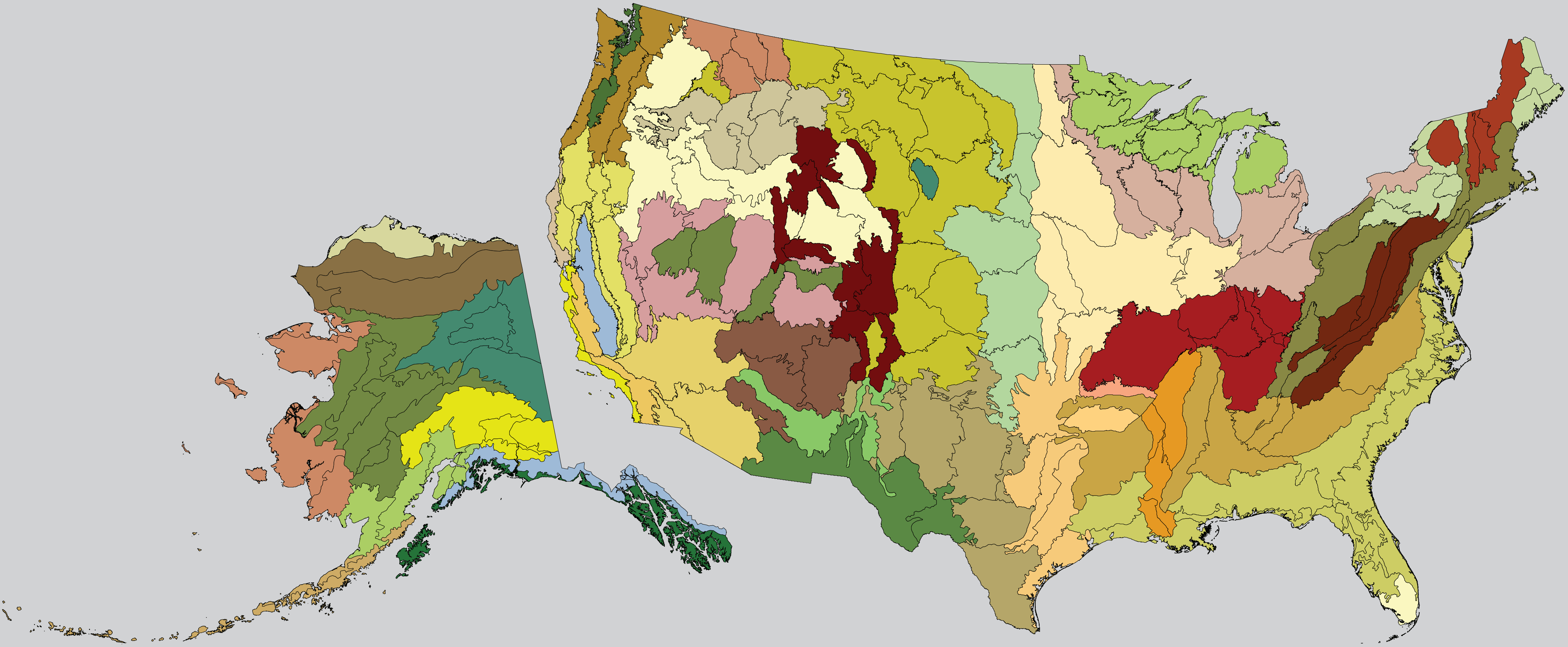
Forest Health Monitoring: National Status, Trends, and Analysis 2011

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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Kevin M. Potter, Research Assistant Professor, North Carolina State University, Department of Forestry and Environmental Resources, Raleigh, NC 27695

Barbara L. Conkling, Research Assistant Professor, North Carolina State University, Department of Forestry and Environmental Resources, Raleigh, NC 27695

ABSTRACT

The annual national report of the Forest Health Monitoring Program of the Forest Service, U.S. Department of Agriculture, presents forest health status and trends from a national or multi-State regional perspective using a variety of sources, introduces new techniques for analyzing forest health data, and summarizes results of recently completed Evaluation Monitoring projects funded through the national Forest Health Monitoring Program. Survey data are used to identify geographic patterns of insect and disease activity. Satellite data are employed to detect geographic clusters of forest fire occurrence. Data collected by the Forest Inventory and Analysis Program of the Forest Service are employed to detect regional differences in tree mortality. Fragmentation status of forest types in the Eastern United States is evaluated and the area of intact forest is estimated by forest type. The presence and abundance of introduced plant species in the Northeastern United States are examined to determine what broad-scale factors might predict their distribution. Results from 16 years of ozone damage biomonitoring are presented, demonstrating overall declines in damage over time. Three recently completed Evaluation Monitoring projects are summarized, addressing forest health concerns at smaller scales.

Keywords—Drought, fire, forest health, forest insects and disease, fragmentation, nonnative 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). The Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture, with cooperating researchers within and outside the Forest Service and with State partners, quantifies the health of U.S. forests (chapter 1). The analyses and results outlined in sections 1 and 2 of this FHM annual national report offer a snapshot of the current condition of U.S. forests from a national or multi-State regional perspective, incorporating baseline investigations of forest ecosystem health, examinations of change over time in forest health metrics, and assessments of developing threats to forest stability and sustainability. For data sets collected on an annual basis, analyses are presented from 2010 data. For data sets collected over several years, analyses are presented at a longer temporal scale. Chapters describe new techniques for analyzing forest health data as well as new applications of established techniques. Finally, section 3 of this report presents summaries of results from recently completed Evaluation Monitoring (EM) projects that have been funded through the national FHM program to determine the extent, severity and/or causes of specific forest health problems (FHM 2011).

