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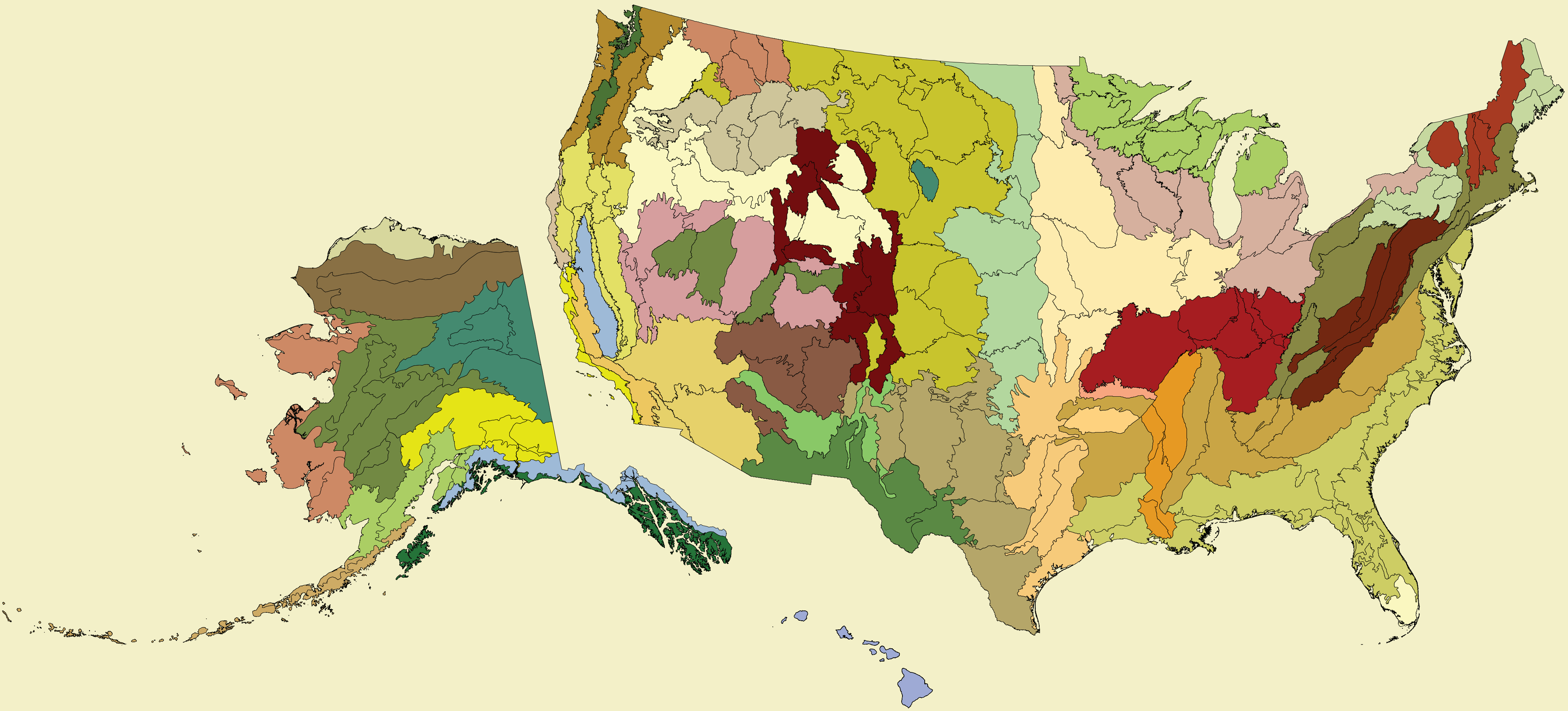
Forest Health Monitoring: National Status, Trends, and Analysis 2013

Editors Kevin M. Potter Barbara L. Conkling



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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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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 13th edition in a series of annual reports, survey data are used to identify geographic patterns of 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. A satellite-derived change detection system operating across the conterminous United States is described. A conceptual organization of existing and future technologies to support and improve forest health monitoring is presented. FIA data are used to produce a national map of invasive plant species infestation and to evaluate changes in crown conditions during the last decade. Five recently completed Evaluation Monitoring projects are summarized, addressing forest health concerns at smaller scales.

Keywords—Change detection, crown conditions, disturbance, drought, fire, forest health, forest insects and disease, invasive plants, 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). The Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture, with cooperating researchers within and outside the Forest Service and with State partners, quantifies status and trends in the health of U.S. forests (chapter 1). The analyses and results outlined in sections 1 and 2 of this FHM annual national report offer a snapshot of the current condition of U.S. forests from a national or multi-State regional perspective, incorporating baseline investigations of forest ecosystem health, examinations of change over time in forest health metrics, and assessments of developing threats to forest stability and sustainability. For datasets collected on an annual basis, analyses are presented from 2012 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 2013).

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 2012 by the Forest Health Protection Program of the Forest Service identified 82 different mortality-causing agents and complexes on 1.67 million ha in the conterminous United States, and 81 defoliating agents and complexes on approximately 3.64 million ha. Geographic hot spots of forest mortality were associated with mountain pine beetle in the West. Hot spots of defoliation were associated with western spruce budworm, aspen defoliation, pine butterfly, and larch needle cast in the West, and with fall cankerworm in the East. Mortality was recorded on a very small proportion of the surveyed area in Alaska, while aspen leafminer and willow leaf blotchminer were the most important identified agents of defoliation in that State.

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 2012, more satellite-detected forest fire occurrences were recorded for the conterminous United States than in any other year since the beginning of full-year data collection in 2001. Ecoregions in Idaho, Wyoming, Nebraska, Montana, and South Dakota experienced the most fires per 100 km^2 of forested area. Geographic hot spots of high fire occurrence density were detected throughout the Interior West. Ecoregions in the Interior West, Northwest, Great Lakes States, Northeast, and Middle Atlantic States experienced greater fire occurrence density than normal compared to the 11-year mean and accounting for variability over time. Alaska experienced low fire occurrence density in 2012.

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 2012 to map drought conditions across the conterminous United States at a fine scale (chapter 4). It was a very dry year relative to historical data. Most of the Central United States, including much of the Great Lakes and Southwest regions, experienced at least mild drought conditions. A large contiguous area of extreme drought extended from the northwestern portion of the Great Plains and into the eastern portion of the Central and Northern Rocky Mountains. Areas with a moisture surplus were limited to the Pacific Northwest and northern California, New England, and coastal areas of the Southeast. Longer-term moisture deficits existed in the Interior West, the South Central and Great Lakes States, and Florida.

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 across multiple measurement cycles from 37 States found that the highest ratios of annual mortality to gross growth occurred in ecoregion sections located in the northern Plains and the Southern Mississippi Alluvial Plain. 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. The exception was high ponderosa pine mortality in one ecoregion. In the Southern Mississippi Alluvial Plain, meanwhile, several hardwood species experienced high mortality.

National-scale satellite-based forest monitoring can provide uniform and timely insights into forest health. *ForWarn*, a satellite-derived change detection system operating across the conterminous United States, has been used since January 2010 to detect a wide array of environmental threats to forests (chapter 6). *ForWarn* disturbance detections rely on changes in the timing of vegetation “greenness,” as measured by the Normalized Difference Vegetation Index (NDVI) derived from Moderate Resolution Imaging Spectroradiometer (MODIS) satellite sensors. Of its four detection capabilities—occurrence, severity, progression and recovery—*ForWarn*'s ability to monitor and track forest recovery may be among the most significant for aiding forest management in the future. *ForWarn*'s multiple baselines and cross-seasonal product lines provide a rich context for understanding the duration of disturbance effects and the cumulative effects of management in the months to years that follow.

While effective as standalone applications, the value of individual forest health protection technologies can be dramatically improved if developed in concert with or as part of an organized system (chapter 7). A conceptual organization of existing and future technologies aims to support and improve the monitoring of forest health. Additionally, a strategic solution is needed to integrate monitoring assessments. As a one-stop shopping system, the Forest Health Protection Mapping and Reporting Portal (FMRP) combines inventory, real-time tracking, and reporting tools to allow for better planning and integration of separate technologies. As part of FMRP, the Forest Disturbance Monitor (FDM) is a Web-based data delivery system designed to enhance efforts to allocate resources and plan forest health surveys. Additionally, a new Insect and Disease Survey (IDS) strategy will prioritize operator safety, maximize the quality and value of aerial sketchmapping, and improve other data streams.

Long-term monitoring and assessment of invasive plant species on the forest landscape is necessary to managers and policymakers for the obligation and direction of funds and other resources (chapter 8). Given the importance of monitoring invasive plants, units in the FIA Program have implemented efforts to track invasive plants in their regions. Here, a national map of invasive species infestation is presented; this map may be used to identify potential hot spots of invasion and could serve as a baseline for future monitoring efforts. Nationwide, 39 percent of sampled forested subplots contained at least one invasive species. In general, excluding Hawaii, invasive species were more prevalent on forest subplots in the East than in the West. In Northern States, multiflora rose, reed canarygrass, garlic mustard, and Japanese honeysuckle were the most commonly detected. In Southern States, Japanese honeysuckle, Chinese/European privets, nonnative roses, and Chinese lespedeza were the most common.

Tree crown conditions are visually assessed by the FIA Program as an indicator of forest health. These assessments are useful because an individual tree's photosynthetic capacity is dependent upon the size and condition of its crown (chapter 9). In general, crown conditions across the United States were stable during the last decade. Though some changes in crown condition were observed, many statistically significant changes were relatively small and likely biologically unimportant. Notable exceptions to this were the declining crown conditions among the hardwoods, western hemlock, and true firs in the West Coast region. The 2010 crown density moving averages for lodgepole pine, ponderosa pine, and Jeffrey pine in the West Coast region were substantially lower than the average conditions observed between 1996 and 1999. The ash group has maintained a high mean level of crown dieback in the Northern U.S. since the late 1990s.

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

- Identify potentially beech bark disease-resistant trees and established permanent plots in four Mid-Atlantic States to monitor general health conditions of beech trees (chapter 10).
- Use large-scale forest inventory data to describe the quality of wildlife habitat in the forests of Maine, particularly of species that frequent mature forest (chapter 11).
- Provide a scientific basis for managing the white pine blister rust invasion in Arizona and New Mexico by extending previous research on southwestern white pine ecology and documenting the distribution and effects of the disease on it (chapter 12).
- Characterize forest attributes and fuel loads of riparian and upland forest stands in southern Rocky Mountain watersheds infested by mountain pine beetle (chapter 13).
- Identify the extent to which nonnative sawflies and alder canker contribute directly and synergistically to alder dieback in Alaska (chapter 14).

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.

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CHAPTER 1.
Introduction

Kevin M. Potter

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

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 U.S. Department of Agriculture, Forest Service, 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).

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. 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 (Kolb and others 1994, Raffa and others 2009, Edmonds and others 2011). 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 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 13th 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, forest disturbance, invasive plants, and crown conditions, 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).

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, presented in selected chapters in section 2, is to introduce new techniques for analyzing forest health data and new applications of established techniques. Examples in this report are chapter 6, which describes a satellite-derived change detection system operating across the conterminous United States; chapter 7, which outlines a conceptual organization of existing and future technologies to support and improve forest health monitoring; and chapter 8, which presents a national county-level map of invasive plant species infestation based on FIA subplot data.

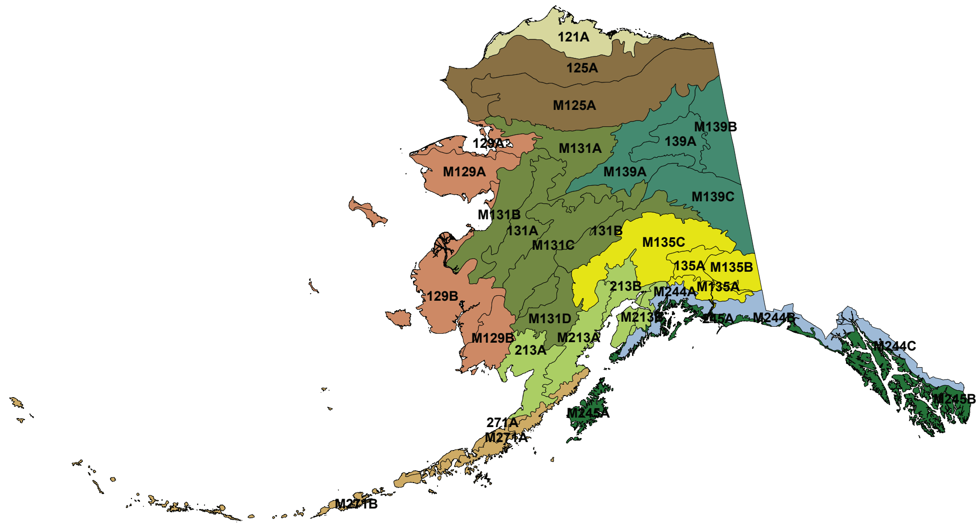
The third objective of the report is to present results of recently completed Evaluation Monitoring (EM) projects funded through the FHM national program. These project summaries, presented in section 3, determine the extent, severity, and/or cause of forest health problems (FHM 2012), generally at a finer scale than that addressed by the analyses in sections 1 and 2. Each of the five chapters in section 3 contains an overview of an EM project and key results.

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. In Bailey's hierarchical system, the two broadest ecoregion scales, domains and divisions, are based on large ecological climate zones, while each division is broken into provinces based on vegetation macro features (Bailey 1995). Provinces are further divided into sections, which may be thousands of square kilometers in extent and are expected to encompass regions similar in their geology, climate, soils, potential natural vegetation, and potential natural communities (Cleland and others 1997).

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 2012). 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



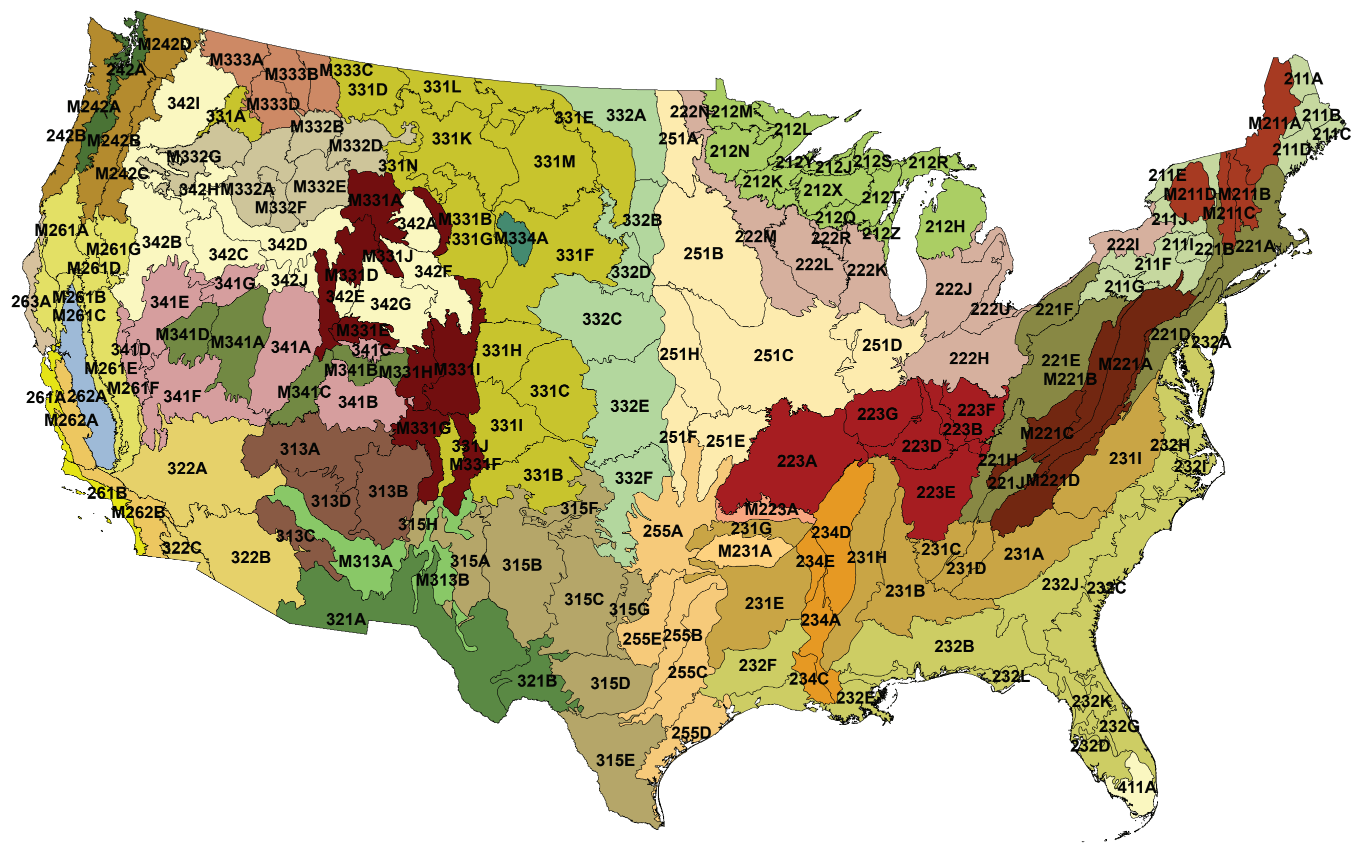


Figure 1.1—Ecoregion provinces and sections for the conterminous United States (Cleland and others 2007) and Alaska (Nowacki and Brock 1995). Ecoregion sections within each ecoregion province are shown in the same color.

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)

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

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

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 Steinman (2004) and Harris and others (2011).

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 (Cumming and others 2006, 2007, Lake and others 2006), health conditions in national forests (Morin and others 2006), crown conditions (Randolph and Moser 2009, Randolph 2010, Schomaker and others 2007), sampling and estimation procedures for vegetation diversity and structure (Schulz

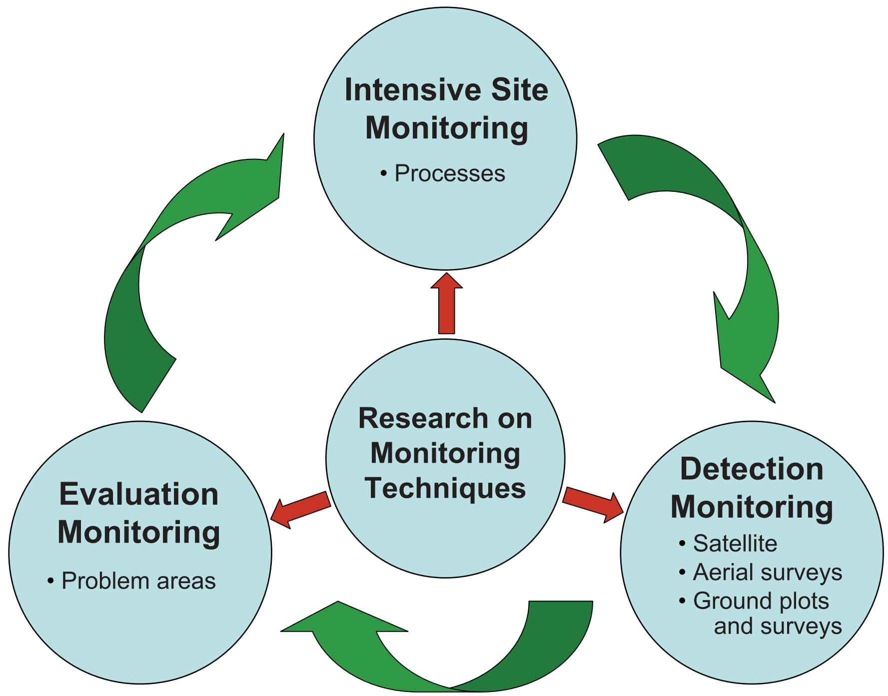


Figure 1.2—The design of the Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture (FHM 2003). A fifth component, Analysis and Reporting of Results, draws from the four FHM components shown here and provides information to help support land management policies and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

and others 2009), ozone monitoring (Rose and Coulston 2009), establishment of alien-invasive forest insect species (Koch and others 2011), spatial patterns of land cover (Riitters 2011), changes in forest biodiversity (Potter and Woodall 2012), and the overall forest health indicator program (Woodall and others 2010). For more information, visit the FHM Web site at www.fs.fed.us/foresthealth/fhm.

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 Detection Survey forest mortality and defoliation data for 2012 (FHM 2005), Moderate Resolution Imaging Spectroradiometer (MODIS) Active Fire Detections for the United States database for 2012 (USDA Forest Service 2013), and forest cover data developed from MODIS satellite imagery by the U.S. Forest Service Remote Sensing Applications Center. Other sources of data are: Parameter-elevation Regression on Independent Slopes (PRISM) climate mapping system data (Daly and Taylor 2000, PRISM Group 2004, PRISM Group 2013) and Normalized Difference Vegetation Index (NDVI) data derived from MODIS.

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 one 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, Pacific Southwest,

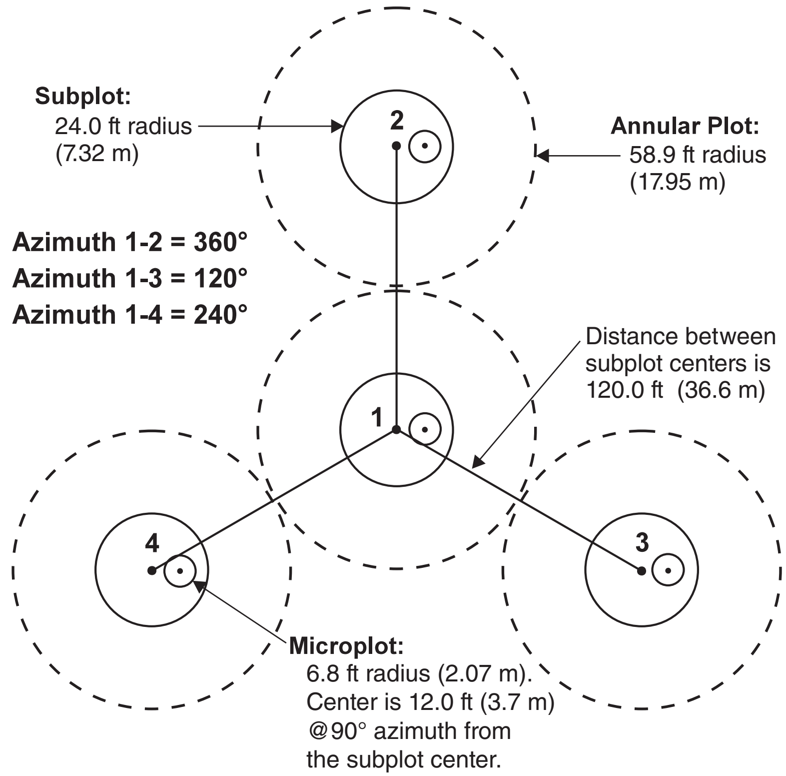


Figure 1.3—The Forest Inventory and Analysis Program mapped plot design. Subplot 1 is the center of the cluster with subplots 2, 3, and 4 located 120 feet away at azimuths of 360°, 120°, and 240°, respectively. (Woudenberg and others 2010)

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 m^2 or 1/24 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 m^2 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 originate at the center of each subplot (Woudenberg and others 2010).

FIA phase 3 plots represent a subset of these phase 2 plots, with one phase 3 plot for every 16 standard FIA phase 2 plots. In addition to traditional forest inventory measurements, data for a variety of important ecological indicators are collected from phase 3 plots, including tree crown condition, lichen communities, down woody material, soil condition, and vegetation structure and diversity, 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 2000[1] (Palmer and others 1991).

FHM REPORT PRODUCTION

This FHM national report, the 13th 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 Forest Restoration Act 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.

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SECTION 1.

Analyses of Short-Term Forest Health Data

CHAPTER 2.

Large-Scale Patterns of Insect and Disease Activity in the Conterminous United States and Alaska from the National Insect and Disease Survey, 2012

Kevin M. Potter
Jeanine L. Paschke

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 clusters 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 (FHM 2005) consists of information from low-altitude aerial survey and ground survey efforts. This database can be used to identify forest landscape-scale patterns associated with geographic hot spots of forest insect and disease activity in the conterminous United States, and to summarize insect and disease activity by ecoregion in Alaska (Potter 2012, 2013; Potter and Koch 2012; Potter and Paschke 2013, 2014). In 2012, IDS surveys covered approximately 142.61 million ha of the forested area in the conterminous United States (approximately 56 percent of the total), and 6.91 million ha of Alaska's forested area (approximately 13.4 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 and hemlock woolly adelgid), diseases (such as laurel wilt, Dutch elm disease, white pine blister rust, and thousand cankers disease), and mortality complexes (such as oak decline) are not easily detected or thoroughly quantified through aerial detection surveys. Such pests may attack hosts that are widely dispersed throughout 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 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 2012 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 and complexes for Alaska. Because of the insect and disease aerial sketchmapping process, all quantities are "footprint," or approximate, areas for each agent or complex, to delineate 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® 9.2 (ESRI 2006) 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 3,382 hexagonal cells, each approximately 2500 km^2 in area, generated in a 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 2500-km^2 hexagon size allows for analysis at a medium-scale resolution of approximately

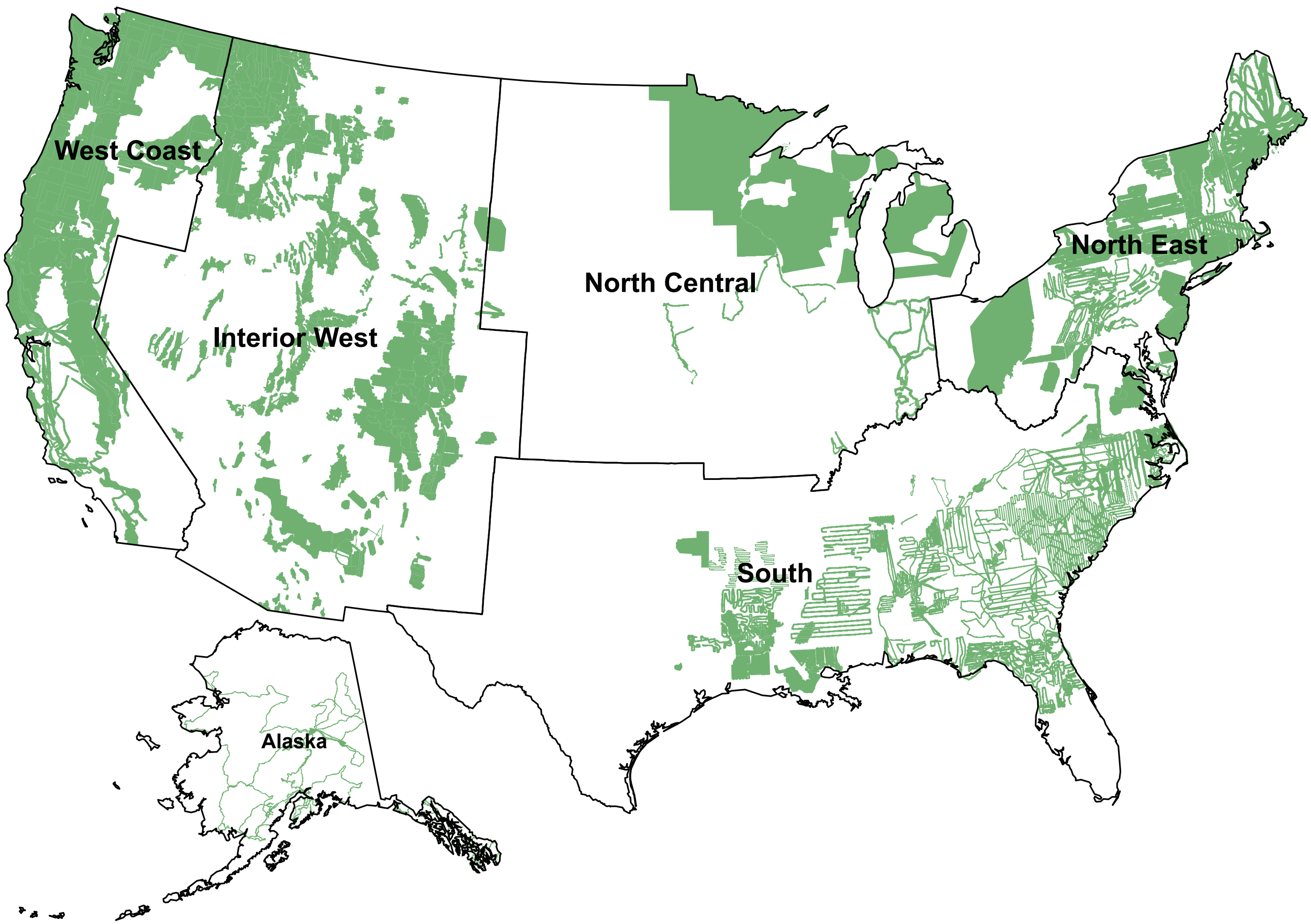

Figure 2.1—The extent of surveys for insect and disease activity conducted in the conterminous United States and Alaska in 2012. The black lines delineate Forest Health Monitoring regions. Note: Alaska is not shown to scale with map of the conterminous United States. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection)

the same area as a typical county. The variable used in the hot spot analysis was the percent 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 with a forest cover map (1-km^2 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 IDS-surveyed areas. Finally, the percent of surveyed forest exposed to mortality or defoliation agents and complexes 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 percent of surveyed forest exposed to mortality or defoliation agents and complexes 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 non-stationarities 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 United States. It is 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 standard deviations of the mean (Laffan 2006). In other words, a G_i* value of 1.96 indicates that the local mean of percent forest exposed to mortality-causing or defoliation-causing agents and complexes for a hexagon and its 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 6 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 2006).

The low density of survey data from Alaska in 2012 (fig. 2.1) precluded the use of Getis-Ord hot spot analyses for the State. Instead, mortality and defoliation data were summarized by ecoregion section (Nowacki and Brock 1995), calculated as the percent of the forest within the surveyed areas affected by agents and complexes of mortality or defoliation. (As with the mortality and defoliation data, the flown-area polygons were first dissolved to create an overall footprint.) 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

The IDS survey data identified 82 different mortality-causing agents and complexes on approximately 1.67 million ha across the conterminous United States in 2012, an area slightly smaller than that of Connecticut and Delaware combined. (Three of these mortality-cause categories were "rollups" of multiple agents.) By way of comparison, forests are estimated to cover approximately 304 million ha of the conterminous United States (Smith and others 2009).

Mountain pine beetle (*Dendroctonus ponderosae*) was the most widespread mortality agent, detected on 969 037 ha (table 2.1). This area has declined considerably in recent years, from 3.47 million ha in 2009 (Potter 2013) to 2.77 million ha in 2010 (Potter and Paschke 2013) and to 1.54 million ha in 2011 (Potter and Paschke 2014. Other mortality agents and complexes detected on more than 100 000 ha were spruce beetle (*Dendroctonus rufipennis*), five-needle pine decline, subalpine fir (*Abies lasiocarpa*) mortality complex, and western pine beetle (*Dendroctonus brevicomis*). Mortality from the western bark beetle group, encompassing 23 different agents and complexes in the IDS data (table 2.2), was detected on a total of more than 1.48 million ha in 2012. This represents a large majority of the total area on which mortality was recorded across the conterminous United States.

Table 2.1—Mortality agents and complexes affecting more than 5000 ha in the conterminous United States during 2012

Agents/complexes causing mortality, 2012	Area
	ha
Mountain pine beetle[a]	969 037
Spruce beetle	172 697
Five-needle pine decline[a]	130 050
Subalpine fir mortality complex[a]	114 834
Western pine beetle	101 999
Fir engraver	84 656
Douglas-fir beetle	65 540
Ips engraver beetles	39 397
Spruce budworm	32 131
Emerald ash borer	23 721
Sudden oak death	21 994
Balsam woolly adelgid	13 686
Decline	12 520
Forest tent caterpillar	11 915
Bark beetles	10 156
Pinyon ips	9 253
Eastern larch beetle	7 783
Hemlock decline	6 836
Unknown	6 233
Western balsam bark beetle[b]	5 936
Jeffrey pine beetle	5 089
Other mortality agents (61)	32 037
Total, all mortality agents	**1 670 707**

Note: All values are "footprint" areas for each agent or complex. The sum of the individual agents is not equal to the total for all agents due to the reporting of multiple agents per polygon.

[a] Rollup of multiple agent codes.

[b] Also included in the subalpine fir mortality rollup.

Table 2.2—Beetle taxa included in the "western bark beetle" group

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

NA= not applicable.

Additionally, the survey identified 81 defoliation agents and complexes affecting approximately 3.64 million ha across the conterminous United States in 2012, an area slightly larger than the combined land area of New Jersey and Hawaii. (Two of these defoliation-cause categories were "rollups" of multiple agents.) The most widespread defoliators were western and eastern spruce budworms (*Choristoneura occidentalis* and *C. fumiferana*), affecting 1.49 million ha (table 2.3). Fall cankerworm (*Alsophila pometaria*), tent caterpillars (*Malacosoma* spp.), pinyon needle scale (*Matsucoccus acalyptus*), and larch needle cast (*Meria laricis*) each affected more than 100 000 ha.

The Interior West region (as defined by the Forest Health Monitoring (FHM) Program of the Forest Service) had, by far, the largest area on which mortality-causing agents and complexes were detected in 2012, approximately 1.11 million ha (table 2.4). A large majority of mortality within that area was associated with mountain pine beetle, although spruce beetle, subalpine fir mortality complex, Douglas-fir beetle (*Dendroctonus pseudotsugae*) and *Ips* engraver beetles were also important mortality agents and complexes.

The Getis-Ord analysis detected four major hot spots of intense mortality exposure in the Interior West region (fig. 2.2). The most intense was centered on the border between eastern Idaho and western Montana, in ecoregions

Table 2.3—Defoliation agents and complexes affecting more than 5000 ha in the conterminous United States in 2012

Agents/complexes causing defoliation, 2012	Area
	ha
Spruce budworm (eastern and western)[a]	1 494 127
Fall cankerworm	1 086 930
Tent caterpillars[a]	382 788
Pinyon needle scale	200 215
Larch needle cast	106 115
Aspen defoliation	55 534
Pine butterfly	36 864
Pinyon sawfly	33 736
Cherry scallop shell moth	26 842
Anthracnose	22 666
Jack pine budworm	21 866
Pear thrips	21 302
Unknown defoliator	20 294
Douglas-fir tussock moth	19 269
Unknown	17 571
Baldcypress leafroller	16 439
Pine engraver	16 198
Gypsy moth	15 861
Leafroller/seed moth	12 002
Needlecast	11 656
Fall webworm	11 073
Marssonina blight	9 941
Larch casebearer	8 763
Winter moth	7 937
Balsam fir sawfly	7 407
Other defoliation agents (56)	50 012
Total, all defoliation agents	**3 638 748**

Note: All values are "footprint" areas for each agent or complex. The sum of the individual agents is not equal to the total for all agents due to the reporting of multiple agents per polygon.

[a] Rollup of multiple agent codes.

Table 2.4—The top five mortality agents or complexes for each Forest Health Monitoring region in 2012

Mortality agents and complexes, 2012	Area
	ha
Interior West	
Mountain pine beetle[a]	727 683
Spruce beetle	147 645
Subalpine fir mortality complex[a]	112 021
Douglas-fir beetle	47 104
Ips engraver beetles	39 177
Other mortality agents and complexes (23)	93 515
Total , all mortality agents and complexes	**1 110 409**
North Central	
Spruce budworm	32 278
Emerald ash borer	23 404
Eastern larch beetle	17 428
Mountain pine beetle[a]	12 095
Decline	3 405
Other mortality agents and complexes (20)	9 011
Total, all mortality agents and complexes	**97 609**
North East	
Forest tent caterpillar	15 335
Beech bark disease	4 806
Emerald ash borer	3 907
Decline	2 874
Unknown	2 745
Other biotic mortality agents and complexes (41)	**14 739**
Total, all mortality agents and complexes	**42 995**
South	
Bark beetles	7 865
Unknown	1 208
Southern pine beetle	846
Ips engraver beetles	23
Black turpentine beetles	0
Total, all mortality agents and complexes	**9 942**
West Coast	
Mountain pine beetle[a]	229 259
Western pine beetle	75 926
Fir engraver	72 033
Spruce beetle	25 019
Sudden oak death	21 994
Other biotic mortality agents and complexes (25)	52 980
Total, all mortality agents and complexes	**409 751**
Alaska	
Yellow-cedar decline	7 044
Spruce beetle	6 726
Northern spruce engraver	4 652
Total, all mortality agents and complexes	**18 422**

Note: 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.

M322B–Northern Rockies and Bitterroot Valley, M332A–Idaho Batholith, and M332E–Beaverhead Mountains. Mortality in this area was attributed almost entirely to mountain pine beetle in lodgepole pine (*Pinus contorta*) forests, although Douglas-fir beetle also caused mortality in Douglas-fir (*Pseudotsuga menziesii*) stands. The hot spot extended beyond those ecoregions into several others, including M332D–Belt Mountains, M332F–Challis Volcanics, M333D–Bitterroot Mountains, M333C–Northern Rockies, and 331K–North Central Highlands.

Two hot spots of intense mortality were detected in north-central Colorado and south-central Wyoming where mountain pine beetle and subalpine fir mortality complex caused extensive mortality in two ecoregion sections, M331I–Northern Parks and Ranges and M331H–North-Central Highlands and Rocky Mountains. The clustering of mortality exposure extended into northeast Utah (M331E–Uinta Mountains). In the south-central part of Colorado, spruce beetle, subalpine fir mortality complex, fir engraver (*Scolytus ventralis*), and Douglas-fir beetle were associated with a cluster of mortality in M331G–Southern Central Highlands, M331F–Southern Parks and Rocky Mountain Range, 313B–Navajo Canyonlands, and 341B–Northern Canyonlands.

In west-central Wyoming, mountain pine beetle, five-needle pine decline, spruce beetle, and subalpine fir mortality complex were associated with a hot spot of intense mortality centered on the M331J–Wind River Mountains

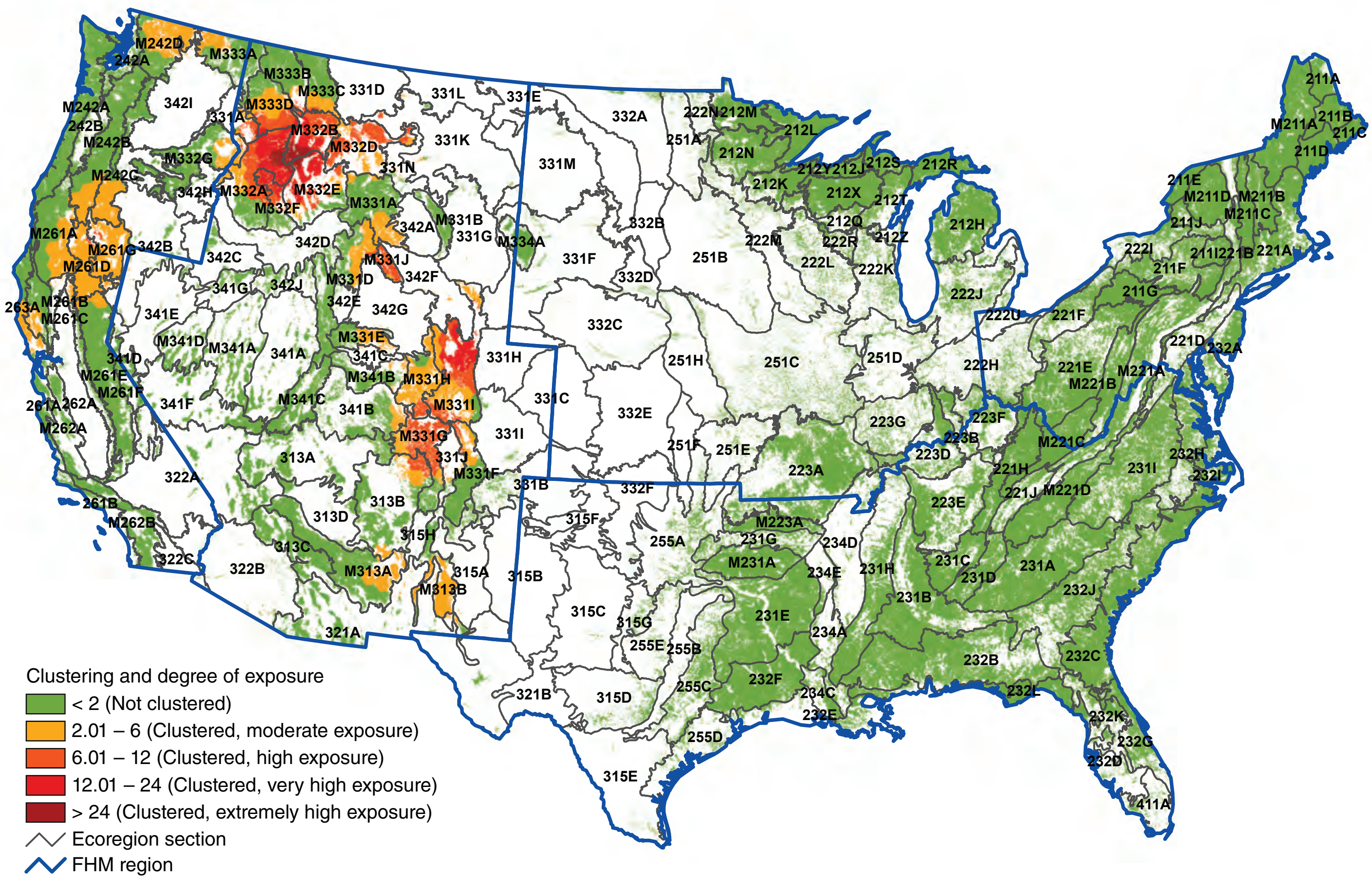

Figure 2.2—Hot spots of exposure to mortality-causing insects and diseases in 2012. Values are Getis-Ord G_i scores, with values >2 representing significant clustering of high percentages of forest area exposed to mortality agents. (No areas of significant clustering of low percentages of exposure, <-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 Moderate Resolution Imaging Spectroradiometer (MODIS) imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection)*

ecoregion, and extending west and north into M331D–Overthrust Mountains and M331A–Yellowstone Highlands.

Finally, a less intense hot spot of mortality was located in southern New Mexico, in M313A–White Mountains-San Francisco Peaks-Mogollon Rim and M313B–Sacramento-Monzano Mountains. Western pine beetle (*Dendroctonus brevicomis*) and *Ips* engraver beetles were the primary mortality agents.

Mountain pine beetle also was the leading cause of mortality in the West Coast region, where detection efforts recorded mortality-causing agents and complexes on nearly 410 000 ha (table 2.4). Several other types of bark beetles, especially western pine beetle, fir engraver, and spruce beetle, were also important causes of mortality in this region. Mountain pine beetle and western bark beetle, in particular, were associated with a relatively extensive geographic hot spot of mortality in northeastern California and south-central Oregon, which encompassed five ecoregion sections: M261G–Modoc Plateau, M261D–Southern Cascades, M261A–Klamath Mountains, M242B–Western Cascades, and M242C–Eastern Cascades (fig. 2.2).

A less intense hot spot in north-central Washington (M242D–Northern Cascades and M333A–Okanagan Highland) was caused primarily by the activity of a wide range of bark beetles, including mountain pine beetle, spruce beetle, Douglas-fir beetle, western balsam bark beetle, and fir engraver. Sudden oak death mortality in tanoak (*Lithocarpus densiflorus*) forests caused another mortality hot spot in California (263A–Northern California Coast).

No geographic hot spots of mortality were detected in the North Central, North East, and South FHM regions. In the North Central region, however, the FHP survey recorded mortality-causing agents and complexes on approximately 98 000 ha (table 2.4). Spruce budworm was the most widely detected mortality agent in the region, followed by emerald ash borer (*Agrilus planipennis*), eastern larch beetle (*Dendroctonus simplex*), and mountain pine beetle.

In the North East FHM region, mortality was recorded on almost 43 000 ha, where forest tent caterpillar (*Malacosoma disstria*) was the most widely identified causal agent. Beech bark disease and emerald ash borer also affected somewhat large areas. In the South, mortality was detected on fewer than 10 000 ha, with bark beetles being the most commonly detected agent (table 2.4).

As with agents of mortality, the Interior West FHM region had the largest area on which defoliating agents and complexes were detected in 2012, approximately 1.6 million ha (table 2.5). Western spruce budworm was by far the most widely detected defoliator in the region, followed by pinyon needle scale, larch needle cast *(Meria laricis)*, and general aspen *(Populus tremuloides)* defoliation.

Three geographic hot spots of intense defoliation occurred in the region (fig. 2.3). The largest, caused by western spruce budworm,

Table 2.5—The top five defoliation agents or complexes for each Forest Health Monitoring region in 2012

Defoliation agents and complexes, 2012	Area
	ha
Interior West	
Western spruce budworm	1 189 386
Pinyon needle scale	198 976
Larch needle cast	84 262
Aspen defoliation	55 534
Pinyon sawfly	33 732
Other defoliation agents and complexes (25)	67 242
Total, all defoliation agents and complexes	**1 593 110**
North Central	
Forest tent caterpillar	136 611
Spruce budworm	65 692
Cherry scallop shell moth	24 237
Jack pine budworm	21 866
Pine engraver	16 198
Other defoliation agents and complexes (10)	35 520
Total, all defoliation agents and complexes	**300 028**
North East	
Anthracnose	22 666
Pear thrips	21 302
Fall webworm	11 038
Forest tent caterpillar	9 041
Fall cankerworm	8 938
Other defoliation agents and complexes (30)	43 904
Total, all defoliation agents and complexes	**90 669**

Defoliation agents and complexes, 2012	Area
	ha
South	
Fall cankerworm	1 077 993
Forest tent caterpillar	232 598
Baldcypress leafroller	16 439
Emerald ash borer	1 767
Unknown	125
Defoliators	7
Total, all defoliation agents and complexes	**1 321 007**
West Coast	
Western spruce budworm	238 896
Pine butterfly	36 851
Larch needle cast	21 852
Balsam fir sawfly	7 407
Needlecast	5 267
Other defoliation agents and complexes (23)	24 219
Total, all defoliation agents and complexes	**333 934**
Alaska	
Defoliators	97 256
Aspen leafminer	27 926
Willow leaf blotchminer	19 218
Large aspen tortrix	4 934
Birch aphid	4 346
Other defoliation agents and complexes (10)	9 588
Total, all defoliation agents and complexes	**161 981**

Note: 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.

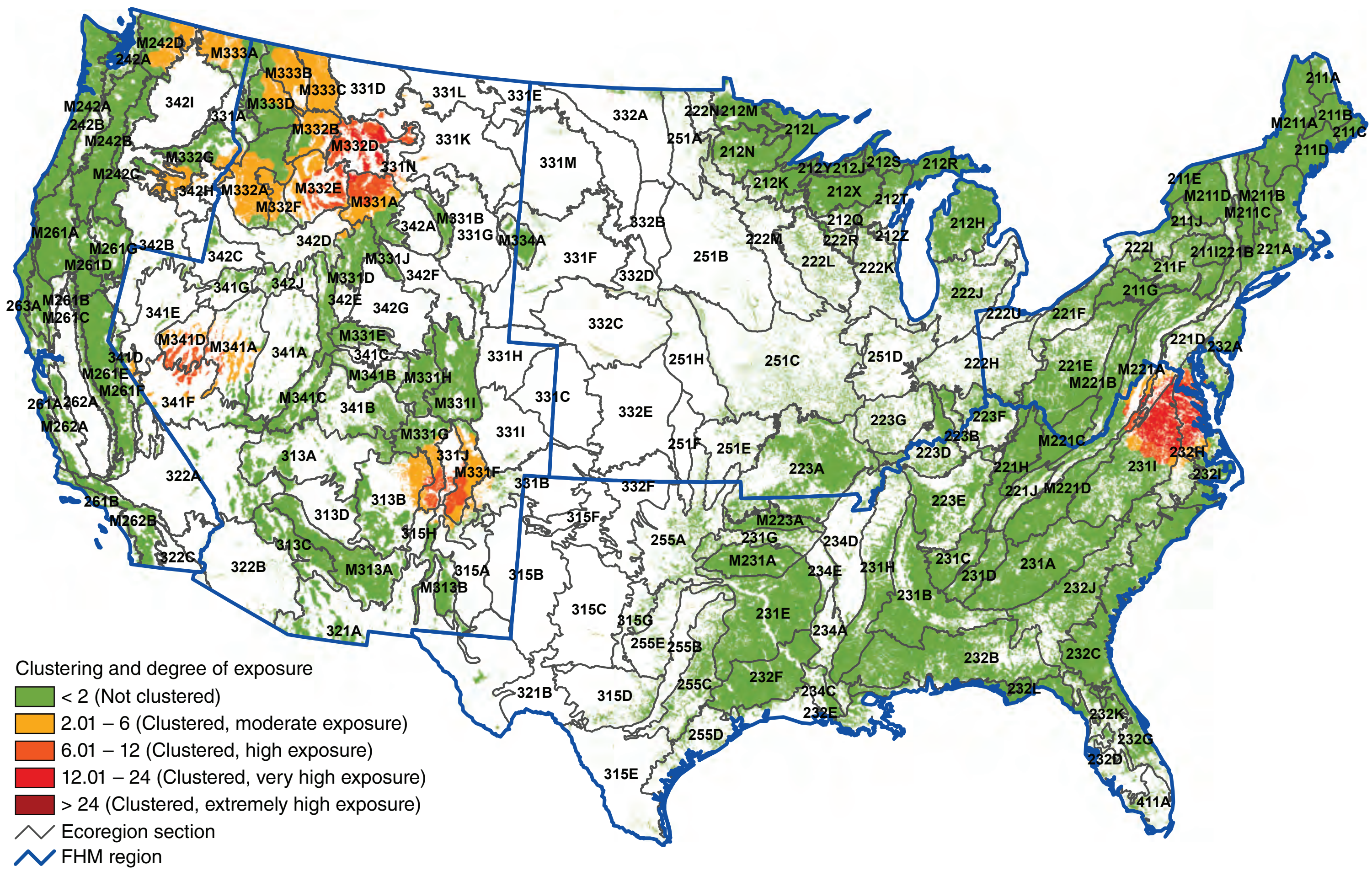

Figure 2.3—Hot spots of exposure to defoliation-causing insects and diseases in 2012. Values are Getis-Ord G_i^* scores, with values >2 representing significant clustering of high percentages of forest area exposed to defoliation agents. (No areas of significant clustering of low percentages of exposure, <-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 Moderate Resolution Imaging Spectroradiometer (MODIS) imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection)

was centered on M332D–Belt Mountains, M332E–Beaverhead Mountains, and M331A–Yellowstone Highlands in western Montana. Western spruce budworm was also the main cause, along with a smaller amount of larch needle cast, of two nearby geographic hot spots that were less intense: one in M332A–Idaho Batholith and M332F–Challis Volcanics, and the other in M333D–Bitterroot Mountains, M333B–Flathead Valley, and M333C–Northern Rockies.

Also in the Interior West Region, western spruce budworm and aspen defoliation were the causal factors associated with a geographic hot spot of defoliation in northern New Mexico and southern Colorado (M331G–South Central Highlands and M331F–Southern Parks and Rocky Mountain Range). Meanwhile, pinyon needle scale (*Matscoccus acalyptus*) and, to a lesser degree, pinyon sawfly (*Neodiprion edulicolus*) were the agents of defoliation associated with a relatively large geographic hot spot in the forested areas of central Nevada (M341D–West Great Basin and Mountains, M341A–East Great Basin and Mountains, and 341F–Southeastern Great Basin).

Western spruce budworm, meanwhile, accounted for about 70 percent of the 334 000 ha of defoliation detected in the FHM West Coast region (table 2.5). While not nearly as widespread, pine butterfly *(Neophasia menapia)* and larch needle cast were the second and third leading defoliators in the region. One geographic hot spot of defoliation occurred in north-central Washington State (M242D–Northern Cascades and M333A–Okanagan Highland), associated with western spruce budworm and some larch needle cast. Another hot spot was located in M332G–Blue Mountains of eastern Oregon, which was caused by western spruce budworm and pine butterfly.

In the North Central FHM region, forest tent caterpillar was the leading defoliator, recorded on about 137 000 ha, or nearly half of the 300 000 ha of defoliation detected in the region (table 2.5). Spruce budworm, cherry scallop shell moth *(Hydria prunivorata)*, and jack pine budworm *(Choristoneura pinus)* were also important agents of defoliation in the region. No geographic hot spots of defoliation were detected in the North Central or North East regions. In the North East, the FHP survey recorded 91 000 ha of forest exposed to defoliators, with anthracnose (*Gnomonia* spp.) and pear thrips *(Taeniothrips inconsequens)* having the greatest geographic extent (table 2.5).

