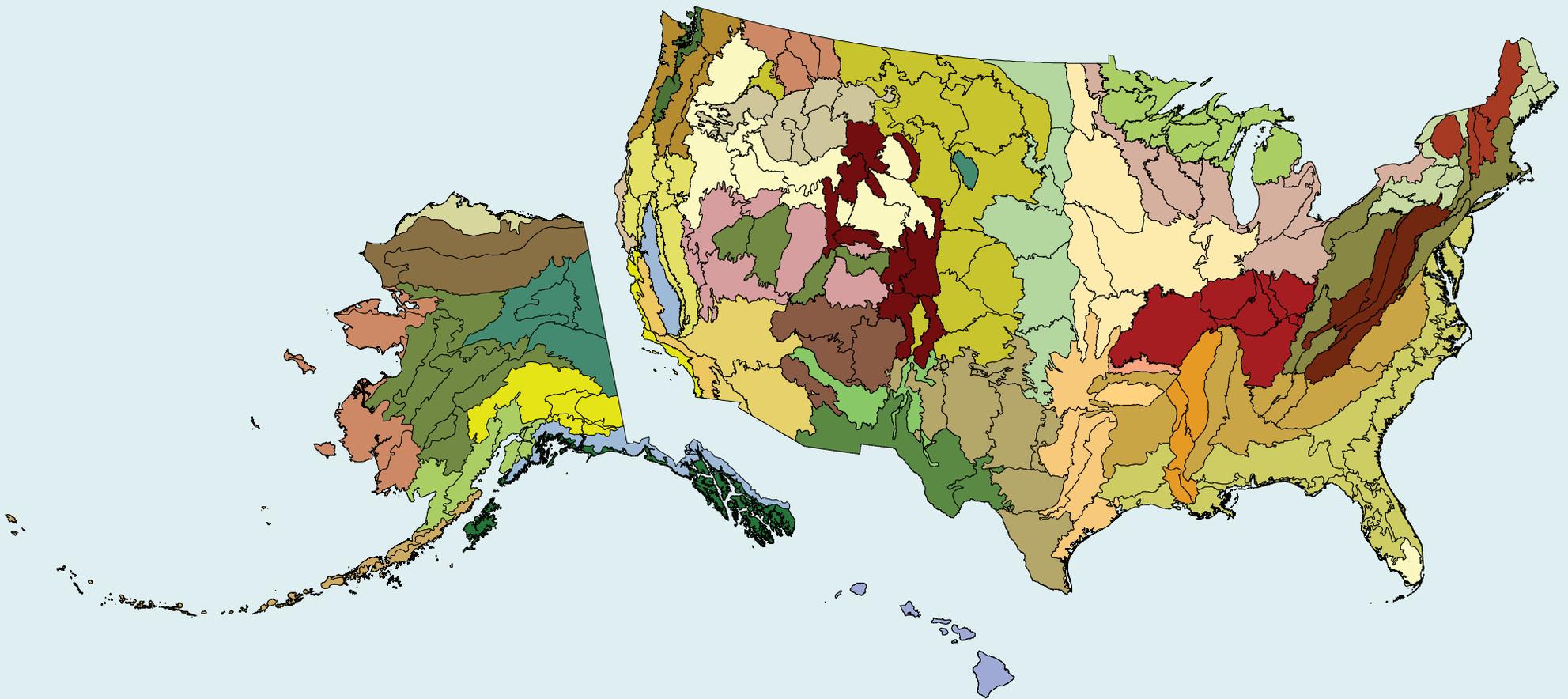


Forest Health Monitoring: National Status, Trends, and Analysis 2014

Editors Kevin M. Potter Barbara L. Conkling



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.

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Forest Health Monitoring: National Status, Trends, and Analysis 2014

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ABSTRACT

The annual national report of the Forest Health Monitoring (FHM) 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 FHM national program. In this 14th edition in a series of annual reports, survey data are used to identify geographic patterns of forest insect and disease activity. Satellite data are employed to detect geographic patterns of forest fire occurrence. Recent drought conditions

are compared across the conterminous United States. Data collected by the Forest Inventory and Analysis (FIA) Program are employed to detect regional differences in tree mortality. Results of a national insect and disease forest risk assessment, including maps, are presented. Using FIA and national land cover data, decline of intact forest is assessed by forest type and ownership. Ten recently completed Evaluation Monitoring projects are summarized, addressing forest health concerns at smaller scales.

Keywords—Change detection, drought, fire, forest health, forest insects and disease, fragmentation, risk assessment, tree mortality.

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EXECUTIVE SUMMARY

Healthy ecosystems are those that are stable and sustainable, able to maintain their organization and autonomy over time while remaining resilient to stress (Costanza 1992). Healthy forests are vital to our future (Edmonds and others 2011), and consistent, large-scale, and long-term monitoring of key indicators of forest health status, change, and trends is necessary to identify forest resources deteriorating across large regions (Riitters and Tkacz 2004). This national report, the 14th in an annual series sponsored by the Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture, quantifies the status of, changes to, and trends in several broadly defined indicators of forest health. These indicators include, among others, forest insect and disease activity, wildland fire occurrence, drought, tree mortality, and forest fragmentation.

The national FHM Program is 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 that encompasses the Forest Service, State foresters, and other State and Federal agencies and academic groups (chapter 1). 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. 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 2013 data. For datasets collected over several years, analyses are presented at a longer temporal scale. Chapters describe new techniques for collecting and 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 2014).

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). National Insect and Disease Survey data collected in 2013 by the Forest Health Protection Program of the Forest Service and by partners in State agencies identified 73 different mortality-causing agents and complexes on 1.53 million ha in the conterminous United States, and 83 defoliating agents and complexes on approximately 2.94 million ha. Geographic hot spots of forest mortality were associated with bark beetle infestations in the West. Hot spots of defoliation were associated with fall cankerworm, baldcypress leafroller, forest tent caterpillar, and gypsy moth in the East, and with western spruce budworm, larch needle cast, and aspen

defoliation in the West. Mortality was recorded on a very small proportion of the surveyed area in Alaska and Hawaii. The most important defoliation agent was birch leafroller in Alaska and koa looper moth in Hawaii.

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 density 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 2013, more satellite-detected forest fire occurrences were recorded for the conterminous States than for all but one year (2012) since the beginning of data collection in 2001. Ecoregions in California, Idaho, and Oregon experienced the most fires per 100 km² of forested area. Geographic hot spots of high fire occurrence density were detected throughout the Interior West and in the Gulf Coast States of the South. Ecoregions in the Pacific West, Interior West, Great Lakes States, Northeast, and Southern States experienced greater fire occurrence density than normal compared to the 12-year mean and accounting for variability over time. Alaska experienced low fire occurrence density in 2013, except in the south-central areas of the State.

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. A standardized drought-indexing approach was applied to

monthly climate data from 2013 to map drought conditions across the conterminous United States at a fine scale (chapter 4). Much of the country experienced moisture surplus conditions. The most dramatic exception was the southern portion of the Pacific Coast, where unusually severe drought conditions existed. Moderate to somewhat worse drought conditions also occurred in neighboring areas of the West, although typically in areas with little or no forest, as well as in central Florida and in the vicinity of New York's Long Island and the Connecticut coast. Most of the conterminous United States displayed an increase in moisture difference z-scores from 2012 to 2013, indicating widespread recovery from drought conditions.

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 on a relatively spatially intense basis of approximately 1 plot per 6,000 acres (chapter 5). An analysis of FIA plots from 37 States found that the highest ratios of annual mortality to gross growth occurred in ecoregion sections located in western South Dakota and Nebraska, southern Kansas, eastern Louisiana/southeastern Arkansas, and southern Louisiana/southeastern Texas. In Plains ecoregions with the highest mortality relative to growth, tree growth is quite low, and most of the species experiencing the greatest mortality are commonly found in riparian areas. Two exceptions are ponderosa pine in multiple ecoregions and aspen in one.

Bottomland hardwood species experienced high mortality in the Southern Mississippi Alluvial Plain section, while the invasive Chinese tallowtree had the greatest mortality in the Louisiana Coastal Prairies and Marshes section.

The 2012 National Insect and Disease Risk Map (NIDRM) represents a nationwide strategic assessment of the hazard of tree mortality due to insects and diseases, displayed as a series of maps (chapter 6). NIDRM was developed within a highly collaborative process led by the FHM Program with participation from throughout the Forest Service and from State forestry agencies. It encompasses 186 individual insect and disease hazard models, integrated within a common Geographic Information System-based, multicriteria framework that can account for regional variations in forest health. Root diseases, bark beetles, and oak decline were the leading contributor to the risk of mortality in the conterminous United States, while spruce beetle was the most significant contributor in Alaska. The confluence of bark beetles and root diseases has resulted in large contiguous areas at risk across much of the Western United States. Emerald ash borer was the most significant exotic forest pest. Tree species with the potential to lose more than 50 percent of their host volume include redbay and whitebark pine.

Forest loss and the fragmentation of remaining forest threaten the sustainability of many ecological attributes and processes that depend on extensive forest cover (chapter 7). Direct loss of intact forest is an obvious threat; less obvious are the indirect threats posed by

isolation and edge effects, which encompass a wide range of biotic and abiotic influences on remnant forest. Forest maps from the National Land Cover Database (NLCD) were used, along with FIA data, to assess the degree of decline of intact forest as assessed by forest type and ownership between 2001 and 2006. The net loss of core forest during the period was 3.8 million ha, representing 3.9 percent of the core area in 2001 and reducing total core area to 94.6 million ha (39 percent of total forest land area) in 2006. Changes in core area were driven primarily by forest cover changes on private lands. Meanwhile, most forest types exhibited a net loss of core area. Among the 94 forest types with a net loss, the percentage of core forest that was lost ranged from near zero to 31 percent; 35 forest types exhibited net losses larger than 5 percent.

Finally, 10 recently completed Evaluation Monitoring project summaries 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):

- Identified and tested American beech trees that may be resistant to the insect that spreads beech bark disease (chapter 8)
- Investigated potential causes of tree mortality recently detected in Vermont, and identified site conditions that contributed to this mortality (chapter 9)
- Established long-term monitoring plots in the Upper Great Lakes region to characterize

changes in sugar maple dieback symptoms and to investigate relationships between dieback and ownership (chapter 10)

- Evaluated the impacts of beech bark disease in Michigan by assessing overstory composition, size, and condition; species composition of regeneration; and coarse woody material (chapter 11)
- Used LiDAR to evaluate forest landscape and health factors in the Pinaleno Mountains of Arizona, and assessed their relationship to habitat of the endangered Mount Graham red squirrel (chapter 12)
- Quantified and characterized the vegetation, surface wood, and soil characteristics associated with successive fire events within the Scapegoat Wilderness in Montana (chapter 13)
- Investigated factors contributing to shore pine mortality and damage in southeast Alaska, and described a permanent plot network installed to provide baseline information about the species' health (chapter 14)
- Evaluated the resiliency of ponderosa pine stands to bark beetle infestations a decade following fuel-reduction and forest-restoration treatments in California (chapter 15)
- Documented the long-term impacts of mountain pine beetle outbreaks on residual stand structure and composition in lodgepole pine forests of the Intermountain West (chapter 16)
- Described an assessment that generated maps of the risk of climate change-associated

extirpation and genetic degradation across the range of more than 300 North American tree species (chapter 17)

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 environmental monitoring 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. This Web site includes links to all past national forest health reports (www.fs.fed.us/foresthealth/fhm/pubs), information about funded Evaluation Monitoring projects (www.fs.fed.us/foresthealth/fhm/em), and annual State reports highlighting forest health issues (www.fs.fed.us/foresthealth/fhm/fhh/fhmusamap.shtml).

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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 the capacity to provide a broad range of goods and services to current and future generations, to safeguard biological diversity, and to contribute to the resilience of ecosystems, societies, and economies (USDA Forest Service 2011). Their ecological roles include supplying large and consistent quantities of clean water, preventing soil erosion, and providing habitat for a broad diversity of plant and animal species. Their socioeconomic benefits include wood products, nontimber goods, recreational opportunities, and pleasing natural beauty. Both the ecological integrity and the continued capacity of these forests to provide ecological and economic goods and services are of concern, however, 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.

Natural and anthropogenic stresses vary among biophysical regions and local environments; they also change over time and interact with each other. These and other factors make it challenging to establish baselines of forest health and to detect important departures from normal forest ecosystem functioning (Riitters and Tkacz 2004). Monitoring the health of forests is a critically important task, however, reflected within the Criteria and Indicators for the Conservation and Sustainable Management of Temperate and Boreal Forests (Montréal Process Working Group 1995), which the Forest Service, U.S. Department of Agriculture,

uses as a forest sustainability assessment framework (USDA Forest Service 2004, 2011). The primary objective of such monitoring is to identify ecological resources whose condition is deteriorating in subtle ways over large regions in response to cumulative stresses, which requires consistent, large-scale, and long-term monitoring of key indicators of forest health status, change, and trends (Riitters and Tkacz 2004). This is best accomplished through the participation of multiple Federal, State, academic, and private partners.

While the concept of a healthy forest has universal appeal, forest ecologists and managers have struggled with how exactly to define forest health (Teale and Castello 2011), and there is no universally accepted definition. Such a definition can help guide efforts to monitor and assess forest health, with which managers and policymakers have also struggled. Most definitions of forest health can be categorized as representing an ecological or a utilitarian perspective (Kolb and others 1994). From an ecological perspective, 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 (Edmonds and others 2011, Kolb and others 1994, Raffa and others 2009). On the other hand, the utilitarian perspective holds that a forest is healthy if management objectives are met, and that a forest is unhealthy if not (Kolb and others 1994). While this definition may be appropriate

CHAPTER 1.

Introduction

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when a single, unambiguous management objective exists, such as the production of wood fiber or the maintenance of wilderness attributes, it is too narrow when multiple management objectives are required (Edmonds and others 2011, Teale and Castello 2011). Teale and Castello (2011) incorporate both ecological and utilitarian perspectives into their two-component definition of forest health: First, a healthy forest must be sustainable with respect to its size structure, including a correspondence between baseline and observed mortality; and second, a healthy forest must meet the landowner's objectives, provided that these objectives do not conflict with sustainability.

This national report, the 14th in an annual series sponsored 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 broadly defined indicators of forest health. The indicators described in this report encompass forest insect and disease activity, wildland fire occurrence, drought, tree mortality, and fragmentation, among others. The previous reports in this series are Ambrose and Conkling (2007, 2009), Conkling (2011), Conkling and others (2005), Coulston and others (2005a, 2005b, 2005c), and Potter and Conkling (2012a, 2012b, 2013a, 2013b, 2014, 2015).

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. Such repeated analyses of regularly collected indicator measurements allow for the detection of trends over time and help establish a baseline for future comparisons (Riitters and Tkacz 2004). 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 presents the 2012 National Insect and Disease Risk Map, and chapter 7, which assesses the decline of intact forest by forest type and ownership using FIA and national land cover data.

The third objective of the report is to present results of recently completed Evaluation Monitoring (EM) projects funded through

the FHM national program. These project summaries, presented in section 3, determine the extent, severity, and/or cause of forest health problems (FHM 2014), generally at a finer scale than that addressed by the analyses in sections 1 and 2. Each of the 10 chapters in section 3 contains an overview of an EM project, key results, and contacts for more information.

When appropriate throughout this report, authors use the USDA Forest Service 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. Bailey's hierarchical system bases the two broadest ecoregion scales, domains and divisions, 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).

THE FOREST HEALTH MONITORING PROGRAM

The national FHM Program is 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 2014). 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. The FHM Program has five major components (fig. 1.2):

- 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).

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)

- 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).

The FHM Program, in addition to national reporting, generates regional and State reports, often in cooperation with FHM partners both within the Forest Service and in State forestry and agricultural departments. For example, the FHM regions cooperate with their respective State partners to produce the annual Forest Health Highlights report series, available on the FHM Web site at www.fs.fed.us/foresthealth/fhm. Other examples include Harris and others (2011) and Steinman (2004).

The FHM Program and its partners also produce reports and journal articles on monitoring techniques and analytical methods, including forest health data (Smith and Conkling 2004); soils as an indicator of forest health (O'Neill and others 2005); urban forest health monitoring (Bigsby and others 2014; Cumming and others 2006, 2007; Lake and others 2006); health conditions in National forests (Morin and others 2006); crown conditions (Randolph 2010a, 2010b, 2013; Randolph and Moser 2009; Schomaker and others 2007); vegetation diversity and structure (Schulz and Gray 2012, Schulz and others 2009); forest lichen communities (Jovan and others 2012, Root and others 2014); downed woody materials in forests (Woodall and others 2012, 2013); ozone monitoring (Rose

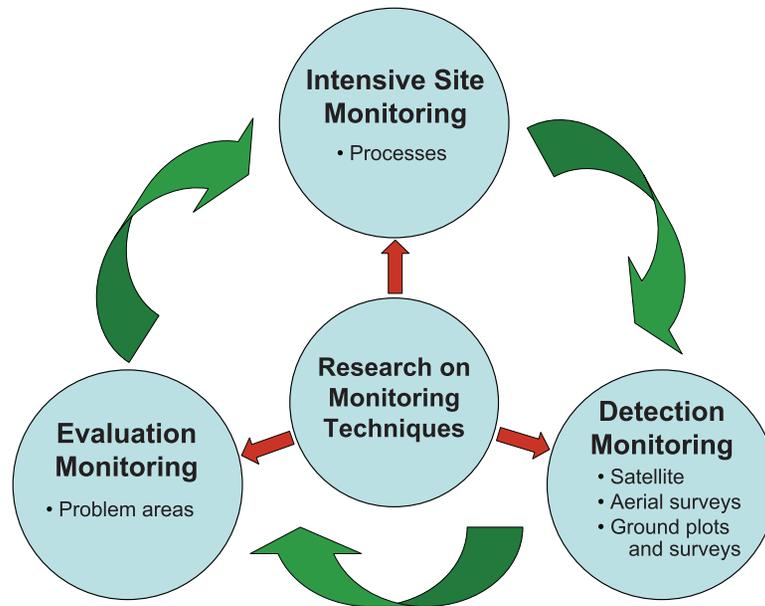


Figure 1.2—The design of the Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture (FHM 2003). A fifth component, Analysis and Reporting of Results, draws from the four FHM components shown here and provides information to help support land management policies and decisions.

and Coulston 2009); predicting establishment of alien-invasive forest insect species (Koch and others 2011, 2014; Yemshanov and others 2014); spatial patterns of land cover (Riitters 2011, Riitters and Wickham 2012, Riitters and others 2012); broad-scale assessments of forest biodiversity (Potter and Koch 2014; Potter and Woodall 2012, 2014); predictions of climate

change effects on forest tree species (Potter and Hargrove 2013); and the overall forest health indicator program (Woodall and others 2011).

For more information about the FHM Program, visit the FHM Web site at www.fs.fed.us/foresthealth/fhm. Among other things, this Web site includes links to all past national forest health reports (www.fs.fed.us/foresthealth/fhm/pubs), information about funded Evaluation Monitoring projects (www.fs.fed.us/foresthealth/fhm/em), and annual State Forest Health Highlights reports (www.fs.fed.us/foresthealth/fhm/fhh/fhmusamap.shtml).

DATA SOURCES

Forest Service data sources included in this edition of the FHM national report are: FIA annualized phase 2 and phase 3 survey data (Bechtold and Patterson 2005, Woodall and others 2010, Woudenberg and others 2010), FHP national Insect and Disease Survey forest mortality and defoliation data for 2013 (FHP 2014), Moderate Resolution Imaging Spectroradiometer (MODIS) Active Fire Detections for the United States database for 2013 (USDA Forest Service 2014), forest cover data developed from MODIS satellite imagery by the U.S. Forest Service Remote Sensing Applications Center, and 186 individual insect and disease hazard models developed by the Forest Health Technology Enterprise Team (FHTET). Other sources of data are the Parameter-elevation Regression on Independent Slopes Model (PRISM) climate mapping system data (Daly and Taylor 2000, PRISM Climate Group 2014) and

2001 and 2006 National Land Cover Database land cover maps (U.S. Geological Survey 2011a, 2011b).

As a major source of data for several FHM analyses, the FIA Program merits detailed description. The FIA Program collects forest inventory information across all forest land ownerships in the United States and 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 1 plot per 2428 ha. FIA phase 2 encompasses the annualized inventory measured on plots at regular intervals, with each plot surveyed every 5 to 7 years in most Eastern States, but with plots in the Rocky Mountain and Pacific Northwest regions surveyed once every 10 years (Reams and others 2005). The standard 0.067-ha plot (fig. 1.3) consists of four 7.315-m radius subplots (approximately 168.6 m² or 1/24th acre), on which field crews measure trees at least 12.7 cm in diameter. Within each of these subplots is nested a 2.073-m radius microplot (approximately 13.48 m² or 1/300th acre), on which crews measure trees smaller than 12.7 cm in diameter. A core-optional variant of the standard design includes four “macroplots,” each with radius of 17.953 m (or approximately 0.1012 ha) 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 1 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, while data on ozone bioindicator plants are collected on a separate grid of plots (Woodall and others 2010, 2011). Most of these additional forest health indicators were measured as part of the FHM Detection Monitoring ground plot system prior to 2000¹ (Palmer and others 1991).

FHM REPORT PRODUCTION

This FHM national report, the 14th in a series of such annual documents, is produced by forest health monitoring researchers at the Eastern Forest Environmental Threat Assessment Center (EFETAC) in collaboration with North Carolina State University cooperators. EFETAC, a unit of the Southern Research Station of the Forest Service, was established under the Healthy Forests Restoration Act of 2003 to generate the 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.

¹ U.S. Department of Agriculture Forest Service. 1998. Forest health monitoring 1998 field methods guide. Research Triangle Park, NC: U.S. Department of Agriculture Forest Service, Forest Health Monitoring Program, 473 p. On file with: Forest Health Monitoring Program, 3041 Cornwallis Rd., Research Triangle Park, NC 27709.

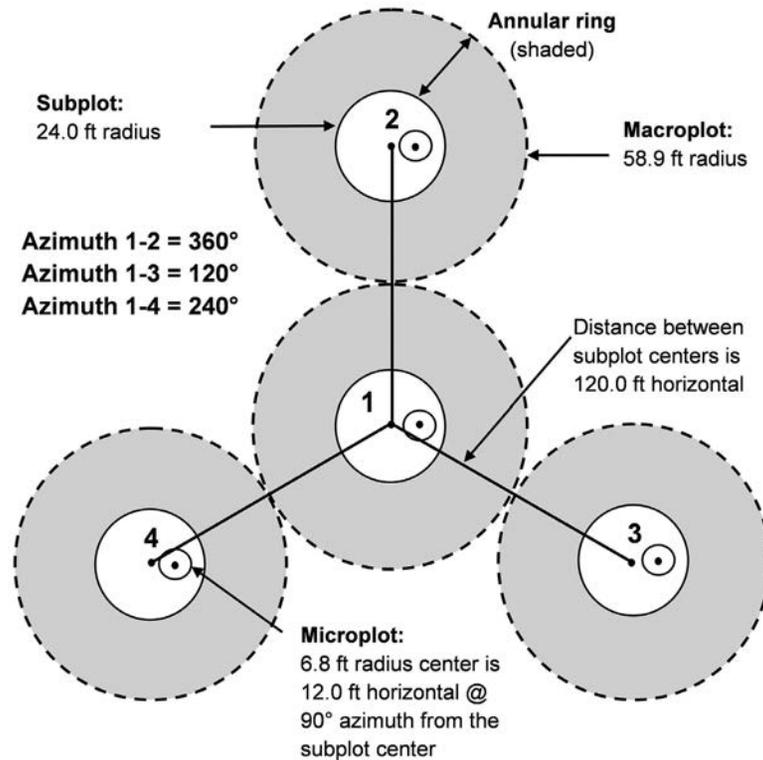


Figure 1.3—The Forest Inventory and Analysis Program mapped plot design. Subplot 1 is the center of the cluster with subplots 2, 3, and 4 located 120 feet away at azimuths of 360°, 120°, and 240°, respectively (Woudenberg and others 2010).

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SECTION 1.

Analyses of
Short-Term Forest
Health Data

INTRODUCTION

Diseases and insects cause changes in forest structure and function, species succession, and biodiversity, which may be considered negative or positive depending on management objectives (Edmonds and others 2011). An important task for forest managers, pathologists, and entomologists is recognizing and distinguishing between natural and excessive mortality, a task which relates to ecologically-based or commodity-based management objectives (Teale and Castello 2011). The impacts of insects and diseases on forests vary from natural thinning to extraordinary levels of tree mortality, but insects and diseases are not necessarily enemies of the forest because they kill trees (Teale and Castello 2011). If disturbances, including insects and diseases, are viewed in their full ecological context, then some amount can be considered “healthy” to sustain the structure of the forest (Manion 2003, Zhang and others 2011) by causing tree mortality that culls weak competitors and releases resources that are needed to support the growth of surviving trees (Teale and Castello 2011).

Analyzing patterns of forest insect infestations, 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). Few forests remain unaffected by invasive species, and their devastating impacts in forests are undeniable, including, in some cases, wholesale changes to the structure and function of an ecosystem (Parry and Teale 2011).

Examining insect 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 insect pest outbreaks (Holdenrieder and others 2004). One such landscape-scale approach is the detection of geographic patterns of disturbance, which allows for the identification of areas at greater risk of significant ecological and economic impacts and for the selection of locations for more intensive monitoring and analysis.

METHODS

Data

Forest Health Protection (FHP) national Insect and Disease Survey (IDS) data (FHP 2014) consist of information from low-altitude aerial survey and ground survey efforts by FHP and partners in State agencies. These data can be used to

CHAPTER 2.

Large-Scale Patterns of Insect and Disease Activity in the Conterminous United States, Alaska, and Hawaii from the National Insect and Disease Survey, 2013

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identify forest landscape-scale patterns associated with geographic hot spots of forest insect and disease activity in the conterminous 48 States and to summarize insect and disease activity by ecoregion in Alaska (Potter 2012; Potter 2013; Potter and Koch 2012; Potter and Paschke 2013, 2014, 2015). In 2013, IDS surveys covered about 152.48 million ha of the forested area in the conterminous United States (approximately 59.8 percent of the total), 8.09 million ha of Alaska's forested area (approximately 15.7 percent of the total), and about 666 000 ha of forest in Hawaii (approximately 14 percent of the total) (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 (*Agrilus planipennis*) and hemlock woolly adelgid (*Adelges tsugae*)], 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 forests with high tree-species diversity 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 other 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 complex” or “aspen defoliation”). Additionally, differences in data collection, attribute recognition, and coding procedures among States and regions can complicate data analysis and interpretation of the results.

The 2013 mortality and defoliation polygons were used to identify the select mortality and defoliation agents and complexes causing damage on more than 5000 ha of forest in the conterminous United States in that year, and to identify and list the most widely detected mortality and defoliation agents for Alaska and Hawaii. Because of the insect and disease aerial sketchmapping process, all quantities are approximate “footprint” areas for each agent or complex, delineating areas of visible damage within which the agent or complex is present. Unaffected trees may exist within the footprint, and the amount of damage within the footprint is not reflected in the estimates of forest area affected. 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.

Analyses

A Getis-Ord hot spot analysis (Getis and Ord 1992) was employed in ArcMap® 10.1 (ESRI 2012) to identify surveyed forest areas with the greatest exposure to the detected mortality-causing and defoliation-causing agents and complexes. The units of analysis were 9,810 hexagonal cells, each approximately 834 km² in area, generated in a

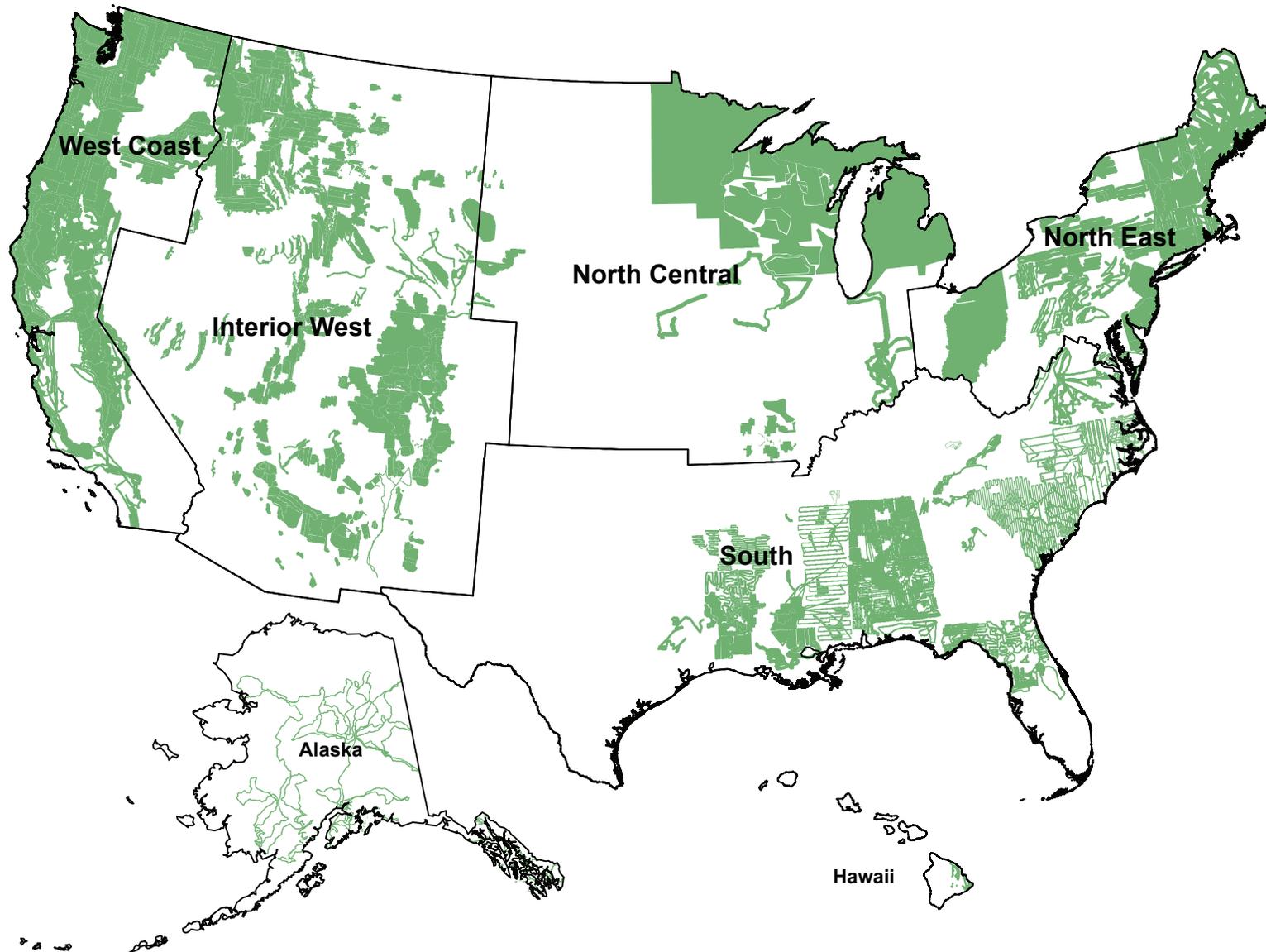


Figure 2.1—The extent of surveys for insect and disease activity conducted in the conterminous United States, Hawaii, and Alaska in 2013. The black lines delineate Forest Health Monitoring regions. Note: Alaska and Hawaii are not shown to scale with the conterminous United States. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection)

lattice across the conterminous United States using intensification of the Environmental Monitoring and Assessment Program (EMAP) North American hexagon coordinates (White and others 1992). The variable used in the hot spot analysis was the percentage of surveyed forest area in each hexagon exposed to either mortality-causing or defoliation-causing agents. This required first separately dissolving the mortality and defoliation polygon boundaries to generate an overall footprint of each general type of disturbance, then masking the dissolved polygons using a forest cover map (1-km² resolution) derived from Moderate Resolution Imaging Spectroradiometer (MODIS) satellite imagery by the U.S. Forest Service Remote Sensing Applications Center (USDA Forest Service 2008). The same process was undertaken with the polygons of the surveyed area. Finally, the percentage of surveyed forest exposed to mortality or defoliation agents was calculated by dividing the total forest-masked damage area by the forest-masked surveyed area.

The Getis-Ord G_i^* statistic was used to identify clusters of hexagonal cells in which the percentage of surveyed forest exposed to mortality or defoliation agents was higher than expected by chance. This statistic allows for the decomposition of a global measure of spatial association into its contributing factors by location, and is therefore particularly suitable for detecting nonstationarities in a data set, such as when spatial clustering is concentrated in one subregion of the data (Anselin 1992).

The Getis-Ord G_i^* statistic for each hexagon summed the differences between the mean values in a local sample, determined by a moving window consisting of the hexagon and its 18 first- and second-order neighbors (the 6 adjacent hexagons and the 12 additional hexagons contiguous to those 6), and the global mean of all the forested hexagonal cells in the conterminous 48 States. It was then standardized as a z-score with a mean of 0 and a standard deviation of 1, with values >1.96 representing significant ($p < 0.025$) local clustering of high values and values <-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 (exactly 1.96) standard deviations of the mean (Laffan 2006). In other words, a G_i^* value of 1.96 indicates that the local mean of the percentage of forest exposed to mortality-causing or defoliation-causing agents for a hexagon and its 18 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 18 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, mortality or defoliation damage within it and its 18 neighbors is not statistically different from a normal expectation.

It is worth noting that the -1.96 and 1.96 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 used to define the local sample around the target observation is large enough to include several neighbors for each feature (ESRI 2012).

The low density of survey data from Alaska and Hawaii in 2013 (fig. 2.1) precluded the use of Getis-Ord hot spot analyses for these States. Instead, Alaska mortality and defoliation data were summarized by ecoregion section (Nowacki and Brock 1995), calculated as the percentage of the forest within the surveyed areas affected by agents of mortality or defoliation. (As with the mortality and defoliation data, the flown-area polygons were first dissolved to create an overall footprint.) No corresponding ecoregion treatment exists for Hawaii, however, so it was not possible to summarize mortality and defoliation for that State similarly. For reference purposes, ecoregion sections (Cleland and others 2007) were also displayed on the geographic hot spot maps of the conterminous 48 United States.

RESULTS AND DISCUSSION

Conterminous United States Mortality

The national IDS survey data identified 73 different mortality-causing agents and complexes on approximately 1.53 million ha across the conterminous United States in 2013, slightly larger than the combined land area of Connecticut and Rhode Island. (Three of these mortality-cause categories were “rollups” of multiple agents.) By way of comparison, forests are estimated to cover approximately 252 million ha of the conterminous 48 States (Smith and others 2009).

Mountain pine beetle (*Dendroctonus ponderosae*) was the most widespread mortality agent in 2013, detected on 653 700 ha (table 2.1), continuing a downward trend in the area affected by this insect in recent years, from 3.47 million ha in 2009 (Potter 2013), to 2.77 million ha in 2010 (Potter and Paschke 2013), to 1.54 million ha in 2011 (Potter and Paschke 2014), and to 969 037 ha in 2012 (Potter and Paschke 2015). The total footprint, or nonoverlapping sum of areas, of detected mountain pine beetle mortality from 2000 through 2013 exceeds 9.54 million ha, with the large majority occurring in the Forest Health Monitoring (FHM) Program Interior West region (as defined by the FHM Program) (table 2.2). This footprint is slightly larger than the State of Indiana.

Three other mortality agents and complexes were detected on more than 100 000 ha in

Table 2.1—Mortality agents and complexes affecting more than 5000 ha in the conterminous United States during 2013

Agents/complexes causing mortality, 2013	Area <i>ha</i>
Mountain pine beetle ^a	653 700
Spruce beetle	216 296
Ips engraver beetles	105 449
Fir engraver	103 755
Subalpine fir mortality complex ^a	98 594
Western pine beetle	94 047
Douglas-fir beetle	91 565
Five-needle pine decline ^a	89 865
Emerald ash borer	70 974
Pinyon ips	39 187
Sudden oak death	19 231
Jeffrey pine beetle	17 668
Spruce budworm	15 463
Pine engraver	13 333
Unknown	10 530
Eastern larch beetle	10 329
Multidamage (insect/disease)	10 026
Balsam woolly adelgid	9 952
Armillaria root disease	9 877
Flatheaded fir borer	6 723
Western balsam bark beetle ^b	5 947
Bark beetles	5 462
Twig beetles	5 336
Other mortality agents (50)	40 197
Total, all mortality agents	1 529 050

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 due to the reporting of multiple agents per polygon.

^a Rollup of multiple agent codes from the Insect and Disease Survey database.

^b Also included in the subalpine fir mortality rollup.

Table 2.2—Footprint area affected by mountain pine beetle (*Dendroctonus ponderosae*), by Forest Health Monitoring (FHM) region, from 2000 through 2013

FHM region	Area <i>ha</i>
Interior West	7 481 640
West Coast	1 900 240
North Central	161 616
Total, all regions	9 543 496

2013: spruce beetle (*Dendroctonus rufipennis*), ips engraver beetles (*Ips* spp.), and fir engraver (*Scolytus ventralis*). Mortality from the western bark beetle group was detected on more than 1.35 million ha in 2013, representing a large majority of the total area on which mortality was recorded across the conterminous States. This group encompasses 24 different agents in the IDS data (table 2.3).

The Interior West region had approximately 992 000 ha on which mortality-causing agents and complexes were detected in 2013, an area far greater than that of any other FHM region (table 2.4). About 43 percent of this was associated with mountain pine beetle, although spruce beetle (20 percent), ips engraver beetles (11 percent), subalpine fir (*Abies lasiocarpa*) mortality complex (10 percent), and Douglas-fir beetle (*Dendroctonus pseudotsugae*) (8 percent) also constituted a considerable portion of the entire area. A total of 27 mortality agents and complexes were detected in the region.

The Getis-Ord analysis detected several major hot spots of intense mortality exposure in the Interior West region (fig. 2.2). As in 2012, the most intense was centered on the border between eastern Idaho and western Montana, especially in ecoregions M332B–Northern Rockies and Bitterroot Valley and M332E–Beaverhead Mountains. Mortality in this area was attributed almost entirely to mountain pine beetle in lodgepole pine (*Pinus contorta*) and ponderosa pine (*Pinus ponderosa*) forests, although smaller areas of mortality were associated with Douglas-fir beetle and white pine blister rust (caused by *Cronartium ribicola*). The hot spot extended beyond those ecoregions into several others, including M332A–Idaho Batholith, M332D–Belt Mountains, and M333D–Bitterroot Mountains. A smaller hot spot, a short distance to the east and also associated with mountain pine beetle mortality, was centered on 331K–North Central Highlands and M332D–Belt Mountains.

In M331E–Uinta Mountains of northeastern Utah, a high-intensity hot spot was mainly associated with mountain pine beetle infestations in lodgepole pine stands, with spruce beetle-caused mortality in Engelmann spruce (*Picea engelmannii*) stands, and with subalpine fir mortality complex in subalpine fir stands (fig. 2.2).

Nearly all of central Colorado constituted a mortality hot spot, with the highest intensities occurring in M331G–South-Central Highlands and M331I–Northern Parks and Ranges. The hot spots extended into M331F–Southern Parks and Rocky Mountain Range, M331H–North-Central

Table 2.3—Beetle taxa included in the “western bark beetle” group

Western bark beetle mortality agents	Genus and species
California fivespined ips	<i>Ips paraconfusus</i>
Cedar and cypress bark beetles	<i>Phloeosinus</i> spp.
Douglas-fir beetle	<i>Dendroctonus pseudotsugae</i>
Douglas-fir engraver	<i>Scolytus unispinosus</i>
Fir engraver	<i>Scolytus ventralis</i>
Five-needle pine decline	—
Flatheaded borer	<i>Buprestidae</i>
Ips engraver beetles	<i>Ips</i> spp.
Jeffrey pine beetle	<i>Dendroctonus jeffreyi</i>
Mountain pine beetle	<i>Dendroctonus ponderosae</i>
Pine engraver	<i>Ips pini</i>
Pinyon ips	<i>Ips confusus</i>
Pinyon pine mortality	—
Red turpentine beetle	<i>Dendroctonus valens</i>
Roundheaded pine beetle	<i>Dendroctonus adjunctus</i>
Silver fir beetle	<i>Pseudohylesinus sericeus</i>
Southern pine beetle	<i>Dendroctonus frontalis</i>
Spruce beetle	<i>Dendroctonus rufipennis</i>
Subalpine fir (<i>Abies lasiocarpa</i>) mortality complex	—
True fir (<i>Abies</i>) pest complex	—
Western balsam bark beetle	<i>Dryocoetes confusus</i>
Western cedar bark beetle	<i>Phloeosinus punctatus</i>
Western pine beetle	<i>Dendroctonus brevicornis</i>
Bark beetles (nonspecific)	—

— = not applicable.

Table 2.4—The top five mortality agents or complexes for each Forest Health Monitoring region, and for Alaska and Hawaii, in 2013

2013 mortality agents and complexes	Area	2013 mortality agents and complexes	Area
	<i>ha</i>		<i>ha</i>
Interior West		South	
Mountain pine beetle ^a	421 829	Hemlock woolly adelgid	197
Spruce beetle	202 728	Unknown	191
Ips engraver beetles	104 135	Southern pine beetle	186
Subalpine fir mortality complex ^a	97 315	Black turpentine beetle	54
Douglas-fir beetle	78 492	Ips engraver beetles	23
Other mortality agents and complexes (22)	126 636	Bark beetles (nonspecific)	2
Total, all mortality agents and complexes	992 139	Total, all mortality agents and complexes	654
North Central		West Coast	
Emerald ash borer	70 561	Mountain pine beetle ^a	218 608
Spruce budworm	15 463	Fir engraver	61 261
Mountain pine beetle ^a	13 263	Western pine beetle	58 339
Eastern larch beetle	10 329	Sudden oak death	19 231
Pine engraver	9 766	Jeffrey pine beetle	17 664
Other mortality agents and complexes (21)	24 612	Other mortality agents and complexes (22)	63 809
Total, all mortality agents and complexes	133 303	Total, all mortality agents and complexes	387 584
North East		Alaska	
Forest tent caterpillar	2 726	Spruce beetle	10 932
Beech bark disease	2 560	Yellow-cedar decline	5 403
Southern pine beetle	2 284	Northern spruce engraver	3 259
Balsam woolly adelgid	1 525	Total, all mortality agents and complexes	19 594
Unknown	1 508	Hawaii	
Other mortality agents and complexes (23)	5 561	Unknown	15
Total, all mortality agents and complexes	15 371	Total, all mortality agents and complexes	15

The total area affected by other agents is listed at the end of each section. 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 due to the reporting of multiple agents per polygon.

^a Rollup of multiple agent codes from the Insect and Disease Survey database.

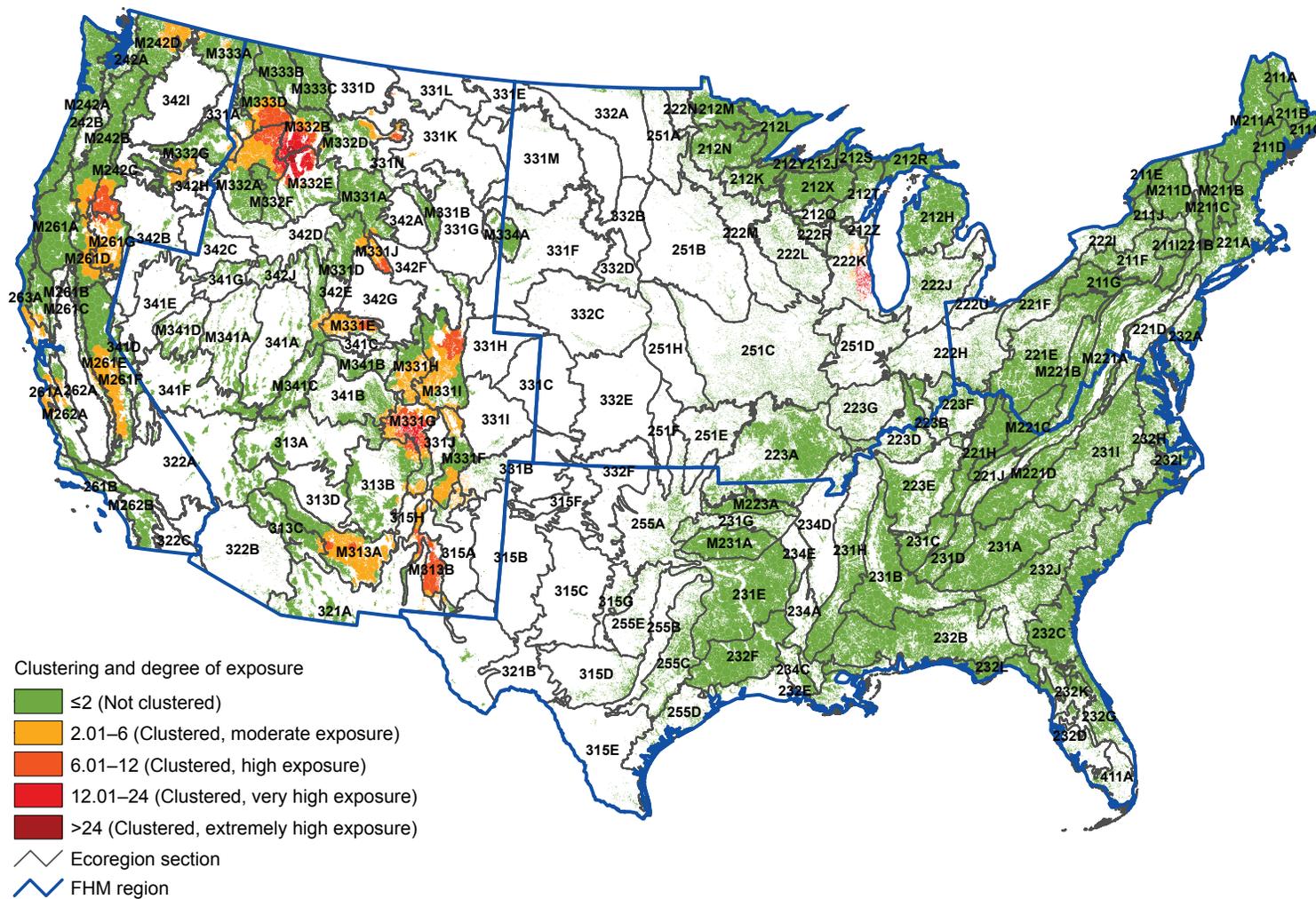


Figure 2.2—Hot spots of exposure to mortality-causing insects and diseases in 2013. Values are Getis-Ord G_i^* scores, with values >2 representing significant clustering of high percentages of forest area exposed to mortality agents. (No areas of significant clustering of low percentages of exposure, <-2 , were detected.) The gray lines delineate ecoregion sections (Cleland and others 2007), and the blue lines delineate Forest Health Monitoring (FHM) 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)

Highlands and Rocky Mountains, 313B–Navaho Canyonlands, and 331J–Northern Rio Grande Basin. Most of the mortality in this area was caused by spruce beetle in Engelmann spruce stands, although mortality was also associated with fir engraver in white fir (*Abies concolor*) forests, with subalpine fir mortality complex in subalpine fir forests, and with Douglas-fir beetle in Douglas-fir forests. Another hot spot of high mortality was located to the south in New Mexico, centered in M313B–Sacramento-Monzano Mountains and extending into 315H–Central Rio Grande Intermontaine, M331G–South-Central Highlands, and M331F–Southern Parks and Rocky Mountain Range. Mortality in this area was associated with a mixture of several mortality agents, including ips engraver beetles, western pine beetle (*Dendroctonus brevicomis*), western balsam bark beetle (*Dryocoetes confusus*), Douglas-fir beetle, fir engraver, and pinyon ips (*Ips confusus*). Moderate-to-high mortality extended west through New Mexico into Arizona in M313A–White Mountains-San Francisco Peaks-Mogollon Rim.

The FHM West Coast region had the second largest area on which mortality agents and complexes were detected, about 388 000 ha (table 2.4). Of the 27 agents and complexes detected, mountain pine beetle was the leading cause of mortality. It was identified on about 219 000 ha, approximately 56 percent of the entire area. Other bark beetles, including fir engraver, western pine beetle, and Jeffrey pine beetle (*Dendroctonus jeffreyi*), were also widespread causes of mortality in the region, as

was sudden oak death (caused by *Phytophthora ramorum*).

Bark beetles were the primary agent associated with four large hot spots of mortality in the West Coast region. The largest of these encompassed much of four ecoregions in northern California and south-central Oregon: M242C–Eastern Cascades, M261G–Modoc Plateau, M242B–Western Cascades, and M261D–Southern Cascades (fig. 2.2). Here, the most common mortality agents were mountain pine beetle in stands of lodgepole pine, ponderosa pine, and western white pine (*Pinus monticola*); western pine beetle in ponderosa pine stands; fir engraver in white fir stands; and Jeffrey pine beetle in Jeffrey pine (*Pinus jeffreyi*) stands. The mortality causes were similar in a hot spot to the northeast in M332G–Blue Mountains.