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). Low-altitude survey data, collected, in 2010, by the Forest Health Protection Program of the Forest Service, identified 67 different mortality-causing agents on nearly 3.68 million ha of forest in the conterminous United States, and 70 defoliating agents on approximately 3.72 million ha. Significant large geographic hot spots of forest mortality were associated with mountain pine beetle in the West, while a smaller hot spot was associated with *Ips* engraver beetles in the South. Hot spots of defoliation were spread throughout the conterminous United States, the largest and most intense associated with pinyon needle scale, western spruce budworm, forest tent caterpillar, and gypsy moth. Spruce beetle was the most important cause of mortality in Alaska, while willow leaf blotchminer and aspen leafminer were the most important defoliating agents.

Forest fire occurrence outside the historic range of frequency and intensity can result in extensive economic and ecological impacts. The detection of regional patterns of fire occurrence can allow for the identification of areas at greatest risk of significant impact and for the selection of locations for more intensive analysis (chapter 3). In 2010, the South Central and Red Bed Plains of Oklahoma experienced the most fires per 100 km^2 of forested area, while ecoregions in Utah, Idaho, and Florida also had high densities of forest fire occurrence. In Alaska, a moderate density of forest fires occurred in the Yukon Flats ecoregion. Geographical hot spots of fire occurrence were detected in eastern Oregon, the Southeastern Coastal Plain, northeastern Oklahoma, and in scattered locations across the West. When looking at the last 10 years of fire occurrence data, ecoregions in southern California and central Idaho had the highest mean number of fires per year relative to forested area.

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 2010 to map drought conditions across the conterminous United States at a fine scale (chapter 4). Most of the Western United States had more moisture than average in 2010, although there were scattered pockets of moderate to extreme drought; this was a departure from a decade-long trend. In contrast, there were fairly extensive areas of drought in the Eastern United States, including along the central Gulf of Mexico coast and in the western Great Lakes region. A separate analysis mapped, for the 100-year period from 1911 to 2010, the frequency of 2, 3, 4, and 5 consecutive years of moderate to extreme drought conditions during the late spring-early summer season.

Mortality is a natural process in all forested ecosystems, but high levels of mortality at large scales may indicate that the health of forests is declining. Phase 2 data collected by the Forest Inventory and Analysis (FIA) Program of the Forest Service offer tree mortality information on a relatively spatially intense basis of approximately 1 plot per 6,000 acres (chapter 5). An analysis of FIA plots from 36 States found that the highest ratios of annual mortality to gross growth occurred in ecoregion sections of the Plains States. Mortality was also high in parts of Florida and eastern Texas. In all ecoregions, the ratio of average dead tree diameter to average surviving live tree diameter indicated that the trees that died were similar in size to the trees that survived. In three ecoregions with the highest mortality relative to growth, the predominant vegetation is grassland, where few forest plots are measured and where tree growth rates are slow.

Fragmentation is a continuing threat to the sustainablity of forests in the Eastern United States. Currently, however, little detail is available about the degree to which forest types are fragmented. Such information could improve land management and policy by identifying forest types of special concern for conservation or remediation. Landcover data were combined with FIA field plot information to evaluate the fragmentation status of forest types in the Eastern United States and to estimate the area of intact forest by forest type (chapter 6). The percentage area in the intact forest area density class varied from 13 percent to 78 percent among individual forest types. Fragmentation would be considered a natural attribute of many of the forest types that exhibited low percentages of intact forest. For the forest types that are not naturally fragmented and that occur in relatively accessible locations, typically less than half of the forest type area qualified as intact forest.

The spread of introduced species into natural plant communities can threaten native plant diversity and ecosystem functions. Surveys have been conducted to assess the extent of the most harmful known species, but many introduced species do not become invasive until many years after their initial introduction. The presence and abundance of introduced species across the forests of the northeastern portion of the United States were examined to determine what broad-scale factors can be used to predict their distribution (chapter 7). The distributions of introduced species were examined over the entire Northern Research Station region, by level of forest intactness, using ecoregion provinces as subpopulations. The results indicate a strong association between forest fragmentation and the regional distribution of introduced species. Occupancy of introduced species varied across provinces; those with a higher proportion of forest-edge plots had the highest occupancy by introduced species.

Ozone is a highly toxic air contaminant that has been shown repeatedly to damage tree growth and cause significant disturbance to forest ecosystems. It also causes distinct foliar injury symptoms to certain plant species that can be used to detect and monitor ozone stress in the forest environment (chapter 8). Biomonitoring surveys, begun in 1994 in the East and 1998 in the West, provide important regional information on ozone air quality, and a field-based measure of ozone injury and probable impact unavailable from any other data source. Results from the North indicate that injury indices have fluctuated annually in response to seasonal ozone concentrations and site moisture conditions. There is an overall declining trend in percent injured plots and injury severity, especially after 2002. Results from the Pacific Northwest also suggest a declining trend in foliar injury severity, while results from the South show a steady decline in percent injured plots.

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

- Replicated a landscape scale experiment to restore oak savanna ecosystems in central Iowa, with an objective of collecting sensitive process-level ecosystem indicators of restored and degraded savannas for long-term monitoring (chapter 9);
- Developed a distribution and incidence database and established the extent of the host range for *Xylella fasidiosa*, the xylem-inhabiting bacterium that causes bacterial leaf scorch in shade trees (chapter 10);
- Tested whether the fuel estimations derived from FIA phase 3 plots capture multiple and distinct fuel complexes in the Southern Appalachian Mountains (chapter 11).

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.