In the South, meanwhile, fall cankerworm was by far the leading defoliation agent, detected across more than 1 million ha in eastern Virginia (table 2.5). The insect outbreak resulted in a geographic hot spot in two Virginia ecoregions, 231I–Central Appalachian Piedmont and 232H–Middle Atlantic Coastal Plains and Flatwoods (fig. 2.3). Across the South, defoliation was recorded on about 1.3 million ha. Forest tent caterpillar was the second most important defoliation agent, detected on 233 000 ha.

Alaska

In 2012, mortality was recorded on 18 000 ha in Alaska, associated with three agents and complexes (table 2.4). This is a very small proportion of the forested area in Alaska that was surveyed in 2012 (approximately 6.91 million ha). Yellow-cedar (*Chamaecyparis nootkatensis*) decline was the most widely detected mortality agent, found on about 7000 ha in the Alaska panhandle, followed by spruce beetle, also affecting about 7000 ha, and northern spruce engraver (*Ips perturbatus*), which was detected on about 4600 ha. The percent of surveyed forest exposed to mortality agents did not exceed 1 percent in any of Alaska's ecoregions (fig. 2.4).

Defoliators affected a much larger area of Alaska during 2012, when 15 defoliating agents and complexes were recorded on nearly 162 000 ha (table 2.5). For much of that area, approximately 97 000 ha, non-specific defoliators were the assigned cause. Aspen leafminer *(Phyllocnistis populiella)* was detected on 28 000 ha, mostly in the central parts of Alaska. Meanwhile, willow leaf blotchminer *(Micrurapteryx salicifoliella)* was found on approximately 19 000 ha, large aspen tortrix *(Choristoneura conflictana)* was detected on about 5000 ha, and birch aphid *(Euceraphis betulae)* was recorded on about 4000 ha.

The three Alaska ecoregions with the highest proportion of surveyed forest area affected by defoliators were all located in the southwestern portion of the State (fig. 2.5). Defoliators were detected on 6.67 percent of surveyed forest in M213A–Northern Aleutian Range, 6.34 percent of surveyed forest in M131D–Nushagak-Lime Hills, and 4.12 percent of the surveyed forest in M129B–Ahklun Mountains. A few central and east-central Alaskan ecoregions also had relatively high levels of defoliation detection, including 131A–Yukon Bottomlands (3.08 percent), M139C–Dawson Range (2.83 percent), and 139A–Yukon Flats and M139A–Ray Mountains (2.38 percent in each).

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 geographic hot spot detection analyses, offer a useful approach for identifying geographic areas where the concentration of monitoring and management activities might be most effective.

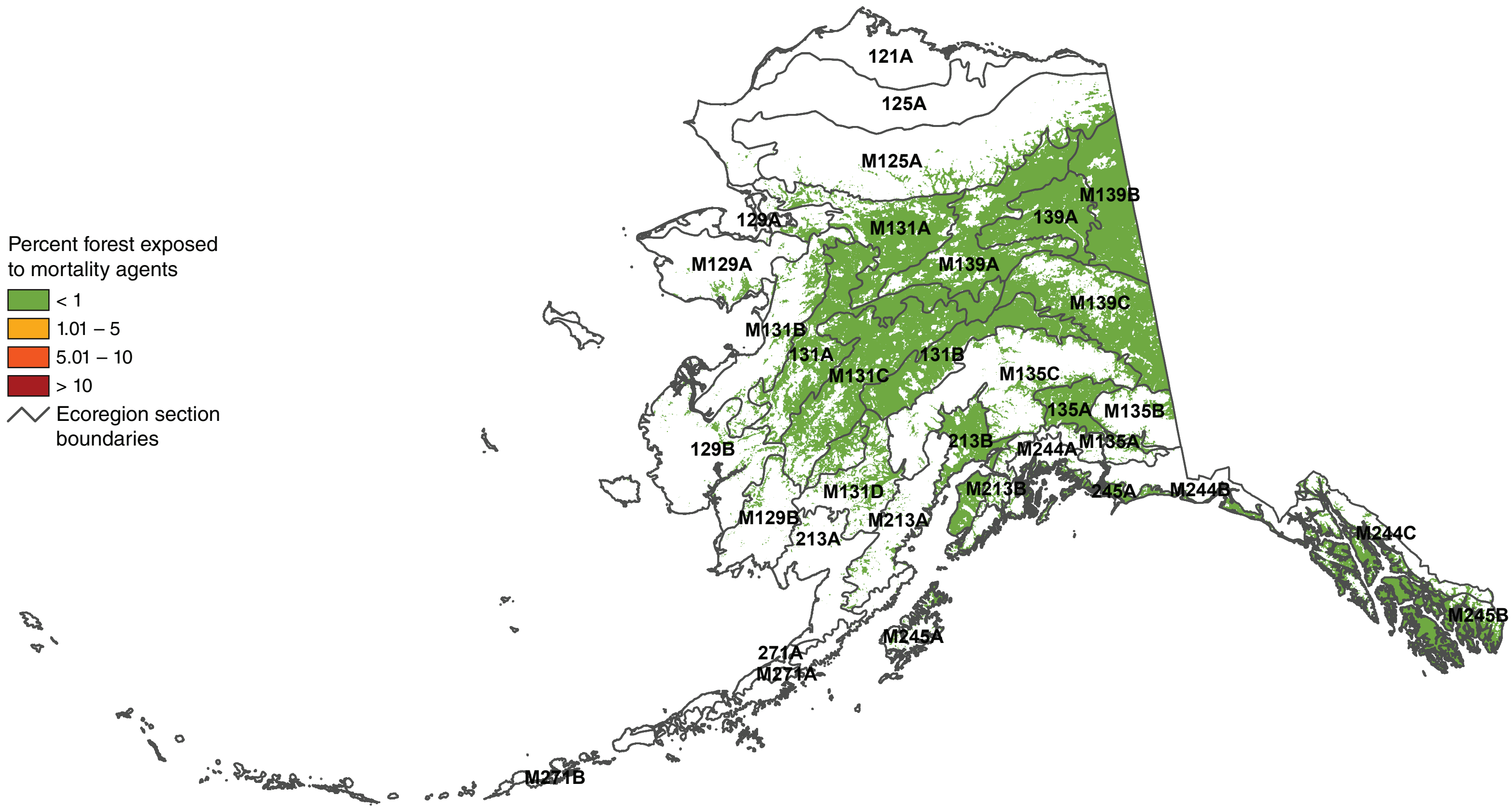

Figure 2.4—Percent of surveyed forest in Alaska ecoregion sections exposed to mortality-causing insects and diseases in 2012. The gray lines delineate ecoregion sections (Nowacki and Brock 1995). Background forest cover is derived from Moderate Resolution Imaging Spectroradiometer (MODIS) imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection)

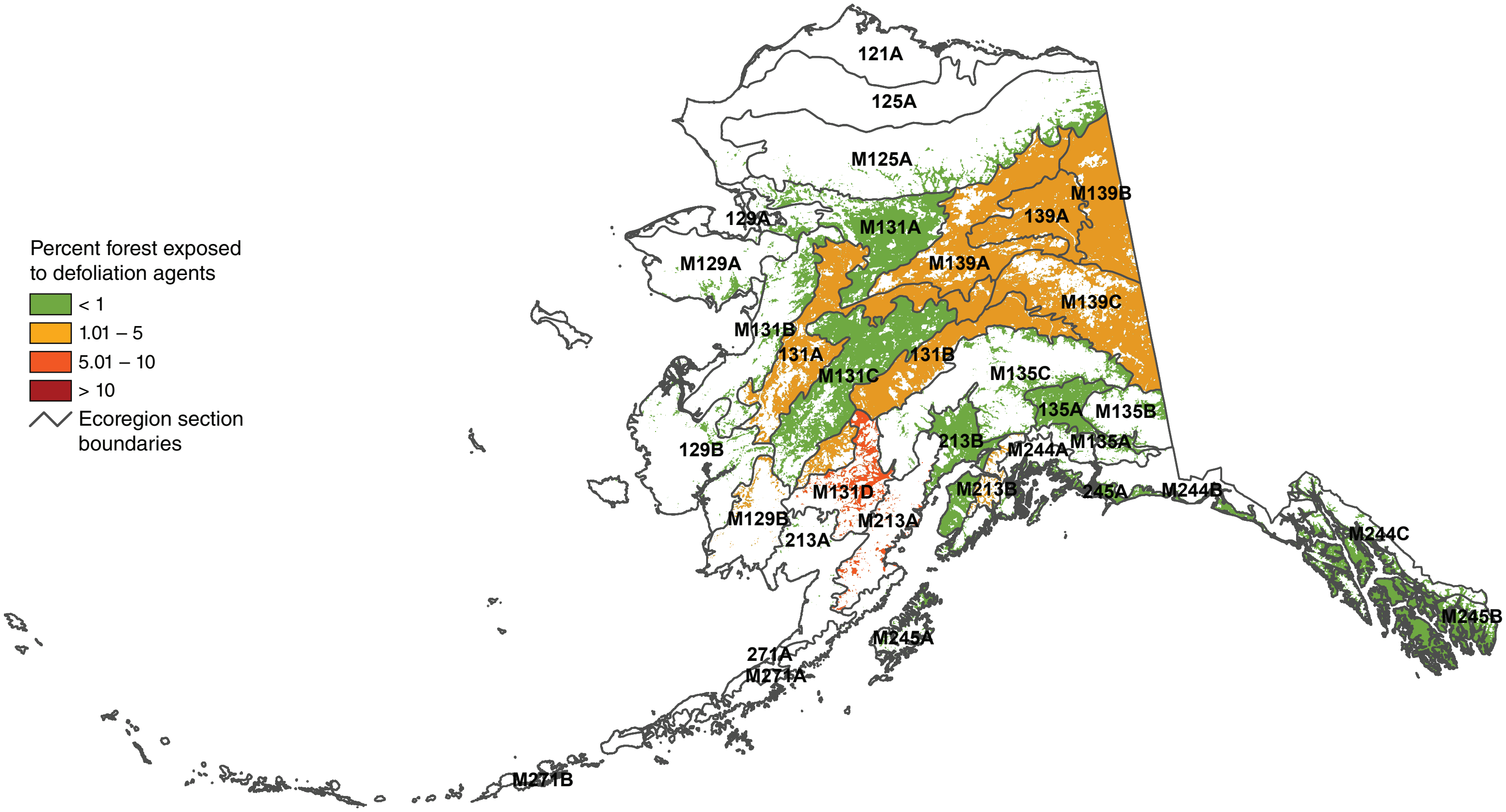

Figure 2.5—Percent of surveyed forest in Alaska ecoregion sections exposed to defoliation-causing insects and diseases in 2012. The gray lines delineate ecoregion sections (Nowacki and Brock 1995). Background forest cover is derived from Moderate Resolution Imaging Spectroradiometer (MODIS) imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection)

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CHAPTER 3.

Large-Scale Patterns of Forest Fire Occurrence in the Conterminous United States and Alaska, 2012

Kevin M. Potter

INTRODUCTION

Free-burning wildland fire has been a frequent ecological presence 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 despite causing extensive tree mortality (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 certain ecosystems (Edmonds and others 2011). Specifically, fire outside its historic range of frequency and intensity in a given forest ecosystem 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 number of acres burned annually decreased from approximately 16-20 million ha (40-50 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 because 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).

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 2012. It also, within a geographic context, compares 2012 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 necessary.

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 2013) allows analysts to spatially display and summarize fire occurrences across broad geographic regions (Coulston and others 2005; Potter 2012a, 2012b, 2013a, 2013b, 2014). A fire occurrence is defined as one daily satellite detection of wildland fire in a 1-km^2 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 km^2, with the center of a pixel recorded as a fire occurrence (USDA Forest Service 2013). 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 thermal bands do not differentiate between a hot fire in a relatively small area (0.01 km^2, for example) and a cooler fire over a larger area (1 km^2, for example). The MODIS Active Fire database does well at capturing large fires 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 2012 were subjected to Geographic Information System (GIS) processing to determine number of fire occurrences per 100 km^2 (10 000 ha) of forested area for each ecoregion section in the conterminous United 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 (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 2012 was then compared to the mean fire density values for the first 11 full years of MODIS Active Fire data collection (2001–11). Specifically, the difference of the 2012 value and the previous 11-year mean for an ecoregion was divided by the standard deviation across the previous 11-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 if the fire occurrence density in the ecoregion in 2012 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. 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® 9.2 (ESRI 2006) was employed to identify forested areas in the conterminous United States with higher-than-expected fire occurrence density in 2012. The spatial units of analysis were 3,382 cells of approximately 2500 km^2 from a hexagonal lattice of the conterminous United States, intensified from Environmental Monitoring and Assessment Program (EMAP) North America hexagon coordinates (White and others 1992). This cell size allows for analysis at a medium-scale resolution of approximately the same area as a typical county. Fire occurrence density values for each hexagon were quantified as the number of forest fire occurrences per 100 km^2 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., non-stationarities) in a data set, 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 United 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 ($p < 0.025$), since 95 percent of the observations under a normal distribution should be within approximately 2 standard deviations of the mean (Laffan 2006). Values between -1.96 and 1.96 have no statistically significant concentration of high or low values; a hexagon and its 18 neighbors, in other words, have a range of both high and low numbers of fire occurrences per 100 km^2 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 2006).

RESULTS AND DISCUSSION

The MODIS Active Fire database captured 138,000 wildland forest fire occurrences across the conterminous United States in 2012, the most of any year of MODIS data collection (fig. 3.1). This number was approximately 77 percent greater than in 2011 (78,235 forest fire occurrences) and more than twice the 64,929 mean annual forest fire occurrences over the previous 11 full years of data collection. In contrast, the MODIS database captured only 687 forest fire occurrences in Alaska in 2012, the third fewest since 2001 and a small fraction of the previous 11-year annual mean of 13,428.

The increase in the total number of fire occurrences across the conterminous States is generally consistent with the official wildland

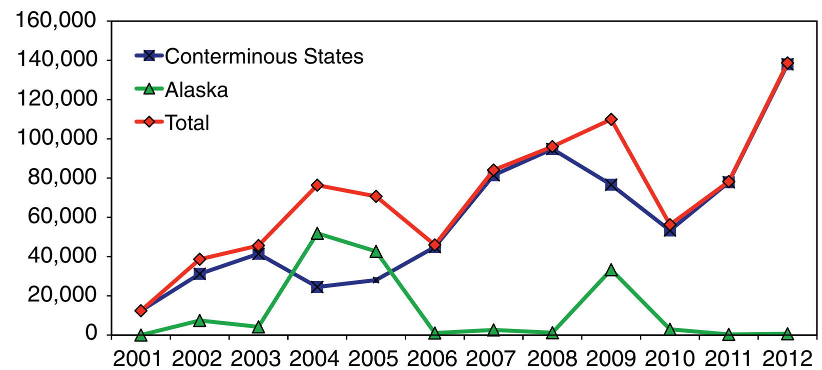


Figure 3.1—Forest fire occurrences detected by Moderate Resolution Imaging Spectroradiometer (MODIS) from 2001 through 2012 for the conterminous United States, Alaska, and the two regions combined. (Data source: U.S. Department of Agriculture Forest Service, Remote Sensing Applications Center)

fire statistics; the area burned nationally in 2012 (3 774 195 ha) was 128 percent of the 10-year average, with 51 fires exceeding 16 187 ha (10 more than in 2011) (National Interagency Coordination Center 2013). The total area burned nationally represented a 7-percent increase from 2011 (3 525 365 ha) (National Interagency Coordination Center 2012). It is important to note that estimates of burned area and calculations of MODIS-detected fire occurrences are 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-km^2 pixel. Analyses of the MODIS-detected fire occurrences, therefore, measure the total number of 1-km^2 pixels each day with fire, as opposed to quantifying only the area on which fire occurred at some point during the course of the year.

In 2012, the highest forest fire occurrence densities occurred in ecoregions of the Interior West (fig. 3.2), where a summer heat wave combined with record to near-record dryness following below-normal winter snowpack. Colorado and Wyoming, for example, had their warmest summers on record, while Wyoming, South Dakota, and New Mexico had one of the driest summers in history (National Interagency Coordination Center 2013). The drought conditions resulted in below-normal fuel moisture and above-normal Energy Release Component indices from New Mexico west to California and north to southern Oregon, Idaho, and Wyoming (National Interagency Coordination Center 2013).

The forested ecoregion with the highest wildland forest fire occurrence density in 2012 (a remarkable 93.5 fires per 100 km^2 of forest) was section M332A–Idaho Batholith (fig. 3.2). This ecoregion section is located in the Eastern Great Basin Geographic Region where official wildland fire statistics recorded nearly 800 000 ha burned (National Interagency Coordination Center 2013), including the 138 179-ha Mustang Complex fire. To the southeast, the M331J–Wind River Mountains ecoregion in western Wyoming experienced a fire occurrence density of 31.9 fires per 100 km^2 of forest. Meanwhile, several ecoregions that contain relatively small amounts of forest (and therefore do not stand out as easily on fig. 3.2) had even higher fire occurrence densities than the Wind River Mountains:

- 331G–Powder River Basin in northeastern Wyoming and southeastern Montana (135.0 fire occurrences per 100 km^2 of forest)
- 331F–Western Great Plains in northwestern Nebraska and southwestern South Dakota (49.6 per 100 km^2 of forest)
- 342B–Northwestern Basin and Range in northwestern Nevada and southeastern Oregon (43.3 per 100 km^2 of forest)
- 331K–North Central Highlands in eastern Montana (38.0 per 100 km^2 of forest)

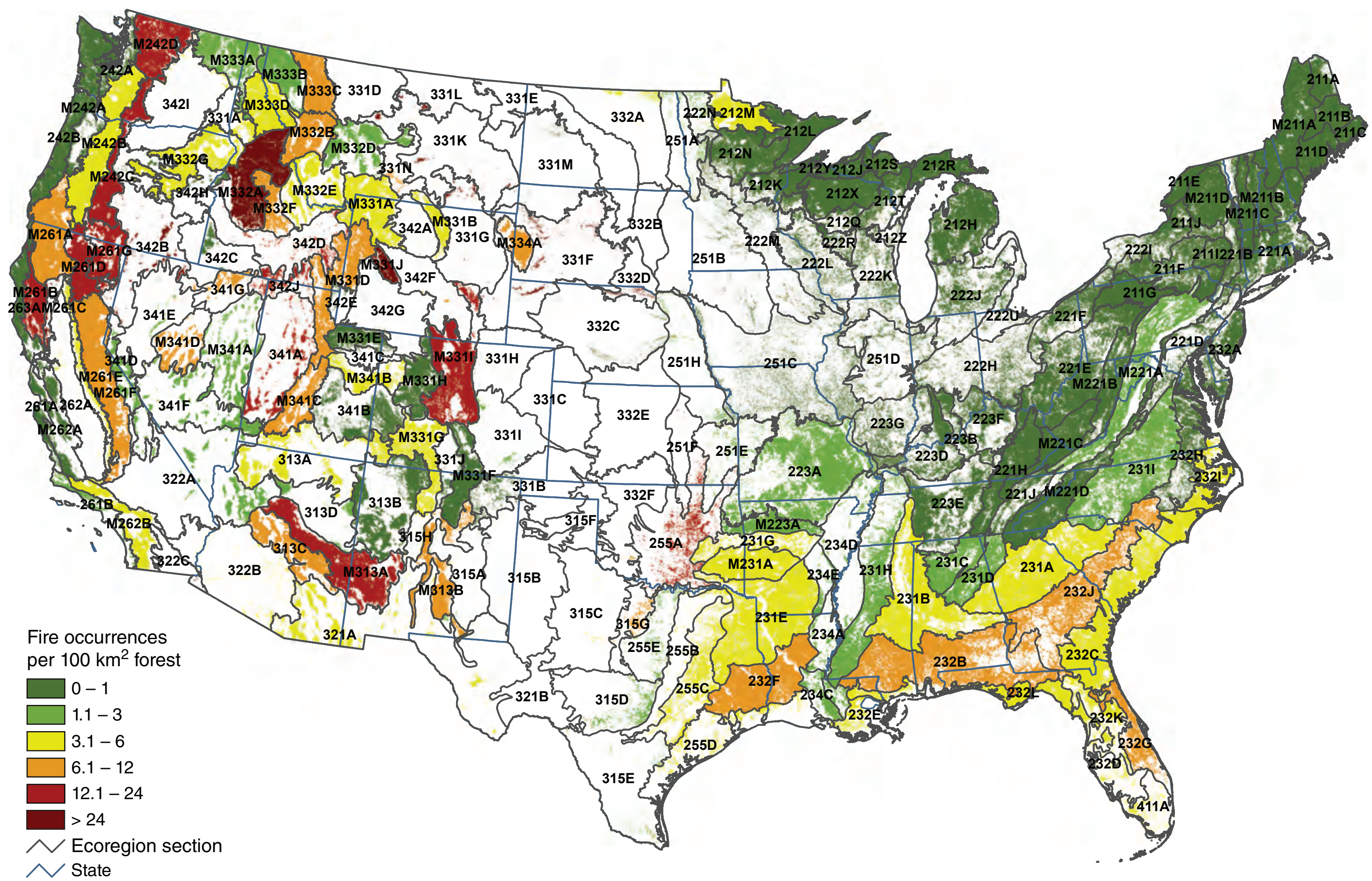


Figure 3.2—The number of forest fire occurrences per 100 km^2 (10 000 ha) of forested area, by ecoregion section within the conterminous United States for 2012. The gray lines delineate ecoregion sections (Cleland and others 2007). Forest cover is derived from Moderate Resolution Imaging Spectroradiometer (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)

Elsewhere in the West, high fire occurrence densities were detected from northern California (M261C–Northern California Interior Coast Ranges, 24.9 fire occurrences per 100 km^2 of forest; and M261B–Northern California Coast Ranges, 20.5 fire occurrences) along the Cascade Mountains into Oregon and Washington (M261D–Southern Cascades, 18.6 fire occurrences; M261G–Modoc Plateau, 23.4 fire occurrences; M242C–Eastern Cascades, 12.8 fire occurrences; and M242D–Northern Cascades, 12.7 fire occurrences).

Meanwhile, the M313A–White Mountains-San Francisco Peaks-Mogollon Rim ecoregion experienced 18.6 fire occurrences per 100 km^2 of forest, driven in part by the 120 534 ha Whitewater-Baldy Complex fire, the largest in New Mexico history. In north-central Colorado, several fires, including the highly destructive High Park and Waldo Canyon fires, resulted in 12.1 fire occurrences for each 100 km^2 of forest in M331I–Northern Parks and Ranges. High fire occurrence densities were also evidenced in western Utah (18.8 for both 342J–Eastern Basin and Range and 341A–Bonneville Basin).

Ecoregions of the Southeastern United States generally experienced moderate fire occurrence densities in 2012, fewer than recent years in many locations. One exception incorporated the forested areas of central Oklahoma (255A–Cross Timbers and Prairie), where 12.9 fires were detected per 100 km^2 of forest (fig. 3.2). Southeastern ecoregions with relatively high fire densities included 232F–Coastal Plains and Flatwoods-Western Gulf (Louisiana and east Texas, 8.0 fire occurrences), 232G–Florida Coastal Lowlands-Atlantic (eastern Florida, 7.3 fire occurrences), 232B–Gulf Coast Plains and Flatwoods (7.0 fire occurrences), and 232J–Southern Atlantic Coastal Plains and Flatwoods (6.8 fire occurrences).

Fire occurrence densities, meanwhile, were almost universally low in the Northeastern and Midwestern States, with two exceptions: 332A–Northeastern Glaciated Plains (in northern North Dakota, 3.9 fire occurrences) and 212M–Northern Minnesota and Ontario (in northern Minnesota, 3.6 fire occurrences).

Meanwhile, few fire occurrences were detected in Alaska, which experienced near-normal summer temperatures and above-normal precipitation (National Interagency Coordination Center 2013). No Alaskan ecoregion had more than a single fire occurrence per 100 km^2 of forest (fig. 3.3). The M131A–Upper Kobuk-Koyukuk ecoregion had the highest fire occurrence density, with only 0.7 fire occurrences detected per 100 km^2 of forest, followed by 131B–Kuskokwim Colluvial Plain (0.6 fire occurrences per 100 km^2 of forest).

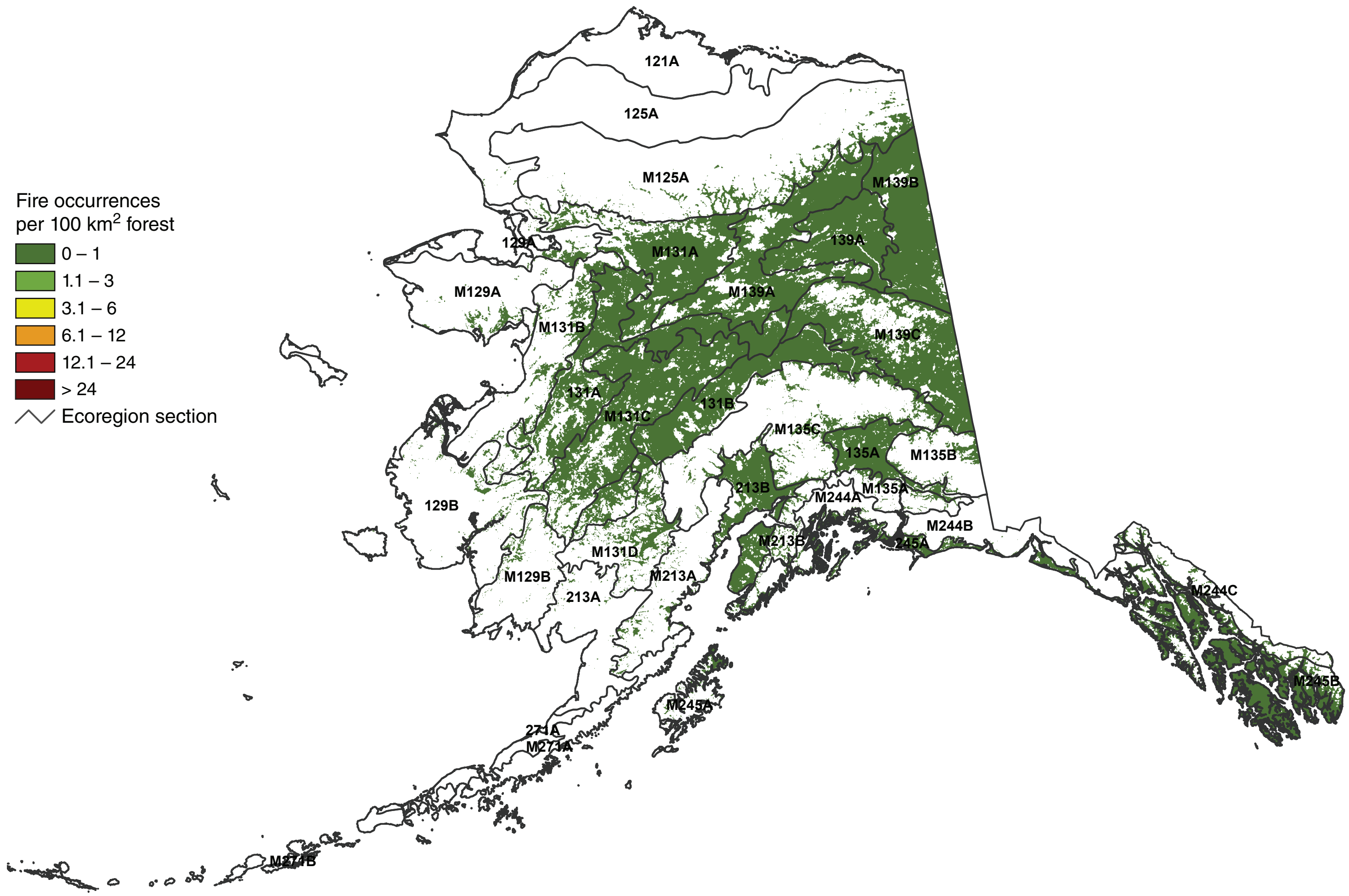


Figure 3.3—The number of forest fire occurrences per 100 km^2 (10 000 ha) of forested area, by ecoregion section in Alaska for 2012. The gray lines delineate ecoregion sections (Nowacki and Brock 1995). Forest cover is derived from Moderate Resolution Imaging Spectroradiometer (MODIS) imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: U.S. Department of Agriculture Forest Service, Remote Sensing Applications Center)

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 11 full years of MODIS Active Fire data collection (2001–11). In general, most ecoregions within the Northeastern, Midwestern, Middle Atlantic, Appalachian, and Central Rocky Mountain regions experienced < 1 fire per 100 km^2 of forest over that period, with means higher in the Northern Rocky Mountain, California, Southeastern, and Southwestern regions (fig. 3.4A). Heavily forested ecoregions that have experienced the most fires on average are located in central Idaho, near the southern California coast, and in north-central Texas (mean annual fire occurrence densities of 6.1–12.0). Ecoregions with the greatest variation in fire occurrence densities over time based on the standard deviation from 2001–11 were also located along the California coast and in central Idaho, with moderate variation in western Montana, central and southeastern Arizona and southwestern New Mexico, and eastern North Carolina (fig. 3.4B). Lesser degrees of variation occurred throughout the Southeast, central California, noncoastal Oregon and Washington, northwestern Wyoming, and northern Minnesota. The least variation was apparent throughout most of the Midwest and Northeast.

In 2012, large areas of the conterminous United States experienced greater fire occurrence densities than normal, compared to the previous 11-year mean and accounting for variability over time based on a standardized *z*-score (fig. 3.4C). This included much of the Rocky Mountain region, and parts of the Pacific Northwest, Middle Atlantic, Great Lakes, and Southeastern regions. Several of these were ecoregions that had very high fire occurrence densities in 2012, including M332A–Idaho Batholith (in Idaho), M331J–Wind River Mountains (in Wyoming), M331I–Northern Parks and Ranges (in Colorado and Wyoming), M313A–White Mountains–San Francisco Peaks-Mogollon Rim (in Arizona and New Mexico), M261D–Southern Cascades (in California and Oregon), M261G–Modoc Plateau (in California and Oregon), and M242C–Eastern Cascades (in Oregon and Washington). Others had moderate fire occurrence densities in 2012 that still deviated considerably from the previous 11-year mean, including 212M–Northern Minnesota and Ontario (in Minnesota), 232F–Coastal Plains and Flatwoods-Western Gulf (in Texas and Louisiana), M331G–Central Highlands (in Colorado and New Mexico), M341C–Utah High Plateau (in Utah), M334A–Black Hills (in South Dakota and Wyoming), M331B–Bighorn Mountains (in Wyoming), and M332E–Beaverhead Mountains (in Montana and Idaho).

Of perhaps greater interest are the many ecoregions across much of the Eastern United States that had low fire occurrence densities in 2012 that were still higher than the longer-term mean, accounting for variability over time (fig. 3.4C). In the Southeast, these included 234C–Atchafalaya and Red River Alluvial Plains

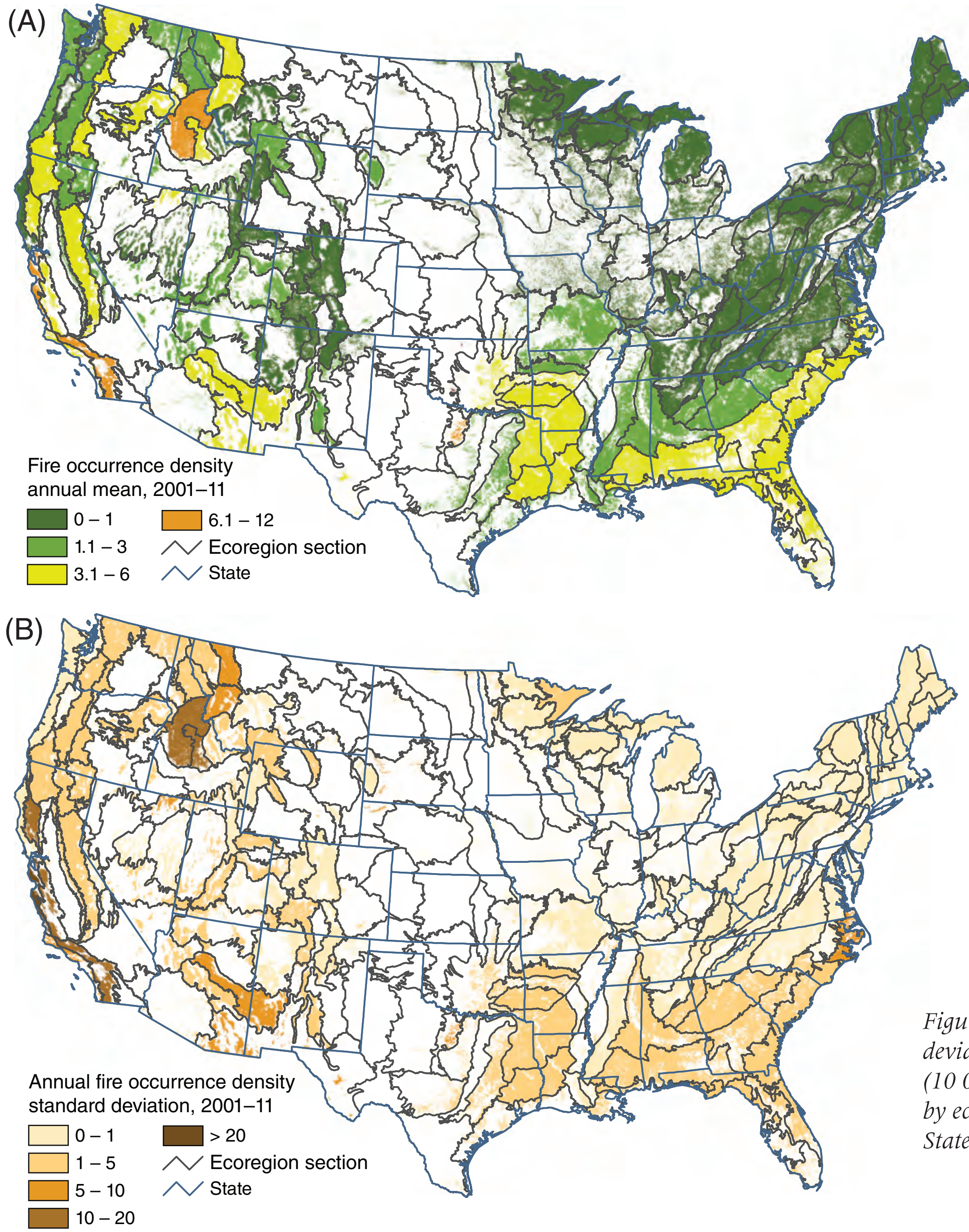


Figure 3.4—(A) Mean number and (B) standard deviation of forest fire occurrences per 100 km^2 (10 000 ha) of forested area from 2001 through 2011, by ecoregion section within the conterminous United States. *(continued on next page)*

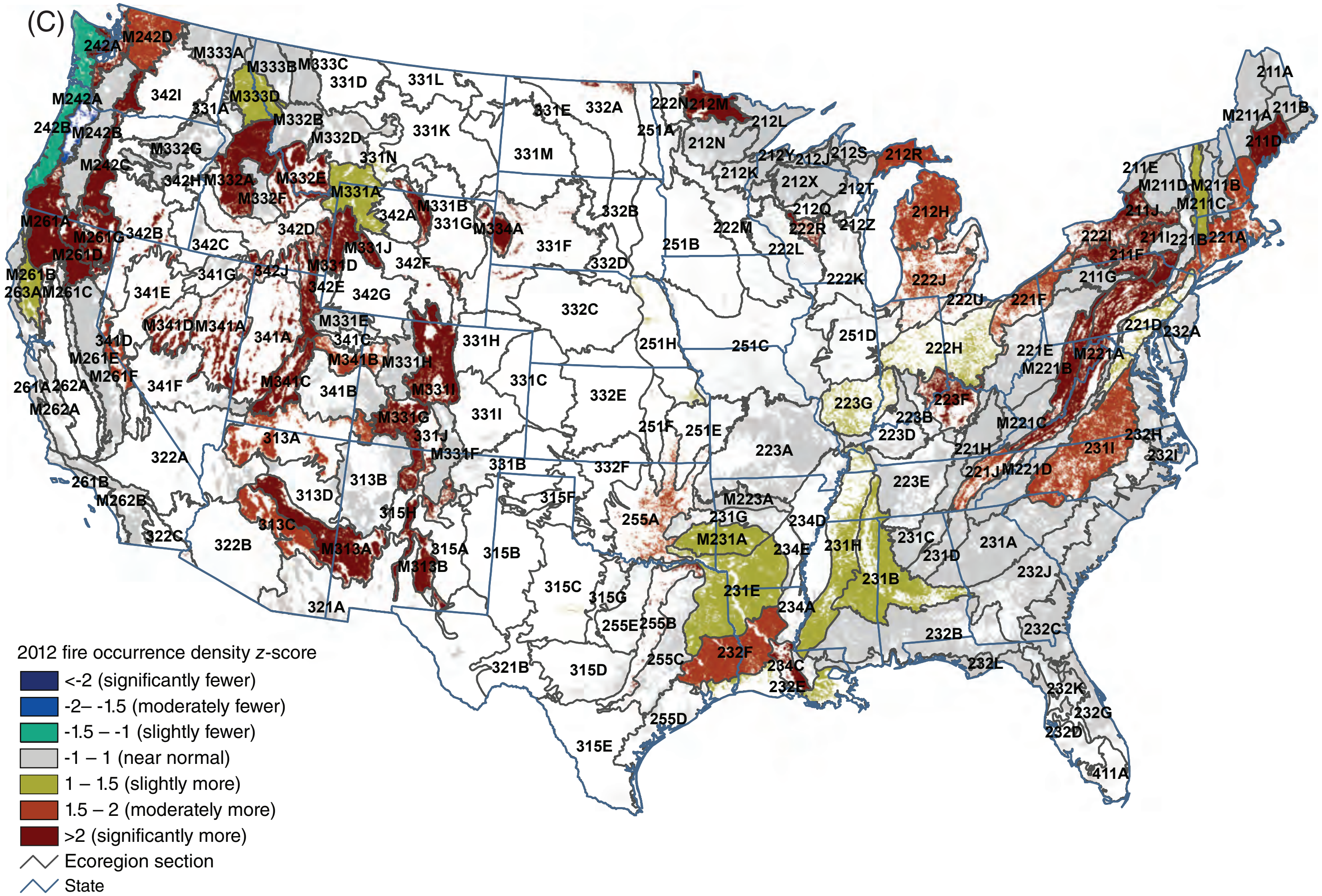


Figure 3.4 (continued)—(C) Degree of 2012 fire occurrence density excess or deficiency by ecoregion relative to 2001–11 and accounting for variation over that time period. The gray lines delineate ecoregion sections (Cleland and others 2007). Forest cover is derived from Moderate Resolution Imaging Spectroradiometer (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 Louisiana), 221J–Central Ridge and Valley (in Tennessee), and 231I–Central Appalachian Piedmont (in North Carolina and Virginia). In the vicinity of the Great Lakes, these included 222R–Wisconsin Central Sands (in Wisconsin); 221F–Western Glaciated Allegheny Plateau (in Ohio and Pennsylvania); 222I–Erie and Ontario Lake Plain (in New York, Pennsylvania, and Ohio); and 212R–Eastern Upper Peninsula, 212H–Northern Lower Peninsula, 222J–South Central Great Lakes, and 222U–Lake Whittlesey Glaciolacustrine Plain (in Michigan). In the Central and Northern Appalachians, there were four such ecoregions: M221B–Allegheny Mountains, M221A–Northern Ridge and Valley, 211F–Northern Glaciated Allegheny Plateau, and 221J–Tug Hill Plateau-Mohawk Valley. In New England, meanwhile, two ecoregions with low fire occurrence density in 2012 had fire densities exceeding the long term mean: 221A–Lower New England and 221D–Central Maine Coastal and Embayment.

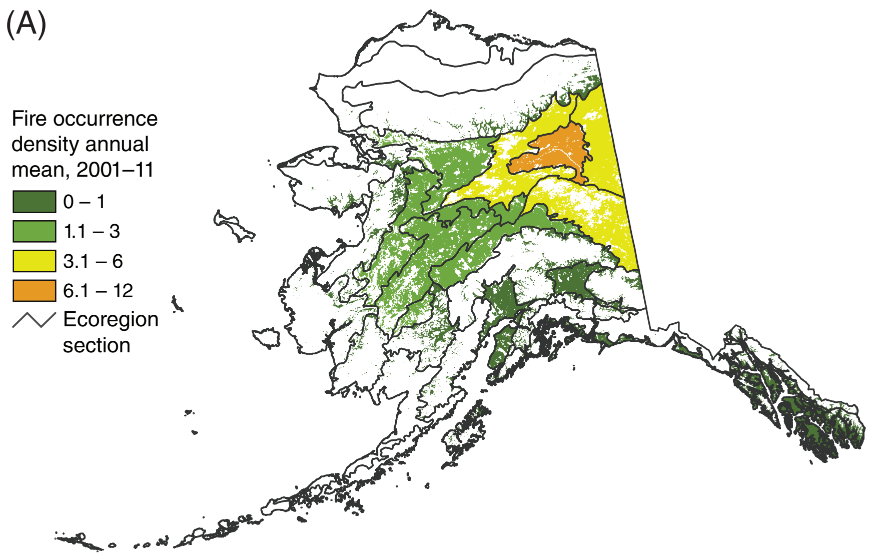
Only one ecoregion in the conterminous United States had a lower fire occurrence density in 2012 compared to the longer-term: M242A–Oregon and Western Coast Ranges (fig. 3.4C). This is a region with a relatively low annual mean fire occurrence density (1.37 fires per 100 km^2 of forest per year) and a low level of variability in that mean. With above-average spring and summer precipitation (National Interagency Coordination Center 2013), it had a fire occurrence density of only 0.72 fires per 100 km^2 of forest in 2012.

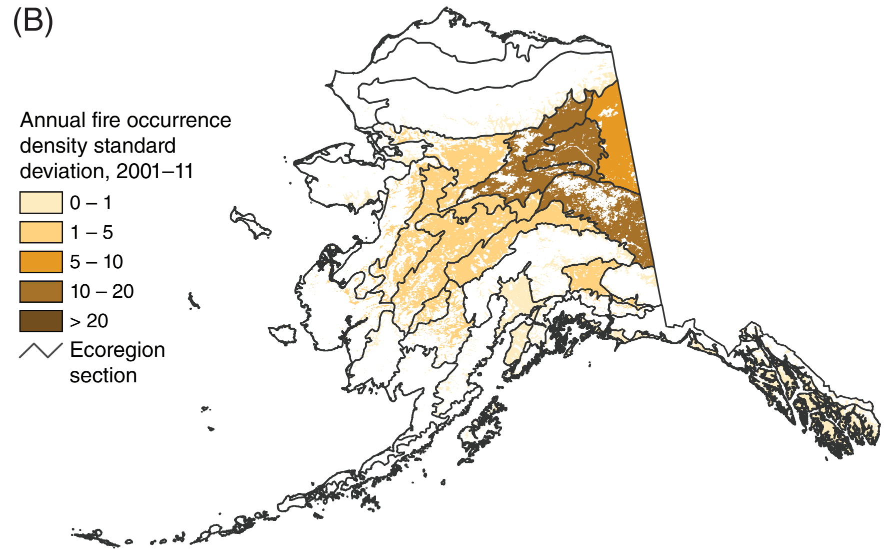
In Alaska, meanwhile, the highest mean annual fire occurrence density between 2001 and 2011 occurred in the east-central and central parts of the State (fig. 3.5A) in the 139A–Yukon Flats ecoregion, with moderate mean fire occurrence density in neighboring areas. Many of those same areas experienced the greatest degree of variability over the 11-year period (fig. 3.5B). In 2012, no ecoregions were outside the range of near-normal fire occurrence density, compared to the mean of the previous 11 years and accounting for variability (fig. 3.5C).

Geographic 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 geographic hot spot analysis can offer insights into where, statistically, fire occurrences are more concentrated than expected by chance. In 2012, the most intense geographic hot spots of fire density within the conterminous United States were located in the Northern Rocky Mountain region (fig. 3.6). The largest of these occurred across parts of seven ecoregion sections in central Idaho and western Montana:

- M332A–Idaho Batholith
- M332F–Challis Volcanics
- M332E–Beaverhead Mountains
- M332B–Northern Rockies and Bitterroot Valley
- M333D–Bitterroot Mountains
- 331A–Palouse Prairie
- M332G–Blue Mountains





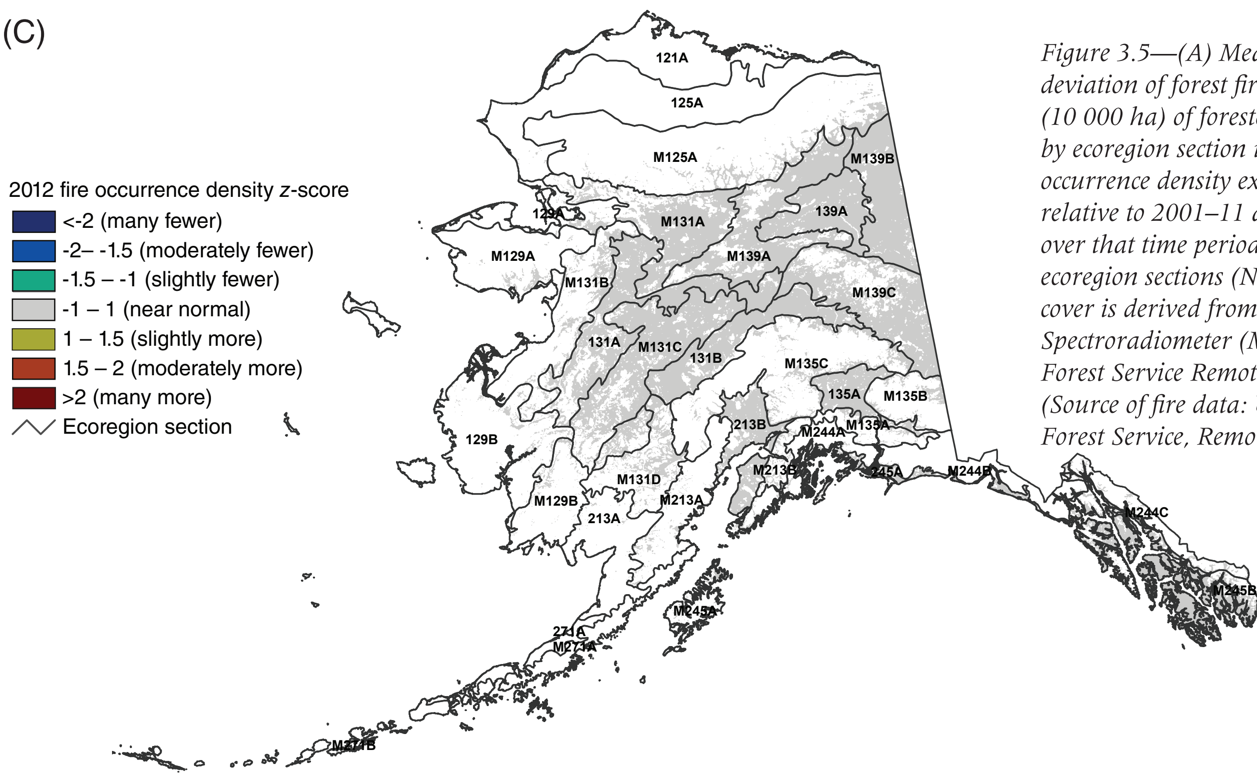


Figure 3.5—(A) Mean number and (B) standard deviation of forest fire occurrences per 100 km² (10 000 ha) of forested area from 2001 through 2011, by ecoregion section in Alaska. (C) Degree of 2012 fire occurrence density excess or deficiency by ecoregion relative to 2001–11 and accounting for variation over that time period. The gray lines delineate ecoregion sections (Nowacki and Brock 1995). Forest cover is derived from Moderate Resolution Imaging Spectroradiometer (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)

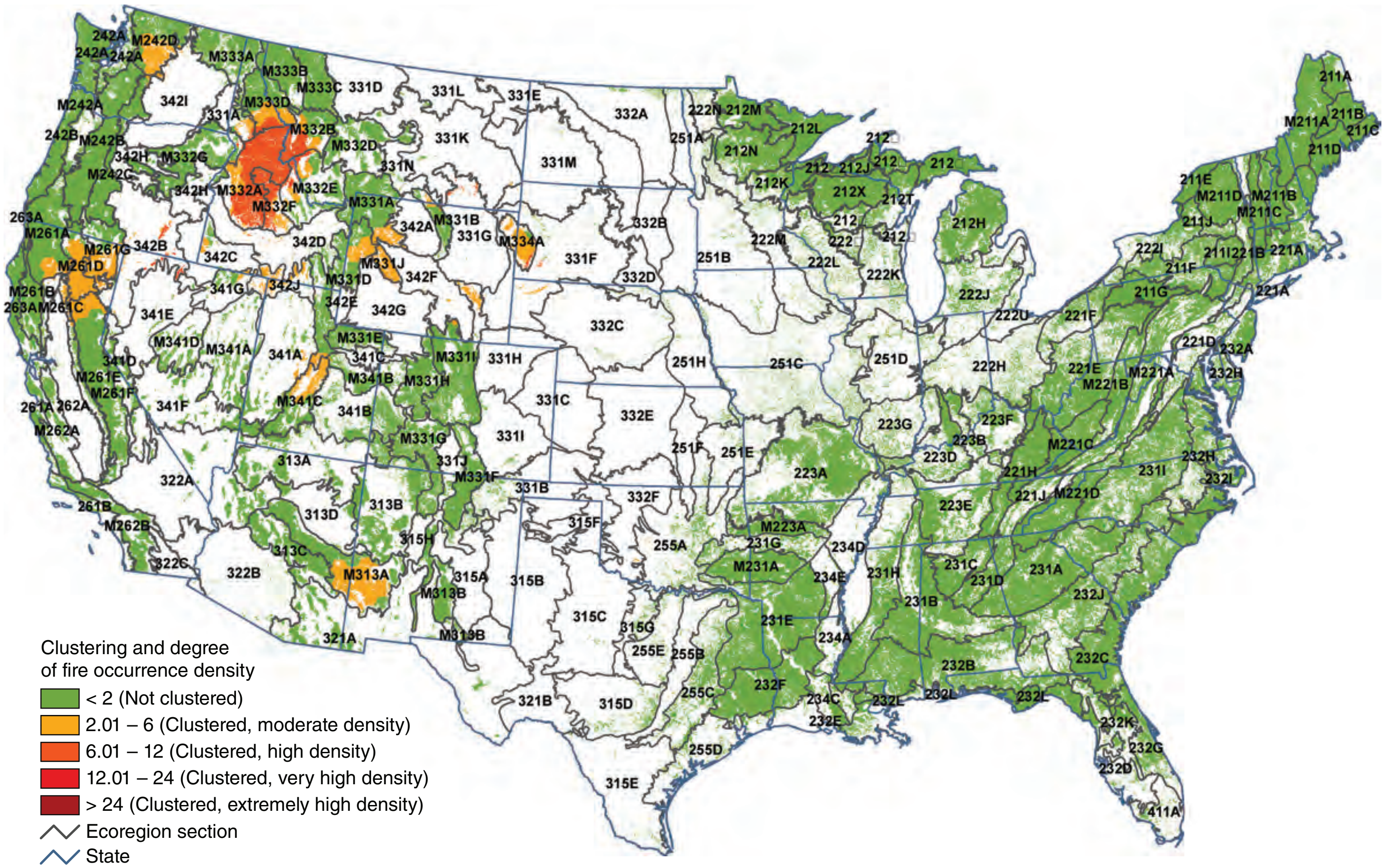


Figure 3.6—Hot spots of fire occurrence across the conterminous United States for 2012. Values are Getis-Ord G_i** *scores, with values >2 representing significant clustering of high fire occurrence densities. (No areas of significant clustering of low fire occurrence densities, <-2, were detected.) The gray lines delineate ecoregion sections (Cleland and others 2007). Background forest cover is derived from Moderate Resolution Imaging Spectroradiometer (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)*

About 200 km to the southwest, a smaller but also intense hot spot was located in the forested portions of northwestern Nevada and southeastern Oregon (342B–Northwestern Basin and Range ecoregion). Two other small areas of intense forest fire occurrence clustering were detected in southeastern Montana (331G–Powder River Basin) and western South Dakota (M334A–Black Hills and 331F–Western Great Plains). Several less intense geographic hot spots of fire occurrence density were also detected in the Northern Rocky Mountains, including:

- Western Wyoming (M331J–Wind River Mountains, M331D–Overthrust Mountains, and M331A–Yellowstone Highlands)
- Southeastern Wyoming (M331I–Northern Parks and Ranges)
- Northern Utah, southern Idaho, and northeastern Nevada (342J–Eastern Basin and Range and 341G–Northeastern Great Basin)
- Southwestern Idaho (342C–Owyhee Highlands)
- Central Utah (M341C–Utah High Plateau and 341A–Bonneville Basin)

The Getis-Ord hot spot analysis also detected less-intense concentrations of forest fire occurrence density in western New Mexico/eastern Arizona (M313A–White Mountains-San Francisco Peaks-Mogollon Rim), in north-central Washington (M242D–Northern Cascades and M242C–Eastern Cascades), and in northern California (M261D–Southern Cascades, M261G–Modoc Plateau, M261E–Sierra Nevada, M261F–Sierra Nevada Foothills, and M261A–Klamath Mountains). No hot spots of fire occurrence density were detected in the Eastern United States in 2012.