A hot spot of mortality in M261E–Sierra Nevada and M261F–Sierra Nevada Foothills was associated primarily with mountain pine beetle in stands of lodgepole pine, western white pine, whitebark pine (*Pinus albicaulis*), and sugar pine (*Pinus lambertiana*); with western pine beetle in ponderosa pine forests; with Jeffrey pine beetle in Jeffrey pine forests; and with fir engraver in stands of California red fir (*Abies magnifica*) and white fir (fig. 2.2). A pair of mortality hot spots in north-central Washington State (in M242D–Northern Cascades and M333A–Okanogan Highland) was caused by infestations of spruce beetle in spruce (*Picea* spp.) forests and mountain pine beetle in lodgepole pine forests.

Sudden oak death mortality in tanoak (*Lithocarpus densiflorus*) and coast live oak (*Quercus agrifolia*) forests was the leading agent of mortality associated with two other mortality hot spots along the California coast. The northern hot spot was located north of San Francisco Bay within 263A–Northern California Coast and M261B–Northern California Coast Ranges. Here, additional sources of mortality were pitch canker (caused by *Fusarium circinatum*) in bishop pine (*Pinus muricata*) stands, flatheaded fir borer (*Phaenops drummondi*) in Douglas-fir forests, and California flatheaded borer (*Phaenops californica*) in knobcone pine (*Pinus attenuata*) stands. The southern hot spot, south of San Francisco Bay, was located within 261A–Central California Coast and M262A–Central California Ranges. Other than sudden oak death, western pine beetle in Coulter pine (*Pinus coulteri*) stands, multiagent damage in gray pine (*Pinus sabaliana*), and flatheaded fir borer in bristlecone fir (*Abies bracteata*) were causes of mortality in this area.

In the North Central FHM region, mortality was recorded on more than 133 000 ha, with emerald ash borer the most widely identified causal agent, found on almost 71 000 ha (table 2.4). Of the 26 agents and complexes detected in the region, spruce budworm (*Choristoneura fumiferana*), mountain pine beetle, eastern larch beetle (*Dendroctonus simplex*), and pine engraver (*Ips pini*) each also affected areas exceeding 9000 ha. Emerald ash borer was the cause of the single mortality hot spot in the region, in 222K–Southwestern Great Lakes Morainial in southeastern Wisconsin (fig. 2.2).

No geographic hot spots of mortality were detected in the North East and South FHM regions. In the North East region, the FHP survey recorded mortality-causing agents and complexes on approximately 15 000 ha (table 2.4). Forest tent caterpillar (*Malacosoma disstria*) was the most widely detected mortality agent, followed by beech bark disease, southern pine beetle (*Dendroctonus frontalis*), and balsam woolly adelgid (*Adelges piceae*). In the South, mortality was detected on about 700 ha, with hemlock woolly adelgid and southern pine beetle being the most commonly detected agents (table 2.4).

Conterminous United States Defoliation

In 2013, the national IDS survey identified 83 defoliation agents and complexes affecting approximately 2.94 million ha across the conterminous United States, slightly larger than the combined land area of Vermont and Delaware. (Two of these defoliation-cause categories were “rollups” of multiple agents.) The most widespread defoliator was fall cankerworm (*Alsophila pometaria*), detected on approximately 962 000 ha, followed by western and eastern spruce budworms (*Choristoneura occidentalis* and *C. fumiferana*), affecting slightly more than 728 000 ha (table 2.5). Three other insects—tent caterpillars (*Malacosoma* spp.), gypsy moth (*Lymantria dispar*), and baldcypress leafroller (*Archips goyerana*)—each also affected more than 100 000 ha.

The South FHM region had the largest area on which defoliating agents and complexes were detected in 2013, approximately 1.1 million ha

Table 2.5—Defoliation agents and complexes affecting more than 5000 ha in the conterminous United States in 2013

Agents/complexes causing defoliation, 2013	Area
	ha
Fall cankerworm	961 855
Spruce budworm (eastern and western) ^a	728 003
Tent caterpillars ^a	608 523
Gypsy moth	232 219
Baldcypress leafroller	117 768
Loopers	80 307
Phoberia moth	80 052
Aspen defoliation	54 597
Spruce budworm	52 367
Lophodermella needle cast of pines	42 046
Birch leaf fungus	32 649
Large aspen tortrix	28 971
Unknown defoliator	24 017
Pinyon needle scale	23 063
Anthracnose	22 354
Unknown	21 914
Leafroller/seed moth	11 310
Other defoliator (known)	11 092
Larch needle cast	10 335
Winter moth	9 724
Tent caterpillars	9 628
Larch casebearer	7 504
Pinyon sawfly	6 556
Other gallmaking insect (known)	5 899
Western blackheaded budworm	5 752
Other defoliation agents (57)	43 429
Total, all defoliation agents	2 941 264

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 due to the reporting of multiple agents per polygon.

^a Rollup of multiple agent codes from the Insect and Disease Survey database.

(table 2.6). Fall cankerworm affected the greatest area, approximately 922 000 ha, but forest tent caterpillar and baldcypress leafroller were also surveyed across large areas. A large area of mostly low-severity defoliation (≤ 50 percent) caused by fall cankerworm caused a hot spot of high-defoliation exposure in northern Virginia and southern Maryland (in the North East FHM region), centered on 231I–Central Appalachian Piedmont and 232H–Middle Atlantic Coastal Plains and Flatwoods (fig. 2.3). Defoliation by baldcypress leafroller and forest tent caterpillar, meanwhile, resulted in a high-defoliation hot spot in southern Louisiana in ecoregions 232E–Louisiana Coastal Prairies and Marshes and 234C–Atchafalaya and Red River Alluvial Plains.

Thirty defoliation agents and complexes were identified on about 327 000 ha in the North East FHM region, with gypsy moth the most widely detected on nearly 206 000 ha. Gypsy moth was the cause of the single defoliation hot spot in the region, centered on ecoregion 211G–Northern Unglaciated Allegheny Plateau in northwestern Pennsylvania and southwestern New York (fig. 2.3).

In the North Central FHM region, defoliators were identified on approximately 650 000 ha, with forest tent caterpillar the most widely detected on slightly more than 434 000 ha, followed by loopers and Phoberia moth (*Phoberia atomaris*). A total of 20 agents and complexes were identified in the region. Forest tent caterpillar was the cause of a high-exposure hot spot of defoliation in two ecoregions in northern Minnesota, 212N–Northern Minnesota Drift and

Table 2.6—The top five defoliation agents or complexes for each Forest Health Monitoring region, and for Alaska and Hawaii, in 2013

2013 defoliation agents and complexes	Area <i>ha</i>	2013 defoliation agents and complexes	Area <i>ha</i>
Interior West		South	
Western spruce budworm	601 271	Fall cankerworm	922 062
Aspen defoliation	54 597	Forest tent caterpillar	161 973
Lophodermella needle cast of pines	42 046	Baldcypress leafroller	117 768
Pinyon needle scale	23 040	Unknown	932
Unknown defoliator	19 240	Total, all defoliation agents and complexes	1 098 609
Other defoliation agents and complexes (26)	41 906	West Coast	
Total, all defoliation agents and complexes	765 460	Western spruce budworm	72 922
North Central		Larch needle cast	5 849
Forest tent caterpillar	434 032	Western blackheaded budworm	5 752
Loopers	80 307	Douglas-fir tussock moth	2 600
Phoberia moth	80 052	Western tent caterpillar	2 469
Spruce budworm	52 367	Other defoliation agents and complexes (18)	10 610
Large aspen tortrix	27 030	Total, all defoliation agents and complexes	100 178
Other defoliation agents and complexes (15)	56 409	Alaska	
Total, all defoliation agents and complexes	650 126	Birch leafroller	133 962
North East		Defoliators	66 869
Gypsy moth	205 585	Western blackheaded budworm	49 041
Fall cankerworm	39 553	Aspen leafminer	40 236
Birch leaf fungus	32 649	Willow leaf blotchminer	11 420
Anthracnose	22 354	Other defoliation agents and complexes (8)	16 536
Unknown	17 122	Total, all defoliation agents and complexes	312 515
Other defoliation agents and complexes (25)	33 529	Hawaii	
Total, all defoliation agents and complexes	326 891	Koa looper moth	26 301
		Total, all defoliation agents and complexes	26 301

The total area affected by other agents is listed at the end of each section. 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 due to the reporting of multiple agents per polygon.

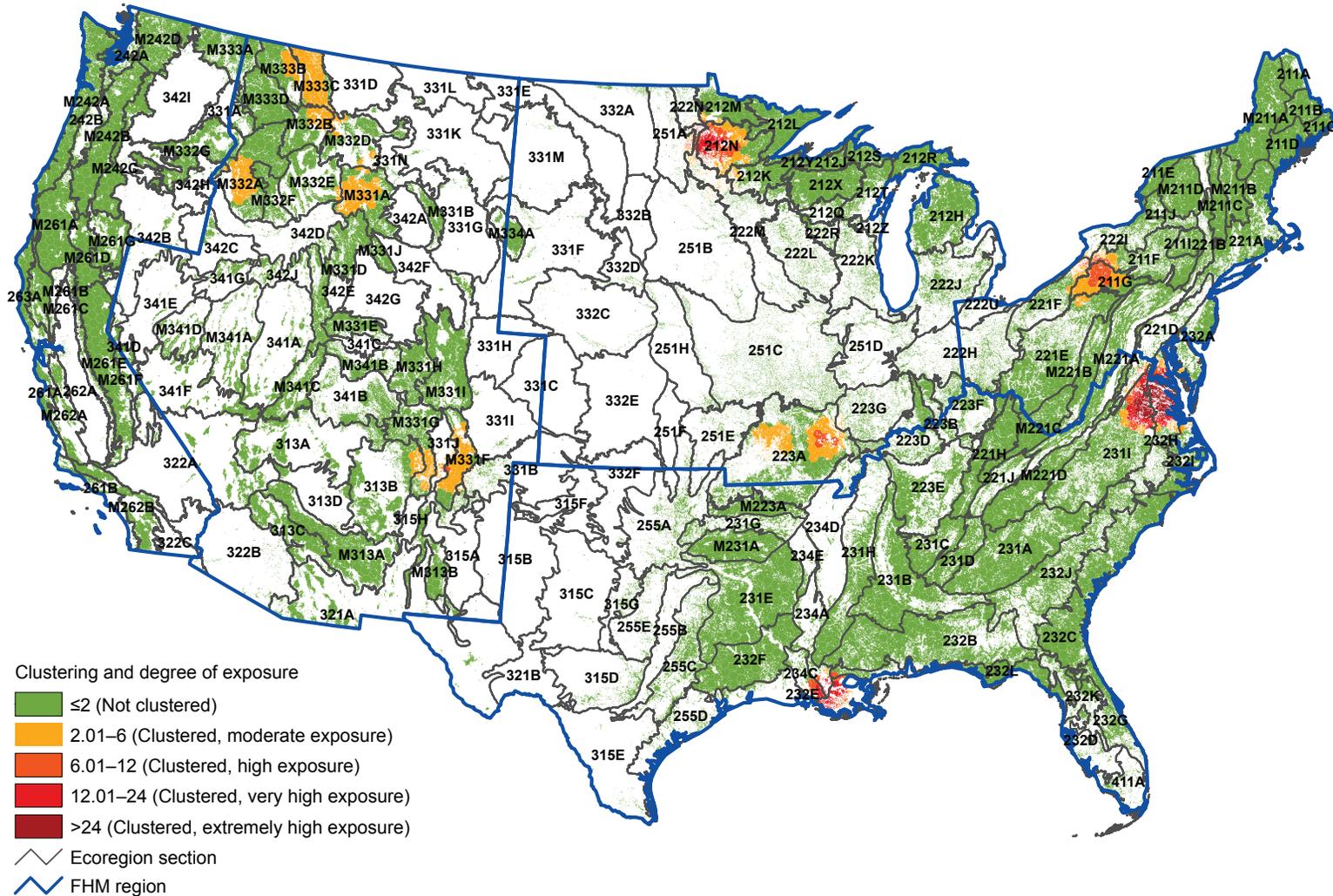


Figure 2.3—Hot spots of exposure to defoliation-causing insects and diseases in 2013. Values are Getis-Ord G_i^* scores, with values >2 representing significant clustering of high percentages of forest area exposed to defoliation agents. (No areas of significant clustering of low percentages of exposure, <-2 , were detected.) The gray lines delineate ecoregion sections (Cleland and others 2007), and the blue lines delineate Forest Health Monitoring (FHM) 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)

Lake Plains and 222M–Minnesota and Northeast Iowa Morainal-Oak Savanna (fig. 2.3). Areas of looper infestation resulted in defoliation hot spots in 223A–Ozark Highlands in southern Missouri.

Of the approximately 765 000 ha of defoliation in the Interior West FHM region, 79 percent (about 601 000 ha) was attributed to western spruce budworm (table 2.6). Aspen defoliation and Lophodermella needle cast of pines (*Lophodermella* spp.) were the next most widely detected defoliation agents of the 31 that were identified. All four defoliation hot spots in the region (fig. 2.3) were associated with western spruce budworm, along with other agents or complexes. In the northernmost of these hot spots, in M333B–Flathead Valley, M333C–Northern Rockies, and M332B–Northern Rockies and Bitterroot Valley, the primary defoliation agents were western spruce budworm in fir forests and larch needle cast (*Meria laricis*) in western larch (*Larix occidentalis*) stands. To the southeast in M331A–Yellowstone Highlands (southwestern Montana and northwestern Wyoming), a defoliation hot spot was caused by western spruce budworm in fir and Lophodermella needle cast of pines in lodgepole pine stands. To the southwest in M332A–Idaho Batholith, a defoliation hot spot was associated with western spruce budworm in subalpine fir and Douglas-fir stands.

Finally, a defoliation hot spot in northern New Mexico and southern Colorado (M331G–South-Central Highlands and M331F–Southern Parks and Rocky Mountain Range) was associated with

western spruce budworm and aspen (*Populus tremuloides*) defoliation.

Western spruce budworm, meanwhile, accounted for about 73 percent of the approximately 100 000 ha of defoliation detected in the FHM West Coast region (table 2.6). The second and third leading defoliators in the region were larch needle cast and western blackheaded budworm (*Acleris gloverana*). No geographic hot spots of defoliation were identified in the region, where a total of 23 defoliation agents and complexes were detected.

Alaska and Hawaii

In Alaska, approximately 8 million ha of forested area was surveyed, 15.7 percent of the total forested land in the State. Mortality was recorded on nearly 20 000 ha in 2013, associated with three agents and complexes (table 2.4). This is a very small proportion (<1 percent) of the forested area surveyed. Spruce beetle was the most widely detected mortality agent, found on about 10 900 ha, mostly in the southern parts of the State. Yellow-cedar (*Chamaecyparis nootkatensis*) decline was identified on about 5400 ha in the Alaska panhandle, while northern spruce engraver (*Ips perturbatus*) was detected on about 3300 ha in the central and northern forested areas of the State. The percentage of surveyed forest exposed to mortality agents did not exceed 1 percent in any of Alaska's ecoregions (fig. 2.4).

Meanwhile, defoliators were detected on a much larger area of Alaska during 2013, with

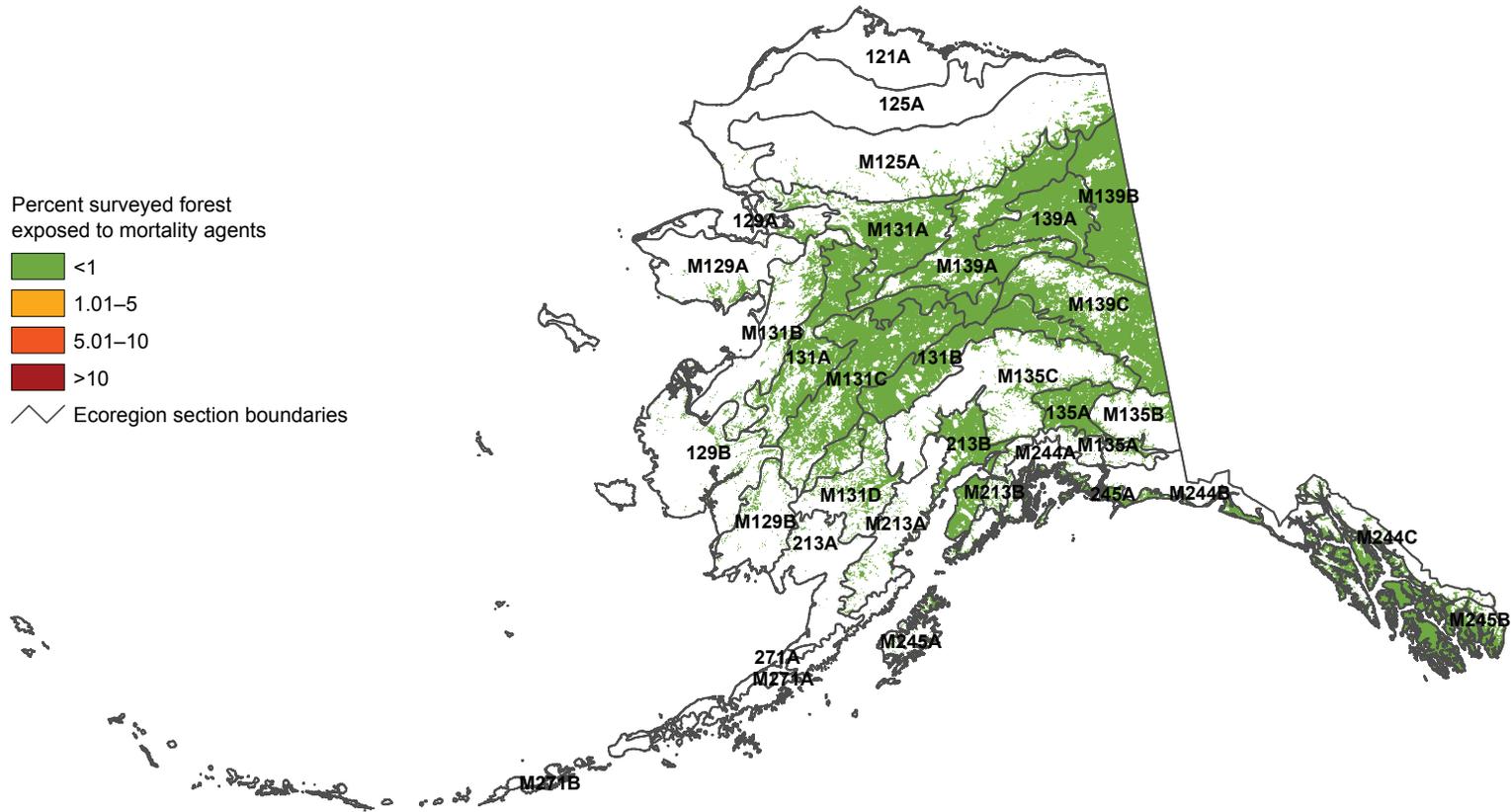


Figure 2.4—Percent of surveyed forest in Alaska ecoregion sections exposed to mortality-causing insects and diseases in 2013. The gray lines delineate ecoregion sections (Nowacki and Brock 1995). Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection)

13 defoliating agents recorded on more than 312 000 ha (table 2.6). Birch leafroller (*Epinotia solandriana*) was by far the most commonly recorded defoliator, recorded on approximately 134 000 ha. Nonspecific defoliators were the causal agent of defoliation on almost 67 000 ha. Western blackheaded budworm was detected on 49 000 ha, while aspen leafminer (*Phyllocnistis populiella*) was detected on 40 000 ha, mostly in the central parts of Alaska. Willow leaf blotchminer (*Micrurapteryx salicifoliella*) was found on approximately 11 000 ha.

The Alaska ecoregions with the highest proportion of surveyed forest area affected by defoliators in 2013 were located in the west-central and southwestern parts of the State (fig. 2.5). M131B–Nulato Hills had the highest proportion of area affected by defoliators (76.6 percent), but only a small proportion of this ecoregion section was surveyed. This was also the case for 213A–Bristol Bay Lowlands, where defoliators were detected on 32.1 percent of the surveyed area. Defoliators were detected on 13.4 percent of surveyed forest in M213A–Northern Aleutian Range and 11.9 percent of 129B–Yukon-Kuskokwim Delta. The primary agent of defoliation in these ecoregions was birch leafroller in stands of Alaska paper birch (*Betula neoalaskana*). A lower proportion of defoliation

was identified in the central, east-central, and south-central portions of the State (between 1 and 5 percent).

Finally, almost no mortality was detected in Hawaii in 2013 (table 2.4), but more than 26 000 ha were identified as having been defoliated by koa looper moth (*Scotorythra paludicola*) (table 2.6). This was about 4 percent of the forested area surveyed in the State.

CONCLUSION

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 include all the biotic forest health threats that should be considered when making management decisions and budget allocations. However, large-scale assessments of mortality and defoliation exposure, including geographical hot spot detection analyses, offer a useful approach for identifying geographic areas where the concentration of monitoring and management activities might be most effective.

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INTRODUCTION

Free-burning wildland fire has been a frequent ecological phenomenon on the American landscape, and its expression has changed as new peoples and land uses have become predominant (Pyne 2010). As a pervasive disturbance agent operating at many spatial and temporal scales, wildland fire is a key abiotic factor affecting forest health both positively and negatively. In some ecosystems, wildland fires have been essential for regulating processes that maintain forest health (Lundquist and others 2011). Wildland fire, for example, is an important ecological mechanism that shapes the distributions of species, maintains the structure and function of fire-prone communities, and acts as a significant evolutionary force (Bond and Keeley 2005).

At the same time, wildland fires have created forest health problems in some ecosystems (Edmonds and others 2011). Specifically, fire outside the historic range of frequency and intensity can impose extensive ecological and socioeconomic impacts. Current fire regimes on more than half of the forested area in the conterminous United States have been 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). Understanding existing fire regimes is essential to properly assessing the impact of fire on forest health because changes to historical fire regimes can alter forest developmental patterns, including the establishment, growth, and mortality of trees (Lundquist and others 2011).

As a result of intense suppression efforts during most of the 20th century, the forest area burned annually decreased from approximately 16 million to 20 million ha (40 to 50 million acres) in the early 1930s to about 2 million ha (5 million acres) in the 1970s (Vinton 2004). In some regions, plant communities have experienced or are undergoing rapid compositional and structural changes as a result of fire suppression (Nowacki and Abrams 2008). At the same time, fires in some regions and ecosystems have become larger, more intense, and more damaging because of the accumulation of fuels as a result of prolonged fire suppression (Pyne 2010). Such large wildland fires also can have long lasting social and economic consequences, which include the loss of human life and property, smoke-related human health impacts, and the cost of fighting the fires themselves (Gill and others 2013, Richardson and others 2012).

CHAPTER 3.

Large-Scale Patterns of Forest Fire Occurrence in the Conterminous United States and Alaska, 2013

KEVIN M. POTTER

Fire regimes have been dramatically altered, in particular, by fire suppression (Barbour and others 1999) and by the introduction of nonnative invasive plants, which can change fuel properties and in turn both affect fire behavior and alter fire regime characteristics such as frequency, intensity, type, and seasonality (Brooks and others 2004). Additionally, changes in fire intensity and recurrence could result in decreased forest resilience and persistence (Lundquist and others 2011), and fire regimes altered by global climate change could cause large-scale shifts in vegetation spatial patterns (McKenzie and others 1996).

This chapter presents analyses of high temporal fidelity fire occurrence data, collected nationally by satellite, that map and quantify where fire occurrences have been concentrated spatially across the conterminous United States and Alaska in 2013. It also, within a geographic context, compares 2013 fire occurrences to all the recent years for which such data are available. Quantifying and monitoring such broad-scale patterns of fire occurrence across the United States can help improve the understanding of the ecological and economic impacts of fire as well as the appropriate management and prescribed use of fire. Specifically, large-scale assessments of fire occurrence can help identify areas where specific management activities may be needed, or where research into the ecological and socioeconomic impacts of fires may be required.

METHODS

Data

Annual monitoring and reporting of active wildland fire events using the Moderate Resolution Imaging Spectroradiometer (MODIS) Active Fire Detections for the United States database (USDA Forest Service 2014) allows analysts to spatially display and summarize fire occurrences across broad geographic regions (Coulston and others 2005; Potter 2012a, 2012b, 2013a, 2013b, 2014, 2015). A fire occurrence is defined as one daily satellite detection of wildland fire in a 1-km² pixel, with multiple fire occurrences possible on a pixel across multiple days. The data are derived using the MODIS Rapid Response System (Justice and others 2002, 2011) to extract fire location and intensity information from the thermal infrared bands of imagery collected daily by two satellites at a resolution of 1 km², with the center of a pixel recorded as a fire occurrence (USDA Forest Service 2014). The Terra and Aqua satellites' MODIS sensors identify the presence of a fire at the time of image collection, with Terra observations collected in the morning and Aqua observations collected in the afternoon. The resulting fire occurrence data represent only whether a fire was active, because the MODIS data bands do not differentiate between a hot fire in a relatively small area (0.01 km², for example) and a cooler fire over a larger area

(1 km², for example). The MODIS Active Fire database does well at capturing large fires during cloud-free conditions, but may underrepresent rapidly burning, small, and low-intensity fires, as well as fires in areas with frequent cloud cover (Hawbaker and others 2008). For more information about the performance of this product, see Justice and others (2011).

Analyses

These MODIS products for 2013 were processed in ArcMap[®] (ESRI 2012) to determine the number of fire occurrences per 100 km² (10 000 ha) of forested area for each ecoregion section in the conterminous 48 States (Cleland and others 2007) and Alaska (Nowacki and Brock 1995). This forest fire occurrence density measure was calculated after screening out wildland fires on nonforested pixels using a forest cover layer derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center (RSAC) (USDA Forest Service 2008). The total numbers of forest fire occurrences were also determined separately for the conterminous States and for Alaska.

The fire occurrence density value for each ecoregion in 2013 was then compared with the mean fire density values for the first 12 full years of MODIS Active Fire data collection (2001–12). Specifically, the difference of the 2013 value and the previous 12-year mean

for an ecoregion was divided by the standard deviation across the previous 12-year period, assuming normal distribution of fire density over time in the ecoregion. The result for each ecoregion was a standardized *z*-score, which is a dimensionless quantity describing whether the fire occurrence density in the ecoregion in 2013 was higher, lower, or the same relative to all the previous years for which data have been collected, accounting for the variability in the previous years. The *z*-score is the number of standard deviations between the observation and the mean of the previous observations. Approximately 68 percent of observations would be expected within one standard deviation of the mean, and 95 percent within two standard deviations. Near-normal conditions are classified as those within a single standard deviation of the mean, although such a threshold is somewhat arbitrary. Conditions between about one and two standard deviations of the mean are moderately different from mean conditions, but are not significantly different statistically. Those outside about two standard deviations would be considered statistically greater than or less than the long-term mean (at $p < 0.025$ at each tail of the distribution).

Additionally, a Getis-Ord hot spot analysis (Getis and Ord 1992) in ArcMap[®] 10.1 (ESRI 2012) was employed to identify forested areas in the conterminous 48 States with higher-than-expected fire occurrence density in 2013. The

spatial units of the analysis were 9,810 cells of approximately 834 km² 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). Fire occurrence density values for each hexagon were quantified as the number of forest fire occurrences per 100 km² 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. This statistic allows for the decomposition of a global measure of spatial association into its contributing factors, by location, and is therefore particularly suitable for detecting outlier assemblages of similar conditions (i.e., nonstationarities) in a dataset, such as when spatial clustering is concentrated in one subregion of the data (Anselin 1992).

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 its 18 first- and second-order neighbors (the 6 adjacent hexagons and the 12 additional hexagons contiguous to those 6), and the global mean of all the forested hexagonal cells in the conterminous 48 States. G_i^* is standardized as a z-score with a mean of 0 and a standard deviation of 1, with values >1.96 representing significant local clustering of higher fire occurrence densities ($p < 0.025$) and values <-1.96 representing significant clustering of lower fire occurrence densities ($p < 0.025$), because 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 18 neighbors, in other words, have a range of both high and low numbers of fire occurrences per 100 km² of forested area. It is worth noting that 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 2012).

RESULTS AND DISCUSSION

The MODIS Active Fire database recorded 98,682 wildland forest fire occurrences across the conterminous United States in 2013, the second largest annual number of fire occurrences since the first full year of data collection in 2001 (fig. 3.1). This number was approximately 28 percent fewer than in 2012 (138,000 forest fire occurrences, the most since the beginning of data collection), but about 68 percent more than the annual mean of 58,709 forest fire occurrences across the previous 12 full years of data collection. In contrast, the MODIS database

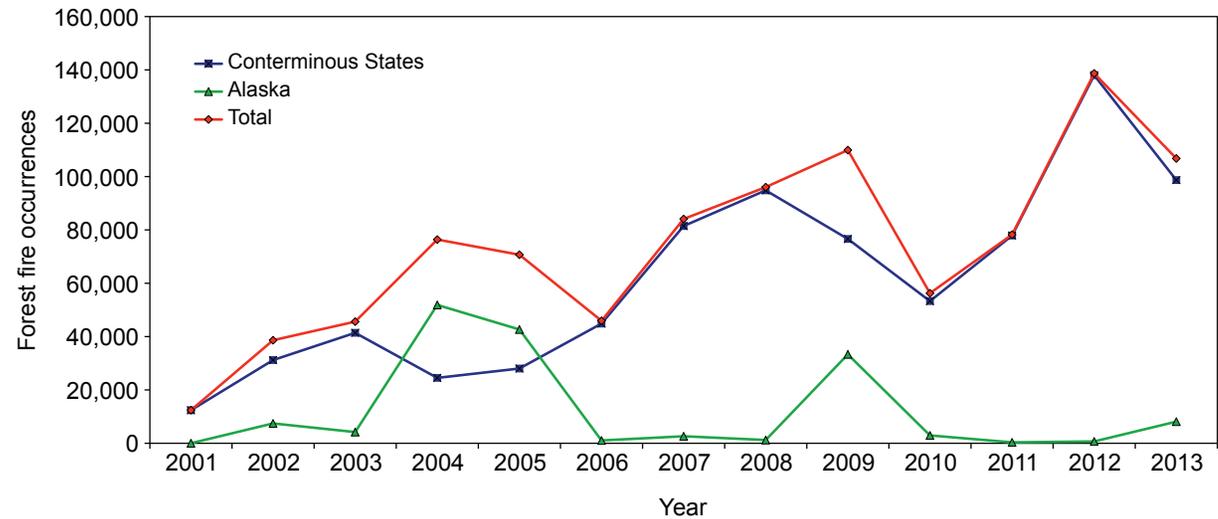


Figure 3.1—Forest fire occurrences detected by MODIS from 2001 through 2013 for the conterminous United States and Alaska and for the two regions combined. (Data source: U.S. Department of Agriculture Forest Service, Remote Sensing Applications Center, in conjunction with the NASA MODIS Rapid Response group)

captured only 8,110 forest fire occurrences in Alaska in 2013, the fourth most since 2001 and about 66 percent of the previous 12-year annual mean of 12,366.

The decrease in the total number of fire occurrences across the conterminous United States is generally consistent with the official wildland fire statistics. In 2013, 47,579 wildfires were reported nationally, compared to 67,774 the previous year. The area burned nationally in 2013 (1 748 058 ha) was 59 percent of the 10-year average, with 20 fires exceeding 16 187 ha (31 fewer than in 2012) (National Interagency Coordination Center 2014). The total area burned nationally represented a 54-percent decrease

from 2012 (3 774 195 ha) (National Interagency Coordination Center 2013). It is important to underscore that estimates of burned area and calculations of MODIS-detected fire occurrences are two different metrics for quantifying fire activity within a given year. Most importantly, the MODIS data contain both spatial and temporal components, since persistent fire will be detected repeatedly over several days on a given 1-km² pixel. In other words, a location can be counted as having a fire occurrence multiple times, once for each day a fire is detected at the location. Analyses of the MODIS-detected fire occurrences, therefore, measure the total number of daily 1-km² pixels with fire during a year, as

opposed to quantifying only the area on which fire occurred at some point during the course of the year.

In 2013, the highest forest fire occurrence densities occurred in Idaho, California, Oregon, and Colorado (fig. 3.2), after summer drought conditions allowed fire fuels to become extremely dry, particularly in northern California and southwestern Oregon (National Interagency Coordination Center 2014). The forested ecoregion with the highest wildland forest fire occurrence density in 2013 (32.8 fire occurrences per 100 km² of forest) was section M332F–Challis Volcanics (fig. 3.2) in central Idaho. The adjacent M332A–Idaho Batholith, meanwhile, experienced 18 fires per 100 km² of forest. These ecoregion sections are located in the Eastern Great Basin Geographic Area, where official wildland fire statistics recorded nearly 311 000 ha burned in 2013 (National Interagency Coordination Center 2014), including the Pony Complex, Elk Complex, and Beaver Creek fires (60 453, 53 118, and 45 118 ha, respectively). Meanwhile, M261E–Sierra Nevada, in central California, saw nearly 30.8 fire occurrences per 100 km² of forest. This area included the Nation’s largest fire in 2013, the 104 131-ha Rim fire, which also was the third largest wildfire in recorded California history, costing an estimated \$127.35 million (National Interagency Coordination Center 2014). In northwestern California and southwestern Oregon, M261A–Klamath Mountains experienced a fire occurrence density of 16.2 fires per 100 km² of forest.

Additionally, two ecoregions that contain relatively small amounts of forest (and therefore do not stand out as easily on fig. 3.2) also had high fire occurrence densities in 2013: 331I–Arkansas Tablelands in southeast Colorado (17.8 fires per 100 km² of forest) and 342I–Columbia Basin in central Washington (16.2 fires per 100 km² of forest).

Elsewhere in the West, several ecoregions had moderate fire occurrence densities, including M331G–South-Central Highlands (south-central Colorado and north-central New Mexico); 322A–Mojave Desert (southeastern California, southern Nevada, and northwestern Arizona); M333C–Northern Rockies (northwestern Montana); and M262B–Southern California Mountain and Valley.

Ecoregions of the Southeastern United States generally experienced moderate fire occurrence densities in 2013 (fig. 3.2). Southeastern ecoregions with relatively high fire densities included 232B–Gulf Coast Plains and Flatwoods (Louisiana, Mississippi, Alabama, Georgia, and Florida, 10.7 fire occurrences); 232G–Florida Coastal Lowlands-Atlantic (eastern Florida, 7.3 occurrences); 232J–Southern Atlantic Coastal Plains and Flatwoods (Georgia, South Carolina, and North Carolina, 9.7 fire occurrences); 232F–Coastal Plains and Flatwoods-Western Gulf (Louisiana and east Texas, 8.4 fire occurrences); and 232D–Florida Coastal Lowlands-Gulf (southwest Florida, 6.8 fire occurrences). Fire occurrence densities, meanwhile, were almost universally low in the Northeastern, Mid-Atlantic, and Midwestern States.

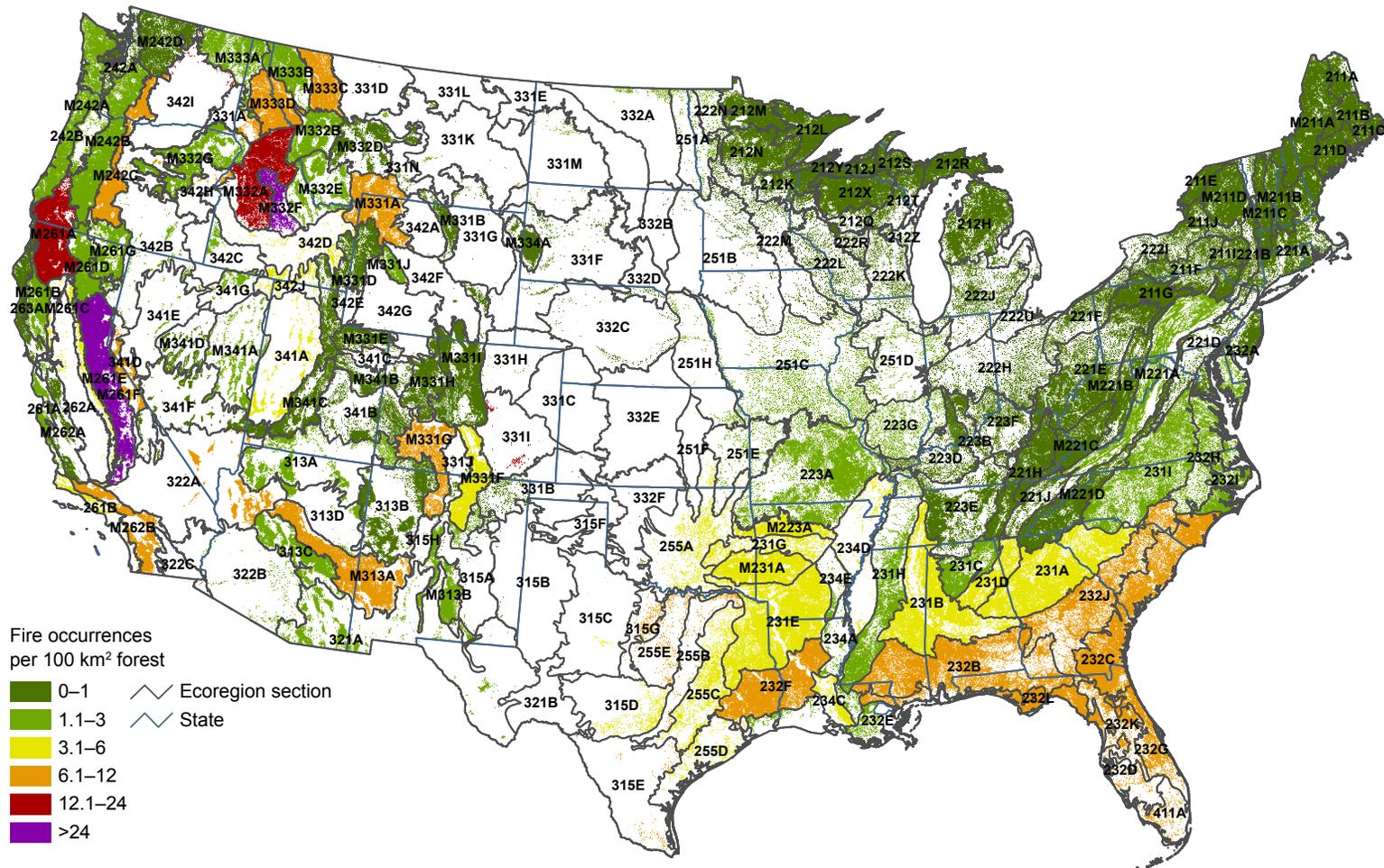


Figure 3.2—The number of forest fire occurrences per 100 km² (10 000 ha) of forested area, by ecoregion section within the conterminous 48 States, for 2013. The gray lines delineate ecoregion sections (Cleland and others 2007). Forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: U.S. Department of Agriculture Forest Service, Remote Sensing Applications Center, in conjunction with the NASA MODIS Rapid Response group)

Meanwhile, Alaska experienced an increase in fire occurrences during its second warmest summer on record, which was coupled with significant dryness that resulted in a later-than-normal fire season (National Interagency Coordination Center 2014). Three Alaskan ecoregions had moderate fire occurrence densities (fig. 3.3). The M139C–Dawson Range ecoregion had the highest fire occurrence density, with 5.4 fire occurrences detected per 100 km² of forest, followed by 131B–Kuskokwim Colluvial Plain (3.7 fire occurrences per 100 km² of forest) and M131D–Nushagak-Lime Hills (3.3 fire occurrences per 100 km² of forest). The Lime Hills fire burned 81 668 ha during the course of 3 months from the end of May to the end of August (National Interagency Coordination Center 2014).

Comparison to Longer Term Trends

Contrasting short-term (1-year) wildland forest fire occurrence with longer term trends is possible by comparing these results for each ecoregion section to the first 12 full years of MODIS Active Fire data collection (2001–12). In general, most ecoregions within the Northeastern, Midwestern, Middle Atlantic, and Appalachian regions experienced <1 fire per 100 km² of forest during that period, with means higher in the northern Rocky Mountain, California, Southeastern, and Southwestern regions (fig. 3.4A). The forested ecoregion that experienced the most fires on average was M332A–Idaho Batholith in central Idaho (mean annual fire occurrence density of 13.6). Other ecoregions with mean fire occurrence densities

of 6.1 to 12.0 were located near the southern California coast, in central Arizona and New Mexico, and in north-central Texas. Ecoregions with the greatest variation in fire occurrence densities from 2001 to 2012 were also located in central Idaho, along the California coast, and in southeastern Oregon, with moderate variation in northeastern California, north-central Washington, western Montana, western Utah, central and southeastern Arizona and southwestern New Mexico, and eastern North Carolina (fig. 3.4B). Less variation occurred throughout the Southeast, central California, noncoastal Oregon and Washington, the Rocky Mountain States, and northern Minnesota. The least variation was apparent throughout most of the Midwest and Northeast.

In 2013, ecoregions scattered across the conterminous United States experienced greater fire occurrence densities than normal, compared to the previous 12-year mean and accounting for variability over time, as determined by the calculation of standardized fire occurrence z-scores (fig. 3.4C). These included ecoregions in central and northwestern California, northern and central Idaho, northwestern Wyoming, southern Colorado and north-central New Mexico, and north-central Minnesota. This was also the case for much of the Southeast and New England. The New England ecoregions had high z-scores despite a relatively low density of fire occurrences in 2013 because these were slightly higher than normal in areas that typically have very little variation over time in fire occurrence density. Several of the western ecoregions also had very high fire occurrence

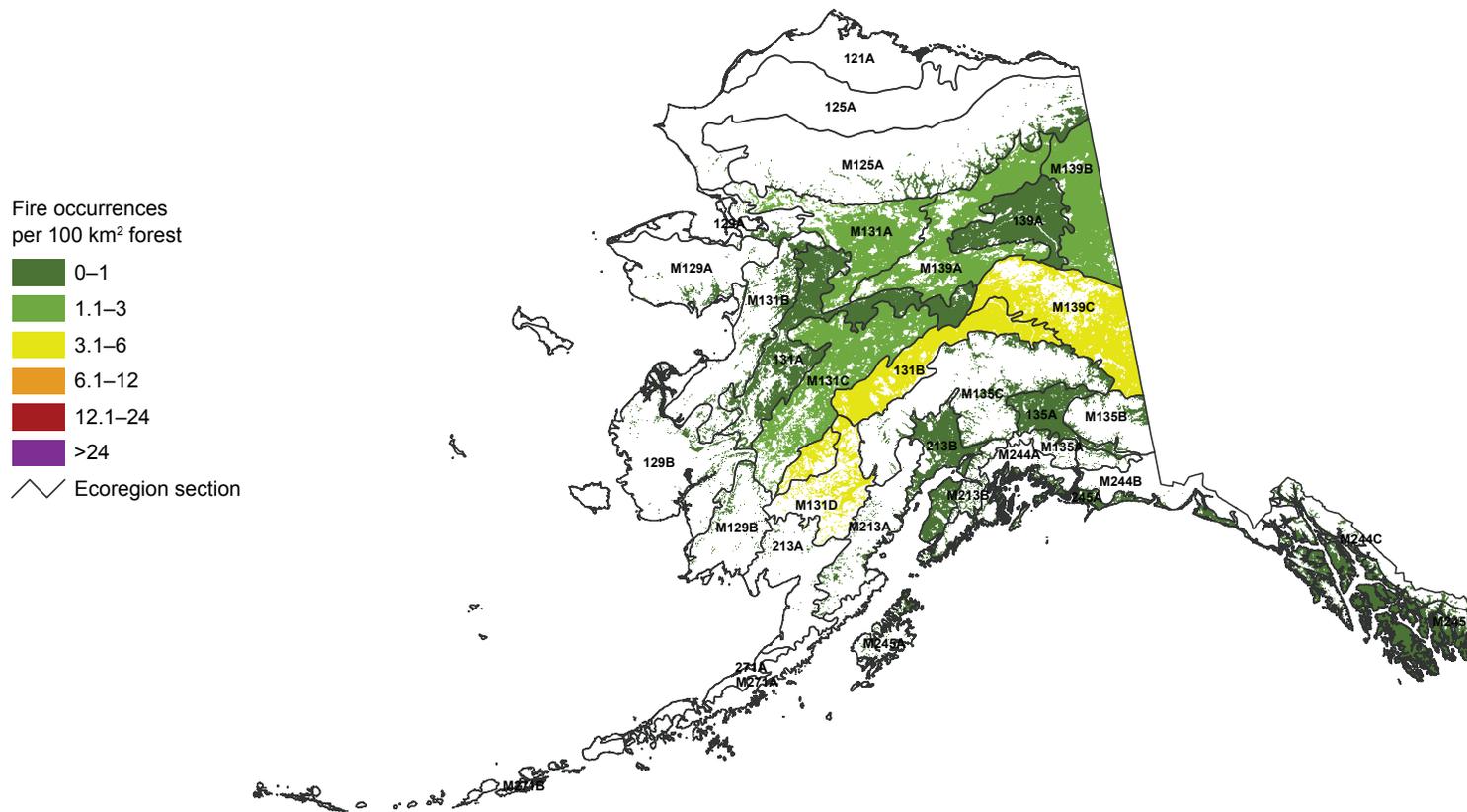


Figure 3.3—The number of forest fire occurrences per 100 km² (10 000 ha) of forested area, by ecoregion section within Alaska, for 2013. The gray lines delineate ecoregion sections (Nowacki and Brock 1995). Forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: U.S. Department of Agriculture Forest Service, Remote Sensing Applications Center, in conjunction with the NASA MODIS Rapid Response group)

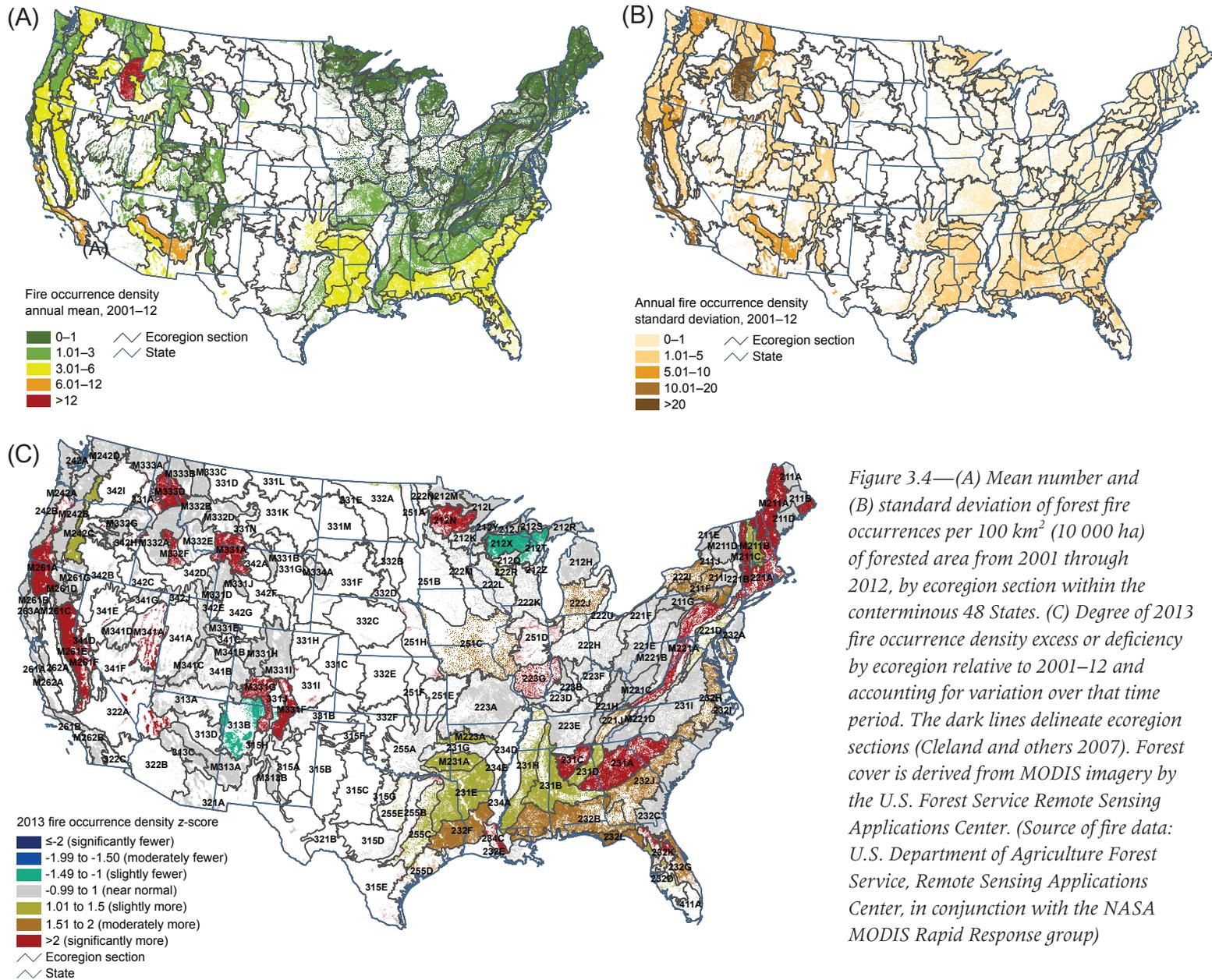


Figure 3.4—(A) Mean number and (B) standard deviation of forest fire occurrences per 100 km² (10 000 ha) of forested area from 2001 through 2012, by ecoregion section within the conterminous 48 States. (C) Degree of 2013 fire occurrence density excess or deficiency by ecoregion relative to 2001–12 and accounting for variation over that time period. The dark 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 Applications Center, in conjunction with the NASA MODIS Rapid Response group)

densities in 2013 (fig. 3.2), including M261E–Sierra Nevada in California, M261A–Klamath Mountains in northwestern California and southwestern Oregon, M332F–Challis Volcanics in central Idaho, and 331I–Arkansas Tablelands in southeastern Colorado. Others had moderate fire occurrence densities in 2013 (fig. 3.2) that still deviated from the previous 12-year mean (fig. 3.4C), including M242C–Eastern Cascades (in central Washington and Oregon), M333D–Bitterroot Mountains (in northern Idaho), M331A–Yellowstone Highlands (in Wyoming), 322A–Mojave Desert (in southern Nevada and northwestern Arizona), and M331G–South-Central Highlands and M331F–Southern Parks and Rocky Mountain Range (in southern Colorado and northern New Mexico). In the Southeastern United States, these included 231A–Southern Appalachian Piedmont, 232K–Florida Coastal Plains Central Highlands, and 234C–Atchafalaya and Red River Alluvial Plains.