Assessing and monitoring the health of these forests are critical and challenging tasks. This is reflected within the Criteria and Indicators for the Conservation and Sustainable Management of Temperate and Boreal Forests (Montréal Process Working Group 1995), which the Forest Service, U.S. Department of Agriculture, uses as a forest sustainability assessment framework (USDA Forest Service 2004, 2011a). While there is no universally accepted definition of forest health, the current understanding of ecosystem dynamics suggests that healthy ecosystems are those that are able to maintain their organization and autonomy over time while remaining resilient to stress (Costanza 1992), and that evaluations of forest health should emphasize factors that affect the inherent processes and resilience of forests (Kolb and others 1994, Raffa and others 2009). This report, the 11th in an annual series produced by the Forest Health Monitoring (FHM) Program of the Forest Service, attempts to quantify the status of, changes to, and trends in a wide variety of such indicators of forest health. These indicators encompass forest insect and disease activity, wildland fire occurrence, drought, tree mortality, forest fragmentation, introduced plant species, lichen diversity, and ozone injury.

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

The second objective of the report is to present new techniques for analyzing forest health data as well as new applications of established techniques, presented in selected chapters of section 2. Examples in this report are chapter 6, which demonstrates an approach to improve national assessments of forest fragmentation by incorporating information about the specific forest types that are fragmented; and chapter 7, which uses FIA phase 3 data to examine factors important in determining the regional distribution of invasive plants in the upper Midwest and in the Northeastern United States.

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 2011), generally at a finer scale than that addressed by the analyses in sections 1 and 2. Each chapter in section 3 contains an overview of an EM project, key results, and contacts for more information.

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

Alaska Ecoregion Provinces

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

Conterminous States Ecoregion Provinces

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

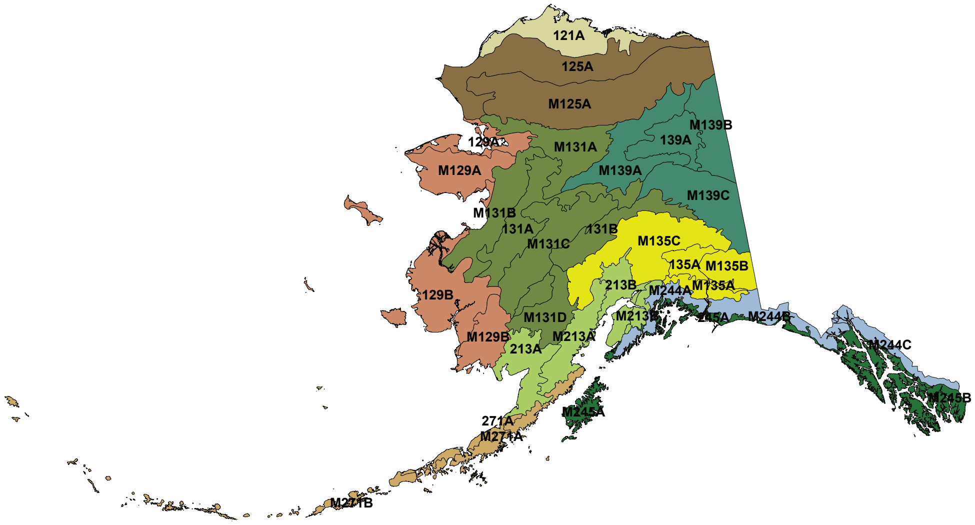
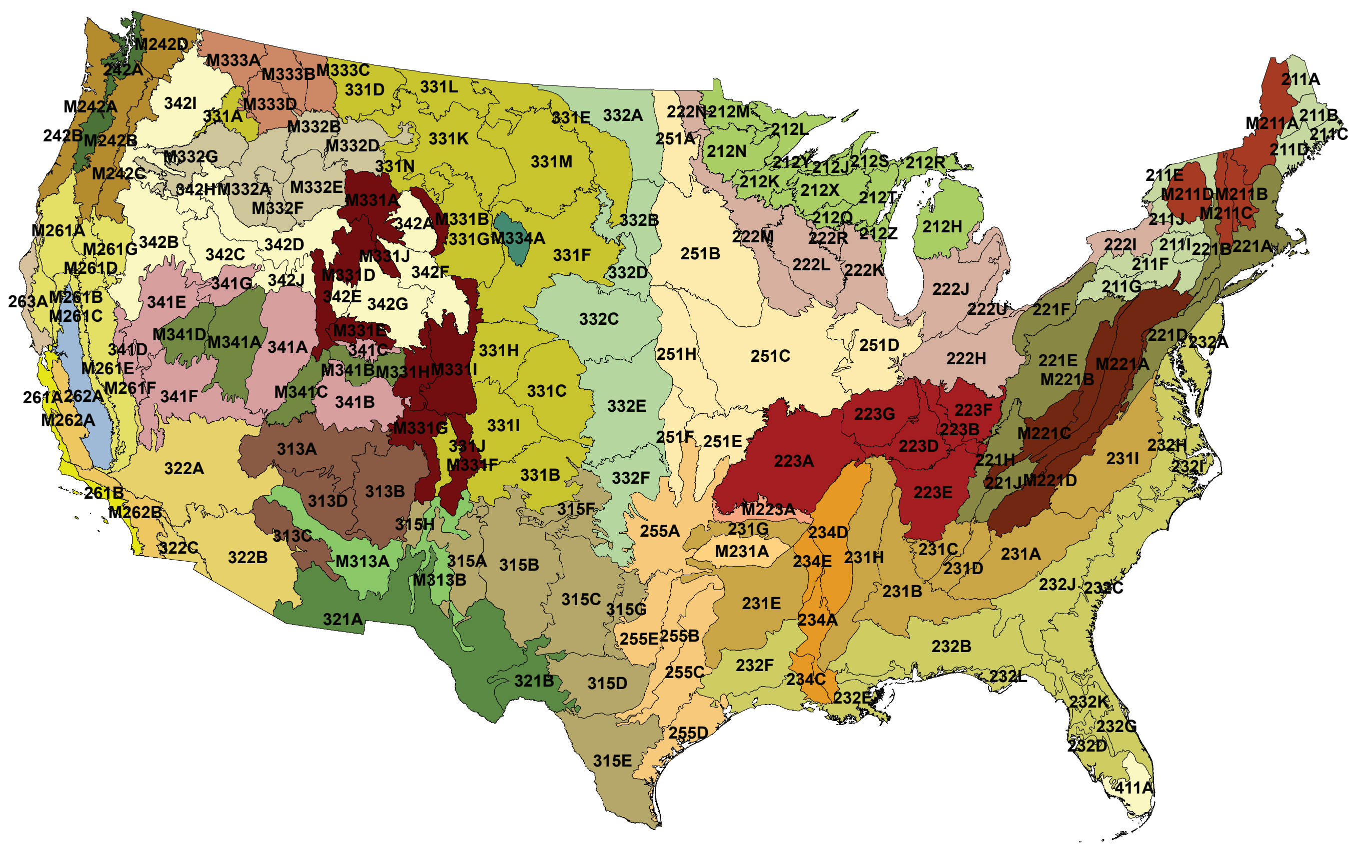