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-km 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 of 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 to 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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CHAPTER 4.

Drought Patterns in the Conterminous United States, 2012

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William D. Smith
John W. Coulston

INTRODUCTION

Droughts are a regular occurrence in most U.S. forests. However, the frequency and intensity of these droughts 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 usually 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).

Plants initially 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 becomes moderate to severe (Kareiva and others 1993, Mattson and Haack 1987). Drought stress often makes forests prone 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).

In general, forests are relatively 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). The duration of a drought event may be more important than its intensity (Archaux and Wolters 2006); for instance, multiple consecutive years of drought (2-5 years) are more likely to cause high tree mortality than one 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 multi-year time windows.

In the 2010 FHM national report, we presented a methodology for mapping drought conditions across the conterminous United States (Koch and others 2013). Our goal with this methodology was to generate drought-related spatial data sets that are finer scale than similar products available from sources such as the National Climatic Data Center (2007) 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. In this chapter, we apply the methodology to the most currently available climate data (i.e., the monthly PRISM data through 2012), thereby providing a fourth time step in an ongoing annual record of drought status in the conterminous United States from 2009 forward (Koch and others 2013a, 2013b, 2014).

METHODS

We acquired monthly PRISM grids for total precipitation, mean daily minimum temperature, and mean daily maximum temperature for the conterminous United States from the PRISM group Web site (PRISM Group 2013). At the time of these analyses, gridded data sets were available for all years from 1895 through 2012. However, the grids for December 2012 were only provisional versions (i.e., the PRISM group had not yet released a finalized grid 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 km^2). 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 km^2) 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), 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):

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

where

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

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

T_m = the mean temperature for month *m* in 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 the empirical derivation of *a*]

To implement equation 1 spatially, we created a grid of latitude values for determining the *L* adjustment for any given grid cell (and any given month) in the conterminous United States. We calculated the mean monthly temperature grids as the mean of the corresponding PRISM daily minimum and maximum monthly temperature grids.

Moisture Index Maps

We used the precipitation *(P)* and *PET* grids to generate baseline moisture index grids for the past 100 years (i.e., 1913–2012) for the conterminous United States. We used a moisture index, *MI'*, described by Willmott and Feddema (1992), with the following form:

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

where

P = precipitation

PET = potential evapotranspiration

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

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

For each year in our study period (i.e., 1913–2012), 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 2012 is the mean of the monthly *MI′* grids from January to December 2012, while the MI_3' grid is the mean of the grids from January 2010 to December 2012 and the MI_5' grid is the mean of the grids from January 2008 to December 2012.

Annual and Multi-year 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:

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

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., 1913–2012)

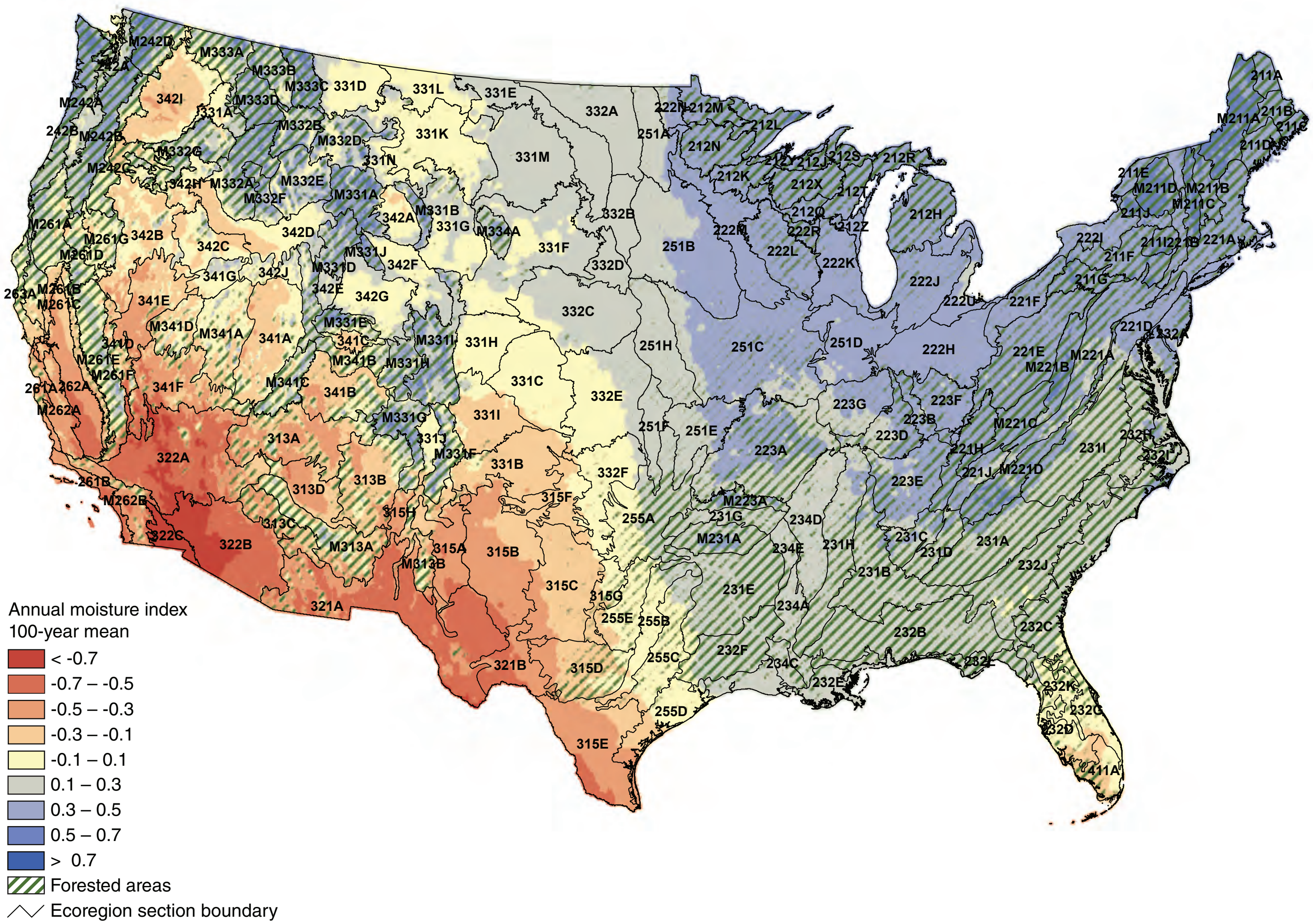

Figure 4.1—The 100-year (1913–2012) mean annual moisture index, or $\mathrm{MI}_1{}'_{\mathrm{norm}}$*, for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries and labels are included for reference. Forest cover data (overlaid green hatching) derived from Moderate Resolution Imaging Spectroradiometer (MODIS) imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: PRISM Group, Oregon State University)*

MDZ scores may be classified in terms of degree of moisture deficit or surplus (table 4.1). The classification scheme is composed of the same categories (e.g., severe drought, extreme drought) as those used in the Palmer Drought Severity Index (Palmer 1965) and widely adopted for other drought indices. Although the breakpoints between the categories in table 4.1 are defined somewhat arbitrarily, they yield theoretical frequencies of occurrence for each category that are comparable to the frequencies observed with other indices, especially the Standardized Precipitation Index (see table 4.2 in Koch and others 2012a). Importantly, because of the standardization in equation 3, the breakpoints between categories remain the same regardless of the size of the time window of interest. For comparative analysis, we generated classified *MDZ* maps of the conterminous United States, based on all three time windows, for the target year 2012. 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.

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

MDZ score	Category	Frequency
		%
<-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

RESULTS AND DISCUSSION

The 100-year (1913–2012) mean annual moisture index, or $MI_1'_{norm}$, grid (fig. 4.1) offers a general overview of climatic regimes in the conterminous United States. (The 100-year $MI_3'_{norm}$ and $MI_5'_{norm}$ grids did not differ substantially from the mean $MI_1'_{norm}$ grid and are not shown here.) Wet climates ($MI' > 0$) are common in the Eastern United States, particularly the Northeast. A noteworthy exception 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 partially driven by large amounts of winter snowfall in these regions.

Figure 4.2 shows the annual (i.e., 1-year) *MDZ* map for 2012 for the conterminous United States. Most of the Central United States, including much of the Great Lakes and Southwest regions, experienced at least mild drought conditions during 2012. Most prominently, the map displays a large contiguous area of extreme drought (*MDZ* < -2) extending from the northwestern portion of the Great Plains and into the eastern portion of the central and northern Rocky Mountains. Much of this contiguous area is spread across ecoregion sections that are partially or sparsely forested, such as 331I–Arkansas Tablelands, 332C–Nebraska Sandhills, 332D–North Central Great Plains, and 331F–Western Great Plains. However, it also extends into more heavily forested sections such as M331H–North Central Highlands and Rocky Mountains, M331I–Northern Parks and Ranges, and M334A–Black Hills.

Beside this large contiguous area of extreme drought, there were a few additional “hot spots” of severe to extreme drought (*MDZ* < -1.5) in the central portion of the country. The first of these spanned the southern portion of the Great Lakes region, extending from section 251B–North Central Glaciated Plains in the West to 222U–Lake Whittlesey Glaciolacustrine Plain in the East, although the affected area is only sparsely forested. Another hot spot included forested portions of sections 223A–Ozark Highlands, M223A–Boston Mountains, 231G–Arkansas Valley, M231A–Ouichita Mountains, and 255A–Cross Timbers and Prairie, as well as the sparsely forested sections 251E–Osage Plains and 251F–Flint Hills. A third hot spot occurred in the Southwest, primarily in sections 313C–Tonto Transition, M313A–White Mountains-San Francisco Peaks-Mogollon Rim, M313B–Sacramento-Monzano Mountains, and the sparsely forested section 313D–Painted Desert.

Overall, 2012 was a very dry year relative to historical data. The percent area of the conterminous United States with moderate or worse drought conditions according to the U.S. Drought Monitor peaked at 65.5 percent in September, which was a record in the 13-year history of the Drought Monitor (National Climatic Data Center 2013). Similarly, the percent area of the country in moderate or worse drought according to the Palmer Drought Severity Index reached 61.8 percent in July, representing the highest recorded percentage since December 1939 (National Climatic Data Center 2013). These record-setting extents are clearly reflected in the 2012 annual *MDZ* map (fig. 4.2). Indeed, the areas of the conterminous United States that experienced a moisture

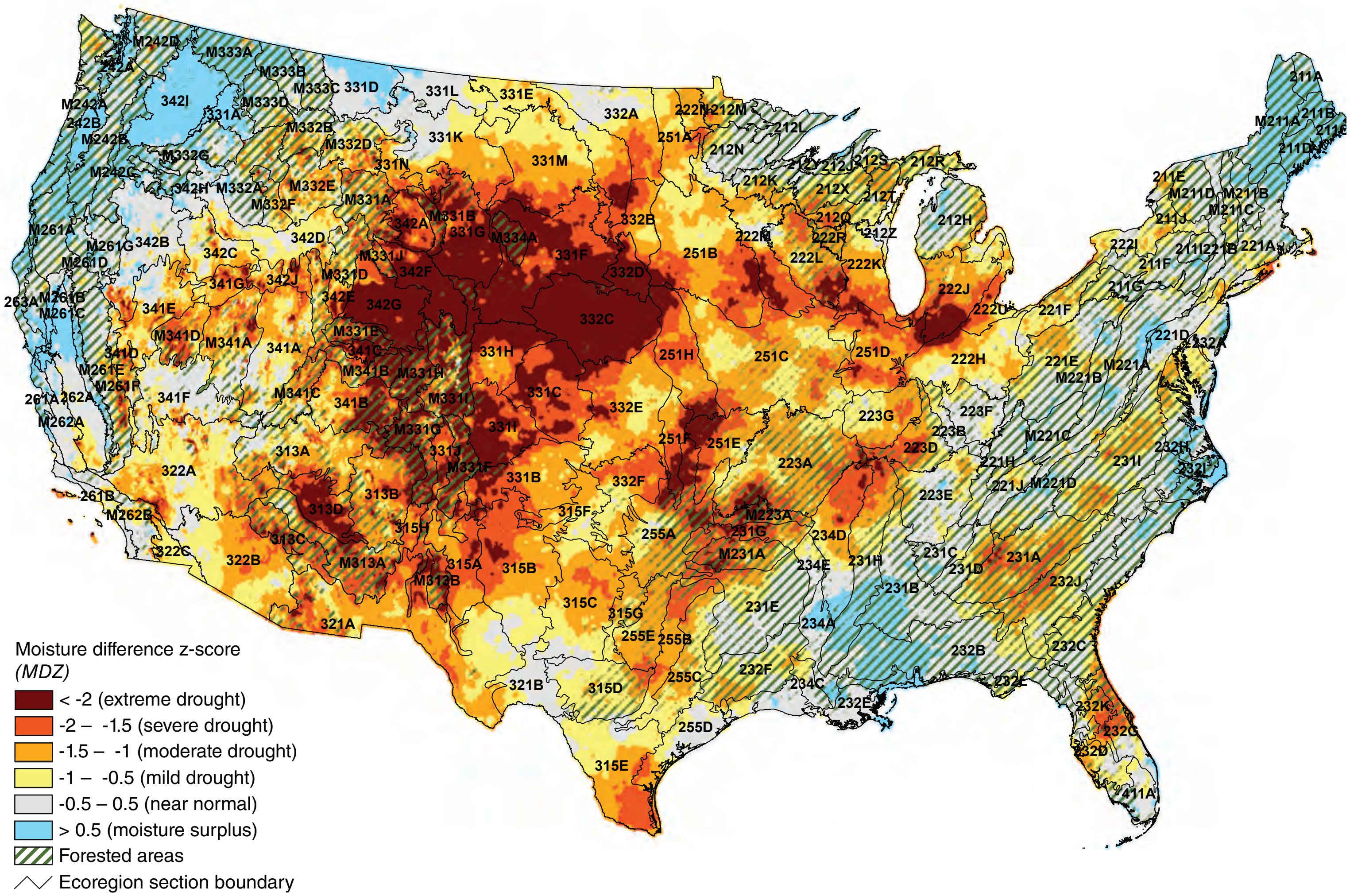

Figure 4.2—The 2012 annual (i.e., 1-year) moisture difference z-score, *or* MDZ, *for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries and labels are included for reference. Forest cover data (overlaid green hatching) derived from Moderate Resolution Imaging Spectroradiometer (MODIS) imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: PRISM Group, Oregon State University)*

surplus in 2012 were primarily limited to a small portion of the Southeastern United States along the Gulf of Mexico, eastern North Carolina (portions of sections 232H–Middle Atlantic Coastal Plains and Flatwoods and 232I–Northern Atlantic Coastal Flatwoods), New England, as well as the Pacific Northwest and northern California.

Figure 4.3 shows a map of the change in *MDZ* category between 2011 and 2012 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 2011 to extreme drought in 2012, 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 2011 to extreme drought in 2012.

Most of the aforementioned areas of the Central United States that were in extreme drought in 2012 displayed a five- or four-category decrease in *MDZ* from 2011 (fig. 4.3). This represents a dramatic decline from surplus or near normal moisture conditions in just 1 year. Conversely, an area near the Gulf of Mexico, particularly in eastern Texas and Louisiana, displayed a three- to five-category increase in *MDZ*. Both of these States were historically dry in 2011, and also experienced record high temperatures in the summer months (National Climatic Data Center 2012). Fortunately, it appears that these conditions abated substantially by the following year. Another area in the Southeast, primarily in the part of section 232I that falls in eastern North Carolina, displayed a similarly large improvement in moisture conditions between 2011 and 2012.

The 3-year (fig. 4.4) and 5-year (fig. 4.5) *MDZ* maps illustrate the recent history of moisture conditions in the conterminous United States. For instance, the Southwestern United States has been regularly subject to intense and widespread droughts for more than two decades (Groisman and Knight 2008, Mueller and others 2005; National Climatic Data Center 2010, 2011; O'Driscoll 2007). The persistence of these conditions is partially reflected in the 3-year and 5-year *MDZ* maps, which both show numerous areas of severe to extreme drought in this region. In fact, the 5-year *MDZ* map displays more extensive or severe drought conditions in the Southwest than the 3-year map. This difference likely reflects a short-term temporal fluctuation in a long-term pattern of persistent drought for the region. Additionally, the 3-and 5-year *MDZ* maps show that severe to extreme drought conditions are persistent elsewhere in the West, such as a relatively small area

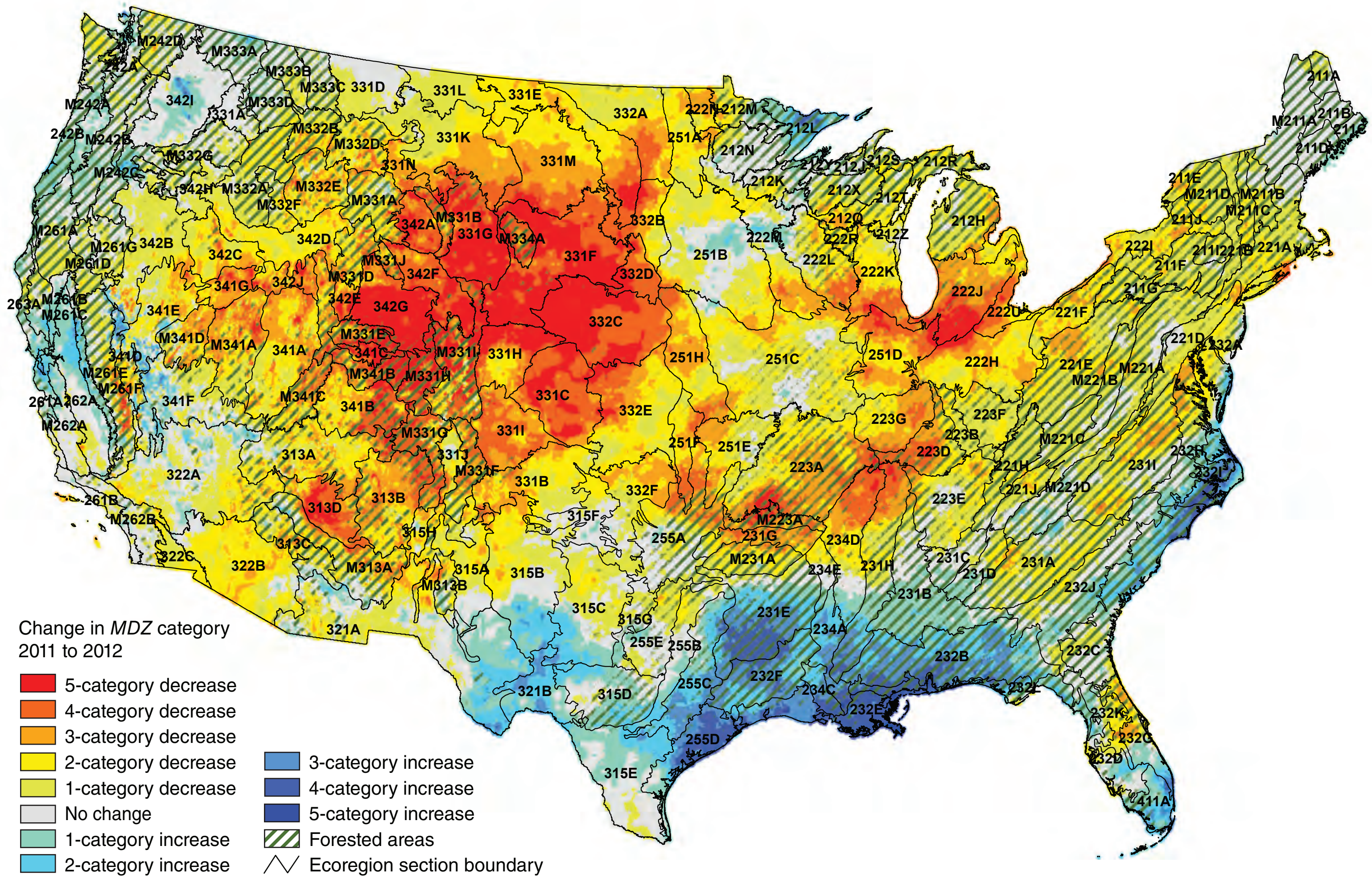

Figure 4.3—Change in moisture difference z-score *(*MDZ*) category between 2011 and 2012. See table 4.1 for a list of the* MDZ *categories used in this analysis; a five-category decrease indicates a change from moisture surplus in 2011 to extreme drought in 2012, while a five-category increase indicates a change from extreme drought in 2011 to moisture surplus in 2012. 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. Forest Service Remote Sensing Applications Center. (Data source: PRISM Group, Oregon State University)*

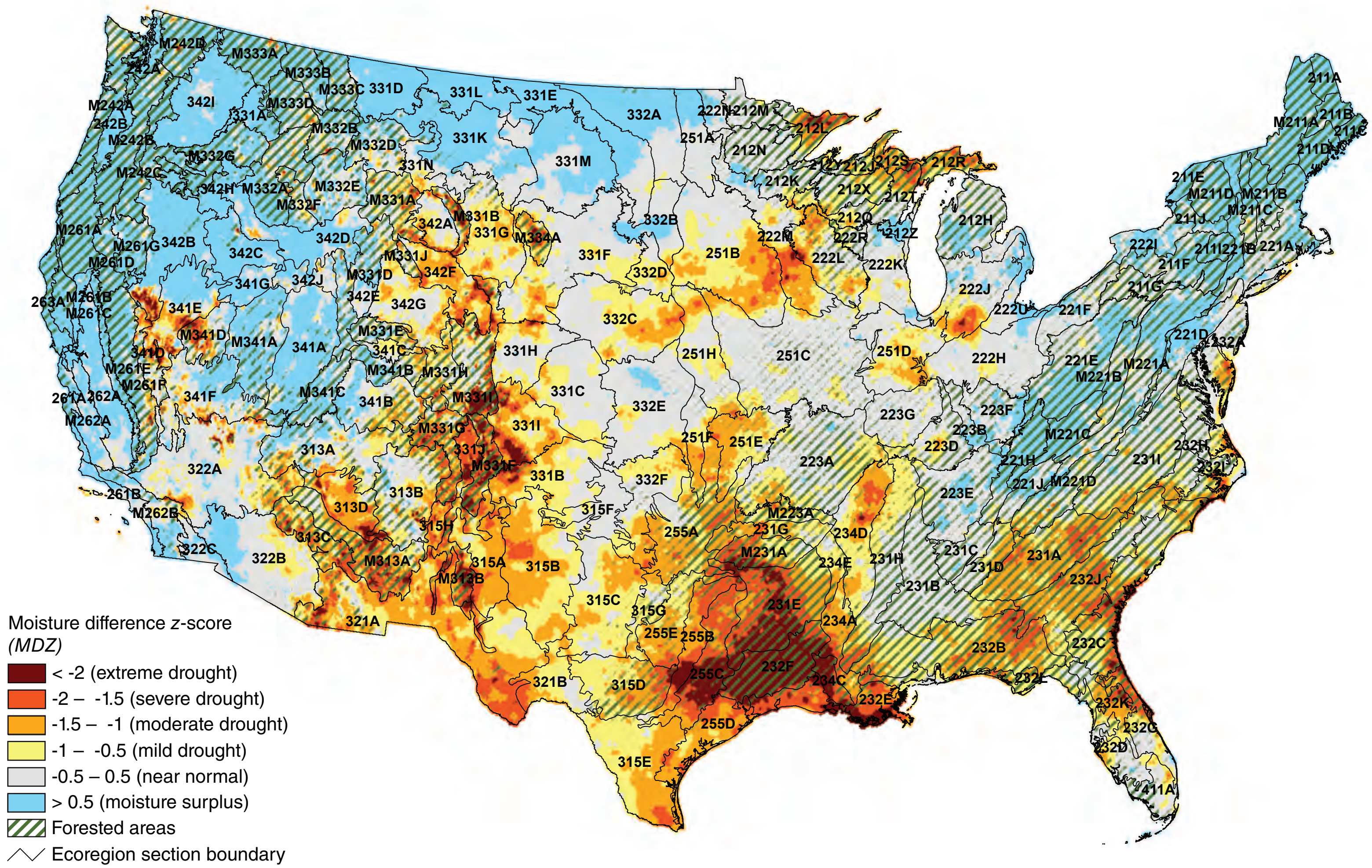

Figure 4.4—The 2010–12 (i.e., 3-year) moisture difference z-score (MDZ) for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries are included for reference. Forest cover data (overlaid green hatching) derived from Moderate Resolution Imaging Spectroradiometer (MODIS) imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: PRISM Group, Oregon State University)

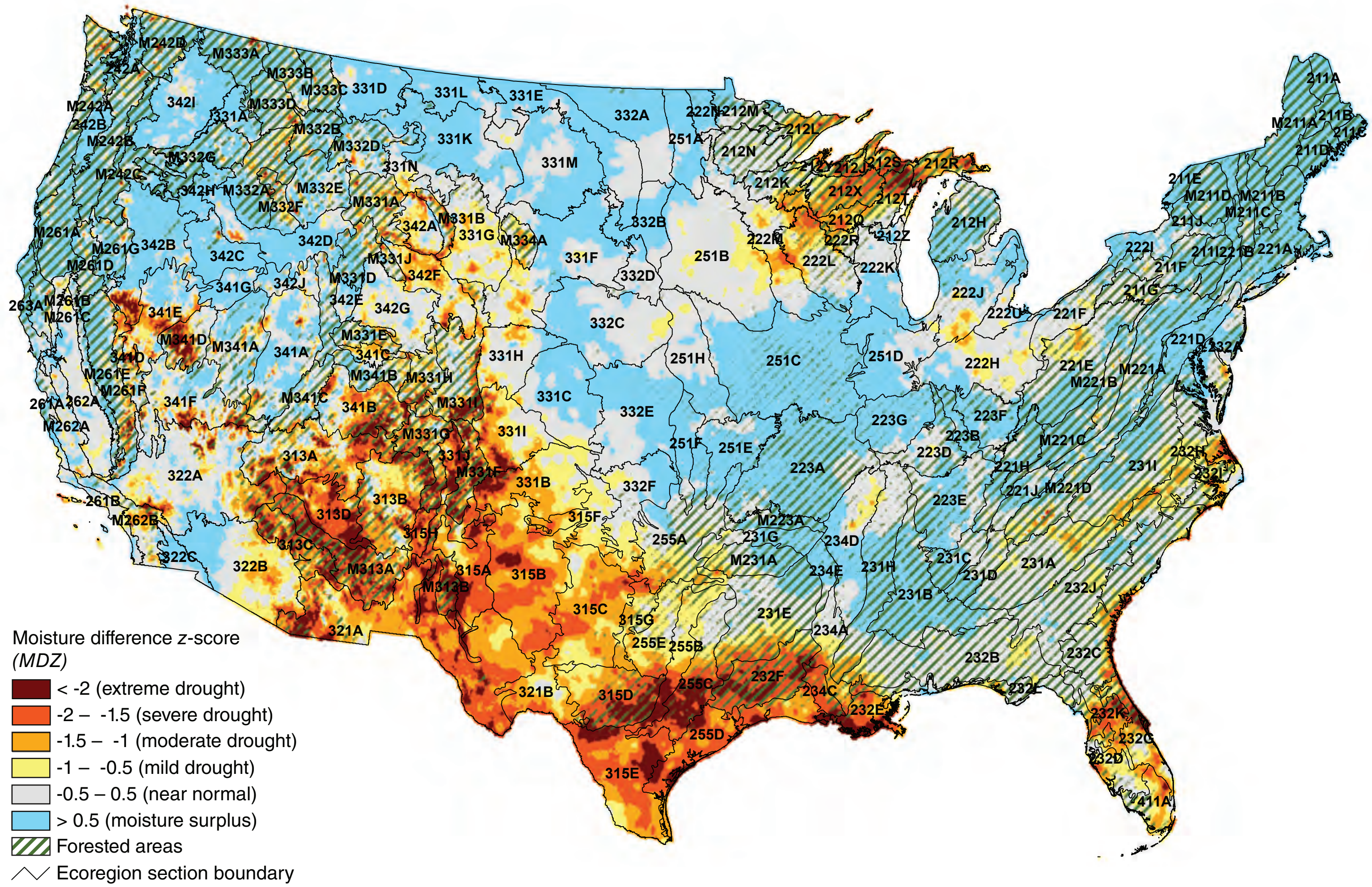

Figure 4.5—The 2008–12 (i.e., 5-year) moisture difference z-score (MDZ) for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries are included for reference. Forest cover data (overlaid green hatching) derived from Moderate Resolution Imaging Spectroradiometer (MODIS) imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: PRISM Group, Oregon State University)

consisting of portions of sections 341D–Mono, 341E–Northern Mono, 342B–Northwestern Basin and Range, and M341D–West Great Basin and Mountains. However, only the latter two sections have substantial forest in drought-affected areas.

The 3-year *MDZ* map (fig. 4.4) displays some influence of the major drought event that affected the Central United States in 2012 (see fig. 4.2), with several areas of severe to extreme drought occurring in the northern Rocky Mountain and Great Plains regions. While the latter region contains few such areas in the 5-year *MDZ* map (fig. 4.5), numerous small pockets of moderate to extreme drought (*MDZ* < -1.5) still appear in the northern Rocky Mountains, suggesting that drought has been fairly persistent at a local scale in this region. Because the region's forests may not be as well adapted to drought as those in the Southwest, these persistent conditions may represent a more immediate threat to forest health.

Regardless, the 3-year map's most pronounced feature is a sizeable area of extreme drought near the Gulf of Mexico, especially in sections 231E–Mid Coastal Plains-Western and 232F–Coastal Plains and Flatwoods-Western Gulf. This area also displays severe or extreme drought conditions in the 5-year *MDZ* map. Notably, this is largely the same area that, in figure 4.3, showed a substantial improvement in moisture conditions between 2011 and 2012, which should have positive implications for affected forests. Likewise, a drought hot spot in the upper Great Lakes region (i.e., in ecoregion section 212L–Northern Superior Uplands) that is clearly visible in both the 3-year and 5-year *MDZ* maps may have been partially counteracted by a moisture surplus in 2012 (see figs. 4.2 and 4.3). Unfortunately, moisture conditions do not appear to have improved as dramatically in other parts of the Great Lakes region. For instance, for sections 212R–Eastern Upper Peninsula, 212S–Northern Upper Peninsula, 212T–Northern Green Bay Lobe, and 212X–Northern Highlands, severe to extreme drought conditions occupy a smaller area in the 3-year *MDZ* map than in the 5-year map. Nonetheless, while severe to extreme conditions occupy even less area in the 1-year *MDZ* map (fig. 4.2), mild to moderate drought conditions extend almost entirely throughout these four ecoregion sections.

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* maps presented here cautiously. Although the maps use a standardized index scale that remains consistent 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 reasonably comprehensive short-term overview, it may also be important to consider a particular region's longer-term pattern of moisture deficit or surplus when assessing the current health of its forests. In future work, we hope to provide forest managers and other decisionmakers with better quantitative evidence regarding some of these critical relationships between deviations in moisture availability and forest health.

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CHAPTER 5.

Tree Mortality

Mark J. Ambrose

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 (USDA Forest Service 2003). 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 discernible. 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 being 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:

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, 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 2014). 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 in order to observe emerging trends or sudden shifts that may indicate forest health problems.

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

Table 5.1—States from which repeated Forest Inventory and Analysis phase 2 measurements were available, 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 3 times
1999–11	IN	1 plot: 6,000 acres	2/5
1999–11	ME	1 plot: 6,000 acres	3/5
1999–11	WI	1 plot: 3,000 acres[a]	2/5
1999–12	MN	1 plot: 3,000 acres[a]	3/5
1999–12	MO	1 plot: 6,000 acres[b]	3/5
2000–11	PA, VA	1 plot: 6,000 acres	2/5
2000–12	IA	1 plot: 6,000 acres	3/5
2000–12	MI	1 plot: 2,000 acres[c]	3/5
2000–12	AR	1 plot: 6,000 acres	2/5
2001–11	OH	1 plot: 6,000 acres	0
2001–11	TX[d]	1 plot: 6,000 acres	3/5
2001–11	GA, KS, NE, TN	1 plot: 6,000 acres	1/5
2001–11	LA	1 plot: 14,000 acres	0
2001–12	AL	1 plot: 6,000 acres	0
2001–12	IL, ND, SD	1 plot: 6,000 acres	2/5
2002–11	FL	1 plot: 10,000 acres	0
2002–11	KY, SC	1 plot: 7,500 acres	0
2002–11	NY	1 plot: 7,500 acres	0
2002–12	NH	1 plot: 6,000 acres	0
2003–11	CT, MA, RI, VT	1 plot: 7,500 acres	0
2003–11	NC	1 plot: 14,000 acres	0
2004–11	DE, MD, NJ, WV	1 plot: 10,000 acres	0
2006–12	MS	1 plot: 10,500 acres	0
2008–11	OK[e]	1 plot: 15,000 acres	0

[a] In Minnesota and Wisconsin, the phase 2 (P2) inventory was done at twice the standard Forest Inventory and Analysis Program (FIA) sample intensity, approximately 1 plot per 3,000 acres.

[b] In Missouri, the P2 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 P2 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.

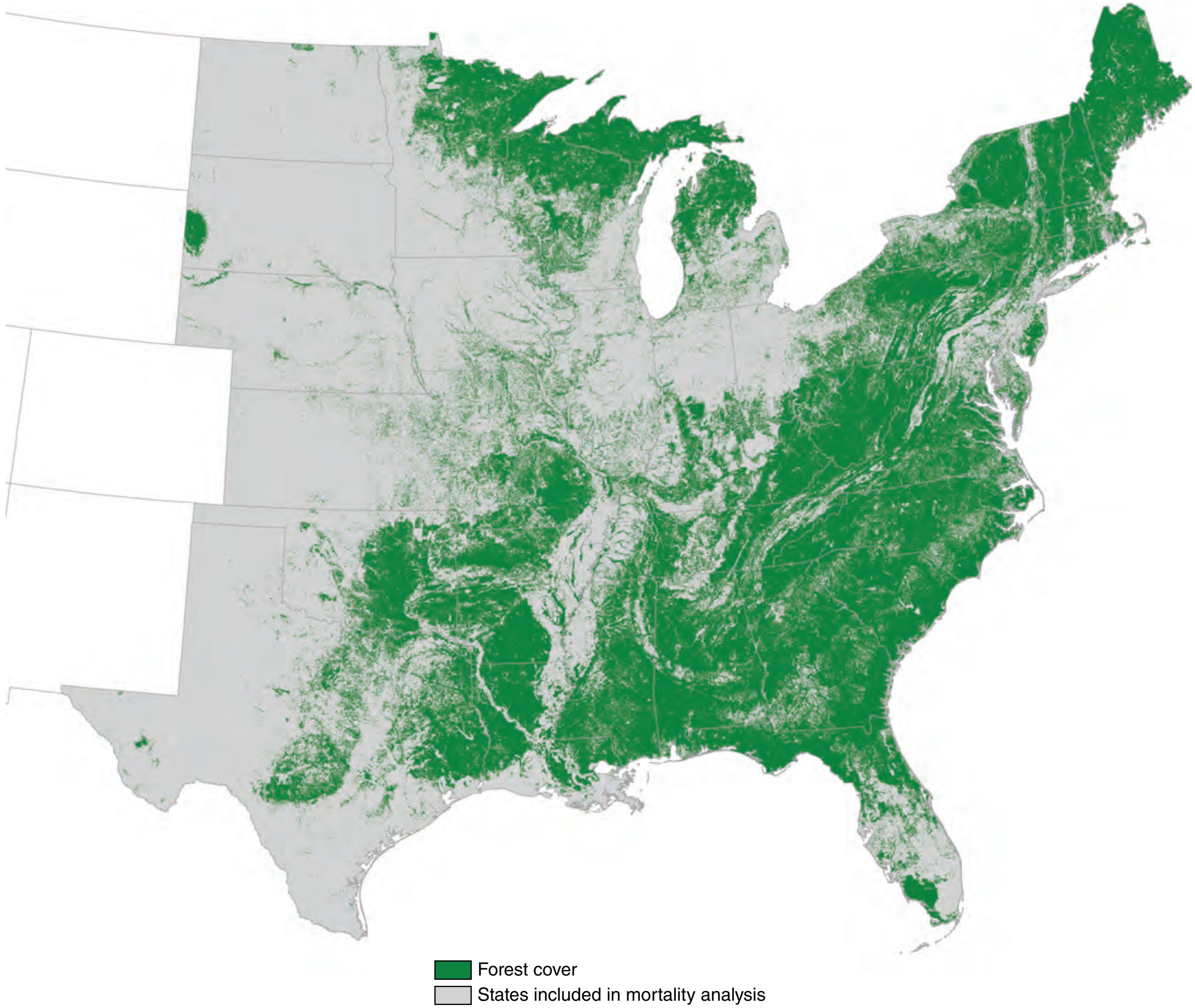

Figure 5.1—Forest cover in the States where mortality was analyzed. Forest cover was derived from Moderate Resolution Imaging Spectroradiometer (MODIS) satellite imagery (USDA Forest Service 2008).

METHODS

The methods used in this analysis were originally developed for earlier FHM national reports (2001–2004) using FIA phase 3 (P3) data. In this report, FIA P2 tree [≥ 5 inches diameter at breast height (d.b.h.)] and sapling (1 inch ≤ 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 2013). The biomass represented by each tree was calculated by FIA (USDA Forest Service 2011). 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 when using data where not all plots have been measured over identical time intervals (Gregoire and others 1995). In the estimation procedure, within-plot temporal correlation was modeled using a Toeplitz matrix. MRATIOs were then calculated from the growth and mortality rates. For details on the method, see Appendix A–Supplemental Methods in *Forest Health Monitoring 2001 National Technical Report* (Coulston and others 2005c) and Appendix A–Supplemental Methods in *Forest Health Monitoring 2003 National Technical Report* (Coulston and others 2005a).

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

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

In addition, 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 stratified mixed stands (i.e., stands consisting of multiple cohorts of different ages). When the age of trees that die is relatively low compared to 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 = 1.42) and 332C–Nebraska Sand Hills (MRATIO = 1.40) in South Dakota and Nebraska, 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 =0.65), M334A–Black Hills (MRATIO = 0.87) in South Dakota, and 234A–Southern Mississippi Alluvial Plain in Louisiana, Mississippi, and Arkansas (MRATIO = 0.86). 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 DDLD ratio, are shown in table 5.3. The DDLD 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 DDLDs were rather close to one, meaning that the trees that died were similar in size to the trees that survived. However, there were some plots with extremely high DDLD values. This same pattern of mean and median DDLD being close to one with some high DDLD 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, 332C–Nebraska Sand Hills, and 332D–North-Central Great Plains), the predominant vegetation is not forest land, but rather grassland (see the forest cover in

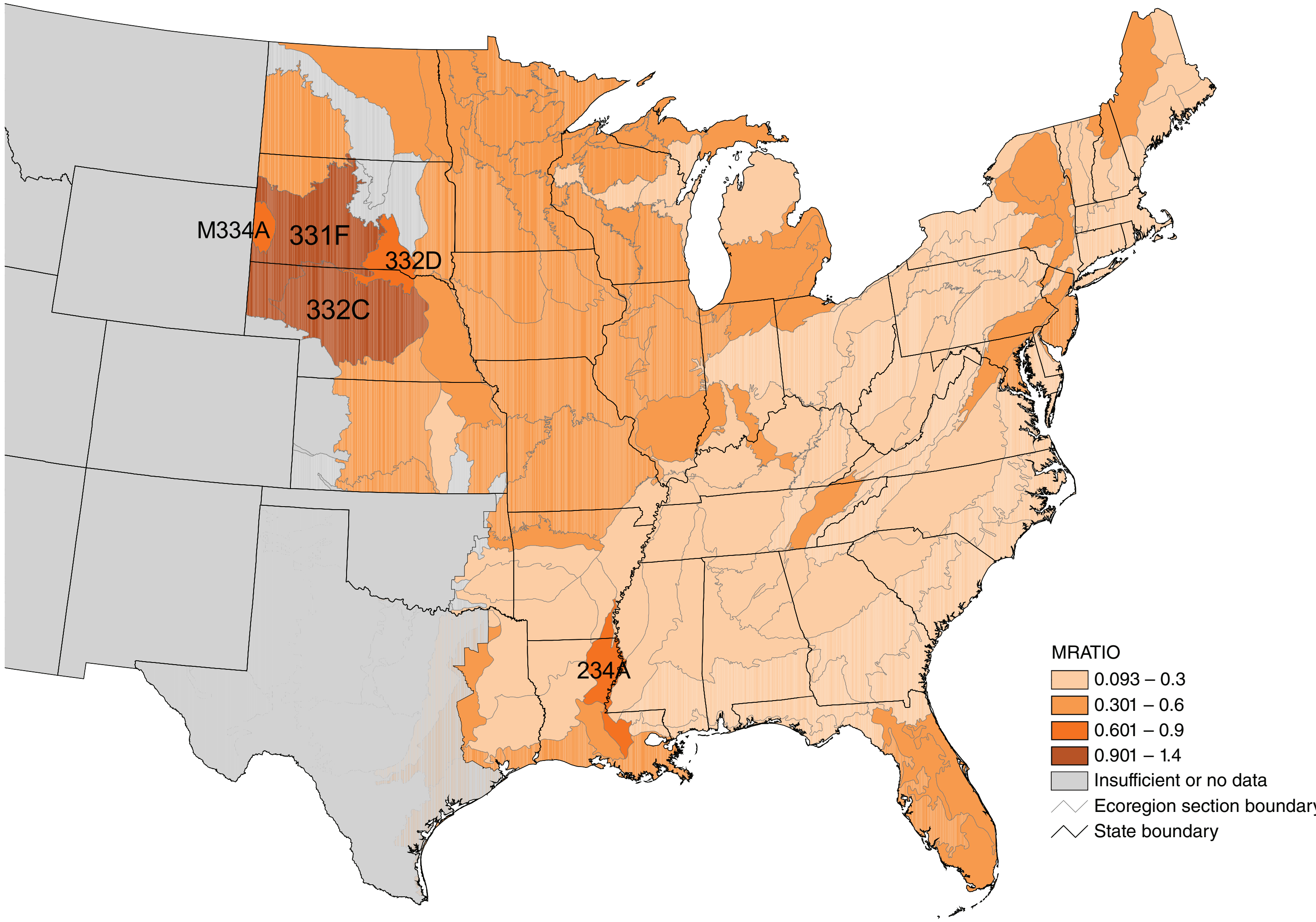

Figure 5.2—Tree mortality expressed as the ratio of annual mortality of woody biomass to gross annual growth in woody biomass (MRATIO) by ecoregion section (Cleland and others 2007). (Data source: U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis Program)

Table 5.2—Tree species comprising at least 5 percent of the mortality (in terms of biomass) for ecoregions where the MRATIO was ≥ 0.60

Ecoregion section	MRATIO	Tree species	Percent of total ecoregion mortality biomass	Mean age of dead trees[a]	Species percent mortality	
					Biomass	Stems
				years		
234A–Southern Mississippi Alluvial Plain	0.86	Black willow *(Salix nigra)*	36.49	38	52.66	66.05
		Water oak *(Quercus nigra)*	8.88	65	10.67	20.73
		Sugarberry *(Celtis laevigata)*	7.45	52	6.53	7.93
		Green ash *(Fraxinus pennsylvanica)*	6.99	50	11.30	14.35
		Eastern cottonwood *(Populus deltoides)*	6.00	54	23.40	16.89
		Swamp chestnut oak *(Quercus michauxii)*	5.88	63	43.64	80.00
331F–Western Great Plains	1.42	Ponderosa pine *(Pinus ponderosa)*	67.04	52	8.53	10.70
		Green ash *(Fraxinus pennsylvanica)*	14.37	44	13.86	12.45
		Eastern cottonwood *(Populus deltoides)*	8.18	79	3.87	7.69
332C–Nebraska Sand Hills	1.40	Eastern cottonwood *(Populus deltoides)*	40.08	56	55.21	33.86
		Green ash *(F. pennsylvanica)*	14.61	52	15.14	14.09
		Eastern redcedar *(Juniperus virginiana)*	12.45	40	6.91	21.22
		American elm *(Ulmus americana)*	6.43	54	22.01	31.88
332D–North-Central Great Plains	0.65	Ponderosa pine *(Pinus ponderosa)*	25.45	44	24.90	34.60
		American elm *(U. americana)*	20.55	49	22.75	25.22
		Bur oak *(Quercus macrocarpa)*	17.85	61	3.57	4.63
		Green ash *(F. pennsylvanica)*	12.69	62	15.17	18.00
		Hackberry *(Celtis occidentalis)*	10.59	60	11.24	0.72
		Eastern redcedar *(J. virginiana)*	6.51	37	4.22	6.53
M334A–Black Hills	0.87	Ponderosa pine *(Pinus ponderosa)*	61.28	22	19.80	45.58
		Quaking aspen *(Populus tremuloides)*	5.22	16	28.83	59.51

MRATIO= ratio of annual mortality of woody biomass to gross annual growth in woody biomass.
[a] Ages are estimated from the stand age as determined by the Forest Inventory and Analysis 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 (DDLD) ratios for ecoregion sections where the MRATIO was ≥0.60

Ecoregion section	Mean DDLD	Maximum DDLD	Median DDLD	Minimum DDLD
234A–Southern Mississippi Alluvial Plain	0.97	3.72	0.77	0.18
331F–Western Great Plains	0.98	3.29	0.91	0.08
332C–Nebraska Sand Hills	1.16	6.75	0.87	0.16
332D–North-Central Great Plains	0.93	2.17	0.91	0.29
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.

fig. 5.1), and subsequently there were relatively few forested plots measured (98 plots in region 331F, 85 plots in region 332C, and 57 plots in region 332D). Both ecoregions 331F and 332D have had high mortality relative to growth in recent years (Ambrose 2013, 2014), 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 forest, 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 mortality in ecoregion section 331F–Western Great Plains. Ponderosa pine is not a riparian species, but like the riparian tree species, it occurs in a relatively small area of the ecoregion, only on discontinuous mountains, plateaus, canyons, and breaks in the plains (Burns and Honkala 1990).

DDLD values vary widely within each of these sections. There are a small number of plots with high DDLDs, 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 comparatively low, as would be the case when remnant larger trees die, leaving younger 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). In section M334A, this mortality represented nearly half of the ponderosa pine stems and nearly 20 percent of the biomass. There has been an ongoing mountain pine beetle (*Dendroctonus ponderosae*) outbreak in the Black Hills (South Dakota Department of Agriculture 2011, 2012), so this pine mortality is very likely related to the outbreak.

In the adjacent ecoregion section 331F, where the MRATIO was highest, ponderosa pine also made up the vast majority of trees that died (67 percent), but this mortality represented a relatively small proportion of the ponderosa pine (biomass and stems) in the region. Scattered mountain pine beetle-related mortality has been reported in the Wildcat Hills and Pine Ridge areas of western Nebraska (Nebraska Forest Service 2010, 2011), which are part of this ecoregion. Green ash, although with an ecoregion mortality less than one quarter that of ponderosa pine, suffered a slightly larger proportional loss of the total ash stock.

In ecoregion section 332D–North-Central Great Plains, six species experienced the highest total mortality in terms of biomass and together represent over 90 percent of the mortality in the ecoregion: ponderosa pine, American elm, bur oak, green ash, hackberry, and eastern redcedar (table 5.2). Of these, ponderosa pine and American elm made up the largest proportion of total mortality and suffered the largest proportional loss in terms of both biomass and number of stems. There is not a lot of ponderosa pine in this region, and much of it is located in shelterbelts. The pine mortality is mostly related to three factors: over-mature trees, drought, and *Diplodia* tip blight. Many of the pines that died were 50 to 100 years old, which is quite old for this species when growing out of its native range and in the harsh environment of the Great Plains. In 2011 and 2012, the region experienced a severe drought that may be a cause of much of the mortality in the region, including that of ponderosa pine. Finally, *Diplodia* tip blight, which has been widely reported in shelterbelts in South Dakota (South Dakota Department of Agriculture 2011) was a third stressor, which finally killed the already severely stressed pines.[2] The elm mortality is probably related to Dutch elm disease, which is reported to be a problem throughout Nebraska (Nebraska Forest Service 2010, 2011). In the case of hackberry, the mortality in terms of biomass (11.24 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.

In ecoregion section 234A–Southern Mississippi Alluvial Plain, a number of hardwood species experienced high mortality, including black willow, water oak, sugarberry, green ash, swamp chestnut oak, and eastern cottonwood. The cause of mortality in this wide range of species is not immediately obvious. However, willow, oak, and cottonwood are among the genera preferred by the forest tent caterpillar (*Malacosoma disstria*), which has affected parts of Louisiana and Arkansas in recent years (Arkansas Forestry Commission 2011; Louisiana Department of Agriculture 2009, 2010), so this defoliator may have played some role in the observed mortality. In addition, the growth and mortality of trees in flood plains can be strongly

[2] Personal Communication. 2013. John Ball, South Dakota Dept. of Agriculture, Resource Conservation and Forestry Division, 523 East Capitol, Pierre, SD 57501.

affected by river and groundwater levels. Thus, the observed mortality may be related to either flooding or drought. This may warrant further investigation.

The mortality patterns shown in these analyses do not immediately suggest large-scale forest health issues. Mortality is relatively low in most of the areas for which data are available. The areas of highest mortality occur in the mostly riparian forests of Great Plains ecoregions. A characteristic of most of these Great Plains ecoregions with high mortality is that they are on the margins of land suitable for forest growth. As a result, the implications of the high mortality are unclear. Trees growing in these marginal situations may be especially susceptible to new or changed biotic or abiotic stressors. Because of the small number of forested plots used to analyze these ecoregions, it is difficult to determine whether the mortality is localized or more widespread. Therefore, further study of the health of these forests may be warranted.

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SECTION 2.

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

CHAPTER 6.

Monitoring Forest Disturbances across Seasons Using the *ForWarn* Recognition and Tracking System

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William W. Hargrove
Joseph P. Spruce
William M. Christie

INTRODUCTION

National-scale satellite-based forest monitoring can provide uniform and timely insights into forest health. Monitoring across jurisdictions satisfies a basic need, since disturbances such as insects and diseases, wildfire, or severe weather do not respect Federal, State or local boundaries. Monitoring across seasons at high frequency satisfies a second basic need, because early detections can affect the choice of actions that managers take, such as altering the progress of a defoliator or deciding where to prioritize post-disturbance remediation.

These two strengths of satellite-based monitoring—cross-jurisdictional uniformity and high-frequency detection—do not come without effort. One of the greatest challenges for development of a high-frequency change detection product is knowing what to expect from healthy forests in terms of a meaningful baseline normal, since such expectations change from place to place, throughout the year, and according to the various needs of land managers. With an appropriate baseline in place, clouds, snowpack, and the seasonal effects of variations in temperature and precipitation may still cause anomalies that appear similar to damage caused by insects and disease, wildland fire, extreme weather, and other disturbances. Actual change recognition often requires not only maps of change, but an integration of knowledge that includes ancillary datasets presented in an accessible way.

This chapter highlights recent monitoring results from *ForWarn*, a satellite-derived change detection system operating across the conterminous United States that can be accessed at http://forwarn.forestthreats.org (Hargrove and others 2009). *ForWarn* is a joint effort of the U.S. Department of Agriculture Forest Service's Eastern and Western Threat Assessment Centers and NASA-Stennis Space Center that is designed to monitor and interpret all types of forest change. *ForWarn* has been largely funded by the National Forest System in response to a Congressional mandate to develop an early warning system for forests as part of the Healthy Forests Restoration Act of 2003.