Only three ecoregions in the conterminous United States had a lower fire occurrence density in 2013 compared to the longer term: 313B–Navaho Canyonlands (in northwestern New Mexico), 212X–Northern Highlands (in northern Wisconsin), and 212T–Northern Green Bay Lobe (in northeastern Wisconsin and the Upper Peninsula of Michigan) (fig. 3.4C). This is the case because these are all regions with relatively low annual mean fire occurrence densities (<1 fire per 100 km² of forest per year) and low levels of variability in those mean densities, where a slightly smaller-than-usual number of fire occurrences in 2013 was coupled with low variability over time.

Of additional interest are the several ecoregions across the Midwestern, Northeastern, and Mid-Atlantic States that had 2013 fire occurrence densities that were low but still had relatively high z-scores (fig. 3.4C). Among these are 212N–Northern Minnesota Drift and Lake Plains (in northern Minnesota); M221A–Northern Ridge and Valley (in Tennessee, Virginia, West Virginia, Maryland, and Pennsylvania); 221A–Lower New England (stretching from Pennsylvania to Maine); and M211A–White Mountains (in Maine, New Hampshire, and Vermont).

In Alaska, meanwhile, the highest mean annual fire occurrence density between 2001 and 2012 occurred in the east-central and central parts of the State (fig. 3.5A) in the 139A–Yukon Flats ecoregion, with moderate mean fire occurrence density in neighboring areas. As expected, many of those same areas experienced the greatest degree of variability over the 12-year period (fig. 3.5B). In 2013, three ecoregions were outside the range of near-normal fire occurrence density, compared to the mean of the previous 12 years and accounting for variability. All are located in south-central Alaska: M213A–Northern Aleutian Range, M131D–Nushagak-Lime Hills, and M135C–Alaska Range (fig. 3.5C).

Geographical Hot Spots of Fire Occurrence Density

While summarizing fire occurrence data at the ecoregion scale allows for the quantification of fire occurrence density across the country, a geographical hot spot analysis can offer insights

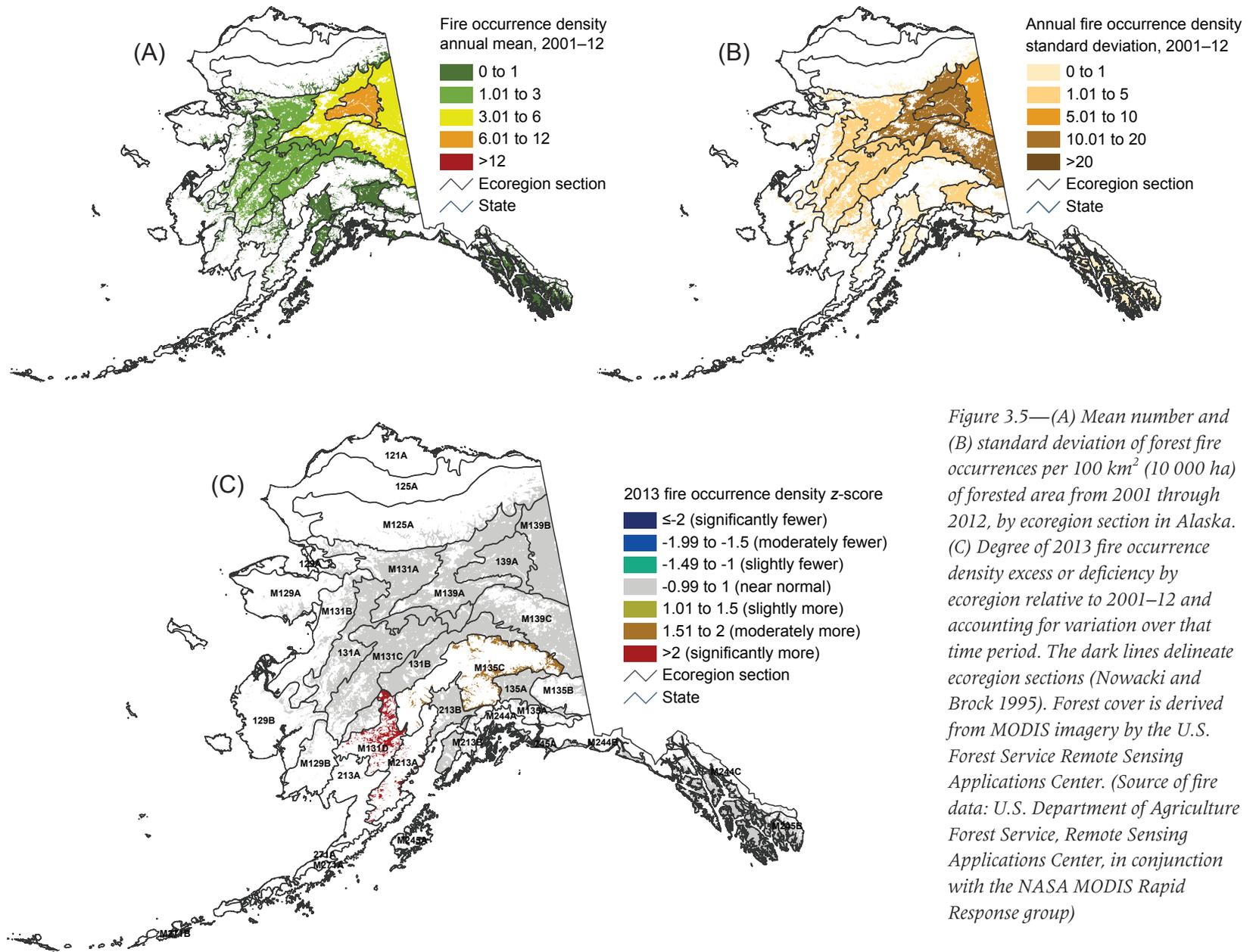


Figure 3.5—(A) Mean number and (B) standard deviation of forest fire occurrences per 100 km² (10 000 ha) of forested area from 2001 through 2012, by ecoregion section in Alaska. (C) Degree of 2013 fire occurrence density excess or deficiency by ecoregion relative to 2001–12 and accounting for variation over that time period. The dark 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 Applications Center, in conjunction with the NASA MODIS Rapid Response group)

into where, statistically, fire occurrences are more concentrated than expected by chance. In 2013, the two geographical hot spots with the highest fire occurrence density were located in California and Idaho (fig. 3.6). The larger of these was centered in M261E–Sierra Nevada, the location of the 2-month Rim fire in and around Yosemite National Park. The second high-density hot spot was located in M332A–Idaho Batholith, M332F–Challis Volcanics, and 342D–Snake River Basalts and Basins.

Three other hot spots of high fire occurrence density were located in south-central Washington (M242C–Eastern Cascades); south-central New Mexico (321A–Chihuahu Desert Basin and Range, M313B–Sacramento-Monzano Mountains, and M313A–White Mountains-San Francisco Peaks-Mogollon Rim); and south-central Georgia (232B–Gulf Coastal Plains and Flatwoods, 232J–Southern Atlantic Coastal Plains and Flatwoods, and 232L–Gulf Coastal Lowlands).

Several hot spots of moderate fire density were scattered across the Western United States (fig. 3.6), including in:

- Northwestern California (M261A–Klamath Mountains, 263A–Northern California Coast, and M261B–Northern California Coast Ranges)
- Southwestern Oregon (M242A–Oregon and Washington Coast Ranges, M261A–Klamath Mountains, and M242B–Western Cascades)

- Southern California (M262B–Southern California Mountain and Valley and 261B–Southern California Coast)
- Central Washington (M242D–Northern Cascades)
- Southern Nevada (322A–Mojave Desert)
- Northwestern Montana (M333C–Northern Rockies)
- Northern Utah and southern Idaho (342J–Eastern Basin and Range and M331D–Overthrust Mountains)
- Southwestern Colorado (M331G–South-Central Highlands)
- Central Colorado (M331I–Northern Parks and Ranges and 331I–Arkansas Tablelands)

The Getis-Ord hot spot analysis also detected a handful of areas with moderate concentrations of forest fire occurrence density in the Southeast:

- West-central Louisiana (232F–Coastal Plains and Flatwoods-Western Gulf)
- South-central Alabama and northwestern Florida (232B–Gulf Coastal Plains and Flatwoods)
- Central Florida (232G–Florida Coastal Lowlands-Atlantic, 232D–Florida Coastal Lowlands-Gulf, 232K–Florida Coastal Plains Central Highlands, and 411A–Everglades)

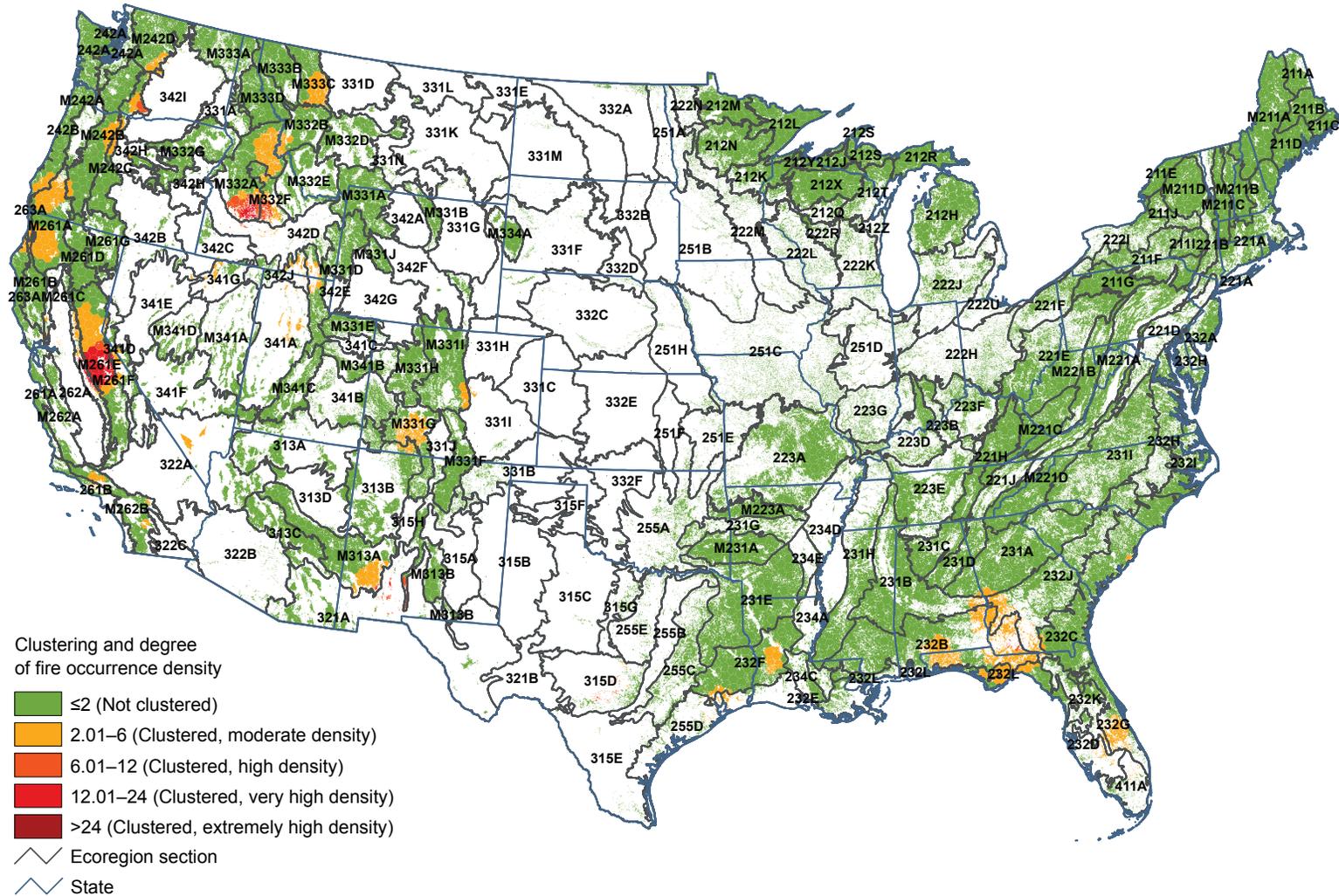


Figure 3.6—Hot spots of fire occurrence across the conterminous United States for 2013. Values are Getis-Ord G_i^* scores, with values >2 representing significant clustering of high fire occurrence densities. (No areas of significant clustering of low fire occurrence densities, <-2 , were detected). 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 Applications Center, in conjunction with the NASA MODIS Rapid Response group)

CONCLUSION

The results of these geographic analyses are intended to offer insights into where fire occurrences have been concentrated spatially in a given year and compared to previous years, but are not intended to quantify the severity of a given fire season. Given the limits of MODIS active fire detection using 1-km² resolution data, these products also may underrepresent the number of fire occurrences in some ecosystems where small and low-intensity fires are common. These products can also have commission errors. However, these high temporal fidelity products currently offer the best means for daily monitoring wildfire impacts. Ecological and forest health impacts relating to fire and other abiotic disturbances are scale-dependent properties, which in turn are affected by management objectives (Lundquist and others 2011). Information about the concentration of fire occurrences may help pinpoint areas of concern for aiding management activities and for investigations into the ecological and socioeconomic impacts of wildland forest fire potentially outside the range of historic frequency.

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INTRODUCTION

Droughts occur regularly in most U.S. forests, but their frequency and intensity vary widely between, as well as within, forest ecosystems (Hanson and Weltzin 2000). In the Western United States, forests commonly experience annual seasonal droughts. In the Eastern United States, forests tend to exhibit one of two prevailing drought patterns: random (i.e., occurring at any time of year), occasional droughts, as typically seen in the Appalachian Mountains and the Northeast; or frequent late-summer droughts, as typically seen in the southeastern Coastal Plain and the eastern edge of the Great Plains (Hanson and Weltzin 2000).

Initially, plants respond to drought stress by decreasing fundamental growth processes such as cell division and enlargement. Photosynthesis, which is less sensitive than these basic processes, decreases slowly when drought stress is low, but more sharply when the stress is moderate to severe (Kareiva and others 1993, Mattson and Haack 1987). Drought stress often makes forests susceptible to attack by tree-damaging insects and diseases (Clinton and others 1993, Mattson and Haack 1987, Raffa and others 2008). Moreover, drought increases wildland fire risk by inhibiting organic matter decomposition and diminishing the moisture content of downed woody materials and other potential fire fuels (Clark 1989, Keetch and Byram 1968, Schoennagel and others 2004).

Forests are generally resistant to short-term drought conditions (Archaux and Wolters 2006),

although individual tree species have differing degrees of resistance (Hinckley and others 1979, McDowell and others 2008). Because of this resistance, the duration of a drought event may be more important than its intensity (Archaux and Wolters 2006); for instance, multiple consecutive years of drought (2 to 5 years) are more likely to cause high tree mortality than a single very dry year (Guarín and Taylor 2005, Millar and others 2007). Therefore, a comprehensive account of drought impact in forested areas should include analysis of moisture conditions over multiyear time windows.

In the 2010 Forest Health Monitoring (FHM) national report, we presented a methodology for mapping drought conditions across the conterminous United States (Koch and others 2013a). Our goal with this methodology was to generate drought-related spatial data sets that are finer in scale than similar products available from sources such as the National Climatic Data Center (2015) or the U.S. Drought Monitor Program (Svoboda and others 2002). The principal inputs are gridded climate data (i.e., monthly raster maps of precipitation and temperature over a 100-year period) created with the Parameter-elevation Regression on Independent Slopes (PRISM) climate mapping system (Daly and others 2002). Notably, the methodology employs a standardized drought-indexing approach that allows us to compare a given location's moisture status during different time windows, regardless of their length. Our drought index is easier to calculate than the commonly used Palmer Drought Severity Index, or PDSI (Palmer

CHAPTER 4.

1-Year (2013), 3-Year (2011–13), and 5-Year (2009–13) Drought Maps for the Conterminous United States

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1965), and sidesteps some criticisms of the PDSI (summarized by Alley 1984) regarding its underlying assumptions and limited comparability across space and time. In this chapter, we apply the methodology to the most currently available climate data (i.e., the monthly PRISM data through 2013), thereby providing a fifth time step in an ongoing annual record of drought status in the conterminous United States from 2009 forward (Koch and others 2013a, 2013b, 2014, 2015).

METHODS

We acquired grids for monthly precipitation and monthly mean temperature for the conterminous United States from the PRISM Climate Group Web site (PRISM Climate Group 2014). At the time of these analyses, gridded data sets were available for all years from 1895 to 2013. However, the grids for December 2013 were only provisional versions (i.e., finalized grids had not yet been released for this month). For analytical purposes, we treated these provisional grids as if they were the final versions. The spatial resolution of the grids was approximately 4 km (cell area = 16 km²). For future applications and to ensure better compatibility with other spatial data sets, all output grids were resampled to a spatial resolution of approximately 2 km (cell area = 4 km²) using a nearest neighbor approach. The nearest neighbor approach is a computationally simple resampling method that avoids the smoothing of data values observed with methods such as bilinear interpolation or cubic convolution.

Potential Evapotranspiration Maps

As in our previous drought mapping efforts (Koch and others 2012a, 2012b, 2013a, 2013b, 2014, 2015), we adopted an approach in which a moisture index value for each location of interest (i.e., each grid cell in a map of the conterminous United States) was calculated based on both precipitation and potential evapotranspiration values for that location during the time period of interest. Potential evapotranspiration measures the loss of soil moisture through plant uptake and transpiration (Akin 1991). It does not measure actual moisture loss, but rather the loss that would occur if there was no possible shortage 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.

To complement the available PRISM monthly precipitation grids, we computed corresponding monthly potential evapotranspiration (*PET*) grids using Thornthwaite's formula (Akin 1991, Thornthwaite 1948):

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

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 degrees 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 m 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 calculation of I and the empirical derivation of a]

Although only a simple approximation, a key advantage of Thornthwaite's formula is that it has modest input data requirements (i.e., mean temperature values) compared to more sophisticated methods of estimating PET such as the Penman-Monteith equation (Monteith 1965), which requires less readily available data on factors such as humidity, radiation, and wind speed. To implement equation 1 spatially, we created a grid of latitude values for determining the L adjustment for any given grid cell (and any given month) in the conterminous United States. We extracted the T_m values for the grid cells from the corresponding PRISM mean monthly temperature grids.

Moisture Index Maps

To estimate baseline conditions, we used the precipitation (P) and PET grids to generate moisture index grids for the past 100 years (i.e., 1914–2013) for the conterminous United States. We used a moisture index described by Willmott and Feddema (1992), which has been applied in a variety of contexts, including global vegetation modeling (Potter and Klooster 1999) and climate change analysis (Grundstein 2009). Willmott and Feddema (1992) devised the index as a refinement of one described earlier by Thornthwaite (1948) and Thornthwaite and Mather (1955). Their revised index, MI' , has the following form:

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

where

P = precipitation

PET = potential evapotranspiration

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

This set of equations yields a symmetric, 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 its moisture index as the mean of the MI' values for all months in the time 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 (i.e., 1914–2013), we used the mean-of-months approach to calculate moisture index grids for three different time windows: 1 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 two years prior through December of the target year, and the MI_5' grids are the mean of the 60 consecutive monthly MI' grids from January four years prior to December of the target year. For example, the MI_1' grid for the year 2013 is the mean of the monthly MI' grids from January through December 2013, while the MI_3' grid is the mean of the grids from January 2011 through December 2013 and the MI_5' grid is the mean of the grids from January 2009 through December 2013.

ANNUAL AND MULTIYEAR DROUGHT MAPS

To determine degree of departure from typical moisture conditions, we first created a normal grid, $MI_{i\ norm}'$, 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 fig. 4.1). We also created a standard deviation grid, $MI_{i\ SD}'$, for each time window, calculated from the window’s 100 individual moisture index grids as well as its $MI_{i\ norm}'$ grid. We subsequently calculated moisture difference z-scores, MDZ_{ij} , for each time window using these derived data sets:

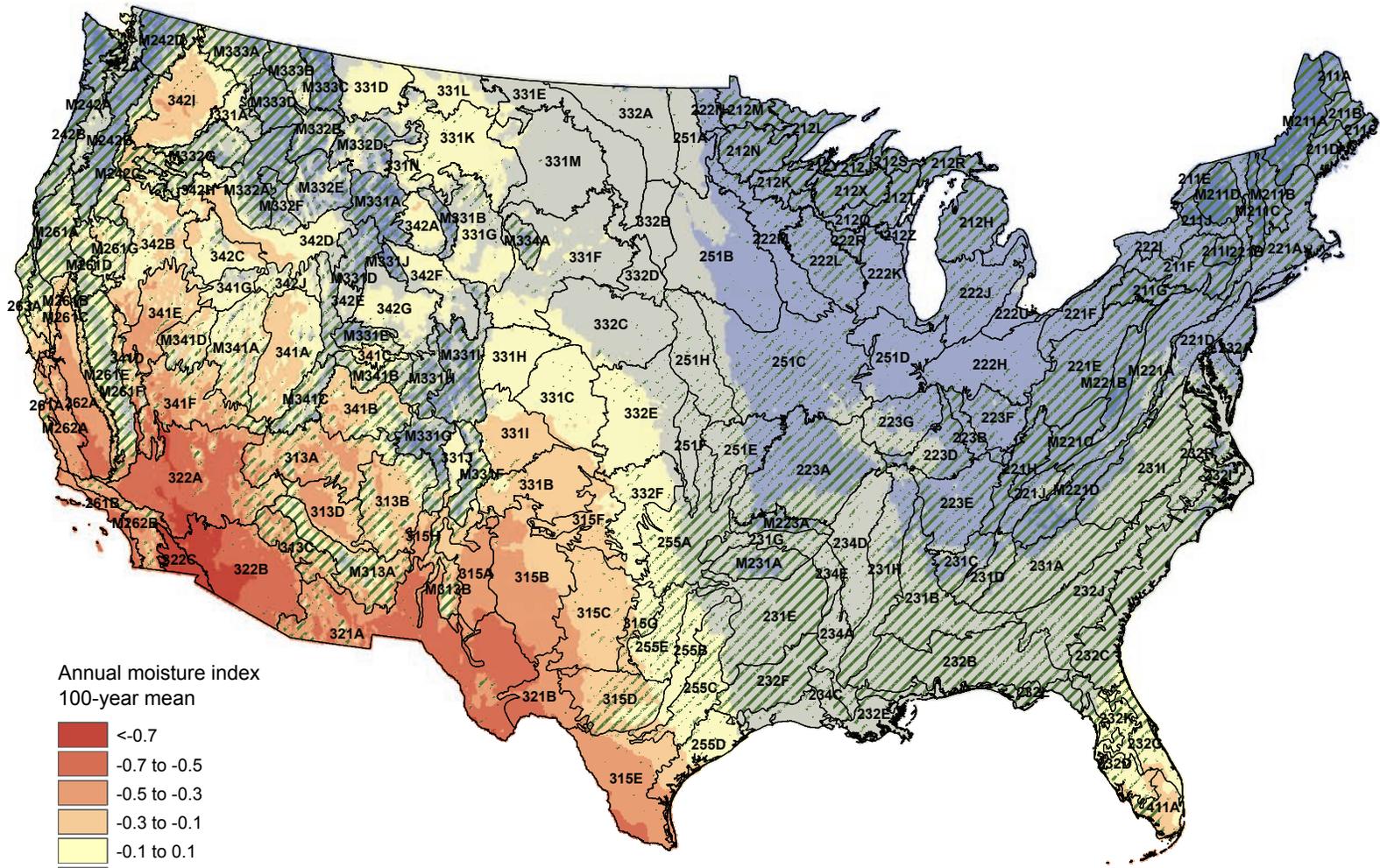
$$MDZ_{ij} = \frac{MI_i' - MI_{i\ norm}'}{MI_{i\ SD}'} \quad (3)$$

where

i = the analytical time window (i.e., 1, 3, or 5 years)

j = a particular target year in our 100-year study period (i.e., 1914–2013)

MDZ scores may be classified in terms of degree of moisture deficit or surplus (table 4.1). The classification scheme includes categories (e.g., severe drought, extreme drought) like those associated with the PDSI. The scheme has also been adopted for other drought indices such as the Standardized Precipitation Index, or SPI (McKee and others 1993). Moreover, the breakpoints between MDZ categories resemble those used for the SPI, such that we expect the MDZ categories to have theoretical frequencies of occurrence that are similar to their SPI counterparts (e.g., approximately 2.3 percent of the time for extreme drought; see McKee and others 1993, Steinemann 2003). More importantly, because of the standardization in equation 3, the breakpoints between categories



Annual moisture index
100-year mean

- <-0.7
- 0.7 to -0.5
- 0.5 to -0.3
- 0.3 to -0.1
- 0.1 to 0.1
- 0.1 to 0.3
- 0.3 to 0.5
- 0.5 to 0.7
- >0.7

- Forested areas
- Ecoregion section boundary

Figure 4.1—The 100-year (1914–2013) mean annual moisture index, or $MI_1'_{norm}$, 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 Moderate Resolution Imaging Spectroradiometer (MODIS) imagery by the U.S. Department of Agriculture Forest Service, Remote Sensing Applications Center. (Data source: PRISM Climate Group, Oregon State University)

Table 4.1—Moisture difference z-score (*MDZ*) value ranges for nine wetness and drought categories, along with each category’s approximate theoretical frequency of occurrence

<i>MDZ</i> 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
-0.5 to 0.5	Near normal conditions	38.2
0.5 to 1	Mild moisture surplus	15.0
1 to 1.5	Moderate moisture surplus	9.2
1.5 to 2	Severe moisture surplus	4.4
>2	Extreme moisture surplus	2.3

remain the same regardless of the size of the time window of interest. For comparative analysis, we generated classified *MDZ* maps of the conterminous United States, based on all three time windows, for the target year 2013. Because our analysis focused on drought (i.e., moisture deficit) rather than surplus conditions, we combined the four moisture surplus categories from table 4.1 into a single category for map display.

RESULTS AND DISCUSSION

The 100-year (1914–2013) mean annual moisture index, or $MI'_{1\text{norm}}$, grid (fig. 4.1) provides an overview of climatic regimes in the conterminous United States. (The 100-year $MI'_{3\text{norm}}$ and $MI'_{5\text{norm}}$ grids did not differ substantially from the mean $MI'_{1\text{norm}}$ grid, and

are not shown here.) Wet climates ($MI' > 0$) are common in the Eastern United States, particularly the Northeast. A noteworthy anomaly is southern Florida, especially ecoregion sections 232G–Florida Coastal Lowlands-Atlantic, 232D–Florida Coastal Lowlands-Gulf, and 411A–Everglades. This region appears to be dry relative to other parts of the East. Although southern Florida usually receives a high level of precipitation over the course of a year, this is countered by a high level of potential evapotranspiration, which results in negative MI' values. This is fundamentally different from the pattern observed in the driest parts of the Western United States, especially the Southwest (e.g., sections 322A–Mojave Desert, 322B–Sonoran Desert, and 322C–Colorado Desert), where potential evapotranspiration is very high but precipitation levels are very low. In fact, dry climates ($MI' < 0$) are typical across much of the Western United States because of generally lower precipitation than the East. Nevertheless, mountainous areas in the central and northern Rocky Mountains as well as the Pacific Northwest are relatively wet, such as ecoregion sections M242A–Oregon and Washington Coast Ranges, M242B–Western Cascades, M331G–South-Central Highlands, and M333C–Northern Rockies. This may be driven in part by large amounts of winter snowfall in these regions.

Figure 4.2 shows the annual (i.e., 1-year) *MDZ* map for 2013 for the conterminous United States. Much of the country experienced moisture surplus conditions during the year. The southern portion of the Pacific Coast was a prominent exception, as most ecoregion

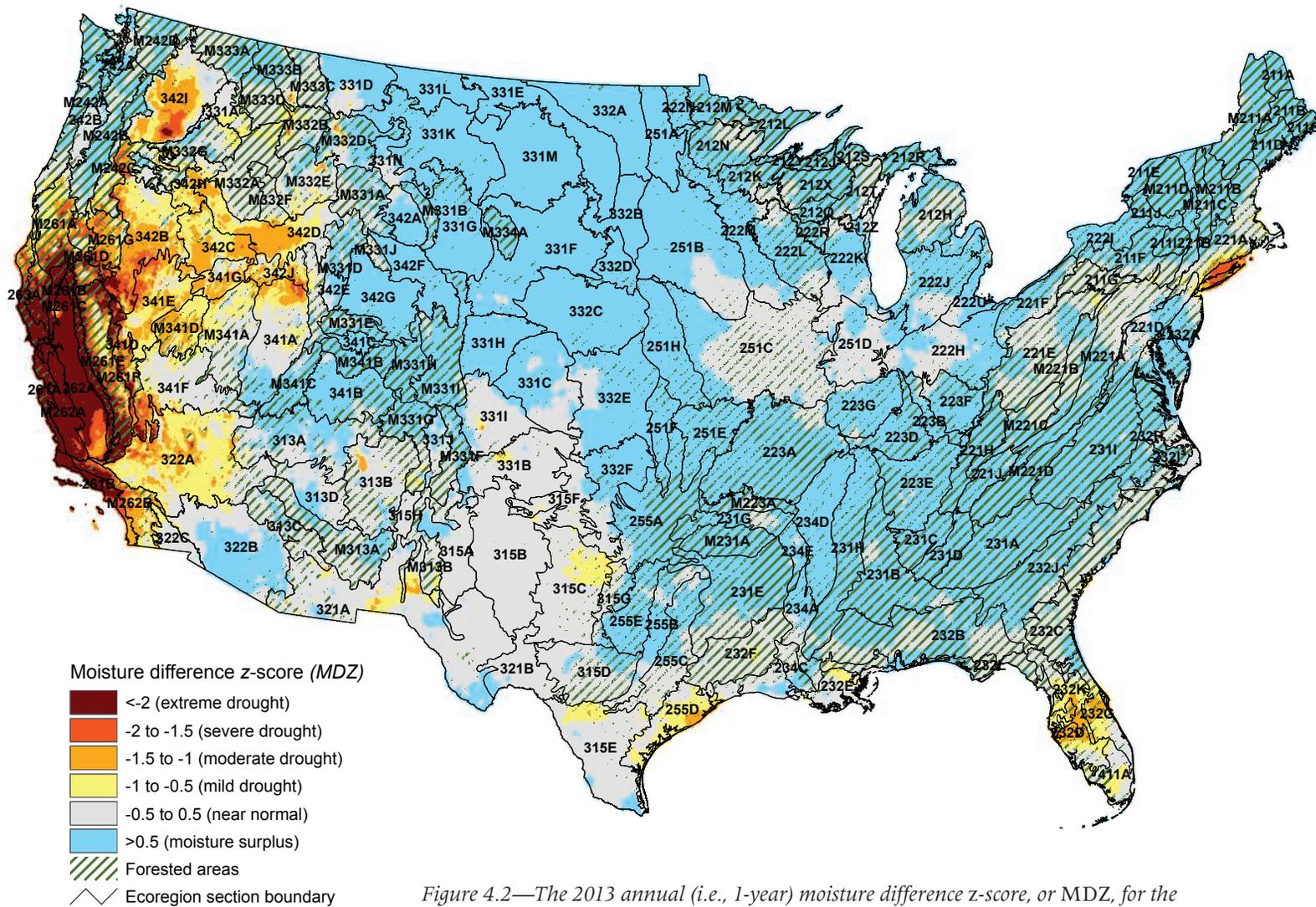


Figure 4.2—The 2013 annual (i.e., 1-year) moisture difference z-score, or MDZ, for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries and labels are included for reference. Forest cover data (overlaid green hatching) derived from MODIS imagery by the U.S. Department of Agriculture Forest Service, Remote Sensing Applications Center. (Data source: PRISM Climate Group, Oregon State University)

sections (Cleland and others 2007) falling at least partially within the State of California experienced extreme drought conditions ($MDZ < -2$) in 2013. Forested sections in this region that contained sizeable areas of extreme drought included M261A–Klamath Mountains, M261B–Northern California Coast Ranges, M261E–Sierra Nevada, M262B–Southern California Mountain and Valley, 263A–Northern California Coast, and 341D–Mono. In addition, extreme drought conditions extended across all of section 261A–Central California Coast and nearly all of M261F–Sierra Nevada Foothills. Furthermore, other forested sections in this region, such as M261D–Southern Cascades and M261G–Modoc Plateau, were subjected to moderate or worse drought conditions ($MDZ < -1$) during 2013. Moderate or worse drought conditions also occurred in portions of neighboring geographic regions (i.e., the Pacific Northwest and the Intermountain West), but were typically limited to areas with little or no forest; the only ecoregion sections where moderate or worse drought conditions and substantial forest cover coincided were M341D–West Great Basin and Mountains and the western portion of 342H–Blue Mountain Foothills.

Although droughts have been common throughout the West during recent decades (Andreadis and Lettenmaier 2006, Andreadis and others 2005, Groisman and Knight 2008), the conditions in California in 2013 were unusually severe. It was the State’s driest calendar year in the historical record, which

began in 1895 (National Climatic Data Center 2014b). The city of San Francisco received only 142 mm of precipitation over the course of the year, while Los Angeles received only 91 mm (National Climatic Data Center 2014c). These amounts represented 24 percent of each city’s normal annual precipitation.

The conditions in California and adjacent regions are especially striking because so few geographic hot spots of moderate or worse drought conditions appeared elsewhere in forested portions of the United States. Two hot spots worth mentioning are an area of moderate drought in central Florida, extending across sections 232G–Florida Coastal Lowlands-Atlantic, 232D–Florida Coastal Lowlands-Gulf, and 232K–Florida Coastal Plains Central Highlands, as well as an area of moderate to extreme drought in section 221A–Lower New England, primarily along New York’s Long Island and the Connecticut coast. On balance, 2013 was a wet year relative to historical data. The percentage of the conterminous United States with moderate or worse drought conditions based on the Palmer Drought Severity Index reached a low of 13.4 percent in October, but increased somewhat to 18.5 percent by the end of December (National Climatic Data Center 2014b).

Figure 4.3 shows a map of the change in MDZ category between 2012 and 2013 for the conterminous United States. The depicted increases and decreases reference the MDZ categories listed in table 4.1. As was the case

for figure 4.2, all of the moisture surplus categories in table 4.1 have been combined into a single category, yielding a six-point scale from extreme drought to moisture surplus. Thus, a five-category decrease indicates a change from moisture surplus in 2012 to extreme drought in 2013, while a five-category increase indicates a change from extreme drought to moisture surplus. The other map classes depict less extreme changes between years. For instance, a two-category decrease represents one of four possibilities: a change from moisture surplus to mild drought, from near-normal conditions to moderate drought, from mild to severe drought, or from moderate drought in 2012 to extreme drought in 2013.

Most of the conterminous United States displayed an increase in *MDZ* from 2012 to 2013 (fig. 4.3). In the Central United States, several large, contiguous areas exhibited a four- or five-category increase in *MDZ*, indicating widespread recovery from drought conditions between 2012 and 2013. Forested ecoregion sections that were in these contiguous areas included M331B–Bighorn Mountains, M331H–North Central Highlands and Rocky Mountains, M334A–Black Hills, M341B–Tavaputs Plateau, and several others. In contrast, the four- or five-category decrease in *MDZ* exhibited by most ecoregion sections in California indicates that the severe drought conditions observed in 2013 were preceded by near-normal or surplus moisture conditions throughout much of the State in 2012.

The 3-year (2011–13) (fig. 4.4) and 5-year (2009–13) (fig. 4.5) *MDZ* maps document the recent history of moisture conditions in the conterminous United States. Both maps, but especially the 3-year map (fig. 4.4), reflect some impact from the drought conditions that occurred in California in 2013. More broadly, the maps also suggest that drought conditions have persisted throughout the Southwestern United States for the last several years; both maps display extensive areas of severe to extreme drought ($MDZ < -1.5$) in this region. Indeed, intense and widespread droughts have occurred regularly in the Southwest for more than two decades (Groisman and Knight 2008; Mueller and others 2005; National Climatic Data Center 2010, 2011). The most severe multiyear drought conditions appear to have occurred in areas with little forest cover (e.g., 313D–Painted Desert, 331I–Arkansas Tablelands, and 341E–Northern Mono), but there are notable counter-examples, such as M313B–Sacramento-Monzano Mountains.

Elsewhere, both maps show a large area of extreme drought along the Gulf of Mexico coast in Texas and Louisiana, particularly in the heavily forested ecoregion section 232F–Coastal Plains and Flatwoods-Western Gulf. Both States were historically dry in 2011 and had record heat waves during the summer of that year (National Climatic Data Center 2012). A modest reduction in the area of extreme drought shown in the 3-year map (fig. 4.4) versus the 5-year map (fig. 4.5) indicates that moisture conditions improved

in this region during 2012 and 2013. This improvement is partially captured in the 1-year *MDZ* map for 2013 (fig. 4.2).

The hot spot of moderate drought in central Florida that appeared in the 1-year *MDZ* map also appeared in the 3- and 5-year maps, but with higher levels of drought severity. Drought conditions also extended over a larger area of the State in the multiyear maps. This seems consistent with the historical record; according to long-term climatological rankings (National Climatic Data Center 2014a), the period from July 2010 to June 2011 was Florida's driest 12-month period to date (i.e., since 1895). In contrast, although Florida experienced somewhat dry conditions at the beginning and end of 2013, this was countered by historically wet conditions during June and July.

Future Efforts

If the appropriate spatial data (i.e., high-resolution maps of precipitation and temperature) remain available for public use, we will continue to produce our 1-year, 3-year, and 5-year *MDZ* maps of the conterminous United States as a regular yearly component of national-scale forest health reporting. However, users should interpret and compare the *MDZ* drought maps presented here cautiously. Although the maps use a standardized index scale that applies regardless of the size of the time window, the window size may still merit some consideration; for example, an extreme drought that persists over a 5-year period has substantially different forest health implications than an extreme drought over a

1-year period. Furthermore, while the 1-year, 3-year, and 5-year *MDZ* maps may together provide a comprehensive short-term overview, it may also be important to consider a particular region's longer term drought history when assessing the current health of its forests. For example, in geographic regions where droughts have historically occurred on a frequent (e.g., annual or nearly annual) basis, certain tree species may be better adapted to a regular lack of available moisture (McDowell and others 2008). Because of this variability in species' drought tolerance, a long period of persistent and severe drought conditions could ultimately lead to changes in a region's forest composition (Mueller and others 2005). In future work, we hope to provide forest managers and other decisionmakers with better quantitative evidence regarding critical relationships between drought and significant forest health impacts such as regional-scale tree mortality (e.g., Mitchell and others 2014). We also intend to examine the role of drought as an inciting factor for other forest threats such as wildfire or pest outbreaks.

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INTRODUCTION

Tree mortality is a natural process in all forest ecosystems. However, extremely high mortality 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 forest in a particular region is made up of older, senescent stands.

The mission of the Forest Health Monitoring (FHM) Program is to monitor, assess, and report on the status, changes, and long-term trends in forest ecosystem health in the United States (FHM 1994). Thus, the approach to mortality presented here seeks to detect 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. However, sometimes the proximate cause of mortality may be discernable. In such cases, the cause of mortality is reported, both because it is of interest in and of itself to many readers and because understanding such proximate causes of mortality might provide insight into whether the mortality is within the range of natural variation or reflects more fundamental changes to ecological processes.

DATA

Mortality is analyzed using Forest Inventory and Analysis (FIA) phase 2 (P2) data. FIA P2 data are collected across forested land throughout

the United States, with approximately 1 plot per 6,000 acres of forest, using a rotating panel sample design (Bechtold and Patterson 2005). Field plots are divided into spatially balanced panels, with one panel measured each year. A single cycle of measurements consists of measuring all panels. This “annualized” method of inventory was adopted, State by State, beginning in 1999. Any analysis of mortality requires data collected at a minimum of 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). For this report, the repeated P2 data were available for all of the Central and Eastern States, and data for some States include a third cycle of measurements (i.e., a third measurement of the plots).

Once all P2 plots have been remeasured in a State, mortality estimates generally will be based on a sample intensity of approximately 1 plot per 6,000 acres of forest.¹ However, at this time not all plots have been remeasured in all the States included in this analysis. When not all plots have been remeasured, mortality estimates are based on a lower effective sample intensity. Table 5.1 shows the 37 States from which consistent, repeated P2 measurements were available,

¹ In some States more intensive sampling has been implemented. See table 5.1 for details.

CHAPTER 5.

Tree Mortality

MARK J. AMBROSE

Table 5.1—States from which repeated Forest Inventory and Analysis (FIA) phase 2 measurements were available in 2013, the time period spanned by the data, and the effective sample intensity (based on plot density and proportion of plots that had been remeasured) in the available data sets

Time period	States	Effective sample intensity	Proportion of plots measured three times
1999–2012	IN	1 plot: 6,000 acres	3/5
1999–2012	ME	1 plot: 6,000 acres	4/5
1999–2012	WI	1 plot: 3,000 acres ^a	3/5
1999–2012	MN	1 plot: 3,000 acres ^a	3/5
1999–2012	MO	1 plot: 6,000 acres ^b	3/5
2000–2012	PA, VA	1 plot: 6,000 acres	3/5
2000–2012	IA	1 plot: 6,000 acres	3/5
2000–2012	MI	1 plot: 2,000 acres ^c	3/5
2000–2012	AR	1 plot: 6,000 acres	2/5
2001–2012	OH	1 plot: 6,000 acres	1/5
2001–2012	TX ^d	1 plot: 6,000 acres	4/5
2001–2012	GA, KS, NE	1 plot: 6,000 acres	2/5
2001–2011	TN	1 plot: 6,000 acres	1/5
2001–2012	LA	1 plot: 10,500 acres	0
2001–2012	AL	1 plot: 6,000 acres	0
2001–2012	IL, ND, SD	1 plot: 6,000 acres	2/5
2002–2012	FL	1 plot: 7,500 acres	0
2002–2012	SC	1 plot: 6,000 acres	1/5
2002–2011	KY	1 plot: 7,500 acres	0
2002–2012	NY	1 plot: 6,000 acres	0
2002–2012	NH	1 plot: 6,000 acres	0
2003–2012	CT, MA, RI, VT	1 plot: 6,000 acres	0
2003–2012	NC	1 plot: 10,500 acres	0
2004–2012	DE, MD, NJ, WV	1 plot: 7,500 acres	0
2006–2012	MS	1 plot: 10,500 acres	0
2008–2012	OK ^e	1 plot: 15,000 acres	0

^a In Minnesota and Wisconsin, the phase 2 inventory was done at twice the standard FIA sample intensity, approximately 1 plot per 3,000 acres.

^b In Missouri, the phase 2 inventory was done at twice the standard FIA sample intensity, approximately 1 plot per 3,000 acres, on national forest lands, and at the standard intensity of 1 plot per 6,000 acres on all other lands.

^c In Michigan, the phase 2 inventory was done at triple the standard FIA sample intensity, approximately 1 plot per 2,000 acres.

^d Annualized growth and mortality data were only available for eastern Texas.

^e Annualized growth and mortality data were only available for eastern Oklahoma.

the time period spanned by the data, and the effective sample intensity. Also shown is the proportion of plots measured for a third time. The States included in this analysis, as well as the forest cover within those States, are shown in figure 5.1.

Because the data used here are collected using a rotating panel design and all available annualized data are used, the majority of data used in this mortality analysis were also used in the analysis presented in the previous FHM national report (Ambrose 2015). Using the data in this way, it would be very unusual to see any great changes in mortality patterns from one annual report to the next. Nevertheless, it is important to look at mortality patterns every year so as not to miss detecting mortality patterns that may be indicative of forest health problems as soon as they become discernable.

METHODS

The methods used in this analysis were developed for earlier FHM national reports (2001–04) using FHM and FIA phase 3 (P3) data. FIA P2 tree (≥ 5 inches d.b.h.) and sapling (1 inch \leq d.b.h. < 5 inches) data were used to estimate average annual tree mortality in terms of tons of aboveground biomass per acre. The data were obtained from the public FIA Database-version 5.1 (USDA Forest Service 2011). The biomass represented by each tree was calculated by FIA and provided in the FIA Database (USDA Forest Service 2013). 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 2005b). Gross growth rate and mortality rate, in terms of tons of biomass per acre, were independently calculated for each ecoregion section (Cleland and others 2007, McNab and others 2007) 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 estimation using data where 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 the Forest Health Monitoring 2001 National Technical Report (Coulston and others 2005c) and appendix A—Supplemental Methods in the Forest Health Monitoring 2003 National Technical Report (Coulston and others 2005a).

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

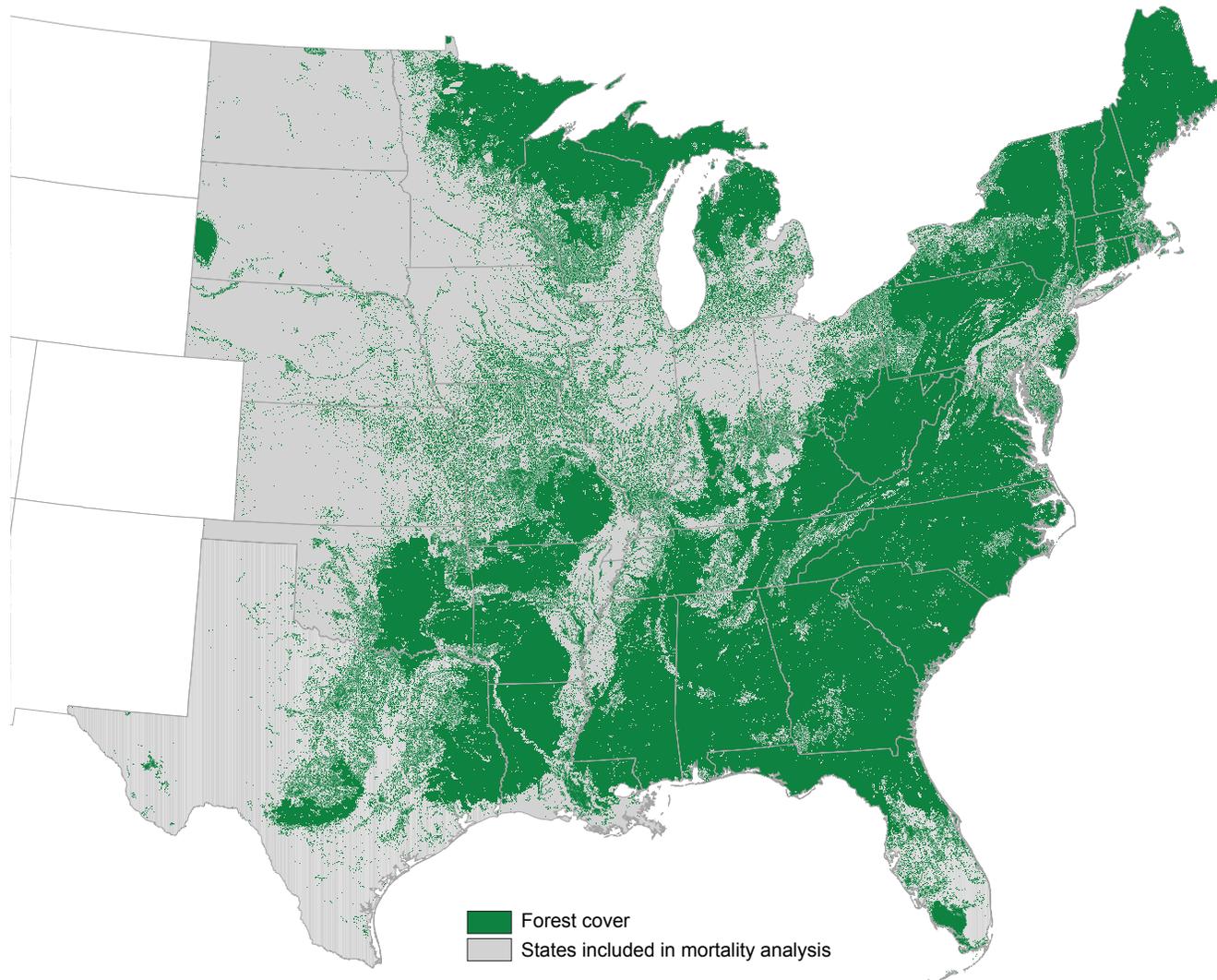


Figure 5.1—Forest cover in the States where mortality was analyzed. Forest cover was derived from Moderate Resolution Imaging Spectroradiometer (MODIS) satellite imagery (USDA Forest Service 2008).