Figure 1.1—Ecoregion provinces and sections for the conterminous United States (Cleland and others 2007) and Alaska (Nowacki and Brock 1995). Ecoregion sections within each ecoregion province are shown in the same color.



DATA SOURCES

Forest Service data sources included in this report are FIA annualized phase 2 and phase 3 survey data and ozone bioindicator plant data (Bechtold and Patterson 2005), FHP insect and disease detection survey forest mortality and defoliation data for 2010, Moderate Resolution Imaging Spectroradiometer (MODIS) Active Fire Detections for the United States database for 2010 (USDA Forest Service 2011b), 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 (PRISM Group 2010) and the 2001 National Land Cover Database (NLCD) map (Homer and others 2007).

A major source of data for FHM analyses has been the FIA program, which collects forest inventory information across all forest land ownerships in the United States. FIA maintains a network of more than 125,000 permanent forested ground plots across the conterminous United States and southeastern Alaska, with a sampling intensity of approximately one plot per 2 428 ha. FIA phase 2 encompasses the annualized inventory measured on plots at regular intervals, with each plot surveyed every 5 to 7 years in most Eastern States, but with plots in the Rocky Mountain and Pacific regions surveyed once every 10 years (Reams and others 2005). The standard 0.067-ha plot (fig. 1.2) consists of four 7.315-m radius subplots (approximately 168.6 m^2 or 1/24 acre), on which field crews measure trees at

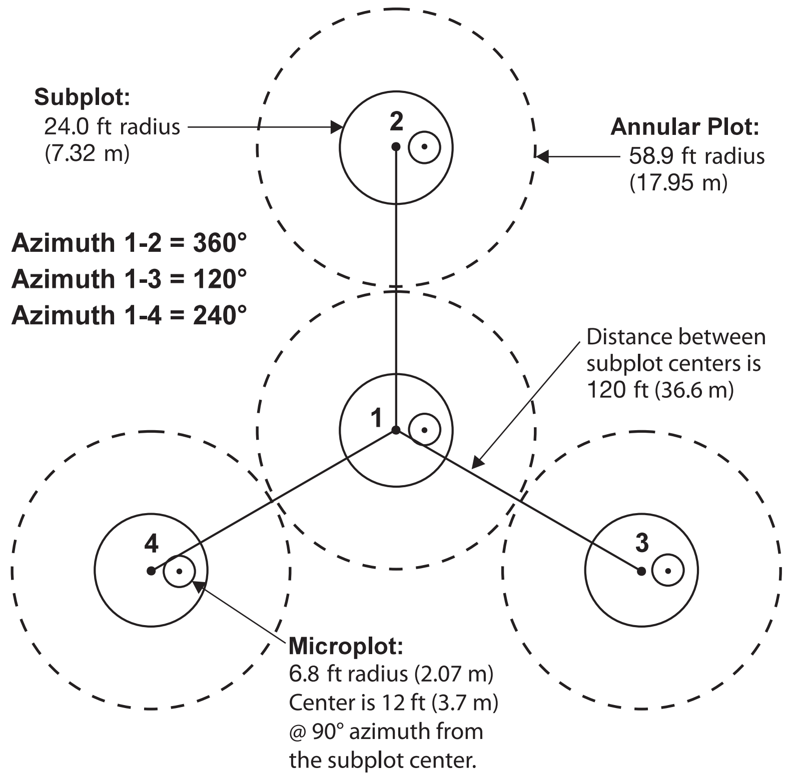


Figure 1.2—The Forest Inventory and Analysis mapped plot design. Subplot 1 is the center of the cluster with subplots 2, 3, and 4 located 120 feet away at azimuths of 360°, 120°, and 240°, respectively (Woudenberg and others 2010).

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 originates at the center of each subplot (Woudenberg and others 2010).