THE SATELLITE-BASED TECHNOLOGY

Since January 2010, the *ForWarn* system has been used to detect environmental threats to forests caused by insects and disease, wildfires, extreme weather, and other natural and man-made events (see Norman and Hargrove 2012, Norman and others 2013, and Spruce and others 2011 for case studies). *ForWarn* disturbance detections rely on changes in the timing of vegetation "greenness" as measured by the Normalized Difference Vegetation Index (NDVI) derived from the MODIS (Moderate Resolution Imaging Spectroradiometer) satellite sensors. NDVI compares wavelengths in the red and infrared range to measure and track changes in the health status of vegetation. Seasonal periodicities in NDVI reflect the collective leaf phenologies of plants on the land surface, while departures from this natural rhythm denote stress or disturbance (Spruce and others 2011).

Phenology can be thought of as both a driver and a response variable. Acting as a driver, phenology controls ecosystem functions and services like productivity, leaf area, and biomass, and it is strongly correlated with vegetation structure, standing stocks, and carbon across a broad range of forested ecosystems. As a response variable, phenology is largely influenced by vegetation reactions to primary growing conditions such as climatic and edaphic factors, but localized changes in phenology can indicate disturbance. Because of these sensitivities, phenology has been broadly recognized as a useful indicator of ecosystem health and function.

ForWarn captures seasonal phenological change by dividing the year into 46 overlapping 24-day periods. Each period overlaps the next by 16 days to minimize cloud contamination (table 6.1). The maximum NDVI observed for each pixel in each period is used as the current greenness value for all but one of the *ForWarn* change products, while the most recent clear value is used for the *Early Detect* product. The *Early Detect* product may sometimes suffer from clouds, smoke, or haze, but it is better than other baselines for detecting disturbances quickly.

Forest disturbance and recovery can be rapid—appearing from one *ForWarn* period to the next—or they can occur gradually over the course of months or years. Our ability to detect all these changes requires a suite of baseline normals that capture different prior conditions. *ForWarn*'s baselines empower analysts to detect change relative to the prior year, the maximum of the last three years, and the maximum since January of 2000, when MODIS first became available. Some landscapes experience considerable variability in NDVI from year to year due to climatic variation, and that seasonal variability can make the use of simple maximums problematic for detecting disturbance during seasonal transitions. *ForWarn* addresses this problem by including two additional baselines that define normal from the average conditions at each period over time. One baseline uses the average of each site's maximum NDVI since 2000, and the other combines phenologically similar neighbors for a landscape average condition during each interval through time. When baselines are carefully selected and used comparatively, forest disturbances from fast- or slow-acting insects

Table 6.1—Basic characteristics of *ForWarn* forest disturbance products

Attribute	Value
Cell spatial resolution	231.7 m (5.4 ha)
Spatial extent	Conterminous United States
Temporal window length	1 to 24 days
Product frequency	8 days (46 NDVI values per year)
Seasonal coverage	Year round
Years of *ForWarn* change products	January 2010 to present
Years of baseline data	January 2000 to present

NDVI = Normalized Difference Vegetative Index.

and disease, flooding, hail, wind, or wildfire can be distinguished from the effects of extreme interannual variation in temperature and precipitation (table 6.2).

ForWarn products can be viewed by anyone using the online *Forest Change Assessment Viewer* (http://forwarn.forestthreats.org/fcav). The *Assessment Viewer* contains all current and historical *ForWarn* maps, along with co-registered maps of insect and disease outbreaks, wildfire perimeters, and other relevant information on disturbances, vegetational cover types, terrain, hydrography, land ownership, and climate. Users can also click on any pixel to obtain a pop-up graph showing the entire NDVI greenness history of vegetation for that location. The ability to quickly review past forest performance at any location shows evidence of many disturbances, and aids greatly in interpretation and attribution of the causative agents. Such NDVI time-series graphs are included in several figures here. During the growing season, the *ForWarn* Team uses the *Assessment Viewer* to examine potential forest disturbances detected in each new set of *ForWarn* products, alerting Federal, State, and local forest managers when warranted.

Table 6.2—*ForWarn* change detection products and their suggested uses

Product name	Current state	Baseline state	Suggested use
Early Detect	Most recent clear view (1 to 24 days)	Prior year's 24-day maximum for one site	Used to detect initial change for new disturbances
1 Year	Maximum of 24-day window	Prior year's 24-day maximum for one site	Used to detect disturbance and recovery < 1 year old when the prior year had normal seasonal weather
3 Year Max	Maximum of 24-day window	Maximum of 24-day maximums of 3 prior years' values for one site	Used to detect disturbances that are < 3 years old when the prior 3 years had normal seasonal weather
All Year Max	Maximum of 24-day window	Maximum of 24-day maximums since 2000 for one site	Used to detect slow or sequential disturbances and to monitor recovery relative to the greenest conditions ever observed
All Year Mean of Maximums	Maximum of 24-day window	Mean of 24-day maximums since 2000 for one site	Used to isolate disturbance and current year weather effects on sites that are sensitive to year-to-year climate variability
Similar Neighbor Mean	Maximum of 24-day window	Mean of 24-day maximums since 2000 for phenologically similar sites	Used to isolate disturbance and current year weather effects on landscapes sensitive to year-to-year climate variability

Note: *ForWarn* works by comparing the current and baseline states of vegetation for corresponding periods of the year for each individual site.

CROSS-SEASONAL EXAMPLES

Change During the Growing Season

Detecting a drop in vegetation greenness at the peak of the growing season is straightforward for most forest types. Monitoring the condition of open stands with grassy or herbaceous understories or in areas with mixed land cover can be challenging, however, as non-woody vegetation is sensitive to variation in moisture. A reduction in forest NDVI may result from defoliation from insects, discoloration from disease, leaf damage or loss from wind or hail, deforestation, or combustion from fire. Drought can result in the broad-scale loss of vigor in the canopy or visible understory. NDVI can drop from either a decline in a portion of a site (e.g., scattered tree defoliation or partial deforestation) or from a more uniform decline across all vegetation at a site (e.g., drought or a uniform loss). The amount of decline reflects the severity of the stressor(s), given the limitations of the product's resolution and vegetation homogeneity.

ForWarn can detect and track certain native and nonnative insects and diseases that occur during the growing season. In the Malheur National Forest of central Oregon, *ForWarn* captured an outbreak of the native, but normally uncommon, pine butterfly (*Neophasia menapia*) that peaked in 2011. According to the national Insect and Disease Survey (IDS), defoliation in this National Forest is more usually caused by the western spruce budworm (*Choristoneura occidentalis*) and mountain pine beetle (*Dendroctonus ponderosae*), but not so in 2011.

A comparison of targeted aerial sketchmapping of pine butterflies with *ForWarn*'s 1-year change for September 29, 2011 shows close agreement (fig. 6.1).

Wildfires cause some of the steepest declines in NDVI, particularly in the Western United States, where large forest fires of high intensity are common. Prior to severe fire, dense evergreen trees are associated with high NDVI values with low interseasonal amplitude. After fire, an increase in grass dominance is often

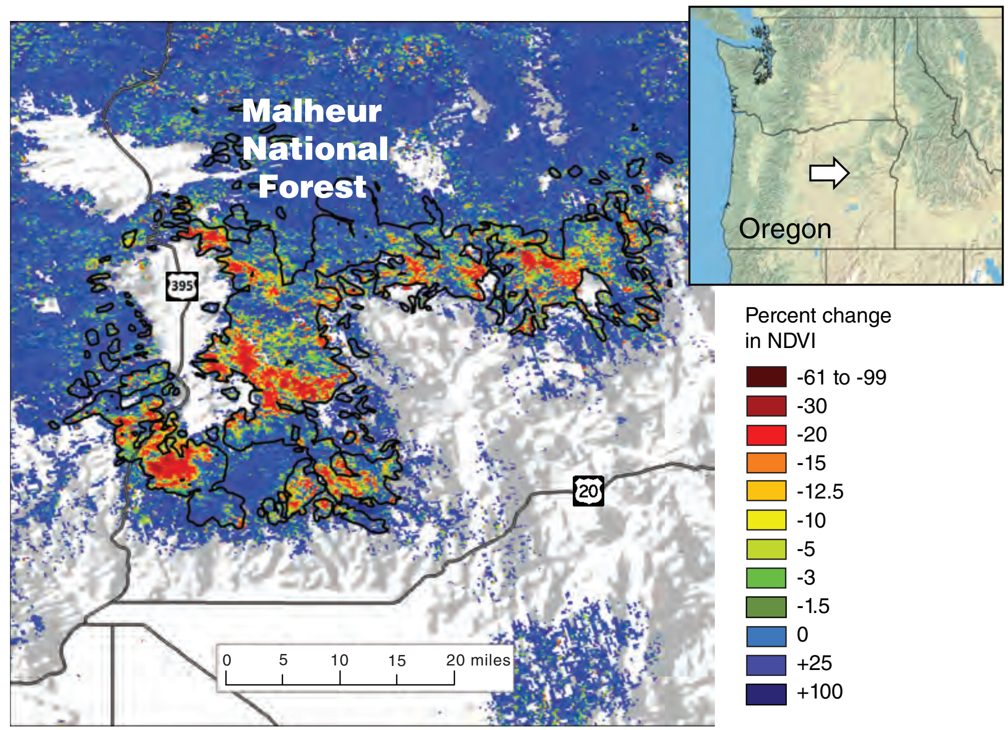

Figure 6.1—Areas defoliated by pine butterflies as mapped by aerial surveys (black outlines) compare well to ForWarn's *1-year change anomalies for the Malheur National Forest, Oregon, for September 29, 2011. NDVI= Normalized Difference Vegetative Index.*

indicated by lower NDVI and sharp growing-season peaks in the NDVI time series. In low burn-severity areas, a more limited drop in greenness occurs, and this shows the spatial heterogeneity in burn severity. *ForWarn*'s initial patterns of fire severity compare well with the higher resolution burn severity mapping efforts of the Forest Service's Remote Sensing Applications Center (RSAC).

ForWarn's long-term monitoring capabilities empower managers to monitor in a time frame that extends beyond immediate fire effects. For example, in the Gila National Forest of New Mexico, large wildfires are now reburning vegetation that previously burned during the MODIS period (fig. 6.2). In these forests, such frequent fires could reduce undesired fuel loads and stand densities, and restore desired

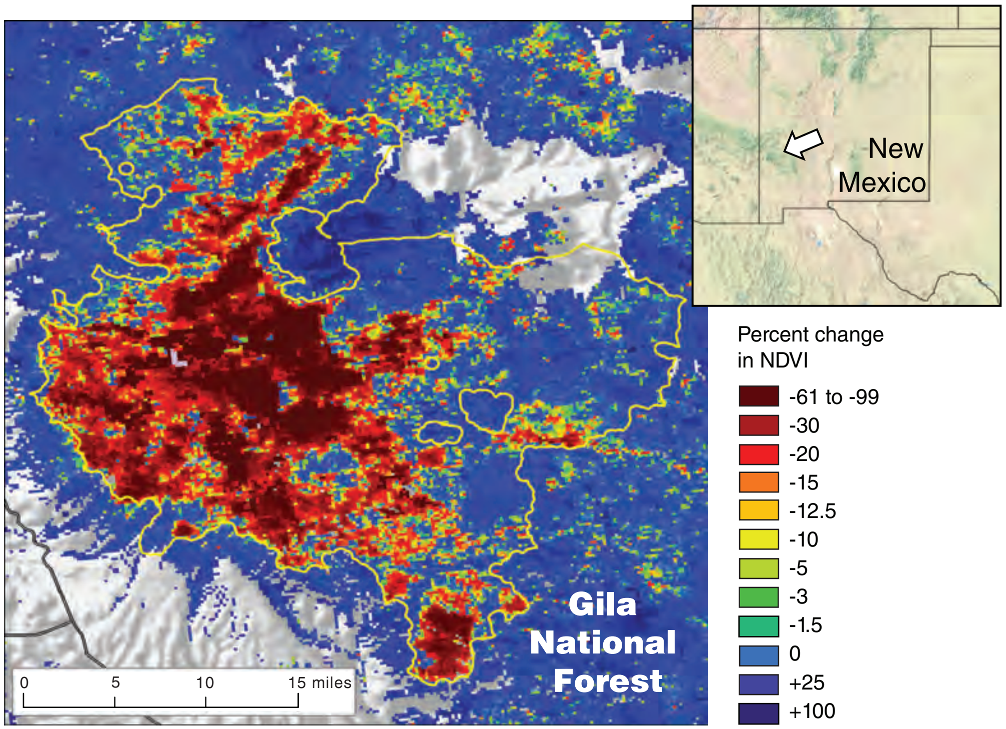

Figure 6.2—The severity of the Whitewater-Baldy Complex fire on the Gila National Forest, New Mexico, varied greatly. This ForWarn *map shows change as of August 3, 2012, using the 1-year baseline with the fire perimeter shown as a yellow line. The majority of the eastern portion of this fire, shown in blue, had burned in recent years, and that likely reduced the severity of this portion of the event. NDVI= Normalized Difference Vegetative Index.*

conditions overall, or they could contribute to sudden or incremental type conversion. This long-term condition can be measured by integrating observations of forest fire's immediate effects and the cross-seasonal pattern of NDVI recovery.

Drought stresses both woody and herbaceous vegetation, and these can be difficult to distinguish in open or mixed forests. The exceptional drought of 2011 in the southwestern United States resulted in the documented decline and mortality of trees in Texas, particularly in the western half of the State. These conditions also led to one of the most intense wildfire seasons ever experienced in this region. By distinguishing the degree of change, *ForWarn* captured the regional decline of vegetation due to drought and the even more extreme change from wildfire (fig. 6.3). While grass areas somewhat recovered during 2012, areas in which tree mortality was predominant experienced a sustained decline.

Change During Winter

In northern latitudes and high elevations, winter can be the most challenging time of year to detect disturbance, due both to the general absence of leaves on deciduous vegetation and to the episodic masking effects of snowpack. Heavy snow can blanket understory evergreen vegetation and, in some climates, can persist on conifer boughs for weeks. The resultant reduction in NDVI from this winter-to-winter snow variation can be very difficult to separate from actual damage to trees caused by severe winter weather or other agents. At lower elevations, lower latitudes, or during years where or when there is no persistent snowpack, snow effects are not a problem because of *ForWarn*'s 24-day sampling period.

The most practical insights into winter change in NDVI come from mixed evergreen–deciduous forest types and how they change over multiple years. Pure evergreen forests can be effectively monitored from above at any time of year, but change in the evergreen fraction of a mixed evergreen–deciduous forest may be most apparent when deciduous vegetation no longer dilutes the NDVI signal. In the southern Appalachians, eastern and Carolina hemlock (*Tsuga canadensis* and *T. caroliniana*) are experiencing a rapid decline due to the nonnative hemlock woolly adelgid *(Adelges tsugae)*. In the NDVI signal, this decline shows up as a gradual reduction in the winter minimum

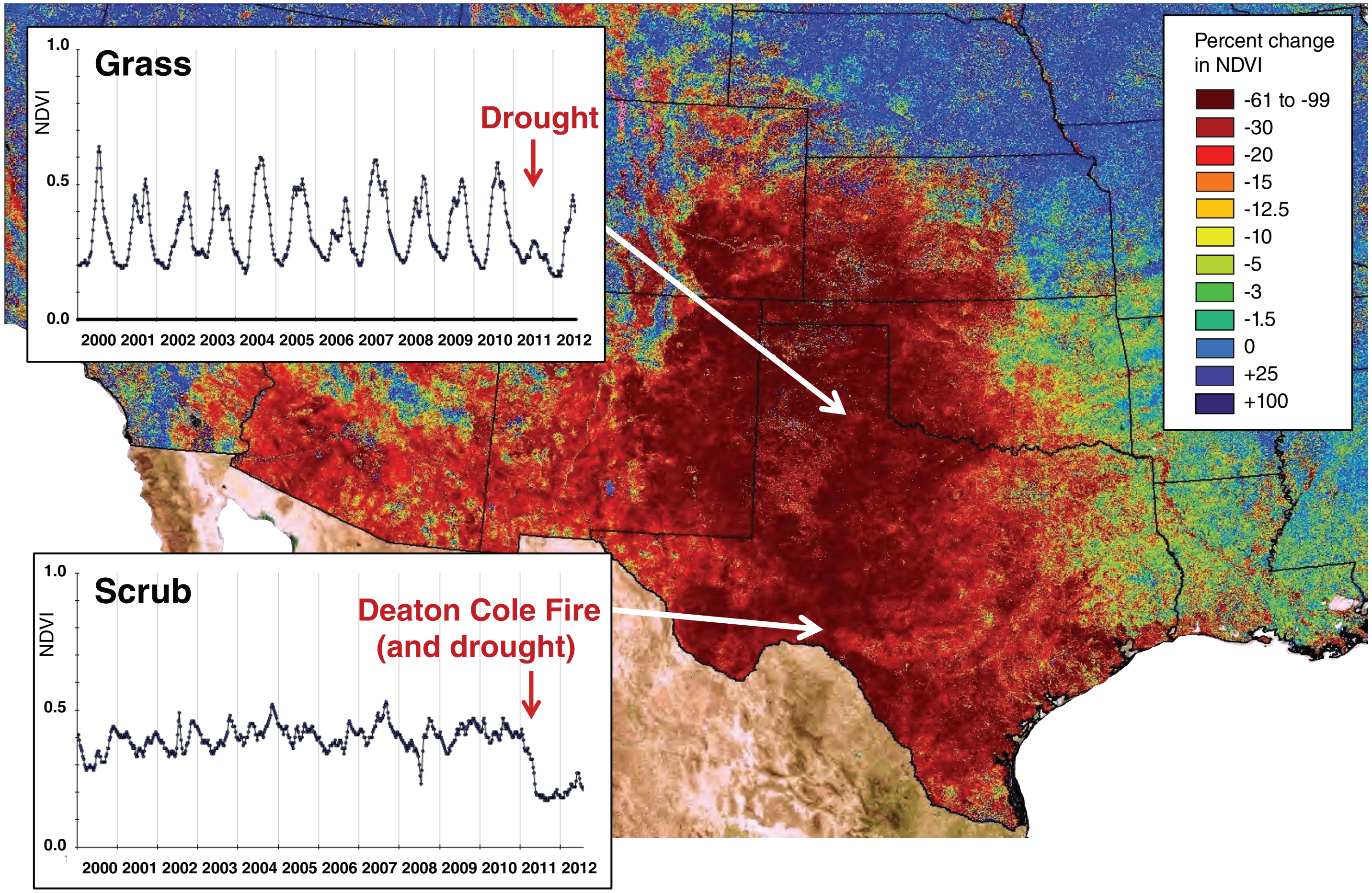

Figure 6.3—This ForWarn *map for July 11, 2011, shows regional departure from the all-year baseline that resulted from an exceptionally severe drought and localized wildfire. The graph insets show the 12.5-year Moderate Resolution Imaging Spectroradiometer (MODIS) history of two areas: at top, the effects of drought on grass are reflected by the strong annual drop in Normalized Difference Vegetative Index (NDVI); at bottom, the effects of wildfire on evergreen scrub resulted in a stronger and more sustained departure from the period of record.*

over a 4- to 6-year period, which is consistent with the time required for defoliation and tree mortality in this region (Vose and others 2013). The intensity of region-wide leaf-off trends in NDVI indicate that the hemlock-rich Cataloochee Valley of Great Smoky Mountains National Park ranks among the areas most significantly affected (fig. 6.4).

Rapid disturbances that occur during winter in deciduous forests are the most difficult to detect. Late winter wind, ice, and hail storms can damage branches and pre-emerged buds and reduce subsequent greenness. Such a storm occurred in western Virginia on March 24, 2012, the most intense portion of which formed

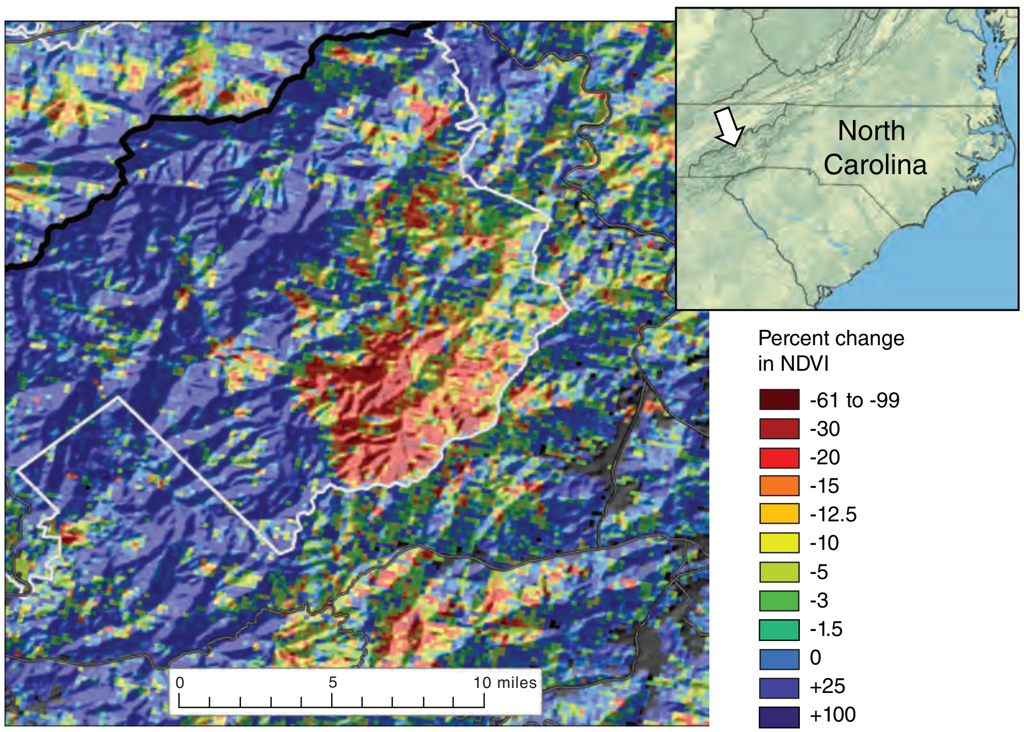

Figure 6.4—An epicenter of hemlock woolly adelgid mortality in the Southern Appalachians is the Cataloochee Valley of Great Smoky Mountains National Park, North Carolina. This ForWarn *map shows change on February 17, 2012, compared to the maximum value observed during the prior decade. That long-term baseline is necessary here because hemlock decline is a slow process that takes several years. NDVI= Normalized Difference Vegetative Index.*

a 1-mile wide track over Smith Mountain Lake, southeast of Roanoke (fig. 6.5). When they occur in the spring instead, such physical defoliation from storm events can be followed by rapid secondary leaf flushes, making the damage ephemeral. This Virginia hail event showed up clearly once canopy green-up occurred, apparently due to the severity of this portion of the storm.

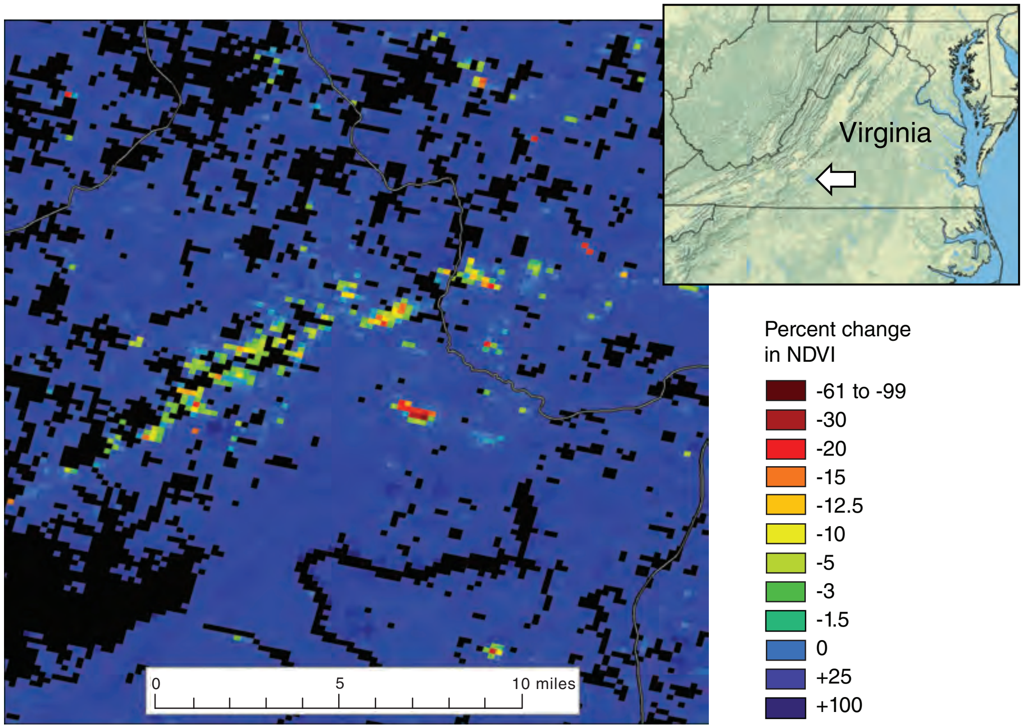

Figure 6.5—This hail storm damage in western Virginia occurred prior to green-up based on ground observations. The storm track became apparent by April 29, 2012 (1-year baseline). NDVI= Normalized Difference Vegetative Index.

Change During Spring Green-up and Fall Senescence

During the spring and fall, detecting forest change from disturbances can be challenging due to the variability in baseline conditions. If spring green-up and fall senescence were timed exactly the same across years this would not be a problem, but one to three weeks difference in spring green-up is not uncommon in temperate deciduous forests of the United States. Disturbances occurring during these seasonal transitions may be more likely to go unnoticed. Compare the variable progression of 2007, 2009, and 2012 for deciduous forests within Great Smoky Mountains National Park (fig. 6.6).

Using a 1-year baseline, *ForWarn* identified extreme hail damage during early May of 2012 in the Asheville Watershed—an ephemeral loss of leaves that occurred when leaves had only half emerged (fig. 6.7). The value of the 1-year baseline is also evident in NDVI profiles, such as that of the historical April 5-9, 2007 spring freeze event in the forests of western Kentucky that reduced NDVI in some areas, while it slowed the rate of green-up in others (figs. 6.6 and 6.8). This regional freeze caused widespread damage to crops and fruit trees (Gu and others 2008).

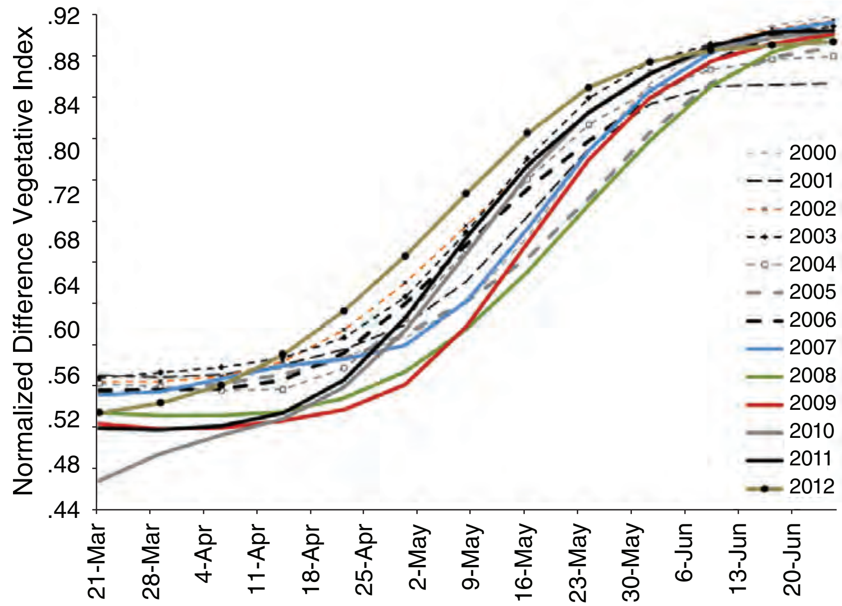

Figure 6.6—The onset and progression of spring varied considerably from 2000–12 across Great Smoky Mountains National Park in Tennessee and North Carolina. This graph shows the average behavior of spring vegetation for 18,030 Moderate Resolution Imaging Spectroradiometer (MODIS) pixels that were identified as pure deciduous forest using the 2006 National Land Cover Dataset. Note that 2012 experienced the earliest spring during the 13-year period of MODIS data collection.

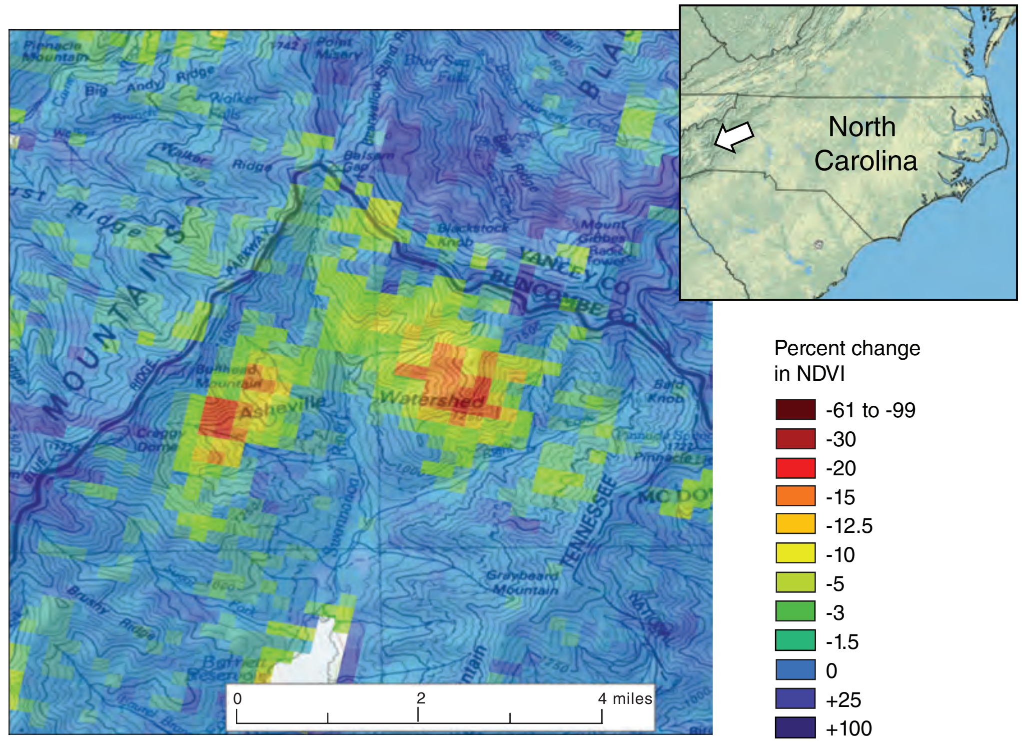

Figure 6.7—This spring hail damage near Asheville, NC, occurred when most deciduous leaves were half emerged, based on field observations. This damage went unnoticed until it was detected in ForWarn *despite this being a heavily monitored urban water source. This map shows departure on June 16, 2012, using the 1-year baseline. NDVI= Normalized Difference Vegetative Index.*

Relatively few insect defoliators are active in the fall in the Northeastern United States other than the fall webworm (*Hyphantria cunea*). These caterpillars cause leaf defoliation during the same months as natural seasonal NDVI decline. Because early leaf loss may occur normally following the early arrival of cold temperatures, fall defoliation can be difficult to detect. Despite this challenge, *ForWarn* captured two successive outbreaks in the western portion of the Allegheny National Forest, PA, in 2011 and 2012 (fig. 6.9).

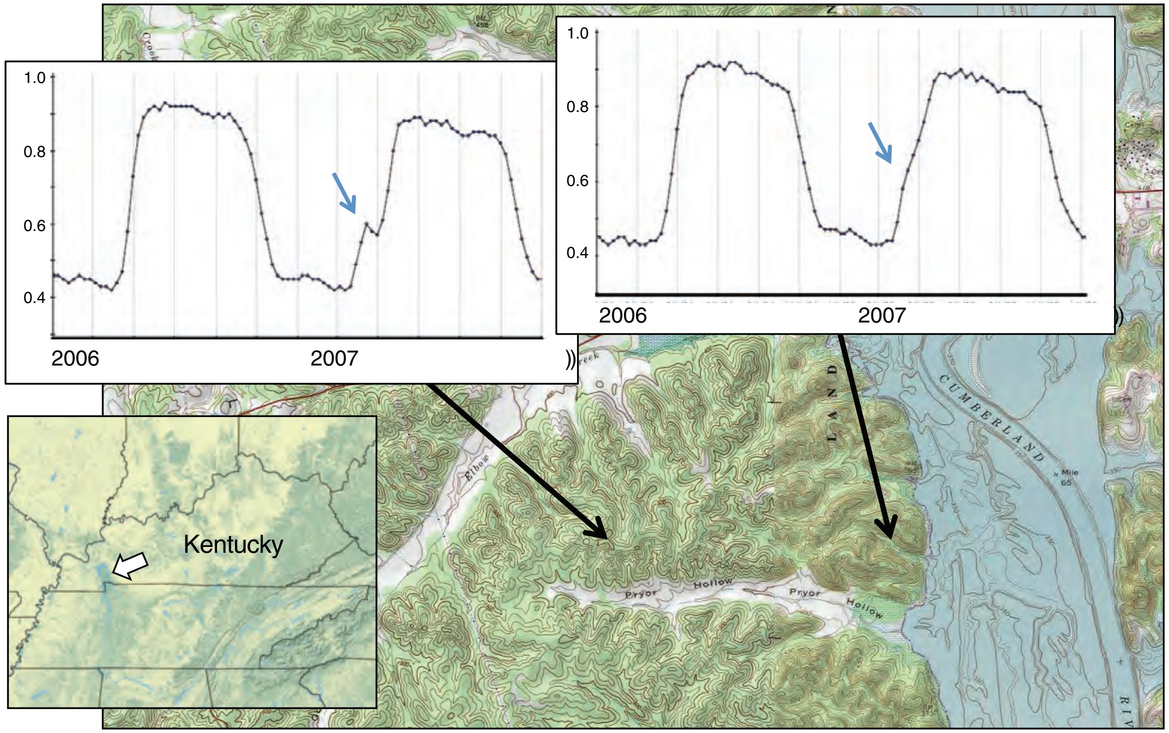

Figure 6.8—The variable effects of the April 5-9, 2007, spring freeze are shown here using Moderate Resolution Imaging Spectroradiometer-(MODIS) based Normalized Difference Vegetative Index profiles for two sites at Land Between the Lakes in Kentucky. These sites are roughly a mile and a half apart and suggest that the lake may have provided a partial temperature buffer. A normal profile for 2006 is included for comparison.

Fall is often a time of damaging storm events, such as hurricanes. Hurricane Sandy struck New Jersey, New York and New England on October 29, 2012 with severe, sustained winds. A comparison with the normal NDVI decline for all prior MODIS years indicates that these forests were roughly halfway through their fall decline, although that varied with forest type and location. Figure 6.10 shows change relative to the prior year after the 24-day rolling window excluded any pre-storm values. Note the coast-to-interior gradient, the variable mainland intensity with respect to Long Island and Cape Cod, and the linear streaks that conform to exposed ridgelines across New Jersey, upstate New York, and eastern Pennsylvania. These patterns are consistent with expectations of leaf loss and tree damage from wind across the landscape. The long-term persistence of such effects can help distinguish severe forest damage from the more ephemeral effects of wind-stripped leaves.

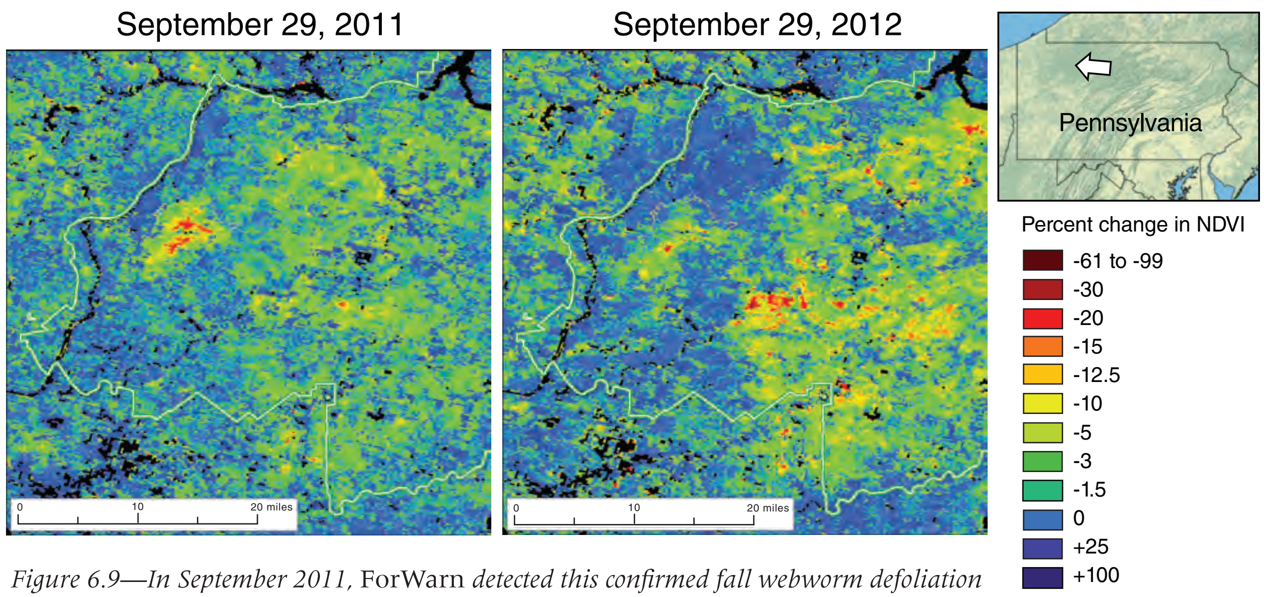

Figure 6.9—In September 2011, ForWarn *detected this confirmed fall webworm defoliation in the Allegheny National Forest using the 3-year baseline. A subsequent outbreak appears to have occurred to the southeast in the fall of 2012. These detections were made despite seasonal leaf decline. NDVI= Normalized Difference Vegetative Index.*

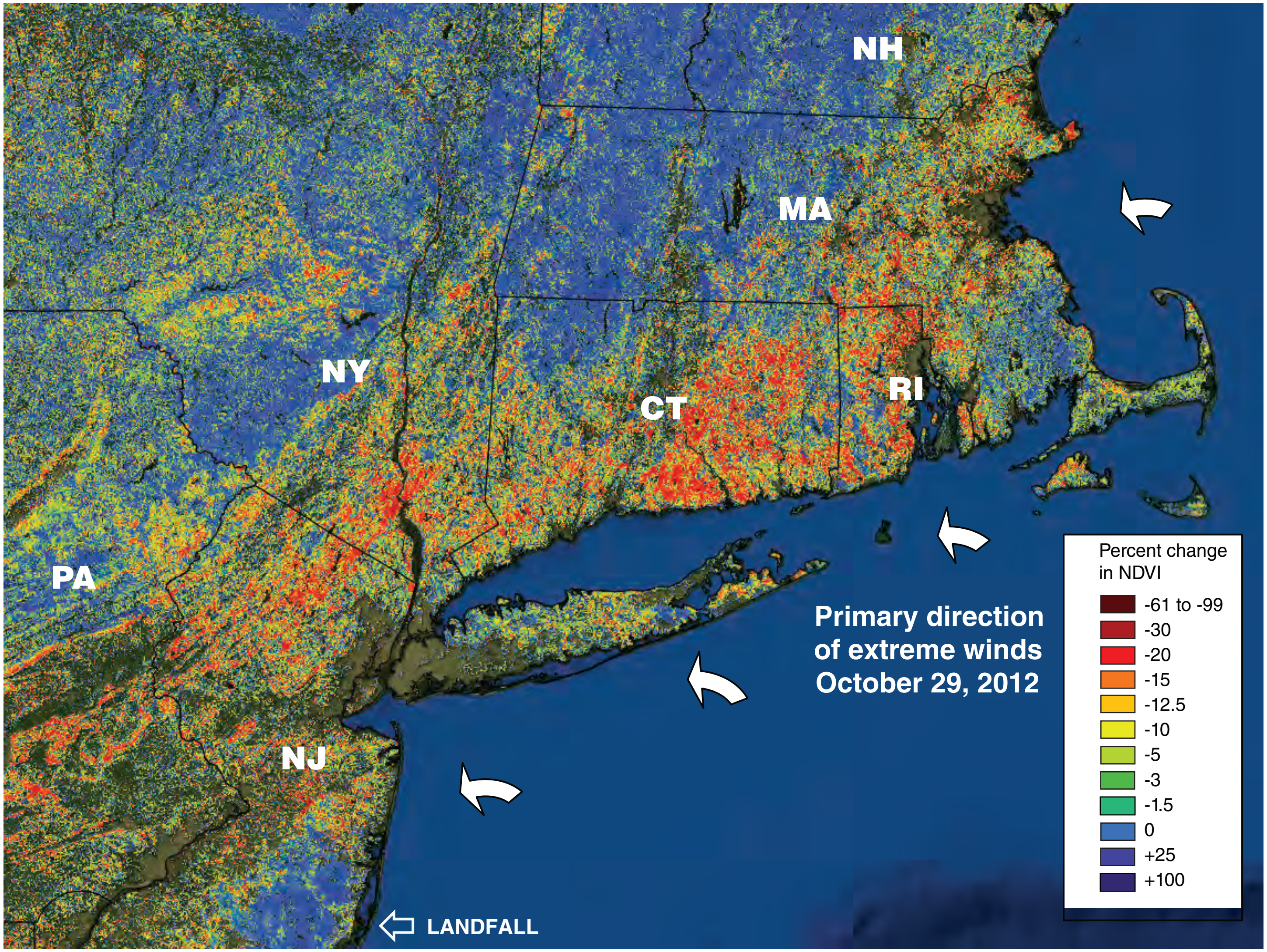

Figure 6.10—Hurricane Sandy struck the Northeast on October 29, 2012, with mixed effects to forests. The change in Normalized Difference Vegetative Index (NDVI) shown here soon after the event likely reflects accelerated fall leaf loss as much as more severely damaged trees. This ForWarn *map may not reflect damage to extensive conifer stands, such as the New Jersey Pine Barrens near landfall, as evergreen trees retain their needles after blowdown; however, tree loss could materialize during subsequent months as a change in winter greenness.*

DISCUSSION

Satellite-based forest monitoring in near real time presents fundamental challenges related to normal seasonal change and interannual climate variation from drought, snowpack, and the variable timing of spring and fall. *ForWarn* overcomes the problem of seasonal change by taking a phenology-based approach that includes multiple perspectives on what is expected for that time of year using a suite of baseline normals. This shifting seasonal sense of normal is analogous to the way the National Weather Service compares current monthly weather conditions to that of last year and to the average or record monthly values of the last thirty years or the prior century. Having this seasonally adjusted context is how *ForWarn* can detect the occurrence, severity, progression, and recovery of a broad range of disturbances within and across years.

ForWarn's ability to monitor and track forest recovery may be significant for aiding forest management in the future. *ForWarn*'s multiple baselines and cross-seasonal product lines provide a rich context for understanding the duration of disturbance effects and the cumulative effects of management in the months to years that follow. This ability to efficiently quantify the long-term consequences of disturbances has long evaded us, preventing the adoption of a more thorough risk-based approach for forest management. Forest managers are generally well informed about the likelihood of particular disturbances in their forests, such as insects and disease, logging, wildfire, or severe weather events, thanks to existing monitoring and extension. They are less well informed about how conditions have changed or recovered from a decade earlier. Having a more effective means to monitor, with both a short- and long-term perspective, can empower forest managers to recognize a broader set of concerns so they may better achieve their goals.

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CHAPTER 7.

Working towards an Integrated Approach for Monitoring Forest Disturbance at Multiple Scales and Frequency

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INTRODUCTION

Forest Health Protection and its cooperators have a long history of technological development. Many applications regarding pesticide application technology, data collection technology, pest modeling technology, and remote sensing technology have all been developed independently without regard for integration. While effective as stand-alone applications, the value of these individual technologies can be dramatically improved if developed in concert with or as part of an organized system. Integration of design with regard to efficiency and optimization of natural resource management can dramatically improve the utility of individual software programs, Web sites, databases, models, and monitoring activities. The purpose of this paper is to describe a conceptual organization of existing and future technologies that would support and ultimately improve the monitoring of forest health as well as provide a framework for developing new technologies.

FOREST DISTURBANCE AND FOREST HEALTH MONITORING ACTIVITIES

Forest disturbance refers to changes in the forest structure, composition, or density that have occurred either through natural, human-caused, or catastrophic events. Disturbance intensity can vary from low intensity over a small area to severe intensity over a large area. While disturbance is a natural process in healthy forests, patterns of disturbance—i.e., source, size, and frequency—are difficult to assess in relation to the range of natural variability.

Forest Health Monitoring (FHM) is a national program designed to determine the status, changes, and trends in specific forest damage agents, as well as indicators of forest condition. The present national program began as an amalgam of national and regional programs. As it has evolved, the emphasis has been on building consistency among regional programs and efficient integration of national activities. This conceptual paper describes recent and ongoing improvements to provide an integrated strategy for monitoring forest disturbance at multiple scales and frequencies.

FHM incorporates the following activities:

- Detection Monitoring (DM) is intended to be a nationally standardized aerial and ground inventory program to evaluate status and change in conditions of forest ecosystems.
- Evaluation Monitoring (EM) activities are used to determine extent, severity, and causes of undesirable changes in forest condition identified through DM.
- Intensive Site Monitoring (ISM) activities are used to enhance understanding of cause–effect relationships at multiple spatial scales by linking DM to studies of ecosystem processes and assessments of specific issues, such as calcium depletion and carbon sequestration.
- Research on Monitoring Techniques (RMT) activities are used 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.
- Analysis and Reporting (AR) activities are used to synthesize 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.

PREDISPOSED CONDITIONS—RISK/HAZARD ASSESSMENT

Disturbance can occur anywhere on the landscape; however, the likelihood of disturbance varies. A priori knowledge of disturbance agents along with knowledge of existing forest conditions can yield information that would be conducive to the monitoring of anticipated disturbances. A collection of information about known insect and disease disturbance agents, hosts, ranges, and environmental conditions has been developed, along with a suite of tools, to facilitate the production of risk/hazard assessments nationally.

Predisposed risk/hazard assessment involves a thorough understanding of the disturbance agents' hosts, range, and impacts. Until recently, there has been no organized effort to document pest ranges nationally. A simple query of the Web will produce multiple conflicting range maps for any given species. In 2010, the Forest Health Technology Enterprise Team (FHTET) began a project to develop an authoritative database of pest ranges. For a limited though ever-increasing number of pests, the outward expression of this database is the Forest Pest Conditions Portal (USDA Forest Service 2013a). This portal provides access to the Pest Range geospatial dataset and Pest Host database, which incorporates published literature and reported occurrences of individual pests. The portal also incorporates nationwide forest pest mapping and reporting on an annual basis. These maps, which locate specific pests on the landscape, are a fundamental basis for forest pest risk assessments.

2012 National Insect and Disease Risk Map

To aid in the development of risk and hazard assessments, the Risk Modeling Application (RMAP) was developed to support the development of risk models using state-of-the-art knowledge for individual pests. Data from the Spatial Data Library (SDL) were improved from various national datasets to provide a set of predictor layers for generating geospatial risk products. This suite of tools and data was used in the 2012–27 National Insect and Disease Forest Risk Assessment (Krist and others 2014), which identified the potential impact of both endemic and non-endemic forest pests in the conterminous United States, Alaska, and Hawaii. The collection of each individual agent–host risk assessment is an effort to provide a 5-year strategic appraisal of the risk of tree mortality due to major insects and diseases.

The National Insect and Disease Risk Map (NIDRM) report contains a nationwide strategic assessment of the hazard of tree mortality due to major insects and diseases, displayed as a series of maps. The risk in NIDRM is defined as the potential that, without remediation, 25 percent or more of the standing live basal area (BA) of trees >1 inch in diameter will die over the next 15 years due to insects and diseases.

The NIDRM represents the integration of 186 individual insect and disease hazard models, all constructed within a common GIS-based multi-criteria framework that can account for regional variations in forest health concerns. The 2012 modeling process, applied to all 50 States, provides a consistent, replicable, transparent, peer-reviewed process through which interactive spatial and temporal hazard assessments can be conducted. Each individual model is based upon the best science and data known to the developers. The modeling process allows for flexible analysis to produce hazard assessments for specific insects and diseases. For the underlying host tree conditions, the National Tree Species Extent and Parameter geospatial dataset was developed to support NIDRM and utilizes Forest Inventory and Analysis (FIA) field data along with predictor layers from the SDL and remotely sensed imagery. The NIDRM can support the prioritization of regional plans and can possibly be implemented to support forest and project plans for project-level support. In addition to NIDRM, individual pest species risk assessments have been developed for a number of anticipated and recently introduced invasive species (USDA Forest Service 2013c) such as Sirex woodwasp (*Sirex noctilio)* and emerald ash borer *(Agrilus planipennis)*.

The predisposed risk/hazard assessment details risk in relation to the health of the forest; however, the ability to observe forest health conditions can pose a risk to the observer. An assessment of aerial safety risk was developed as the Spatial Tool for Aviation Risk (STAR) project. The STAR project prioritizes the safety of areas based upon weighted factors of known pest risks, topography, land cover, and distance to landing sites. Coupled with NIDRM, STAR-based analysis may reveal that—due to landscape characteristics, road density, and other factors—changes to flight mission profiles, aircraft selection, or detection methods are required in order to ensure safe coverage of high priority forested landscapes.

MULTI-SCALED MONITORING—A SPATIAL CONTEXT

Given a priori knowledge of known disturbance agents, improvements in efficiency can be realized by creating targeted and more focused monitoring activities. To assist in the allocation of monitoring resources, inventory designs that alter the intensity and frequency of samples would achieve better utility of monitoring resources. The use of risk/hazard assessments can be used as a tool to focus and prioritize monitoring areas. The predisposing risk assessments can be considered an initial Tier 1a monitoring effort.

Risk assessments show *potential* for disturbance, while Detection Monitoring (DM) and Evaluation Monitoring (EM) measure actual disturbances. Different DM and EM activities can be organized hierarchically (fig. 7.1 and table 7.1). At the top of the spatial hierarchy, broad-scale disturbance detection monitoring is achieved through Synoptic Surveillance and can be referred to as Tier 1b monitoring. Using large-area monitoring systems, disturbances are mapped and tracked through periodic surveys. The monitoring of the extent, severity, and causes of undesirable changes is achieved through Focused Surveillance (Tier 2a monitoring). Directed Monitoring (Tier 2b monitoring) is used for fine-scale mapping to validate or invalidate positive detections from Tiers 1b and 2a. Project Assessment (Tier 3 monitoring) is needed to identify site-specific existing and desired conditions for project implementation (e.g., National Environmental Policy Act, pest management or State planning and analyses, decisions and implementation, and ecosystem restoration).

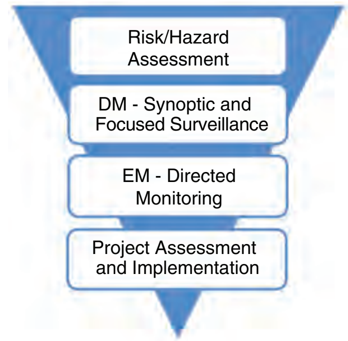

Figure 7.1—Hierarchical organization of Forest Health Monitoring and Forest Disturbance Monitor activities. DM= Detection Monitoring; EM= Evaluation Monitoring.

Table 7.1—Organizational hierarchy of Detection Monitoring and Evaluation Monitoring activities

	Tier	Decadal	Annual	Seasonal	Monthly	Daily	Hourly
	0-Risk Assess	National Insect and Disease Risk Map					
	1-Synoptic	Landscape change	Harvest and development				
Spatial	2-Focused	LandTrendr recovery	Burn severity		Defoliation from insects and diseases		
	3-Directed		Mortality from insects and diseases	Mortality and defoliation from insects and diseases	Defoliation from insects and diseases	Active large fire	
	4-Project					Active fire	Fire behavior

Detection Monitoring/Synoptic Surveillance: Tier 1b

Survey intelligence regarding ongoing forest disturbances comes from multiple sources. Often the best source of information regarding a pest outbreak or forest decline comes from the public. These reports can be more timely than regular survey methods, but they typically lack information regarding the scale or scope of the problem. The public may see a stretch of defoliation along a roadside but may not be able to comment on the overall extent, which may occur over several counties or multiple regions. Public reports often lack proper identification of the causal agent of a forest disturbance event. Informal observations from trained specialists yield more accurate determinations of causal agents; however, due to the nature of limited, localized observations, they may not identify the full extent of the disturbance agent. More extensive observational systems from aerial detection surveys allow for better overviews of disturbance agents, although these sacrifice some of the determination accuracy since observations are limited to areas and times flown, which may not coincide with the peak signature of the disturbance event.