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 DDL.

To further analyze tree mortality, the number of stems and the total biomass of trees that 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 may be affecting the forests. Although determining particular causal agents associated with all 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.

Finally, a biomass-weighted mean mortality age was calculated by ecoregion and species. For each species experiencing mortality in an ecoregion, the mean stand age was calculated, weighted by the dead biomass on the plot. This value gives a rough indicator of the average age of the stands in which trees died. However, the age of individual trees may differ significantly from the age assigned to a stand by FIA field crews, especially in mixed species stands. When the age of trees that die is relatively low compared with the age at which trees of a particular species usually become senescent, it suggests that some pest, pathogen, or other forest health problem may be affecting the forest.

RESULTS AND DISCUSSION

The MRATIO values are shown in figure 5.2. 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 >1 indicates that mortality exceeds growth and live standing biomass is actually decreasing.

The highest MRATIOS occurred in ecoregion sections 331F–Western Great Plains (MRATIO = 1.35) in South Dakota and Nebraska and 332F–South-Central and Red Bed Plains (MRATIO = 1.16) in southern Kansas, where mortality actually exceeded growth. Other areas of high mortality relative to growth were sections 332D–North-Central Great Plains, also in South Dakota and Nebraska (MRATIO = 0.98); M334A–Black Hills (MRATIO = 0.87); 234A–Southern Mississippi Alluvial Plain in Louisiana, Mississippi, and Arkansas (MRATIO = 0.86); and 232E–Louisiana Coastal Prairies and Marshes (MRATIO = 0.69). Table 5.2 shows the tree species experiencing the greatest mortality in those ecoregions.

The results of the analysis of the relative sizes of trees that died to those that lived, the DDL ratio, are shown in table 5.3. The DDL ratio is a plot-level indicator, so I obtained summary statistics for the ecoregions where mortality relative to growth was highest. In all cases, the mean and median DDLs were rather close to 1, meaning that the trees that died were similar in size to the trees that survived. However, there were some plots with extremely high DDL values. Interestingly, the same pattern of

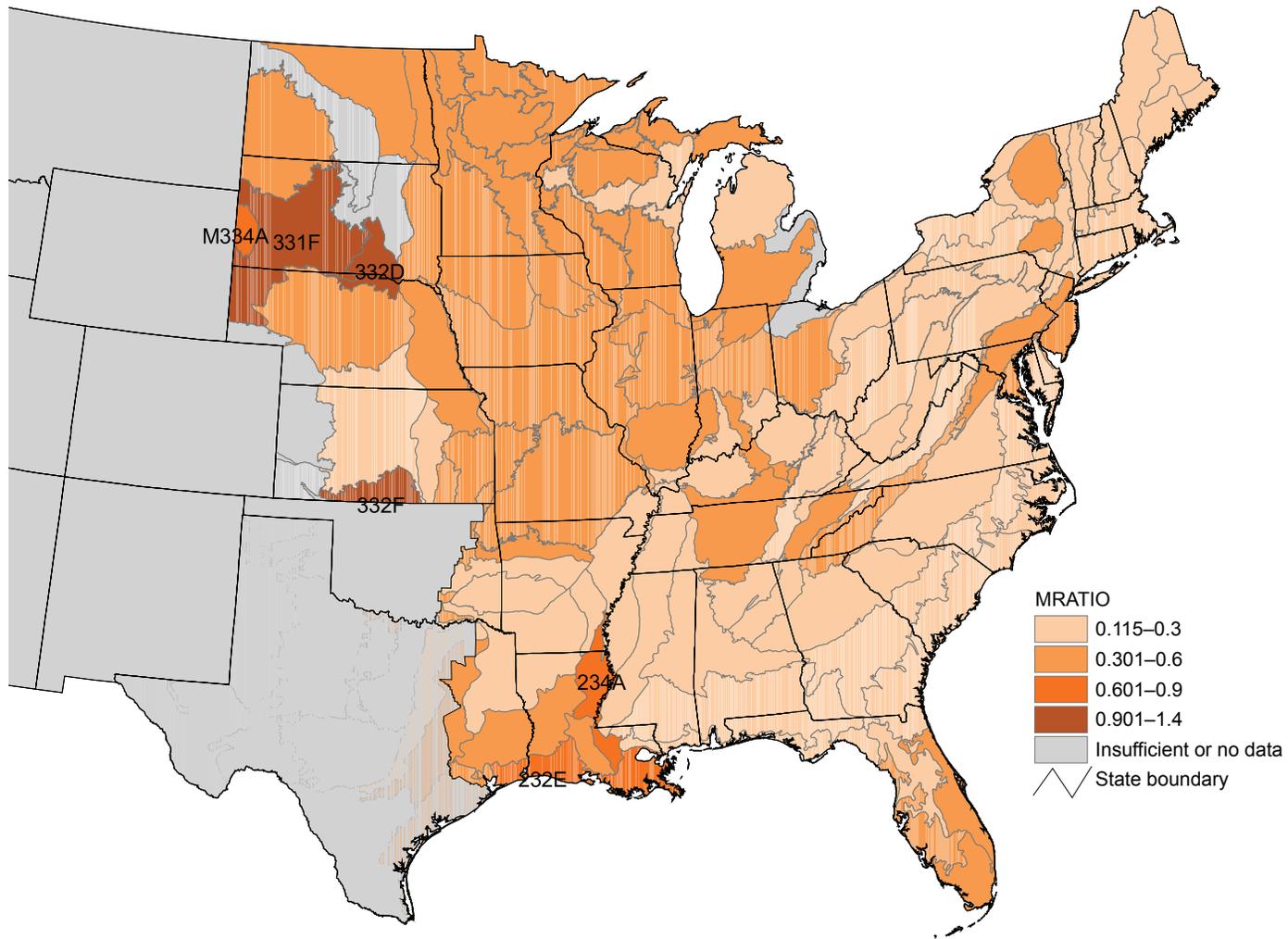


Figure 5.2—Tree mortality expressed as the ratio of annual mortality of woody biomass to gross annual growth in woody biomass (MRATIO) by ecoregion section (Cleland and others 2007). Ecoregions with high MRATIOS are identified by section number. (Data source: U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis Program)

Table 5.2—Tree species responsible for least 5 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)	
232E—Louisiana Coastal Prairies and Marshes	0.69	Chinese tallowtree (<i>Sapium sebiferum</i>)	20.22	29	11.54	13.20
		Laurel oak (<i>Quercus laurifolia</i>)	13.89	51	32.49	16.03
		Loblolly pine (<i>Pinus taeda</i>)	12.71	26	4.99	7.21
		Willow oak (<i>Quercus phellos</i>)	9.86	46	26.89	7.61
		Water oak (<i>Quercus nigra</i>)	8.99	35	9.45	15.06
		Swamp chestnut oak (<i>Quercus michauxii</i>)	6.73	45	58.53	6.25
234A—Southern Mississippi Alluvial Plain	0.86	Black willow (<i>Salix nigra</i>)	73.06	38	68.02	50.75
		Water oak (<i>Quercus nigra</i>)	10.41	25	22.19	29.12
331F—Western Great Plains	1.35	Ponderosa pine (<i>Pinus ponderosa</i>)	62.50	54	8.36	10.40
		Green ash (<i>Fraxinus pennsylvanica</i>)	14.98	44	15.13	12.83
		Eastern cottonwood (<i>Populus deltoides</i>)	10.22	61	4.31	21.89
332D—North-Central Great Plains	0.98	Eastern cottonwood (<i>Populus deltoides</i>)	23.88	85	10.80	4.50
		American elm (<i>Ulmus americana</i>)	19.46	49	26.49	22.00
		Ponderosa pine (<i>Pinus ponderosa</i>)	17.65	44	22.18	30.72
		Bur oak (<i>Quercus macrocarpa</i>)	12.93	63	3.21	4.87
		Green ash (<i>Fraxinus pennsylvanica</i>)	8.80	62	15.15	16.24
		Hackberry (<i>Celtis occidentalis</i>)	7.34	60	10.24	0.72
		Eastern redcedar (<i>Juniperus virginiana</i>)	5.54	43	3.89	6.90
332F—South-Central and Red Bed Plains	1.16	Eastern cottonwood (<i>Populus deltoides</i>)	22.53	37	9.95	5.78
		Black willow (<i>Salix nigra</i>)	21.57	55	57.59	62.50
		Black locust (<i>Robinia pseudoacacia</i>)	19.49	42	23.59	39.25
		American elm (<i>Ulmus americana</i>)	13.78	39	28.11	2.87
		Red mulberry (<i>Morus rubra</i>)	6.04	51	3.92	2.05
		Hackberry (<i>Celtis occidentalis</i>)	5.88	44	10.22	1.93
M334A—Black Hills	0.87	Ponderosa pine (<i>Pinus ponderosa</i>)	83.91	70	5.10	8.08
		Quaking aspen (<i>Populus tremuloides</i>)	7.83	74	28.74	28.06

MRATIO = ratio of annual mortality of woody biomass to gross annual growth in woody biomass.

^a Ages (in years) are estimated from the stand age as determined by the 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.

Table 5.3—Dead diameter–live diameter (DDL) ratios for ecoregion sections where the MRATIO was 0.60 or greater

Ecoregion section	Mean DDL	Maximum DDL	Median DDL	Minimum DDL
232E–Louisiana Coastal Prairies and Marshes	1.11	2.91	0.85	0.32
234A–Southern Mississippi Alluvial Plain	1.01	3.45	0.70	0.26
331F–Western Great Plains	0.99	3.29	0.92	0.08
332D–North-Central Great Plains	1.04	5.38	0.89	0.15
332F–South-Central and Red Bed Plains	1.17	3.11	1.20	0.14
M334A–Black Hills	1.04	7.02	0.77	0.16

MRATIO = ratio of annual mortality of woody biomass to gross annual growth in woody biomass.

mean and median DDL close to 1 and some high DDL values was observed in nearly all ecoregions, regardless of the overall mortality level.

In three of the ecoregion sections exhibiting highest mortality relative to growth (331F–Western Great Plains, 332D–North-Central Great Plains, and 332F–South-Central and Red Bed Plains), the predominant vegetation is grassland (see the forest cover in fig. 5.1), and there were relatively few forested plots measured (98 plots in region 331F, 58 plots in region 332D, and 26 plots in region 332F). Both ecoregions 331F and 332D have had high mortality relative to growth in recent years (Ambrose 2013, 2014, 2015), so the observed mortality is not a new phenomenon. Tree growth rates in these regions (especially in 331F) are quite low, so the high MRATIOS are due to a combination of low growth and high mortality. Much of the forest in these sections is riparian, and, indeed, most

of the species experiencing greatest mortality (table 5.2) are commonly found in riparian areas. The one exception was high ponderosa pine (*Pinus ponderosa*) mortality in ecoregion section 331F–Western Great Plains. Ponderosa pine is not a riparian tree species, but like the riparian species, it only occurs in a relatively small area of the ecoregion, on discontinuous mountains, plateaus, canyons, and breaks in the plains (Burns and Honkala 1990).

Within the three ecoregions exhibiting the highest mortality relative to growth, DDL values vary widely. There are a small number of plots with high DDLs, and these plots represent most of the biomass that died in these sections. However, on many of these plots the overall level of mortality is fairly low, as would be the case when remnant larger trees die, leaving young, vigorous stands behind. Tree growth is generally slow in these ecoregion sections because of naturally dry conditions. Where the number of

sample plots is small and tree growth is slow, care must be taken in interpreting mortality relative to growth over short time intervals.

In ecoregion section M334A–Black Hills, by far the largest amount of biomass that died was ponderosa pine (table 5.2); however, this represented a relatively small proportion of the ponderosa pine in the ecoregion (about 8 percent of ponderosa pine stems and 5 percent of biomass). This pine mortality is very likely related to mountain pine beetle (*Dendroctonus ponderosae*). There has been an ongoing pine beetle outbreak in the Black Hills (South Dakota Department of Agriculture 2011, 2012) and mountain pine beetle mortality has been reported in western Nebraska (Nebraska Forest Service 2011, 2012). In contrast, aspen (*Populus tremuloides*) mortality made up a much smaller portion of total mortality in the ecoregion, but it represented a much higher mortality rate in aspen (about 28 percent of aspen, in terms of both stems and biomass, died). This suggests that aspen may be affected by more serious forest health issues.

In the adjacent ecoregion section 331F–Western Great Plains, where the MRATIO was highest, ponderosa pine also made up the majority of trees that died (63 percent). Here, too, this mortality represented a relatively small proportion of the ponderosa pine (biomass and stems) in the region. Green ash (*Fraxinus pennsylvanica*), which made up less than one quarter of the ecoregion mortality as ponderosa

pine, suffered a slightly larger proportional loss of the total ash stock.

In ecoregion section 332D–North-Central Great Plains, seven species experienced high total mortality in terms of biomass and together represent over 90 percent of the mortality in the ecoregion: eastern cottonwood (*Populus deltoides*), American elm (*Ulmus americana*), ponderosa pine, bur oak (*Quercus macrocarpa*), green ash, hackberry (*Celtis occidentalis*), and eastern redcedar (*Juniperus virginiana*) (table 5.2). Of these, ponderosa pine and American elm suffered the largest proportional loss in terms of both biomass and number of stems and, together with eastern cottonwood, made up the largest proportion of total mortality. In the case of hackberry, the mortality in terms of biomass (10.24 percent) was much higher than the mortality in terms of number of stems (0.72 percent), which means that the trees that died were a relatively small number of very large trees. A number of different factors may be responsible for the high mortality in the ecoregion. Drought in 2012, as well as associated winter desiccation, has been reported as affecting much of South Dakota and Nebraska. Dutch elm disease has been responsible for elm mortality in both States as well (South Dakota Department of Agriculture 2012, Nebraska Forest Service 2012). Cedar bark beetles (*Phloeosinus* spp.) and juniper blight have been reported as affecting eastern redcedar in South Dakota (South Dakota Department of Agriculture 2012). Green ash has been affected by ash/lilac borer (*Podosesia*

syringae) in South Dakota (South Dakota Department of Agriculture 2012). In addition, a variety of insects and disease has been reported as affecting ponderosa pine in South Dakota and Nebraska; their activity may have produced increased mortality in trees stressed by drought conditions.

In ecoregion 332F–South-Central and Red Bed Plains in south-central Kansas, a wide range of species suffered high mortality, including eastern cottonwood, black willow (*Salix nigra*), black locust (*Robinia pseudoacacia*), American elm, red mulberry (*Morus rubra*), and hackberry. It is unlikely that a single pest or pathogen would produce mortality in this range of species. The most likely factor associated with this mortality is drought. Both 2011 and 2012 were extremely dry years in most of Kansas, with the areas of most severe drought including this ecoregion (Kansas Forest Service 2011, 2012). Such severe drought could lead to tree mortality either directly or by stressing the trees so that they succumb to pests or pathogens that would normally be nonlethal.

In section 232E–Louisiana Coastal Prairies and Marshes, the species experiencing the highest mortality was Chinese tallowtree (*Sapium sebiferum*). This species is an invasive exotic, so the tallowtree mortality is not necessarily a bad thing. However, the high mortality may indicate some stressors affecting the forest more generally that may be of concern in the future. The other species having high mortality in the ecoregion include loblolly pine (*Pinus taeda*) and several bottomland oak species. In ecoregion section 234A–Southern Mississippi Alluvial Plain, the

species experiencing high mortality were also bottomland species: black willow and water oak (*Quercus nigra*).

Drought may be at least partially responsible for the mortality observed in these two ecoregions. During 2010 and 2011, much of Louisiana, including the areas of ecoregions 234A and 232E, suffered from severe drought (Koch and others 2014; National Climatic Data Center 2012a), and 2011 was the seventh driest year on record in Louisiana. In addition, 2011 was an extremely warm year across much of the continental United States, and Louisiana experienced its warmest summer on record in 2011 (National Climatic Data Center 2012b). Such severe and extended drought may be responsible for tree mortality either directly or by weakening trees so that they succumb to insects or disease that might otherwise be nonlethal.

This analysis shows that in most of the Eastern and Central United States, mortality has been low relative to tree growth. Mortality has been rather low in most of the areas for which data are available. The areas of highest recent mortality occurred in the mostly riparian forests of Great Plains ecoregions. A common characteristic of many of the ecoregions having high mortality, those on the Great Plains, is that they are on the margins of land suitable for forest growth, being very dry. Thus, they tend to be vulnerable to changes in weather patterns that might produce prolonged and/or extreme drought. Other areas having high mortality, those in the Gulf Coast, have much wetter climates but also experienced extreme drought

conditions in recent years. A variety of other biotic or abiotic stressors, together with drought, likely have had a role in the mortality observed.

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SECTION 2.

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

OVERVIEW

The 2013–2027 National Insect and Disease Forest Risk Assessment represents a nationwide strategic assessment of the hazard of tree mortality due to insects and diseases displayed as a series of maps, the National Insect and Disease Risk Maps (NIDRM) (Krist and others 2014) (figs. 6.1, 6.2, and 6.3). Risk, or more appropriately termed, *hazard*, is defined in the assessment as *the expectation that, without remediation, at least 25 percent of standing live basal area >1 inch in diameter will die over a 15-year time frame (2013–27) due to insects and diseases.*

NIDRM is more than just maps. It is a nationwide, science-based, administrative planning tool that is the product of a process whereby, every 5 years, the forest health community works together to determine the severity and extent of tree-mortality hazard due to insects and diseases. NIDRM was developed within a highly collaborative process led by the Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture, with participation from FHM staffs from all regions, State forestry agencies, the Forest Service's Forest Health Protection Program, and Forest Service Research and Development.

NIDRM represents 186 individual insect and disease hazard models, integrated within a common Geographic Information System-based, multicriteria framework that can account for

regional variations in forest health. Applied to all 50 States, and based on the best-available science and data, NIDRM's modeling process provides a consistent, repeatable, transparent, peer-reviewed process through which interactive spatial and temporal hazard assessments can be conducted (Krist and others 2014). This process is consistent with the 2006 effort (Krist and others 2007), allowing for flexible analysis to produce hazard assessments for specific insects and diseases, and can be used to inform other agency assessments such as the Integrated Resource Restoration, Watershed Condition Framework, Terrestrial Ecosystem Condition Assessment, Existing Vegetation Classification Mapping and Inventory, and Hazardous Fuels Prioritization Allocation System.

NIDRM products are compiled on a national extent with a 240-m spatial resolution (approximately 14 acres) and can be updated as new data and/or models become available. This “live” or near-real-time approach will greatly facilitate the production of new hazard maps. This chapter provides a brief overview of NIDRM and describes how geospatial and tabular data from this 2013–27 National Insect and Disease Forest Risk Assessment can be accessed. The full report (Krist and others 2014) and associated data are available from the NIDRM Web site at <http://www.fs.fed.us/foresthealth/technology/nidrm.shtml>.

CHAPTER 6.

2013–2027 National Insect and Disease Forest Risk Assessment: Summary and Data Access

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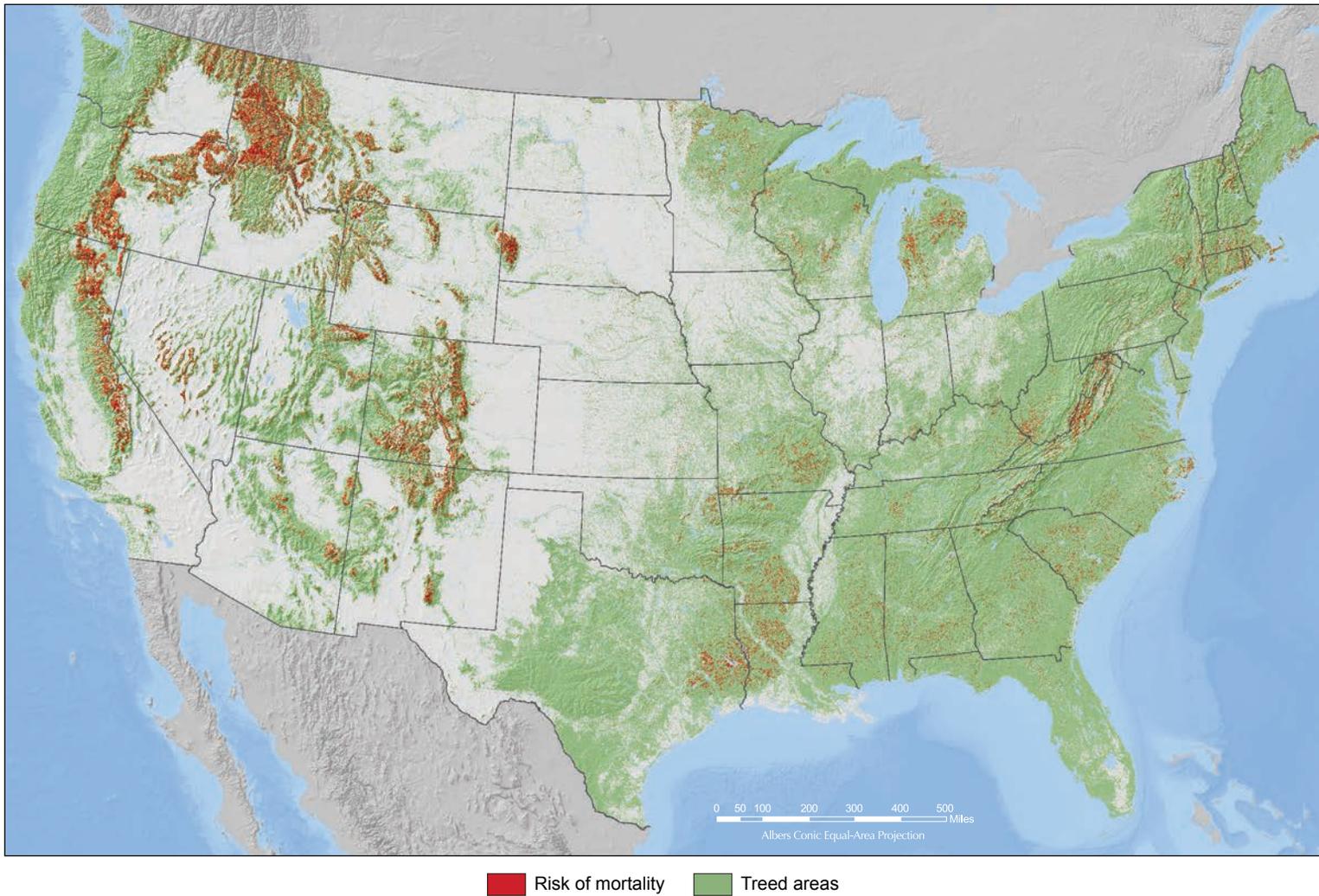


Figure 6.1—The 2013–27 national insect and disease composite risk map for the conterminous United States. Risk, or more appropriately termed, “hazard,” is defined as the expectation that, without remediation, at least 25 percent of the standing live basal area >1 inch in diameter will die over a 15-year time frame (2013–27) due to insects and diseases. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection)

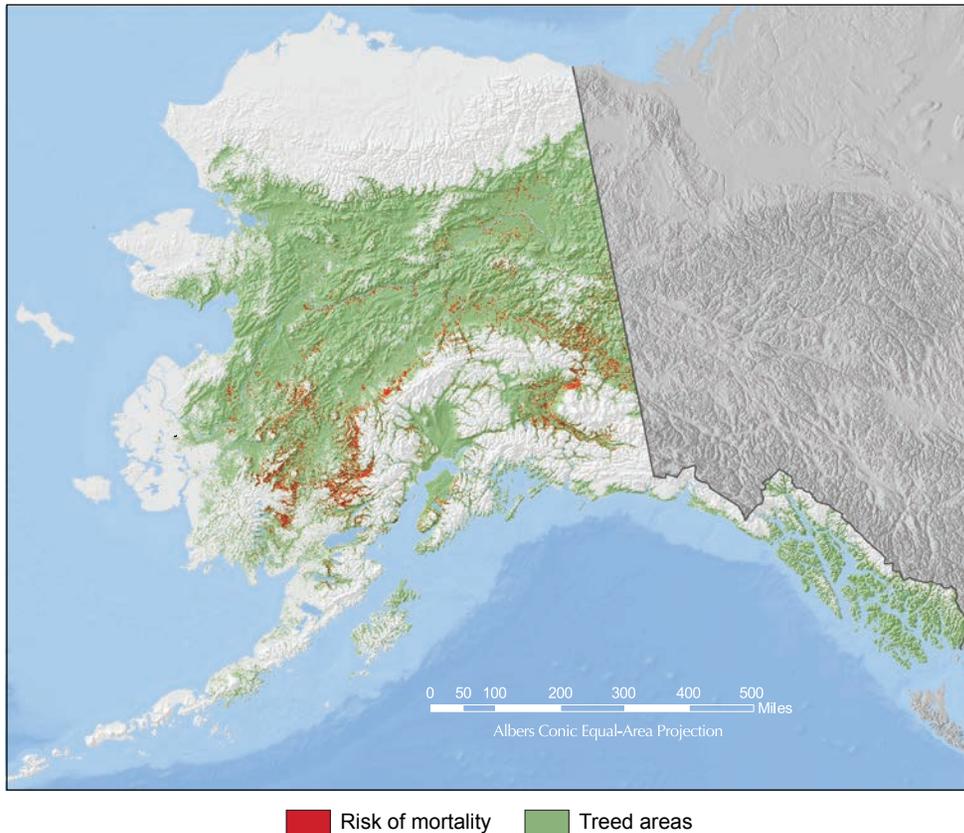


Figure 6.2—The 2013–27 national insect and disease composite risk map for Alaska. Risk, or more appropriately termed, “hazard,” is defined as the expectation that, without remediation, at least 25 percent of the standing live basal area >1 inch in diameter will die over a 15-year time frame (2013–27) due to insects and diseases. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection)

PURPOSE

NIDRM’s primary purpose is as a strategic, broad-scale planning tool that can be used for administrative activities and work planning. In certain landscapes and at appropriate scales, NIDRM maps may be helpful for on-the-ground tactical management. NIDRM was included in section 8204 of the 2014 Farm Bill as an evaluation criterion for States requesting designation of landscape-scale areas to address insect or disease threats on one or more of their National forests.

DATA SOURCES AND PROCESSING

Previous NIDRM assessments (Krist and others 2007, Lewis 2002) defined forests as lands containing at least 10-percent tree canopy cover, including land that formerly had such tree cover and will be naturally or artificially regenerated. By this definition, there are approximately 749 million acres of forested land in the conterminous United States and Alaska. For the 2012 hazard assessment, we extended these limits and modeled 1.2 billion acres of *treed* land—areas of measurable tree presence—across the United States (whether or not these treed lands met some standard definition of forested) (Krist and others 2014). This approach improves coverage for rural areas of the Great Plains and urban areas nationally. The 2012 hazard assessment estimates that 81 million of these acres are in a hazardous condition for insects and diseases. Almost 72 million acres susceptible to hazard are in the conterminous United States,

and 9.5 million acres are in Alaska. In Hawaii, not previously assessed, just under a half-million acres are estimated to be in a hazardous condition. These estimates do not include hazard due to projected climate changes, although this NIDRM report includes an examination of future climate impacts on insect and disease hazards.

With significant improvements in coverage, accuracy, and precision of the data, the 2012 NIDRM was better able to model risk in the Great Plains, urban areas, National forests, and National parks. These improvements also allowed us to model pests, such as emerald ash borer (*Agrilus planipennis*) and laurel wilt, which infest rare and/or widely distributed host species.

MAJOR HAZARDS

Collectively, root diseases, bark beetles, and oak decline were the leading contributors to the risk of mortality in the conterminous United States, while spruce beetle (*Dendroctonus rufipennis*) was the most significant contributor in Alaska. The confluence of bark beetles and root diseases has resulted in large contiguous areas at risk across much of the Western United States. Emerald ash borer was the most significant exotic forest pest. Tree species with the potential to lose more than 50 percent of their host volume include redbay (*Persea borbonia*) and whitebark pine (*Pinus albicaulis*).

While future climate change is not modeled within NIDRM, we expect that the climate changes projected over the next 15 years will significantly increase the number of acres at

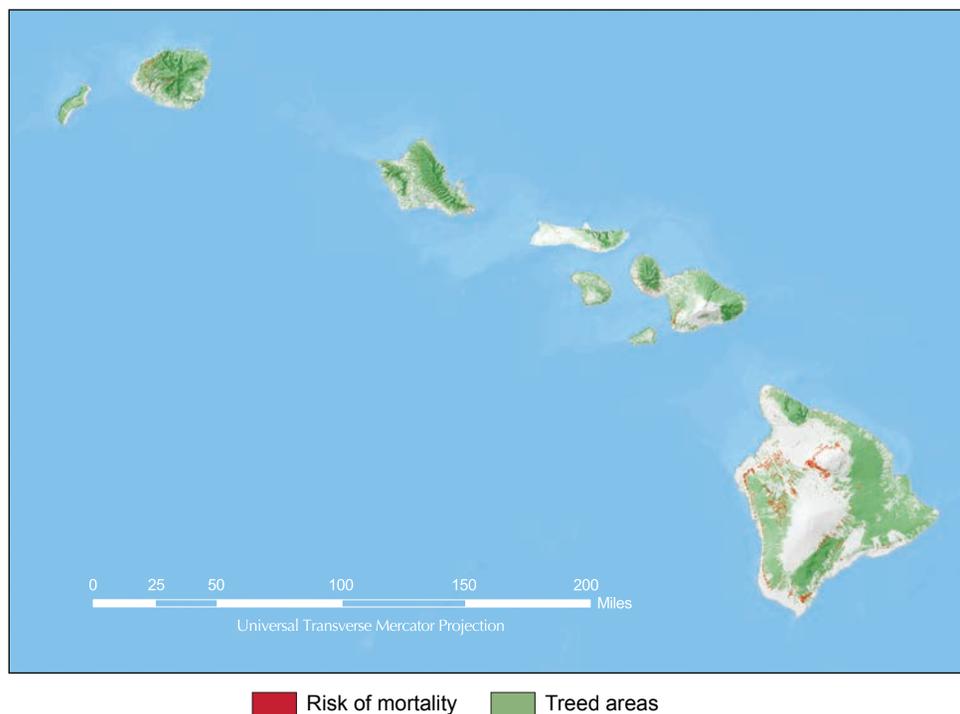


Figure 6.3—The 2013–27 national insect and disease composite risk map for the main islands of Hawaii. Risk, or more appropriately termed, “hazard,” is defined as the expectation that, without remediation, at least 25 percent of the standing live basal area >1 inch in diameter will die over a 15-year time frame (2013–27) due to insects and diseases. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection)

risk, and will include elevated risk from already highly destructive pests such as mountain pine beetle (*Dendroctonus ponderosae*) and engraver beetles (*Ips* spp.). Host trees such as whitebark pine would be at increased risk in future climate change scenarios.

DATA ACCESS

NIDRM's outputs of tabular and geospatial data can be used to:

- Identify the potential impacts of pests and pathogens to forest ecosystems throughout the United States for the 2013–27 timeframe.
- Generate forest pest and pathogen risk maps at a scale useful for resource planning and management in many of our National forests, National parks, and other local units.
- Inform ecosystem assessments and focus forest pest-management resources across geographic regions and individual pest distributions; in other words, prioritize investment for areas where both hazard is great and effective treatment can be efficiently implemented.
- Detect areas where hazardous fuel conditions coincide with forests at high risk for insect and disease activity; this analysis can maximize fire and forest health budgets because, in many cases, the same silvicultural treatments are effective for reducing both fire and pest hazards.

Access to NIDRM data is provided in three ways from the 2012 NIDRM Data Download

Web site (<http://www.fs.fed.us/foresthealth/technology/nidrm2012.shtml>):

- (1) *Tabular and geospatial data download*—Downloadable information includes tabular data that summarize results from the 2013–27 National Insect and Disease Forest Risk Assessment by National forest, National park, and sixth-level (12-digit) Hydrologic Unit Code watersheds units. Geographic Information System (GIS) data available for download include ArcINFO® GRID format raster layers depicting estimated losses from individual forest pests or pathogens, and impacts by tree species.
- (2) *Direct connect to geospatial data within ArcGIS®*—Map and image Web services can be easily added to ArcGIS® and incorporated into maps. Map services provide ready-to-use layers with fixed symbology, while image services provide a wider range of flexibility such as data download, access to metadata, customizable symbology, and inputs to ArcGIS® raster analysis tools. All the GIS layers available for download are offered as map or image services. The 2012 NIDRM Data Download Web site explains how to access and use these services.
- (3) *Viewing data through one of two Web applications*:
 - (a) 2013–27 National Insect and Disease Risk Map Viewer allows users to easily navigate the library of map services across all treed lands in the United States (<http://foresthealth.fs.usda.gov/nidrm/>).

- (b) The Forest Health Advisory System combines pest images, tables, and an embedded map viewer to summarize the 2012 NIDRM by National forest and National park units (<http://foresthealth.fs.usda.gov/fhas>). The site also provides contact information for local forest health experts and publications where managers can obtain additional details on the biology and management for the insect and disease hazards identified on their unit.

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INTRODUCTION

Forest loss and fragmentation of the remainder threaten the sustainability of many ecological attributes and processes that depend on extensive forest cover. The direct loss of intact forest is an obvious threat; less obvious are the indirect threats posed by isolation and edge effects, which encompass a wide range of biotic and abiotic influences on remnant forest (Forman and Alexander 1998, Harper and others 2005, Laurance 2008, Murcia 1995, Ries and others 2004). Because fragmentation is a spatial process, monitoring the threats posed by forest fragmentation necessarily involves analysis of forest maps. The forest maps from the National Land Cover Database (NLCD) have proven useful for synoptic analyses because they provide consistent and complete coverage for the Nation. However, those synoptic analyses provide no details about the forest types or ownerships that are being fragmented. That information is important when considering conservation and restoration alternatives such as where to add or remove forest cover and whether the effort should be a public or private concern. A recent analysis of forest fragmentation trends using NLCD maps from 2001 and 2006 documented a decline in relatively intact forest in the conterminous United States (Riitters and Wickham 2012). The objective of this report is to reevaluate that decline in relation to forest types and ownerships by incorporating *in situ* data from the Forest Inventory and Analysis (FIA) databases.

METHODS

Briefly, we combined forest fragmentation data derived from the 2001 and 2006 NLCD land cover maps with field plot information from the FIA Program. We used the same general methods as have been applied in previous analyses (Riitters and others 2012). Here we extended those earlier analyses of forest-type fragmentation in the Eastern United States in 2001 by (a) evaluating trends from 2001 to 2006, (b) extending the geographic coverage to the conterminous United States, and (c) including forest ownership in addition to forest types.

Forest Inventory and Analysis Data

Bechtold and Patterson (2005) provide a detailed description of the FIA inventory, which may be summarized as follows. The FIA inventory uses a permanent, national, grid-based, equal-probability sample design across all land. Each sample location is determined to be either a forest land use (“forest land”) or a nonforest land use. For those locations determined to be a forest land use, a field inventory plot is installed to collect additional information. A variety of site and vegetation measurements are taken on a cluster of four fixed-area subplots spanning approximately 0.4 ha, which may extend into more than one forest type and/or ownership class. FIA uses a poststratified estimator that accounts for different sampling intensities that arise because of intentional increases in sample size or unintentionally because of survey nonresponse. In effect, each plot has a weight

CHAPTER 7.

Detailed Assessment of the Decline of Core Forest in the Conterminous United States

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factor that accounts for those differences. In addition, each within-plot forest type and/or ownership is weighted by its relative area on the field plot. We refer to the within-plot differences as “partial plot” observations. The area estimates that we report were derived by combining the two weight factors (Bechtold and Patterson 2005).

We used forest land data from 140,584 plot or partial plot observations across the conterminous United States. The plots came from the same sample that was used in the most recent (as of 2012) FIA State report for each State. Unlike the NLCD data, for which observations were available for the years 2001 and 2006, only one date of observation was available during that time interval for each of the FIA plot locations. When combining FIA and NLCD data, it was assumed that the FIA observations represented both of the NLCD observations. Forest types and ownership classes were defined by FIA protocols (USDA Forest Service 2010). The original set of FIA plots represented approximately 275 million ha of forest land and 151 forest types. That sample was screened to eliminate observations of the nonstocked and unassigned types (19.6 million ha, 2 types); exotic types (0.8 million ha, 8 types); types with <50 000 ha each (0.3 million ha, 17 types); and woodland hardwood types (13.5 million ha, 6 types). The data carried into the analysis considered 118 forest types representing 240.5 million ha of forest land. The FIA ownership classes were condensed into four classes, called Federal, State and local government, corporate private, and noncorporate private.

National Land Cover Database Data

Fragmentation was measured using the 2001 and 2006 NLCD land cover maps (U.S. Geological Survey 2011a, 2011b). The NLCD maps identify 16 land cover types at a spatial resolution of 0.09 ha/pixel and a minimum mapping unit of 0.45 ha. The 16 NLCD land cover types were combined into 2 generalized land cover types called forest (including the NLCD deciduous forest, evergreen forest, mixed forest, and woody wetlands classes) and nonforest (including all other NLCD classes). Each inventory plot location was classified as “core” or “not core” by evaluating the NLCD forest data within a 4.41-ha (7 pixel by 7 pixel) neighborhood centered on the inventory plot center location (Riitters and others 2012). The “core” locations were those with 100-percent forest land cover in the 4.41-ha neighborhood, and the “not core” locations were those with <100-percent forest cover. That neighborhood size was large enough to reliably assess core status yet small enough to characterize fragmentation in the immediate vicinity of a field plot. Core (or not core) status was assigned to all of the plot (and partial plot) records for a given FIA plot location. In other words, core status was treated as a new plot-level attribute when using the FIA weight factors to summarize core status by forest types and owner classes.

RESULTS AND DISCUSSION

Of the 240.5 million ha of forest land in 2001 that was included in this study, the total core area was 98.4 million ha, or 41 percent of total

forest land area. Between 2001 and 2006, the gross gain of core forest was 1.2 million ha and the gross loss was 5.0 million ha. The net loss of 3.8 million ha, representing 3.9 percent of the core area in 2001, reduced total core area to 94.6 million ha (39 percent of total forest land area in 2006).

Changes in core area were driven primarily by forest cover changes on private lands (table 7.1). Because most of the total core area was privately owned, it is not surprising that most of

the gains and losses of core occurred on private land. However, the changes on private land were not directly proportional to total core area. While 57 percent of the core area in 2001 was privately owned, private land accounted for more than 80 percent of the gross gains and losses and 78 percent of the net loss of core area. Changes in core area were larger on corporate private land than on noncorporate private land even though approximately two-thirds of private core area in 2001 was noncorporate private land.

Table 7.1—Core area and change from 2001 to 2006 by ownership class in the conterminous United States

(A) Core area and change^a

Owner	2001	Gross loss	Gross gain	Net loss	2006
<i>million hectares</i>					
State & local government	10.99	0.31	0.05	0.26	10.73
Corporate private	19.36	2.33	0.63	1.70	17.65
Noncorporate private	36.68	1.71	0.43	1.28	35.39
Federal	31.35	0.61	0.05	0.56	30.79
Total area	98.38	4.96	1.15	3.81	94.57

(B) Percent of total core area or total change area^b

Owner	2001	Gross loss	Gross gain	Net loss	2006
<i>percent of total area from table 7.1A</i>					
State & local government	11.2	6.3	4.5	6.9	11.3
Corporate private	19.7	47.0	54.3	44.7	18.7
Noncorporate private	37.3	34.5	37.2	33.7	37.4
Federal	31.9	12.2	3.9	14.7	32.6

^a Columns may not sum to total area due to rounding.

^b Due to rounding, columns may not sum to 100 percent, and individual percentages may be different than those calculated from raw hectare totals.

Public lands accounted for 43 percent of the core area in 2001, with Federal land contributing approximately three times more core area than State and local government land (table 7.1). Approximately 22 percent of the total net loss of core area occurred on public lands. Overall, the differences between the changes of core area on public and private lands resulted in a 1-percent increase in the overall share of total core area being located on public lands by 2006.

The total area of individual forest types ranged from 0.1 to 20.4 million ha, with average and median areas of 2.0 and 0.8 million ha,

respectively. In 2001, the percentage of total forest type area that was core area ranged from 2 percent to 79 percent, with average and median area percentages of 38 percent and 36 percent, respectively. Just a few forest types dominated both total forest area and core area. Half (50.5 percent) of total forest area consisted of the 11 forest types with more than 5 million ha each. Those 11 forest types, along with 8 additional forest types with more than 1.5 million ha each of core area in 2001, accounted for 65 percent of total core area in 2001 (fig. 7.1). This report focuses on that group of 19 “dominant” forest

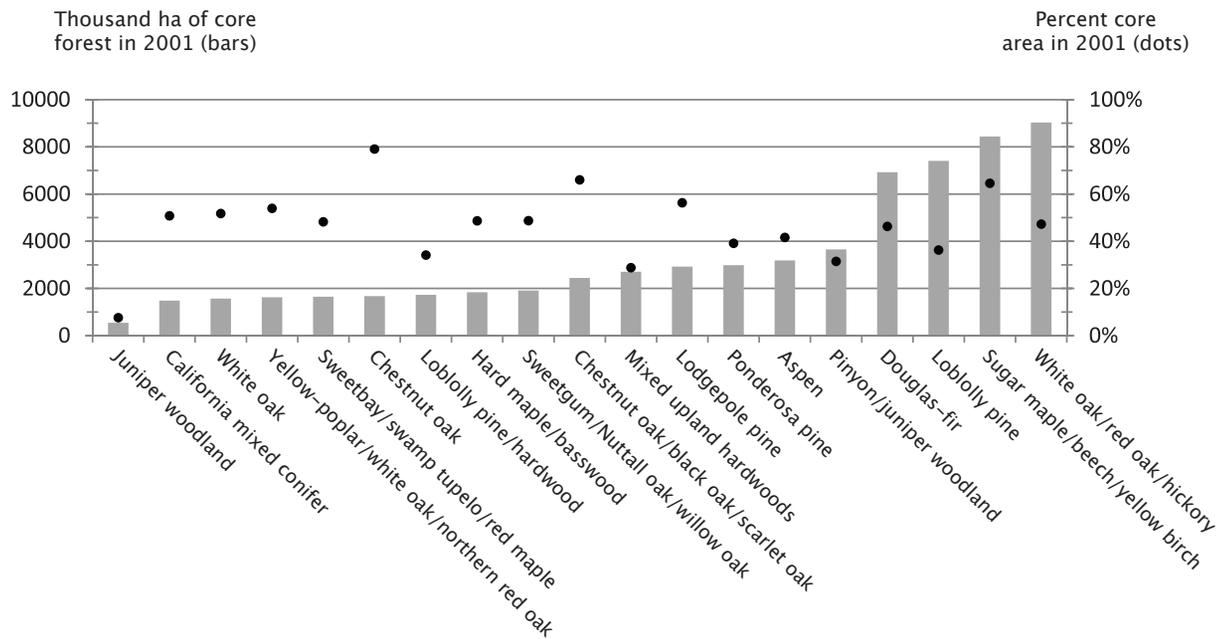


Figure 7.1—Core area and percentage of total area that was core in 2001 for the 19 forest types in the “dominant” group of forest types.

types because their dynamics dominated changes in total core area.

Most forest types exhibited a net loss of core area. Only 5 forest types had net gains (totaling only 5000 ha), and 19 forest types had no change in core area. Among the 94 forest types with a net loss, the percentage of core forest that was lost ranged from near 0 to 31 percent, and 35 forest types (including 5 in the “dominant” group) exhibited net losses larger than 5 percent. The net losses for the 19 forest types in the “dominant” group accounted for 66.7 percent of the total net loss of core area between 2001 and 2006 (fig. 7.2).

By 2006, the range of core area percentage among forest types was between 2 percent and 78 percent, and the 19 forest types in the “dominant” group still accounted for 65 percent of total core area. However, for most forest types, the percentage of total area that was core was lower in 2006 compared to 2001. As a result, the average and median percentages of core area both decreased by 2 percent, and the core area of many individual forest types decreased accordingly. The core areas in 2006 for the “dominant” group of forest types are shown for comparison in figure 7.3.

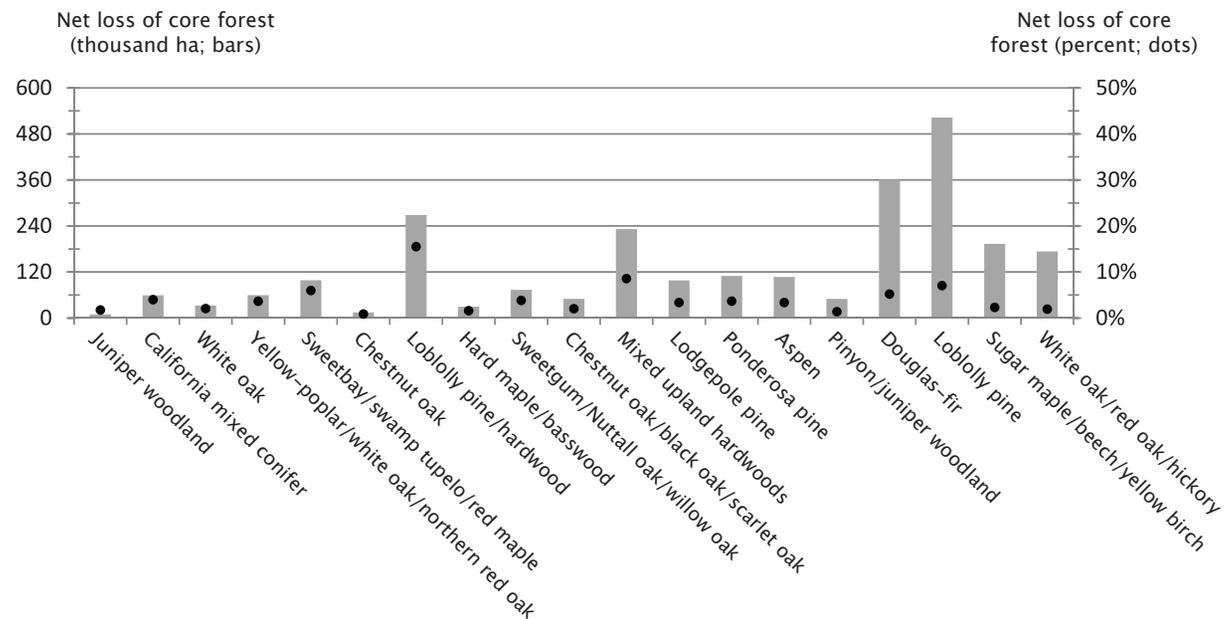


Figure 7.2—Net loss of core area and percentage of core area that was lost from 2001 to 2006 for the 19 forest types in the “dominant” group of forest types.

One approach to prioritizing conservation or restoration efforts is to identify the forest types and/or ownerships that have experienced large net losses (table 7.2) or large percentage reductions (table 7.3) of core area from 2001 to 2006. For example, public efforts could be directed at the giant chinkapin forest type because 71 percent of the remaining core area is on public land, while private efforts could be directed at the loblolly pine or gray birch forest types because 89 and 90 percent, respectively, of the remaining core area is on private lands (table 7.3). There may be particular concerns for the 5 forest types (slash pine, sweetgum/yellow-poplar, post oak/blackjack oak, western hemlock, and cherry/white ash/yellow-poplar) that were not

in the “dominant” group of forest types but were among the top 20 in terms of net loss of core area (table 7.2). Similarly, there may also be particular concerns for the three forest types (loblolly pine/hardwood, mixed upland hardwoods, and loblolly pine) which are in the “dominant” group and experienced reductions of core area ranging from 7 percent to 16 percent (table 7.3).