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

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 2011). 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. 1.3):

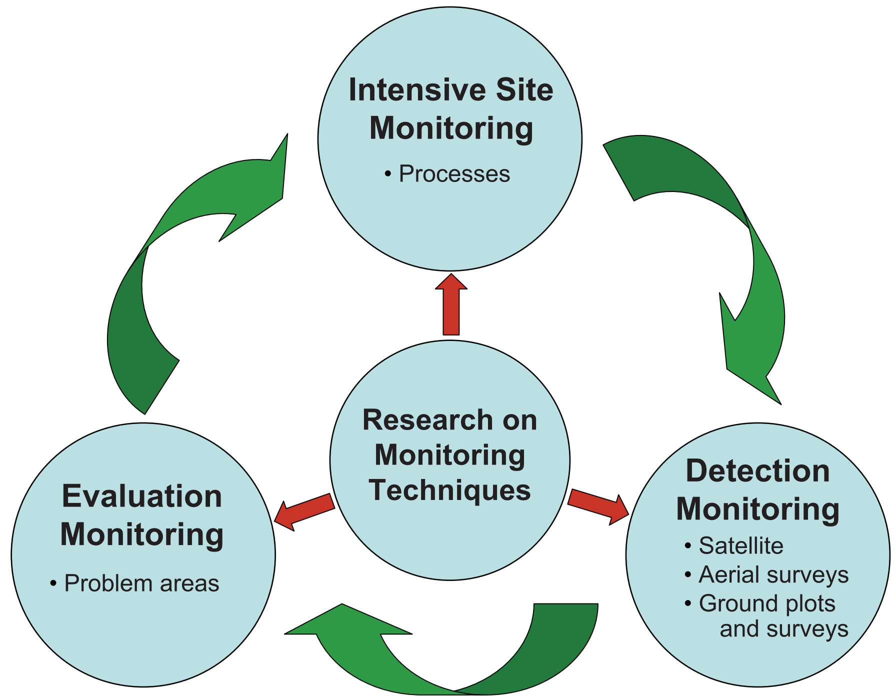


Figure 1.3—The design of the Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture (FHM 2003). A fifth component, Analysis and Reporting of Results, draws from the four FHM components shown here and provides information to help support land management policies and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

[1] USDA Forest Service. 1998. Forest health monitoring 1998 field methods guide. Research Triangle Park, NC: USDA Forest Service, National Forest Health Monitoring Program. 473 p. On file with: Forest Health Monitoring Program, 3041 Cornwallis Rd., Research Triangle Park, NC 27709.

- Detection Monitoring—nationally standardized aerial and ground surveys to evaluate status and change in condition of forest ecosystems (sections 1 and 2 of this report);
- Evaluation Monitoring—projects to determine extent, severity, and causes of undesirable changes in forest health identified through Detection Monitoring (section 3 of this report);
- Intensive Site Monitoring—projects to enhance understanding of cause-effect relationships by linking Detection Monitoring to ecosystem process studies and to assess specific issues, such as calcium depletion and carbon sequestration, at multiple spatial scales (section 3 of this report);
- Research on Monitoring Techniques—work to develop or improve indicators, monitoring systems, and analytical techniques, such as urban and riparian forest health monitoring, early detection of invasive species, multivariate analyses of forest health indicators, and spatial scan statistics (section 2 of this report);
- Analysis and Reporting—synthesis of information from various data sources within and external to the Forest Service to produce issue-driven reports on status and change in forest health at national, regional, and State levels (sections 1, 2, and 3 of this report).

The FHM program, in addition to national reporting, generates regional and State reports. These reports may be produced with FHM partners, both within the Forest Service and in State forestry and agricultural departments. Some examples include reports on disturbance and forest conditions (Steinman 2004), urban monitoring methods (Lake and others 2006), health conditions in national forests (Morin and others 2006), urban forest health monitoring (Cumming and others 2006, 2007), crown conditions (Randolph 2010, Randolph and Moser 2009), and ozone monitoring (Rose and Coulston 2009). The Forest Health Highlights report series, available on the FHM Web site at www.fs.fed.us/foresthealth/fhm, is produced by the FHM regions in cooperation with their respective State partners.

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), crown-condition classification (Schomaker and others 2007), sampling and estimation procedures for vegetation diversity and structure (Schulz and others 2009), and the overall forest health indicator program (Woodall and others 2010). For more information, visit the FHM Web site at www.fs.fed.us/foresthealth/fhm.

This FHM national report is produced by national forest health monitoring researchers at the Eastern Forest Environmental Threat Assessment Center, which was established under the Healthy Forest Restoration Act to generate knowledge and tools needed to anticipate and respond to environmental threats. For more information about the research team and about threats to U.S. forests, please visit www.forestthreats.org/about.

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SECTION 1.
Analyses of Short-Term Forest Health Data

CHAPTER 2.

Large-Scale Patterns of Insect and Disease Activity in the Conterminous United States and Alaska from the National Insect and Disease Detection Survey Database, 2010

Kevin M. Potter

Jeanine L. Paschke

INTRODUCTION

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

METHODS

Nationally compiled low-altitude aerial survey and ground survey data collected by the Forest Health Protection (FHP) Program of the Forest Service, U.S. Department of Agriculture, can be used to identify forest landscape-scale patterns associated with hot spots of forest insect and disease activity in the conterminous United States, and to summarize insect and disease activity by ecoregion section in Alaska (Potter and Koch 2012, Potter 2012, Potter 2013). Surveys covered approximately 155.6 million ha (61 percent) of the forested area in the conterminous United States in 2010, and 9.1 million ha (17.7 percent) of Alaska's forested area (fig. 2.1).