Real-time forest disturbance mapping technology from coarse spatial resolution satellite imagery allows for a consistent and replicable assessment of the extent and magnitude of disturbances. The Forest

Disturbance Monitor (FDM) Web site (USDA Forest Service 2013b) was developed to host the near real-time forest disturbance (RTFD) mapping data as a first-cut disturbance monitoring system. By tracking interseasonal changes, early-detected disturbances can be monitored subsequently using finer scale remote sensing applications. Though Moderate Resolution Imaging Spectroradiometer (MODIS) satellite imagery is collected twice daily, frequent cloud contamination limits the ability to gather adequate samples to produce timely composites. Even with compositing periods of 16 and 24 days, RTFD is somewhat limited in producing a consistently reliable early-warning system. Due to the coarse resolution of the imagery, the system is not sensitive to all types of forest disturbance. These shortcomings limit the utility of this system as a stand-alone system; however, when integrated with casual observations, data analysis, and aerial and ground survey, there is the potential to create a more robust sound and timelier survey system.

Insect and Disease Survey (IDS) is the main focus of the Tier 1b monitoring effort and currently relies largely on aerial detection surveys (ADS). For this data stream, information is collected using a hardware/software system called the Digital Aerial Sketchmapping System (DASM) (USDA Forest Service 2013a). Though this system is designed for aerial observation of pest damage, it can be used to capture areas of damage from the ground. Tier 1a and Tier 1b monitoring can be utilized to prioritize IDS missions.

DASM Next Gen is a redesign of the Digital Aerial Sketchmapping System. This computer tablet-based data-collection system will retain much of the original functionality but will be based upon less-expensive technology and will be designed to fully support both ground-based survey and aerial observations.

Detection Monitoring/Focused Surveillance: Tier 2a

Sadly, in 2010 FHP lost a survey crew, consisting of a pilot and two aerial surveyors, in a crash. This tragedy forced FHP to examine the value of information gained versus the risk of conducting aerial surveys, and the methods by which it collects insect and disease information. The FHP staff director and management team concluded that whereas aerial survey remains a key data-collection tool, FHP needs to decrease its reliance on flying. While FHTET and FHP have been working on multiple tools for collecting field data in a consistent fashion for some time, this tragedy reinforced the need to develop an integrated strategy. As no one technique offers a complete assessment of forest health on a national scale, an integration of existing technologies and an identification of new opportunities seem warranted. A remote sensing strategy and new tools are being developed by FHTET to help focus monitoring and evaluate the extent of forest disturbances.

The following is a list of existing forest pest reporting technologies that are being integrated into FHP's monitoring system:

- The Early Detection and Rapid Response (EDRR) database was developed to support State cooperators and FHP personnel for early detection of key invasive pests using traps, periodic collections, and taxonomist expertise.
- Pest Event Reporter (PER) supplements IDS and standardizes the data entry process across the regions to allow local experts to integrate and interpret pest observations from multiple sources and enter the records into the database.
- The Southern Pine Beetle Information System (SPBIS) has survey, management, and cost reporting functions for the management of southern pine beetle (*Dendroctonus frontalis*) on Federal lands.
- The Southern Pine Beetle Portal (SPB portal) application focuses on southern pine beetle survey on all lands, including both trap counts and spot delimitation.
- Pest Observations (PestOBS) allows users to capture forest pest damage locations, typically based on ground observations. While PER operates at the scale of a pest event that typically involves many observations over a multi-county area, PestOBS records individual observation sites and may eventually include crowd-sourced data.
- Operationalizing Remote Sensing for Forest Health Protection (ORS) has the potential to be a companion, alternative, and gap-filler for aerial survey, and to facilitate the evolution of aerial survey to a more reliable insect and disease survey. The increased availability of imagery, combined with lower imagery costs, makes operational remote sensing a practical opportunity. Finer-scaled remotely sensed products can gap-fill missing coverages from aerial survey and serve as an alternative survey where aviation safety concerns are high and where potential pest impacts are of high concern.

For Tier 2 monitoring, FHTET proposes a multi-staged system that takes advantage of synoptic sensors for phenological trend and high resolution sensors for detecting damage and mortality. Extension of FDM products to incorporate interseasonal trend data with finer-scaled imagery can be utilized to track on-going mortality and defoliation extents. At a resolution of 22 m, the Disaster Monitoring Constellation (DMC) collects imagery every three to five days over any given point during the growing season (Bethel 2013). Alternative imagery from U.S. Geological Survey Landsat 8 (now operational) can also be utilized, as can the European Space Agency's future Sentinel satellite constellation.

Evaluation Monitoring/Directed Monitoring: Tier 2b

ORS in a second stage can be implemented to create a Directed Monitoring production environment. The production environment can be conducted as a stand-alone environment as an extension of Detection Monitoring or in conjunction with ground investigations in support of Evaluation Monitoring efforts. In a stand-alone environment, ORS can be used as fine-scale Detection Monitoring by utilizing available remotely sensed high-resolution data as a surrogate for field data collection. High-resolution imagery from WorldView-2 and other high-resolution satellites can be utilized for individual tree counts for both recent and older mortality and can serve as an alternative surveying technique or for gap-filling missing aerial survey coverage areas. Automation of this process can be refined through the development of object-recognition algorithms.

When combined with ground investigations, Directed Monitoring ORS can be used to determine extent, severity, and causes of disturbance. A team of specialists working closely with regional aerial survey programs can conduct damage assessments. In areas where the causal agent cannot be determined through remote sensing alone, aerial and ground observers can augment the image analysis with appropriate attribution.

Project Assessment and Implementation: Tier 3

FHTET also develops treatment technologies for direct control of forest health concerns. Bio-control technologies are used to limit the biological processes of the targeted species. Aerial spray technologies are designed to aid in the dispersal of chemical and biological control agents. Semiochemicals (pheromones, anti-aggregation chemicals) can be utilized to protect or concentrate localized populations of pests.

OBSERVATION AND REPORTING FREQUENCY (TEMPORAL CADENCE)

Surveys of disturbance events are temporally limited in that the duration of an observation is, in effect, a snapshot of the conditions at a given point in time. Ephemeral disturbance events can be entirely missed if the observation snapshots are too far apart, while persistent disturbance events can be observed through multiple observation periods.

Intraseasonal (short cadence) needs for tracking short-duration (e.g., defoliators) and multi-generation-per-year disturbance agents (e.g., southern pine beetles) require frequent observations. Ground observations, multiple aerial survey flights, and frequent remote sensing collections can be utilized to track the extent of a known ephemeral disturbance agent.

In an effort to differentiate between persistent and ephemeral disturbances, two FDM persistence products were developed. The 3-year and 5-year FDM Persistence of Disturbance (POD) products track deviations from prior season normals by looking for deviations over three consecutive overlapping 16-day compositing periods (spanning 32 days). This reduces some of the system noise; however, it does not track subtle changes due to the large class bins. A cumulative seasonal product may be able to track some of the more subtle changes, but this has not been developed. Additional noise reduction efforts to aid in tracking more subtle, but persistent, changes might be achieved through developing a drought-normalized adjustment based upon current monthly drought indices.

Interseasonal (moderate cadence) needs for tracking longer duration disturbance agents (e.g., mortality) require less frequent observation. *ForWarn* (chapter 6) produces several products that track these disturbance agents: full-archive change detection, 5-year change detection, 3-year change detection, and annual change detection (Hargrove and others 2009). Each dataset is a 24-day gap-filled composite of daily imagery produced every 8 days. FDM also produces two products that track longer duration disturbance agents: 5-year trend analysis and 3-year change detection products, where each dataset is a 16-day non-filled gap composite of daily imagery produced every 8 days. While the differences between the *ForWarn* and FDM approaches are subtle, the systems have significantly different intents. *ForWarn* is predicated on creating an absolute deviation from normal for quantitative assessments, while FDM is predicated on creating a user-oriented system to allow for direct interpretation of disturbance events to facilitate an observational survey and create a refined extent for the disturbance through IDS.

Annual reporting needs for area impacted by damage agents are currently being met with the IDS program. However, concerns over safety as well as customer needs for damage intensity information are driving the development of a revised system. A grid-based organization of geospatial data will be incorporated in the DASM Next Gen system, which will allow for better estimates of damage intensity over limited extents. The grid-based organization also will allow for the transition to an ORS-based system and a ground-based survey. It may be possible to incorporate sampling strategies, but this is not currently being considered.

Periodic national assessments, such as the 2010 National Report on Sustainable Forests (USDA Forest Service 2011), focus on semi-decadal tracking (long cadence) and are accomplished through a compilation of the FIA State reports and an aggregation of the annual IDS survey data. Though most States' inventory schedules data are annualized, mortality estimates are based upon observed recent mortality on plot samples (recent mortality within the last 5 years), which does not fit well with the annualized sample design. Mortality estimates can be improved using an intermediate assessment of mid-scale disturbance products. Possibilities exist to develop a calibrated dataset based upon the MODIS phenology dataset (2001–12) together with two separate archives from the USGS/Earth Resources Observation and Sciences (EROS) Data Center phenology dataset (at 250-m resolution) and the NASA/Stennis phenology dataset (a 232-m resolution dataset for 2001 through 2009). FHTET utilized the NASA/Stennis phenology dataset to calibrate against remeasured FIA plots to adjust the NDIRM host layers and bring the dataset to a 2011 vintage (Ellenwood and others [in press]).

Additional tools under development would allow for the compilation of long-term disturbance monitoring assessments. Trends in Canopy Change (TCC) is part of the National Land Cover Dataset to track decadal change in forest canopy (Coulston and others 2012). Monitoring Trends in Burn Severity (MTBS) utilizes the historical Landsat archive to map burn severity of large fires (Eidenshink 2007). A more complete method that tracks all disturbances throughout the Landsat archive is the LandTrendr/TimeSync (Cohen and others 2010). Though currently existing for the West Coast and a few isolated Landsat scenes, this dataset tracks disturbance at a 30-m scale throughout the 28-year historic record and allows for the characterization of the initial disturbance impact as well as the subsequent recovery. The long-term implications and utility of this dataset have yet to be fully realized. The ability to calibrate these disturbances can yield benefits for modernizing our estimates of existing conditions and the potential of future impacts of disturbance agents.

STRATEGIC SOLUTION

An overarching strategy is needed to integrate each piece of the monitoring assessments. As a one-stop shopping system, the FHP Mapping and Reporting Portal (FMRP) combines the inventory, real-time tracking, and reporting tools to allow for better planning and the potential for better integration of these separate technologies (fig. 7.2). The applications are independent of one another and operate at disparate scales. The integration of the toolsets requires skilled analysis to produce statistically robust and viable projects.

One path to better integration is to establish a standard evaluation grid. This grid framework would accommodate all survey methods (ground, aerial sketchmapping, and remote sensing), integrate with other geospatial data layers (tree species distribution, pest ranges, land cover, etc.), and provide set resolutions. FHTET is in the process of developing a national fixed grid at a 240-m cell resolution (approximately 6 ha) as a basis for all survey reporting. This 240-m grid aligns with key national, geospatial datasets, including National Land Cover Database (NLCD) and LANDFIRE. The 240-m standard has already been utilized to build and align a comprehensive Spatial Data Library of Geographic Information System (GIS) layers (soils, climate, species-specific tree host parameters, etc.) for NDIRM. By tying all forest health survey observations (ground survey points, sketch-mapped polygons and sample surveys) to the 240-m cells on which pest events occur, FHP can standardize the scale of its own data. By establishing a framework that borrows from a widely accepted national standard, FHTET has established a convenient basis for integrating forest health surveys, the NIDRM, remotely sensed data, and other important datasets and new reporting technologies such as HTML5/mobile device-based platform software.

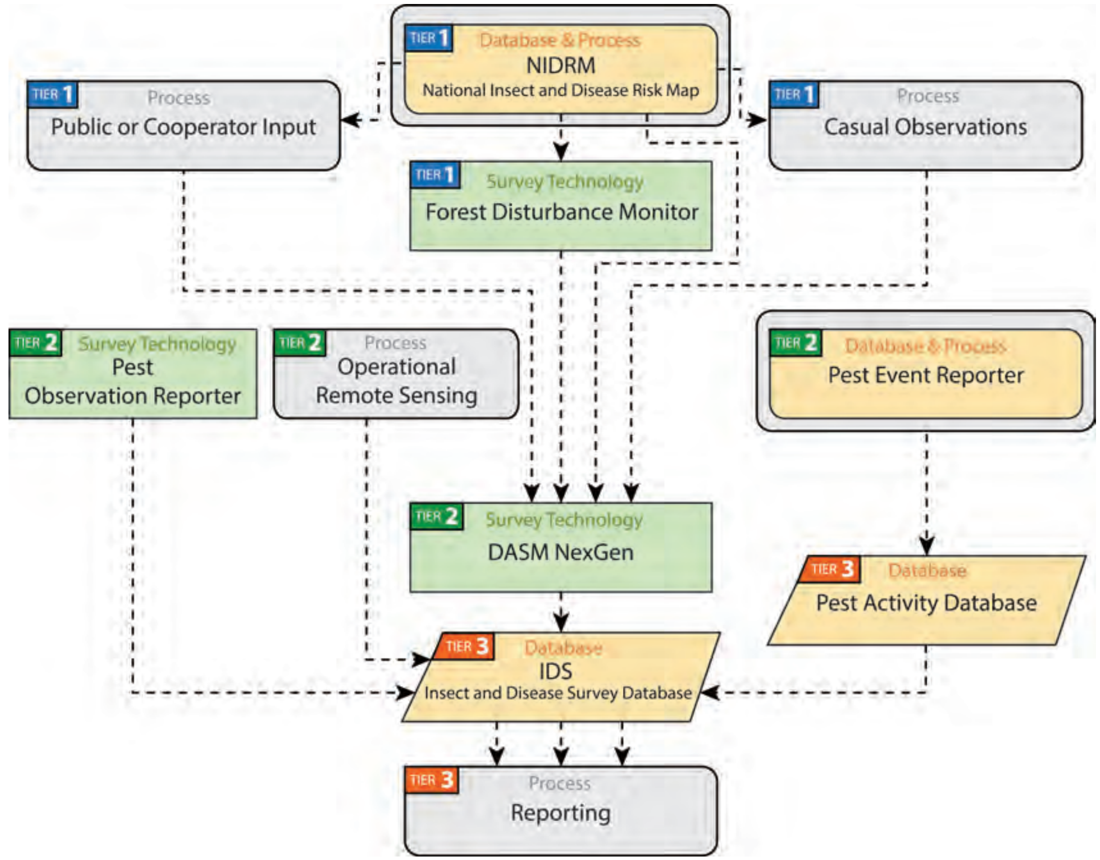

Figure 7.2—The Forest Health Protection Mapping and Reporting Portal combines the inventory, real-time tracking, and reporting tools to allow for better planning and the potential for better integration of these separate technologies. DASM= Digital Aerial Sketchmapping.

CASE STUDY 1—FOREST DISTURBANCE MONITOR

Forest Disturbance Monitor (FDM) is a Web-based forest disturbance data delivery system that was specifically designed for integration with IDS. FDM is designed to enhance the ability of forest health personnel to allocate resources and plan missions for aerial and ground forest health surveys.

The forest disturbance products used in the FDM are based on 240-m MODIS satellite data and are created from 16-day (RSAC) or 24-day (NASA) composites updated every 8 days. FDM utilizes two main types of forest disturbance products: the 3-year Real Time Forest Disturbance (RTFD) data, produced by both RSAC (Nielson 2008) and NASA (Hargrove and others 2009), and the 5-year Trend Disturbance Data (TDD) produced by RSAC (Chastain and others 2013). Collectively these are referred to as the Disturbance Composites. The 3-year RTFD dataset is a change-detection product that compares the current RTFD Normalized Difference Vegetation Index (NDVI) to a 3-year baseline of NDVI. RTFD is sensitive to defoliation events in deciduous forests, such as those caused by gypsy moth. The 5-year TDD is derived from a regression analysis of the past 5-year NDVI data with the slope parameter normalized. The TDD is sensitive to longer-term disturbances in coniferous forests such as those caused by mountain pine beetle.

Persistence of Disturbance

In addition to the composites, FDM produces 240-m 3-year and 5-year Persistence of Disturbance (POD) products. The POD is created by combining the negative departure values from normal NDVI values of the latest three Disturbance Composite products. The 3-year persistence is defined as those pixels whose NDVI values are significantly below normal as compared to the 3-year baseline and whose NDVI have remained substantially below normal for at least the latest two of the last three or all three RTFD composite periods (32 total days). The 5-year POD is defined as those pixels whose 5-year regression trend of forest NDVI have had a negative regression slope for at least the latest two of the last three or all three RTFD composite periods. A 5-year POD would indicate significant changes in forest structure.

Histogram Threshold Tool

The FDM is designed to allow the user to directly interact with the 3-year RFTD and 5-year TDD products using the FDM Histogram Threshold Tool. This tool is used to develop forest disturbance signatures by reclassifying the continuous Disturbance Composite data in real time. The Disturbance Composite data use only the reduced (below-normal) NDVI values of the disturbance data and exclude the above-normal NDVI values. This enhances the signal for the negative departures from normal and allows finer thresholding for the classification of the disturbance. Different levels of departure from normal NDVI are located at different positions within the Disturbance Composite histogram. The FDM Histogram Threshold Tool allows the user to select specific threshold ranges that best represent a potential forest disturbance. The severity of forest damage varies for different types of forest disturbances, as does the threshold range that best captures these disturbances in the MODIS-based Disturbance Composite data. This is illustrated in figure 7.3, where locations represented on the left side of the histogram have a much greater departure from normal NDVI and represent areas of severe disturbance such as fires, storm damage, and extreme insect and disease activity. Similarly, moderate severity disturbance values are typically found within the shoulder of the histogram, and the more subtle or less detectable disturbances are associated with the right side portion of the histogram. FDM also includes important ancillary data such as IDS data for

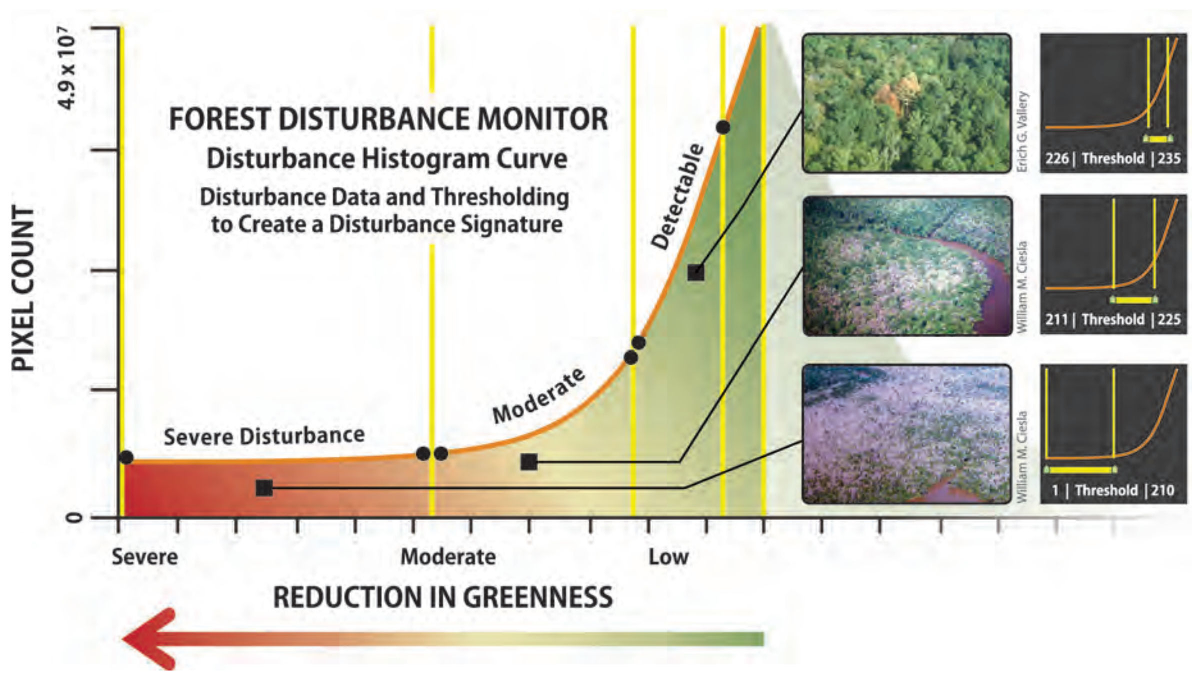

Figure 7.3—Examples of the application of the Forest Disturbance Monitor Histogram Threshold Tool and the degree to which the disturbances are reflected by departures from the normal Normalized Difference Vegetation Index.

the previous 5 years (dominant tree species, recent drought severity data, and past fire data) that allow the user to eliminate non-insect and disease forest disturbances.

FDM Process Procedures

The FDM system incorporates an intuitive three-step process that allows the user to (1) quickly scan and assess large areas for potential forest disturbances, (2) create spatial forest disturbance signatures using the FDM Disturbance Composite threshold tool, and (3) create downloadable datasets (polygon and raster) that can be used in aerial and ground survey missions and in subsequent GIS analyses (fig. 7.4).

The first step in the FDM process is to examine the Persistence of Disturbance data. The persistence data are designed as the initial targeting layer, allowing the user to quickly locate potential forest disturbance over a wide area. Compared to the Disturbance Composite data, the Persistence of Disturbance is a more conservative spatial estimate of a potential forest disturbance. The persistence data have other important advantages including reducing the amount of false positives due to atmospheric contamination from clouds and haze, and work as a reference guide when thresholding the Disturbance Composite data. The Persistence of Disturbance data are made up of three basic classes: (1) disturbance that is detectable, (2) moderate disturbance, and (3) severe disturbance. These classes are based on a combination of the temporal persistence and the degree of departure from normal NDVI. The second step in the FDM process involves creating forest disturbance signatures using the Histogram Threshold Tool and the selected current Disturbance Composite data. The thresholding is accomplished by adjusting the left and right slider located just under the histogram graphic tool. Any adjustment made to the histogram slider is updated immediately in the viewer, allowing the user to quickly create a disturbance signature that best represents the potential disturbance area. All pixels that fall within the adjusted range between the left and right slider will be visible in the viewer frame. The recommended starting thresholds are 100 and 225, which primarily capture moderately severe disturbances. The user can adjust the thresholds until the desired extent is reached. In the third step, the user creates polygons using the disturbance thresholded areas as a guide. These polygons are downloaded and utilized in the DASM system. These polygons contain attribute data based on the disturbance data including the creation date, the data source (RSAC or NASA), the name and date range of the Disturbance Composite, and the threshold range. The user may also assign additional attributes, such as the rationale for the polygon's delineation and the hypothesized (or verified) causal agent of disturbance, among other data needs.

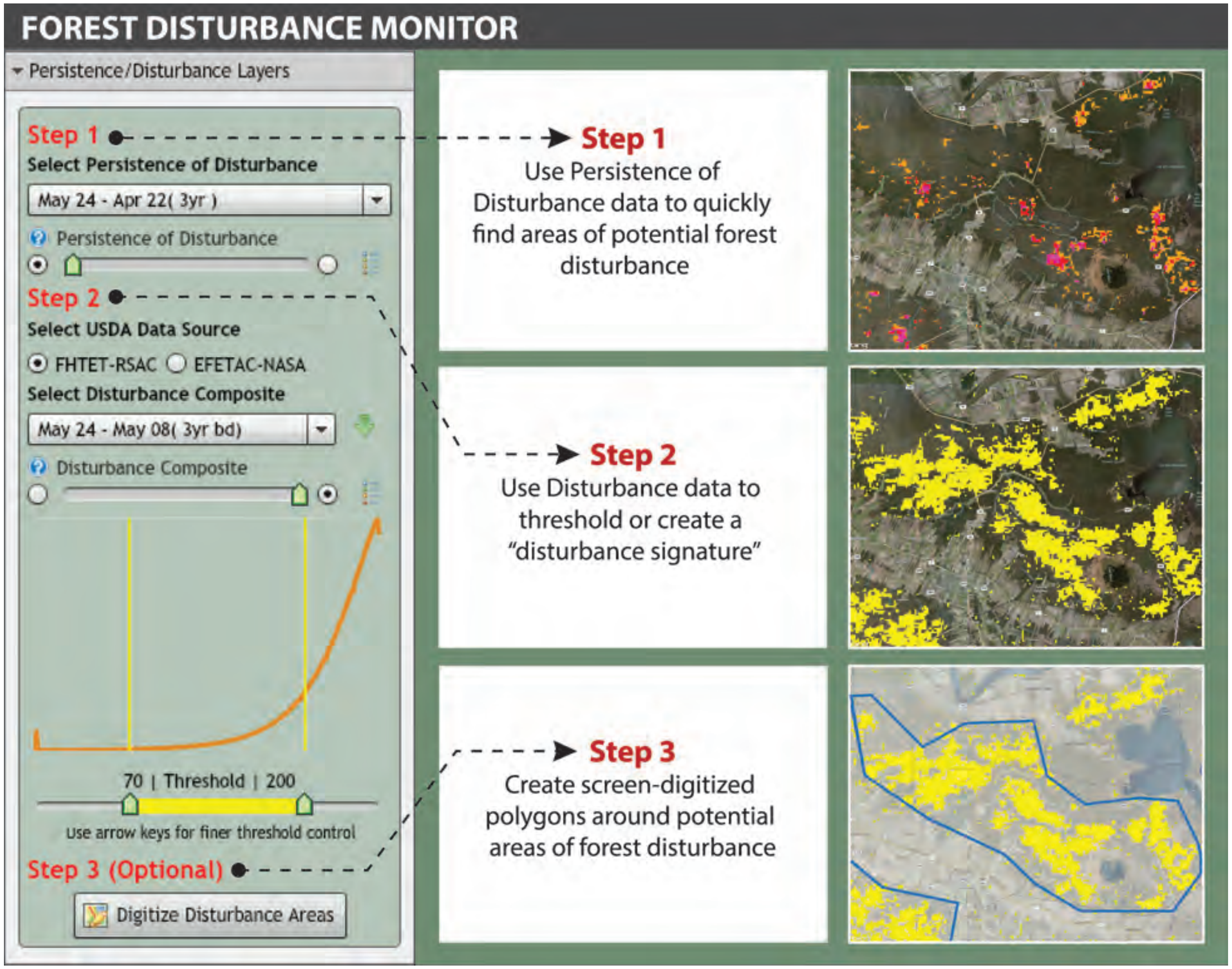

Figure 7.4—The Forest Disturbance Monitor (FDM) system incorporates an intuitive three-step process that allows the user to (1) quickly scan and assess large areas for potential forest disturbances, (2) create spatial forest disturbance signatures using the FDM Disturbance Composite threshold tool, and (3) create downloadable datasets that can be used in aerial and ground survey missions and in subsequent geographic information system analyses.

Example 1: Charles County, Maryland

In Charles County, MD, a fall cankerworm (*Alsophila pometaria*) outbreak occurred in April and May 2013. The FDM was used to create apparent forest disturbance signatures using the April 22 RSAC Disturbance Composite data using thresholds of 100–210. Screen-digitized polygons were created over the general area of the disturbance signature. An aerial survey was flown (by Biff Thompson, Forest Health Technician, Maryland Department of Agriculture) on May 15, 2013, using a DASM system. The resulting IDS polygons were then spatially assessed with the Disturbance Composite data (fig. 7.5). The results show good general spatial congruency of the disturbance signatures and the resulting IDS polygons.

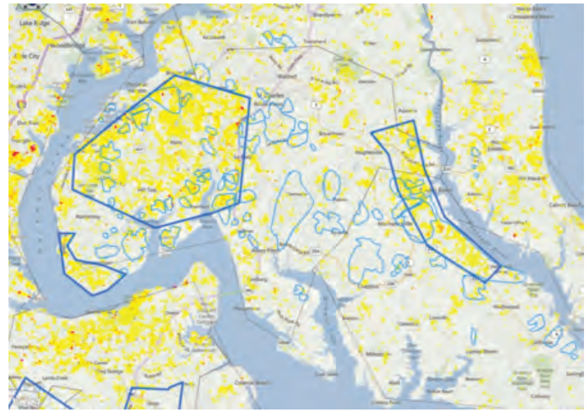

Figure 7.5—A good general spatial congruency was found between Insect and Disease Survey polygons and apparent forest disturbance signatures created by the Forest Disturbance Monitor for a fall cankerworm outbreak in Charles County, MD, in April and May 2013.

An analysis of the Disturbance Composite data and the IDS data with Charles County, MD, shows 51 total IDS polygons, of which 33 polygons (65 percent) correctly overlay areas identified as potential forest disturbances using the FDM and the Disturbance Composite data. Eighteen of the 51 IDS polygons (35 percent) did not overlay any disturbance identified with FDM. This indicates that the forest disturbance data do not consistently detect all forest disturbances and are dependent on the severity and size of the disturbance. In addition, there were also areas that were identified as having forest disturbance but where no IDS polygons were generated. Overall, the results of this assessment show the FDM system performed as designed and successfully located a majority of the disturbance in an area verified as affected by a fall cankerworm outbreak.

Example 2: Atchafalaya Valley Area, Louisiana

An outbreak of forest tent caterpillar (*Malacosoma disstria*) began west and southwest of Lake Pontchartrain and New Orleans, LA (in St. James, Assumption, Lafourche and Terrebonne Parishes), in April and May 2013. The FDM was used to create apparent forest disturbance signatures using the April 30 RSAC Disturbance Composite data using thresholds of 100 and 220. Screen-digitized polygons were created over the general area of the disturbance signature. An aerial survey was flown by Region 8 personnel on May 6, 2013, using a DASM. The resulting IDS polygons were then spatially assessed with the Disturbance Composite data (fig. 7.6). The results show excellent spatial congruency between the disturbance data and the IDS polygons and indicated the FDM screen-digitized polygons adequately indicated the general areas of disturbance that required survey.

An analysis of the 114 total IDS polygons shows that 69 percent overlaid the thresholded forest disturbance signatures. There was an area of disturbance commission where forest disturbance was indicated by FDM but not by IDS to the west of Baton Rouge (West Baton Rouge Parish). This signature is most likely due to phenological responses of later-than-normal leaf-out in the dominant sugarberry forest type. Approximately 31 percent of the area had cankerworm (as identified by IDS) that the thresholded forest disturbance signature did not detect. Based on the IDS data, this was because these areas had light defoliation disturbance that was not detected by the disturbance data. This indicates that the forest disturbance data do not consistently detect all forest disturbances and are dependent on the severity and size of the disturbance event. Overall, the system performed as designed and indicated that disturbance was occurring in the area, providing data that the aerial surveyor could use as information when flying the overall affected area.

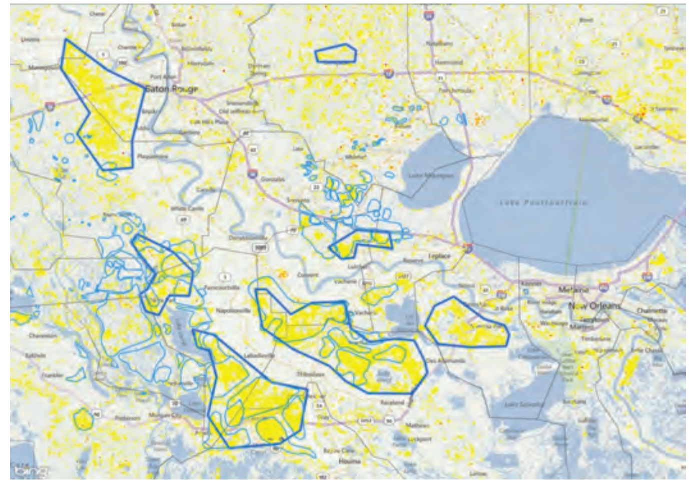

Figure 7.6— Excellent spatial congruency was found between Insect and Disease Survey polygons and apparent forest disturbance signatures created by the Forest Disturbance Monitor for a forest tent caterpillar outbreak near New Orleans, LA, in April and May 2013.

CASE STUDY 2—INSECT AND DISEASE SURVEYS

From 2000 through 2011, FHP's Insect and Disease Survey (IDS) program mapped and reported more than 18 million ha across the United States with significant tree mortality, with additional areas of other damage types as well. At a national scale, this mortality information is relatively coarse: areas mapped as having tree mortality can range from landscape-level epidemics to pockets of dead trees or widely scattered mortality intermixed with areas of healthy trees, and to lands that are not forested (USDA Forest Service 2009). Natural resource managers need more specific, consistently mapped, and appropriately scaled information about where dead trees exist. Other damage surveys are also warranted. Meaningful damage assessments that can be derived from focused post-processing of the national data are difficult because of different and uncertain spatial and temporal scales of the source data. Business needs can vary within each monitoring tier and at all administrative levels of the agency. There is not only a need to measure damage intensity but also an increasing interest in more specific information regarding the residual standing live trees remaining in pest-infested areas. To meet those needs, FHP is evaluating whether to shift from capturing data from "areas *with* tree mortality," to "areas *of* tree mortality," and to more accurately and usefully represent data about relative loss expressed as percentages of forest.

Digital Aerial Sketchmapping is FHP's primary method to survey the presence and impacts of forest insects, diseases, and other disturbances that lead to tree mortality or other damage impacting growth, health, and vigor. A network of FHP crews and State partners conducts aerial and ground-based surveys across State, private and Federal lands. Regional IDS data are edited to meet national reporting standards and transmitted to FHTET on an annual basis. FHTET then integrates the data into the FHP national Insect and Disease Survey (IDS) database and creates regional and national maps detailing forest mortality and other damage.

Data Standards and Guidelines

The data and methods FHTET uses to produce these maps and analyses are well documented, but due to the increasing importance, utility, and client base of the data, there is a national interest in improving accuracy and meeting data standards. However, despite significant investments to develop new standards, provide compliance training, and enhance technology, national standards are still too broad and provide surveyors too much latitude in how damage is recorded.

The Benefits of Insect and Disease Survey

FHP partners derive considerable benefit from their own aerial survey and pest-damage maps because they have an intimate understanding of both the pest patterns in their local landscapes and of the idiosyncrasies and tendencies of the mapping crews who observe and record the data. These benefits include:

- Assisting managers in meeting requirements for reporting on the status of forest insects and diseases on an annual basis (USDA Forest Service 1997)
- Summarizing disturbance events and trends for Forest Monitoring Reports and Biological Evaluations

- Contributing to regional and State highlights reports
- Detecting and mapping outbreaks of native and exotic insects and diseases (within limits of aerial surveys and levels of effort to investigate developing threats)
- Communicating with the public and elected officials
- Describing landscape-level impacts over time (within the statistical and biological constraints of these data)
- Locating studies to determine causal agents, extents, and severity of disturbance events
- Providing an approximate (first cut) at extent and severity of disturbance events in support of salvage and suppression projects
- Linking to risk-rating systems (appropriate when used as trend data and not for precise locations)
- Providing dynamic visual representations of insect/disease disturbance over time
- Providing timely disturbance information to assist land managers in prioritizing work
- Estimating acres impacted summarized by jurisdictional boundaries (e.g., Forests, State districts, State lands, and counties)
- Estimating snag densities at appropriate scales
- Training remotely sensed data

These benefits produce significant challenges when FHP aggregates regional data nationally. The sheer scale of uncertainties within, and inconsistencies between, the regional-level surveys inhibits FHP's ability to collaborate and share reliable information with its research partners. The uncertainties and inconsistencies restrict how the data can be (1) used to validate the NIDRM models and (2) integrated with cell-based, remotely sensed imagery or ground-based surveys.

Geospatial Technology

FHP is a leader in the collection and dissemination of geospatial data about forest health and, as such, recognizes the need to make significant improvements to the data. FHP understands that aerial survey data cannot reliably answer some basic management questions. Often, the IDS estimate of damage area is overestimated and the severity of mortality (dead trees per acre) is underestimated. Inconsistencies among datasets exist with regard to (1) spatial resolution of the delineated polygon, (2) thresholds of damage severity used to designate an area as damaged, (3) variability among damage signatures from region to region, and (4) variation with host/agent relationships among regions.

FHP has a need and an opportunity to better quantify the extent and severity of many of the most significant insect and disease outbreaks of the past decade, including the mountain pine beetle, beech bark disease, and emerald ash borer epidemics. ADS struggles with accurately estimating higher levels of damage associated with epidemic pest events. The ability to track multiple-year pest trends is often limited due to lack of consensus as to what areas will be flown in a particular year. A more cohesive structured program of work that considers priority lands regardless of ownership, predisposed conditions, and a monitored seasonal trend can improve the ability to track pest trends. New remote sensing methods can provide more accurate spatial location and disturbance intensity measurements; however, cost and timeliness are not comparable to the current aerial survey.

The Path to Better Insect and Disease Survey

With the rapid evolution of GIS technology and the Internet, it has become significantly easier to distribute geospatial data in general, and for FHP to collect, combine, and analyze information from a multitude of sources. Coincidently, these advances are coming at the same time as demands and expectations on FHP to improve the quality of its IDS data are increasing and as aviation risk management is pressuring surveyors to fly less due to concern for employee safety. The path to a better insect and disease survey includes a new survey strategy, refreshed DASM hardware and software, and a more centralized and coordinated national planning effort.

New survey strategy—FHP is considering developing a forest health survey strategy that gives top priority to operator safety, maximizes the quality and value of aerial sketchmapping, and improves other data streams. For example, to overcome limitations on aerial sketchmappers to observe, identify, and record pest events accurately during flight, the system needs to do a better job of accommodating data from ground observations. To address safety concerns, FHP is considering a new survey strategy that:

1. Combines remote sensing, risk modeling techniques, and local expertise to prioritize areas where sketchmapping flights are most needed;
2. Identifies modifications to mission profile (pattern, altitude, maneuvers) and/or choice of aircraft; and,
3. Augments aerial surveys with remote sensing wherever possible and prudent (e.g., where alternative technology can be shown to capture useful forest health data such as mountain pine beetle damage in remote, high-altitude terrain).

Refresh Digital Aerial Sketchmapping hardware and software— FHTET is upgrading the DASM and will deliver a mobile data-collection platform and back-end process that integrates aerial detection, ground survey, and remote sensing data with a standard spatial resolution. Current DASM technology allows for the delineation of forest-pest damage as well as the (1) identification of a causal agent, (2) damage type, (3) damage intensity, and (4) host type. Delineation of damage is done through on-screen digitizing, using a pen-tablet personal computer. FHTET is examining possible improvements to this delineation process within the DASM refresh. Improvements may include the introduction of a new delineation method based on the selection of predefined areas, or grid cells, at fixed resolutions. This new data-collection platform needs to be mobile and compatible with both aerial and ground surveys.

The refreshed DASM will utilize the standardized grid as its back-end framework. For example, instead of sketchmappers making rapid, and typically flawed, trees-per-acre (TPA) mortality estimates, they would simply attribute selected grid cells from a list of percent-of-trees-affected categories (e.g., 1-3 percent, 4-10 percent, 11-29 percent). Then, by applying that percent-class selection to TPA information already loaded in its grid-aligned GIS, DASM will be able to calculate a much more reliable and consistent measure of tree mortality. This will be accomplished by providing the forest cover and host layers used as a baseline to indicate whether an area is "forested," as well as by providing the presence and correct stocking of specific species, regularly maintained to reflect current composition/basal area over time.

The need for more centralized and coordinated national planning—State and regional partners have considerable leeway as to where, when, and how much area to survey. Although FHTET expects its partners to submit spatial data about the areas flown and not flown within their jurisdictions, it does not have adequate information to assess whether a decision not to fly certain areas was based on the local knowledge that these areas are likely pest-free, or based on information or evidence unrelated to pest biology (e.g., funding, aircraft availability, travel restrictions, and weather limitations). Recent experience shows that local flight planning decisions can subvert the ability of the national IDS database to track trends of major outbreaks, such as the mountain pine beetle epidemic. FHTET and the aerial survey standards committee are working to revamp a survey coding system that, in order to accommodate a variety of region-specific concerns, has grown too complex, inconsistent, and inflexible. Each of these potential developments suggests the need for a more centralized and coordinated national planning program.

SUMMARY

Effective use of limited financial resources is driving the need to increase efficiencies at all levels of natural resources management. In this era of data overload, the complex processes of information management are a distraction from the real need to focus information analysis into useable knowledge and decision support. Disturbance is a natural process, but sorting out which disturbances are significant can be burdensome. Does a single tree dying constitute a significant impact? It does if it is the start of an outbreak. Gathering historical data, analyzing current conditions, and projecting future issues is a sound pathway toward the endeavor of providing useable knowledge. The development of the technologies to support each step of the pathway has been highlighted in this chapter. It is important that future development focuses on incorporating new pathways to better integrate these individual technologies.

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CHAPTER 8.

Invasive Plants on Forest Land in the United States

Christopher M. Oswalt

Sonja N. Oswalt

INTRODUCTION

The Forest Service, U.S. Department of Agriculture, considers a species to be invasive if it meets the following criteria: (1) the species must be nonnative to the ecosystem under consideration, and (2) the species' introduction must cause or be likely to cause economic or environmental harm or harm to human health (Executive Order No. 13112, Federal Register/Vol. 64, No. 25/February 8, 1999). Indeed, Pimentel and others (2005) last estimated the cost of prevention and eradication of invasive plant species in crop, pasture, and forest settings at approximately $27 billion every year. In fact, the cost of combating just the invasive tree melaleuca *(Melaleuca quinquefolia)* in the State of Florida was estimated at between $3 and $6 billion dollars in 2005 (Pimentel and others 2005).

On the Invasive Species Program Web site of the Forest Service, 54 plant species are recognized as invasive, presumably in forested systems (USDA Forest Service 2014). Long-term monitoring and assessment of invasive species occupation on the forest landscape is necessary to managers and policymakers for the obligation and direction of funds and other resources. Monitoring at a national scale can be difficult and expensive, however, given the regional nature of species distributions. Given the importance of monitoring invasive plants on U.S. forest land, units in the Forest Inventory and Analysis Program of the Forest Service have implemented efforts to track invasive plants in their regions. Up to this point in time, efforts by individual units have been unique and specific to those units (e.g., sample intensities and field protocols differ), thus no consistent method for identifying and tracking invasive plants has been applied nationwide. Efforts are underway to establish some modicum of consistency in measurement; however, for this paper we use data collected and compiled by each regional office. Our objectives were to produce a national map of invasive species infestation based on regionally collected data, which may be used to identify potential "hot spots" of invasion and which may serve as a baseline for future monitoring efforts. Additionally, we present regional analyses of data from the Southern United States, where a large number of invasive species impact forests in the continental United States.

METHODS

Data collected by the Forest Service Forest Inventory and Analysis (FIA) Program were assembled from each region of the United States. Occurrence, measured as the percent of forested subplots within a county with observed invasive plant species, was calculated across the continental United States and Hawaii. Each region and, in some cases, each State maintains a specific watch list to constrain monitoring to only the most important invasive plant species within a given area. Therefore, occurrence is based on regionally important species and is inconsistently measured across the United States.

The data used in the analysis spanned 1999 through 2011, depending on the State and region (table 8.1). Each region uses a distinct program for collecting invasive species data, though plans are underway to provide a nationally consistent method for future surveys. For this paper, data collection methods differed by region and, in some cases, State. Data were normalized to minimize differences between regions by calculating the number of forested subplots present in a county, the number of forested subplots with at least one invasive species present, and by generating a "percent invaded" statistic so that counties across the country could be compared in a consistent manner. County and regional comparisons are based on visual observations of mapped data. Rudis and others (2004) described data collection methods for the various regions, and specific data collection details are available through the FIA Web site at http://www.fia.fs.fed.us/library/field-guides-methods-proc/.

The temporal richness of the southern invasive plant data allowed for more detailed investigations of individual species across the region and spatial change in occurrence over time. Most plots in the southern region have been measured multiple times, including observations of 33 regionally important invasive plants. The number of invaded subplots and number of invaded plots were calculated.

Table 8.1—Regions, States, and data collection periods for invasive plants referred to in this publication

Region	States	Data collection period
Pacific Northwest	CA, OR, WA	1999–2009
Intermountain West	AZ, CO, ID, MT, NV, NM, UT, WY	1999–2009
North	CT, DE, IL, IN, IA, KS, ME, MD, MA, MI, MN, MO, NE, NH, NJ, NY, ND, OH, PA, RI, SD, VT, WV, WI	2007 –2011
South	AL, AR, FL, GA, KY, LA, MS, NC, OK, SC, TN, TX, VA	2001–2011
Alaska		2004–2009
Hawaii		2010

Additionally, the most recent observations were compared to previous observations (typically a 5-year remeasurement period) for a select list of representative invasive plants [Japanese honeysuckle *(Lonicera japonica)*, Chinese tallowtree *(Triadica sebifera)*, cogongrass *(Imperata cylindrica)*, nonnative roses (*Rosa* spp.), and garlic mustard (*Alliaria petiolata*)]. Plots were categorized into one of four infestation categories: no infestation (no observation at time 1 or time 2); newly infested (no observation at time 1 and positive observation at time 2);

stable infestation (positive observations at time 1 and 2); or elimination (positive observation at time 1 but no observation at time 2). Plots categorized as elimination for a given invasive plant were removed from the analysis due to the difficulty in determining the exact cause of the change in status. Areas of high expansion pressure were mapped based on the density of newly infested and stable plots using a simple inverse distance weighting imputation approach on a dummy variable based on weights assigned to each of the infestation categories (Roberts and others 2004).

RESULTS AND DISCUSSION

Nationwide, 39 percent of forested subplots sampled for invasive plants contained at least one invasive species. Hawaii had the highest percentage of subplots with invasive plants present at 70 percent. In general (excluding Hawaii), invasive species were more prevalent on forested subplots in the East than in the West (fig. 8.1), while Alaska and the Intermountain region had the fewest incidences of invasion. Approximately 46 percent of forested subplots in the broader eastern region had at least one

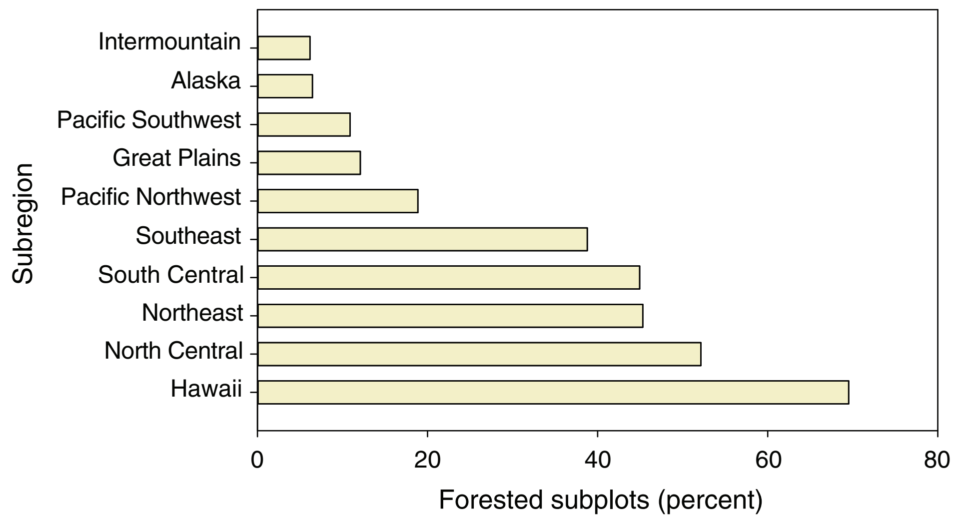

Figure 8.1—Percent of forested subplots containing at least one invasive plant, by subregion.

invasive species compared to only 6 percent in Alaska and 11 percent in the broader western region (fig. 8.2). Patterns of invasive species presence/absence also vary at the county level, as shown in figure 8.3. Highly fragmented landscapes (as in the North Central region) and major travel corridors (as in the Southeast) tended to exhibit higher percentages of forested subplots with at least one invasive species. Relationships between fragmentation and invasive species have been studied and recorded (Brothers and Spingam 1992, Luken and others 1997); however, other factors influence the abundance of invasive plants in particular regions. For example, some species like Chinese tallowtree, multiflora rose (*Rosa multiflora*), and autumn olive (*Elaeagnus umbellata*) were introduced and planted intentionally for various purposes including industrial use, hedgerows, and wildlife food.

In the Northern States, species infesting >5 percent of sampled plots included multiflora rose, reed canarygrass *(Phalaris arundinacea)*, garlic mustard, Japanese honeysuckle, buckthorn (*Rhamnus cathartica*), *Lonicera* spp., autumn olive, black locust (*Robinia pseudoacacia*), and Nepalese browntop (*Microstegium vimineum*; table 8.2). Multiflora rose had the highest rate of occurrence, appearing on 2,169 (25 percent) of 8,769 forested plots where invasive plants were monitored. Multiflora rose was over

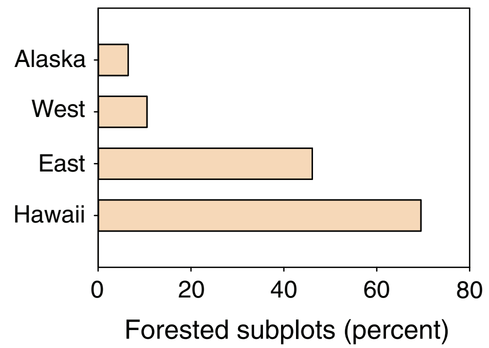

Figure 8.2—Percent of forested subplots containing at least one nonnative invasive plant, by major region.

three times as prevalent as the next most frequently recorded species, reed canarygrass, which appeared on 8 percent of monitored plots. Species infestation differed by State. In Delaware, for example, 44 percent of measured plots contained Japanese honeysuckle, compared to 19 percent with multiflora rose, while 71 percent of plots in Indiana contained multiflora rose. Ohio was overwhelmingly infested with multiflora rose, with 86 percent of measured plots containing the species. Nebraska differed from other Northern States in that Siberian elm (*Ulmus pumila*) was the most frequently recorded invasive, noted on 11 percent of plots. This is unsurprising given that Siberian elm experimental plantations were established throughout the Prairie States, and the tree

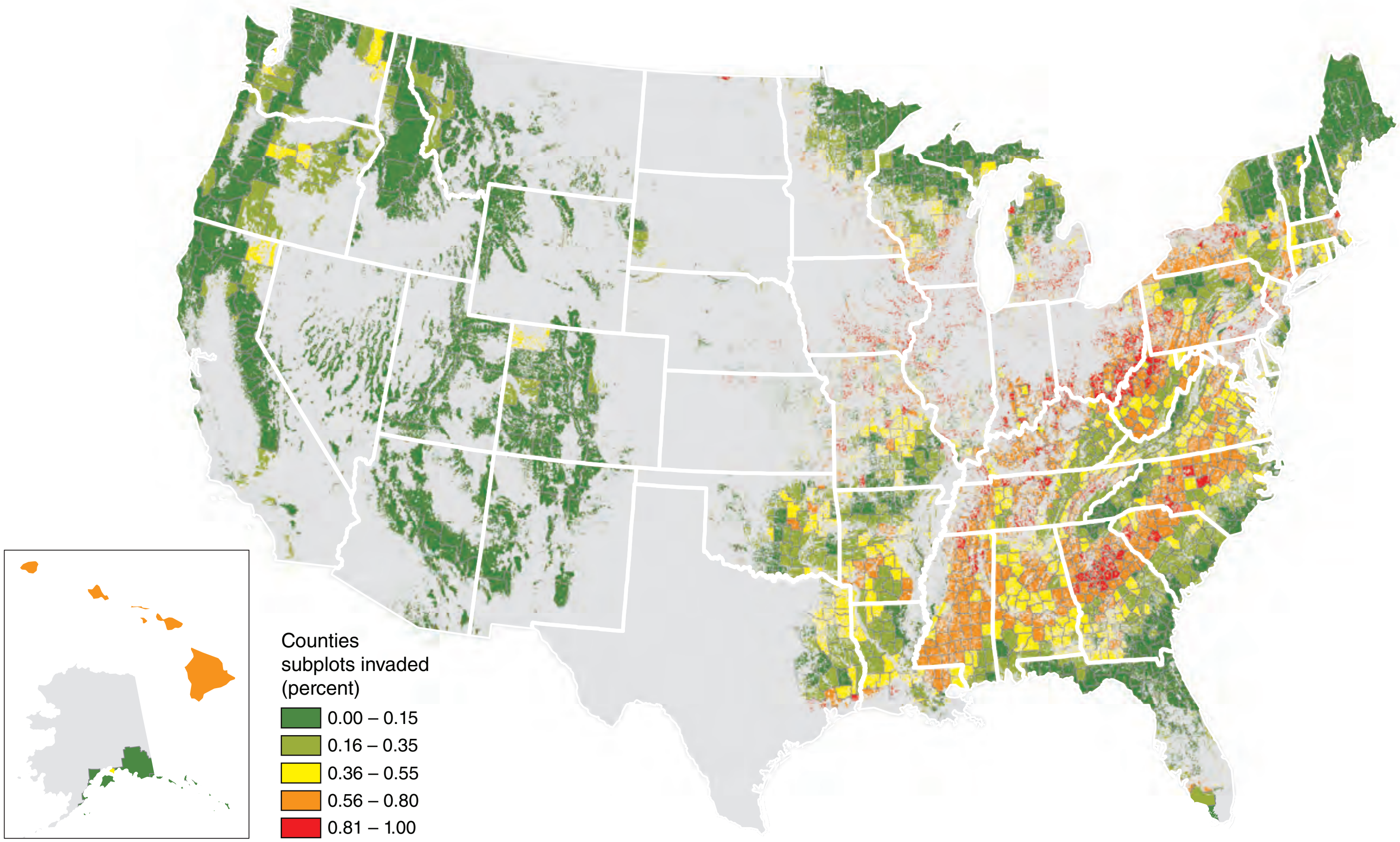

Figure 8.3— National map showing percent of forested subplots with at least one nonnative invasive plant, calculated at the county level. Forest/nonforest mask applied to the conterminous United States.