Previous national studies of forest fragmentation had much lower thematic resolution than this study because they were based only on land cover data derived from synoptic mapping from Landsat satellites. By combining high thematic resolution data on forest communities obtained from field observations with satellite-based land cover

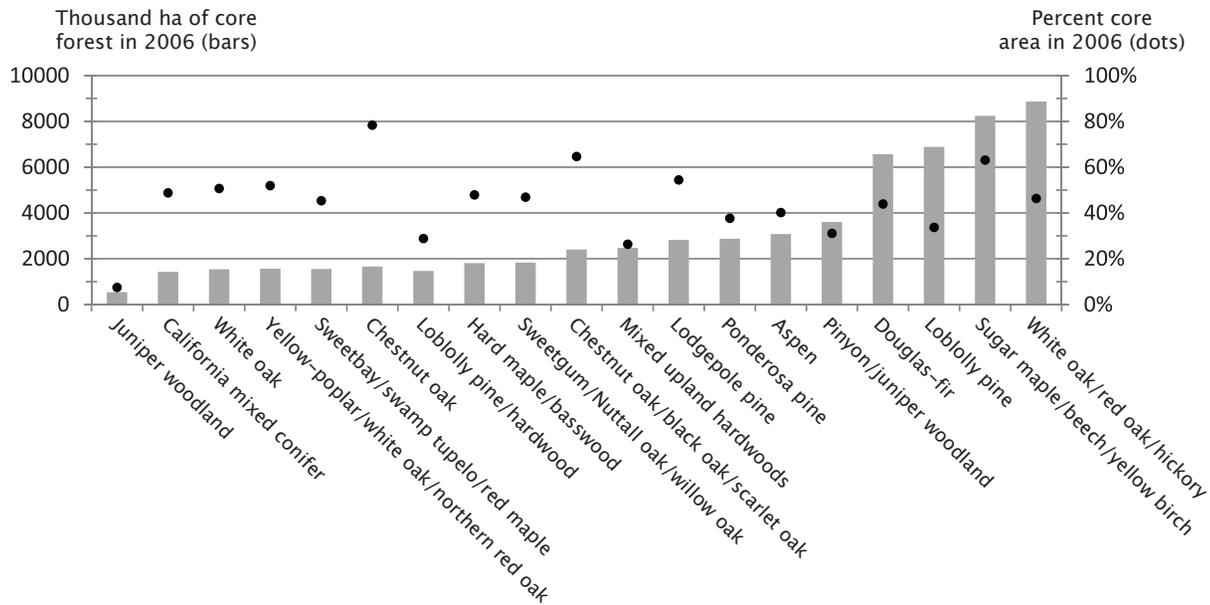


Figure 7.3—Core area and percentage of total area that was core in 2006 for the 19 forest types in the “dominant” group of forest types.

Table 7.2—The 20 forest types in the conterminous United States with the largest net losses of core area from 2001 to 2006

Forest type ^a	Net loss of core area	Share of 2006 core area in public ownership
	<i>thousand ha</i>	<i>percent</i>
Loblolly pine*	522	11.1
Douglas-fir*	361	75.2
Loblolly pine/hardwood*	269	12.0
Mixed upland hardwoods*	232	15.4
Sugar maple/beech/yellow birch*	193	30.0
White oak/red oak/hickory*	174	22.4
Ponderosa pine*	110	72.0
Aspen*	107	61.7
Slash pine	102	28.8
Sweetgum/yellow-poplar	100	13.0
Sweetbay/swamp tupelo/red maple*	99	27.6
Lodgepole pine*	98	95.0
Sweetgum/Nuttall oak/willow oak*	73	19.2
Post oak/blackjack oak	63	10.7
Yellow-poplar/white oak/northern red oak*	60	23.8
California mixed conifer*	59	76.1
Chestnut oak/black oak/scarlet oak*	50	37.8
Pinyon/juniper woodland*	50	86.5
Western hemlock	46	77.9
Cherry/white ash/yellow-poplar	43	15.9

^a Asterisks indicate forest types included in the “dominant” group of forest types (see text for explanation).

Table 7.3—The 20 forest types in the conterminous United States with the largest percentage of core area loss from 2001 to 2006

Forest type ^a	Net loss of core area	Share of 2006 core area in public ownership
	<i>percent</i>	
Sitka spruce	31.3	36.7
Giant chinkapin	24.9	71.1
Loblolly pine/hardwood*	15.5	12.0
Oregon white oak	14.4	26.8
Longleaf pine/oak	14.3	50.3
Southern scrub oak	13.6	35.0
Sweetgum/yellow-poplar	11.9	13.0
Cottonwood	11.3	40.3
Bigleaf maple	10.7	42.3
Longleaf pine	9.7	56.5
Baldcypress/pondcypress	9.6	55.3
Sassafras/persimmon	8.8	19.1
Western larch	8.7	83.5
Mixed upland hardwoods*	8.6	15.4
Gray birch	8.4	10.2
Willow	8.4	46.7
Tanoak	8.3	38.1
Slash pine	7.6	28.8
Red alder	7.4	39.3
Loblolly pine*	7.1	11.1

^a Asterisks indicate forest types included in the “dominant” group of forest types (see text for explanation).

data, we were able to substantially increase the thematic resolution of forest to determine which forest types and ownerships were experiencing more or less fragmentation. The lack of core forest in a relatively small (4.41-ha) neighborhood is a sensitive indicator of local fragmentation, and forest types or ownerships that are not fully forested over such small extents are also (by definition) not intact over larger extents. We expect that all estimates of percentage intact forest would be dramatically lower if larger neighborhood sizes (e.g., >10 ha) were tested (Riitters and others 2002).

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Each year the Forest Health Monitoring (FHM) Program 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 (DM) and other means” (FHM 2009). In addition, EM projects can produce information about forest health improvements. EM projects are submitted, reviewed, and selected in two main divisions: base EM projects and 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 FHM national report contains summaries of recently completed EM projects. Each summary provides an overview of the project and its 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 FHM and include enough information for readers to pursue specific interests. Ten project summaries are included in this report.

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SECTION 3. Evaluation Monitoring Project Summaries

INTRODUCTION

Beech bark disease (BBD) has long been negatively impacting the American beech (*Fagus grandifolia*), an important component of hardwood and mixed hardwood forests throughout eastern North America that provides food and habitat for over 40 species of birds and mammals (McCullough and others 2001). BBD is initiated by feeding activities of the beech scale insect (*Cryptococcus fagisuga*), which create wounds that act as entry points for the *Neonectria* spp. fungi. It is the fungal component of the disease complex that weakens and kills the tree. Mortality levels in the first wave of the disease can be as high as 50 percent (Miller-Weeks 1983). Surviving beech trees are often severely deformed, and their tendency to produce root sprouts can result in the formation of “thickets” that prevent regeneration of resistant beech or other species. The deformed trees offer no economic value and severely reduced ecological value as the disease continues to kill susceptible beech over time (Morin and others 2007). Fortunately, there are American beech trees that remain healthy despite intense BBD pressure. Studies have shown that when eggs are directly affixed to the bark of such trees, scale insects fail to establish, indicating that these trees are resistant to the scale insect (Houston 1983, Koch and others 2010). In the absence of feeding by the beech scale insect, there is little opportunity for *Neonectria* to invade, minimizing impact of the fungus. Large-scale mortality levels in beech due to *Neonectria* have never been reported in the absence of the insect, so resistance to the beech

scale insect equates to resistance to beech bark disease.

Genetic studies have confirmed that resistance to the scale insect can be successfully selected and bred for because it is a heritable trait (Koch and others 2010). In a single generation, the proportion of resistant progeny can be increased from the 1 to 5 percent estimated to occur in natural stands to 50 percent by using two resistant parents. Genetic improvement of stands can be accomplished either through traditional tree improvement (seedling development and planting), through silvicultural methods designed to manipulate stand genetics by favoring resistant trees (remove susceptible beech), or a combination of both (Koch and others 2010). Both State and National Forest managers have been including beech bark disease-related silvicultural treatments as well as plans for restoration/regeneration of beech as part of their resource management plans. However, there is a lack of genetically diverse, regionally adapted, disease-resistant planting stock for forest managers to use to carry out such plans. The goal of this study was to survey for healthy beech trees in heavily BBD-infested areas, then test them by applying scale eggs to confirm their resistance. Scion would then be collected for grafting from these validated scale-resistant beech trees for inclusion in ongoing efforts to establish seed orchards. The long-term goal of this work, which builds off previously funded projects, is the completion of four interagency BBD-resistant regional beech seed orchards in the following locations: the Hardwood Tree

CHAPTER 8. Beech Seed Orchard Development: Identification and Propagation of Beech Bark Disease-Resistant American Beech Trees (Project NE-EM-B-11-03)

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Regeneration and Improvement Center in Indiana, the Oconto River Seed Orchard in Wisconsin, the Monongahela National Forest, and the Pennsylvania State Nursery. Each seed orchard will consist of scale-resistant beech trees collected regionally, within an approximate 200-mile radius, from both National forest and State lands.

METHODS

Survey for Candidate Resistant Beech

Surveys for resistant beech were carried out in sites where the beech component was at least 20 percent of the basal area and there was a long history of significant BBD infestation. To be considered a candidate BBD-resistant tree, the following criteria had to be met: diameter at breast height >9 inches; located within 50 feet of an infested beech tree; healthy crown; and no signs of scale infestation (rough, cracked

bark) or fungal infection (cankers, tarry spots, fruiting structures). Surveys were conducted across 73 stands on 19 sites (over 5,300 acres) on the Hiawatha National Forest, Michigan. In West Virginia, four sites were surveyed on the Monongahela National Forest and five sites on State forest land. In Pennsylvania, four sites on State forest land were surveyed (table 8.1).

Field Assay to Confirm Scale Resistance

Workshops were conducted at each site to instruct participating personnel on the methods used to collect scale eggs and set up field assays for scale resistance (fig. 8.1A). A detailed protocol and instructional video has now been published (Koch and Carey 2014). To briefly summarize, between mid-July and early August, a paintbrush was used to gently brush the white, waxy clumps containing adult scale insects and eggs from an infested tree into a sealable collection bag. The mixture was passed through a 250-micron nylon

Table 8.1—Summary of candidate American beech trees identified and results of field assays for beech scale insect resistance

Location	No. of sites	No. of candidate trees tested	Resistant ^a	Susceptible ^b	Inconclusive ^c	Total no. resistant genotypes ^d
MI-Hiawatha National Forest	19	52	19	9	24	19
WV-State lands	5	17	9	7	1	9
WV-Monangahela National Forest	4	11	1	6	4	2
PA-State lands	4	24	9	14	1	22

^a No egg clusters on foam pad or tree, <2 adults.

^b One or more egg clusters on foam and ≥2 adults.

^c Tests were inconclusive if both pads were missing on either the test or control tree.

^d Includes trees tested in previous years.

mesh to separate the eggs from the adults and debris. Approximately 500 eggs were counted out using a dissecting microscope and sprinkled across a piece of moistened polyethylene foam. The foam was then tied to the candidate tree, with the eggs facing the bark, and covered with a synthetic home barrier wrap (Tyvek®). A minimum of two egg-containing pads were placed on each candidate tree, and additional pads were placed on visibly susceptible trees as a control. A year later, the pads were carefully removed, and each tree and pad was inspected with a hand lens or dissecting microscope to determine the number of adult scale insects and egg clusters that were present (fig. 8.1B). A tree is determined to be resistant if no egg clusters and no more than two adult scale insects are found on both the tree and the pad. The control tree pad was also removed and inspected for the presence of adult scale insects and egg clusters to confirm the viability of the eggs and the validity of the test.

Scion Collection and Grafting

Branch sections 1 to 2 m long were collected from candidate trees throughout the months of January and February in 2013 and 2014 using a shotgun, pole pruner, or rope saw, or by climbing and hand pruning. The cut ends of the branches were wrapped with moistened paper towels for shipment. Hot callus grafting was carried out in winter 2013 as described in Carey and others (2013), but due to less-than-optimal grafting success rates (table 8.2) the following changes were implemented in winter of 2014. To minimize fungal contamination of

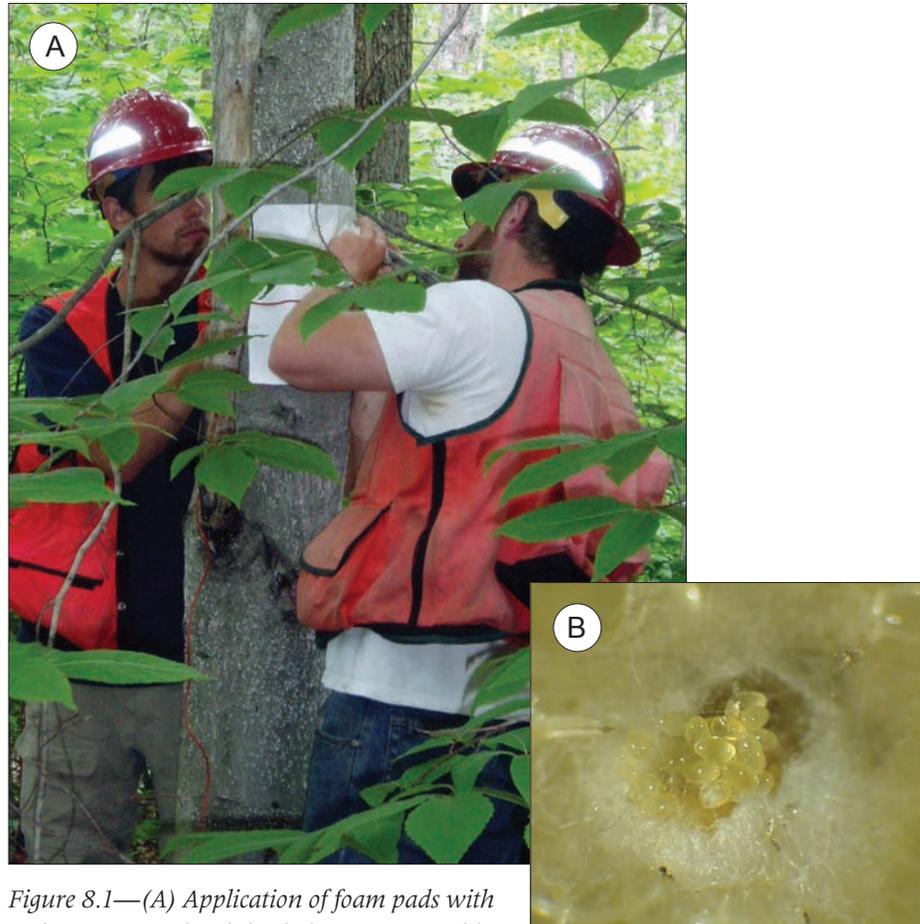


Figure 8.1—(A) Application of foam pads with scale eggs onto a beech bark disease-susceptible beech tree as a control for field assays of nearby candidate beech trees. (Photo by Paul Berrang, U.S. Department of Agriculture Forest Service)

(B) Adult beech scale insect and egg cluster established in foam test pad viewed under 10X dissecting microscope. (Photo by David Carey, U.S. Department of Agriculture Forest Service)

rootstock, beechnuts were sown in a media mix containing 40 L Fafard[®] aged pine bark, 40 L BFG M2 Professional Mix, 225 g horticultural grade perlite, 94 g Osmocote[®] Plus (15–9–12), 25 g Micromax[®], 75 g gypsum, and 255 g Actino-Iron[®] Biological Fungicide mixed together and moistened with a solution of Subdue[®] Maxx[®] (1.25 ml/L water). Germinants were treated throughout the growing season with foliar applications of fungicides, alternating Subdue[®] Maxx[®] (.04 ml/L) with Alude[™] (3.0 ml/L). A soil drench application of Subdue[®] Maxx[®] (.66 ml/L) was given in October prior to putting the germinants into winter storage. Meanwhile, to reduce sources of contamination from the scion, the branches cut in January and February were surface sterilized by spraying with a solution of ZeroTol[®] (4 ml/L) upon

receipt. The proximal ends of the branches were given fresh cuts and placed into buckets of water for storage at 4 to 6 °C. Water was changed, buckets were cleaned, and fresh cuts were made weekly until grafting was completed. Scions were cut from the branches. Prior to making the final cut for veneer grafting, the scion pieces were dipped in molten paraffin wax (50 °C) to push out excess sap, which would otherwise flood the graft union and promote contamination. The sap was blotted away prior to placing the scion on the rootstock. Once banded, the scion was again dipped in paraffin as previously described (Carey and others 2013).

RESULTS

The number of candidate trees identified and field tested at each location is listed in table 8.1,

Table 8.2—Summary of American beech grafting attempts and success rates in 2013 and 2014

Location	Total number of genotypes grafted	Number of grafts attempted, 2013	2013 percent success	Number of grafts attempted, 2014	2014 percent success ^a
MI—Hiawatha National Forest ^b	8	0	—	240	NA ^e
WV—State lands ^c	5	216	21	0	—
PA—Allegheny National Forest ^d	8	19	26	70	84
PA—State lands ^d	10	28	82	245	80
Total	31	263	28	555	81

— = Not applicable (zero grafts attempted).

^aThis represents a preliminary estimate, final success rates will be determined based on 1-year survival rates.

^bGrafting done at the Oconto River Seed Orchard, White River, WI.

^cGrafting done at the Northern Research Station Forestry Laboratory, Delaware, OH.

^dA portion of grafting done at each facility.

^eSuccess rates not yet available.

along with the number of trees confirmed to be scale resistant. In some sites, a large number of the field tests were inconclusive due to the loss of both test pads on either the test tree or the control tree. Often this appeared to be the result of bear activity. Of the 74 candidate trees that were successfully tested, just over half (38) were confirmed to be resistant. Grafting results are listed in table 8.2. In 2013, the overall success rate on the 263 total grafts attempted was only 28 percent based on 1-year survival rates, well below the average success rate of 52 percent previously reported (Carey and others 2013). The preliminary estimate of the overall success rate in 2014, based on the 315 total grafts attempted for which grafting results were available, is 81 percent. This indicates that the changes in 2014 to the potting media and grafting protocol, along with the addition of regular treatments with fungicides, contributed to improved graft success.

DISCUSSION AND CONCLUSIONS

Surveying stands with a beech component of at least 20 percent in areas long infested with beech bark disease was an effective approach for identifying candidate trees with resistance to the beech scale insect. The proportion of the candidate trees that was confirmed to be resistant through field testing varied across sites. It is possible that in areas that are considered more of a “killing front,” characterized by high beech scale populations, that candidate trees can be selected with better efficiency compared to sites that are “aftermath forest,” where the beech scale populations are much lower. Given that about 50 percent of the candidate trees tested

demonstrated some level of susceptibility (at least one egg cluster and one adult), field testing prior to investing resources into scion collection and grafting is a more cost-effective approach to identifying scale-resistant parent trees for inclusion in seed orchards.

To capture a significant portion (>90 percent) of the genetic variation in each beech population, a minimum of 20 to 25 unrelated scale-resistant trees (8 to 10 grafted ramets of each) are needed for each regional seed orchard (Johnson and Lipow 2002). With the addition of the scale-resistant trees identified as part of this project, there are now 17 genotypes from State lands in the Upper Peninsula of Michigan and 8 from the Hiawatha National Forest that will be installed in a seed orchard at the Hardwood Tree Regeneration and Improvement Center in Lafayette, IN, in the spring of 2016. A separate seed orchard is also slated for installation at the Oconto River Seed Orchard that will include the 19 resistant trees identified at the Hiawatha National Forest, with a target of identifying 6 additional genotypes, possibly from re-assaying the trees that were inconclusive due to the loss of the test pads. Installation of the seed orchard at the Pennsylvania State Nursery was initiated in 2012 with the planting of 20 grafted ramets that represented 9 resistant genotypes. With the addition of the 9 new resistant trees identified in Pennsylvania, there are now 22 scale-resistant trees that will be added to this orchard. Efforts to identify three additional resistant trees will continue. There are currently 11 resistant trees identified in West Virginia, only 2 of which are from the Monongahela National Forest. However,

the Monongahela National Forest recently conducted additional surveys and located over 100 additional candidate trees that were slated to undergo field testing in the summer of 2014. Once fully established, these regional seed orchards will provide a source of beechnuts enriched for resistance to beech bark disease that can be used by State and Federal forest managers for restoration of healthy American beech for decades to come.

ACKNOWLEDGMENTS

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CONTACT INFORMATION

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INTRODUCTION

In preparing the 5-year Forest Inventory and Analysis (FIA) report for Vermont in 2007–08, analyses showed an increase in tree mortality since the previous inventory of 1996–97. Adjacent States had a similar spike in mortality. Further evaluation of FIA plot data helped focus the investigation. Some general characteristics of the mortality included: (1) distribution in both northern and southern Vermont, (2) high-elevation forests and timberland forests were affected, and (3) mortality was not related to stocking levels. Several species seemed to be affected, including red spruce (*Picea rubens*), paper birch (*Betula papyrifera*), balsam fir (*Abies balsamea*), red maple (*Acer rubrum*), and American beech (*Fagus grandifolia*). Local concern about white ash (*Fraxinus americana*) decline led to its inclusion in this investigation even though FIA data did not show significant mortality.

Our major goals were to:

- Investigate potential causes of tree mortality detected in recent FIA data for Vermont.
- Identify site conditions that contributed to tree mortality in order to inform forest management strategies to maintain future forest health.

METHODS, RESULTS, AND DISCUSSION

The investigation results are grouped into three sections:

- (1) Existing forest health data on forest disturbance factors;
- (2) Spatial analyses of FIA data and potential inciting and contributing factors; and
- (3) Field assessments of sites with high mortality.

Summary of Forest Disturbance Factors

Existing data from Vermont forest health monitoring programs were used to identify stress events that had occurred since the last FIA inventory. Data sources included those from the national Insect and Disease Survey (IDS), tree health plots visited annually since 1988 as part of the North American Maple Project (30 sites), Vermont Monitoring Cooperative forest health plots (19 sites), and pest survey plots. Analysis of the most likely stress agents involved focused on:

- ice storm in 1998
- droughts in 1999 and 2001–02
- birch decline agents
- spruce winter injury

CHAPTER 9.

Investigating Causes of Mortality in Vermont

(Project NE-EM-B-10-01)

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- forest tent caterpillar (*Malacosoma disstria*) defoliation
- beech bark disease
- stand-replacing windstorms
- balsam woolly adelgid (*Adelges piceae*) damage
- air pollution effects (acid deposition and ozone)

The 1998 ice storm was initially thought to be a major factor in the observed mortality spike because it caused an increase in dead and dying trees for several years following the event. However, the initial effects from the ice storm were resurveyed in 1998, and the 1997 FIA data were adjusted in a subsequent analysis to account for ice storm mortality and tree volume reductions. In addition, follow-up crown health monitoring showed that most species recovered well from the ice storm (Kelley and others 2002).

However, lingering effects from the ice storm were later shown to be a significant contributor to birch decline. Studies of white birch (*Betula papyrifera*) declines identified the 1998 ice storm as the inciting factor, with tree recovery influenced by drought and soil nutrition (Halman and others 2011).

Significant winter injury on red spruce was reported across the Northeast in 2003. This event was extensively studied, and published articles indicated this event was responsible for red spruce mortality (Schaberg and others 2011). Due to results from these studies on birch and red spruce declines, these species were not included in our investigation.

Spatial Analysis

Aerial survey maps of forest disturbances were intersected with FIA plots to compare known locations of disturbance history and incidence of mortality on ground plots. Aerial survey maps showed that approximately 45 percent of Vermont's forest land received damage between 1997 and 2005. Data from FIA plots within mapped disturbance polygons indicated that tree growth was slower where disturbances occurred than on undamaged plots for all forest type groups, but no statistically significant differences were detected. Similarly, the spruce/fir, oak/hickory, and aspen/birch forest type groups showed more standing dead trees per acre where disturbances occurred than on the undamaged plots, but no statistically significant differences were found. Additional spatial data layers were used to assess site conditions in relation to aerial survey maps. Soil moisture and water availability were site factors of importance, and several datasets were used to evaluate these.

Soil moisture and site characteristics—

A soil drainage index (DI) (Schaetzl 1986), also referred to as the soil dryness index, is designed to represent the amount of water that a soil potentially contains and makes available to plants under normal climatic conditions. This index was used to characterize aerial survey damage polygons and FIA plots. A high DI corresponds to sites with high water-holding capacity. A low DI corresponds to sites with low water-holding capacity. The main factors that influence the DI are the depth to the water table and the soil volume available for rooting. In

addition, a depth-to-bedrock map was created for Vermont that included finer scale data than the DI map. We used shallow-to-bedrock polygons from this map as another means of addressing potential drought susceptibility of FIA sites.

The DI and shallow-to-bedrock maps were compared to aerial survey damage maps and FIA plots. Results indicated that:

- Significantly more acres with damage were mapped on sites shallow to bedrock.
- FIA plots on shallow soils were more likely to have 1 or more years of damage.
- The greatest relationship between sites with shallow soils and standing dead trees per acre was found in the aspen/birch forest type group.
- Greater mortality was reported on sites that were either dry or wet (from DI) when combined with shallow soils.

Long-term (30-year) average growing-season precipitation maps were used with FIA plots to compare average precipitation with forest land in each forest type group. General relationships between precipitation and forest type groups were assessed. Several growth and mortality trends were found in relation to the average growing-season precipitation. The white/red/jack pine forest type group grew more often in areas of relatively low average precipitation, and the maple/beech/birch group grew more often in areas of relatively high average precipitation. The oak/hickory type group had higher net

growth per acre per year on sites with moderate precipitation. The spruce/fir type group was more prevalent in areas with higher average precipitation.

Specific years when precipitation was below normal (1999, 2001, and 2002) were also compared to FIA plots. It should be noted that nearly the entire State was under drought conditions during these years. The only specific mortality relationship that was statistically significant was that the spruce/fir forest type group had the highest number of standing dead trees per acre at sites that experienced drought during the 1999 growing season. No other statistically significant relationships were found between precipitation deficits, dryness index, and tree mortality.

The spatial distribution of standing dead basal area (DBA) was assessed using Global Moran's I spatial autocorrelation statistic, a tool used to evaluate whether the pattern of mortality is clustered, dispersed, or random. DBA showed patterns of spatial autocorrelation for all species, and individually for red maple, balsam fir, and American beech (fig. 9.1). Spatial analysis of DBA of all species showed positive correlations with stand age and elevation. A northern cluster of declining plots was positively correlated with elevation. A southern cluster was positively correlated with stand age. A central cluster was positively correlated with both elevation and stand age. On a species level, high red maple DBA was most prevalent on FIA plots in southern Vermont. High balsam fir DBA was concentrated in two separate clusters, one in the

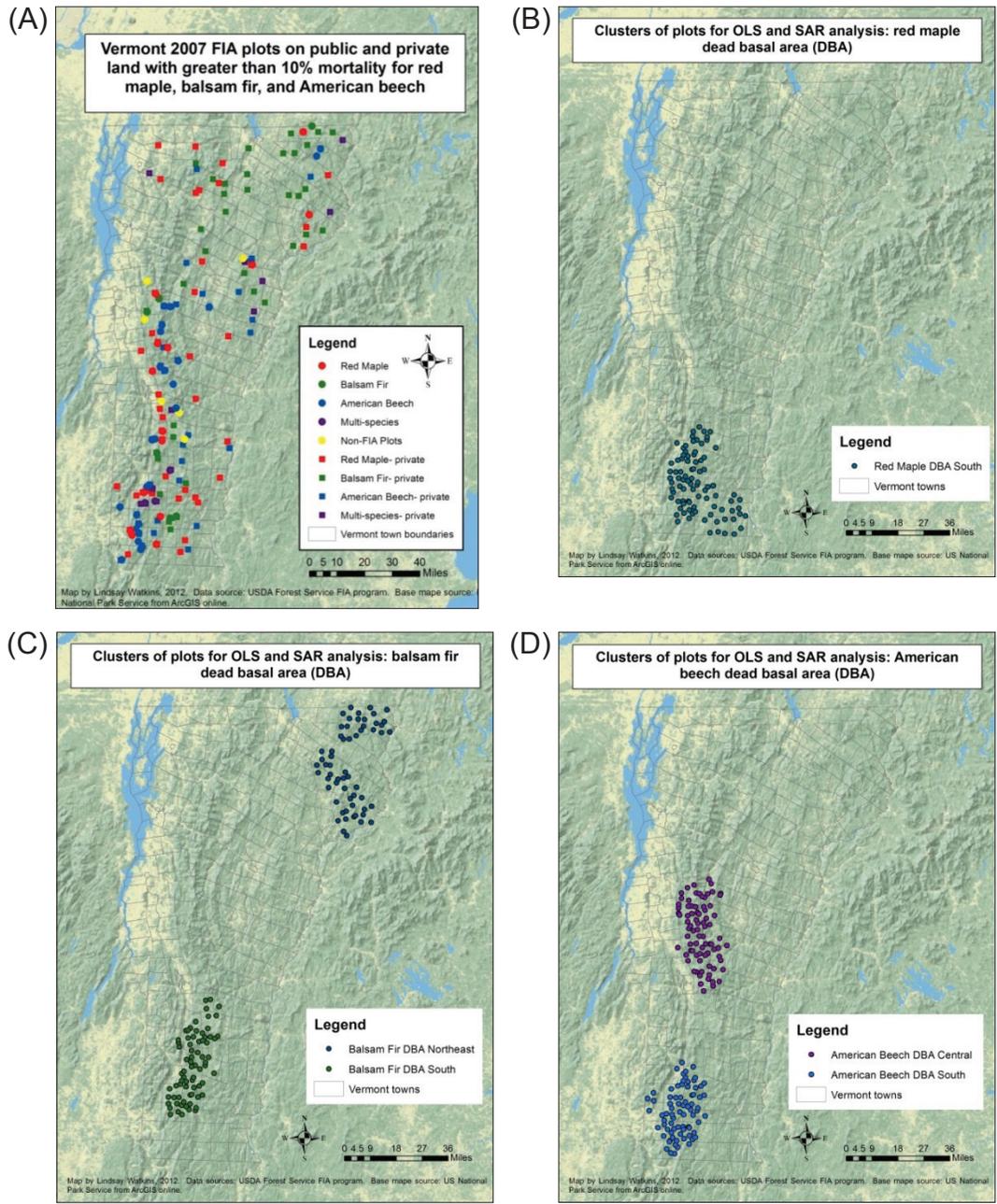


Figure 9.1—(A) High mortality FIA plots in Vermont in 2007 identifying the tree species involved; cluster analysis identifying geographic groupings of dead basal area for (B) red maple, (C) balsam fir, and (D) American beech. Plot locations are approximate.

Green Mountains of southern Vermont and a second cluster in the Northeast Kingdom. High beech DBA was clustered in central Vermont in the Green Mountains, with a separate cluster in the southern Vermont Green Mountains.

Spatial analysis of DBA for each species cluster was compared to site features. For total balsam fir DBA, slope was negatively correlated with dead basal area. In the northeast, elevation was positively correlated with DBA, and 30-year average growing-season precipitation was negatively correlated with DBA. In the south, slope and soil dryness index were both negatively correlated with DBA. For total American beech DBA, stand age and elevation were positively correlated with DBA, and 30-year average growing-season precipitation was negatively correlated with DBA. For the central cluster, stand age and elevation were again positively correlated with DBA, and 30-year average growing-season precipitation was negatively correlated with DBA. In the southern cluster, only stand age was positively correlated with DBA. Stand age, elevation, and precipitation deficit were the factors most associated with DBA for beech plot clusters. No significant relationships were found between red maple DBA and stand or site factors.

Ozone and ozone bioindicator plant trends—During the interval between FIA samplings, ozone concentration dropped, and the severity of ozone symptoms on bioindicator plants was reduced (Smith and others 2012, Vermont Monitoring Cooperative 2009). While it is possible that previously high ozone levels

affected long-term tree health, the absence of ozone symptoms observed in forests over the previous decade does not support any long-term ozone effects.

Acid deposition effects on tree health—Previous work developing critical load and exceedance maps for nitrogen and sulfur have been used successfully to demonstrate relationships between forest decline and areas exceeding critical load (Pardo and others 2010, Schaberg and others 2010, Sullivan and others 2013). Our analysis using FIA plots with varying mortality levels did not find any relationships between mortality and acid deposition indicators.

Field Assessments

Field assessments were used to find on-the-ground evidence for the timing of mortality events, site characteristics contributing to mortality, and any inciting factors still in evidence at high mortality sites. Sites with more than 10-percent mortality were identified through FIA plot data (15 sites on public lands) and using aerial survey records or through referrals from foresters (16 sites). Field assessments focused on sites and tree variables for four species: red maple, balsam fir, American beech, and white ash. Thirty-one mortality sites were visited, but only 15 sites met our criteria for sampling. At sampled sites, data were collected on past disturbances, physiography, soil type and drainage, ground cover species, and regeneration. In addition, 20 trees were measured for diameter, height, crown class, dieback, defoliation, vigor, seed, bole damage, presence of standing water,

and other potential damage agents. Tree cores were collected on a subset of trees. It was our intent to collect cores from live, declining, and dead trees, but this was not possible due to limited numbers of sample trees in each class for each species and a high amount of internal decay, especially on declining trees. As mentioned previously, research on paper birch and red spruce mortality was conducted by others.

Red Maple is susceptible to mechanical injury, defects, diseases, and sapsucker damage (Burns and Honkala 1990b). Fungal diseases often attack the stem through branch stubs and wounds and then advance quickly. Red maple can be found in early-, mid-, and late-successional forests. It can be a pioneer species, though it is more shade tolerant than other pioneers. While it can also be considered a climax species in some forest types (usually on wet sites), in the Northern Hardwood Forest it is usually replaced by more shade-tolerant species. Depending on growing conditions, trees can reach maturity at 70 to 80 years and rarely live past 150 years.

Field results showed that trees were on average more than 80 years of age at seven of the nine declining sites visited. Three of the sites had trees that were 100 to 125 years old. Also present at most sites was evidence of past logging (93 percent of sites) in the form of basal bole wounds. At one site, 70 percent of the trees were either wounded or dead. Red maple is particularly susceptible to wound-induced decay. Although

dead basal area was not significantly related to stand age (see above), the combination of aging trees and wound-induced decay seems likely to have contributed to some, if not most, of the red maple mortality observed.

White ash crown assessments were measured on 45 FIA phase 3 plots. White ash had the second highest incidence of poor crown condition, with 25 percent of the live basal area displaying poor crowns. Poor crowns were defined by dieback >20 percent, crown density <35 percent, or foliage transparency >35 percent (Morin and others 2011). Crown health was especially poor on plots in southern Vermont.

Field assessments included observations of symptoms of ash yellows disease. These symptoms include tufted foliage due to slow twig growth and short internodes, small leaf size, deliquescent branching or loss of apical growth dominance that resulted in lateral branching, presence of witches' brooms at the trunk base, and vertical cracks on the trunk near the ground. Very few of these symptoms were observed at the sites visited. Root collar samples were collected from declining ash trees and analyzed for ash yellows disease. Tests were negative in all cases except one site in southeastern Vermont. Site characteristics in declining ash stands indicated that the sites were prone to water deficits. As a ring-porous species, white ash is susceptible to decline following drought years. This was likely involved as a contributing factor to ash declines.

Balsam fir grows at upper elevations along the spine of the Green Mountains as well as

in lowland spruce/fir forests and several of Vermont's swamp communities (Thompson and Sorenson 2000). Maximum size and age vary with climate, soil, and biotic conditions, but trees can grow to 40 to 60 feet and 12 to 18 inches in diameter and reach a maximum age of 200 years (Burns and Honkala 1990a). Balsam fir has several important insect pests, including spruce budworm (Rose and Lindquist 1994), which defoliates trees and causes extensive root damage, and the balsam woolly adelgid, an introduced species that attacks tree stems, twigs, and buds and can kill trees within 3 years (Quiring and others 2008). Spruce budworm populations have been at low levels in Vermont since the 1980s, but mortality from balsam woolly adelgid was mapped during aerial surveys on nearly 11,000 acres in 2004, when it was confined to central and southern Vermont (Vermont Department of Forests, Parks, and Recreation 1997–2011).

Several fungal pathogens can cause decay in the trunk, branches, and roots of balsam fir. These pathogens can affect trees without any visible external symptoms and may be more severe on drier sites. Unfortunately, tree cores from balsam fir did not remain intact when extracted from trees, so they were unavailable for assessment of tree age, and we were unable to ascertain the timing of mortality. The presence of balsam woolly adelgid in southern Vermont will remain a strong candidate as an inciting factor for mortality. In northern Vermont, precipitation

deficit seems likely to be the inciting factor for mortality.

American beech in Vermont has experienced decades of injury from beech bark disease (BBD), a nonnative pathogen vectored by beech scale insects (Vermont Department of Forests, Parks, and Recreation 1997–2011). Diseased trees tend to decline when water availability is reduced, such as during the recent droughts of 1999 and 2001–02. Field assessments at one location showed that 95 percent of sample trees were infested with BBD or were dead.

CONCLUSIONS

Determining causes for mortality after the fact is a challenge, and thus they are difficult to assign with certainty. Thankfully, our annual detection surveys and periodic ground plot measurements allowed us to get a head start in this investigation.

Our findings point to the following inciting and contributing factors for species and locations involved in this mortality study:

- Balsam fir trees in southern Vermont experienced mortality because of balsam woolly adelgid damage, with drought being a likely contributor to mortality.
- Balsam fir trees in northern Vermont likely experienced water fluctuations during drought years, resulting in decline and mortality.
- Beech trees showed reduced growth and poor crown condition due to beech bark disease

and stand age; elevation and precipitation deficit were contributing factors in increased mortality.

- Red maple mortality was likely related to internal decay from past logging wounds that reduced tree vigor, but did not cause mortality until other compounding factors reached critical levels. Aging trees and multiple years of drought may have been contributing factors.
- White ash decline was likely initiated by water fluctuations at well-drained sites brought on by recent drought events. Ash yellows disease was confirmed at only one location in southeastern Vermont.

The role of soil nutrition in this mortality event could not be fully explored. Initial results from other research on red spruce and paper birch declines revealed a role for calcium in stress recovery (Halman and others 2011, Schaberg and others 2011). Paper birch initially damaged by the 1998 ice storm varied in recovery success depending on site levels of available calcium. Mortality was greater on low-calcium sites. Similar results were reported for red spruce following damage from winter injury. Site levels of calcium were correlated with recovery and subsequent growth.

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INTRODUCTION

Crown dieback and declines in tree health of sugar maple (*Acer saccharum*) have been reported on various land ownerships in the western Upper Great Lakes region in recent years (MDNR 2009, 2010, 2012). In some areas, the crown dieback has affected high-value crop trees. Historically, sugar maple dieback (fig. 10.1) has been reported more frequently in the eastern part of its range and has not been described on the same scale in the Great Lakes region since the 1950s and 1960s (Bal and others 2015, Millers and others 1989). As a result, fewer studies of canopy health of sugar maple exist in the Midwest than in the Eastern United States. Dieback and decline episodes of sugar maple often appear to be driven by local conditions, mainly predisposed or incited by poor soil nutrient status and further exacerbated by severe drought or other weather extremes, local insect or disease damage, or management activities (Horsley and Long 1999, St. Clair and others 2008).

Reports of current sugar maple dieback in the Great Lakes region came recently from both public and private land managers across various locations. The extent to which the recent sugar maple dieback was related to management activities and to biotic or abiotic factors was unclear. Furthermore, the severity and geographic scope of the dieback across the region was unknown. This project established long-term monitoring plots in the Upper Great Lakes region to characterize changes in dieback symptoms in sugar maple and characterize



Figure 10.1—Severe canopy dieback and tree mortality in a sugar maple stand in Upper Michigan. (Photo by Tara Bal, Michigan Technological University)

CHAPTER 10.

Evaluation of Sugar Maple Dieback Trends in the Upper Great Lakes Region (Project NC-EM-B-10-02)

TARA L. BAL

ANDREW J. STORER

relationships between dieback and ownership. These plots were located on a range of soil types and encompassed areas both with and without differing amounts of dieback. Other factors related to the dieback etiology continue to be examined, including climate variables, soil and foliar nutrients, and other biotic and abiotic plot variables.

METHODS

A network of permanent sugar maple health evaluation plots (0.04 ha, fixed radial) was established across Upper Michigan, northern Wisconsin, and eastern Minnesota, with

variable mean dieback levels on public (State and federally owned, 59 plots established in 2010) and private (industry owned, 61 plots established in 2009) lands (fig. 10.2). The private lands have been managed by industry since stand initiation. Plots were not random, but were identified by industrial and agency foresters as having varying degrees of sugar maple dieback (from none to severe). Sugar maple dieback symptoms were evaluated annually from the plot establishment year through 2012. Plots were located on multiple soil types, and stand basal areas varied from 12 to 61 m²/ha with a mean basal area of 30.6 m²/ha. Plots were located at least 40 m away

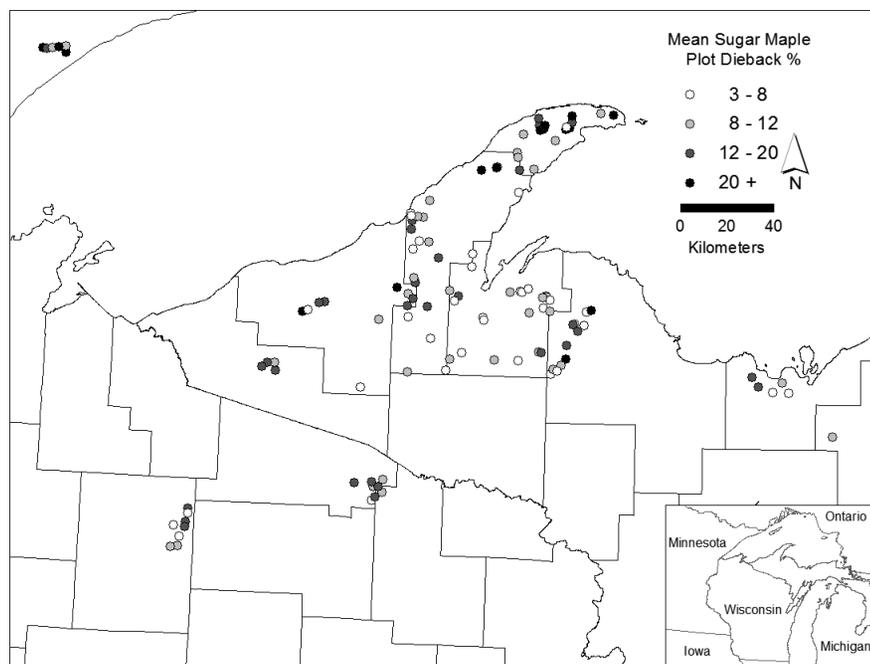


Figure 10.2—Sugar maple health evaluation plot distribution on public and private land across Michigan, northern Wisconsin, and eastern Minnesota (circles), and mean plot sugar maple dieback from year of establishment (2009–10) through 2012, excluding initially 100-percent dead trees (shading of circles).

from established roadways and in most cases included at least 10 sugar maple trees.

All trees with at least a 10-cm diameter at breast height (d.b.h.) (merchantable size) were measured, permanently tagged, and had canopies assessed using Forest Health Monitoring protocols¹ to determine canopy dieback, transparency, density, and other measurable factors. Crown dieback was estimated as percentage of the whole crown that had dieback present (0 to 99 percent, estimated to 1- to 5-percent intervals), including recently dead branches, peeling branches, or fine twigs lacking foliage or live buds in the upper and outer portions of the crown. In order to calculate the mean sugar maple dieback for plots in each year, harvested trees were not included (11 of 120 plots had at least one tree removed during the 4-year period). Saplings that reached the minimum 10 cm d.b.h. were added into the plot measurements after the plot establishment year. Trees that were 100-percent dead during the plot establishment year were not included in the plot mean canopy dieback for that year, but subsequent mortality was included in annual dieback estimations to capture dead and dying trees. Other tree and plot variables were also assessed for use in future studies, including tree bole conditions, foliage and soil nutrients in plots, soil density, canopy density, regeneration and herbaceous density,

¹ U.S. Department of Agriculture Forest Service. 1999. Forest health monitoring 1999 field methods guide. Research Triangle Park, NC: U.S. Department of Agriculture Forest Service, Forest Health Monitoring Program. 480 p. On file with: Forest Health Monitoring Program, 3041 Cornwallis Road, Research Triangle Park, NC 27709.

growth rates from increment cores, and forest floor disturbance.

RESULTS AND DISCUSSION

On 120 sites, a total of 2,763 trees were evaluated (the majority annually), with 2,065 being sugar maple. The overall mean plot sugar maple dieback level on industry land decreased from 14 to 9 percent from 2009–12 across the region; on public land, dieback decreased from 10 to 9 percent from 2010–12 (fig. 10.3). There were significant differences in mean plot dieback between years ($p < 0.001$), with a general increase until 2011 and decline in

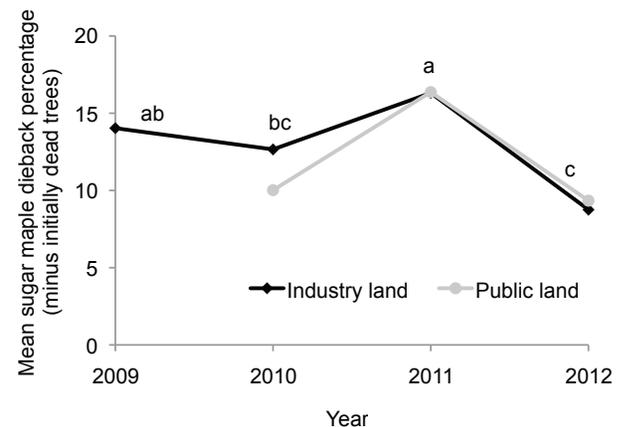


Figure 10.3—Mean sugar maple crown dieback percentage (excluding initially dead trees) from 2009–12 on industry (61 plots) and public (59 plots) lands in the western Upper Great Lakes region (Bal 2013). Letters indicate significant differences (repeated-measures analysis of variance, $\alpha = 0.05$) between years of means, independent of ownership.

dieback symptoms in 2012. Mean dieback values ranged considerably across the study area, with individual plots having from 0.8-percent (apparently healthy) to 75.5-percent (severely unhealthy) mean sugar maple dieback. A similar evaluation of more than 1,000 sugar maples across a wider regional gradient (the North American Sugar Maple Project) reported an average dieback percentage between 6 and 9 percent during 1988–90, which is lower than those found here (Allen and others 1992). The stresses currently impacting trees in the Upper Great Lakes region are likely ongoing or may be more severe than those that have occurred historically.

Mean plot values over 10 percent are usually indicative of an unhealthy stand, and values over 20 percent are typically considered moderate to severe (e.g., Allen and others 1992, Horsley and Long 1999). Although overall mean dieback decreased during the 4 years of this study, some individual trees and stands continued to decline rapidly. Four years of data collection may establish a good baseline for long-term monitoring, though interannual variation in precipitation and data collection timing may influence the amount of crown foliage and dead twigs seen and thereby limit the interpretation of long-term trends. In cases where the cause of the dieback is occurring gradually, such as a nutrient deficiency that weakens trees over decades, dead branches will snap off, reducing measured dieback levels (Watmough and others 1999). In addition, harvesting removed some

trees present with heavy dieback. Examination of basal area indices and growth rates are needed in some cases to determine evidence of decline that is not necessarily evident from mean dieback values. Crown conditions for sugar maple varied between plots, with mean crown transparency following a similar pattern as dieback over time, and mean crown density followed an inverse pattern over time, as would be expected given the relationships between these variables (fig. 10.4). Mean foliage density and canopy transparency plot values for each year have significant linear correlations with mean dieback plot values except for crown density in 2009 and

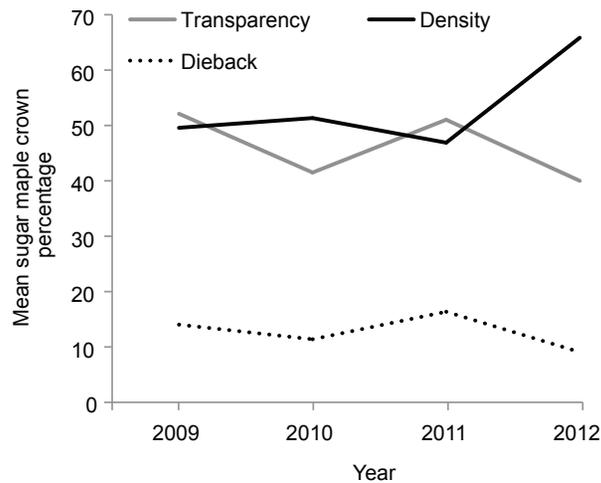


Figure 10.4—Mean sugar maple plot percentages of foliage transparency, crown density, and crown dieback during study period across the western Upper Great Lakes region (Bal 2013). Note: transparency in 2012 from only a subset of plots.

foliage transparency in 2011 (all others, $p < 0.05$). The increase in foliage density in 2012 could be masking dieback from surveyors, causing them to underestimate signs of stress in the canopies.