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

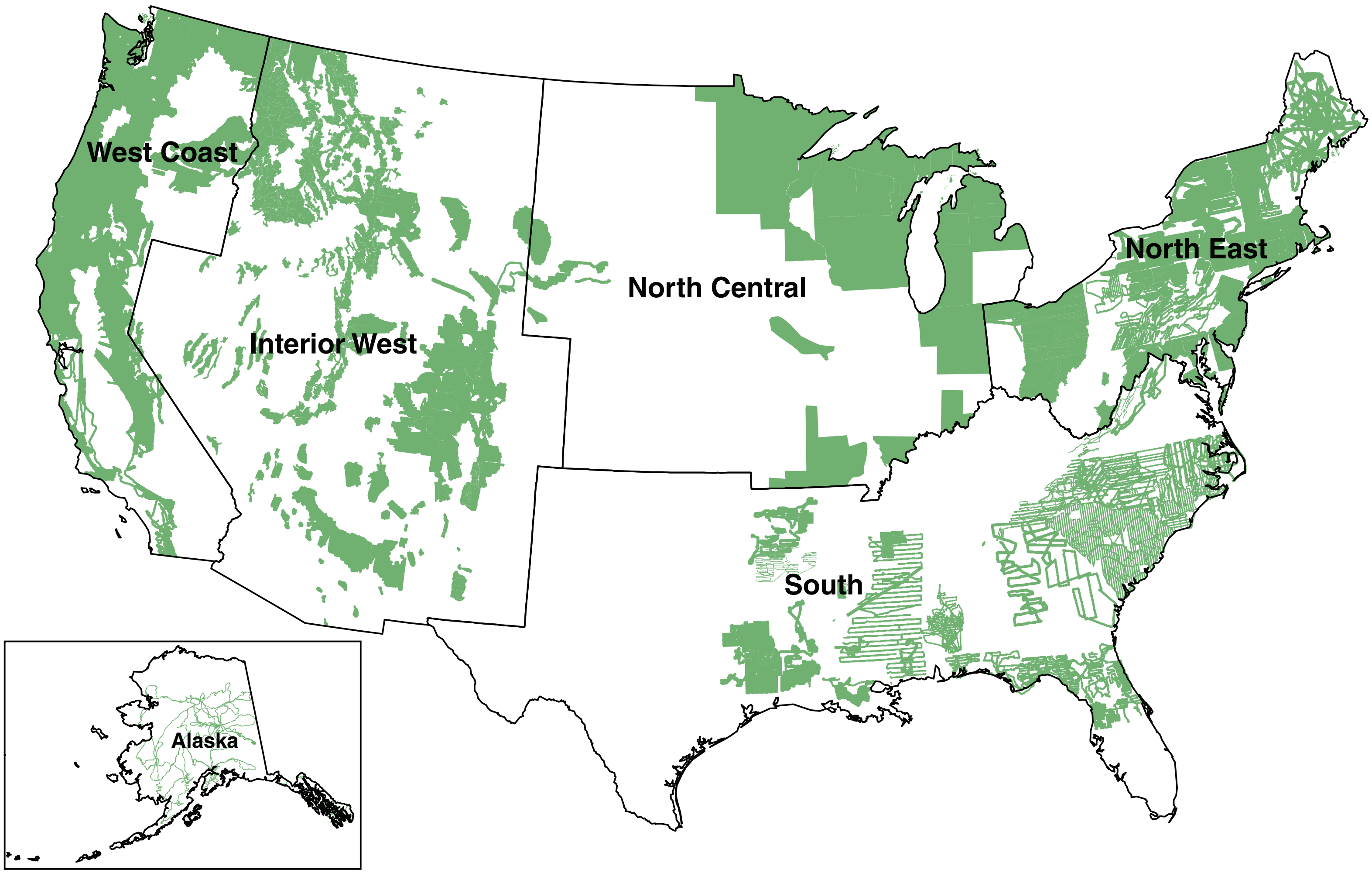

Figure 2.1—The extent of surveys for insect and disease activity conducted in the conterminous United States and Alaska in 2010. The black lines delineate Forest Health Monitoring regions. (Data source: USDA Forest Service, Forest Health Protection Program.)

The mortality and defoliation polygons were used to identify the mortality and defoliation agents and complexes in the conterminous United States found on more than 5 000 ha of forest, and to identify and list the five most widely detected defoliation and mortality agents and complexes for Alaska. As a result of the insect and disease sketchmapping process, all quantities are "footprint" areas for the agent or complex, outlining the areas 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.

A Getis-Ord hot spot analysis (Getis and Ord 1992) in ArcMap 9.2 (ESRI 2006) was employed to identify surveyed forest areas with the greatest exposure to the detected mortality-causing and defoliation-causing agents and complexes. Hexagon coordinates for North America, taken from the Environmental Monitoring and Assessment Program (White and others 1992), were intensified to develop a lattice of hexagonal cells, of approximately 2 500 km^2 extent, for the conterminous United States. This cell size allows for analysis at a medium-scale resolution of approximately the same area as a typical county. The percent of surveyed forest area in each hexagon exposed to either mortality-causing or defoliation-causing agents was then calculated by masking the surveyed area and mortality and defoliation 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 percent of forest exposed to the identified mortality or defoliation agents and complexes was calculated by dividing the 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 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). Non-stationarities are processes whose statistical properties vary over time or space.

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

The threshold values are not exact, because the correlation of spatial data violates the assumption of independence required for statistical significance (Laffan 2006). The Getis-Ord approach does not require that the input data be normally distributed because the local G_i* values are computed under a randomization assumption, with G_i* equating to a standardized *z* score that asymptotically tends to a normal distribution (Anselin 1992). The *z* scores are reliable, even with skewed data, as long as the distance band 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 data from Alaska in 2010 (fig. 2.1) precluded the use of hot spot analyses for the State. Instead, mortality and defoliation data were summarized by ecoregion section (Nowacki and Brock 1995), calculated as the percent of the forest within the surveyed areas affected by agents of mortality or defoliation. For reference purposes, ecoregion sections (Cleland and others 2007) were also displayed on the geographic hot spot maps of the conterminous United States.