Table 8.2—Number and percent of forested plots in the Northern United States where monitored nonnative invasive plants were detected

Common name	Scientific name	Plots	
		number	*%*
Multiflora rose	*Rosa multiflora*	2,169	24.7
Reed canarygrass	*Phalaris arundinacea*	699	8.0
Garlic mustard	*Alliaria petiolata*	667	7.6
Japanese honeysuckle	*Lonicera japonica*	562	6.4
Common buckthorn	*Rhamnus cathartica*	511	5.8
Honeysuckle	*Lonicera*	489	5.6
Autumn olive	*Elaeagnus umbellata*	475	5.4
Black locust	*Robinia pseudoacacia*	461	5.3
Nepalese browntop	*Microstegium vimineum*	452	5.2
Japanese barberry	*Berberis thunbergii*	417	4.8
Morrow's honeysuckle	*Lonicera morrowii*	342	3.9
Canada thistle	*Cirsium arvense*	297	3.4
Amur honeysuckle	*Lonicera maacki*	270	3.1
Oriental bittersweet	*Celastrus orbiculatus*	238	2.7
Bull thistle	*Cirsium vulgare*	215	2.5
Glossy buckthorn	*Frangula alnus*	194	2.2
Tree-of-heaven	*Ailanthus altissima*	162	1.8
European privet	*Ligustrum vulgare*	157	1.8
Tatarian honeysuckle	*Lonicera tatarica*	122	1.4
Creeping jenny	*Lysimachia nummularia*	114	1.3
Showy fly honeysuckle	*Lonicera x bella*	96	1.1
Spotted knapweed	*Centaurea stoebe* ssp. *micranthos*	71	0.8
Dames rocket	*Hesperis matronalis*	68	0.8
Norway maple	*Acer platanoides*	55	0.6
Common barberry	*Berberis vulgaris*	49	0.6
Siberian elm	*Ulmus pumila*	49	0.6
Russian olive	*Elaeagnus angustifolia*	45	0.5
Common reed	*Phragmites australis*	31	0.4
Japanese knotweed	*Polygonum cuspidatum*	26	0.3
European cranberrybush	*Paulownia tomentosa*	20	0.2
Princesstree	*Viburnum opulus*	20	0.2
Leafy spurge	*Euphorbia esula*	18	0.2
Spotted knapweed	*Centaurea biebersteinii*	17	0.2
Purple loosestrife	*Lythrum salicaria*	17	0.2
Japanese meadowsweet	*Spiraea japonica*	14	0.2
Giant knotweed	*Polygonum sachalinense*	5	0.1
English ivy	*Hedera helix*	4	0.0
Louise's swallow-wort	*Cynanchum louiseae*	3	0.0
Bohemian knotweed	*Albizia julibrissin*	2	0.0
Saltcedar	*Polygonum x bohemicum*	2	0.0
Silktree	*Tamarix ramosissima*	2	0.0
Chinaberrytree	*Cynanchum rossicum*	1	0.0
European swallow-wort	*Melia azedarach*	1	0.0

was promoted as a hedge species in the 1950s (Klingaman 1999). Northern States with no single invasive species occupying more than 10 percent of measured plots included Maine, Michigan, New Hampshire, and Vermont (fig. 8.4). Maine had the lowest rates of infestation compared with other North Central and Northeastern States; no one invasive species occupied more than 2 percent of measured plots in Maine.

In the Southern States, Japanese honeysuckle was observed on >17,000 forested plots, or 43 percent of all forested plots where invasive plants were monitored (table 8.3). The prevalence of Japanese honeysuckle on southern forested plots obscured patterns for all other species; therefore, Japanese honeysuckle was removed from the analysis and select metrics were recalculated. The distribution of the number of southern counties across categories of the percent of subplots invaded by any monitored invasive plant (invaded class) was considerably different when Japanese honeysuckle was included compared to when it was not (fig. 8.5). Southern counties were

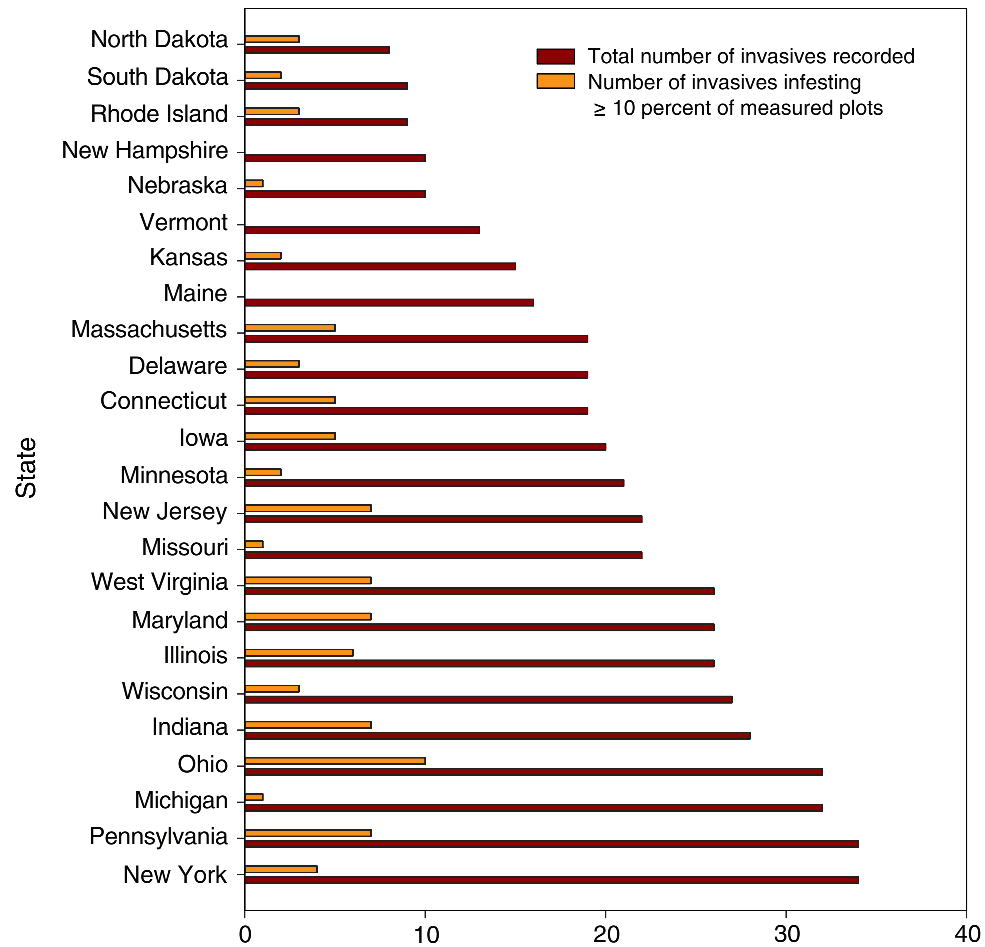

Figure 8.4— Number of invasive plants infesting ≥10 percent of measured plots compared with the total number of invasive plants recorded by State.

Table 8.3—Number and percent of forested plots in the Southern United States where monitored nonnative invasive plants were detected

Common name	Scientific name	Plots number	Plots %
Japanese honeysuckle	*Lonicera japonica*	17,212	43.3
Chinese/European privet	*Ligustrum sinense/L. vulgare*	8,260	20.8
Nonnative roses	*Rosa* spp.	3,469	8.7
Chinese lespedeza	*Lespedeza cuneata*	2,154	5.4
Nepalese browntop	*Microstegium vimineum*	2,067	5.2
Japanese climbing fern	*Lygodium japonicum*	1,731	4.4
Tallowtree, popcorntree	*Triadica sebifera (Sapium sebiferum)*	1,427	3.6
Shrubby lespedeza	*Lespedeza bicolor*	940	2.4
Tree-of-heaven	*Ailanthus altissima*	932	2.3
Silktree, mimosa	*Albizia julibrissin*	720	1.8
Tall fescue	*Lolium arundinaceum*	662	1.7
Chinaberry	*Melia azedarach*	550	1.4
Japanese/glossy privet	*Ligustrum japonicum/L. lucidum*	480	1.2
Bush honeysuckles	*Lonicera* spp.	468	1.2
Autumn olive	*Elaeagnus umbellata*	378	1.0
Kudzu	*Pueraria Montana* var. *lobata (Pueraria lobata)*	299	0.8
Princesstree, royal paulownia	*Paulownia tomentosa*	268	0.7
Sacred bamboo, nandina	*Nandina domestica*	161	0.4
Garlic mustard	*Alliaria petiolata*	143	0.4
Chinese/Japanese wisteria	*Wisteria sinensis/ W. floribunda*	141	0.4
Silverthorn, thorny olive	*Elaeagnus pungens*	134	0.3
English ivy	*Hedera helix*	120	0.3
Cogongrass	*Imperata cylindrica*	109	0.3
Wintercreeper	*Euonymus fortunei*	108	0.3
Nonnative vincas, periwinkles	*Vinca minor/V. major*	101	0.3
Nonnative climbing yams (air yam, Chinese yam)	*Dioscorea bulbifera/ D. oppositifolia*	96	0.2
Oriential or Asian bittersweet	*Celastrus orbiculatus*	77	0.2
Tropical soda apple	*Solanum viarum*	71	0.2
Winged burning bush	*Euonymus alata*	55	0.1
Nonnative bamboos	*Phyllostachys* spp., *Bambus* spp.	40	0.1
Russian olive	*Elaeagnus angustifolia*	22	0.1
Chinese silvergrass	*Miscanthus sinensis*	20	0.1
Giant reed	*Arundo donax*	6	0.0

distributed relatively evenly among many of the invaded classes below 75 percent when Japanese honeysuckle was included. However, when Japanese honeysuckle was removed, the distribution of counties was skewed heavily toward classes with smaller percentages of invaded subplots. The removal of Japanese honeysuckle reduced the mean percent of subplots invaded by 15 percent (fig. 8.6). Alabama, Mississippi, and Virginia each experienced the largest change when Japanese honeysuckle was removed from the analysis.

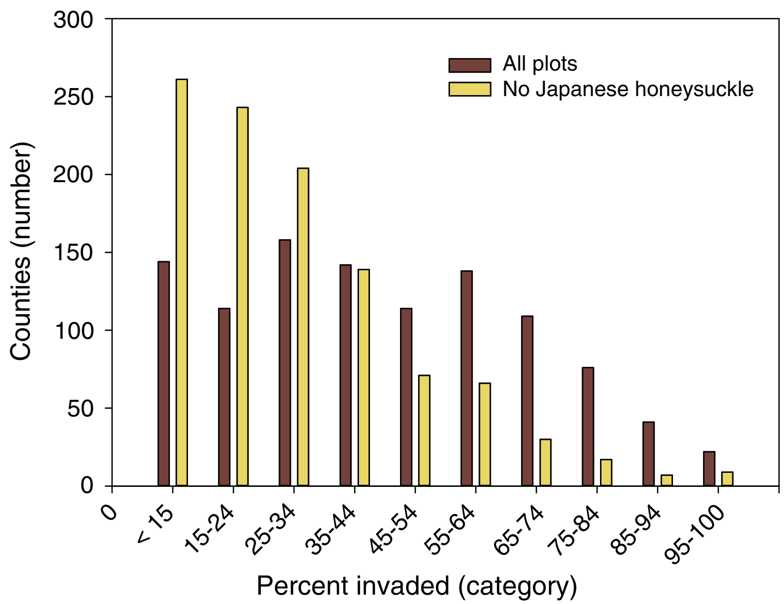

Figure 8.5— Comparison of the distribution of counties infested by percent invasion category, with and without Japanese honeysuckle.

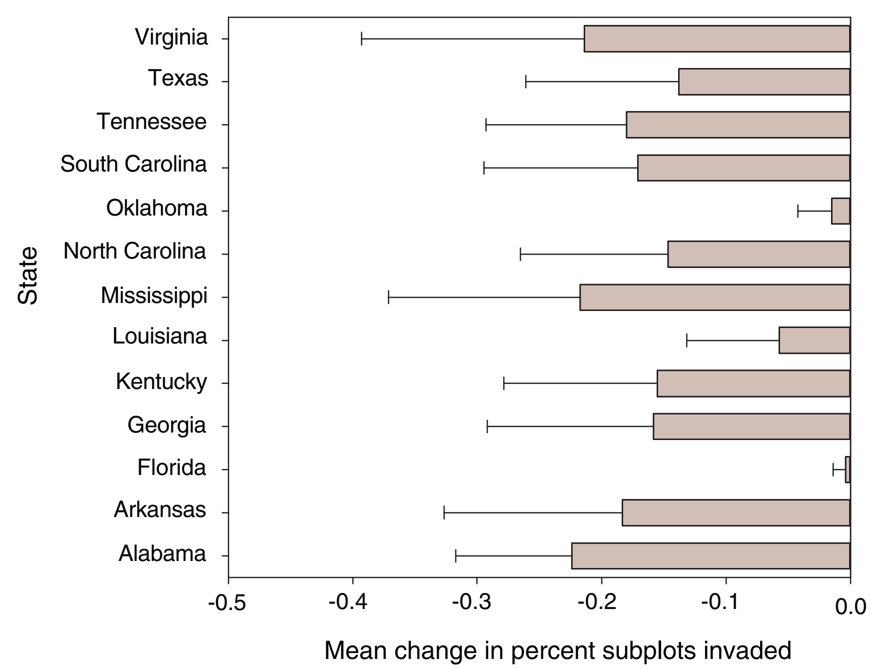

Figure 8.6— Change in mean percent of subplots invaded when Japanese honeysuckle was removed from analysis of southern Forest Inventory and Analysis plots.

The southern invasive plant data included numerous invasive plant species that currently cover large areas (e.g., Japanese honeysuckle) or are emerging as regionally significant forest invaders (e.g. cogongrass), and offered a valuable opportunity to evaluate recent changes in invasive plant distribution because of its temporal richness. Analysis of select species representing each of the plant life forms monitored by the Southern FIA program revealed that garlic mustard and cogongrass had the highest percentages of new infestations (table 8.4). For every plot where garlic mustard was previously noted there are now 12 newly infested plots. Such a high ratio of newly infested plots to stable plots suggests a high degree of active expansion relative to the observed population. In comparison, 76 percent of the plots where Japanese honeysuckle was observed during time 2 contained Japanese honeysuckle during the time 1 plot visit. At time 2, there were only 0.32 plots newly infested with Japanese honeysuckle for every stable plot. Tallowtree expansion activity relative to the observed population was one new plot for every stable plot. A smaller new-to-stable ratio for tallowtree, an invasive tree understood to be rapidly invading southern forests (Oswalt 2010), may be a result of a longer establishment time for trees versus herbaceous plants like garlic mustard. On the other hand, such a large new-to-stable ratio for herbaceous plants like garlic mustard may be influenced by improvements in identification skills by field personnel at time 2 after being exposed to additional invasive plant identification training. Additionally, year-round collection of invasive plant data in the Southern United States could potentially introduce some error or bias (Oswalt and others 2012).

Table 8.4—Percent of invaded plots and all monitored plots by select species and infestation class, showing changes in invasive plant distribution

Infestation class	Japanese honeysuckle	Cogongrass	Garlic mustard	Nonnative roses	Tallowtree
			percent of currently invaded plots		
Newly infested	24	68	92	47	51
Stable infestation	76	32	8	53	49
			percent of all monitored plots		
Newly infested	11.51	0.25	0.41	4.53	1.66
Stable infestation	36.30	0.12	0.04	5.17	1.59
New/stable ratio	0.32	2.09	11.50	0.88	1.05

Notable expansion activity was expected with an analysis of cogongrass due to the recent research activity focused on the species (Grebner and others 2010, Holzmueller and Jose 2011, Minogue and others 2012). Indeed, cogongrass had a new-to-stable infested plot ratio of >2. This indicates that for every plot where cogongrass has been found currently and in the past, there are two newly infested plots. The expansion pressure (fig. 8.7A) encircles the area where cogongrass is believed to have been introduced: the Gulf Coast of Mississippi, Alabama, and Florida around Mobile Bay

(Bryson and Carter 1993). The highest pressure exists in southern Mississippi, central Alabama, and the Florida panhandle.

Garlic mustard expansion pressure is located in the northernmost States included in the southern region (fig. 8.7B). Unlike cogongrass, garlic mustard was introduced in the North, probably New England (Welk and others 2002), and is currently spreading southward. The estimate of current expansion pressure for garlic mustard includes northern and central Kentucky, Virginia, western North Carolina, and northwestern Arkansas. While cogongrass is expanding rapidly north from the Gulf Coast, garlic mustard appears to be spreading rapidly from the North into the South.

This research illuminates the value of continuous, long-term monitoring of invasive plant species in understanding where resources may best be allocated to deter expansion, particularly into ecologically sensitive or protected areas. Knowledge of the expansion rates and directions of invasive plants is increasingly important in the context of global warming, which could increase rates of northern expansion of some species. Regional data viewed in a national context provides insight to policymakers and stakeholders, while national data provides context for local and regional research. Harmonization of data collection procedures across regions in the future will allow for further cross-region exploration of the data.

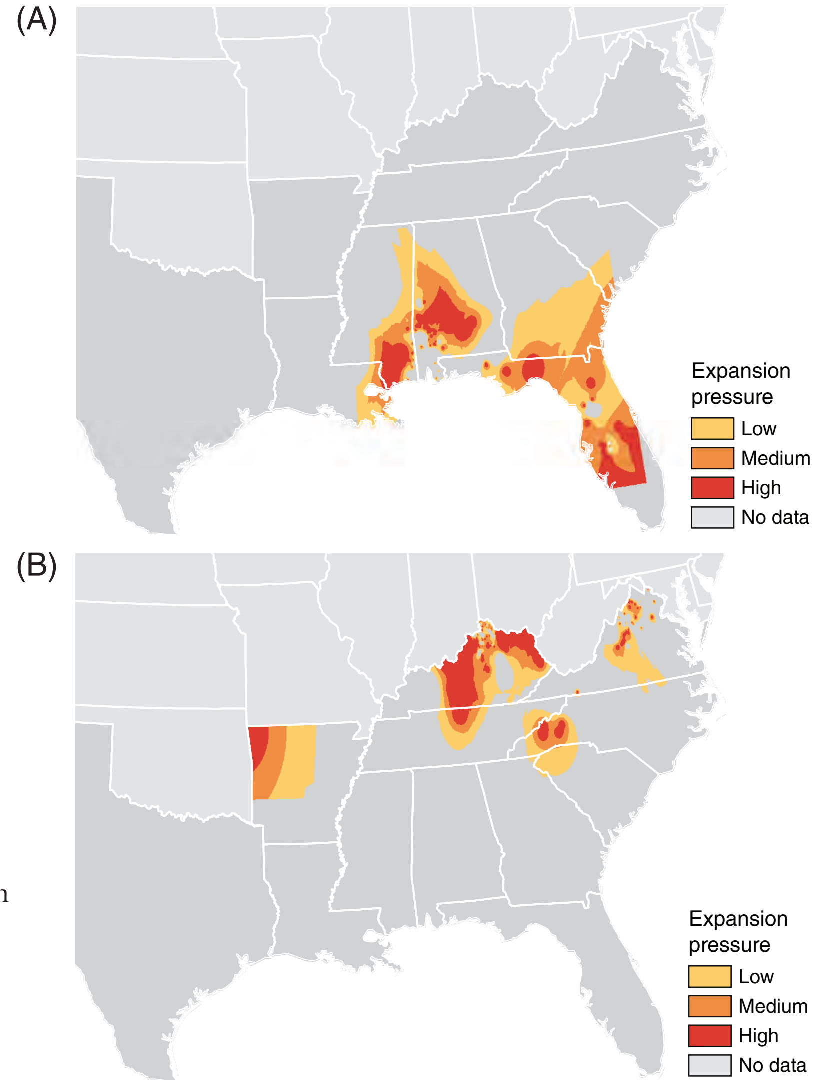

Figure 8.7— Expansion pressure of (A) cogongrass and (B) garlic mustard based on previously infested and newly infested plot locations.

ACKNOWLEDGMENTS

Numerous hard-working individuals throughout the United States collected the data in this chapter, and we thank them for their tireless efforts to document the presence of invasive plants on our Nation's forests. Additionally, we are grateful to Keith Moser, Cassandra Kurtz, Beth Schulz, and Andrew Gray for assembling the data from their respective regions. The authors also owe thanks to Ted Ridley and Jeff Turner for their help in processing and assembling data from the southern region. Thanks to Mark Nelson and Songlin Fei for their helpful review comments.

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CHAPTER 9.
Crown Condition

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INTRODUCTION

Tree crown conditions are visually assessed by the U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis (FIA) Program as an indicator of forest health. These assessments are useful because an individual tree's photosynthetic capacity is dependent upon the size and condition of its crown. In general, trees with full, vigorous crowns are associated with more vigorous growth rates (Zarnoch and others 2004), and when trees undergo stress, the first symptoms are often visible in the crown. Furthermore, tree crowns form the overstory structure of the forest and directly influence the composition and structure of the understory, thereby making them an integral component of the forest ecosystem.

Initially implemented by the Forest Service, Forest Health Monitoring (FHM) Program, crown conditions have been measured in the United States since 1990 (Randolph 2013). After a series of field tests and reviews in the early 1990s, the crown condition indicator was formalized to include a set of eight variables: vigor class, uncompacted live-crown ratio, crown light exposure, crown position, crown density, crown dieback, foliage transparency, and crown diameter (Schomaker and others 2007). When the FHM Detection Monitoring plots were incorporated into FIA in the year 2000, assessment of these and other forest health indicators was continued by FIA. Due to budget uncertainties in 2011, FIA halted collection of the forest health indicators, including crown condition (USDA Forest Service 2012). Along with budget constraints, emergent user needs and evolving forest health science have led FIA to incorporate some of its forest health indicators, among them crown condition, into a new framework termed "Phase 2 Plus / Ecosystem Indicator Program" (USDA Forest Service 2013). This new framework collects fewer variables on a greater number of plots in an effort to improve flexibility without compromising long-term analytical capabilities. Specific protocols for the new framework are under development by FIA. Recent analyses suggest that at minimum FIA should continue assessing uncompacted live-crown ratio and crown dieback as part of the Phase 2 Plus / Ecosystem Indicator Program (Morin and others 2012).

The last national reporting of crown condition was included in the 2006 *Forest Health Monitoring National Technical Report* (Randolph 2009) where I summarized data collected from 2000 through 2004. Geographic areas and species groups with poor conditions were identified, and those with unknown causes were investigated further (Randolph and others 2012). In the same manner, this report summarizes crown conditions for major species groups in the United States (2006–2010) and evaluates changes in crown condition during the last decade. Also included are comparisons to the crown conditions observed by the FHM Program between 1996 and 1999 (Randolph 2006; Randolph and Thompson 2010; Randolph and others 2010a, 2010b, 2010c).

METHODS

Data

I used publicly available crown condition data collected by the FIA Program between 2000 and 2010 (table 9.1) (O'Connell and others 2013). Crown density, crown dieback, and foliage transparency for live trees with a diameter of at least 5.0 inches at breast height were summarized by FIA species groups within each FIA region (fig. 9.1). Crown condition definitions and data collection protocols are outlined by Schomaker and others (2007). Briefly, crown density is the amount of crown biomass, i.e., branches, foliage, and reproductive structures, that blocks light visibility through the projected crown outline. Foliage transparency is the amount of skylight visible through the live, normally foliated portion of the crown. Crown dieback is the recent mortality of branches with fine twigs, which begins at the terminal portion of a branch and proceeds toward the trunk. All three variables are assessed by means of ocular estimation and recorded in 5-percent classes. High levels of crown dieback indicate potentially serious declines in tree health, while low levels of crown density and high levels of transparency may indicate greater amounts of defoliation and signal that a tree may have a reduced capacity for growth.

Table 9.1—Years of data included in crown condition analyses by State

Measurement years	States
2000–10	Indiana, Iowa, Maine, Michigan, Minnesota, Missouri, Pennsylvania, Utah, Wisconsin
2001–10	Arizona, California, Illinois, Kansas, Nebraska, Oregon, South Dakota, Washington
2001, 2003–09	North Dakota
2001–05, 2007–10	Ohio
2002–10	Alabama, Colorado, Florida, Montana, South Carolina, Tennessee, Texas
2002–05, 2007–10	New Hampshire, New York
2002–04, 2006–10	Arkansas
2002–04, 2006, 2009–10	Georgia
2002–05, 2009–10	Louisiana
2002–06, 2008–10	Virginia
2003–10	Connecticut, Kentucky, Massachusetts, Vermont
2003–07, 2009–10	North Carolina
2004–10	Alaska, Delaware, Idaho, Maryland, New Jersey, West Virginia
2004–05	Nevada
2004–07, 2009–10	Rhode Island
2008–10	New Mexico
2009–10	Mississippi, Oklahoma

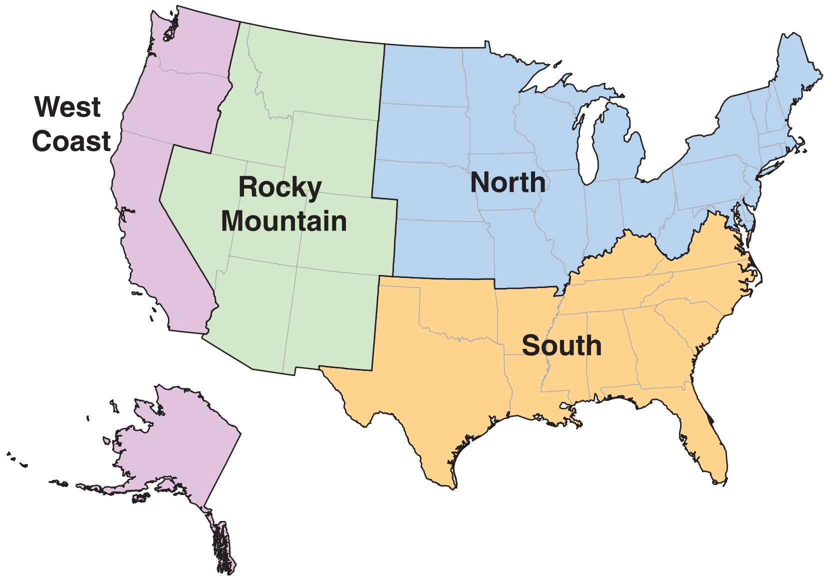

Figure 9.1— Regional breakdown of the United States for the crown condition analysis.

Analysis

The prescribed frequency of measurement for FIA forest health plots is once every five years; however, deviations from this ideal occurred between 2000 and 2010 due to a variety of factors. For example, some States in the East transitioned from a 5-year measurement cycle to a 7-year cycle after the first State-wide inventory was completed. Such factors produced irregular remeasurement periods (i.e., more or less than the ideal 5 years) for a small portion of the data. Analyses for this report dealt with the irregular measurement patterns thusly:

- Using the ratio of means estimator (Cochran 1977, Woodall and others 2011), 5-year moving averages and associated 95-percent confidence intervals were calculated for crown density, crown dieback, and foliage transparency. The moving averages were calculated for the general hardwoods and softwoods groups and for FIA species groups within each region for the years 2004–2010. These moving averages are referenced by their ending year, e.g. the "2007 moving average" is based on data from 2003–2007. Only the most recent assessment for each 5-year time period was used for plots that happened to have two assessments within the selected timeframe. For example, if a plot was measured in 2004 and 2005, the 2005 moving average included the 2005 assessment but not the 2004 assessment.
- Tests for significant changes in crown condition during the 2000s were performed by comparing the 95-percent confidence interval of the 2004 moving average to that of the 2010 moving average. The averages were declared statistically significantly different if the two confidence intervals did not overlap. This method is a more conservative approach (i.e., significant differences may not be detected when they truly exist) than the standard method, which examines the confidence interval for the difference between two means (i.e., $mean_1$ - $mean_2$) (Schenker and Gentleman 2001). This approach was necessary because the groups being compared

included a mixture of paired and non-paired trees.[1] Only species groups measured on at least 100 plots during both time periods were included in the tests. Tests for significant differences were performed on the 2004 and 2010 moving averages for the hardwoods, softwoods, and each species group within each region. Data for the West Coast and South were not available until 2001 and 2002, respectively (table 9.1), which shortened the 2004 5-year moving average to a 4-year moving average for the West Coast and to a 3-year moving average for the South.

FIA has established measurement quality objectives (MQO) for each variable in its inventory. For crown density, crown dieback, and foliage transparency, 90 percent of the assessments by two independent field crews are expected to be within ±10 percent (two classes) of each other (Schomaker and others 2007). Quality assurance data collected between 2002 and 2004 (Westfall and others 2009) indicated that field crews in all regions met the MQO for crown dieback. Field crews in the West Coast region, Rocky Mountain region, and north-central portion of the North region met the MQO for foliage transparency. MQO for crown density were not attained in any region. Given these quality assurance results, it is evident that estimates of crown density and, depending on the region, estimates of foliage transparency include more variation due to inconsistency among observers than estimates of crown dieback. In general, statistically significant changes in crown density that approach ±10 percent from time 1 to time 2 and statistically significant changes in foliage transparency that exceed ±5 percent are worthy of further investigation. Any species or species group with statistically significant changes in more than one of the crown variables, regardless of the magnitude of change, also should be investigated further.

The 5-year moving average was used to assess trends over time and to report the current (2010) conditions. Annual means were calculated for each species group for the years 2000–2010 to shed light on the 5-year moving average trend lines. The spatial distribution of mean crown conditions were examined visually by mapping the plot means. All plots with measurement years of 2006–2010 were displayed for this analysis if the plot contained at least five trees of the species group of interest. Displays were based on the perturbed ("fuzzed") geographic coordinates (McRoberts and others 2005). Maps of the plot means showed nothing extraordinary overall; therefore, only a limited number of examples are presented in the results.

[1] Bechtold, W.A.; Randolph, K.C. 2006. FIA crown-condition indicator workshop outline and class notes. 70 p. Unpublished report. On file with: U.S. Department of Agriculture Forest Service, Southern Forest Inventory and Analysis Program, 4700 Old Kingston Pike, Knoxville, TN 37919.

RESULTS AND DISCUSSION

West Coast Region

A significant increase in crown dieback was observed for the West Coast hardwood group, which consisted mostly of species from the *Quercus* genera (table 9.2). Some extreme year-to-year fluctuations in crown dieback were evident for the hardwoods; particularly notable were averages >6.0 percent in 2005 and 2006 (fig. 9.2). The effect of these years on the 5-year moving average was evident throughout the measurement period, as the moving average increased between 2005 and 2009 and then declined in 2010 when the 2005 high dropped out of the calculation. The 2010 moving average was 4.8 percent, only slightly higher than the average observed for hardwoods in California, Oregon, and Washington between 1996 and 1999 (fig. 9.2).

Significant changes in crown density, crown dieback, and foliage transparency were observed for the West Coast softwood group, which consisted mostly of species from the *Pseudotsuga, Pinus, Tsuga, Abies,* and *Picea* genera (table 9.2). The changes in all three crown variables were indicative of declining crown conditions. Crown

Table 9.2—Mean crown conditions for major species groups in the West Coast region of the United States, 2006–10

			Crown density			Crown dieback			Foliage transparency		
Species group	Plots	Trees	Mean	SE[a]	Change[b]	Mean	SE[a]	Change[b]	Mean	SE[a]	Change[b]
			%			%			%		
Softwoods	518	10,152	39.9	0.54	-	2.9	0.17	+	22.4	0.43	+
Douglas-fir	223	2,565	38.2	1.18	-	1.3	0.17	0	22.4	1.12	0
Lodgepole pine	63	693	35.3	1.45	NA	3.1	0.52	NA	22.5	1.24	NA
Ponderosa and Jeffrey pines	103	825	37.9	1.76	0	2.1	0.31	0	23.7	1.23	0
Sitka spruce	83	555	46.2	1.32	NA	4.8	1.05	NA	25.1	1.26	NA
True fir	134	1,385	40.8	1.47	-	2.8	0.42	+	18.7	0.57	+
Western hemlock	149	1,641	40.8	0.94	-	3.6	0.37	+	25.4	0.78	+
Hardwoods	211	2,739	36.8	0.69	0	4.8	0.46	+	26.2	0.74	0
Oak	92	1,199	36.0	0.92	NA	5.5	0.73	NA	25.0	1.19	NA

[a] Standard error.
[b] Test that the mean for 2006–10 is significantly different from the mean for 2000–04 based on overlapping 95-percent confidence intervals. (+) indicates the mean for 2006–10 was greater than the mean for 2000–04, (–) indicates the mean for 2006–10 was less than the mean for 2000–04, and (0) indicates no significant difference. NA indicates the test for significant difference was not performed. Only species groups measured on at least 100 plots during both time periods were included in the tests.
Source: U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis Program.

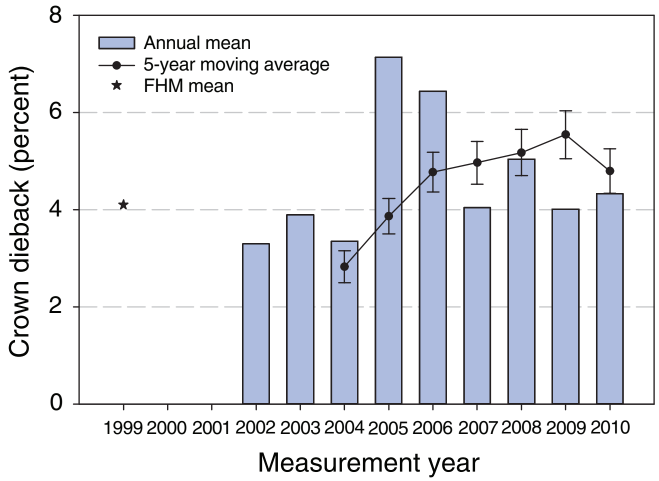

Figure 9.2—Mean crown dieback for hardwood species in the West Coast region of the United States. The annual mean in 2001 was zero percent. The 1999 mean is based on data collected in California, Oregon, and Washington by the Forest Service, Forest Health Monitoring (FHM) Program, 1996–99 (Randolph and others 2010a). Standard error bars are shown for each data point along the 5-year moving average trend line. (Additional data source: U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis Program)

density decreased from 45.0 percent to 39.9 percent, crown dieback increased from 1.4 percent to 2.9 percent, and foliage transparency increased from 19.2 percent to 22.4 percent (fig. 9.3). All of the 2010 moving averages indicated poorer conditions than those observed by FHM between 1996 and 1999 (fig. 9.3). The changes among the softwood crown conditions generally were gradual, with few unusually high or low annual means affecting the moving average. Further examination indicated that the decline in the softwood crown conditions were concentrated in western hemlock (*Tsuga heterophylla*), true fir (*Abies* spp., predominantly *A. concolor*), and Douglas-fir (*Pseudotsuga menziesii*) (table 9.2).

Crown density—Significant declines in crown density were observed for Douglas-fir, western hemlock, and true fir in the West Coast region (table 9.2). Average crown density for all three of these species was 49 to 50 percent between 1996 and 1999 (Randolph and others 2010a). By 2004, the moving averages for all three species had dropped to about 46 percent, and by 2010 had declined to <41 percent (fig. 9.4). The decline for western hemlock was fairly steady throughout the measurement period, whereas the declines for Douglas-fir and the true firs were influenced by large decreases in 2006 (fig. 9.4).

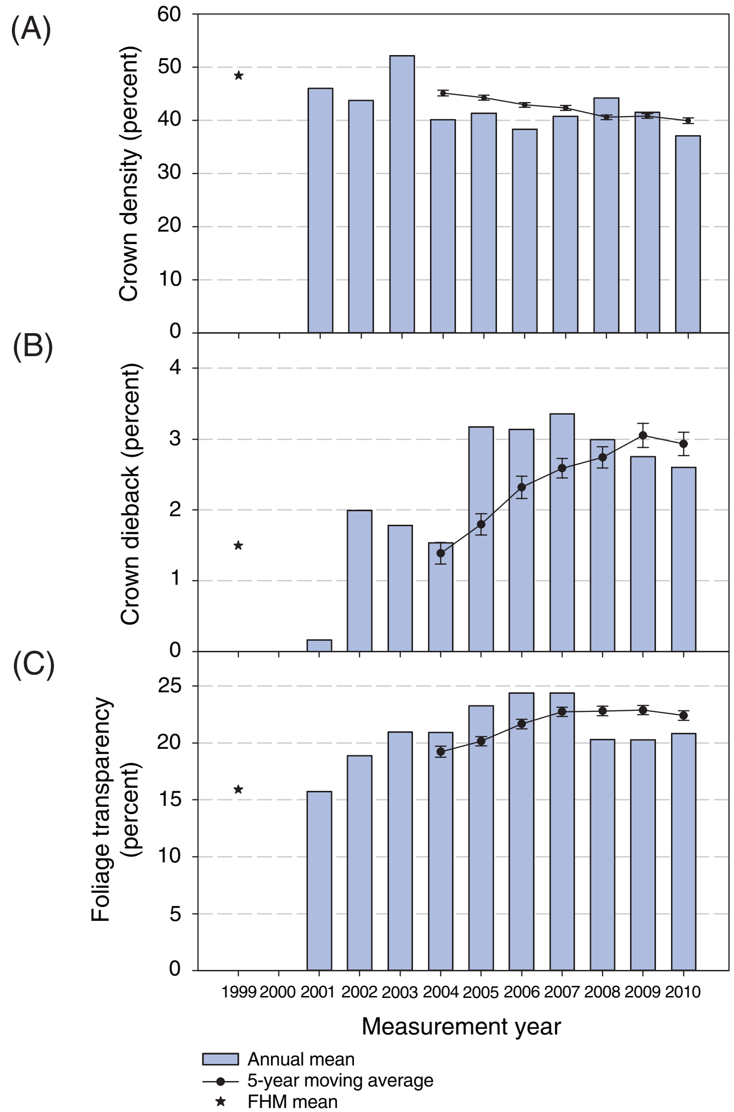

Figure 9.3— (A) Mean crown density, (B) mean crown dieback, and (C) mean foliage transparency for softwood species in the West Coast region of the United States. The 1999 mean is based on data collected in California, Oregon, and Washington by the Forest Service, Forest Health Monitoring Program (FHM), 1996–99 (Randolph and others 2010a). Standard error bars are shown for each data point along the 5-year moving average trend line. (Additional data source: U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis Program)

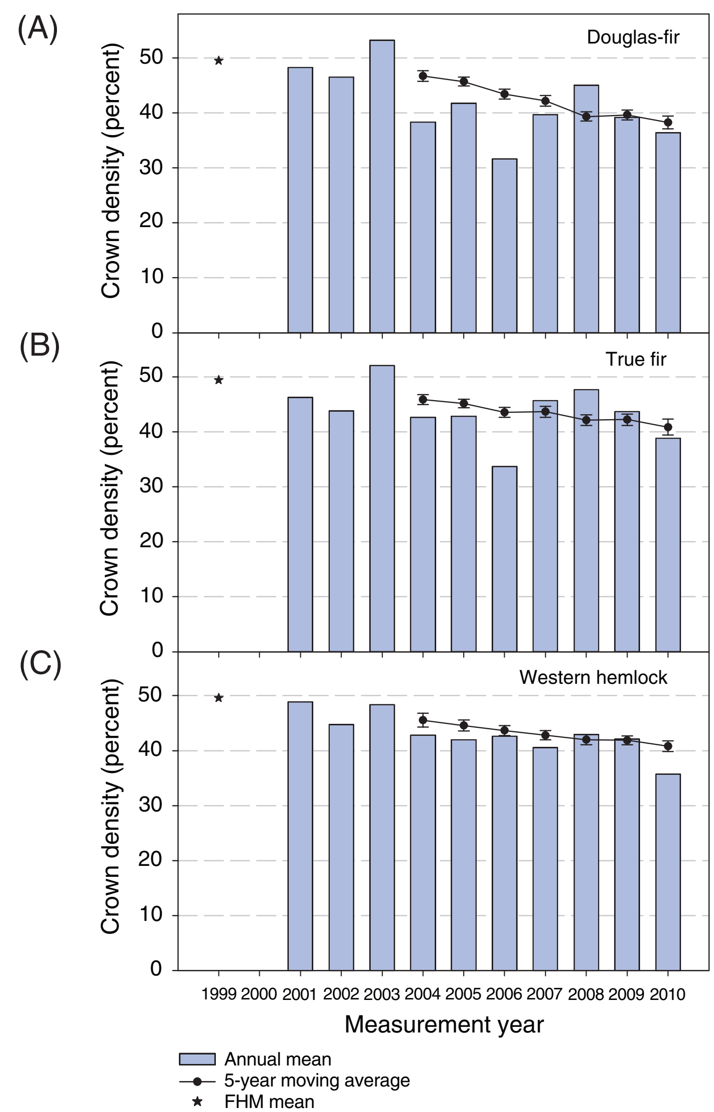

Figure 9.4—Mean crown density for (A) Douglas-fir, (B) true fir, and (C) western hemlock in the West Coast region of the United States. The 1999 mean is based on data collected in California, Oregon, and Washington by the Forest Service, Forest Health Monitoring Program (FHM), 1996–99 (Randolph and others 2010a). Standard error bars are shown for each data point along the 5-year moving average trend line. (Additional data source: U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis Program)

Crown dieback—Significant increases in crown dieback were observed for true fir and western hemlock in the West Coast region during the 2000s (table 9.2). Between 1996 and 1999, both species groups averaged <2.0 percent crown dieback (Randolph and others 2010a), but by 2010, the moving average had risen to 2.8 percent for true fir and 3.6 percent for western hemlock (table 9.2). The pattern of increasing crown dieback was characterized by a relatively steady increase throughout the measurement period for true fir and by a peak in the middle of the measurement period for western hemlock (fig. 9.5). An examination of the 2006–2010 plot averages indicated that western hemlock crown dieback was, in general, higher in southeast Alaska than in coastal Washington and Oregon (fig. 9.6).

Foliage transparency—Significant increases in foliage transparency were observed for true fir and western hemlock in the West Coast region during the 2000s (table 9.2). The change in foliage transparency for the true fir group was steady and statistically significant, but rather small, increasing from a 2004 moving average of 16.1 percent to 18.7 percent in 2010 (fig. 9.7). The change in western hemlock foliage transparency was larger, increasing from 20.5 percent in 2004 to 25.4 percent in 2010 (fig. 9.7). The change for

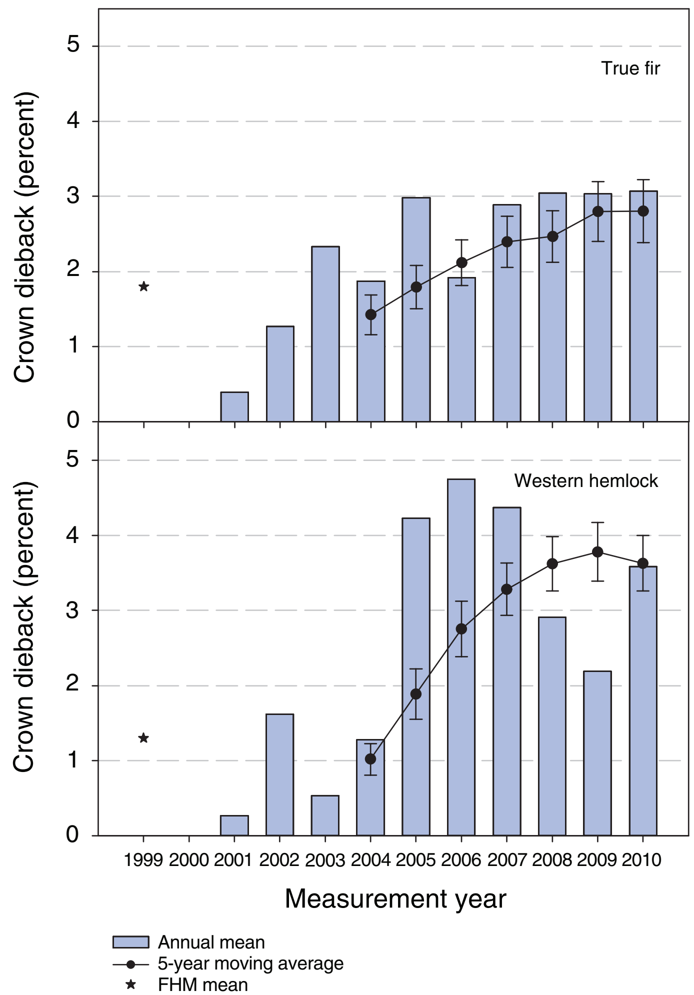

Figure 9.5—Mean crown dieback for true fir and western hemlock in the West Coast region of the United States. The 1999 mean is based on data collected in California, Oregon, and Washington by the Forest Service, Forest Health Monitoring (FHM) Program, 1996–99 (Randolph and others 2010a). Standard error bars are shown for each data point along the 5-year moving average trend line. (Additional data source: U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis Program)

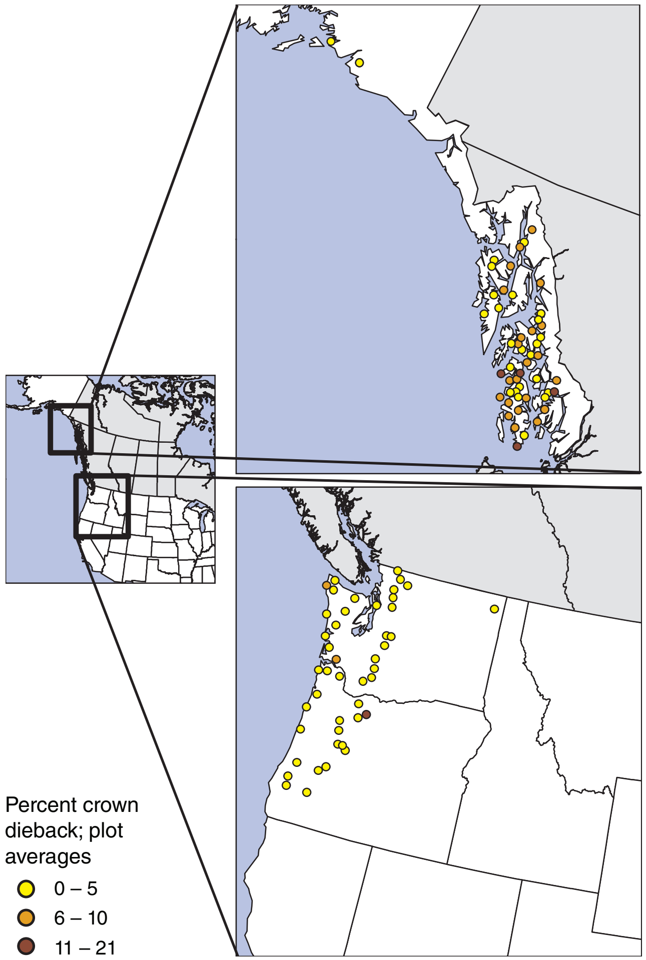

Figure 9.6—Western hemlock crown dieback plot averages, 2006–10. Plot locations are approximate. (Data source: U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis Program)

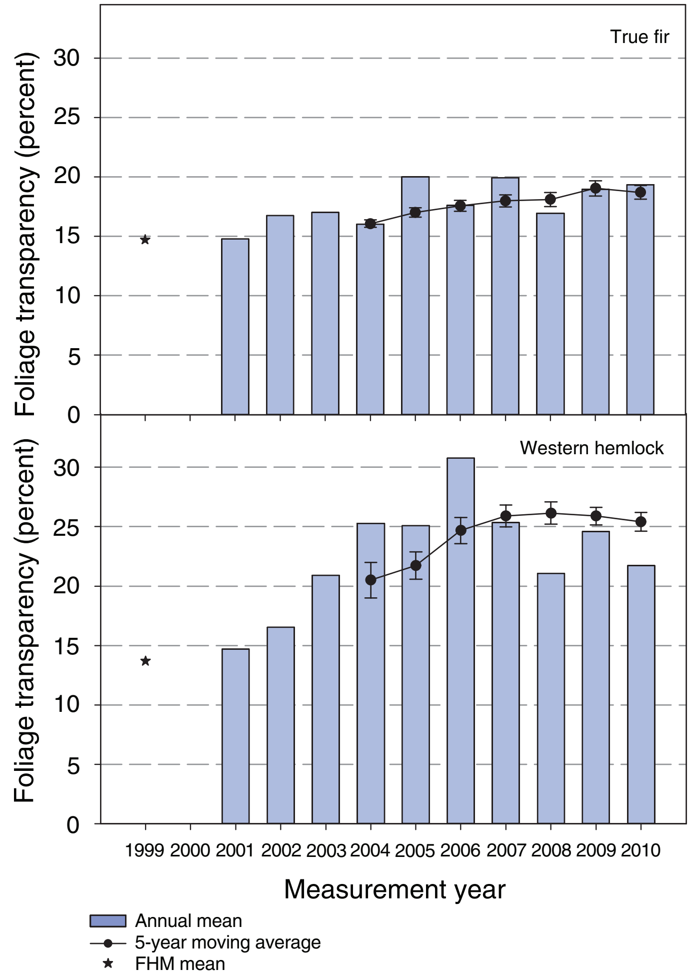

Figure 9.7—Mean foliage transparency for true fir and western hemlock in the West Coast region of the United States. The 1999 mean is based on data collected in California, Oregon, and Washington by the Forest Service, Forest Health Monitoring (FHM) Program, 1996–99 (Randolph and others 2010a). Standard error bars are shown for each data point along the 5-year moving average trend line. (Additional data source: U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis Program)

western hemlock is especially concerning given that the mean observed between 1996 and 1999 was 13.7 percent (Randolph and others 2010a). An examination of the 2006–10 plot averages indicated that western hemlock foliage transparency was poorer in Alaska and the Olympic Peninsula of Washington than farther south along the Pacific Coast and elsewhere in the region (fig. 9.8).

Rocky Mountain Region

A significant increase in crown density was observed for the Rocky Mountain softwood group (table 9.3). This is likely due to the pinyon-juniper group, the only group with a significant change in crown condition (table 9.3). Increases in mean crown density are considered improvements in condition and typically result from added biomass in the crowns. However, increases in the mean also could result from trees with poor conditions at time 1 dying and dropping out of the crown assessments before time 2, which causes an increase in the mean of the remaining trees.

