Koch, Manfred Mielke, KaDonna Randolph, Kurt Riitters, Keith Sprengel, Dale Starkey, Borys Tkacz, Chris Woodall, and Stan Zarnoch.

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Potter, Kevin M.; Conkling, Barbara L., eds. 2013. Forest Health Monitoring: national status, trends, and analysis 2010. Gen. Tech. Rep. SRS-176. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 164 p.

The annual national report of the Forest Health Monitoring Program of the Forest Service, U.S. Department of Agriculture, presents forest health status and trends from a national or multi-State regional perspective using a variety of sources, introduces new techniques for analyzing forest health data, and summarizes results of recently completed Evaluation Monitoring projects funded through the national Forest Health Monitoring Program. In this 10th edition in the annual series of national reports, survey data are used to identify geographic patterns of insect and disease activity. Satellite data are employed to detect geographic clusters of forest fire occurrence. Data collected by the Forest Inventory and Analysis Program of the Forest Service are employed to detect regional differences in tree mortality. Established forest fragmentation assessment protocols are used to characterize and compare the fragmentation of landcover types nationally. A new methodology for the comparison of moisture conditions among different geographical areas and time periods is described. Forest Inventory and Analysis data are used to conduct an empirical assessment of the Nation's standing dead tree resources. The potential impacts of climate change on forest soil critical acid load limits are explored. Seven recently completed Evaluation Monitoring projects are summarized, addressing forest health concerns at smaller scales.

Keywords—Drought, fire, forest health, forest insects and disease, fragmentation, nonnative species, tree mortality.

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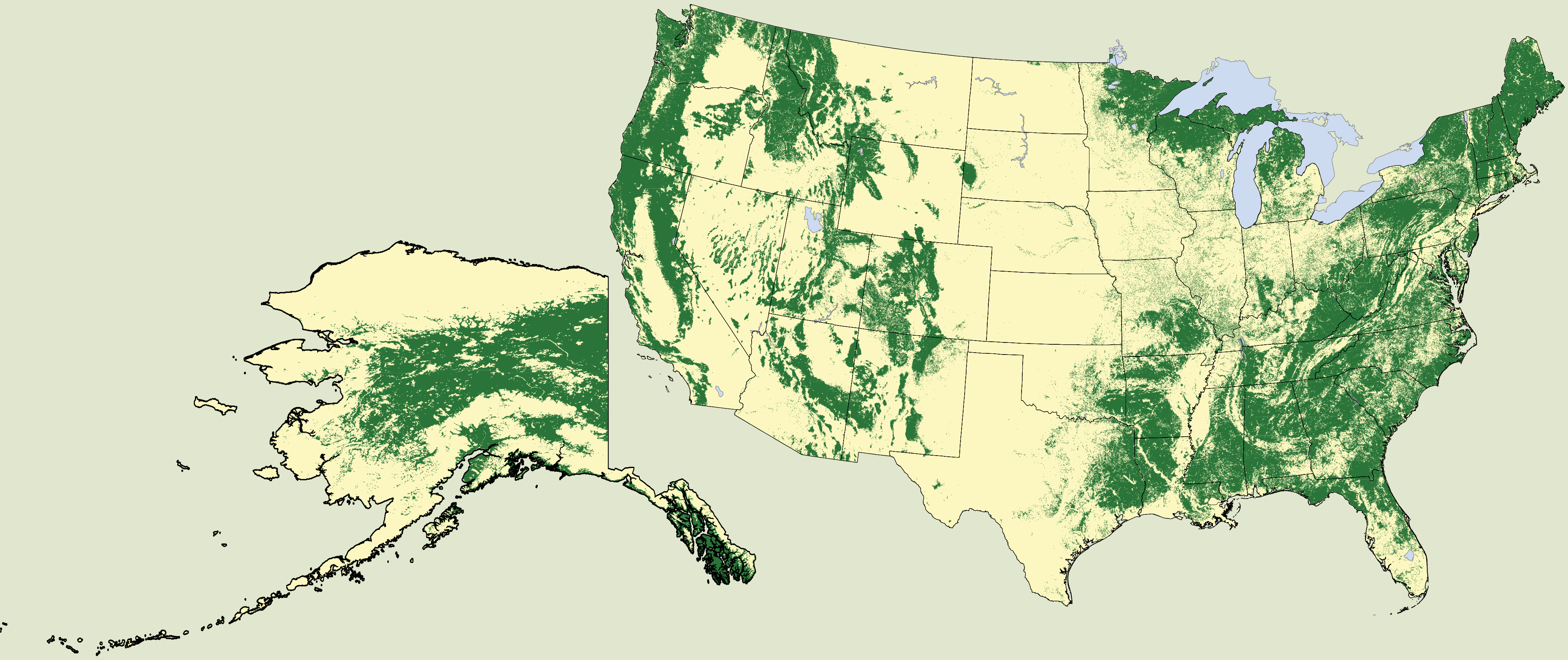

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