RESULTS AND DISCUSSION

The FHP data identified 67 different biotic mortality-causing agents and complexes on approximately 3.68 million ha of forest across the conterminous United States in 2010, an area slightly smaller than the land area of New Hampshire and Connecticut combined. Forests cover approximately 252.7 million ha of the conterminous United States (Smith and others 2009).

Mountain pine beetle (*Dendroctonus ponderosae*) was the most widespread mortality agent, detected on 2.77 million ha (table 2.1). Other mortality agents detected across very large areas, each affecting more than 100 000 ha, were fir engraver (*Scolytus ventralis*), five-needle pine decline, subalpine fir (*Abies lasiocarpa*) mortality, and spruce beetle (*Dendroctonus rufipennis*). Mortality from western bark beetles, when considered as a group (table 2.2), was detected on a total of more than 3.48 million ha in 2010, a vast majority of the total area on which mortality was recorded.

Table 2.1—Mortality agents and complexes affecting more than 5 000 ha in the conterminous United States in 2010

Agents/complexes causing mortality, 2010	Area
	ha
Mountain pine beetle	2 770 492.4
Fir engraver	286 653.5
Five-needle pine decline	229 561.8
Subalpine fir mortality	173 944.4
Spruce beetle	134 062.8
Western pine beetle	93 737.5
Douglas-fir beetle	70 526.8
Gypsy moth	23 163.2
Emerald ash borer	14 711.7
Balsam woolly adelgid	9 411.3
Eastern larch beetle	7 749.5
Forest tent caterpillar	6 883.8
Flathead borer	6 589.9
Jeffrey pine beetle	5 868.3
Southern pine beetle	5 778.5
White pine blister rust	5 708.9
Total, all agents	3 675 135

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

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

Western bark beetle taxa	
Douglas-fir beetle	*Dendroctonus pseudotsugae*
Fir engraver	*Scolytus ventralis*
Flatheaded borer	*Buprestidae*
Ips engraver beetles	*Ips* spp.
Jeffrey pine beetle	*Dendroctonus jeffreyi*
Mountain pine beetle	*Dendroctonus ponderosae*
Northern spruce engraver beetle	*Ips perturbatus*
Roundheaded pine beetle	*Dendroctonus adjunctus*
Silver fir beetle	*Pseudohylesinus sericeus*
Spruce beetle	*Dendroctonus rufipennis*
Tip beetles	*Pityogenes* spp.
Western balsam bark beetle	*Dryocoetes confusus*
Western cedar bark beetle	*Phloeosinus punctatus*
Western pine beetle	*Dendroctonus brevicomis*
Bark beetles	Non-specific

Additionally, the survey identified 70 biotic defoliation agents and complexes affecting approximately 3.72 million ha of forest across the conterminous United States in 2010, an area slightly smaller than the land area of Maryland and Connecticut combined. The most widespread defoliators were western and eastern spruce budworms (*Choristoneura occidentalis* and *C. fumiferana*), affecting 1.08 million ha (table 2.3). Tent caterpillars (*Malacosoma* spp.), pinyon needle scale (*Matsucoccus acalyptus*), gypsy moth (*Lymantria dispar*), aspen (*Populus tremuloides*) decline, and nonspecific defoliators each affected more than 100 000 ha.

Table 2.3—Defoliation agents and complexes affecting more than 5 000 ha in the conterminous United States in 2010

Agents/complexes causing defoliation, 2010	Area
	ha
Spruce budworm (eastern and western)	1 080 861.0
Tent caterpillars	733 803.3
Pinyon needle scale	521 565.3
Gypsy moth	488 579.1
Aspen decline	152 280.4
Defoliators (non-specific)	112 485.9
Larch needle cast	47 036.0
Baldcypress leafroller	35 779.2
Winter moth	31 061.2
Needlecast	14 442.5
Linden looper	11 705.7
Pinyon sawfly	11 025.7
Aspen blotchminer	10 674.8
Pine butterfly	9 716.6
Larch casebearer	7 273.6
Douglas-fir tussock moth	6 664.0
Leaftier	6 539.7
Aspen leafminer	6 344.4
Jack pine budworm	5 468.5
Beech bark disease	5 422.5
Birch leaf fungus	5 288.2
Total, all agents	3 715 292