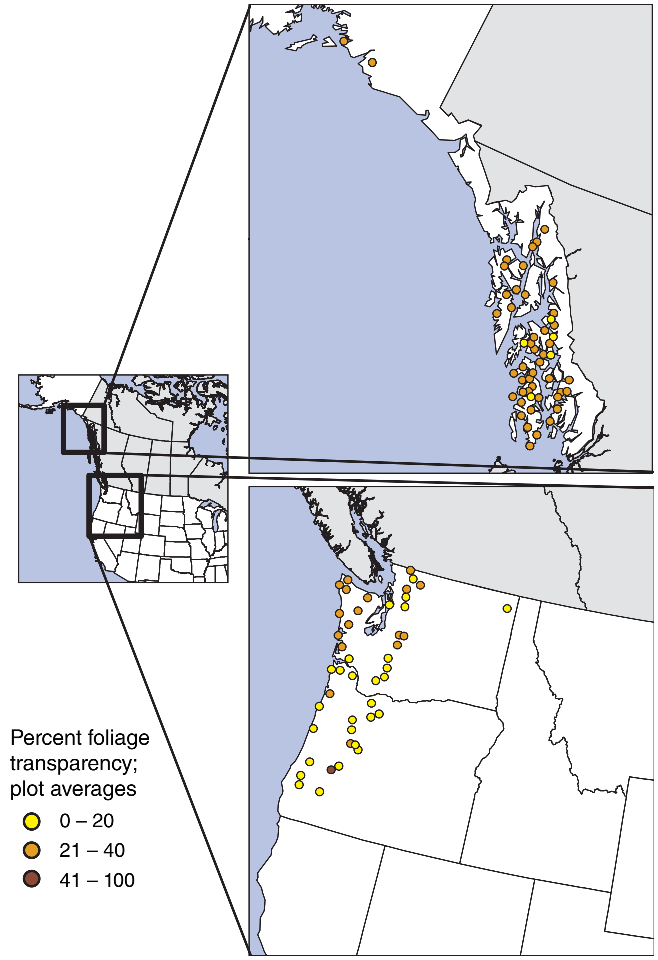

Figure 9.8—Western hemlock foliage transparency plot averages, 2006–10. Plot locations are approximate. (Data source: U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis Program)

Table 9.3—Mean crown conditions for major species groups in the Rocky Mountain region of the United States, 2006–10

Species group	Plots	Trees	Crown density			Crown dieback			Foliage transparency		
			Mean	SE[a]	Change[b]	Mean	SE[a]	Change[b]	Mean	SE[a]	Change[b]
			%			%			%		
Softwoods	574	10,824	45.8	0.47	+	3.6	0.20	0	15.9	0.17	0
Douglas-fir	191	1,720	44.1	0.89	0	2.4	0.46	0	16.0	0.36	0
Engelmann and other spruces	105	1,003	48.4	1.16	0	1.9	0.38	0	14.3	0.37	0
Lodgepole pine	94	1,530	38.7	1.02	NA	3.5	0.61	NA	16.9	0.54	NA
Pinyon-juniper	266	3,381	50.4	0.84	+	5.9	0.32	0	15.7	0.29	0
Ponderosa and Jeffrey pines	118	1,041	40.2	1.09	0	2.1	0.32	0	18.1	0.43	0
True fir	163	1,514	48.3	0.98	0	2.6	0.35	0	14.4	0.34	0
Hardwoods	163	1,660	39.1	1.66	0	6.0	0.66	0	20.2	0.48	0
Cottonwood and aspen	69	888	33.9	1.58	NA	3.4	0.56	NA	19.8	0.61	NA

[a] Standard error.
[b] Test that the mean for 2006–10 is significantly different from the mean for 2000–04 based on overlapping 95-percent confidence intervals. (+) indicates the mean for 2006–10 was greater than the mean for 2000–04, (–) indicates the mean for 2006–10 was less than the mean for 2000–04, and (0) indicates no significant difference. NA indicates the test for significant difference was not performed. Only species groups measured on at least 100 plots during both time periods were included in the tests.
Source: U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis Program.

North Region

In the North region, no changes were observed in crown density, whereas two species groups, beech (*Fagus grandifolia*) and maples (*Acer* spp.), displayed decreases in crown dieback, and five individual species groups, in addition to the general hardwood and softwood groups, displayed increases in foliage transparency (table 9.4). Though decreases in crown dieback are considered improvements in crown condition, the improvement for beech was much more dramatic than that for the maples (fig. 9.9). Although they were statistically significant, the increases in foliage transparency for black walnut (*Juglans nigra*), hickory (*Carya* spp.), maples, northern white-cedar (*Thuja occidentalis*), and spruce (*Picea* spp.) and balsam fir (*Abies balsamea*) were all <4.0 percent, and therefore likely biologically unimportant.

Table 9.4—Mean crown conditions for major species groups in the North region of the United States, 2006–10

Species group	Plots	Trees	Crown density			Crown dieback			Foliage transparency		
			Mean	SE[a]	Change[b]	Mean	SE[a]	Change[b]	Mean	SE[a]	Change[b]
			%			%			%		
Softwoods	952	12,055	48.6	0.33	0	2.4	0.14	0	19.4	0.22	0
Eastern hemlock	207	1,316	47.7	0.91	0	1.5	0.20	0	18.1	0.35	0
Eastern white and red pines	262	2,081	48.5	0.65	0	1.3	0.17	0	20.5	0.50	0
Northern white-cedar	160	2,060	44.3	0.96	0	4.8	0.60	0	22.3	0.79	+
Spruce and balsam fir	426	3,978	52.0	0.51	0	2.2	0.17	0	17.7	0.22	+
Hardwoods	1,874	32,772	47.3	0.16	0	4.1	0.10	0	21.1	0.12	0
Ash	550	2,176	44.7	0.60	0	6.3	0.84	0	23.0	0.39	0
Basswood	172	659	47.0	0.89	0	2.9	0.41	0	19.9	0.37	0
Beech	304	1,208	46.7	0.60	0	4.1	0.36	-	19.7	0.37	0
Black walnut	152	402	46.8	0.61	0	4.4	0.60	0	23.5	0.69	+
Cottonwood and aspen	390	2,270	45.9	0.63	0	3.7	0.35	0	23.4	0.47	0
Hickory	419	1,407	49.7	0.43	0	2.7	0.24	0	20.1	0.29	+
Maples	1,176	9,708	48.2	0.24	0	3.0	0.11	-	19.7	0.19	+
Red oaks	708	2,781	47.8	0.33	0	4.6	0.28	0	21.1	0.26	0
Tupelo and blackgum	150	331	49.9	0.77	NA	2.6	0.73	NA	19.6	0.69	NA
White oaks	578	3,327	46.6	0.36	0	3.9	0.18	0	21.0	0.27	0
Yellow birch	257	845	49.6	0.58	0	3.3	0.34	0	19.4	0.30	0
Yellow-poplar	158	616	52.1	0.72	NA	3.0	0.62	NA	20.3	0.74	NA

[a] Standard error.
[b] Test that the mean for 2006–10 is significantly different from the mean for 2000–04 based on overlapping 95-percent confidence intervals. (+) indicates the mean for 2006–10 was greater than the mean for 2000–04, (–) indicates the mean for 2006–10 was less than the mean for 2000–04, and (0) indicates no significant difference. NA indicates the test for significant difference was not performed. Only species groups measured on at least 100 plots during both time periods were included in the tests.
Source: U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis Program.

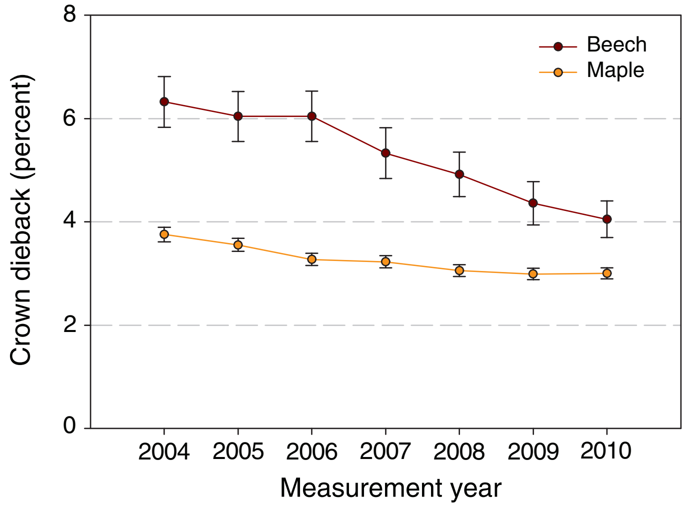

Figure 9.9—Five-year moving average trend line for beech and maple crown dieback in the North region of the United States, 2004–10. (Data source: U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis Program)

South Region

Significant increases in foliage transparency were observed for both the hardwood and softwood groups in the South region of the United States (table 9.5). At the species level, all but one species group (ash, *Fraxinus* spp.) displayed increases in foliage transparency, but as in the North, all changes were <4.0 percent. One species group, loblolly and shortleaf pine (*Pinus taeda* and *P. echinata*), displayed a decrease in crown dieback; however, the decrease was negligible (from 0.3 percent to 0.2 percent). No changes were observed in crown density.

National Observations

In general, crown conditions across the United States were stable during the last decade. Though some changes in crown condition were observed, many of the statistically significant changes were relatively small and likely biologically unimportant. Notable exceptions to this were the declining crown conditions among the hardwoods, western hemlock, and true firs in the West Coast region. These declines may be the result of Sudden Oak Death (*Phytophthora ramorum*) among oaks in California and Oregon, western black-headed budworm (*Acleris gloverana*) on hemlocks in Alaska, and western spruce budworm (*Choristoneura occidentalis*) and fir engraver beetle (*Scolytus ventralis*) on the true firs (Man 2009, Snyder and others 2008). However, the increases in western hemlock and hardwood crown dieback between 2005

Table 9.5—Mean crown conditions for major species groups in the South region of the United States, 2006–10

Species group	Plots	Trees	Crown density Mean	Crown density SE[a]	Crown density Change[b]	Crown dieback Mean	Crown dieback SE[a]	Crown dieback Change[b]	Foliage transparency Mean	Foliage transparency SE[a]	Foliage transparency Change[b]
			%			%			%		
Softwoods	1,164	17,385	41.5	0.28	0	0.4	0.05	0	23.7	0.27	+
Loblolly and shortleaf pines	783	11,830	41.4	0.35	0	0.2	0.03	-	23.5	0.33	+
Longleaf and slash pines	168	2,636	41.1	0.60	NA	0.3	0.10	NA	22.8	0.61	NA
Virginia pine	119	673	39.4	0.79	NA	1.1	0.28	NA	27.6	1.35	NA
Hardwoods	1,631	23,116	43.6	0.21	0	2.4	0.10	0	22.9	0.18	+
Ash	247	759	42.1	0.69	0	2.9	0.46	0	23.1	0.57	0
Beech	106	239	48.5	1.2	NA	1.6	0.55	NA	19.6	0.54	NA
Hickory	525	1,622	46.2	0.43	0	1.4	0.19	0	21.2	0.40	+
Maples	640	2,453	41.9	0.37	0	2.2	0.23	0	22.4	0.32	+
Red oaks	951	3,564	44.7	0.34	0	2.7	0.21	0	22.9	0.27	+
Sweetgum	598	2,389	44.4	0.38	0	2.0	0.24	0	21.1	0.36	+
Tupelo and blackgum	418	1,533	41.1	0.88	0	1.6	0.41	0	23.7	0.61	+
White oaks	771	3,699	44.6	0.39	0	2.1	0.18	0	22.6	0.37	+
Yellow-poplar	404	1,613	44.1	0.53	0	1.4	0.26	0	20.7	0.42	+

[a] Standard error.
[b] Test that the mean for 2006–10 is significantly different from the mean for 2000–04 based on overlapping 95-percent confidence intervals. (+) indicates the mean for 2006–10 was greater than the mean for 2000–04, (–) indicates the mean for 2006–10 was less than the mean for 2000–04, and (0) indicates no significant difference. NA indicates the test for significant difference was not performed. Only species groups measured on at least 100 plots during both time periods were included in the tests.
Source: U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis Program.

and 2007 coincided with an increase in western hemlock foliage transparency and decreases in Douglas-fir and true fir crown density in 2006. This coincident pulse of degraded crown conditions may suggest a stressor event other than the insects and diseases known to be present in the West Coast region.

Regional differences were observed in the crown condition means for several of the species groups that crossed the FIA unit boundaries by which the data were summarized (fig. 9.1) (though no formal tests were performed to determine statistical differences). Four species groups were summarized in both the West Coast and Rocky Mountain regions. With the exception of Douglas-fir and lodgepole pine (*Pinus contorta*) crown dieback, crown conditions in the Rocky Mountain region were better or approximately equal to the crown conditions in the West Coast region (tables 9.2 and 9.3). Eight species groups were summarized for both the North and South regions. Crown density and foliage transparency in the North were approximately equal to or better than the conditions in the South; however, crown dieback was better (i.e., lower) in the South, and sometimes substantially so, e.g., ash, beech, and red oaks (*Quercus* spp.) (tables 9.4 and 9.5). One species group, cottonwood-aspen, spanned the Rocky Mountain and North regions. Crown dieback and foliage transparency conditions in the Rocky Mountain region for this species group were approximately equal to those in the North region; however, crown density was much lower in the Rocky Mountain region than in the North (tables 9.3 and 9.4). The differences within species across regions may signify actual differences in the condition, i.e., health, of the trees, but also potentially reflect differences in climate and other factors that affect growing conditions, e.g., forest management practices.

During the last decade, several species throughout the United States were imperiled by insect and disease outbreaks. Among these were the western pines (*Pinus* spp.), endangered by a host of bark beetles, particularly the mountain pine beetle (*Dendroctonus ponderosae*), and the eastern ashes, threatened by the emerald ash borer (*Agrilus planipennis*). Although no changes in crown condition were observed for either species group during the 2000s, the 2010 crown density moving averages for lodgepole pine, ponderosa pine (*Pinus ponderosa*), and Jeffrey pine (*Pinus jeffreyi*) in the West Coast region were substantially lower than the average conditions observed between 1996 and 1999. Mean crown density for lodgepole pine was 43.3 percent in 1999 (Randolph and others 2010a) and 35.3 percent in 2010 (table 9.2). Mean crown density for the ponderosa and Jeffrey pines was 47.2 percent in 1999 (Randolph and others 2010a) and 37.9 percent in 2010 (table 9.2). The ash group has maintained a high mean level of crown dieback in the Northern United States since the late 1990s, averaging 5.3 percent in the Northeast and 5.7 percent in the North Central States between 1996 and 1999 (Randolph and others 2010b, 2010c) and 6.3 percent across the entire northern region in 2010 (table 9.4).

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SECTION 3.

Evaluation Monitoring Project Summaries

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 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. Five project summaries are included in this report.

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CHAPTER 10.

Multi-State Beech Bark Disease Survey and Beech Scale Resistance

(Project NE-EM-B-11-01)

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INTRODUCTION

Beech bark disease (BBD) remains the most serious threat to American beech (*Fagus grandifolia*) in its native range. Pathologists have historically described BBD as a predictable interaction between the *Cryptococcus fagisuga* scale insect and a *Neonectria* canker-forming fungus. It is well known that beech trees infested by *C. fagisuga* become predisposed to *Neonectria* colonization (Houston and O'Brien 1983).

This cooperative effort involved Delaware, New Jersey, Maryland, and West Virginia. The purposes of this study were (1) to identify potentially BBD-resistant beech trees in areas with BBD-associated decline and mortality, and (2) to establish permanent plots containing beech trees in four States where trees will be monitored for general health conditions including BBD.

METHODS, RESULTS, AND DISCUSSION

In Delaware, BBD had never been recorded, although no formal surveys had yet been carried out. Therefore, the primary goal was to identify suitable survey sites and to initiate a BBD survey. During the 2011 field season, four permanent survey sites were established. Three were in New Castle County, the county closest to the known range of BBD. The fourth site was in Kent County. At each site, a starting point was chosen in the interior of the stand, away from edge influence. The closest tree with diameter at breast height (d.b.h.) of ≥10 inches was selected as the first study tree. Subsequent trees were selected based on closest proximity to the previous study tree. A total of 116 beech trees were surveyed at the four sites. For each tree, the d.b.h. was recorded and visual estimates were made for crown transparency, percent crown dieback, percent of crown with foliar discoloration, and trunk decay. GPS coordinates were recorded for each survey tree, and each was photographed with a digital camera to facilitate follow-up work in coming years. Stand information was also recorded, including slope, aspect, elevation, soil type, and overstory and understory vegetation types at each site.

The 2011 surveys in Delaware did not detect *Cryptococcus fagisuga* or decline due to *Neonectria*. The obvious conclusion based on this first survey is that BBD is not yet established in Delaware. Follow-up surveys using the same methodology were conducted in 2012 at the same four sites, and again no scale or mortality was observed. All 116 trees in the 2011 survey were alive during the 2012 survey, and overall condition of study trees remained the same.

Study sites will continue to be revisited annually to continue gathering baseline data in anticipation of possible introduction of BBD in the future. Should BBD be identified at these sites in coming years, this study will be of value in two ways. First, the exact year of first appearance will be well documented, assisting in rate-of-spread analysis. Second, the effects of this disease on growth rate will be easily quantifiable given multiple years of baseline data. At that time, the study would be able to enter a second phase in which putatively resistant trees are identified.

In West Virginia, BBD has been monitored for more than 30 years and has been well documented as a cause of mortality. Efforts in this State have (1) focused on locating scale-resistant beech in areas where both scale and BBD-induced mortality are known to be present, and (2) determined the etiology and extent of scale infestation and fungal colonization. In 2012, 10 locations were selected as potential survey sites for identification of putatively resistant beech. Nine of these sites were ultimately included as actual survey sites. The sites were located in Tucker, Randolph, and Pocahontas Counties where beech mortality and scale presence were previously documented. Each stand contained up to 20 potentially resistant trees >9 inches in d.b.h. Each of these trees was rated for scale presence and measured for d.b.h.

In addition to these beech resistance monitoring plots, another 10 plots were established throughout the State in 2013 as permanent survey sites. Each survey site consisted of a minimum of 20 mature beech trees > 9 inches in diameter. Each tree was examined for *Cryptococcus fagisuga* presence and quantity, *Neonectria* colonization, and presence of decay-associated fungi. Tree canopy conditions were measured and photographed for dieback, foliar discoloration, and crown transparency. Survey trees also were measured for d.b.h. and located using GPS coordinates.

In Maryland, as in West Virginia, BBD was known to be well established in some areas. For this study, sites in Garrett County were scouted with the goal of locating stands with moderate scale pressure. It was believed that these sites would provide the best opportunity to identify putatively BBD-resistant beech. No sites with moderate scale pressure were located. All sites examined were either very heavily infested, with "whitewashing" of all beech trunks due to scale abundance, or very minimally infested with low scale, and *Neonectria* populations were observed. Foresters, to date, were unable to locate a site in Garrett County that had a moderate scale population.

In 2013, staff established several permanent sites in western and central areas of the State to survey the incidence of scale and fungal populations. Some of these sites might be more appropriate for identification of putatively resistant beech trees.

In New Jersey, the disease was well established for at least 20 years within the Stokes State Forest in northern New Jersey. This State forest and the surrounding county contain the greatest basal area of American beech in New Jersey. It was assumed that scale and disease would spread southward as time passed.

In 2012, surveys to determine BBD incidence and spread were carried out in several northern and central locations. A total of 293 beech trees were examined for scale and fungal colonization.

The survey did not show any evidence of scale or disease spread from north to south. In fact, even within the northern areas of the State, population levels of *Cryptococcus fagisuga* were very low and *Neonectria* fruiting bodies were difficult to find. For the most part, beech trees appeared healthy. With only slight disease pressure, resistant beech could not be identified. It is highly doubtful that the numerous trees examined in areas previously affected by BBD were resistant. Survivors were younger and might have been more disease tolerant than older trees, or may have simply escaped disease exposure.

A novel approach during this survey examined bark integrity based on suggestions that the fungus could form lethal sapwood cankers without necessarily fruiting. Sapwood in this case would be clearly discolored. An effort was made during the New Jersey survey to examine bark for weakness and/or discoloration due to fungal colonization. However, bark even on scale-infested trees appeared intact tightly appressed to sapwood without evidence of fungal colonization. Surveys for BBD often are limited to examinations made of the lower stem. This survey, however, provided a unique opportunity to examine upper-stem sections and branches of mature trees. Beech trees blown over during Hurricane Sandy made possible the examination of upper stems and canopy branches not visible from ground level. Even though only a limited number of fallen beech trees could be examined, it was apparent that bark integrity remained good throughout the entire tree. There was also no incidence of scale or *Neonectria* on upper canopy branches.

CONCLUSIONS

Permanent plots established in Delaware in 2011 as well as those scheduled for installation in West Virginia, New Jersey, and Maryland in 2013 provided baseline data that will most accurately describe scale and disease progression as well as general beech health conditions. This study clearly demonstrates the need for additional research in coming years. Permanent BBD plots will facilitate future research and provide an excellent opportunity to continually monitor and survey for this disease.

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CHAPTER 11.

Assessing Wildlife Habitat from a Large-Scale Forest Inventory

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INTRODUCTION

Large-scale forest inventory data sets are widely available today and include the USDA Forest Service programs Forest Inventory and Analysis (FIA, Phase 2, or P2) and Forest Health Monitoring (FHM, Phase 3, or P3) (Woodall and others 2011). Such data sets can be used to evaluate other aspects besides forest management, including wildlife management (Rudis 1991, Trani and others 2001) and tree and shrub diversity (Torras and others 2012). FIA data are the basis of State-level (Maine Forest Service 2010) and national-level strategic forest assessments. Allen and Plantinga (1999) explored biodiversity and wildlife habitat using FIA data, identifying tree diameter distribution, snags, downed woody debris, stand age, and shrub and herb cover as useful indicators.

Our principal objective was to use existing large-scale forest inventory data to describe the quality of wildlife habitat in Maine's forests particular to several wildlife species. Trani and others (2001) suggested that we might find important trends based on stand successional stages of plots in the current database. Three aspects were involved: (1) categorization of forest stands (subplots) into five successional categories, (2) application of inventory data variables into existing species specific habitat suitability index models, and (3) relation of successional categories in an ordination that emphasizes percent cover of tree species and select understory plants important to wildlife.

METHODS

Data

FIA phase 2 (P2) and phase 3 (P3) data from 2007 were used, representing 46 plots, 48 unique conditions, and 161 subplots. The subplot level, rather than plot or condition, was selected to maximize sample size. Bechtold and Patterson (2005), Schulz and others (2009), and Woodall and others (2011) provide details of FIA sampling design, measurement procedures, and variables. Two of the P3 sampling protocols used in our analysis are the Down Woody Material indicator (DWM) and the Vegetation Structure and Diversity indicator (VEG).

To contrast habitat stages, we used the term "successional category" and assigned FIA conditions according to criteria related to stand age, stand size class, and all live-tree stocking (table 11.1). The number of subplots per successional category is more or less evenly distributed. From an initial dataset that included 23 environmental variables, we used a Discriminant Functions Analysis (DFA) in SYSTAT to determine a subset of variables that had potential to distinguish successional categories.

P2 Variables

P2 variables included tree diameter at breast height (d.b.h.), species, tree history (live/dead/cut), tree class, forest type, elevation, site index, stand size class, stand age, physiographic class, stocking class, stand origin, and land use.

Table 11.1—Successional category definitions, criteria, and number of subplots

Definition	Number of subplots (n=161)
1 – Early succession (ES)—Stand age ≤30 years, stand size class is small diameter (saplings), and all live-stocking class is either well-stocked or overstocked; representing young trees or saplings, with a canopy closure at 4.9 m.	31
2 – Tending toward early succession (TES)	35
3 – Intermediate (INT), broadly defined with a possibly unrelated grab bag of forest types and stocking conditions	34
4 – Tending toward late succession (TLS)	38
5 – Late succession (LS)—Stand age is >60 years, stand size class is large diameter (sawtimber), and all live-stocking class is either moderately stocked, well-stocked, or overstocked	23

P3 Indicators

For DWM we linked decay class for dead wood, duff depth, and features of understory vegetation from both transect and microplot data to habitat requirements of animals that are known to require some specified volume or average piece size. Ground variables from the P3 VEG data and percent cover for vascular plants on the subplot were used by species and by species group (e.g., shrubs, trees, graminoids, etc.).

Habitat Suitability Index Models

To test applicability of the FIA data in assessing wildlife habitat quality, we sought Habitat Suitability Index models (HSI) developed for animals, with habitat preferences that represented a range of successional categories. Three species (ruffed grouse, American woodcock, and snowshoe hare) typically associated with early-succession habitats, one species (red spotted newt) linked to an intermediate successional stage, and four species (barred owl, pine marten, fisher, and pileated woodpecker) associated with later successional habitats were selected (Allen 1982, Allen 1983, Allen 1987, Schroeder 1982). In an effort to use the P3 VEG data, the eastern wild turkey was included and can be associated with either early or later successional categories. The pine marten HSI required percent cover estimates of coarse woody debris (CWD) (≥7.62-cm diameter at the line intersect). We visualized each CWD piece as a shadow on the ground at noon and calculated the area of each piece using the formula for a trapezoid: $A = a*(b1+b2)/2$, in which a = length, b1 = small diameter, and b2 = large diameter; totaling piece area by subplot, and then calculated its percent of the 168.1-m^2 subplot area.

Analytical software included SAS® 9.1 and SYSTAT® 12.0, and for data summaries the Microsoft® Excel pivot table function was useful.

The ordination analysis of successional category and important soft mast plants for migratory birds was inspired by a review of HSIs, current literature, and Martin and others (1951) regarding frugivores of Maine. Percent cover of live foliage was used as a rough proxy for soft mast resources, making a link for the supposed availability of fleshy fruits of understory plants such as *Rubus* spp., Northern fly honeysuckle, low sweet blueberry, viburnum, etc. We also included some P2 variables (e.g., stocking for all live trees, stand age). Three subplots lacked required data and were dropped, and to reduce noise in the 425 species x 158 subplot matrix, we excluded plant species present in <10 of 158 subplots, and all ferns and their allies, grasses, sedges, rushes, and a few herbs. The revised matrix included 57 seed-producing plants, including 14 fleshy-fruited plants. Ordination was conducted using PC-ORD™ 5.0 (MJM Software™, Gleneden Beach, OR) and Nonmetric Multidimensional Scaling (NMS). This technique has been well established as a robust approach toward identification of structure in community data and does not require assumption of normally distributed data. Quantitative variables were log-transformed to relativize data. A joint biplot was used to visualize relationships because some of the environmental variables, shown as vectors, better explain the relationship of species and successional category.

RESULTS

Discriminant functions analysis on the reliability of assigning the succession categories suggests imperfect classifications across all categories, with lowest confidence for the late successional category, with percent correct classification based on 8 variables varying from 18 percent to 100 percent (table 11.2).

Table 11.2—Results of discriminant functions analysis to classify subplots into successional categories

Succession category	1	2	3	4	5	Row total	Percent correct
1	31	0	0	0	0	31	100
2	4	22	0	5	4	35	63
3	1	4	22	0	5	32	69
4	0	3	0	29	4	36	81
5	0	0	8	10	4	22	18
Column total	36	29	30	44	17	156	69

Note: Only a match-up between row and column are a correct classification.

Habitat Suitability Index Models

The four animal species associated with mature forest conditions differed at least slightly in their habitat suitability index values for the two later-successional categories (fig. 11.1). Medians ranged from a low of 0.0, indicating poor habitat for pileated woodpeckers in both these successional categories, to a high of 0.39 for fishers in the late successional category, and overall the models indicate a scarcity of suitable habitat. We were unable to run all HSI models because (1) HSI methods were not specified to enable a match-up to FIA data; and (2) variables essential to some models were not available in the FIA datasets.

In the NMS analysis, the ordination of Axes 1 and 3 is shown with vectors for the six environmental variables and their relationship to species and subplots by their successional category (fig. 11.2). Longer vectors have more explanatory power than shorter ones, and when distance increases in a direction opposite the arrow and beyond the centroid, negative association is then assumed to increase. The ordination suggests a weak relationship between successional category and the plant species favored by migratory frugivores, because subplots within a successional category are scattered and do not form a cohesive group.

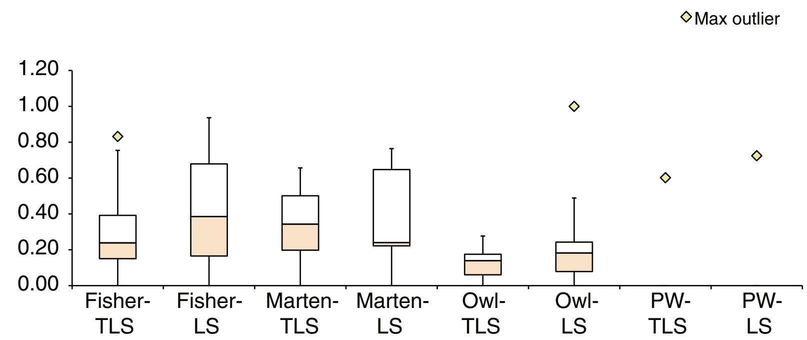

Figure 11.1—Box-whisker plot of the Habitat Suitability Index values for four species (fisher, pine marten, barred owl, and pileated woodpecker) in the "Tending toward late succession" (TLS) and "Late Succession" (LS) categories. PW= pileated woodpecker.

DISCUSSION AND CONCLUSIONS

The subplot was chosen as the primary sampling unit because the other options (plot, condition) delivered more aggregated information. We ignored the potential lack of independence of the subplots for increased replication in the HSI models, and given our objectives in this study, we consider that decision acceptable.

There is strong potential in the overall ability of a large-scale inventory to indicate wildlife habitat, especially for animal species that frequent mature forests. Index values

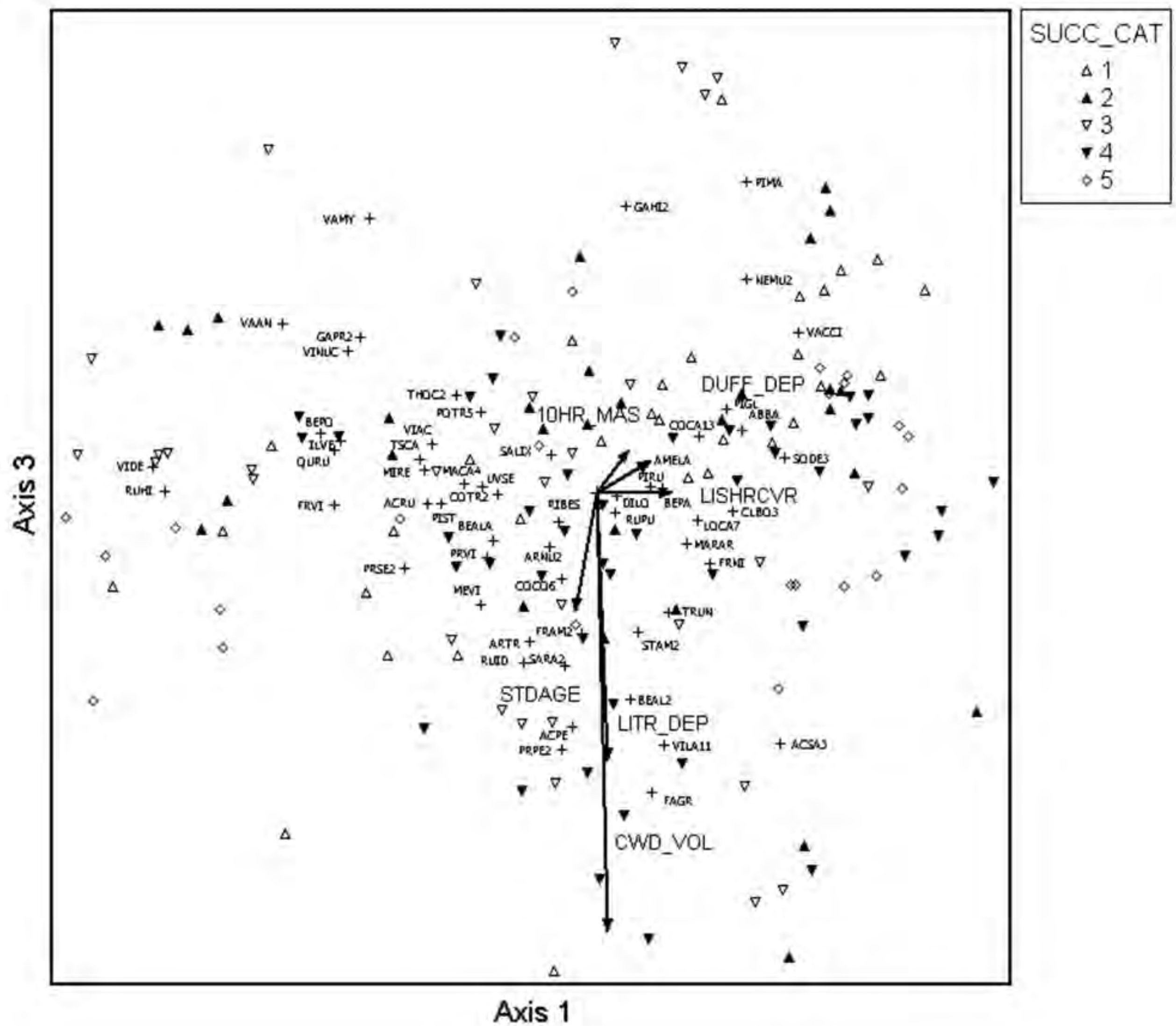

Figure 11.2—Joint biplot (first and third axes) of an ordination resulting from Nonmetric Multidimensional Scaling (NMS) of data from 158 subplots. Symbols are for subplots, by "Successional Category" (SUCC_CAT), and explained in table 11.1. Six select environmental variables, shown as vectors, (10HR_MAS = 10 Hour Fuel Loading; DUFF_DEP = Duff Depth; LISHRCVR = Live Shrub Cover; LITR_DEP = Litter Depth; CWD_VOL = Course Woody Debris Volume; and STDAGE = Stand Age) and 57 vascular plant species are displayed.

were higher in stands categorized as "late successional" compared to the "tending towards late successional" category. Even for pileated woodpeckers, the maximum value was greatest in late successional. The HSI failed to identify suitable habitat for pileated woodpecker due to very low frequency of live trees ≥50.8 cm d.b.h. and for snags ≥38.1 cm d.b.h. This finding is correlated by a real biodiversity benchmark (Maine Forest Service 2010). The pileated woodpecker HSI has such a major focus on large live trees and snags that FIA data within this model might be more appropriately analyzed at the condition or plot level in future studies. For fisher, a few subplots had highly suitable habitat, shown as outliers in figure 11.1. We think this demonstrates that HSI indices based on FIA data are sensitive enough to detect relatively subtle variation in habitat quality.

The needs of early successional species should not be overlooked, many of which are undergoing declines at a continental scale, and some of which are valued as game species. Trani and others (2001) used FIA data for 33 States to assess trends in availability of early successional wildlife habitat from 1946 through 1998. The available HSI models for early successional species require data not encompassed by FIA, such as cover of mature staminate aspen trees (catkins are winter food for ruffed grouse). HSI models for many early successional species have not been developed, presenting an opportunity for researchers to incorporate variables collected by FIA into new model development efforts.

We encountered many areas in which the FIA data suggest stands did not meet described thresholds. For example, the pine marten model ideally calls for 20–50 percent ground surface cover in coarse woody debris (≥7.62-cm diameter), but not >50 percent cover. We had a maximum of only 18 percent, implying that (1) there is inadequate CWD downfall for marten in our sample, (2) the model might not reflect conditions adequately, or (3) our calculations of CWD cover differed from methods used by Allen (1982), which were not specified.

The data were highly adaptable to the ordination technique, suggesting that other studies should follow. Individual subplot summaries reflect the scale at which a plant detects its light environment and vegetative competition. Importance of the shrub layer to some mammals and birds surely varies by animal species, shrub species, canopy conditions, and season. Shrub species are not clustered in the ordination (fig. 11.2), and this suggests that the P3 VEG data are especially important for understanding shrub cover at the level of subplot. Additional research in wildlife habitat should examine not only Maine's add-on P2 live shrub variables, but the P3 VEG data as well. There are many more opportunities for exploration of questions related to wildlife habitat using ordination techniques and the P3 VEG data. The national scope of FIA data provides a probabilistic sample, and is available across large regions of the country.

Our result that coarse woody debris volume is clearly important in explaining seed-bearing plant distribution is a start toward sorting out the relationship.

While FIA data are strong in quantification of components that represent the vertical structure, there is low emphasis on horizontal diversity, or patchiness (DeGraaf and others 2006). Wherever there is a distinct change in vegetation, that edge represents a different set of opportunities for wildlife, and these edges and openings tend to increase wildlife diversity. In the FIA data, a change in condition within a subplot or between subplots comes closest to delineating the desired edge feature.

We think that other crucial aspects of wildlife habitat can be addressed through the FIA data. For example, Meneguzzo and Hansen (2009) demonstrated the utility of Geographic Information Systems and FIA data in a study of fragmentation at three sites in Michigan, comparing plot data to two resolutions of satellite imagery. They concluded that FIA plot data have potential as an alternative to the use of imagery for assessing forest fragmentation. The other P3 core indicators include crown condition, lichen communities, forest soils, and ozone. Each of these could have implications for wildlife habitat quality. We hope to stimulate further research using the FIA data for wildlife habitat projects. This study provides analytical methods and results with wide applicability in other States or regions. Despite some limitations, we found the FIA data highly useful for exploring specific wildlife habitat questions.

CONTACT INFORMATION

Alison Dibble (Email: adibble2@gmail.com) can provide three appendices upon request: Appendix I–Maine Forest Service estimates of trends in mature forest area; Appendix II–Variables extracted from the Maine FIA/FHM data for correspondence to variables in eight HSI models; Appendix III–Summary statistics of select environmental values used in NMS ordination.

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CHAPTER 12.

Monitoring the Health of *Pinus strobiformis:* Early Impacts of White Pine Blister Rust Invasion

(Project INT-EM-B-10-03)

Christopher E. Looney
Kristen M. Waring
Mary Lou Fairweather

INTRODUCTION

Southwestern white pine (*Pinus strobiformis*, abbr. PIST), a tree native to Arizona, New Mexico, and western Texas in the U.S. Southwest (Little 1971), is threatened by a potentially lethal invasive fungal pathogen, *Cronartium ribicola* (white pine blister rust, abbr. WPBR) (Conklin and others 2009). Researchers detected the disease around 1990 in New Mexico's Sacramento Mountains (Hawksworth 1990), where it has since infected as much as 40 percent of the PIST population and inflicted increasing mortality (Conklin and others 2009). By 2004, WPBR was observed in PIST in New Mexico's Gila Mountains, and in 2009, researchers identified the first infected trees on the Fort Apache Indian Reservation in the White Mountains of Arizona (Fairweather and Geils 2011).

The purpose of this study was to provide a scientific basis for managing the WPBR invasion in Arizona and New Mexico by extending previous research on PIST ecology and documenting the distribution and effects of WPBR on PIST. Specifically, our objective for the research presented here was to determine the present distribution of WPBR and other damaging agents in less-investigated areas of Arizona and New Mexico, and how WPBR has impacted trees within infected areas. This research has been published as part of a M.S. thesis at Northern Arizona University (Looney 2012), with portions also published in the peer-reviewed literature (Looney and Waring 2012, 2013).

METHODS

Study Areas

We investigated mixed-conifer stands on the Coconino, Apache-Sitgreaves, Coronado, Gila, and Santa Fe National Forests of Arizona and New Mexico, and the Fort Apache Indian Reservation of Arizona (fig. 12.1). We based sampling on Forest Service, U.S. Department of Agriculture stand examination data, or permanent plot data where stand exam data were unavailable. Sampling intensity was based on the availability of stand data and time constraints. Sampling was more intense in eastern Arizona due to abundant stand data, as well as our goal to better characterize WPBR in this recently discovered infection center. We sampled only stands with PIST basal areas ≥6.9 m^2ha^{-1} and avoided sampling adjacent stands to better characterize the landscape. To generate one random point per stand, we used stratified sampling in Hawth's tools in ArcGIS® 9.3.1 (ESRI 2009) and fTools in QGIS® 1.6.0. (Quantum GIS Development Team 2011). At each point, we installed a single 0.1-ha plot (20 m by 50 m). Each plot included at least five PIST ≥12.7 cm diameter at breast height (1.37 m, d.b.h.) or was randomly relocated. Plots were oriented with the short axis downhill to minimize elevation change. Each plot was subdivided into three 10 m by 10 m and 5 m by 5 m nested subplots (combined area = 0.03 ha and 0.0075 ha, respectively) for measuring saplings (trees <5.0 inches d.b.h. and >1.37 m height), and recording seedlings (trees <1.37 m height), surface cover, and counts of

Ribes plants. We installed 59 plots between 2010 and 2011 (fig. 12.1). Detailed descriptions of plot installation can be found in Looney (2012) and Looney and Waring (2012). We examined additional stands for WPBR infection and other damaging agents in PIST (hereafter 'walk-through surveys') based on the same stand exam data. Across the Southwest, soils are commonly derived from basalt and other volcanic materials, but coarse-grained igneous, sedimentary, and metamorphic parent materials are present (Laing and others 1987, Miller and others 1995). The climate is generally characterized by cold, wet winters and a summer monsoon precipitation pattern (Sheppard and others 2002).

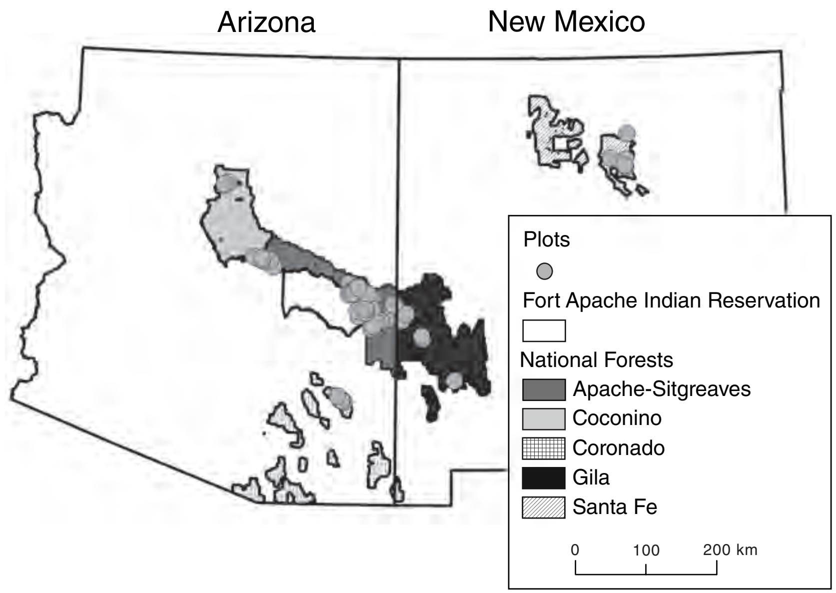

Figure 12.1—Context map of Arizona and New Mexico showing the Pinus strobiformis *plot locations established for this study. Locations were within the National Forest System except for the Fort Apache Indian Reservation in Arizona.* P. strobiformis *found on the Santa Fe National Forest and northern Coconino National Forest are of uncertain taxonomy and may represent intergradation with* P. flexilis *(Conklin and others 2009, Little 1971). Reprinted from Looney and Waring (2012) with permission from Elsevier.*

We identified major abiotic and biotic damaging agents of live overstory PIST. We rated dwarf mistletoe infections using Hawksworth's (1977) dwarf mistletoe rating system and examined all trees of sapling size or larger for signs of both *Atropellis piniphila* canker (a native pathogen with similar signs and symptoms to WPBR) and WPBR. We relied on aecial blisters on PIST as signs of WPBR (Tainter and Baker 1996) from mid-May through mid-July. We also considered trees infected if they bore at least three of the main five WPBR symptoms: flagging, animal chewing, resin flow outside the bark, roughened bark on young trees, and stem or branch swelling (Tomback and others 2005). For all infected overstory trees, we recorded canopy dieback (percent of crown affected by recent death of shoots and branches) using ocular estimates aided by crown profiles drawn on transparent crown grids (Millers and others 1991, Schomaker and others 2007). We divided each tree into three location categories to record canker location: (1) branches, (2) bole, and (3) branches and bole (Arvanitis and others 1984). We did not count cankers due to the large size of many overstory PIST trees and high potential for missing cankers within the upper crown area. We then classified canopy dieback location using the following categories:

0 = no shoot death, 2 = top ¼ of crown, 3 = top ½ of crown, 4 = bottom ½ of crown, 5 = middle crown only, and 6 = entire crown (Innes 1993). We used a rating of leader condition based on Innes (1993): 1 = normal, 2 = shorter than side branches, 3 = dead, 4 = missing, 5 = replaced by side branches, or 6 = exhibiting complete loss of apical dominance. For saplings, we recorded WPBR presence/absence and, if present, WPBR canker location. We did not record WPBR on seedlings. Walk-through stand surveys were limited to presence/absence of WPBR, dwarf mistletoe, and *Atropellis*. We quantified canopy dieback and leader condition for a subsample of 16 trees on uninfected plots for comparison with infected trees. We tested whether canopy dieback was higher in infected trees using a two-sample t-test with unequal variances. We tested whether leader condition was poorer in infected trees using a two-sample Mann-Whitney U test, as those data were non-normal but had comparable variances.

RESULTS

Site characteristics varied (Looney and Waring 2012) and reflected a variety of past management and disturbance histories, including recent mixed-severity fire. Damaging agents of living PIST, including WPBR, were fairly rare and only affected 3.7 percent (S.E. = 0.7) of PIST basal area (fig. 12.2). White pine blister rust was the most common damaging agent, followed by animal damage at 3.2 percent (S.E. = 0.6) and fire at 2.8 percent (S.E. =0.5), though no significant differences were found between these three agents. The majority of animal damage was partial-to-complete girdling from black bear clawing, with minor ungulate antler rubbing on small trees. Fire damage included both old and recent fire scars, as well as crown scorch on several recently burned plots. Abiotic damage agents included lightning damage and sun scald. Logging damage included minor bole and branch damage associated with operations. Bark and twig beetle damage was highly uncommon on living trees.

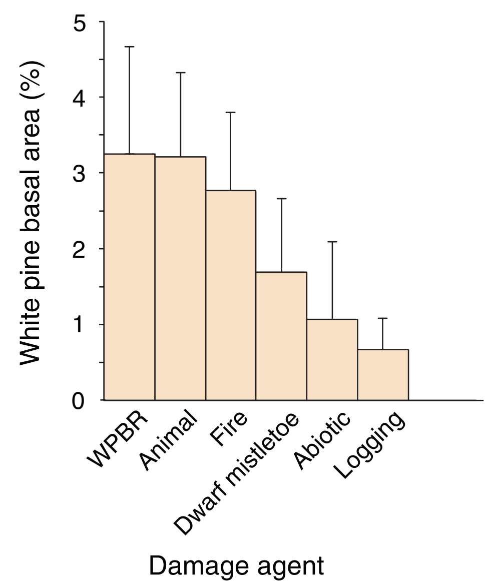

Figure 12.2—Major damage to live southwestern white pines by damaging agent and percent total basal area (mean + 1 standard error). The most common abiotic damages included sun scald and crushing. Reprinted from Looney (2012). WPBR= white pine blister rust.

Dwarf mistletoe (*Arceuthobium apacheum*) occurred on 6.7 percent of plots (fig. 12.2), with an average dwarf mistletoe plot rating of 2.7 (S.E. = 0.6). *Atropellis piniphila* affected four trees on just one plot on the Alpine Ranger District of Apache-Sitgreaves National Forest, AZ. We also found *Atropellis* cankers on saplings adjacent to two plots on the Mogollon Rim district of Coconino National Forest, AZ. Dwarf mistletoe, *Atropellis*, and WPBR did not co-occur on any plot. We found WPBR infection on 18.3 percent of plots sporadically distributed on the Fort Apache Indian Reservation, Apache-Sitgreaves National Forest, and Gila National Forest (fig. 12.3). While affecting 3.3 percent (S.E. = 0.7) of live PIST basal area (fig. 12.2), WPBR infected 4.4 percent of trees ha^{-1} (S.E. = 1.6). Considering only trees within infected plots, WPBR incidence was 22.9 percent of total PIST trees ha^{-1} (S.E. = 6.1).

We performed an additional 23 walk-through surveys for WPBR, dwarf mistletoe, and *Atropellis* (fig. 12.3). Four of these stands were infected with WPBR (17.4 percent), while an additional four were infected with dwarf mistletoe (fig. 12.2). We did not find any additional stands with *Atropellis* infection. All WPBR detections were within the White Mountains region of Apache-Sitgreaves National Forest, AZ. We detected dwarf mistletoe on the White Mountains area of Apache-Sitgreaves National Forest and scattered across Gila National Forest, NM. We made an additional nine incidental detections of WPBR in addition to planned plot measurements or walk-through surveys (fig. 12.3).

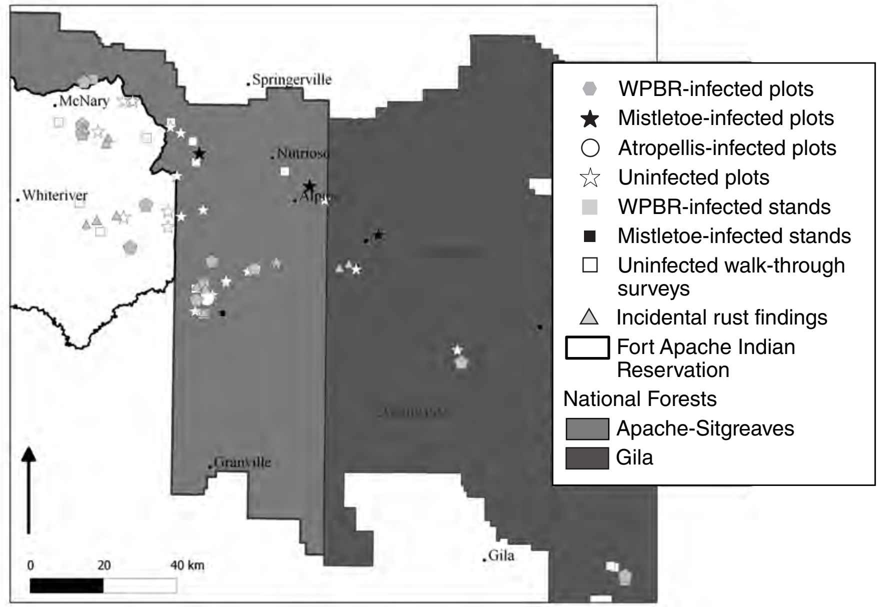

Figure 12.3—Detail of eastern Arizona and western New Mexico showing white pine blister rust (WPBR) detections (other surveyed areas not shown; see fig. 12.1 for all plot locations). Stands are distinguished from plots, as several stands were visited for determining WPBR presence/absence only. In one stand that contained a plot, we found infected trees but these were not located on the plot. Dwarf mistletoe detections are reported for Arceuthobium apacheum *infections on* P. strobiformis *only. Adapted from Looney (2012).*

Within the 11 WPBR-infected plots, we found a total of 31 infected trees (table 12.1). Infection incidence varied by 10-cm diameter class, with most infected trees smaller than 40 cm (table 12.1). Cankers were most common on branches (canker location = 1), with few trees displaying infections on bole (canker location = 2) or both boles and branches (canker location = 3). Canopy dieback was slight, but

Table 12.1—Characteristics [mean (standard error)] of *Pinus strobiformis* within 11 plots with white pine blister rust-infected trees on Apache-Sitgreaves National Forest, Gila National Forest, and the Fort Apache Indian Reservation, Arizona and New Mexico

Diameter class midpoint	Total N PIST	Incidence	Canker location[a]	Canopy dieback[b]	Dieback location[c]	Animal[d]	Leader cond.[e]
cm		%		%		%	
5	15	26.7	0.0 (0.0)	N/A	N/A	N/A	N/A
15	24	41.7	1.0 (0.3)	4.4 (1.7)	3.1 (0.8)	11.1 (0.1)	0.6 (0.6)
25	28	21.4	1.3 (0.3)	4.8 (2.7)	4.2 (0.6)	0.0 (0.0)	1.3 (1.0)
35	24	29.1	1.3 (0.3)	5.8 (0.9)	4.6 (0.4)	14.3 (0.1)	2.3 (0.9)
45	9	11.1	1.0 (0.0)	6.5	6	0	1
55	3	66.7	1.0 (0.0)	5.0 (1.0)	5.0 (0.0)	0.0 (0.0)	0.5 (0.5)
65	6	0	N/A	N/A	N/A	N/A	N/A
75	1	0	N/A	N/A	N/A	N/A	N/A
85	1	100	1	0	4	0	6

N/A= not available.

Note: Southwestern white pines are pooled (Total N PIST), both infected and uninfected, across all 11 plots and by 10-cm diameter class. Incidence refers to white pine blister rust-infected trees as a percentage of PIST by diameter class; animal refers to animal chewing; leader cond. refers to leader condition. Incidence, canker location, canopy dieback, dieback location, animal chewing, and leader condition statistics are calculated for the pooled sample of infected trees only and do not represent inter-plot variability, as not all size classes were present on individual plots. As a result, standard error cannot be calculated for incidence.

[a] (1) branches; (2) bole; (3) branches and bole (Aravanitis and others 1984).

[b] Percent of crown affected by recent death of shoots and branches (Schomaker and others 2007).