There was no significant difference in mean dieback between ownership types, public or industry (fig.10.3). Some difficulties exist in separating the effects of management and ownership type due to the industry land plots being primarily located in Houghton, Keweenaw, Baraga, and Marquette Counties, Michigan (in the northwestern Upper Peninsula of Michigan), while the public lands evaluated during the study had a much wider regional distribution (fig. 10.2). Regional differences in forest composition and biotic and abiotic stress may also be influencing sugar maple health. For example, eastern Upper Michigan stands have beech bark disease (*Fagus grandifolia*, *Cryptococcus fagisuga* and *Neonectria* spp.) present, which can influence sugar maple growth (DiGregorio and others 1999), but this was not the focus of this study.

CONCLUSIONS

The specific etiology of the current sugar maple dieback is still unclear but likely varies across more local rather than regional scales. Sapstreak (*Ceratocystis virescens*) was present in some stands but was ruled out as responsible for the extent of dieback seen across the region (Bal and others 2013). Further analysis of data indicates factors impacting trees may continue to persist, including climate extremes, poor soil

nutrients, and invasive earthworms impacting the forest floor condition (Bal 2013, Larson and others 2010).

This study provides a baseline for monitoring sugar maple health in the Upper Great Lakes region. Occurrence and severity of sugar maple dieback in the Upper Great Lakes region was variable and may be more prevalent than historically reported in the area. No differences in dieback amounts were detected between forest ownership. Forest managers should adopt practices that alleviate additional stresses such as increased soil disturbance and exposure in northern hardwood systems. In some cases, it may be appropriate to promote other species in anticipation of sugar maple dieback or reduced growth that may occur from stressors that will continue to impact forests such as climate change, invasive exotic species, defoliator outbreaks, and soil biogeochemistry perturbation.

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INTRODUCTION

Beech bark disease (BBD) has spread across roughly half of the range of American beech (*Fagus grandifolia*) in North America since it was introduced into Nova Scotia around 1890 (Erllich 1932, 1934; Gwiazdowski and others 2006; Houston 1994). The nonindigenous beech scale insect, *Cryptococcus fagisuga* Lind., mediates BBD by piercing the outer bark of beech trees, facilitating entry of the nonindigenous fungi *Neonectria faginata*, which colonizes only *Fagus* spp., or *Neonectria ditissima* (synonymous with *N. galligena*), which occurs on a variety of hardwood species in North America and Europe (Castlebury and others 2006, Houston and O'Brien 1983). The fungi kill small patches of phloem and cambium, and as dead tissues coalesce, large branches and the trunk are girdled (Burns and Houston 1987, Ehrlich 1934). Terms used to depict the three stages of BBD include the “advancing front” where trees are infested by beech scale, the “killing front” where trees are dying from fungal infection, followed by the “aftermath forest” characterized by beech mortality, infected “cull” trees, and, often, dense thickets of beech sprouts (Houston 1994, Houston and O'Brien 1983, Shigo 1972).

Presence of BBD in Michigan was first noted in 2000 in Mason County in the northwestern Lower Peninsula and Luce County in the eastern Upper Peninsula (UP) (McCullough and others 2001, O'Brien and others 2001). As of 2003, beech scale was present in four counties in northwest Lower Michigan and five counties

in the eastern UP, but there was little evidence of beech decline or mortality related to fungal pathogens (Kearney 2006). Inventory data indicate that more than 15 million merchantable beech [>22 cm diameter at breast height (d.b.h.)] are present on forest land in Michigan (Heyd 2005, McCullough and others 2001). Resource managers, including foresters and wildlife biologists, remain concerned about potential BBD impacts, but their ability to plan and prioritize stand-level operations (e.g., harvest, presalvage, regeneration) is limited by a lack of information about BBD distribution, spread, and effects (Ostrosky and McCormack 1986). Previous efforts to estimate BBD spread and quantify its impacts have been conducted in Eastern States, typically years after BBD establishment. Differences in topography, climate, soils, and forest attributes between the Lake States region and the Northeast could potentially affect BBD progression and impacts.

This project built on previous efforts undertaken following initial identification of BBD in Michigan. Progression of the advancing fronts in Lower and Upper Michigan were monitored from 2005 through 2009 using an adaptive sampling method to delineate distinct beech scale infestations (Schwalm 2009, Wieferich and others 2011) and iterative modeling processes to estimate spread rates of individual infestations. Results through 2009 indicated that spread rates varied considerably among years and among beech scale populations (Wieferich and others 2011) but were consistently lower than

CHAPTER 11.

Beech Bark Disease in Michigan: Spread of the Advancing Front and Stand-Level Impacts

(Project NC-EM-09-02)

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published estimates derived from observations in Northeastern States (Griffin and others 2003, Morin and others 2007). In 2002–03, 62 impact sites were established to collect baseline data on composition and condition of the overstory, regeneration, and coarse woody material. Sites represented three levels of beech basal area (low, moderate, and high) and three levels of beech scale infestation (absent, light, and heavy) (Kearney 2006). In 2003, beech scale was absent in 39 sites, 12 sites had light infestations, while 21 sites had moderate or heavy infestations. There was little evidence of beech decline or mortality attributable to BBD in any site in 2003.

Our primary goals in this project were to continue to monitor and delineate the advancing fronts of beech bark disease (BBD) in Lower and Upper Michigan and to assess spread rates. In addition, we resurveyed the original 62 sites to document current condition of beech and other overstory trees, species composition of regeneration, and the amount and composition of coarse woody material.

METHODS

Progression of the Advancing Front

We monitored the advancing fronts of BBD and delineated individual beech scale infestations using adaptive sampling (Thompson and Seber 1996). Sampling points were established in concentric circles 5 to 8 km from the farthest infested sites. If one or more beech were found to be infested with beech scale at a previously

uninfested point, another point was established farther out until an uninfested point was recorded. If no beech were present within 8 km of an infested site, “no beech” points were recorded to ensure the area would not be revisited, and the surveyor moved on. Each year, the uninfested points closest to infested points were revisited and the process was repeated until the infestation was surrounded by a buffer of uninfested sites (with beech). Infestations were considered to be distinct if there was ≥ 20 km of uninfested habitat between the edge of the infestation and the primary beech scale infestation or other satellites. Over time, as beech scale spread, some infestations coalesced and were subsequently considered as a single infestation.

In forested sites, a variable radius plot (determined using a prism with a basal area factor of 10) was established where beech and infested trees (if present) were concentrated, and d.b.h. was recorded by species for overstory trees (>10 cm d.b.h.). In sites with limited access and in nonforested sites (e.g., roadside trees), the first 10 beech trees encountered were measured. In all plots, beech trees were visually examined and ranked as: (1) beech scale absent; (2) trace (scattered, low-density beech scale); (3) patchy (several clumps of beech scale); or (4) whitewashed (one or more aspects of the trunk were heavily infested). Coordinates of plots were imported into ArcGIS® 10.1 (ESRI 2012), and infestations were mapped annually. Spread rates were estimated and areas of beech scale

infestations were calculated using minimum convex polygons for each distinct infestation and statewide.

Impacts of BBD to Date

The 62 sites with BBD impact plots, including 34 sites in 7 counties in Upper Michigan and 28 sites in 14 counties in Lower Michigan, were revisited in 2011–13 to assess impacts of BBD. Similar methods were used in 2002–03 and 2011–13 to assess beech scale and beech condition. The center plot and four subplots, 18.3 m from the center in each cardinal direction (all plots 7.3-m radius), were relocated, and GPS coordinates were recorded at the center plot. Species, d.b.h., and a visual estimate of beech scale abundance were recorded on trees in all five subplots. Twelve additional beech trees growing at equally spaced azimuths and within 60 m of the perimeter of the four subplots were tagged in 2002 and were also reexamined. These trees were originally selected using prioritized criteria: (1) largest tree with dead tissue or at least one canker, (2) any tree (pole-sized or larger) with a canker, (3) largest tree with *C. fagisuga* present, (4) any tree (pole-sized or larger) with *C. fagisuga*, or (5) the largest tree near the azimuth. We measured d.b.h., number of cavities and estimated crown dieback, transparency, and beech scale densities on each tree. Overstory species composition and tree and beech scale abundance were also recorded along three transects (each 25.5 m by 10 m) in the sites.

Frequency, species, size, and decay class of coarse woody material were recorded along

three transects (25.5 m by 1 m wide) in each site. Decay classes for coarse woody material (CWM) were defined as: (0) fresh material with intact bark and no obvious decay; (1) bark sloughing off or absent, but with solid inner sapwood; (2) some decay; small wood chunks break off under impact, but firm center; (3) decaying; loses form under impact; and (4) decayed and form lost (Kearney and others 2005). Regeneration plots were established equidistantly between the center plot and each of the four subplots in the cardinal directions in the sites. Seedlings (<30.5 cm tall), saplings (>30.5 cm tall; <2.5 cm d.b.h.) and recruits (>2.5 cm d.b.h.) were tallied by species within a 2.4-, 3.5-, and 7.3-m radius of the plot center, respectively.

RESULTS

Progression of the Advancing Front

From 2011 through 2013, we established a total of 544 sites (with beech) in 28 counties in Lower Michigan and 9 counties in Upper Michigan to monitor the advancing front. We examined 1,854 live beech trees in these sites; d.b.h. of these beech ranged from 4.2 to 119.5 cm and averaged 30.0 ± 0.4 cm. There were 187 sites with infested beech. On average, 86.0 ± 1.6 percent of the beech trees within infested plots had at least some beech scale. Across all sites, beech made up 53 ± 1.1 percent of the total basal area (all species), which averaged 16.8 ± 0.4 m²/ha. Along with beech, sugar maple (*Acer saccharum*), red maple (*Acer rubrum*), and northern red oak (*Quercus rubra*) dominated the overstory.

In Lower Michigan, we continued to survey four distinct beech scale infestations that were monitored from 2005 through 2009 (Wieferich and others 2011). The Mason-Wexford infestation, which encompassed 7727 km² in 2009, had expanded to 9200 km² in 2013. The Charlevoix-Crawford-Emmet infestation encompassed <3010 km² in 2011 but had expanded to 7088 km² in 2013. In addition, a relatively new infestation in Midland County occupied <2 km² when detected in 2011 but had increased to 287 km² by 2013. The most recent detection occurred early in 2014, when a small area in a nature center near Grand Rapids (Kent Co.) was found to be infested. More than 16 575 km² in Lower Michigan are now infested. Advancing fronts in Lower Michigan spread at an average rate of 3.3 ± 0.4 km/year from 2011 through 2013 and 3.1 ± 0.3 km/year from 2005 through 2013, but the rate of spread of individual infestations ranged from <2 to 14.8 km/year.

In the UP of Michigan, a single and nearly continuous beech scale infestation has been monitored since 2005. A distinct satellite population encompassing 255 km² was identified in Menominee County in 2009, more than 45 km west of the leading edge of the main front. This infestation coalesced with the primary infestation in 2011. The beech scale-infested area in the UP, estimated at 6214 km² in 2005, encompassed at least 13 530 km² in 2013. The leading edge of the infestation in the central Upper Peninsula is now ringed by no-beech points, although a few scattered areas with beech occur in the northwestern portion of the UP. Annual spread rates of the advancing front in

Upper Michigan have ranged from 6.2 ± 2.2 km/year between 2005 and 2007 to 2.0 ± 0.8 km/year between 2011 and 2013. On average, the infestation spread at 3.9 ± 0.7 km/year from 2005 through 2013.

Impacts of BBD to Date

In 2002–03, beech scale was present in 23 of the 62 sites, but by 2013, 55 sites were infested, including 44 sites that were heavily infested. Overall, 18 percent of the 1,440 beech trees examined in the 62 sites had died by 2013. There was little difference in the size of dead beech (average 34 ± 0.8 cm d.b.h.) and live beech (average 30 ± 0.4 cm d.b.h.). The killing front is progressing in Upper Michigan; 23 percent of the beech trees and 26 percent of the beech basal area in our plots are dead. In the Upper Peninsula sites that were infested with beech scale in 2002, 49 percent of the trees were dead in 2013. In contrast, in Lower Michigan, only 9 percent of the beech trees and <7 percent of the beech basal area was dead by 2013. Even in sites that had beech scale in 2002, only 8 percent of the trees had died. Canopy dieback and transparency levels were generally higher in Upper Peninsula sites (average of 12 percent and 16 percent, respectively) than in Lower Michigan sites (average of 7 and 13 percent, respectively).

We were able to find 724 of the 744 beech trees around the perimeter of the subplots that were tagged in 2002. Most trees (79 percent) were alive, 5 had died and fallen, 80 were dead but standing, and 46 had broken along the trunk (e.g., beech snap). Only 11 of the 314 beech in

Lower Michigan had died, while roughly 30 percent of the 368 beech in the Upper Peninsula were dead. In 2002, 500 of the tagged trees were not yet infested by beech scale, but by 2013, only 189 trees had not been colonized. Average d.b.h. of live and dead beech trees was 33.5 ± 1.0 cm and 23.5 ± 4.7 cm in Lower Michigan and 34.3 ± 0.8 cm and 39.5 ± 1.1 cm in the UP, respectively. Radial growth was higher for trees that were uninfested in 2003 than for trees that were already colonized by beech scale in 2003 (2.5 ± 0.2 and 1.6 ± 0.2 cm, respectively).

Coarse woody material (CWM) was encountered in every site in 2011–13, and 732 pieces were recorded. Beech made up 68 percent of the 310 fresh, identifiable pieces, and beech CWM was most abundant in heavily infested sites. Total CWM volume ranged from 7 to 312 m³/ha and averaged 74 ± 10.5 and 50 ± 8.4 m³/ha in sites in the Upper Peninsula and Lower Michigan, respectively. Beech CWM volume averaged 14.7 ± 3.2 m³/ha across all sites.

A total of 4,978 overstory trees, representing 19 species, were measured in subplots and transects in the 62 sites in 2011–13. Beech and sugar maple dominated the overstory. Other common species (from most to least dominant) included red maple, eastern hemlock (*Tsuga canadensis*), paper birch (*Betula papyrifera*), and red oak. As in 2002, regeneration was dominated by beech and maple. Beech accounted for only 11 percent of seedlings but made up 63 percent of all saplings, followed by sugar maple, which accounted for 18 percent of the saplings.

DISCUSSION

Spread rates of the advancing front of BBD in Michigan, monitored since 2005, have varied among years and among distinct beech scale infestations. Between 2005 and 2007, Schwalm (2009) reported populations spread at rates of 4.0 and 1.5 km/year in Upper and Lower Michigan, respectively. Wieferich and others (2011) monitored 12 distinct infestations and reported spread rates of individual infestations varying from 1.0 to 8.0 km/year in Lower Michigan. The highest spread rate was recorded when two originally distinct infestations coalesced and subsequently expanded at a rate of 14.3 km/year. In Upper Michigan, Wieferich and others (2011) reported the maximum spread rate of the single advancing front was 11.0 km/year. Our estimates indicate spread in the Upper Peninsula has slowed, probably because nearly the entire range of beech has been colonized. Spread rates in Lower Michigan remain highly variable, however, which may reflect the fragmented distribution of beech and forested land in general in this part of the State. We expect spread rates will slow as the advancing front reaches areas in the central and eastern Lower Peninsula where beech volume is relatively low. Recent identification of localized infestations in Midland and Kent Counties indicates long-range dispersal of beech scale continues, probably as a result of birds transporting eggs or crawlers. Uninfested beech within 15 km of beech scale infestations should be treated as high-risk sites, and BBD impacts should be considered as management plans are developed.

Effects of BBD are most pronounced in the Upper Peninsula, where the killing front continues to advance. Little beech mortality attributable to BBD was present in 2002–03 (Heyd 2005, Kearny and others 2005, McCullough and others 2001), but almost 25 percent of the beech trees we examined had been killed. In Northeastern States, 50 percent or more of the overstory beech reportedly died during the first wave of BBD (Kasson and Livingston 2012, Krasny and Whitmore 1992). This mortality rate is comparable to what we observed in our sites that were infested in 2003.

Beech thickets, which are common in many Northeastern forests, may proliferate following the relatively rapid death of previously healthy beech trees caused by BBD, increasing the abundance of susceptible beech stems (Griffin and others 2003, Hane 2003, Houston 1994). Thickets reduce light availability and survival rates of nonbeech seedlings and saplings, including sugar maple and red maple (Hane 2003, Kobe and others 1995, Twery and Patterson 1984). Although beech saplings were common, we observed a single beech thicket in only three sites in the Upper Peninsula, and none were present in Lower Michigan. In Upper Michigan, overstory composition will likely change as overstory beech die. Sugar maple was consistently abundant in our sites, and previous studies indicate canopy gaps resulting from BBD can increase radial growth and recruitment of young sugar maples into the overstory (DiGregorio and others 1999).

Beech mortality in the Upper Peninsula has resulted in an increase in snags and a pulse of coarse woody material. These changes may provide habitat for numerous birds, mammals, and salamanders (Davis 1983, Gilbert and others 1997, Kahler and Anderson 2006, Morrison and others 1986, Strojny and others 2010), although such benefits may be offset by reduced production of mast as overstory trees succumb (Hamelin 2011, Tubbs and Houston 1990). Studies in Northeastern States, for example, indicate bear reproduction is associated with abundance of beech nuts (Hamelin 2011). Our data also indicate beech scale infestation has reduced radial growth. Radial growth rates of trees that were not infested in 2003 were almost 65 percent higher than those of trees that were infested in 2003. Reduced growth of trees infested by beech scale has been similarly noted in other studies (Erlach 1934, Mencuccini and others 2005, Mize and Lea 1979).

It is not clear why beech mortality is so much greater in the Upper Peninsula sites than in Lower Michigan. Beech scale has been present in Lower Michigan for at least as long as it has been present in the Upper Peninsula, and in many of our sites, all or nearly all of the beech have been colonized by beech scale. The difference likely involves *Neonectria* sp. infection rates or virulence. Fruiting bodies have been observed more frequently in the Upper Peninsula than in Lower Michigan, while evidence of *Neonectria* remains difficult to find in Lower Michigan (O'Brien and others 2001). Further study will be needed to understand why beech mortality rates

remain relatively low in Lower Michigan despite the presence of beech scale for more than 15 years.

CONTACT INFORMATION

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INTRODUCTION

The Pinaleño Mountains in southeastern Arizona represent a Madrean sky island ecosystem that contains the southernmost expanse of spruce-fir forest type in North America. This ecosystem is also the last remaining habitat for the Mt. Graham red squirrel (*Tamiasciurus hudsonicus grahamensis*), a federally listed endangered species. Due to a general shift in species composition and forest structure of spruce-fir type forests across the Southwest, the ecosystem is being threatened by large high-severity fires, insect infestation, and a general loss of biodiversity. These risk factors have led the Coronado National Forest to begin a forest restoration effort using LiDAR (light detection and ranging) as a tool for identifying habitat and cataloging forest inventory variables at a landscape level. LiDAR was identified as an efficient tool for filling the data collection needs because field data collection is restricted due to rugged terrain and safety concerns.

This Pinaleño Canopy Mapping Project was divided into three phases. Phase 1 compiled the technical specifications for the LiDAR data acquisition. A request for quotes was posted, the contract was awarded, and airborne LiDAR data collected in 2008 (Laes and others 2008).

Phase 2 evaluated the acquired LiDAR data, applied image analysis techniques, and derived several forestry-based Geographic Information System (GIS) data layers. Eighty 0.05-ha forest

inventory plots were established during this phase in the 2009 field season. These data were used in the subsequent phase to establish statistical relationships between the conditions on the ground and the LiDAR data for modeling (Laes and others 2009).

Phase 3 modeled forest inventory parameters at the landscape level. Regression models were constructed using forest inventory parameters measured on field plots and their associated LiDAR canopy (plot) metrics (Mitchell and others 2012).

In addition to these three initial analysis phases, the effort has led to several ongoing research projects focusing on the Mt. Graham red squirrel and its habitat, an intended outcome of the original project proposal. Led by the University of Arizona Conservation Research Laboratory and Mount Graham Research Program, researchers have utilized the LiDAR GIS layers and modeling products from the Forest Service, U.S. Department of Agriculture Remote Sensing Applications Center (RSAC) to create additional LiDAR analysis layers and habitat models (Merrick and others 2013).

METHODS

The project area covers approximately 85,500 acres (34 600 ha) in the mixed-conifer and spruce-fir zones above 7,000 feet (2133 m) within the Pinaleño Mountains, located southwest of Safford, Graham County, Arizona (fig. 12.1).

CHAPTER 12.

Using LiDAR to Evaluate Forest Landscape and Health Factors and Their Relationship to Habitat of the Endangered Mount Graham Red Squirrel on the Coronado National Forest, Pinaleño Mountains, Arizona¹

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¹This work was previously published as Mitchell and others (2012) and parts of that report are included here in their entirety in this summary by the same authors.



Figure 12.1—Pinaleño Mountains in southeastern Arizona showing the LiDAR acquisition and forest inventory modeling area represented in a three-dimensional virtual globe environment (Mitchell and others 2012).

Data collected for this project included airborne LiDAR data and associated field plot data. The two datasets were collected with similar dates, and field data were collected specifically for use with high-resolution LiDAR data.

LiDAR Data

High pulse-density LiDAR data were acquired over the Pinaleño Mountains project area September 22–27, 2008. The dataset had a nominal pulse density of ≥ 3 pulses/m², > 50

percent side lap, and a scan angle within 14 degrees off nadir. The full LiDAR data collection specifications and quality assessment can be found in the Phase 2 report by Laes and others (2009).

Field Data

Field data were collected with the primary goal of addressing data needs to support LiDAR modeling. Eighty field plots were collected in the summer of 2009 based on a 500-m grid.

Plots were 1/20-ha (0.05-ha) fixed plots with a 12.62-m radius. Only 80 of the 200 potential plot locations were sampled due to extreme terrain, one of the primary reasons for the LiDAR project. All plots were permanently monumented and trees tagged.

Plot location maps for the 80 plots were created to assist the field crew. A color infrared aerial photo was used as a backdrop with the plots marked by a circle (fig. 12.2). LiDAR subsets

were also clipped from the data, which provided the field crew an additional 3-D visualization of the desired plot location. Using map products and predetermined plot coordinates, the field crew navigated to the potential plot location using GPS. The actual plot center location and elevation were recorded using a GPS unit (Trimble® GeoXH™) capable of submeter locational accuracy. Differential corrections were used to increase plot location accuracy. A relatively high level of positional accuracy is needed

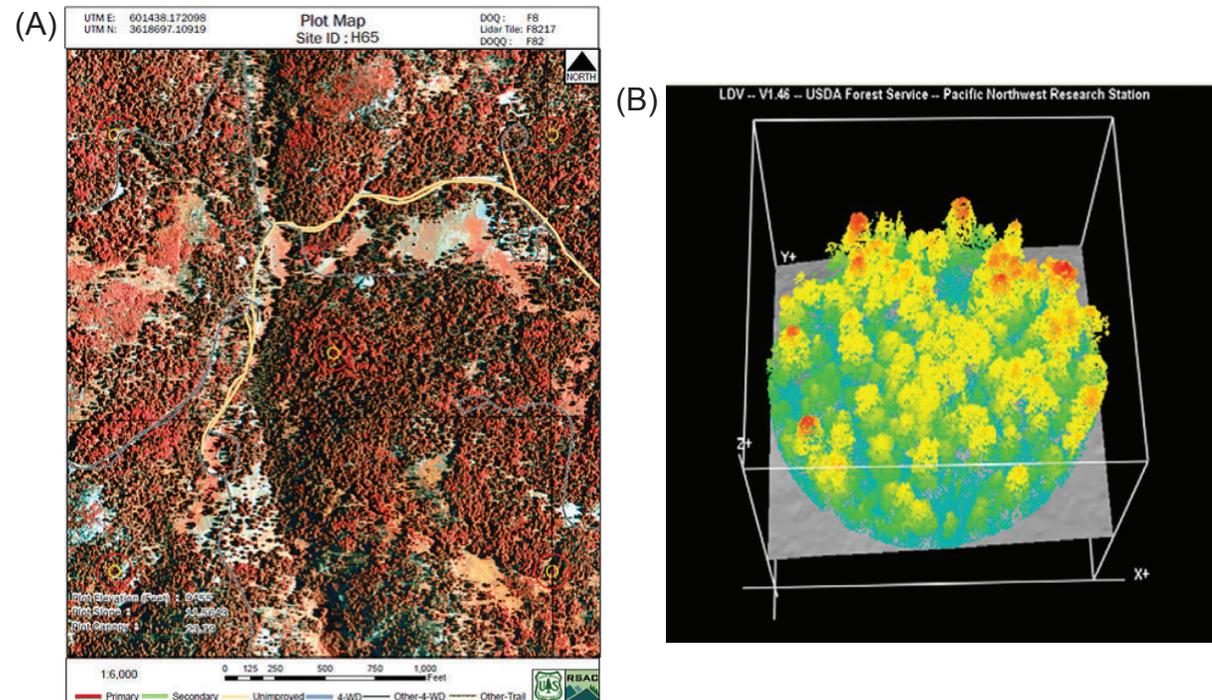


Figure 12.2—(A) Example of a map book page created for each of 80 field plots surveyed in support of the LiDAR modeling analysis for the Pinaleno Mountain study area. These map book pages helped field crews find the correct plot locations (Mitchell and others 2012). (B) Three-dimensional LiDAR point cloud visualization of the plot, which was also provided to the field crew for each potential plot location.

to minimize error and maximize correlation between field and LiDAR data in the modeling methodology.

All trees (live or dead) ≥ 20 cm in diameter at breast height (d.b.h.) and all coarse woody debris (down logs) > 20 cm were measured on each plot. To assess smaller trees and small coarse woody fuel, a 1/60-ha wedge-shaped subplot was created within the plot from a random radius bearing. All trees (live or dead) < 20 cm in d.b.h. and coarse woody debris < 20 cm but ≥ 5 cm in diameter were measured in the subplot. Three Brown's fuel transects (Brown 1974) and understory cover transects were used to measure shrub, forbs, grasses, and regeneration. Three photos were collected at each plot location.

The LiDAR and field inventory data were then processed and prepared for modeling. The goal of the data processing was to ensure a high level of correspondence between the data sets. Field inventory data were summarized to the plot level [e.g., basal area (BA), stand density index (SDI), average tree height, trees/ha (TPH), and quadratic mean diameter (QMD)] with corresponding metrics from the LiDAR data.

LiDAR predictor variables were generated at the plot scale using FUSION software (McGaughey 2012). The ClipData FUSION command was used to subset the LiDAR data. LiDAR metrics were calculated for each plot's point cloud using FUSION's CloudMetrics command. The resulting metrics became the predictor variables for the inventory models.

In summary, the plot level data were used to develop models to estimate the forest inventory variables while LiDAR metrics at the landscape scale were used in subsequent steps to apply the regression models to the entire landscape.

Model Development

Predictive models were created for 23 different forest and fuel inventory parameters (Mitchell and others 2012). Since all parameters being modeled are represented by continuous values, regression techniques were used to perform the modeling. Nonlinear regression was used to provide adequate predictive functions for each of the modeled parameters. All models created and applied to the landscape, along with the selected LiDAR predictor variables, are displayed in table 12.1.

The modeling process was conducted in five principal stages: (1) identify best predictors, (2) create appropriate modeling masks, (3) generate forest inventory models, (4) extrapolation, and (5) validation. To find the best linear predictors, a "leave one out" cross validation with a generalized linear regression model was used to find the LiDAR-derived parameters that best predicted the plot parameters. Forty LiDAR predictor variables were evaluated for each model. Two selected as additional variables did not contribute to model performance. Once these were selected, a curve-fitting application was used to identify the best nonlinear functions to correlate the LiDAR predictors with the modeled parameters of interest. The final models that were regarded as acceptable to apply to the

Table 12.1—Summarizations of the best linear and nonlinear forest inventory models created using LiDAR predictor variables to model field-derived forest inventory parameters (Mitchell and others 2012)

Predicted	Parameter 1	Parameter 2	Linear fit-R ²	Nonlinear fit-R ²	Nonlinear equation
TotBMKg (biomass in kg/ha)	Elev mean	(all returns above mean) / (total first returns) * 100	0.7416	0.8780	$z = (ax/b - cy/d) / (1 + x/b + y/d) + \text{Offset}$
BAwHGT (Lorey's mean height)	Elev mean	Elev skewness	0.8123	0.8265	$z = a + bx + cy + dx^2 + fy^2 + gx^3 + hy^3$
CUMVolTot (volume)	Elev P60	(all returns above mean) / (total first returns) * 100	0.7026	0.8216	$z = (a + b*\ln(x) + c*\ln(y) + d*\ln(x)*\ln(y)) / (1 + f*\ln(x) + g*\ln(y) + h*\ln(x)*\ln(y))$
BATotal (total basal area)	Elev P75	(all returns above mean) / (total first returns) * 100	0.7024	0.7782	$z = a + bx + cy + dx^2 + fy^2 + gx^3 + hy^3 + icy + jx^2y + kxy^2$
SDItotal (stand density index)	Percentage all returns above 3.00	(all returns above mode) / (total first returns) * 100	0.6994	0.7614	$z = a + b*y + c*y^2 + d*y^3 + f*y^4 + g*x + h*x*y + i*x*y^2 + j*x*y^3 + k*x*y^4$
HGTmax (height of tallest tree)	Elev L2	Percentage all returns above 3.00	0.6919	0.7529	$z = a + by + cy^2 + dy^3 + fx + gxy + hxy^2 + ixy^3$
CUMVoid (volume dead)	Percentage all returns above mean	(all returns above mean) / (total first returns) * 100	0.4038	0.7361	$z = (a + b*\ln(x) + c*\exp(y) + d*\ln(x)\exp(y)) / (1 + fx + gy + hxy) + \text{Offset}$
SdHGT (std tree height)	Elev variance	Elev P50	0.5802	0.6817	$z = (a + bx + cy + dxy) / (1 + f*\ln(x) + g*\ln(y) + h*\ln(x)*\ln(y)) + \text{Offset}$
BALive (live basal area)	Percentage all returns above mean	—	0.5993	0.67	$z = 0.0153*x^2 + 0.9306*x - 3.8899$
CBH (canopy base height)	Elev variance	Elev CV	0.4855	0.6187	$z = (a + b*\exp(x) + c*\exp(y) + d*v(x)\exp(y)) / (1 + fx + gy + hxy) + \text{Offset}$
CFL (canopy fuel load)	Percentage all returns above mean	(all returns above mode) / (total first returns) * 100	0.512	0.6	$z = (a + bx + cy + dxy) / (1 + f*\ln(x) + g*\ln(y) + h*\ln(x)*\ln(y)) + \text{Offset}$
QMD ^a (quadratic mean diameter)	Elev minimum	Elev P10	0.3932	0.4909	$z = (a + bx + cy + dxy) / (1 + f*\ln(x) + g*\ln(y) + h*\ln(x)*\ln(y))$
CBD ^a (canopy bulk density)	Elev P20	Percentage all returns above mean	0.3724	0.4378	$z = (a + bx + cy + dxy) / (1 + f\ln(x) + g*\ln(y) + h*\ln(x)*\ln(y)) + \text{Offset}$

— = no parameter 2 exists for BALive.

^a Model did not meet the R² threshold of 0.6, but was included based on interest of cooperators.

landscape based on a smoothness of fit and an R^2 value ≥ 0.6 are displayed in table 12.1. The models for quadratic mean diameter and canopy bulk density were included for further investigation based on interest from the project cooperators even though the models did not meet the R^2 threshold.

Some of the forest inventory parameters that performed poorly included downed woody fuels, TPH, and QMD. The poor performance of downed woody fuels models probably reflects the limitation of LiDAR ground-filtering models to differentiate between the modeled ground surface and coarse woody debris. TPH and QMD are difficult to estimate due to the large variation in tree size. Both QMD and TPH could have benefited from a second round of modeling in which trees under a certain diameter class (< 5 cm) were excluded from the model. This is typically done in traditional forest inventories where numerous small trees wash out the midstory and overstory characteristics of the unit being measured. Parameters governed by larger tree size, such as BA, SDI, volume, biomass, and Lorey's mean height, performed well in our study. Lorey's mean tree height (height of a tree of average BA) holds particular promise ($R^2 = 0.83$) with LiDAR inventories and could be used in a similar fashion as QMD to represent the average tree of a stand. Since all tree heights within the plots were measured, we were able to calculate Lorey's mean tree height and develop models for this parameter.

LiDAR may be superior to field estimates at measuring parameters that are indirectly measured in the field, such as crown bulk density, but ascertaining this will require more intensive studies. For example, crown bulk density is estimated from very costly whole tree clipping studies but can be mathematically modeled using more easily measured tree attributes (height, diameter, and species). Clipping studies utilizing LiDAR data may be required to develop better models.

Our models for estimating continuous forest inventory measurements across the LiDAR acquisition area performed poorly in two areas: areas that fell outside the range of plot measurements and thus required the model to extrapolate estimates, and nonforest areas. Predicted data outliers were masked and clipped in the first circumstance, and in the latter, nonforested areas were excluded from the modeling process.

Continuous inventory parameters were created at the landscape scale by applying the derived equations (table 12.1) to the LiDAR GridMetrics layers. Each calculation produced a new grid of cells with 25-m side lengths, each of which spatially represented the estimated forest parameter of interest derived from the LiDAR data (fig. 12.3). These models were validated using local biological, silvicultural, and ecological knowledge of the study area alongside ancillary GIS data and imagery.

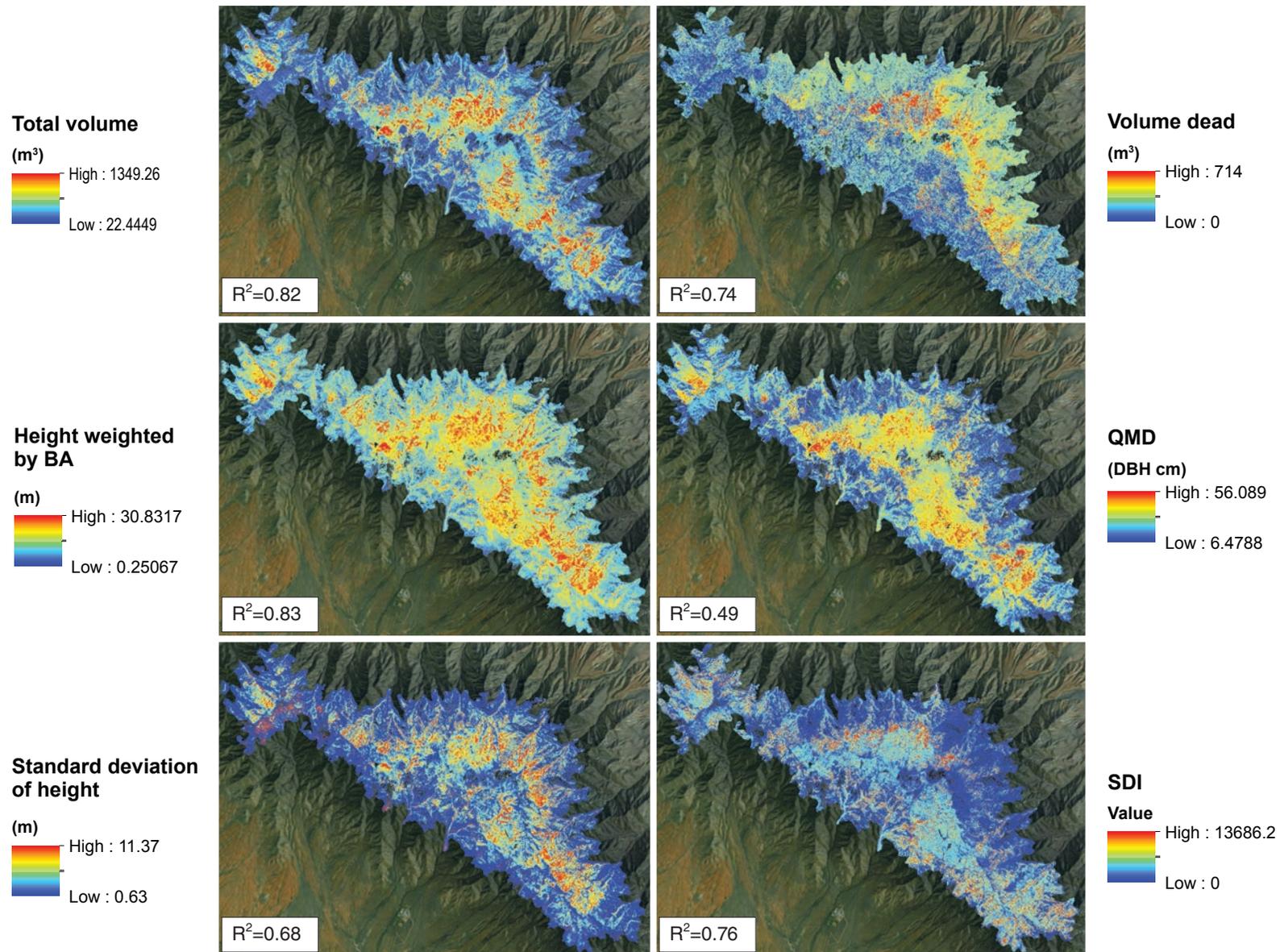


Figure 12.3—The Geographic Information System (GIS) grid layers (25-m cell size) represent the forest inventory parameter models applied at the landscape level (Mitchell and others 2012). These GIS layers are the end user products that will be used for future decisionmaking, analysis, and monitoring for the Pinaleno Mountain study area. Note: QMD = quadratic mean diameter, DBH = diameter at breast height, SDI = stand density index, BA = basal area.

Since LiDAR technology directly and continuously measures the structural characteristics of the forest vegetation across the landscape, and the modeling methodology used has been replicated and accepted internationally (Means and others 2000, Naesset 2002), we are confident that our statistically validated models provide reasonable results representing trends known to exist on the landscape.

CONCLUSIONS

The Pinaleno Canopy Mapping Project illustrates that forest inventory parameters measured in the field can be successfully modeled across the landscape with continuous LiDAR data. Parameters such as biomass (above ground), basal area, Lorey's mean height, and timber volume appear to lend themselves to this methodology, which is not surprising as they are directly related to tree size and density of vegetation. Our methodology failed to adequately model trees/ha or any of the down woody debris parameters.

The landscape GIS layers are also being used by the Coronado National Forest to create better strategies for managing and conserving the Pinaleno sky island ecosystem. The first-order LiDAR derivatives, modeled parameters, and the landscape GIS inventory layers created from this project are currently being incorporated

into habitat characterization studies for the Mt. Graham red squirrel. In particular, the University of Arizona Conservation Research Laboratory's work has utilized the project's products. The lab's interests have focused on factors influencing natal dispersal movements and habitat selection during adult red squirrel settlement. The researchers are currently testing several hypotheses related to natal habitat preference within mixed conifer forest, decision rules habitat selection, habitat fragmentation, wildfire impacts, and Forest Service restoration treatments and their influence on red squirrel dispersal movements and settlement patterns. These studies utilize the Pinaleno LiDAR data in conjunction with other field-based measurements to model red squirrel occurrence and settlement, to characterize forest structure, to develop canopy connectivity indices and identify dispersal thresholds, and to identify forest structural features associated with long-term Mt. Graham red squirrel occupancy (Merrick 2014) (fig. 12.4).

This project, from scoping and data acquisition to production of usable GIS layers, took approximately 65 weeks. We have demonstrated that first-order LiDAR derivatives, such as canopy height and percent canopy cover, can be used in natural resource management activities without the added cost of field data collection and in-depth modeling scenarios

described in this report. The LiDAR approach to obtaining forest structure data had several benefits over a ground-based approach. First, it provided continuous coverage of all forested areas, rather than stand- or plot-level estimates of various parameters created from stratified sampling methods. Second, it sampled areas that field crews could not safely measure due to extreme terrain. Third, it was very cost effective. The Coronado National Forest estimates that obtaining (statistically less valid and complete) data sufficient to implement the Pinaleño forest restoration project and other anticipated projects would cost approximately \$500,000 (assuming that crews could safely work in all areas, which is not the case). By comparison, the Pinaleño LiDAR mapping project cost \$250,000 including acquisition, processing, and analysis.

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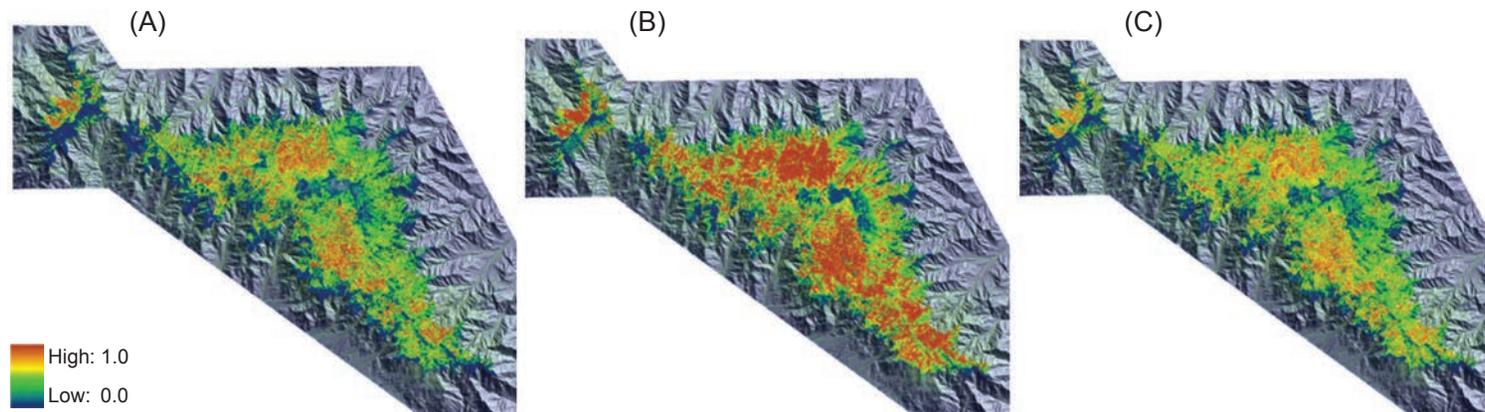


Figure 12.4—Mt. Graham red squirrel habitat maps with probability of use for (A) juveniles, (B) adults, and (C) all ages. (Source: Mitchell and others 2012)

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INTRODUCTION

Prior to fire suppression and exclusion, wildfires and other disturbances (e.g., insects, disease, and weather) sustained ecosystem processes in many landscapes of the Western United States. However, wildfires have been increasing in size, frequency, and intensity in recent years (Kellogg and others 2008). Recognizing the value of wildfire, scientists and land managers now promote allowing non-human-caused fires to burn in these landscapes, hoping fire can recreate the historical distribution and mosaic of presettlement, burned forests.

In some wilderness areas, these large natural fires are now burning multiple times in a given area, and the time between fires is decreasing (Halofsky and others 2011). Reburns have the potential to alter vegetation, fuel, and site and soil characteristics and can also alter landscape fire dynamics. Successive fires in a short amount of time can have major consequences for vegetation community structure and soil organic carbon.

The goal of this study was to quantify and characterize the vegetation, surface wood, and soil characteristics associated with successive fire events within the Scapegoat Wilderness, Montana. Our specific purpose was to address the following: How do multiple burns and their frequency influence (1) the extent and degree of changes in coarse wood (standing and down), aboveground carbon (C) pools, and surface and subsurface C and nitrogen (N) levels; and (2) changes in vegetation composition and structure?

METHODS

Site Description

Study sites were located in the Scapegoat Wilderness, Montana, near the Lake, Cabin, Canyon, and Dry Fork Creeks of the Blackfoot River Drainage. The area has burned multiple times (fig. 13.1). The first of the series of fires was the 240,000-acre (96 000 ha) Canyon Fire, started on June 25, 1988 from a lightning strike that burned 75 percent of the Wilderness. In October 1988 and the following summer, a total of 14 plots were established to monitor vegetation recovery after the fire.¹ Two more fires occurred after plot establishment: the Cabin Creek Fire in 2001 [13,300 acres (5338 ha)] and the Conger Creek Fire in 2007 [14,000 acres (5600 ha)]. In the summer of 2012, we revisited 13 of the 14 previously established plots that had burned in the 1988 Canyon Creek Fire. One plot had been previously burned in the Boy Scout Fire (burned in 1952) and this plot and the remaining 12 plots were subsequently burned in the Cabin Creek Fire (2001) and or Conger Creek Fire (2007). The years of individual fire events and their associated plots are shown in table 13.1.

Soils in this area have developed on glaciated uplands, glacial valley trains, or alluvial deposits. Most soils have developed in volcanic ash-influenced loess overlying metasedimentary rock. The area is generally

¹ Losensky, B.J. 1989. Canyon Creek fire vegetation study. 23 p. Unpublished report. On file with: Lolo National Forest, 24 Fort Missoula Road, Missoula, MT 59804.

CHAPTER 13.

Reburns and their Impact on Carbon Pools, Site Productivity, and Recovery

(Project INT-EM-F-11-02)

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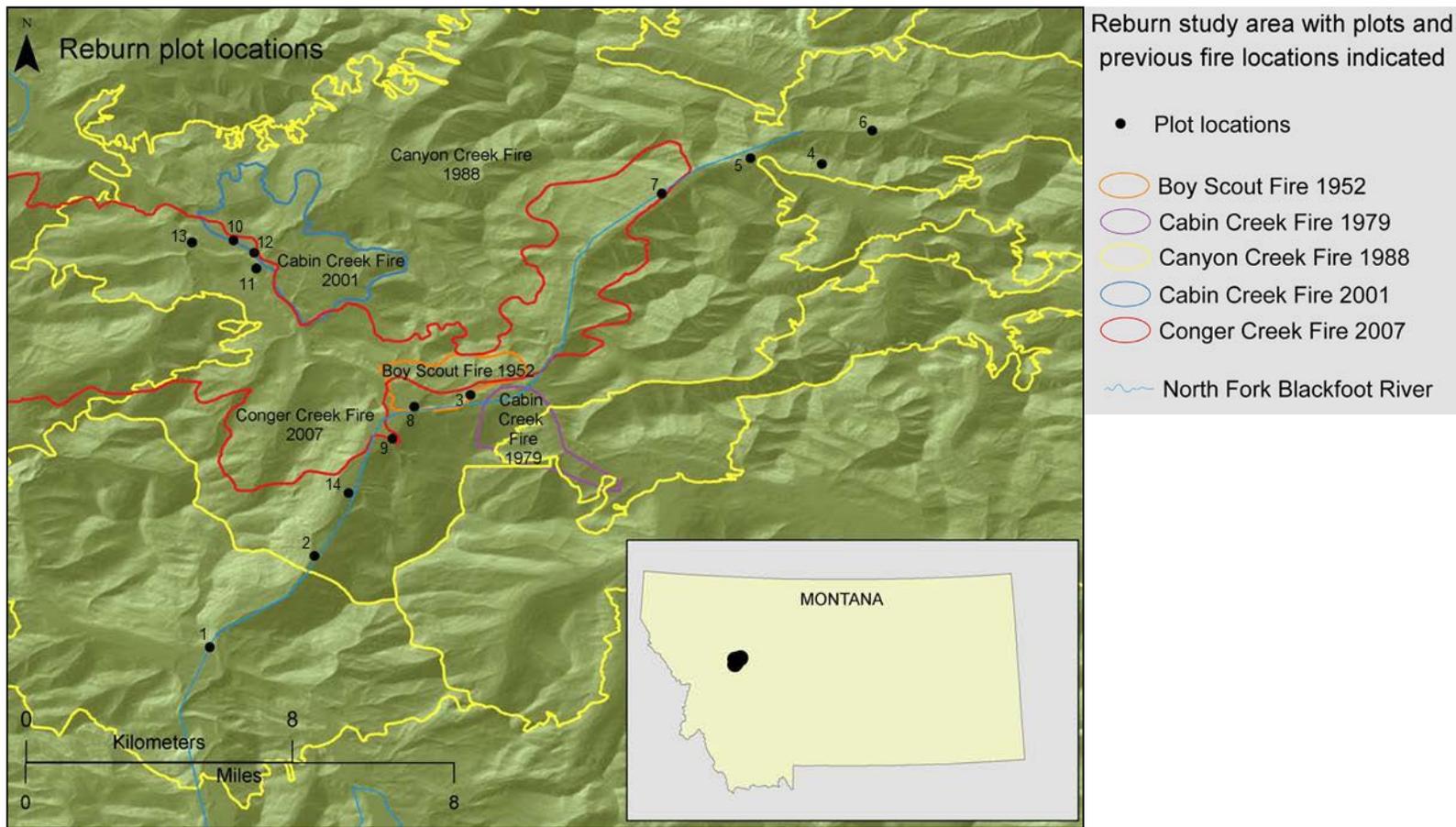


Figure 13.1—Scapegoat Wilderness reburn study area with plots and previous fire locations indicated.