[c] Location of canopy dieback within live crown. Excludes bole and isolated, low branches (Innes 1994). Classifications are as follows: 0 = no shoot death; 2 = top ¼ of crown; 3 = top ½ of crown; 4 = bottom ½ of crown; 5 = middle crown only; 6 = entire crown.

[d] Evidence of small rodent chewing on infected tissues.

[e] Rating of leader damage: 1 = normal; 2 = shorter than side branches; 3 = dead; 4 = missing; 5 = replaced by side branches; 6 = complete loss of apical dominance. Adapted from Innes (1993).

Source: Adapted from Looney (2012).

tended to be dispersed throughout the live crown. Infected trees did not show significantly more dieback than uninfected trees ($t = 0.33$, $p = 0.747$). We detected few cases of animal chewing (7.7 percent). Leader condition was generally unaffected by WPBR, with topkill or disfigurement (ratings >2) rare (table 12.1), and WPBR infected trees did not show significantly poorer leader condition than uninfected trees [$w = 346$, $p = 0.6189$ (adjusted for ties)]. There were no apparent relationships between disease incidence, canker location, and canopy dieback with increasing PIST diameter.

DISCUSSION

Serious damaging agents were rare in PIST. Black bear damage, involving the partial or complete girdling of trees, was the most common form of major animal damage and was generally confined to the White Mountains of Arizona. Black bear damage occurs as a result of the bear feeding on sugary resin and can increase at low stand densities (Nolte and others 2003), though we did not investigate the relationship between bear damage and stand density in our data. Fire damage was also common in mature trees on burned plots, but actual mortality was rare, supporting previous evidence that PIST is fire tolerant when mature (Dieterich 1983). Dwarf mistletoe was fairly uncommon but widespread, with detections in several disjunct PIST populations. Overall incidence of both dwarf mistletoe and *Atropellis* canker were low, and we did not detect either disease on WPBR-infected plots. The low incidence of dwarf mistletoe and *Atropellis* canker should help avoid misidentification of WPBR in the study areas given the similar symptomatology of the three diseases (Geils and Hawksworth 2002, Nevill and others 1989). Bark beetle damage was not detected in our study, although small pockets of PIST killed by mountain pine beetle (*Dendroctonus ponderosae*) were observed in the Pinaleños Mountains in Arizona in 2011 (USDA Forest Service 2012). This is in stark contrast to recent large-scale outbreaks of mountain pine beetle in *Pinus flexilis* and *P. albicaulis* in the Rocky Mountains (Gibson and others 2008). The typically low to moderate densities of PIST, combined with its occurrence in diverse mixed-conifer forests, may put it at relatively low risk of bark beetle attack (Gibson and others 2008).

We better described WPBR incidence within eastern Arizona and portions of New Mexico. The WPBR in western New Mexico and eastern Arizona appears locally intense but sporadic, consistent with recent spread into the region (Kearns and Jacobi 2007). The low frequency of infections likely reflects the relatively recent arrival of WPBR between 1988 and the early 2000s (Fairweather and Geils 2011), whereas the earlier studies investigated infections up to 40 years old (Kearns and Jacobi 2007, Smith and Hoffman 2001). Despite the disease's rarity, WBPR infection incidence was comparable to these two studies in terms of trees bearing infection within infected plots. Burns (2006) reported similar between- and within-plot incidence figures in southern Colorado, where the disease had been present since the early

1990s. White pine blister rust severity on individual trees was light, with many trees having a single evident canker or area of WPBR-related dieback. The prevalence of branch cankers and the general lack of bole damage or topkill suggest relatively recent introduction (Smith and Hoffman 2000). We did not find any trees within the plots that had succumbed to WPBR, further suggestive of recent WPBR infection.

Neither canopy dieback nor leader damage in WBPR-infected trees was significantly elevated compared to uninfected trees. Several trees showed signs of infection without disfigurement, and our severity metrics probably have a response lag of several years. When present, WPBR-related canopy dieback was scattered throughout the live crown, a pattern previously reported in the Southwest (Conklin and others 2009). The uncommonness of animal chewing makes WPBR identification by symptoms more difficult but preserves signs of the disease. The lack of leader damage and topkill suggests WPBR will not rapidly affect PIST height growth.

In our relatively small sample size (n = 31), we found an inconsistent relationship between tree size and infection probability with increasing diameter. Probability of infection typically increases with size, likely reflecting greater crown area (Burns and others 2010, Conklin 2004, Kearns and Jacobi 2007). Compared to studies of shorter *P. albicaulis* and *P. flexilis*, our ability to detect infections on taller PIST was limited. Viewing conditions were difficult given tall trees, dense stands, and high contrast during the monsoon season. Also in contrast to previous studies, an inverse relationship between damage severity and tree diameter was not evident in our data (Conklin 2004, Kearns and Jacobi 2007, Smith and Hoffman 2000). Smaller-diameter trees did not show higher bole canker incidence, canopy dieback, or more frequent topkill compared to larger trees. These patterns will likely change in the near future, as disease progression is often faster in smaller trees given shorter distances between infected foliage and boles (Kearns and Jacobi 2007, Tainter and Baker 1996) and shorter time required to girdle smaller stems (Kearns and others 2009).

The early stage of the WPBR invasion in much of Arizona and New Mexico suggests widespread tree mortality will not occur quickly. Furthermore, the rarity or absence of white pine blister rust alternate hosts could limit the spread of the disease to certain populations, such as the Mogollon Rim of central Arizona (Conklin and others 2009, Looney 2012) and isolated mountains of southern Arizona (Conklin and others 2009). Proactive management, such as silvicultural treatments, intended to conserve *P. strobiformis* would help prepare the landscape for WPBR invasion, particularly given the current limited progression of the disease (Burns and others 2008, Schoettle and Sniezko 2007).

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CHAPTER 13.

Forest Attributes and Fuel Loads of Riparian vs. Upland Stands in Mountain Pine Beetle-Infested Watersheds, Southern Rocky Mountains

(Project INT-EM-F-11-01)

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INTRODUCTION

Extensive outbreaks of mountain pine beetle (MPB), spruce beetle (SB), and other insects are altering forest stand structure throughout the Western United States, and thereby increasing the natural heterogeneity of fuel distribution. Riparian forests frequently occur as narrow linear features in the landscape mosaic and can contribute to the spatial complexity of forest stands and fuel loads. These streamside forests are valued for providing aquatic and terrestrial habitat, serving as sources of in-stream and floodplain large wood, and maintaining streamside microclimates and stream water quality. Despite the ecological importance of riparian forests, few data exist on riparian stand attributes and fuel characteristics in watersheds affected by recent beetle outbreaks. This lack of knowledge, combined with administrative regulations for riparian management, present additional challenges to resource specialists planning fuel treatment projects in beetle-infested watersheds. To address this need, we measured stand characteristics and fuel loads in streamside and adjacent upland stands. Within selected watersheds on the Medicine Bow-Routt and Arapaho-Roosevelt National Forests (Colorado and Wyoming), we sampled 30 paired riparian and upland plots during the 2012 field season. Our objectives were to (1) quantify riparian forest characteristics, specifically species composition, structure, and extent of insect-caused mortality; (2) characterize riparian fuel profiles; and (3) compare riparian forest attributes and fuels with those of adjacent uplands. Here, we present preliminary results on basal area, stand structure, and woody surface fuels in riparian and upland stands.

METHODS

Site Selection

Potential study watersheds (those with >50 percent infestation) were selected using aerial detection survey maps of mountain pine beetle infestation compiled by Forest Service, U.S. Department of Agriculture Region 2 Forest Health Monitoring Program. Criteria were conifer-dominated riparian areas along gentle-to-moderate gradient, moderately confined stream segments with floodplains 10 to 15 times wider than the stream channel within the elevation range 2500–3200 m. Potential riparian study locations were identified through examination of topographic maps, digital elevation models, forest vegetation maps for selected watersheds, and suggestions from local land managers.

Plot-Level Data

The approximate center of each 0.05-ha circular riparian plot was established randomly along the selected stream segment; however, riparian plots were located as close to adjacent steams as possible, with the streamside plot perimeter along the bank. Upland plots were located 200–400 m upslope from each riparian plot. Plot location (UTM point), slope, and aspect were recorded at plot center. Within each plot, information recorded on all live and dead trees [≥5 cm diameter at breast height (d.b.h.)] included: species, d.b.h., crown position class (dominant, codominant, intermediate, or suppressed), percent live crown, and crown base height (noncompacted). Evidence of MPB incidence and damage was recorded for each standing lodgepole pine tree (*Pinus contorta* var. *latifolia*, live or dead); similarly, SB incidence was recorded for each standing Engelmann (*Picea engelmanii*) and Colorado blue spruce tree (*Picea pungens*). Trees were categorized as: uninfested (no evidence of beetle), infested with MPB, or infested with SB. We recorded data on the lower canopy strata (<5 cm d.b.h.) to assess ladder fuels (Lutes and others 2006, Ottmar and others 2007) and advance regeneration (Collins and others 2012, Kayes and Tinker 2012). Information recorded on live saplings in two diameter classes (stems ≥2.5 cm and <5 cm d.b.h., and stems <2.5 cm d.b.h.) included species, d.b.h., and estimated height. Within the inner 0.0125 ha of each plot (radius = 6.31 m), live seedlings were tallied by species and height class (<0.5 m or ≥0.5 m). Surface woody fuel loads were estimated using the planar intersect method (Brown 1974) for 1-hour, 10-hour, and 100-hour size classes. Three transects (12.63 m) were established from plot center on randomly selected bearings. Along each transect, 1-hour fuels were tallied for the first 6.3 m, and 10-hour and 100-hour fuels were tallied for the entire transect (Lutes and others 2006, Riccardi and others 2007). Length and two diameters were measured on each coarse woody fuel piece (1000-hour; diameter >7.6 cm) that occurred within the plot perimeter. Depth of litter and duff was measured every meter along each transect (12 depths per transect; 36 depths per plot).

Data Analysis

We calculated fuel loads for 1-hour, 10-hour, and 100-hour size classes as described in Brown (1974). For large woody fuels (1000 hour), length and diameters of each piece were used to calculate individual piece volume, approximating the piece as a cylinder. Total wood volume was summed for each plot in cubic meters. We assumed an average density of 400 kg m^{-3}, calculated kg $m^{-3}$of coarse fuels, and then converted to Mg ha^{-1} for each plot. We compared stand characteristics and fuel loads in 30 pairs of riparian and upland plots in mountain pine beetle-infested stands using one-way analysis of variance (SAS Institute Inc. 2011). We verified normality and homogeneity of variance (Levene’s test) and assigned statistical significance at the $\alpha = 0.05$ level.

RESULTS

Stand Structure

Compositional differences in the riparian and upland stands are reflected in the relative basal area for the three dominant species, lodgepole pine, Engelmann spruce, and subalpine fir (*Abies lasiocarpa*) (fig. 13.1). Upland stands have higher basal area of lodgepole pine, especially dead basal area, which accounts for nearly 32 percent of the upland total basal area (live + dead). Riparian stands have higher basal area of Engelmann spruce, especially live basal area, which accounts for 31 percent of the riparian total basal area. Live and dead basal area of subalpine fir was similar across stand types (fig. 13.1). Other species that occurred in riparian plots were quaking aspen (*Populus tremuloides*), mountain alder (*Alnus incana* ssp. *tenuifolia*), Colorado blue spruce, and willow (*Salix geyeri*). Collectively, these species contributed about 6 percent of the total riparian basal area. Additional species in upland plots were ponderosa pine (*Pinus ponderosa* ssp. *scopulorum*) and Douglas-fir (*Pseudotsuga menziesii*), which together only contributed about 2 percent of the total upland basal area.

Riparian and upland stands did not differ significantly in either total basal area or dead basal area (table 13.1). However, live basal area was significantly higher in riparian stands, accounting for 59 percent of the total basal area. In upland stands, live basal areas composed only 50 percent of the total basal area. Across all plots, nearly 92 percent of the lodgepole pine dead basal area was attributed to MPB, and approximately 60 percent of the Engelmann spruce dead basal area was attributed to SB.

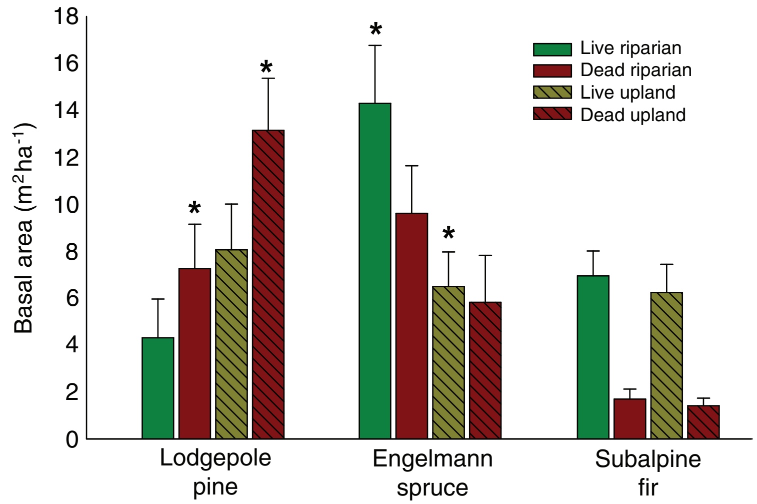

Figure 13.1—Basal area (live and dead, d.b.h. ≥5 cm) for lodgepole pine, Engelmann spruce, and subalpine fir in riparian and upland stands. Data are means and standard errors for 30 riparian and 30 upland plots. Asterisks indicate significant differences between riparian and upland stands.

Table 13.1—Live, dead, and total basal area (d.b.h. ≥5 cm) for riparian and upland stands

Stand type	Live basal area	Dead basal area	Total basal area
		---------- m^2ha^{-1} ----------	
Riparian	26.6 ± 2.7[a] (6.8–64.6)	18.9 ± 2.5 (0.1–48.4)	45.5 ± 3.1 (16.3–79.6)
Upland	21.0 ± 1.9[a] (7.1–50.7)	20.6 ± 2.2 (2.4–50.1)	41.7 ± 3.0 (20.2–87.0)

Note: Data are means, standard error, and range for 30 riparian and 30 upland plots.
[a] Significant difference between riparian and upland stands ($\alpha = 0.05$).

Regeneration

In riparian stands, average density of live saplings was approximately 400 stems ha^{-1} for each diameter size class, slightly lower but not significantly different from densities in upland stands (table 13.2). Subalpine fir dominated the lower canopy strata in all stands, comprising 53 percent of the larger diameter class saplings (stems ≥2.5 cm and <5 cm d.b.h.) in riparian plots and 51 percent in upland plots, and approximately 48 percent of the smaller diameter class saplings (stems <2.5 cm d.b.h.) in both riparian and upland plots. Densities of lodgepole pine saplings were three- to fourfold higher in the upland stands. Densities of Engelmann spruce saplings were not significantly different in riparian and upland stands.

Stem densities of taller seedlings (≥0.5 m and <1.4 m) were greater overall in riparian plots, most notably for Engelmann spruce (table 13.2). Subalpine fir dominated the seedling stratum (<1.4 m) in all plots, comprising approximately 86 percent of the shorter seedling size class (<0.5 m) in both riparian and upland stands, and 56 percent (riparian) and 76 percent (upland) of the taller seedling size class. Densities of lodgepole pine seedlings were twofold higher in the taller size class and eightfold higher in the shorter size class in the upland stands.

Table 13.2—Density (stems ha^{-1}) of advance regeneration, including two diameter classes of saplings (stems ≥2.5 cm and <5 cm d.b.h.; and stems <2.5 cm d.b.h.), and two height classes of seedlings (stem height ≥0.5 m and <1.4 m and stem height <0.5 m)

	Saplings		Seedlings	
Stand type Tree species	D.b.h.: ≥2.5 cm and <5 cm	D.b.h.: stems <2.5 cm	Height: ≥0.5 m and <1.4 m	Height: <0.5 m
	----stems/ha^{-1}----			
Riparian				
Lodgepole pine	39 ± 14[a]	50 ± 25[a]	93 ± 69[a]	24 ± 14[a]
Subalpine fir	214 ± 47	187 ± 39	1513 ± 317	4186 ± 2300
Engelmann spruce	113 ± 27	98 ± 20	793 ± 274[a]	631 ± 210[a]
Colorado blue spruce	3 ± 2	9 ± 6	11 ± 7	8 ± 4
Ponderosa pine	0	0	0	0[a]
Quaking aspen	25 ± 18[a]	27 ± 16[a]	290 ± 190	72 ± 59
Mountain alder	5 ± 2[a]	17 ± 8[a]	0	0
Willow spp.[b]	1 ± 1	10 ± 6	0	0
Total	401 ± 51	397 ± 47	2700 ± 512[a]	4921 ± 2303
Upland				
Lodgepole pine	157 ± 65[a]	140 ± 52[a]	184 ± 91[a]	192 ± 80[a]
Subalpine fir	264 ± 57	204 ± 34	1200 ± 283	3754 ± 1026
Engelmann spruce	91 ± 16	71 ± 15	187 ± 69[a]	369 ± 106[a]
Colorado blue spruce	0	0	0	0
Ponderosa pine	1 ± 1	1 ± 1	0	5 ± 5[a]
Quaking aspen	1 ± 1[a]	2 ± 1[a]	0	0
Mountain alder	1 ± 1[a]	4 ± 4[a]	0	0
Scouler's willow	3 ± 3	2 ± 1	0	0
Total	517 ± 84	423 ± 60	1570 ± 310[a]	4321 ± 1031

D.b.h.= diameter at breast height.
Note: Data are means and standard error for 30 riparian and 30 upland plots.
[a] Significant difference between riparian and upland stands ($\alpha = 0.05$).
[b] Combined stems for Geyer's willow (*Salix geyeri*) and Drummond's willow (*Salix drummondiana*).

Surface Woody Fuels

One-hour fuel loads were higher in riparian stands (table 13.3). However, the mass of 10-hour, 100-hour, and 1000-hour fuel size classes were similar in riparian and upland stands. In addition, the depths of litter and duff layers were comparable in riparian and upland plots (table 13.3).

DISCUSSION

The riparian stands sampled in this study are dominated by the same overstory species as surrounding uplands and are drier than most other riparian plant associations in the region (Carsey and others 2003). Yet, they are managed differently from adjacent uplands to protect and sustain valued functions, notably provision of aquatic and terrestrial habitat. Results from this study provide discussion information to riparian managers and planners.

Comparison of basal area in riparian and upland stands revealed both similarities and differences. Riparian and upland overstories are both dominated by lodgepole pine, Engelmann spruce, and subalpine fir (figs. 13.1 and 13.2). However, stand types differ in percent of standing dead and live trees, and the proportion of basal area accounted for by lodgepole pine and Engelmann spruce (fig. 13.1). Live basal area is currently higher in riparian stands (table 13.1), but may approach upland levels of mortality as SB populations continue to spread. Dead basal area and total basal area (live + dead) did not differ significantly across stand types (table 13.1). Differences in live basal area have been attributed to moister site conditions in riparian areas, as well as successional dynamics among the three dominant species following disturbance (Romme and Knight 1981). Natural disturbance regimes in riparian

Table 13.3—Surface woody fuel loads (Mg ha^{-1}) and litter and duff depths (cm) in riparian and upland stands

Stand type	1 hour	10 hour	100 hour	1,000 hour (sound + rotten)	Litter depth	Duff depth
	---------------Mg ha^{-1}---------------				----cm----	
Riparian	0.73 ± 0.08[a] (0.13–1.97)	1.49 ± 0.14 (0.03–3.46)	2.16 ± 0.30 (0–7.67)	44.56 ± 4.89 (0.15–110.08)	1.7 ± 0.1 (0–30.0)	3.6 ± 0.5 (0–33.0)
Upland	0.49 ± 0.04[a] (0.09–1.23)	1.40 ± 0.10 (0.45–2.57)	2.52 ± 0.25 (0.52–5.37)	46.51 ± 5.77 (2.89–123.30)	1.4 ± 0.1 (0–11.5)	3.2 ± 0.3 (0–24.0)

Note: The 1-, 10-, 100-, and 1,000-hour fuels correspond to woody material 0–0.6, 0.6–2.5, 2.5–7.6, and >7.6 cm in diameter, respectively. Data are means, standard error, and range for 30 riparian and 30 upland plots.
[a] Significant difference between riparian and upland stands ($\alpha = 0.05$).

(A) (B)

(C) (D)

Figure 13.2—A range of stand conditions were sampled in both upland and riparian plots. However, similarities can be seen across stand types in these photos of paired upland and riparian plots. (A, upper left) Bennet Creek, Roosevelt National Forest, Colorado, upland; (B, upper right) Bennet Creek, riparian; (C, lower left) Cortez Creek, Medicine-Bow National Forest, Wyoming, upland; (D, lower right) Cortez Creek, riparian. (Photos by Robert Bazan, USDA Forest Service, Rocky Mountain Research Station)

and upland stands likely differ somewhat in the study area, but have not been studied directly. Generally, fire return intervals are less frequent in valley bottoms (Dwire and Kauffman 2003), and the role of insect infestations on stand development is assumed to be spatially heterogeneous, depending largely on site conditions (Kulakowski and Jarvis 2011). Comparison of relative basal area is also a way to assess standing fuels, which currently appear to be similar across stand types. However, fuel distribution and corresponding fire risk will change as the stands continue to respond to overstory mortality.

Riparian and upland stands contained high densities of understory regeneration in all size classes, similar to results reported in other studies of post-outbreak forest conditions (table 13.2) (figs. 13.2C and 13.2D) (Collins and others 2012, Kayes and Tinker 2012). Higher densities of lodgepole pine in upland plots and Engelmann spruce in riparian plots reflect the influence of site conditions and overstory composition (table 13.2). The dominance of subalpine fir in the lower canopy strata has been observed throughout post-outbreak forests in Colorado and southern Wyoming, and will strongly influence future stand development (table 13.2) (Collins and others 2012, Kayes and Tinker 2012, Pelz and Smith 2012). Although stem densities of taller seedlings are higher in riparian stands, densities of all other size classes of regeneration are comparable across stand types (table 13.2). In general, saplings and other regeneration size classes are considered ladder fuels, i.e., fuels that provide vertical continuity between strata, thereby allowing fire to carry from surface fuels into the crowns of trees (Lutes and others 2006).

Fine surface fuel loads and depths of litter and duff were surprisingly similar in riparian and upland stands (table 13.3). We anticipated that fuel bed depth, primarily litter layer and 1-hour fuels due to high needle and small twig input, may be greater in upland stands due to the condition of the overstory strata, which were largely composed of dead lodgepole pine in the "gray stage" of post-outbreak (Simard and others 2011). Higher 1-hour fuel loads in riparian stands may be due to greater productivity and differences in shrub composition. Preliminary analysis of coarse woody fuels (1000 hour) showed that loads were similar in both stand types (table 13.3), with some differences in extent of decay (data not shown).

In many forested landscapes, riparian areas burn less frequently or less severely than surrounding uplands (Dwire and Kauffman 2003). This has been attributed to the tendency for fire to burn uphill rather than downhill and the reduced probability of lightning strikes in valley bottoms (Romme and Knight 1981). Differences in moisture content of various fuel strata may also be a critical feature in determining how some riparian stands burn relative to uplands. Riparian microclimates, mostly higher humidity and cooler temperatures (Brosofske and others 1997), likely slow the rates of fine fuel drying and decay of coarse wood, thus reducing the probability of fire ignition and spread. In a Wyoming subalpine forest, Romme and Knight (1981) found that late-season moisture of fine woody surface fuels (1–10 cm, which includes 1-hour, 10-hour, and 100-hour fuels) were consistently higher in valley bottoms relative to uplands. In Douglas-fir stands (Blue Mountains, northeastern Oregon), Agee and others (2002) found that understory shrub and herbaceous foliar moisture was considerably higher in riparian areas relative to uplands. Fuel moisture can affect the rate of spread, fuel consumption, and fire-caused mortality for wildland and prescribed fires. More data are needed on relative moisture content of riparian and upland fuelbeds, especially during drought years and late in the fire season.

CONCLUSIONS

1. Sampled stands—both riparian and upland—are dominated by lodgepole pine, Engelmann spruce, and subalpine fir, but differ in relative proportions of lodgepole pine (higher in uplands) and Engelmann spruce (higher in riparian). Although live basal area is higher in riparian stands, neither dead basal area nor total basal area (live + dead) differed significantly across stand types. For current purposes of planning fuel management projects, standing fuels (trees) are similar across stand types. However, fuel distribution and corresponding fire risk will change as the stands respond to overstory mortality.

2. All size classes of understory regeneration were abundant in both riparian and upland plots. Lower canopy strata were dominated by subalpine fir, which accounts for nearly half of the understory stem density across stand types. Lodgepole pine regeneration was higher in upland stands and Engelmann spruce regeneration was slightly higher in riparian stands. Considering only stem density, ladder fuels (regeneration strata) are relatively similar in riparian and upland stands.

3. With the exception of 1-hour fuels, fine fuel loads and depths of litter and duff are similar in riparian and upland stands. Coarse fuel loads (1000 hour) are also similar across stand types.

4. A more complete comparative analysis of riparian and upland stand structure and fuel loads is underway. Examination of the contribution of live shrub biomass and herbaceous biomass to riparian and upland fuel loads is in progress. However, notable differences between riparian and upland stands are not expected. Canopy, size class distribution of live and dead trees, and understory data will be examined in more detail to determine the role of MPB and SB in the development of future forests and fire risk for riparian and upland stands.

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CHAPTER 14.

Alder *(Alnus incana tenuifolia)* Mortality Agent Complex Effects on Riparian Zone Habitat (Project WC-EM-B-10-01)

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INTRODUCTION

The riparian forests of interior and south-central Alaska are arguably the most productive and important ecosystems within the portion of the State covered by the boreal forest. Recent large-scale mortality of thin-leaf alder (*Alnus icana tenuifolia*), one of the dominant species in these areas, has created the potential for deep-seated changes in these ecosystems. Alder is a symbiotic nitrogen-fixer, allowing it to thrive in low-nutrient soil. The long-term productivity of Alaskan riparian forests is directly related to the amount of nitrogen fixed and deposited in the soil during the alder-dominated stages of succession (Ruess and others 2009). In addition, because the streams and rivers of south-central Alaska are a critical resource for salmon reproduction, the Cook Inlet and Prince William Sound fisheries are, in part, dependent on the breeding habitat found in these waters (Roon and others 2012, Wipfli and Musslewhite 2004).

Three defoliating sawfly species feed on thin-leaf alder in riparian areas throughout south-central and interior Alaska. The circumpolar-striped alder sawfly *(Hemichroa crocea)* and two nonnative European species, woolly alder sawfly *(Eriocampa ovata)* and European green alder sawfly *(Monsoma pulveratum)*, are the major sources of alder defoliation in Alaska. The green alder sawfly is the newest detection of a nonnative sawfly in Alaska, representing a new U.S. record (Kruse and others 2010, Smith and Goulet 2000). Significant defoliation by both exotic sawflies has been recorded in south-central Alaska on the Palmer Hay Flats, Eagle River, Little Susitna River, and on the Kenai Peninsula (Cooper Landing, Quartz Creek, and Kenai River).

Discernible defoliation, branch dieback, and mortality of thin-leaf alder in Alaska was documented as early as 2003. By 2005, the green alder sawfly and the canker fungus *Valsa melanodiscus* were both implicated as possible causal agents or contributing factors (Adams and others 2010). While the green alder sawfly is an exotic insect new to Alaska, it has quickly become established on the Kenai Peninsula, the Anchorage bowl, and the Matanuska-Susitna Valley. It has since been found throughout the Pacific Northwest (Kruse and others 2010). In contrast, the fungus that causes alder canker is presumably a native, usually benign fungus for which conditions have changed to its advantage. To a lesser extent, two other alder species are also affected by alder canker, Siberian alder (*A. fruticosa*) and Sitka alder (*A. sinuata*).

Previous roadside surveys have detected widespread canker disease at over 100 locations across south-central and interior Alaska, with mortality reaching over 80 percent at some sites. The primary causal agent of canker on *Alnus tenuifolia* has previously been confirmed as *Valsa melanodiscus* (Stanosz and others 2011, Worrall and others 2010). *V. melanodiscus* also causes similar cankers on *A. fruticosa*, which may be more vulnerable when water stressed (Rohrs-Richey and others 2011a, 2011b). However, differences in canker morphologies and fruiting bodies suggest that other fungal species may also be involved in dieback and mortality (Walker and others 2012). Two *Phytophthora* species have been suggested as possible contributors to the widespread alder mortality in Alaska (Adams and others 2008, Adams and others 2010, Aguayo and others 2013).

Little is known about how sawflies, canker, or other pathogens interact in regards to alder productivity and survival. This project served to investigate alder dieback in riparian areas previously observed via Alaska's aerial detection survey. We attempted to (1) identify the extent to which nonnative sawflies contribute directly to alder dieback, (2) identify the extent to which alder canker contributes directly to alder dieback, (3) identify the extent to which nonnative sawflies and canker may synergize to cause alder dieback, and (4) identify whether nonnative sawflies may serve as infection facilitators or otherwise predispose alder to pathogens.

METHODS

In addition to roadside and opportunistic surveys, a network of monitoring plots was established to estimate the occurrence and severity of canker and sawfly activity along streams in three geographic areas. Nine plots (three in interior, three in south-central, and three on the Kenai Peninsula) were selected in early- and mid-succession alder stands. Plots were chosen in areas with known evidence of sawfly and/or canker. Plots were 18 m square, and divided into three equal transects (6 m by 18 m).

In early spring, five flight traps were placed at each of these nine plot locations prior to bud break for the host plant. Traps were hung at a height of 1 m, and one was placed at each corner of the plot, as well as at plot center. Traps were collected and replaced every 2 weeks throughout the summer during 2 consecutive years. Beat sampling for larva was conducted during early July in both years to provide a quantifiable estimate of larval numbers during peak density. Alder damage levels from sawfly defoliation and canker were each evaluated using ocular estimates to place observations into percentage classes. The naturally occurring gradient of sawfly population densities was used to test for relationships between sawfly feeding and canker infection. Larvae were collected adjacent to the study sites, and a host suitability feeding trial was conducted using leaves from three species of alder and from willow (*Salix* spp.). In addition, we investigated the observations of Pieronek (1980) regarding *M. pulveratum's* unique ability amongst sawflies to overwinter in woody materials.

At each plot, all alder ramets >1 inch in diameter were individually labeled, diameter measured, and assessed for presence, absence and progression of stem cankers. These measurements included all *Alnus* species found within the plots, including *A. tenuifolia*, *A. fruticosa*, and *A. sinuata*. The causal agents of canker-induced dieback and mortality on the three alder species were determined by pathogenicity tests to fulfill Koch's postulates; fungi were isolated from canker margins, identified via DNA sequencing, and inoculated onto alder stems at two sites. Fourteen months after inoculation, the resultant cankers were measured and fungi re-isolated from the margins. Each site was inspected for typical symptoms of *Phytophthora* diseases. *Phythophthora* spp. were baited and trapped at each of these locations from roots and soil using thin-leaf alder twigs.

During both summers of the study, signatures of alders with active canker that could be reliably identified from the air were defined, and surveys for canker damage in Alaska were conducted as part of annual Aerial Detection Surveys for forest insects and diseases. Field verifications of the presence of both sawflies and canker within areas identified by aerial surveys were conducted whenever possible.

RESULTS

The range of green alder sawfly was found to extend from the city of Juneau in southeast Alaska to the city of Fairbanks, approximately 700 miles to the north. The infestation appears to be centralized around the Kenai Peninsula and Parks Highway, with no adults caught on flight traps deployed in more remote portions of the State. The green alder sawfly infestation and defoliation were highest in pure thin-leaf alder stands in south-central Alaska and the Kenai Peninsula. Evidence of sawfly activity was much lower in interior Alaska (fig. 14.1). Siberian and

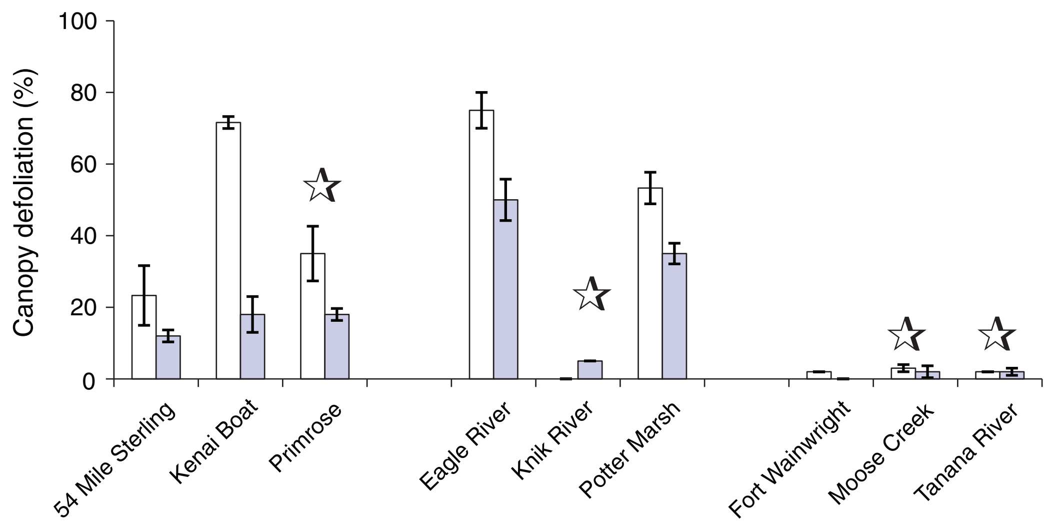

Figure 14.1—Total observed canopy defoliation based on ocular estimates taken during 2010 (white bars) and 2011 (gray bars). Stars indicate sites with strong components of Alnus fruticosa *or* A. sinuata *in addition to* A. incana tenuifolia. *Error bars are +/- 1 standard error.*

Sitka alder do not appear to be suitable hosts for larval feeding, with 100-percent mortality occurring in captive larva when supplied with any food source other than thin-leaf alder. High adult catches in flight traps were correlated with high larva counts and defoliation levels during both study years.

Alder canker, by contrast, is widespread throughout Alaska, and was present to some degree in virtually every thin-leaf alder stand visited in this study. It was also found to infect all three species of alder found in south-central and interior Alaska. Of the nine sites in this study, three had >30 percent mortality (fig. 14.2). At two sites near Anchorage, nearly 100 percent of the mortality was due to canker. The three sites with the least mortality were not in the pure stands of thin-leaf alder, but had a significant component of either Sitka or Siberian alder. Fifty-eight different fungal species were isolated from canker margins, and the 13 most common were used for artificial field inoculations at two sites. Analysis of variance showed highly significant differences in mean canker size among the fungal pathogens at each plot (fig. 14.3). *Valsa melanodiscus* and *Melanconis alni* both showed high levels of virulence on thin-leaf and Siberian alders. The most virulent of the 13 fungi tested on Sitka alder was *Melanconis stilbostoma*, which was not highly virulent on the other alder species. In 2010

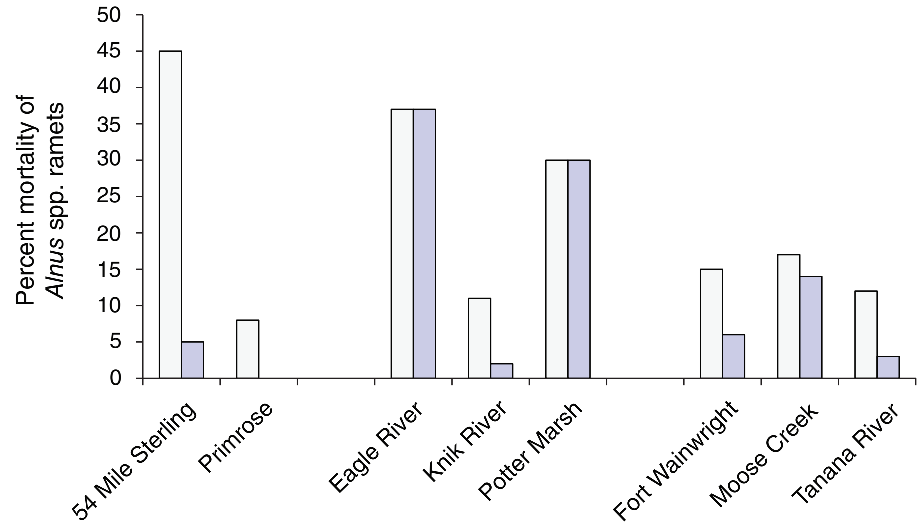

Figure 14.2—Percent of total Alnus *ramets at each site dead (white bars) and percent of total* Alnus *ramets dead and known to be killed by alder canker (gray bars).*

and 2011, over 700 *Phytophthora* isolates were obtained from alder stands in Alaska (Hansen and others 2010). However, no symptoms of *Phytophthora* diseases were observed (Adams and others 2010).

Alaska's Aerial Detection Survey mapped alder dieback for the first time in 2010, when 44,230 acres were recorded. While most of the affected acreage was mapped near streams, many were found up to 2 miles from riparian areas and up to 1,500 feet elevation. In 2011, 142,005 acres of alder dieback were recorded (fig. 14.4).

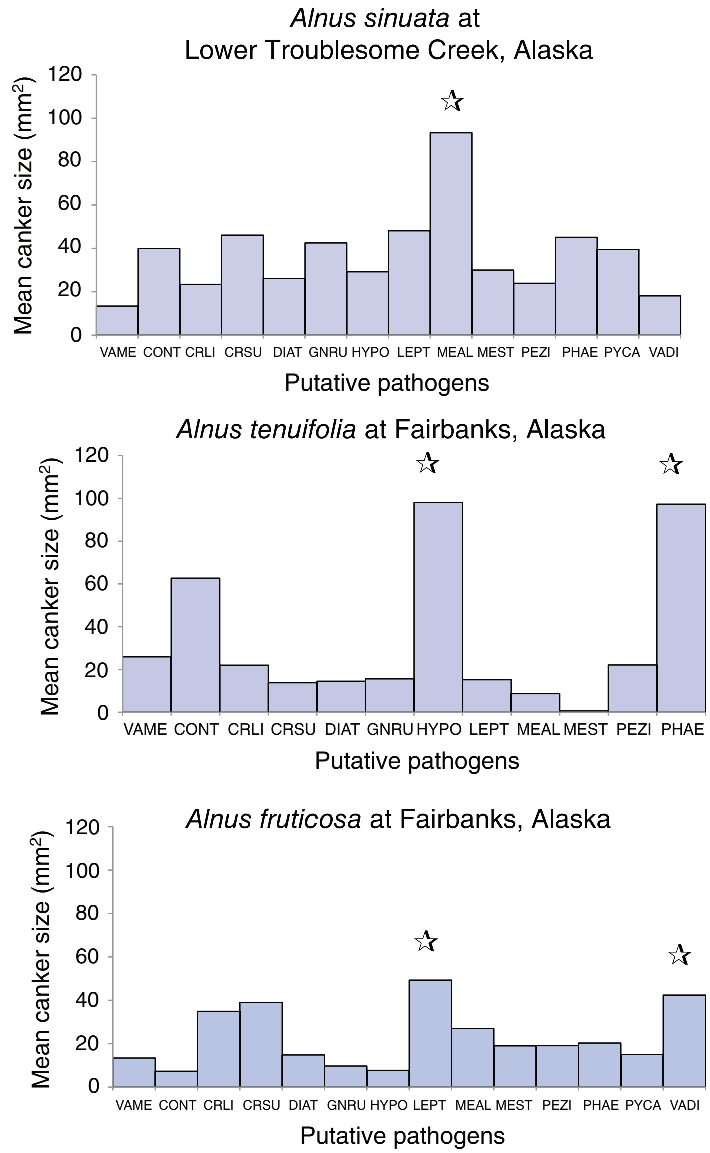

Figure 14.3—Disease response on three alder species to inoculations with the fungal pathogens. VAME= Valsa melanodiscus, *CONT=Control (no pathogen), CRLI=*Cryptosphaera ligniae, *CRSU=*Cryptosporella suffusa, *DIAT=*Diatrype spilocea, *GRNU=*Gnomonia rubi-ideaei *(=*Valsalnicola oxystoma*), HYPO=*Hypoxylon fuscum, *LEPT=*Leptographium piriforme, *MEAL=*Melanconis alni, *MEST=*Melanconis stilbostoma, *PEZI=*Pezicula *sp., PHAE=*Phaeomollisia/Phialocephala fortinii, *PYCA=*Pyrenochaeta cava, *VADI=*Valsa diatrypoides. *Stars indicate species that differ significantly from the controls.*

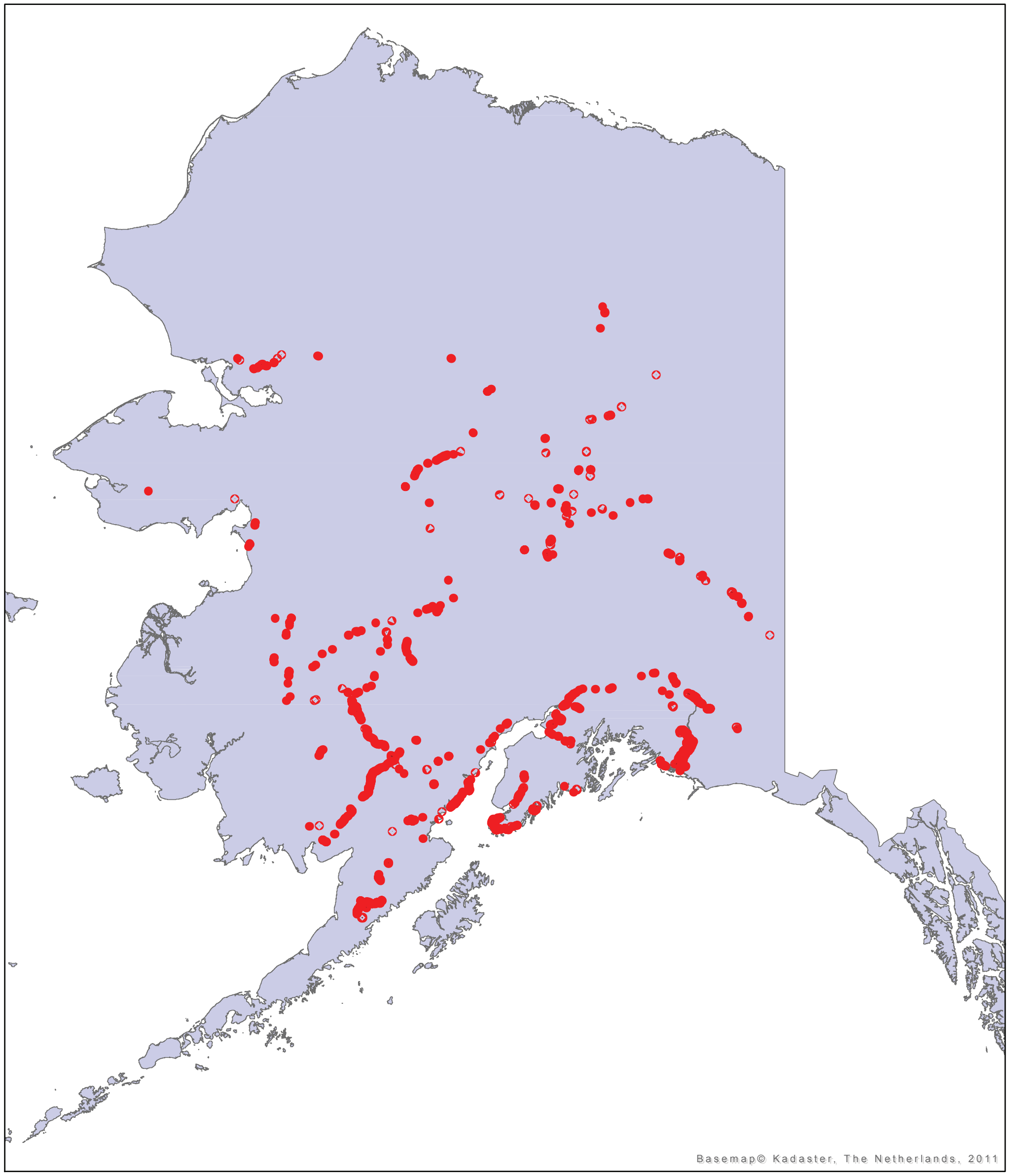

Figure 14.4—Distribution of alder dieback mapped during 2011 Aerial Detection Survey.

Although there appeared to be an ecologically significant correlation between alder canker mortality and sawfly abundance, there was insufficient evidence to support a statistically significant relationship, and the causal mechanisms remain largely unknown (fig. 14.5).

DISCUSSION

Green alder sawfly caused significant defoliation to thin-leaf alder stands in south-central Alaska, including the Kenai Peninsula. Although few alder ramets were killed as a direct result of the sawfly feeding, defoliation at these high levels is known to affect symbiotic nitrogen fixation associated with alder (Ruess and others 2009). Defoliation also reduces the tree's ability to respond to other sources of stress. In comparison, alder canker directly caused significant mortality in thin-leaf alder stands, with limited signs of recovery or new recruitment. Permanent removal or reduction of thin-leaf alder from riparian ecosystems on a landscape scale would adversely affect long-term nutrient cycling and forest productivity, aesthetic value, allochthanous inputs to rivers and streams, reduce shading of streams that would increase temperatures, and reduce prey abundance and quality in salmonid breeding areas.

The extent to which alder sawflies and alder canker may synergize is not yet known. Other defoliators are known to increase susceptibility of their host plants to other insects and diseases, but the presence or absence of alder sawflies or other defoliators did not enhance or impede infection rates or death rates of alder due to alder canker. Alder sawfly and alder canker occurrence are clearly correlated; however, it

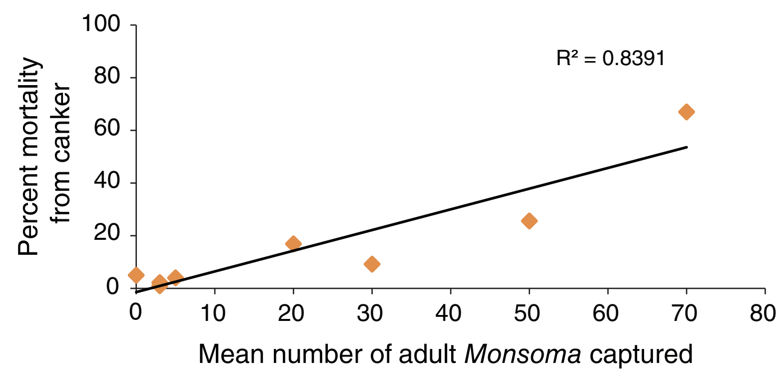

Figure 14.5—The apparent relationship between the number of Alnus *ramets at each site killed by alder canker and the number of* Monsoma pulveratum *sawflies caught by flight traps.*

may be that alder sawfly is somehow attracted to alder canker infestations, or the introduction of alder sawfly is coincident with hot spots of alder canker occurrence. That said, there are still plenty of areas in the State where alder canker exists in high abundance without sawflies, but no known areas where sawfly occurs in high numbers in the absence of alder canker.

Ethanol release by *Phytophthora ramorum* cankers on coast live oak *(Quercus agrifolia)* has been implicated in the attraction of bark and ambrosia beetles (*Scolytinae*) (Kelsey and others 2013). In a pilot study, canker-infested alder ramets contained ethanol concentrations comparable to the amounts recorded in *P. ramorum* cankers on coast live oak.[1] Relatively high tissue concentrations and release rates of ethanol by healthy ramets located near canker-infested ramets on the same genet both suggest a possible sympathetic response, but higher ethanol concentrations within canker-infested stands also potentially increase the attraction of sawflies. This work will be continued in collaboration with the Pacific Northwest Research Station.

CONCLUSION

Canker may occur where sawflies do not, but sawflies do not occur in high numbers in the absence of significant canker infestation.

[1] Personal communication. 2012. Rick Kelsey, Research Forester, U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station, 3200 Southwest Jefferson Way, Corvallis, OR 97331.

Future work will continue to explore possible relationships. Prior to this project, little was known about many basic aspects of *Monsoma pulveratum* natural history in North America, or its relationship to alder canker. This study provided important information regarding the range, extent, and host suitability of *M. pulveratum*, including successful overwintering populations at all nine study locations. This includes all three interior Alaska locations studied, where only two individual specimens had previously been recorded and no established populations were previously known.

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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 13th edition in a series of annual reports, survey data are used to identify geographic patterns of 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. A satellite-derived change detection system operating across the conterminous United States is described. A conceptual organization of existing and future technologies to support and improve forest health monitoring is presented. FIA data are used to produce a national map of invasive plant species infestation and to evaluate changes in crown conditions during the last decade. Five recently completed Evaluation Monitoring projects are summarized, addressing forest health concerns at smaller scales.

Keywords: Change detection, crown conditions, disturbance, drought, fire, forest health, forest insects and disease, invasive plants, tree mortality.

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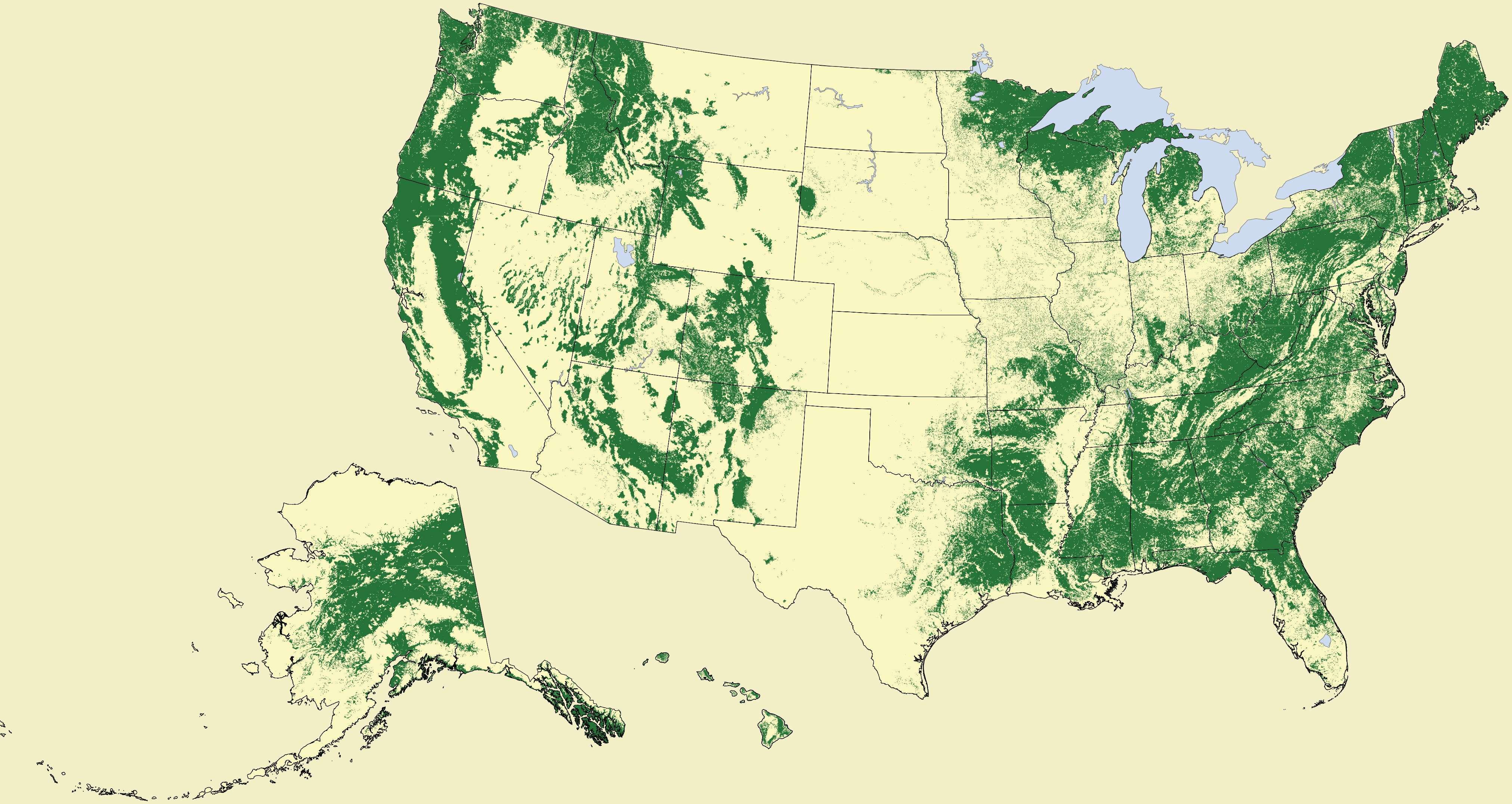

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