Table 13.1—Plots numbers, year of burn(s), and recovery time after reburns of the Canyon Creek, Cabin Creek, Conger Creek, and Boy Scout Wildfire plots revisited in 2012 in the Scapegoat Wilderness, Montana

Plot numbers	Wildfire name	Year(s) of burn	Recovery time since last fire event
			<i>years</i>
1, 2, 4, 5, 6, 9, 14	Canyon Creek	1988	24
7, 13	Canyon Creek and Conger Creek	1988 and 2007	5
8	Boy Scout and Canyon Creek	1952 and 1988	24
10, 12	Canyon Creek, Cabin Creek, and Conger Creek	1988, 2001, and 2007	5
3	Boy Scout, Cabin Creek, and Conger Creek	1952, 1988, and 2007	5

Note: plot locations are shown in figure 13.1. Plot 11 was not revisited in 2012.

aligned in a north-south direction, and has a precipitation gradient decreasing from west to east. The western portion of the study area has a modified maritime climate, while the east side is continental; annual precipitation ranges from 40 cm in the valleys to 350 cm at high elevation (Keane and others 1994). Elevations range from 970 m to 3200 m.

Climax species include subalpine fir [*Abies lasiocarpa* (Hook.) Nutt.] and Douglas-fir [*Pseudotsuga menziesii* (Mirb.) Franco], with some western hemlock [*Tsuga heterophylla* (Raf.) Sarg.] or ponderosa pine (*Pinus ponderosa* Lawson and C. Lawson). Because of wildfire, many seral species, such as larch (*Larix occidentalis* Nutt.), lodgepole pine (*Pinus contorta* Douglas ex Loudon), and aspen (*Populus tremuloides* Michx.), as well as some shrubfields, can be found across much of this area.

Field Methods

Data presented in this summary were collected in 2012. Only vegetation data were collected in the original sampling in 1988 and 1989. Coarse wood and ground cover biomass are not presented here.

Soil sampling—After postfire index category classification (as described below), the forest floor (if present) was sampled from each postfire index category present. Soil cores were collected from 0- to 10-cm and 10- to 20-cm depths with a small-diameter (2 cm by 10 cm) corer. Soil and forest floor samples were placed in zip-type bags for transport. In addition, digital photographs were taken at each point to systematically determine soil cover using the Cover Monitoring Assistant computer program (Steinfeld and others 2011). Soil C and N were analyzed on

sieved (<2 mm) soil using an induction furnace. Soil nutrient concentrations were converted to a mass basis using the fine fraction bulk density (Page-Dumroese and others 1999). Woody material located on the soil surface was quantified by data collection on three transects. Small twigs and branches were tallied in three classes (0 to 60 mm, 61 to 250 mm, and 251 to 750 mm). All logs >750 mm had their diameter and decay class recorded (Sollins 1982).

Quantifying postfire conditions—Based on historical data collected, each plot was assigned an initial postfire index (PFI) as described by Jain and others (2012) to classify vegetation conditions immediately after the 1988 fire. This index is based on the amount of surface organic matter remaining postfire, with low index values indicating that larger amounts of vegetation remain on the site, and subsequently higher values indicating less organic matter remaining postfire. Each plot was revisited in 2012, and a

new soil PFI was assigned for each of four sample points in the plot.

RESULTS AND DISCUSSION

Soil

Soil cover (percent) was surprisingly similar for most plots (table 13.2). However, there were a few exceptions. For example, there was 13 percent bare soil on plots burned three times with 5 years of recovery. Higher levels of bare soil likely indicate most of the forest floor was removed during the three burns, and vegetation has not recovered sufficiently to develop a uniform forest floor layer. In addition, total soil profile C and N pools were also relatively similar (table 13.3). The notable exception here is that plots with three burns and 5 years recovery had the least amount of soil C in the forest floor but the most C in the mineral soil horizons, indicating possible relocation of organic materials after a fire or decomposition of dead roots. Fire

Table 13.2—Ground cover (percent) and total amount of coarse wood as related to recovery time and number of reburns from the Canyon Creek, Cabin, Boy Scout, and Conger Wildfire plots revisited in 2012 in the Scapegoat Wilderness, Montana

Recovery time	Plot numbers	Number of burns	Bare soil	Charcoal	Forest floor	Forbs	Grass	Shrubs	Rocks	Total Coarse Wood (all size classes)
<i>years</i>			----- <i>percent</i> -----							<i>Mg/ha</i>
24	1, 2, 4, 5, 6, 9, 14	1	6	<1	18	28	20	21	2	193
24	8	2	4	<1	29	36	1	12	0	88
5	7, 13	2	8	1	22	38	25	3	1	134
5	3, ^a 10, 12	3	13	1	25	28	24	3	3	80

^aAlthough plot 3 was burned by different fires, for these analyses it was combined with plots 10 and 12 with 3 burns.

Table 13.3—Forest floor and mineral soil C and N pools as affected by recovery time and number of reburns from the Canyon Creek, Cabin, Boy Scout, and Conger Wildfire plots revisited in 2012 in the Scapegoat Wilderness, Montana

Recovery time <i>years</i>	Number of burns	Plot numbers	Forest floor		0-10 cm mineral soil		10-20 cm mineral soil		Total soil profile	Total soil profile
			C <i>Mg/ha</i>	N <i>kg/ha</i>	C <i>Mg/ha</i>	N <i>kg/ha</i>	C <i>Mg/ha</i>	N <i>kg/ha</i>	C <i>Mg/ha</i>	N <i>kg/ha</i>
24	1	1, 2, 4, 5, 6, 9, 14	17.1 (1.8)	419 (37)	25.4 (2.0)	1316 (152)	16.4 (2.0)	889 (86)	58.9	2624
24	2	8	49.1 (4.5)	730 (74)	13.6 (1.5)	571 (87)	6.9 (1.0)	335 (41)	69.6	1636
5	2	7, 13	30.9 (3.2)	1186 (115)	17.5 (2.1)	860 (109)	10.9 (1.7)	572 (87)	59.3	2618
5	3	3, ^a 10, 12	12.3 (2.1)	197 (54)	29.6 (2.3)	1638 (156)	22.6 (2.7)	1284 (183)	64.5	3119

Values in parentheses are standard error of the mean.

^aAlthough plot 3 was burned by different fires, for these analyses it was combined with plots 10 and 12 with 3 burns.

changes to soil surface conditions and mineral soil properties can cover a spectrum depending on burn intensity, duration, fuel loading, combustion type, vegetation type, fire climate, slope, topography, soil texture and moisture, and soil organic matter content (Neary and others 1999). Therefore, the relative lack of differences and variability we detected in 2012 for each of the fire recovery times and number of fires is to be expected. More detailed work is needed to elucidate cause and effect.

Vegetation

As noted earlier, vegetation coverage measurements were made as part of the original site data collected in 1988 and 1989. Based on these measurements, values of PFI were calculated for each plot for the 13 plots revisited in 2012. Four of these plots still had much of the forest floor cover (>85 percent) intact (soil PFI = 1.0), four plots averaged 61 percent forest

floor cover (PFI = 2.1), and four plots averaged 15 percent forest floor cover (PFI > 3.0) (table 13.4). Jain and others (2012) related the PFI values on other postfire sites to the physical, chemical, and biological state of the soil at the Scapegoat Wilderness plots in 2012. These sites correlate with the C and N data we obtained from the soil samples and with the soil cover data derived from the digital photographs at the Scapegoat plots (table 13.2). Percentages of soil covered by gray and black ash were also calculated (table 13.4). Gray ash is used as an indicator that soil nutrients such as N were likely volatilized during the fire. A dominance of black ash indicates that some volatilization may have occurred, but much of the organic C is still present. The multiple fires since 1988 have resulted in a decrease in forest floor cover. However, the plots with soil PFI >3.0 have resulted in a trend toward a shrub response with additional leaf litter inputs; this is most likely contributing to increased forest floor cover.

Table 13.4—Soil Post-fire Index (PFI) based on forest floor and surface soil conditions immediately after 1988 Canyon Creek Wildfire along with forest floor and low shrub (<15 cm tall) covers observed in 2012, Scapegoat Wilderness, Montana

PFI	No. of obs.	Immediate post-burn (1988 Canyon Creek Fire)				2012	
		Forest floor	Black ash	White ash	Mineral soil	Forest floor	Low shrub
----- percent -----							
1.0	4	85.9 (9.2)	2.3 (1.6)	3.1 (4.9)	0.3 (0.4)	26 (41)	6 (8)
2.1	4	60.8 (7.8)	13.4 (5.7)	8.5 (5.9)	3.1 (5.7)	26 (49)	8 (5)
> 3.0	4	15.4 (9.7)	0.8 (0.7)	42.7 (18.9)	0.8 (0.7)	45 (31)	19 (16)

Note: The index was developed by Jain and others (2012) and reflects a full range of post-fire outcomes. The post-fire classification consisted of no evidence of recent fire (0); evidence of fire with > 85% forest floor cover (1.0); evidence of fire with >40% forest floor cover and mineral soil is charred black (2.1), gray/white, or orange (2.2); evidence of fire with <40% forest floor and mineral soil is unburned (3.2), charred black (3.2), gray/white (3.3), or orange (3.4). For each PFI, average cover and standard deviation (in parentheses) are provided for forest floor characteristics immediately after the 1998 fire and in 2012. All values reported in percent cover.

After 24 years, the forest floor cover, canopy cover, shrub cover, and biomass were no longer different among the plots ($p > 0.05$).

Canopy cover and total biomass were related to time since fire (24 years versus 5 years, tested at $p = 0.05$) and not related to number of fires (one, two, or three; data not shown). This is not surprising, as longer recovery times after a disturbance provide an opportunity for the vegetation to respond. The reburns in 2007 killed many established seedlings and consumed much of the woody debris, leaving only grass and forbs with more mineral soil exposed (fig. 13.2A). In contrast, the plots that were 24 years postfire had more trees, shrubs, and overall biomass. Saplings, some above 2 m tall, were established; low shrubs dominated the understory with some forbs and grasses. In addition, there was more woody debris created from snags as they fell over time (fig. 13.2B).

CONCLUSIONS

Few differences in soil surface or mineral soil conditions were found between plots in our 2012 survey, and the similarity is likely due to terrain, soil, and climatic conditions at each plot at the time of each fire. It is notable that the plots with three burns and only 5 years of recovery time had the lowest C and N in the forest floor, highlighting the loss of forest floor mass and the short recovery time. This trend is similar to the results from our vegetation survey. At the same time, sites within these three wildfire burn areas, even after multiple burns, show some level of recovery.

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Figure 13.2—(A) Recent reburn removing seedlings, shrubs and large downed wood; and (B) regeneration and large wood after site recovery. (Photos by Jonathan Sandquist, U.S. Department of Agriculture Forest Service)

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INTRODUCTION

Recent Forest Inventory Analysis (FIA) plot remeasurements revealed a statistically significant 4.6 percent loss of shore pine (*Pinus contorta* subsp. *contorta*) biomass in Alaska despite negligible harvest, with greater losses among larger size-class trees (Barrett and Christensen 2011). Shore pine is one of four distinct subspecies of *Pinus contorta* (Critchfield 1957); it occurs on coasts and wetlands from northern California to Yakutat Bay in southeast Alaska. In Alaska, shore pine is most common on peatland sites with saturated, acidic soils (known as muskegs) and is outcompeted by western hemlock (*Tsuga heterohpylla*) and Sitka spruce (*Picea sitchensis*) on more productive sites that have better drainage and nutrient availability (Bisbing and others 2015, Martin and others 1995). Little is known about the insect and disease agents of shore pine (Reeb and Shaw 2010).

Although shore pine is not a commercial species, it serves an important structural and ecological role in forested wetlands (Lotan and Perry 1983, Martin and others 1995). Few tree species are able to survive the harsh conditions of saturated and acidic peatland bogs and fens; therefore, there is concern that the loss of shore pine from these habitats may create a void that other tree species are unable to fill. This project was initiated to better understand the decline in shore pine biomass from tree mortality in southeast Alaska. Installation of a permanent plot network will provide baseline information about the insects, disease, and other damage

agents that affect the health of shore pine, and it will offer an opportunity to monitor shore pine populations over time. This information will help to determine if there is reason for heightened concern regarding the health and survival of shore pine in southeast Alaska.

METHODS

Site Selection

Sites were established at five locations in southeast Alaska (fig. 14.1). Plot locations were randomly selected from National Wetland Inventory (NWI) polygons (Cowardin and others 1979) known to reliably contain shore pine (palustrine emergent wetland and palustrine scrub-shrub wetland) that were at least 4 acres in size and located within 0.5 miles of a road or trail. Geographic Information System tools from ArcMap® 10.0 (ESRI 2011) were used to assess selected wetland polygons to ensure accessibility and shore pine forest type with satellite imagery and topographic maps.

Plot Layout and Data Collection

Forty-six plots were established using a modified FIA plot layout (USDA Forest Service 2007). Plot positioning maximized shore pine composition and captured a range of shore pine size classes. Live and dead trees ≥ 4.5 feet tall were tagged for long-term monitoring. Data collected from live trees included height; diameter at breast height (d.b.h.); lower crown height; crown dieback (percent, visually estimated); wound type and severity; and presence of conks, decay, or topkill. Height, d.b.h., FIA-defined

CHAPTER 14.

Factors Contributing to Shore Pine Mortality and Damage in Southeast Alaska

(*Pinus contorta* subsp. *contorta*)
(Project WC-EM-B-12-03)

ROBIN MULVEY
TARA BARRETT
SARAH BISBING

decay class from 1 to 5 (USDA Forest Service 2007), and wound or damage information were collected from snags. For shore pine, we also quantified western gall rust (WGR) severity and associated dieback, and we estimated foliage retention, length of branches with foliage, and disease/insect damage type and severity. The WGR severity rating was adapted from the Hawksworth (1977) dwarf mistletoe rating system. Wound severity was determined by the relative circumference of the bole affected. Symptomatic foliage was collected from shore pine to facilitate identification of foliar pathogens. Prism counts, slope, aspect, cover percentage of vegetation types and plant species, and one dominant shore pine tree core were collected in all three subplots.

RESULTS

Data were collected in 2012 and 2013 from 5,452 trees >4.5 feet tall (table 14.1), including 1,031 trees \geq 5 inches d.b.h. Tree species included shore pine, yellow-cedar (*Callitropsis nootkatensis*), mountain hemlock (*Tsuga mertensiana*), western hemlock, Sitka spruce, western redcedar (*Thuja plicata*), and red alder (*Alnus rubra*). On average, there were 62 shore pine and 119 total trees per plot, with 19 pines and 22 total trees >5.0 inches d.b.h.

Percentage Dead and Snag Decay Classes

A higher percentage of shore pine (13 percent) and yellow-cedar (14 percent) trees were dead compared to other tree species (<5 percent). Because mountain and western hemlock were less abundant and at least 18 of 73 snags

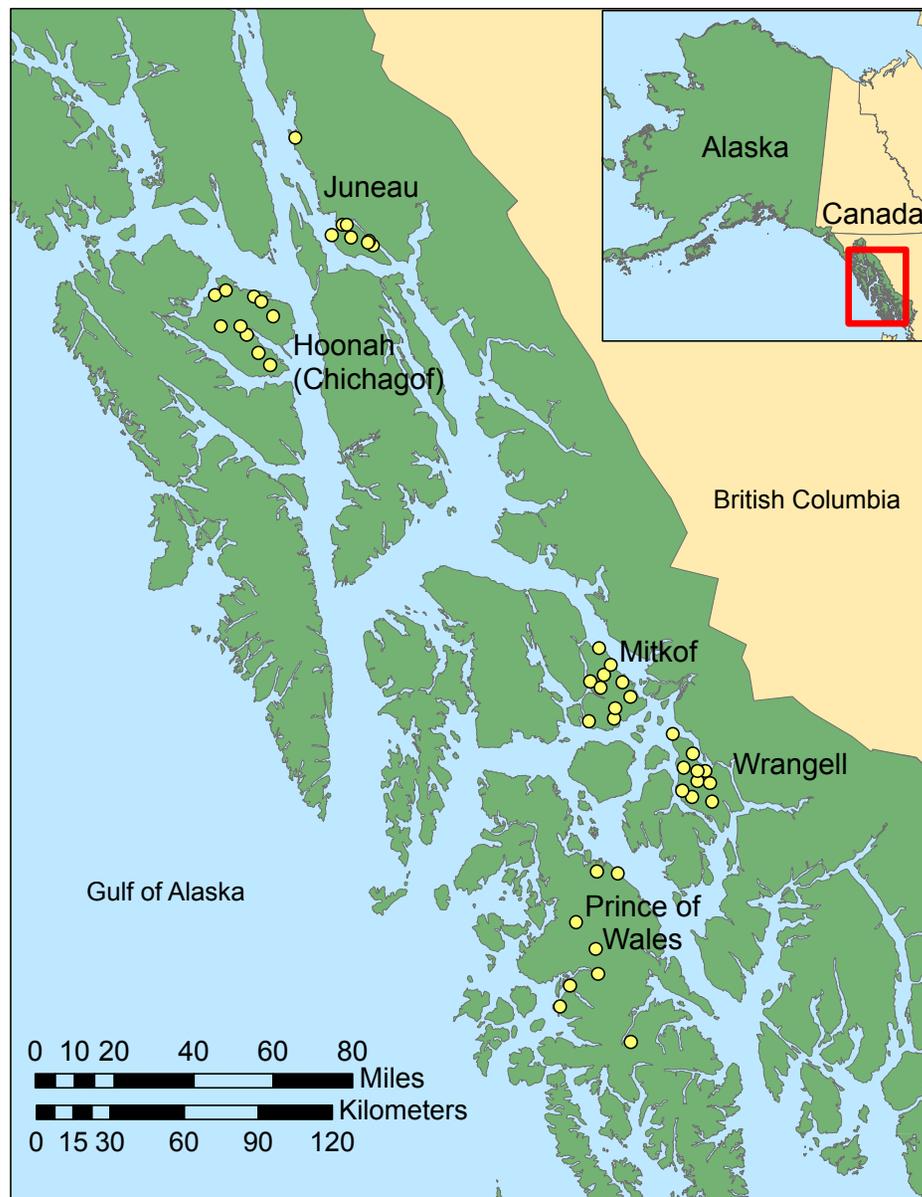


Figure 14.1—Forty-six permanent shore pine plots at five locations in southeast Alaska.

that could not be definitively identified were thought to be hemlock, percentage dead may be underestimated for these species. Among shore pine, the highest percentage of snags occurred in the largest diameter class (15.0–22.5 inches) (table 14.2). This diameter class contained just 25 trees; 5 of the 10 snags were designated decay class 2, with the remainder spread evenly among the other decay classes. There was no discernable trend for pronounced recent mortality in any particular diameter size class.

Western Gall Rust Incidence and Ratings

Western gall rust, caused by the fungus *Peridermium harknessii*, was detected in all subplots and on 85 percent of live shore pine. Among plots, infection incidence ranged from 52 to 100 percent. For snags, WGR incidence was only 32 percent and bole gall incidence was 21 percent. A greater percentage (22 percent) of small trees (0.1–2.4 inches d.b.h.) were uninfected, compared to <8 percent for all other size classes. A 0-to-6 scale quantified WGR on live trees (each vertical one-third of crown has a maximum rating of 2): 39 percent of shore pines were rated 1 to 2 (low severity), 36 percent were rated 3 to 4 (moderate severity), and 10 percent were rated 5 to 6 (high severity). The largest trees (>15 inches d.b.h.) had the lowest proportion of moderate to high severity ratings (33 percent of trees rated 3 to 6). The highest severity rating among the one-third crown portions (2) was more common in the lower crown (47 percent of live trees) than in the upper and middle crown (27 and 14 percent, respectively).

Table 14.1—Number of live and dead trees ≥4.5 feet tall and percentage of trees dead by species in 46 plots in southeast Alaska, 2013

Species	Live	Dead	Total trees	Trees dead <i>percent</i>
Shore pine	2504	361	2865	13
Yellow-cedar	1113	177	1290	14
Mountain hemlock	577	32	609	5
Western hemlock	467	20	487	4
Sitka spruce	60	3	63	5
Western redcedar	60	0	60	0
Red alder	5	0	5	0
Unknown	0	73	73	100

Table 14.2—Number of live and dead shore pine and percentage of trees dead by diameter class in 46 plots in southeast Alaska, 2013

d.b.h. ^a <i>inches</i>	Live trees	Dead trees	Total trees	Trees dead <i>percent</i>
0.1–2.4	1109	137	1246	11
2.5–4.9	655	109	764	14
5.0–6.9	289	45	334	1
7.0–9.9	263	36	299	12
10.0–14.9	173	24	197	12
15.0–22.5	15	10	25	40
Total	2504	361	2865	13

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

Bole galls were observed on 35 percent of live shore pine (9 to 78 percent per plot). WGR-associated topkill was observed on one-fourth of live shore pine (>70 percent of trees with bole galls). When topkill occurred, 40 percent of trees developed new leaders; however, some trees showed one or more iterations of a cycle of (1) topkill, (2) new leader development (40 percent of topkilled trees), (3) subsequent bole infection (37 percent of trees with new leaders), and (4) new topkill (7 percent of trees with new leaders). WGR-associated topkill averaged just 4.4 percent for trees without bole galls, compared to 25, 35, and 48 percent for pines with bole galls in one, two, and three crown-thirds, respectively. Similarly, mean crown dieback associated with WGR was correlated with WGR rating, ranging

from 5 to 43 percent for WGR severity ratings 1 and 6 (fig. 14.2). WGR-associated crown dieback increased sharply when the WGR ratings exceeded 2. The presence of bole galls and the number of crown-thirds affected by bole galls were both significant predictors of crown dieback (analysis of variance $p < 0.001$).

Wound Incidence, Type, and Severity

Wound types recorded included mechanical injury, root exposure, porcupine feeding, antler rub, bole cankers, burls, old dead bole galls, frost cracks, bear scratch, bark rubbing from neighboring trees, sapsucker feeding, and limb or bole harvest for Christmas trees. Wounds were observed on 47 percent of live shore pine; 26 percent had moderate- to high-severity

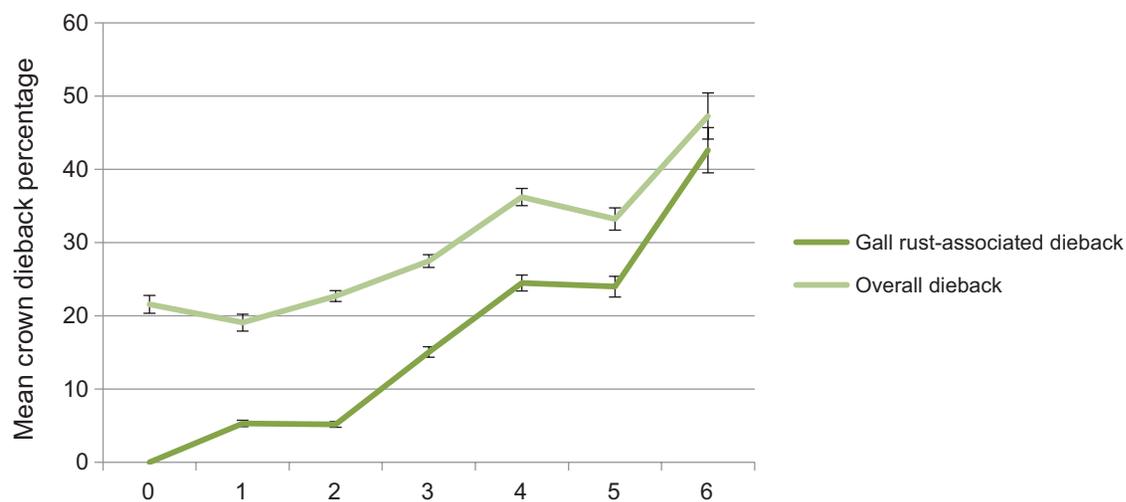


Figure 14.2—Mean crown dieback (percent) and mean crown dieback associated with western gall rust (WGR) (percent) by WGR severity rating (0 to 6) in 46 shore pine plots in southeast Alaska, 2013. Standard error bars are shown. WGR severity rating adapted from Hawksworth (1977).

wounds, and 6 percent had more than one wound type. Wound incidence was lower for dead shore pine (31 percent) than live shore pine, but the incidence of moderate- to high-severity wounds was the same. Wound severity and the proportion of live shore pine wounded increased with diameter (fig. 14.3). Shore pine had the greatest incidence of wounding among live trees, followed by western hemlock and mountain hemlock species (table 14.3).

Bole wounds were significantly more common on live shore pine (32 percent of live trees) compared to associated species (2 to 8 percent). Snow loading and animal feeding or marking are likely major sources of bole wounds, but specific causes were usually unknown. A bole canker

pathogen may create diamond-shaped wounds observed on all size classes of pine, sometimes in great abundance. These cankerlike wounds were recorded as general mechanical damage in 2012. In 2013, cankers were recorded on 22 percent of live shore pine on Prince of Wales Island and 19 percent on northeast Chichagof Island, compared to 5 to 8 percent at all other locations. Moderate to high severity ratings were assigned to all canker wounds on dead trees and 74 percent of cankers on live trees.

Poor root anchorage (root exposure) in saturated soils, mossy mounds, or standing water affected 5 to 32 percent of trees (by species) and was the most common wound for nonpines. Root exposure may only harm trees when severe, as

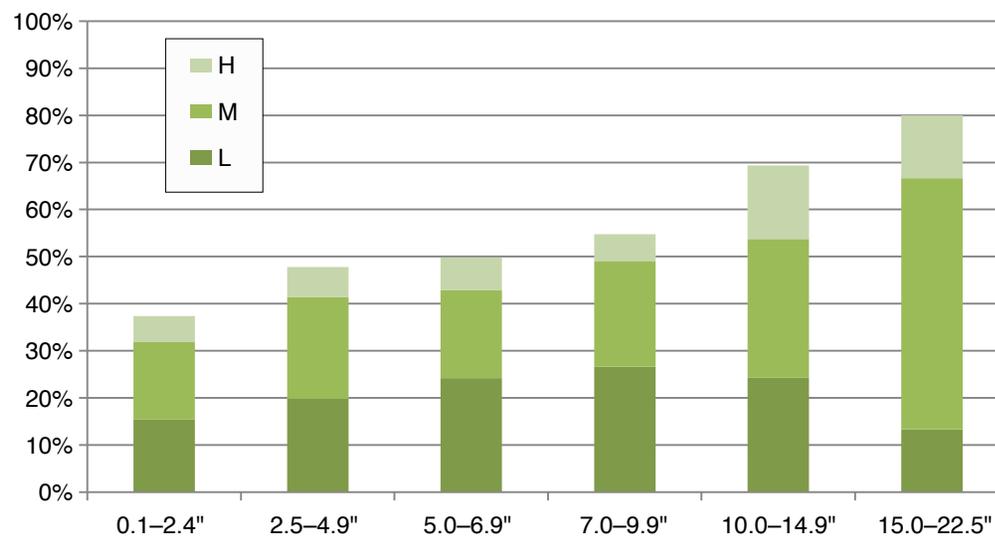


Figure 14.3—Percentage of live shore pine with low- (L), moderate- (M), and high- (H) severity wounds by diameter size class in 46 plots in southeast Alaska, 2013.

Table 14.3—Percentage of live and dead trees with bole wounds, exposed root wounds, and overall wounds by species in 46 plots in southeast Alaska, 2013

Dead/live status	Species	Trees with exposed root wounds	Trees with bole wounds	Overall trees wounded
		-----percent-----		
Dead	Shore pine	14	20	31
Dead	Yellow-cedar	9	0	9
Dead	Mountain hemlock	25	9	31
Dead	Western hemlock	40	0	40
Dead	Sitka spruce	0	0	0
Dead	Unknown	7	0	8
Live	Shore pine	17	32	47
Live	Yellow-cedar	5	2	7
Live	Mountain hemlock	23	3	26
Live	Western hemlock	32	3	36
Live	Sitka spruce	30	8	3
Live	Western redcedar	12	7	18

Note: Some trees had both wound types, and some uncommon wound types did not fall under these wound categories.

was the case in 1 percent of shore pine compared to 7 to 8 percent of spruce and hemlocks.

Foliar Damage and Retention

Foliage disease or leaf feeding insects caused low- to moderate-severity damage to 38 percent of shore pine, while <1 percent had severe foliar damage. Collected symptomatic foliage overwintered in mesh bags most often yielded fruiting bodies of *Dothistroma septosporum*. The foliar pathogens *Lophodermium seditiosum* and *Lophodermella concolor* were usually limited to

scattered individual needles or the previous year's shoots. Feeding damage of the lodgepole needle miner (*Coleotechnites milleri*) and defoliating weevils (*Magdalis* sp. or *Scythropus* sp.) was occasionally observed; these tentative identifications are based on the appearance of the feeding damage. Sawflies were noted in 13 of 46 plots and at all study locations except for Juneau. Reared adults were identified as lodgepole pine sawfly (*Neodiprion nanulus contortae*). Sawfly defoliation was usually restricted to a few branches, but some small trees were heavily

defoliated. On average, shore pine retained 3.3 years (standard deviation: 1.2 years) and 3.2 inches (standard deviation: 1.6 inches) of foliated branch tissue. Years of foliage retention and foliate branch length decreased as foliar damage severity rating increased (data not shown). Juneau was the location with the lowest mean needle retention (2.6 years), while Mitkof Island was the location with the highest (3.6 years), consistent with more moderate- and high-severity foliar damage in Juneau (14 percent) compared to other locations (9 to 11 percent).

Conks and Bark Beetles

Porodaedalea pini (formerly *Phellinus pini*) was the only heart rot decay fungus observed on live shore pine in study plots. *P. pini* conks were detected on 14 live trees and 6 snags. *Laetiporus sulphureus* was observed on a shore pine snag outside of study plots on Douglas Island (Juneau).

Secondary and tertiary bark beetles and galleries were observed on some large dying and recently killed pines. Detected bark beetles included *Pseudips mexicanus*, *Dryocoetes* sp., *Hylurgops porosus*, and the ambrosia beetle *Trypodendron lineatum*. *Pseudips mexicanus* beetles and galleries were most common and often occurred with a blue-staining fungus identified as *Leptographium wingfieldii* or a closely related species. Beetle galleries and staining were also noted on larger size-class shore pine snags. Bark beetle activity may be undercounted because galleries of secondary/tertiary insects (e.g.,

flathead borers) were common on snags and may have obscured evidence of earlier beetle activity.

DISCUSSION

Western gall rust, bole wounds, and Dothistroma needle blight were the most common forms of damage to shore pine detected in this study. All encountered biotic damage agents are presumed to be native, although some new State records were found (e.g., pine sawfly) and more work is needed to verify the causes of some forms of damage (e.g., bole cankers and other bole wounds).

Size and decay class information from snags showed that the largest diameter class of pine had the highest proportion dead (10 of 25 trees) and that much of this mortality was relatively recent (decay class 2). This pulse of mortality mirrors the loss of shore pine detected through the FIA network but is based on a small sample size. Secondary bark beetles and galleries (*Pseudips mexicanus* and *Dendroctonus murryanae*) and black fungal staining were detected on large, actively dying shore pine in our plots, and similar staining was observed on shore pine snags up to decay class 3. Shore pine occurs in a challenging environment and has a high incidence of damage; trees may succumb to injury and environmental stresses directly over time, and large weakened trees may attract bark beetles. Continued monitoring of this plot network will provide more concrete information on what conditions and damage agents are associated with mortality of trees that are currently alive.

WGR was present in all subplots and affected most trees, with variable incidence and severity. The coastal rainforest provides consistently conducive infection conditions. WGR severity was correlated with crown dieback, especially when bole galls were present, because bole galls were frequently associated with topkill. The high incidence of bole gall topkill in the upper crown meant that affected trees often survived, albeit with compromised form, reduced photosynthetic capacity from crown dieback, and a tendency for repeated injury from bole galls on new leaders. Secondary fungi (*Nectria cinnabarina*), caterpillars (Lepidoptera: Pyralidae: *Dioryctria*), and twig beetles (Coleoptera: Scolytidae: *Pityophthorus*) were detected in galls from recently killed branches.

Bole wounds were present on about one-third of live shore pine, with overall wound and bole wound incidence and severity increasing progressively with tree diameter. More than half of the trees over 10 inches d.b.h. had bole wounds, and nearly 40 percent had bole wounds of moderate to high severity. The causes of bole wounding, especially older wounds, were difficult to distinguish. More work is needed to understand the causes of bole wounding and to determine whether a fungal pathogen is responsible for the diamond-shaped cankerlike wounds observed on many shore pine. Large bole wounds can girdle and kill trees directly or increase their vulnerability to other agents.

Mild, wet summers in the coastal rainforest favor spread and infection by *Dothistroma septosporum*. Dothistroma needle blight probably

limits needle retention and negatively affects growth of shore pine across the study area but was not causing mortality in study plots. Foliage disease severity varied by location and was negatively correlated with foliage retention. A localized Dothistroma outbreak observed outside the plot network near Glacier Bay caused shore pine mortality following three consecutive years of severe foliage disease. Shore pine's low needle retention in southeast Alaska may make it particularly sensitive to successive years of foliar damage. Unprecedented damage and mortality from Dothistroma needle blight has occurred in managed lodgepole pine stands of British Columbia in conjunction with recent increases in summer precipitation, demonstrating that moderate changes in local climate can significantly affect severity of native foliage diseases (Woods and others 2005). Damage from pine sawfly had not been previously noted in Alaska. The broad detection of pine sawfly across the study area, its distribution in neighboring Canadian provinces (Ciesla 1976), and its recorded occurrence on shore pine suggest that this insect is native to southeast Alaska.

Porodaedalea pini (formerly *Phellinus pini*) was the only heart rot decay fungus observed fruiting on live shore pine. Conks were uncommon, but larger size-class shore pine often had sapwood decay associated with bole wounds, bole snap at old bole galls or frost cracks, or bole swelling. Bole injuries and WGR topkill create infection points for stem decay fungi. Increment coring the largest diameter pine in each subplot often revealed otherwise undetected heart rot.

Shore pine snags, in particular, frequently had animal holes associated with decayed cavities. In light of the high incidence of other damage agents (girdling bole wounds and bole galls of WGR), stem decays appear to be relatively less important disturbance agents in shore pine stands compared to other old-growth forest types in Alaska but probably create valuable wildlife habitat for cavity nesters.

Root exposure was commonly recorded among all species and included situations in which trees were rooted in water in addition to situations in which trees were rooted in mossy mounds with apparently compromised root anchorage. Root exposure may not stress or damage trees unless severe, and <1 percent of shore pine had high-severity root exposure, usually in standing water. Sitka spruce, western hemlock, and mountain hemlock frequently grew on the relatively drier mossy mounds, microsites that these species appeared to outgrow over time. Sitka spruce was often stunted and chlorotic, an indication of nutrient deficiency or water stress. Hemlocks attained larger size and regenerated more prolifically in mixed-conifer forest on the periphery of shore pine-dominated muskegs. In contrast, shore pine more often regenerated on moderately wet microsites with high light availability and limited competition from other tree species. Yellow-cedar allocates more roots near the soil surface in saturated, nutrient-poor soils of forested wetlands, increasing its vulnerability to decline freezing injury (D'Amore and others 2009, Hennon and

others 2012). Crown dieback and discoloration symptoms of yellow-cedar decline were common. Together, these observations indicate that associated conifers are not well suited to fill the niche that shore pine occupies in forested wetlands.

CONCLUSIONS

Despite the high incidence of injury to shore pine observed in this plot network, the species appears to be regenerating well in the peatland bogs and fens of southeast Alaska, as evidenced by the large number of small and medium trees in sample plots. Shore pine is well adapted to take advantage of the high light environment of forested wetlands, tolerating acidic, saturated soils better than associated conifers. We hypothesize that accumulated stress, particularly from bole wounding and bole galls of WGR, may kill trees directly or attract secondary bark beetles to weakened, larger trees. Weather conditions that support population increases of fungi or insects that attack WGR galls, cause foliar damage, or cause bole cankers may make these agents relatively more important to shore pine health in some years than others. Remeasurement of the 46-plot network every 5 years will further increase our knowledge of the causes and incidence of damage and mortality of shore pine and will help to determine whether the loss of larger diameter shore pine detected through FIA and our plot network continues and warrants concern.

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INTRODUCTION

About 10 million ha of forests are classified as having moderate to high fire hazards in the Western United States (Stephens and Ruth 2005), many of which are characterized by ponderosa pine (*Pinus ponderosa*), an integral component of three forest cover types and a major component of >65 percent of all forests in the Western United States (Burns and Honkala 1990). Prior to Euro-American settlement, many ponderosa pine forests were open and parklike, as frequent thinning of small-diameter [<19 cm diameter at breast height (d.b.h.)] and fire-intolerant trees by low-intensity surface fires and competitive exclusion of tree seedlings by understory grasses maintained such conditions (Covington and Moore 1994). Today, these forests tend to be denser, have more small trees and fewer large trees, and are dominated by more shade-tolerant and fire-intolerant tree species such as white fir (*Abies concolor*). Consequently, fuel-reduction and forest-restoration treatments have been widely promoted to reduce the intensity and severity of future wildfires and to increase resilience to a multitude of other disturbances. When properly applied, prescribed fire, mechanical thinning, and their combination are effective for increasing residual vegetative resilience to wildfire (Agee and Skinner 2005, Stephens and others 2012). For example, Ritchie and others (2007) studied the effects of fuel-reduction and forest-restoration treatments at Blacks Mountain Experimental Forest, California on observed fire severity between treated and

untreated stands impacted by a wildfire. Tree survival was highest in areas that were both thinned and prescribed burned. Survival in thinned-only areas was significantly greater than in untreated areas but less than in areas that had received the combined treatment.

Following prescribed fire, tree mortality may be immediate due to consumption of living tissue and heating of critical plant tissues, or can be delayed, occurring over the course of several years as a result of fire injuries to the crown, bole, or roots. Levels of delayed tree mortality are difficult to predict, and depend on numerous factors including tree species, tree size, phenology, degree of fire-related injuries, initial and postfire levels of tree vigor, the postfire environment, and the frequency and severity of other predisposing, inciting, and contributing factors (Stephens and others 2012). The propensity for bark beetles to attack fire-injured trees has led to questions regarding how the amount and distribution of bark beetle-caused tree mortality may negatively impact efforts to restore fire-adapted forest ecosystems with prescribed fire. Furthermore, fuel-reduction and forest-restoration treatments have functionally different effects on the structure and composition of residual forests and their resiliency to bark beetles. In particular, factors such as stand density, species composition, host density, average tree diameter, and average stand age are consistently identified as primary attributes associated with the severity of bark beetle infestations (Fettig and others 2007).

CHAPTER 15.

Resiliency of Ponderosa Pine Forests to Bark Beetle Infestations Following Fuel-Reduction and Forest-Restoration Treatments

(Project WC-EM-F-11-02)

CHRISTOPHER J. FETTIG

In recent years, numerous studies have examined the effects of fuel-reduction and forest-restoration treatments on the amount, distribution, and causes of tree mortality (Stephens and others 2012). Most were conducted on small experimental plots (<4 ha) for a relatively short period of time (1 to 5 years). The primary objective of this project was to evaluate the resiliency of different stand structures, created by applications of fuel-reduction and forest-restoration treatments, to bark beetle infestations at large spatial scales representing reasonable management scenarios in ponderosa pine forests a decade following prescribed burns.

METHODS

The project was executed at two locations: (1) Goosenest Adaptive Management Area, Klamath National Forest (41°30' N., 121°52' W.; 1500–1780 m elevation), in association with the Fire and Fire Surrogate Study (McIver and others 2013); and (2) Blacks Mountain Experimental Forest, Lassen National Forest (40°40' N., 121°10' W.; 1700–2100 m elevation), in association with the Blacks Mountain Experimental Forest Ecological Research Team (Oliver 2000). Due to page limitations, this summary focuses on Blacks Mountain Experimental Forest and expands on earlier research conducted there that details results 1 to 2 years after prescribed burns (Fettig and others 2008) and 3 to 5 years after prescribed burns (Fettig and McKelvey 2010).

Study Location

Blacks Mountain Experimental Forest was established in 1934 as a research facility for the study of forest management in interior ponderosa pine. The climate is characterized by hot, dry summers and cold, moist winters (Oliver 2000). Prior to treatment, stands were dominated by two age cohorts consisting of 300- to 500-year-old ponderosa and Jeffrey pines (*Pinus jeffreyi*) and incense cedar (*Calocedrus decurrens*) with a dense understory and midstory of 50- to 100-year-old pines, white fir, and incense cedar. There were 32.4 ± 1.5 (mean \pm SEM, standard error of the mean) m^2/ha of basal area and 871 ± 58 trees/ha (Zhang and others 2008).

Treatments

Twelve experimental plots, 77 to 144 ha (mean = 111 ha), were established to create two distinct forest structural types: mid-seral stage (low structural diversity, LoD) and late-seral stage (high structural diversity, HiD). Each structure was randomly assigned to two plots within each of three blocks. Blocking allowed for allocating variation to differences in tree species composition associated with elevational gradients and year of treatment (see Fettig and McKelvey 2014 for schedule). LoD was created by removing large overstory trees and small understory trees leaving only trees of intermediate size, while HiD was attained by thinning smaller trees and retaining larger trees (Oliver 2000). Following harvest, half of each plot was prescribed burned (B). Burns were implemented using

head, backing, and spot fires in fall. Hourly fuel moistures averaged 10.5 percent for 6- to 25-mm fuels (block 1) to 39.2 percent for duff (<20 mm) (blocks 1 and 2) (Fettig and others 2008). Following treatments, LoD and HiD averaged 10 and 25 m²/ha of basal area and 282 and 513 trees/ha, respectively (Zhang and others 2008).

Data Collection and Analyses

A 100-percent cruise (census) was conducted on each plot to locate dead and dying pines and firs 2 (Fettig and others 2008), 5 (Fettig and McKelvey 2010), and 10 years after prescribed burns. While both incense cedar and western juniper (*Juniperus occidentalis*) are minor components of Blacks Mountain Experiment Forest (<8 percent of trees), these species are rarely attacked and killed by bark beetles and were ignored. All recently killed pines and firs >19 cm d.b.h. were tallied and the causal agent of mortality was identified. Tree species; d.b.h. (later placed into five diameter classes: 19.0–29.2, 29.3–39.3, 39.4–49.5, 49.6–59.7, and >59.7 cm); crown color; colonizing bark beetle species; presence of wood borers; and ranking of fire severity (1 to 4, based on external measures of bole char and bark consumption) were recorded (Fettig and others 2008). A section of bark ~625 cm² was removed on each recently killed tree with a hatchet at ~2 m in height on at least two aspects to determine if any bark beetle galleries were present in the phloem or cambium. The shape, distribution, and orientation of galleries were used to distinguish among bark beetle species. Bark removal also

served as a means of separating mortality tallied during each of the three sample periods. This summary focuses on the mean percentage of trees killed by all causes, and by all bark beetle species, across all tree species. Responses for individual bark beetles species and host tree species can be found in Fettig and McKelvey (2014).

The experimental design was a randomized complete block with split plots with three blocks, two treatments (HiD and LoD), and two replicates per treatment. Due to an imbalance in the number of plots (i.e., because of one plot being impacted by mixed-severity wildfire in September 2002, necessitating its removal from the study), the Satterthwaite approximation method was used to estimate the appropriate degrees of freedom. An analysis of variance was performed on each response variable at $\alpha = 0.05$. If a significant treatment effect was detected, Tukey's multiple comparison test (Tukey's HSD) was used for separation of treatment means.

RESULTS AND DISCUSSION

A total of 188,793 pines and firs were monitored for mortality during the 10-year period. Of these, 106,314 were ponderosa pine, 63,636 were white fir, and 18,843 were Jeffrey pine.

Overall Tree Mortality

A total of 16,473 trees (8.7 percent of all trees) died during the 10-year period, of which 42.1, 5.2, and 52.7 percent were ponderosa pine, Jeffrey pine, and white fir, respectively.

The highest levels of tree mortality were observed during the initial sample period (1 to 2 years) (Fettig and others 2008) followed by the second (3 to 5 years) (Fettig and McKelvey 2010) and third (6 to 10 years) sample periods. This was expected, as fire-susceptible trees are often directly killed by the immediate effects of prescribed fire. Overall, tree mortality was concentrated (10,320 trees) in the smallest diameter class (24.1 cm), while the 54.7-cm diameter class had the lowest levels of tree mortality (252 trees).

Higher levels of tree mortality occurred on LoD + B (18.8 percent) compared to HiD (5.7 percent) and LoD (4.6 percent) ($p = 0.017$; all trees). Higher levels of tree mortality were also observed on LoD + B in the two smallest diameter classes ($p < 0.02$; 24.1 and 34.3), but in the two largest diameter classes, higher levels were observed in HiD + B ($p < 0.05$; 54.7 and >59.7). Interestingly, no significant treatment effect was observed for the 44.5-cm diameter classes ($p = 0.90$) (fig. 15.1). Fettig and others (2010) examined the effects of prescribed fire season (spring, fall, and none) on levels of tree mortality in ponderosa and Jeffrey pine forests in the central Sierra Nevada, California, and also reported few significant treatment effects in the intermediate diameter classes.

Bark Beetle-caused Tree Mortality

Western pine beetle and mountain pine beetle (*Dendroctonus ponderosae*) were observed colonizing ponderosa pine, Jeffrey pine beetle (*Dendroctonus jeffreyi*) was observed colonizing

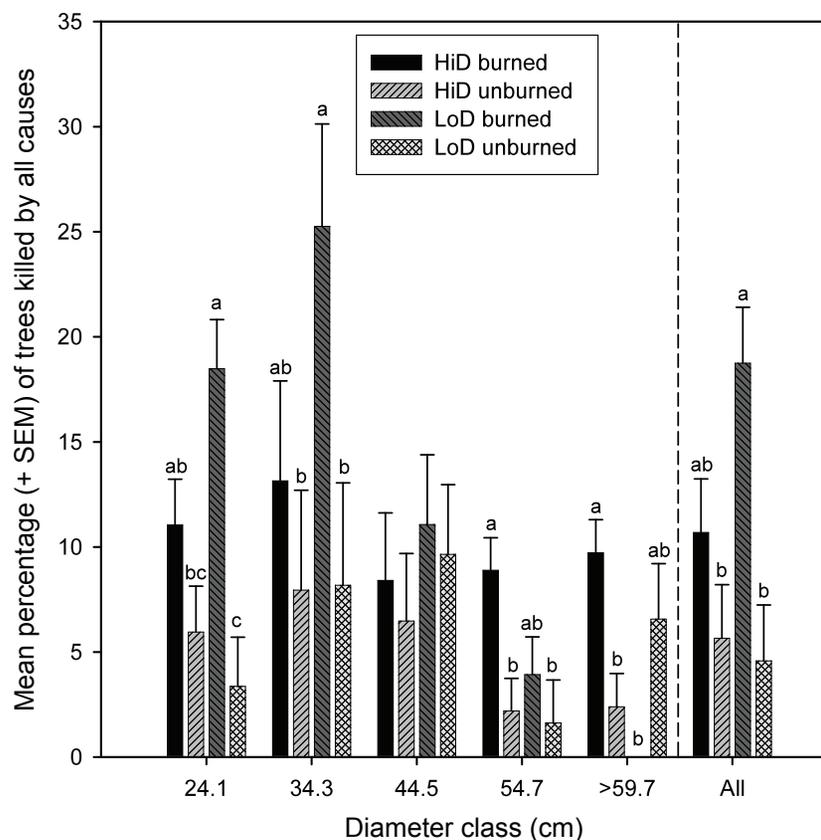


Figure 15.1—Mean percentage of trees killed by all sources by diameter class (midpoint of 10-cm diameter classes) and treatment (LoD = low structural diversity; HiD = high structural diversity) 10 years following prescribed burns, Blacks Mountain Experimental Forest, California. Means (+ SEM, standard error of the mean) followed by the same letter within groups are not significantly different (Tukey's HSD; $p > 0.05$) (from Fettig and McKelvey 2014).

Jeffrey pine, and fir engraver (*Scolytus ventralis*) was observed colonizing white fir. We also found pine engraver (*Ips pini*) and, to a much lesser extent, emarginate ips (*Ips emarginatus*) and *Ips latidens* (= *Orthotomicus*) colonizing ponderosa and Jeffrey pines. *Hylastes* spp., primarily *H. macer*; *Hylurgops* spp., primarily *H. subcostulatus*; and *Pseudohylesinus* spp. were occasionally observed but are not considered sources of tree mortality. Red turpentine beetle (*Dendroctonus valens*) was found colonizing many pines on burned-split plots (Fettig and others 2008), but their activity was largely limited to the 2 years following prescribed burns. Attacks by red turpentine beetle are usually confined to basal portions of stressed, weakened, or dead and dying pines, and are typically not considered a significant threat to tree health.

A total of 10,655 trees (5.6 percent of all trees) were killed by bark beetles (all bark beetle species combined) during the 10-year period (table 15.1). The highest levels of bark beetle-caused tree mortality were observed during the second sample period (4,193 trees), followed by the first (2,684 trees) and third (3,778) sample periods (table 15.2). Overall, bark beetle-caused tree mortality was concentrated (6,141 trees) in the smallest diameter class (24.1 cm), while the 54.7-cm diameter class had the lowest levels (178 trees) (table 15.1). Higher levels of bark beetle-caused tree mortality were observed on LoD + B than LoD in the 34.3-cm diameter class (15.4 and 7.3 percent, respectively) and for all trees (8.7 and 4.2 percent, respectively) ($p < 0.05$).

HiD + B (6.4 percent) exhibited higher levels of bark beetle-caused tree mortality than HiD (1.7 percent) in the 54.7-cm diameter class ($p < 0.05$). HiD + B (6.6 percent) also exhibited higher levels of bark beetle-caused tree mortality than LoD + B (0 percent) in the >59.7-cm diameter class ($p = 0.02$) (fig. 15.2). No other significant differences were observed. Approximately 60.5 percent of all bark beetle-caused tree mortality occurred on burned-split plots.

CONCLUSIONS

During the 10-year period, 8.7 percent of all trees died, most of which was attributed to bark beetles (64.7 percent), primarily fir engraver, mountain pine beetle, and western pine beetle. Bark beetles killed trees of all ages and size classes, but mortality was generally concentrated on HiD (64.3 percent), on burned-split plots (60.5 percent), within the two smallest diameter classes (87 percent), and 3 to 5 years after prescribed burns. These observations were consistent for all bark beetle species, with few exceptions (Fettig and McKelvey 2014). The observation concerning bark beetle-caused tree mortality concentrated 3 to 5 years after prescribed burns differs from many other similar studies that reported mortality was concentrated during the first year or two (Stephens and others 2012). However, many of those studies were of limited duration (<3 years). Relatedly, it is important to note that the treatment effects observed in our study varied by sample period (Fettig and McKelvey 2010, 2014; Fettig

Table 15.1—Numbers of trees killed by bark beetles 10 years following prescribed burns, Blacks Mountain Experimental Forest, California (from Fettig and McKelvey 2014)

Treatment ^a	d.b.h. class ^b	<i>Dendroctonus brevicomis</i>	<i>Dendroctonus ponderosae</i>	<i>Dendroctonus jeffreyi</i>	<i>Ips</i> spp.	<i>Scolytus ventralis</i>	Total
HiD + B	24.1	190	448	7	129	1,354	2,128
	34.3	198	227	11	28	537	1,001
	44.5	102	59	2	1	157	321
	54.7	66	10	2	2	27	107
	>59.7	250	28	2	1	14	295
	All	806	772	24	161	2,089	3,852
HiD	24.1	119	450	1	2	1,071	1,643
	34.3	89	175	1	0	626	891
	44.5	33	37	2	0	200	272
	54.7	27	10	0	0	29	66
	>59.7	100	13	0	0	17	130
	All	368	685	4	2	1,943	3,002
LoD + B	24.1	65	221	5	252	1,151	1,694
	34.3	86	104	3	60	549	802
	44.5	18	13	1	2	66	100
	54.7	0	4	0	0	—	4
	>59.7	—	—	0	—	0	0
	All	169	342	9	314	1,766	2,600
LoD	24.1	12	68	4	6	586	676
	34.3	27	30	1	1	381	440
	44.5	11	3	0	0	67	81
	54.7	1	0	0	0	—	1
	>59.7	0	2	0	0	1	3
	All	51	103	5	7	1,035	1,201
Total		1,394	1,902	42	484	6,833	10,655

— = No hosts present.

^a LoD, low structural diversity; HiD, high structural diversity. LoD was created by removing large overstory trees and small understorey trees leaving only trees of intermediate size, while HiD was attained by thinning smaller trees and retaining larger trees. Following harvest, half of each plot was prescribed burned (+ B).

^b d.b.h. = diameter at breast height in cm; value is the midpoint of the size class (except for the largest class).

Table 15.2—Trees killed by bark beetles by sample period and overall, Blacks Mountain Experimental Forest, California (from Fettig and McKelvey 2014)

Sample period	Interval (years)	<i>Dendroctonus brevicomis</i>	<i>Dendroctonus ponderosae</i>	<i>Dendroctonus jeffreyi</i>	<i>Ips</i> spp.	<i>Scolytus ventralis</i>	Total
1	1 to 2	442	468	18	456 ^a	1,300	2,684
2	3 to 5	747	947	17	15	2,467	4,193
3	6 to 10	205	487	7	13	3,066	3,778
Total		1,394	1,902	42	484	6,833	10,655

^a Fettig and others (2008) erroneously reported this value as 494.

and others 2008), further emphasizing the importance of long-term monitoring.

Oliver (1995) reported maximum stand density index (SDI) for even-aged ponderosa pine stands in northern California was regulated by bark beetle infestations. An SDI value of 230 defined a threshold for a zone of imminent bark beetle-caused tree mortality within which endemic populations kill a few trees but net growth is positive. Maximum SDI was defined at 365. While the SDI relationships described by Oliver (1995) are a tenuous fit for HiD, which does not represent an even-aged structure, it is quite appropriate for LoD. LoD plots averaged SDI values of 118 and 124 for the unburned and burned-split plots, respectively (Fettig and others 2008). These values are much less than the threshold SDI value of 230. As such, higher levels of bark beetle-caused tree mortality should be expected following similar treatments that retain higher residual stand densities, independent of the confounding effects of prescribed fire. No significant differences were observed between

HiD and LoD for any of the variables analyzed within any diameter class (data not shown), suggesting that over the 10-year period, these structures were of similar resilience to bark beetle infestations and other disturbances.

Concerns about maintaining large-diameter trees, particularly pines on HiD + B, have been expressed by members of the interdisciplinary team at Blacks Mountain Experimental Forest (Ritchie and others 2008). The majority (77.9 percent) of mortality in the largest diameter class occurred during the first 5 years following prescribed burns. During this time, significantly higher levels of tree mortality were observed on HiD + B (8.4 percent) compared to HiD (1.2 percent) (Fettig and McKelvey 2010); however, during the second 5 years, no significant difference was observed between these treatments. There are insecticide- and semiochemical-based tools available that could be selectively used to protect trees most susceptible to colonization by bark beetles (Fettig and Hilszczański 2014). Furthermore, methods such

as raking of litter and duff a short distance from the base of large-diameter ponderosa pines have been shown effective for reducing fire severity and subsequent levels of tree mortality when applied prior to burning (Fowler and others 2010). Such techniques might be considered in the future for reducing levels of tree mortality in the large-tree component following similar treatments.

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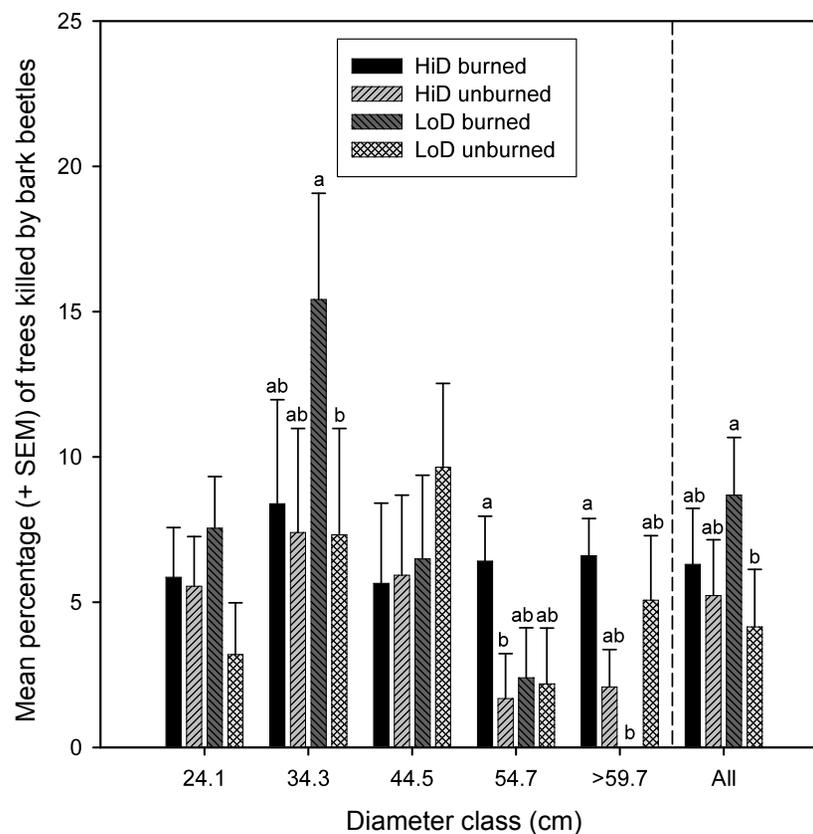


Figure 15.2—Mean percentage of trees killed by bark beetles (all species) by diameter class (midpoint of 10-cm diameter classes) and treatment (LoD = low structural diversity; HiD = high structural diversity) 10 years following prescribed burns, Blacks Mountain Experimental Forest, California. Means (+ SEM, standard error of the mean) followed by the same letter within groups are not significantly different (Tukey's HSD; $p > 0.05$) (from Fettig and McKelvey 2014).

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INTRODUCTION

Mountain pine beetle (*Dendroctonus ponderosae*) is a major disturbance in conifer forests of western North America, where it colonizes several tree species, perhaps most notably lodgepole pine (*Pinus contorta*). Recent outbreaks have been severe, long lasting, and well documented, with more than 27 million ha impacted (British Columbia Ministry of Forests 2012, USDA Forest Service 2012). Mountain pine beetle is an important part of the ecology of these forests, but extensive levels of tree mortality resulting from outbreaks may have undesirable impacts, e.g., negatively affecting aesthetics, recreation, fire risk and severity, human safety, timber production, and real estate values, among many other factors. In some areas, the magnitude of recent outbreaks has exceeded the range of historic variability, triggering concerns about short- and long-term impacts (Bentz and others 2009).

The primary objective of this ongoing project is to document the long-term impacts of mountain pine beetle outbreaks on residual stand structure and composition in lodgepole pine forests of the Intermountain West. Specifically, we concentrate on impacts to surface and aerial fuel loads; tree age, size, and species diversity; regeneration; invasive weeds; fall rates; and snag composition within the same monitoring sites over time. The scope of our work encompasses areas where the majority of tree mortality attributed to mountain pine beetle has occurred in the United States.

METHODS

In 2010, 25 circular 0.08-ha plots were established in each of 5 Western States near Fort Collins, CO, Stanley, ID, Butte, MT, Kamas, UT (near Evanston, WY), and Jackson, WY (125 total plots) (fig. 16.1) in lodgepole pine forests recently impacted by mountain pine beetle. Within each plot, all trees ≥ 7.6 cm diameter at breast height (d.b.h.) were tagged, and the species, d.b.h., total height, height to the base of the live crown, status (live or dead), causal agent of mortality (if applicable), and year of tree death [if applicable, based on Klutsch and others (2009)], among other variables, were recorded. Three 16.1-m Brown's transects (Brown 1974) were established at 0°, 120°, and 240° from plot center to estimate surface fuels. A 1-m² plot was established at the end of each Brown's transect to determine forest floor composition, and a 0.004-ha plot was established at plot center to estimate tree regeneration. Increment cores were collected from the three tallest trees on each plot to determine stand age and site productivity.

Levels of tree mortality and fall rates of trees have been recorded on an annual basis since 2010 (table 16.1). In 2012, methods were expanded to include census of each plot for the presence of exotic, invasive plants; monitoring of bark retention and checking (cracks) on trees, as these influence wood quality and salvage potential; and reconstruction of stand histories through use of dendrochronology (Colorado only). In August 2012, five plots in Idaho were burned in the Halstead fire at high severity and

CHAPTER 16.

The Impacts of Mountain Pine Beetle (*Dendroctonus ponderosae*) Outbreaks on Forest Conditions in the Intermountain West (Project INT-EM-F-10-03)

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Table 16.1—Year of sampling for variables collected across a network of 125 monitoring plots in lodgepole pine forests impacted by mountain pine beetle in 5 Western States

Variables measured	2010	2011	2012	2013	2014	2015	2016
Levels of tree mortality ^{a,b}	X	X	X	X	X	X	X
Fall rates of trees ^b	X	X	X	X	X	X	X
Tree species, d.b.h., and height ^b	X	—	—	—	X	—	—
Height to base of live crown ^b	X	—	—	—	X	—	—
Live crown cover	X	—	—	—	X	—	—
Checking of snags ^b	—	—	X	X	X	X	X
Bark retention of snags ^b	—	—	X	X	X	X	X
Tree regeneration	X	—	—	—	X	—	—
Forest floor composition	X	—	—	—	X	—	X
Invasive plants	—	—	X	—	X	—	X
Ladder fuels	X	—	—	—	X	—	—
Surface fuels	X	—	—	—	X	—	—
Litter and duff	X	—	—	—	X	—	—
Stand age	X	—	—	—	—	—	—

X = variable measured; — = variable not measured in this year.

^aFor trees killed prior to 2010, time of death was based on parameters adapted from Klutsch and others (2009), including: 1 year previous = crown of lime, yellow or yellow/red-colored needles; 2 years previous = ≥50 percent needles remaining; 3 years previous = <50 percent needles remaining; 4 years previous = no needles remaining but small and large twigs present; 5 years previous = only large twigs remaining; ≥6 years = both small and large twigs absent.

^bAll trees ≥7.6 cm diameter at breast height (d.b.h.).

were removed from the network that year. All variables are remeasured every fifth year. For purposes of this summary, we largely focus on causes, distributions, and levels of tree mortality that occurred between 2004 and 2011.

RESULTS AND DISCUSSION

Between 2004 and 2011, levels of tree mortality ranged from <1 percent (Utah, 2004)

to 37 percent (Utah, 2007) on an annual basis. By 2010, mortality declined to near preoutbreak (endemic) levels in all States except Colorado. In particular, a substantial loss of trees was observed in the larger diameter classes (>20 cm d.b.h.) (fig. 16.2), most of which was attributed to mountain pine beetle (fig. 16.3). This agrees with our basic understanding of mountain pine beetle outbreak dynamics. In endemic populations,

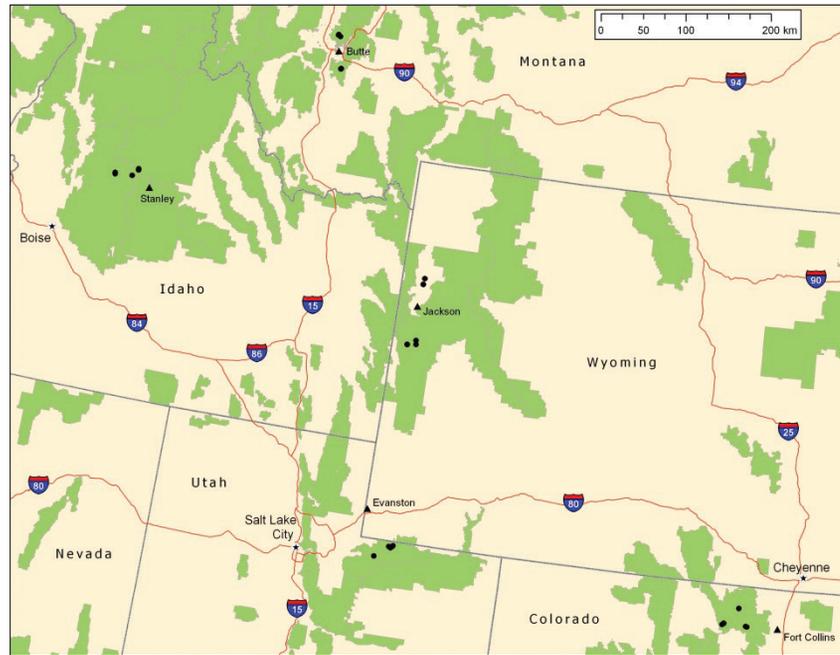


Figure 16.1—Evaluation monitoring plots (filled circles) installed to determine the impacts of mountain pine beetle outbreaks on lodgepole pine forests in the Intermountain West (25 per State, but many overlapped due to scale). Green represents National Forest System lands.

trees weakened by other agents are often first colonized and killed by mountain pine beetle (Boone and others 2011), but as an infestation develops, mountain pine beetle colonizes the largest trees (Shepherd 1966, Rasmussen 1972 for lodgepole pine), with progressively smaller trees being attacked over time as the proportion of uninfested larger trees declines. However, most of the mortality observed in the smallest diameter class (10 cm) was attributed to causes other than mountain pine beetle (fig. 16.3).

Large reductions in live pine volumes were observed between 2004 and 2011, ranging from 49 percent (Idaho) to 67 percent (Montana) (fig. 16.4). Ultimately, all of the woody biomass contained in these trees is transferred to the forest floor after tree fall. We expect these and other changes (Jenkins and others 2014) to have a significant effect on baseline fuel conditions (fig. 16.5) when remeasured in 2014. Mitchell and Preisler (1998) followed the fall rates of >600 lodgepole pines killed by mountain pine

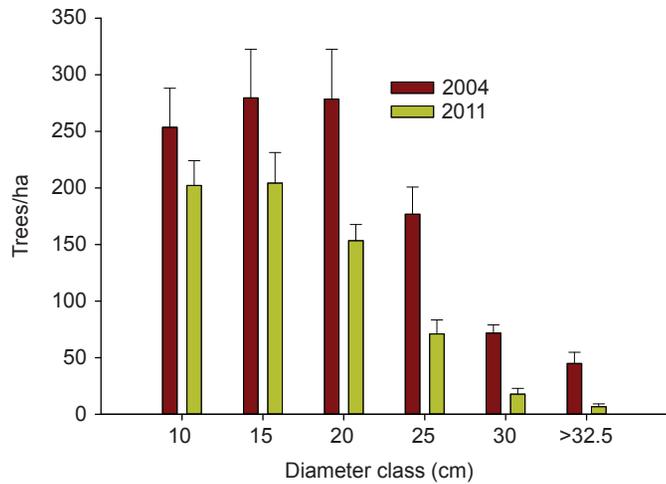


Figure 16.2—Mean (+ SEM, standard error of the mean) number of pines/ha by diameter class in 125 plots in the Intermountain West (midpoint of 5-cm diameter classes shown, except for largest).

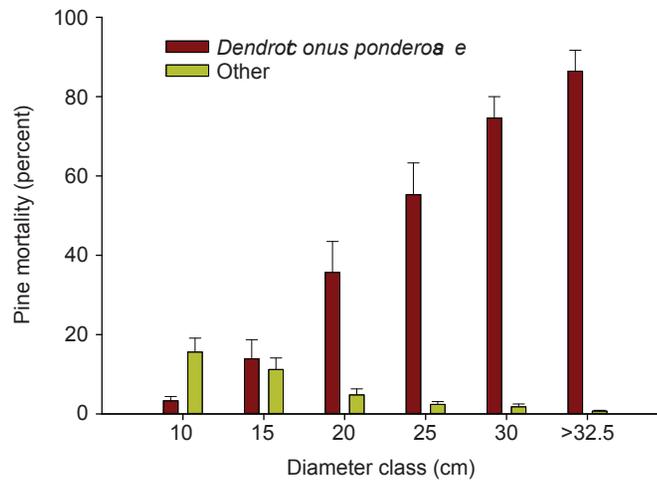


Figure 16.3—Mean (+ SEM, standard error of the mean) percentage of pines killed, by diameter class in 125 plots in the Intermountain West (midpoint of 5-cm diameter classes shown, except for largest).

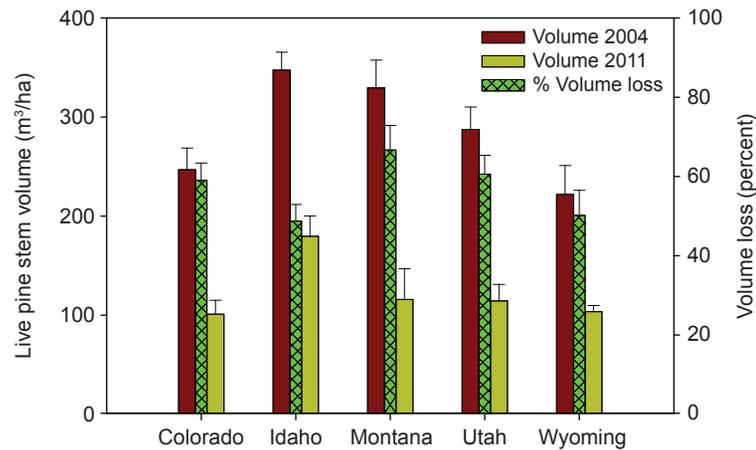


Figure 16.4—Mean (+ SEM, standard error of the mean) live pine stem volume and mean (+ SEM) percentage lost in 125 plots in the Intermountain West.

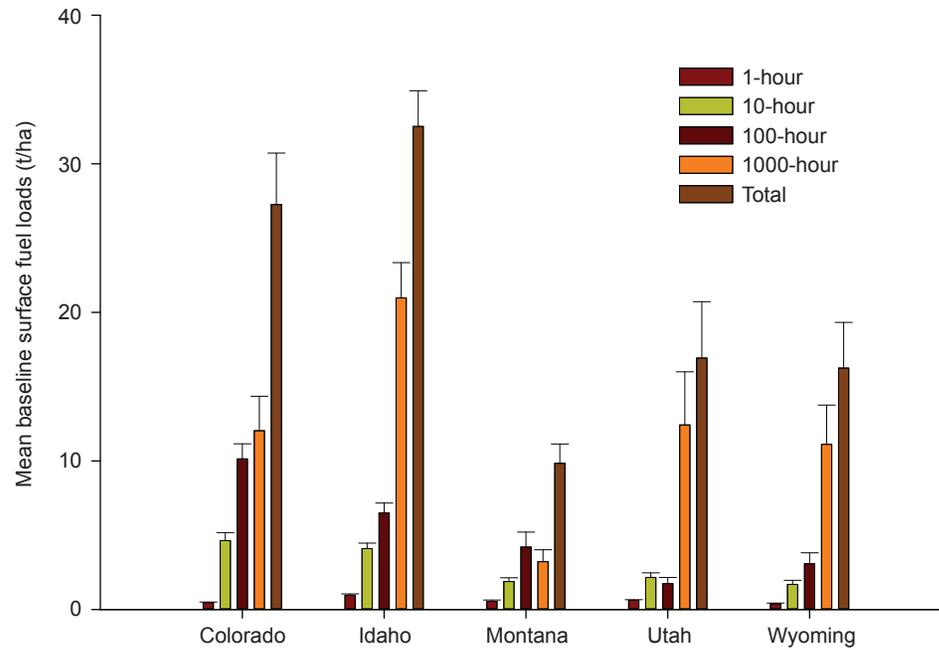


Figure 16.5—Mean (+ SEM, standard error of the mean) baseline surface fuel loads (t/ha) measured in 2010 by time lag categories in 125 plots in the Intermountain West. Fuels were to be remeasured across the network in 2014.

beetle in central Oregon and reported half-lives (i.e., the period of time it takes for half of the population to fall) of 8 years in thinned stands and 9 years in unthinned stands. These results are in contrast with Harvey (1986), who reported that only 25 percent of lodgepole pines killed by mountain pine beetle fell 11 years after death in northeastern Oregon. In our study, fall rates were negligible (<1 percent) for trees killed between 2004 and 2011, but have increased dramatically since (data not shown). Similarly, Lewis and

Thompson (2011) reported negligible (<1 percent) fall rates for trees dead 0 to 6 years and 28 percent for trees dead 6 to 10 years in British Columbia.

Checking was observed on most trees killed by mountain pine beetle. The development of checks is simply a function of changes in wood moisture content over time. As wood dries, it initially releases free water not bound in cells until it reaches the fiber saturation point. As

water is then drawn from the cell walls, they shrink, causing checks to develop. This can influence lumber recovery values and volumes if salvage is planned (Lewis and Thompson 2011).

To date, the invasion of exotic, invasive plants has been minimal. However, Lamb's quarter (*Chenopodium berlandieri*), Canada thistle (*Cirsium arvense*), and bull thistle (*Cirsium vulgare*) are increasing in Colorado plots that have experienced higher levels of tree mortality. Any activity that creates disturbance can promote plant invasions by increasing resource availability and/or decreasing plant competition. Accordingly, we expect to see higher levels of invasion in the future.

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INTRODUCTION

Research indicates that tree species are exhibiting changes in distribution and phenology in response to climate change (Root and others 2003, Woodall and others 2009, Zhu and others 2012). Climate change is expected to have large impacts on the area and location of suitable tree species habitat (Iverson and Prasad 1998, Iverson and others 2008, Schwartz and others 2001) and may pose a threat to the viability of forest tree species, many of which may be forced to either adapt to new conditions or shift to more favorable environments (Aitken and others 2008, Davis and others 2005). Managers and decisionmakers will need tools to assess the potential impacts of climate change on the broad diversity of forest tree species across North America and elsewhere.

Climate change is a priority area for Evaluation Monitoring projects funded by the Forest Health Monitoring (FHM) national program of the Forest Service, U.S. Department of Agriculture. Attendees at the 2008 FHM Working Group meeting approved a resolution calling for a baseline assessment across North American tree species of the risk of genetic degradation, local extirpation, or species-wide extinction associated with climate change.

Known as Forecasts of Climate-Associated Shifts in Tree Species (ForeCASTS), this assessment was conducted across all forest types and ownerships across the North American continent. The central focus of the assessment has been the statistical modeling of environmental niche envelopes that forecast species'

geographic ranges under climate change using the Multivariate Spatio-Temporal Clustering (MSTC) technique developed by Hargrove and Hoffman (2005). The resulting maps predict the future location and quality of habitat for tree species and, along with consideration of species' biological attributes, allow for predictions of the degree to which species are likely to be able to move to areas with the appropriate environmental conditions over time and avoid the loss of extensive genetic variation.

METHODS AND RESULTS

Combining aspects of traditional geographical information systems and statistical clustering techniques, MSTC employs nonhierarchical clustering to classify Geographic Information System (GIS) raster cells with similar environmental conditions into categories (Hargrove and Hoffman 2005). The MSTC process generates output maps that group and display each pixel as part of an "ecoregion" with other pixels possessing similar environmental conditions. Global in scope, MSTC incorporates 17 environmental variables and generates maps at a resolution of 4 km², the finest resolution at which global environmental data are consistently available. It is an appropriate tool for the assessment of the potential genetic effects of climate change on forest tree species because (1) it is able to rapidly identify potential changes in suitable habitat for a large number of species, (2) it allows for flexible occurrence data inputs, (3) it generates relatively high-resolution results applicable at the population level, (4) it has the ability to identify potential suitable habitat

CHAPTER 17.

Assessing Forest Tree Risk of Extinction and Genetic Degradation from Climate Change

(Project SO-EM-09-01)

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beyond the borders of the United States, and (5) it incorporates pertinent environmental variables associated with plant distributions (Potter and others 2010). Details of the technique are presented elsewhere (Hargrove and Hoffman 2005, Potter and Hargrove 2013, Potter and others 2010).

We used MSTC to predict the future location and quality of habitat for 337 forest tree species under four combinations of two general circulation models (GCMs), the Hadley model and Parallel Climate Model, and two emissions scenarios, A1FI and B1, each for years 2050 and 2100. These sets of maps were generated twice for each species, once with elevation as a spatial environmental characteristic and once without. Forest Inventory and Analysis (FIA) plot data (Woudenberg and others 2010) were used as occurrence location training data for most species. For rare species not well sampled by FIA, training data came from the Global Biodiversity Information Facility (2013). Using a grayscale ramp, these maps depict areas of decreasing environmental similarity to the environmental conditions currently present at the tree species training occurrence locations.

Quality of habitat is determined spatially for each species with two sets of maps, those of Minimum Required Movement (MRM), which quantify the distance from each pixel to the nearest *environmentally suitable* location in 2050 under the Hadley low-emissions scenario, and of Optimal Required Movement (ORM) distance, which quantify the distance from each 4-km²

pixel to the nearest *environmentally identical* location in 2050.

We used these MSTC mapped results to calculate, for each of the 337 tree species, several metrics of projected climate change pressure. Four of these were described in Potter and Hargrove (2013): (1) the degree to which the area of suitable environmental conditions is predicted to decrease or increase over time (percentage change in suitable area), (2) the amount of currently suitable area that is expected to remain suitable (range stability over time), (3) the distance that tree populations currently in areas expected to become unsuitable would have to travel to reach the nearest suitable location in the future (range shift pressure to any acceptable future habitat), and (4) the existing environmental variation across the range of a species (realized current niche occupancy). Other statistics include (5) the distance that tree populations currently in areas expected to become unsuitable would have to travel to reach the nearest identical location in the future (range shift pressure to any identical future habitat), and (6) the proportion of current habitat area with no future analogue predicted (proportion of no identical future habitat).

These maps and statistics are available, for all 337 North American tree species, at http://www.geobabble.org/~hnw/global/treeranges5/climate_change/atlas.html. Here, a page exists for each species, containing maps of (1) training occurrence locations, (2) locations with currently suitable environmental conditions, (3) locations expected to be suitable under the

four GCM/scenario combinations in 2050 and 2100 (fig. 17.1), (4) current expected range compared to future expected range under Hadley B1 (low-emissions scenario) in 2050 (fig. 17.2A), and (5) MRM and ORM under Hadley B1 2050 (fig. 17.2B). When they exist, links to corresponding climate change projections from other researchers using different techniques (Crookston 2013, Prasad and others 2013) are included. GIS files are also available for download.

DISCUSSION

A variety of threats, most importantly climate change (Parmesan 2006) and insect and disease infestation (Dukes and others 2009, Logan and others 2003), may increase the likelihood that forest tree species will experience population-level extirpation or species-level extinction during the next century. In the face of multiple threats and uncertainty, an important forest management goal will be to safeguard existing adaptive capacity within tree species and create conducive conditions for future evolution, with a focus on the conservation of variability in adaptive traits (Myking 2002).

Along with the consideration of important species life-history traits and of threats other than climate change (Aitken and others 2008, Myking 2002, Sjoström and Gross 2006), we expect that the ForeCASTS maps and climate change pressure metrics will be valuable for scientists and policymakers attempting to determine which forest tree species, in the face of climate change, should be targeted for

monitoring efforts and for *in situ* and *ex situ* conservation actions such as seed banking efforts, facilitated migration, and genetic diversity studies. For example, the ForeCASTS climate change pressure maps and statistics, along with consideration of species' biological attributes, can allow for the assessment of whether migrating species might be able to track appropriate environmental conditions over time and avoid the loss of extensive genetic variation.

A loss of important adaptive genetic variation may be of particular concern for species that have narrow habitat requirements, are located exclusively at high elevations, and/or are not able to disperse their propagules effectively across long distances. Even if not locally extirpated outright, populations of these and other species could experience significant inbreeding, genetic drift, and decreased genetic variation because of reduced population size. Such populations may then become more susceptible to mortality caused both by nonnative pests and pathogens and by the environmental pressures associated with climate change. This susceptibility could generate a cycle of mortality, loss of genetic variation, and inability to adapt to change, a cycle that could ultimately result in population extirpation (Potter and others 2010).

Measures of predicted climate change pressure may be particularly helpful in multiple-species assessments across broad regional scales that take into account climate change risk to many species. For example, an analysis of climate change pressure results across 172 North American tree species using the Hadley B1 GCM/

Eastern hemlock (*Tsuga canadensis*)

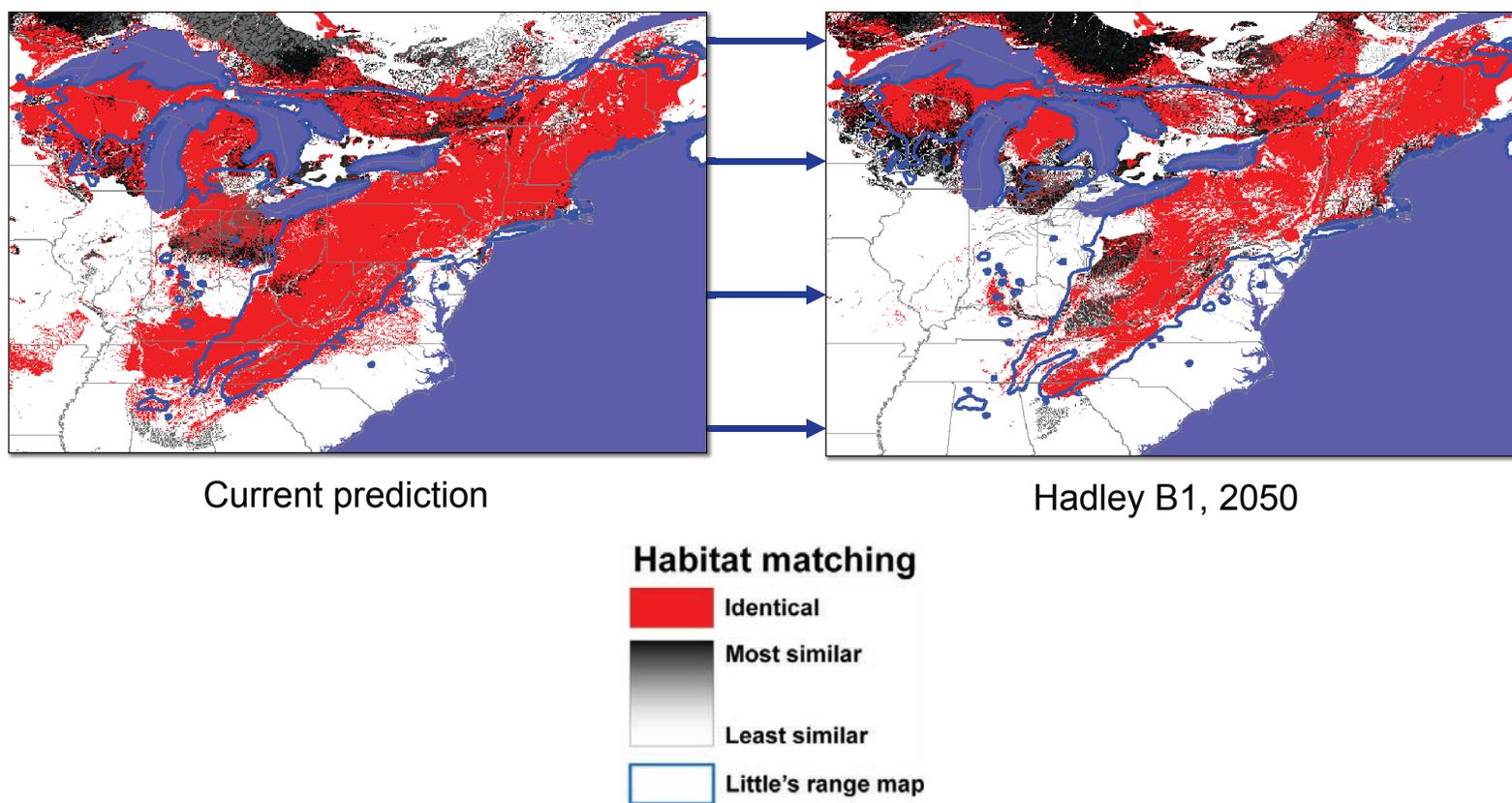


Figure 17.1—Predicted locations of eastern hemlock (*Tsuga canadensis*) current suitable habitat and future suitable habitat in 2050 under the Hadley general circulation model, B1 emissions scenario, using Multivariate Spatio-Temporal Clustering. Dark blue line indicates tree range as delineated by E.L. Little (U.S. Geological Survey 1999).

emissions scenario combination for 2050 found that all but two were projected to decline in suitable area. Eastern species were predicted to experience both a greater decline in suitable area and to maintain less range stability than Western species, although predicted range shift did not differ between the regions (Potter and Hargrove 2013). Additionally, Eastern species were more likely than Western species, on average, to be habitat generalists. In general, most species are expected to need to move a relatively short distance from newly unsuitable to the nearest future suitable locations. That study indicated that Great Basin bristlecone pine (*Pinus longaeva*) and September elm (*Ulmus serotina*) are species that need to be closely monitored, and may need to be considered as candidates for facilitated migration, because the distance from current suitable to future suitable habitat was predicted to be extensive for both.

Additionally, the ForeCASTS climate change pressure metrics are being used as inputs in Project CAPTURE (Conservation Assessment and Prioritization of Forest Trees Under Risk of Extirpation), a cooperative effort across the three Forest Service deputy areas to establish a framework for conservation priority-setting assessments of forest tree species across the entire United States.

Finally, the ForeCASTS species habitat maps can be overlaid to identify areas of high potential species richness and endemism, both in current time and in the future.

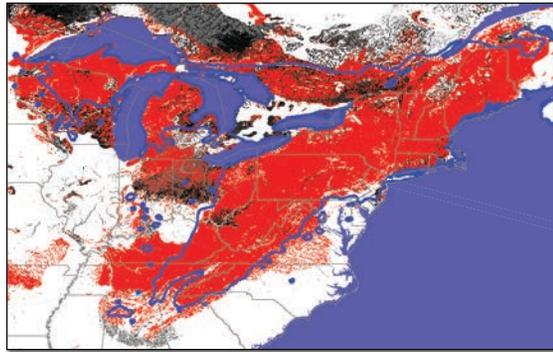
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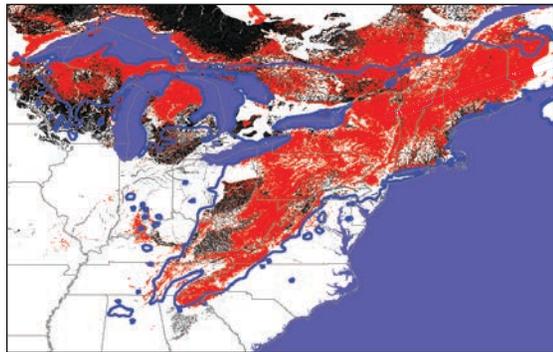
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(A)



Current prediction



Hadley B1, 2050

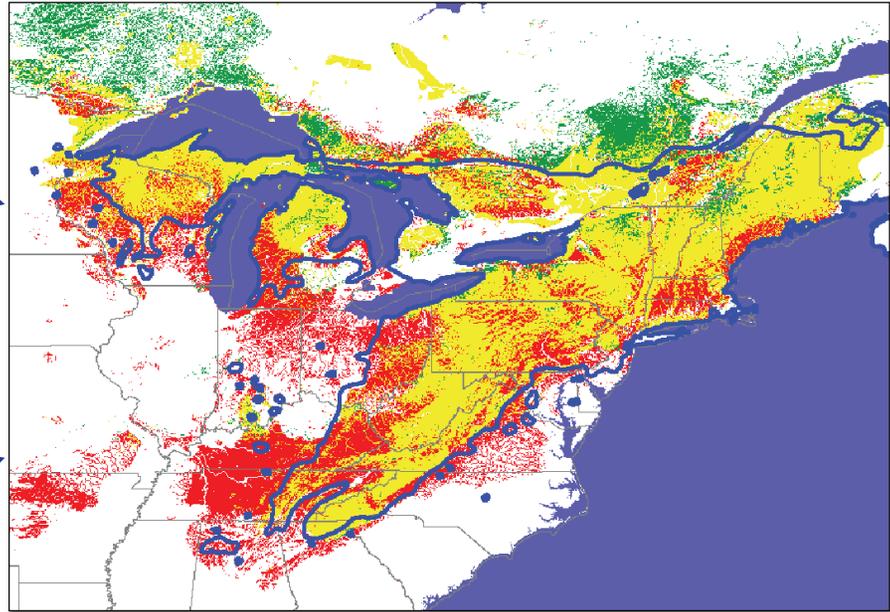


Figure 17.2—Predicted eastern hemlock (*Tsuga canadensis*) (A) environmental suitability comparison for current conditions and for 2050 under the Hadley general circulation model, B1 emissions scenario, and (B) minimum required movement (MRM) distance to nearest future suitable conditions in 2050 using Multivariate Spatio-Temporal Clustering. Maps from figure 17.1 shown for reference. Dark blue line indicates tree range as delineated by E.L. Little (U.S. Geological Survey 1999).

(B)

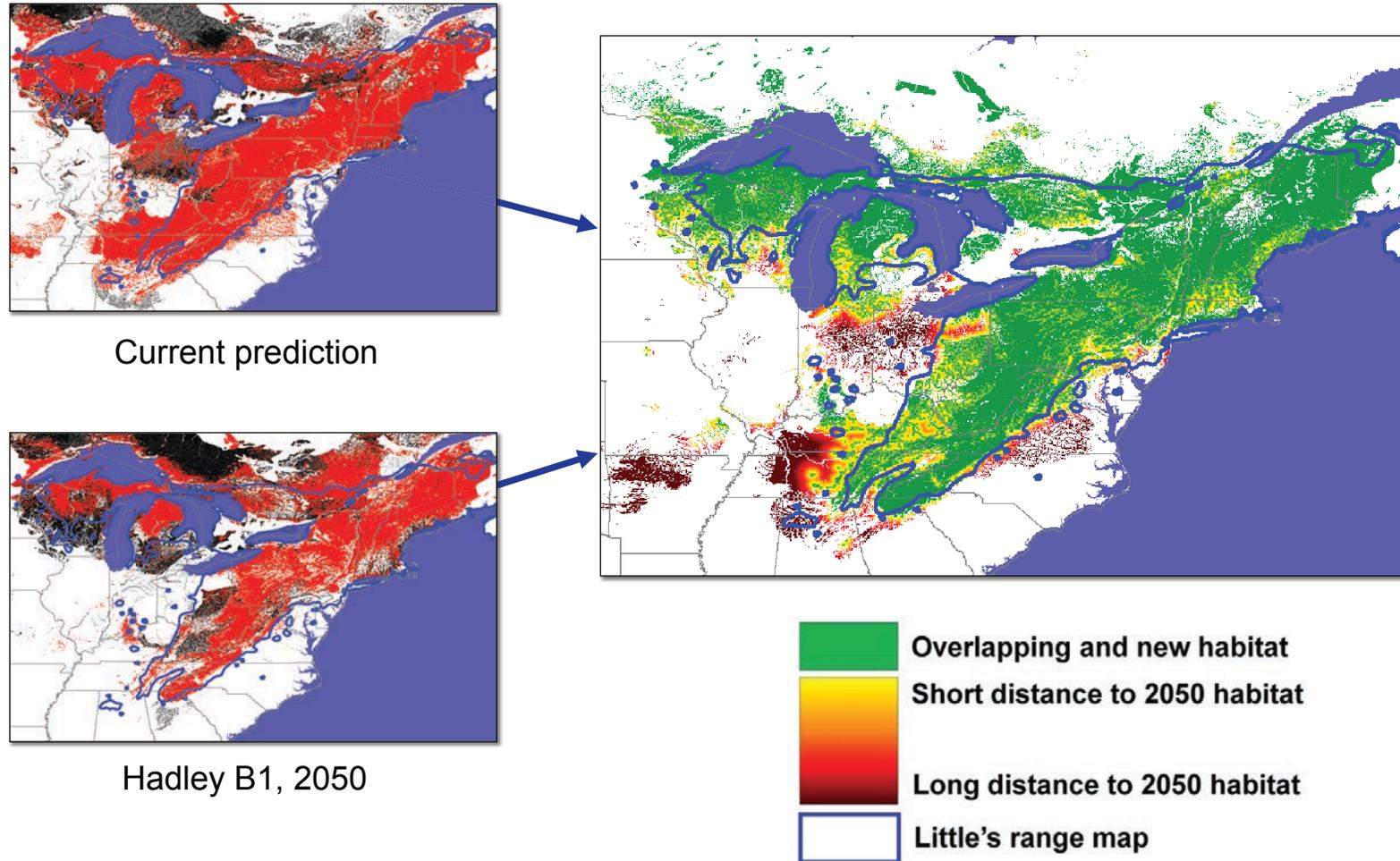


Figure 17.2(continued)—Predicted eastern hemlock (*Tsuga canadensis*) (A) environmental suitability comparison for current conditions and for 2050 under the Hadley general circulation model, B1 emissions scenario, and (B) minimum required movement (MRM) distance to nearest future suitable conditions in 2050 using Multivariate Spatio-Temporal Clustering. Maps from figure 17.1 shown for reference. Dark blue line indicates tree range as delineated by E.L. Little (U.S. Geological Survey 1999).

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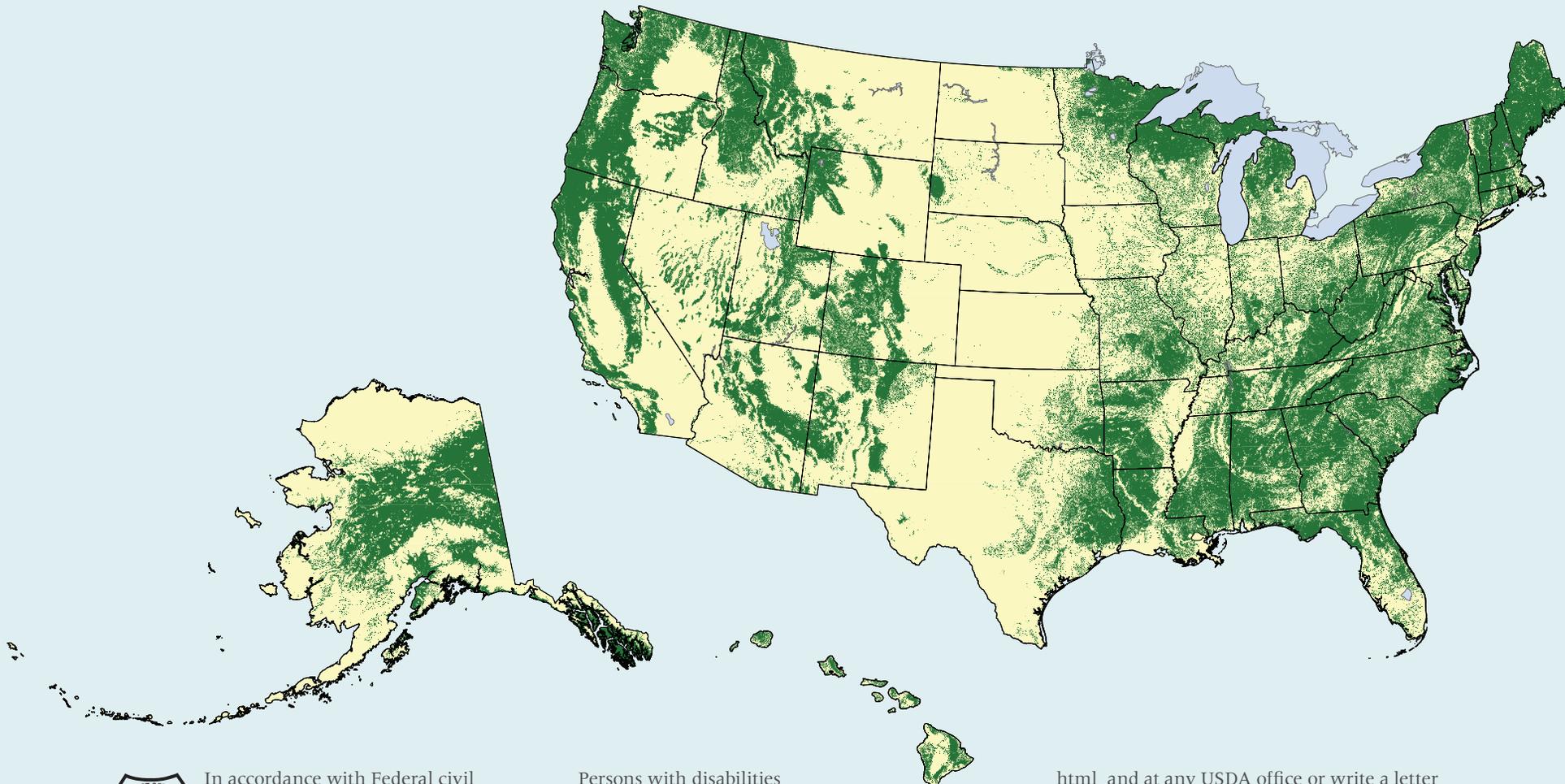
The annual national report of the Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture, presents forest health status and trends from a national or multi-State regional perspective using a variety of sources, introduces new techniques for analyzing forest health data, and summarizes results of recently completed Evaluation Monitoring projects funded through the FHM national program. In this 14th edition in a series of annual reports, survey data are used to identify geographic patterns of forest insect and disease activity. Satellite data are employed to detect geographic patterns of forest fire occurrence. Recent drought conditions are compared across the conterminous United States. Data collected by the Forest Inventory and Analysis (FIA) Program are employed to detect regional differences in tree mortality. Results of a national insect and disease forest risk assessment, including maps, are presented. Using FIA and national land cover data, decline of intact forest is assessed by forest type and ownership. Ten recently completed Evaluation Monitoring projects are summarized, addressing forest health concerns at smaller scales.

Keywords—Change detection, drought, fire, forest health, forest insects and disease, fragmentation, risk assessment, tree mortality.



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