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

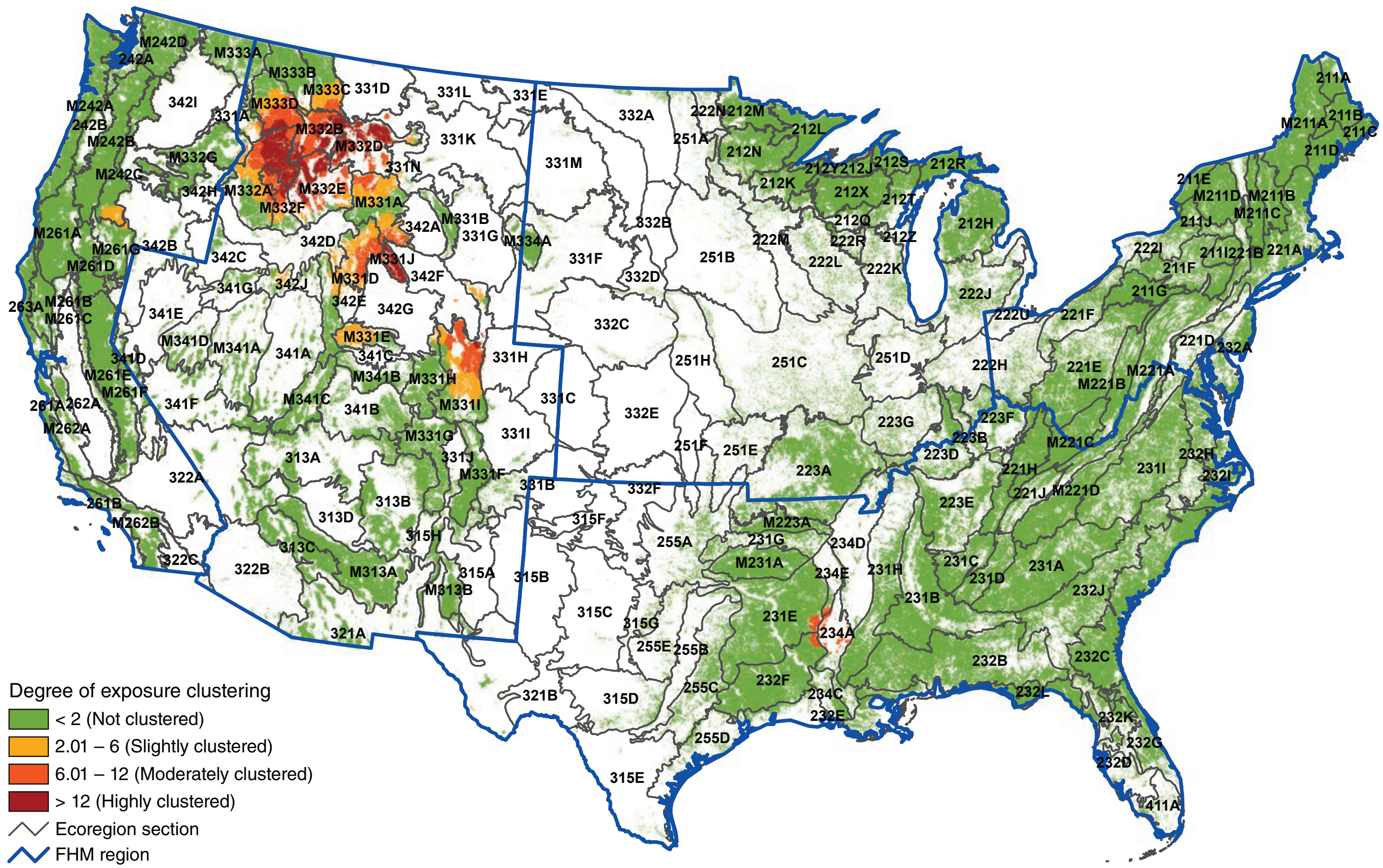

Figure 2.2—Hot spots of exposure to mortality-causing insects and diseases in 2010. Values are Getis-Ord G_i** scores, with values greater than 2 representing significant clustering of high percentages of forest area exposed to mortality agents. (No areas of significant clustering of low percentages of exposure, -2, were detected). The gray lines delineate ecoregion sections (Cleland and others 2007); the blue lines delineate Forest Health Monitoring regions. Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: USDA Forest Service, Forest Health Protection Program.)*

The hot spot analysis detected three major hot spots of insect and disease mortality in the FHM Interior West region (fig. 2.2), the region in which mountain pine beetle was by far the predominant mortality agent. A large and highly intense hot spot occurred in Idaho and Montana, the result of extensive mountain pine beetle mortality and centered on ecoregion sections M332A-Idaho Batholith, M332B-Northern Rockies and Bitterroot Valley, M332E-Beaverhead Mountains, and M332D-Belt Mountains. A second highly intense, but smaller, hot spot was centered on ecoregion section M331J-Wind River Mountains and extending into neighboring ecoregion sections M331D-Overthrust Mountains and M331A-Yellowstone Highlands (all in Wyoming). In addition to mountain pine beetle, five-needle pine decline and subalpine fir mortality were important mortality agents in this hot spot. A third intense, but smaller, mortality hot spot was caused by mountain pine beetle, subalpine fir mortality, and spruce beetle activity in ecoregion section M331I-Northern Parks and Ranges in northern Colorado and southern Wyoming. A less intense hot spot associated with mountain pine beetle occurred in ecoregion section M331E-Uinta Mountains in northeastern Utah, while another, associated with mountain pine beetle, subalpine fir mortality, and Douglas-fir beetle, was detected in ecoregion section 342J-Eastern Basin and Range in southern Idaho and northwestern Utah.

Mountain pine beetle also was an important cause of mortality in the West Coast and North Central regions. The single, relatively low-intensity mortality hot spot in the West Coast region, in ecoregion section M242C-Eastern Cascades in south-central Oregon (fig. 2.2), was associated with mountain pine beetle and, to a lesser degree, with western pine beetle (*Dendroctonus brevicomis*).

No mortality hot spots occurred in the North Central region, where mountain pine beetle mortality occurred in the Black Hills of South Dakota, or in the North East FHM region. The South, meanwhile, contained a single hot spot, in ecoregion section 234A-Southern Mississippi Alluvial Plain in northeastern Louisiana (fig. 2.2), where an outbreak of *Ips* engraver beetles occurred. This ecoregion section is part of a large area affected by acute drought in 2010 (see chapter 4). Extensive *Ips*-caused pine mortality across much of Louisiana was largely in response to these drought conditions, with particularly large areas of damage in Franklin and Evangeline parishes (Louisiana Department of Agriculture and Forestry 2011). Due to the scattered nature of *Ips* occurrence, detection and reporting of *Ips* damage is inconsistent and incomplete; there are likely more areas of unreported damage (Louisiana Department of Agriculture and Forestry 2011).

As with mortality, the Interior West FHM region encompassed several defoliation hot spots. One intense and extensive hot spot in the region was associated with pinyon needle scale defoliation in three Nevada ecoregion sections: M341D-West Great Basin and Mountains, 341F-Southeastern Great Basin, and M341A-East Great Basin and Mountains. A second,

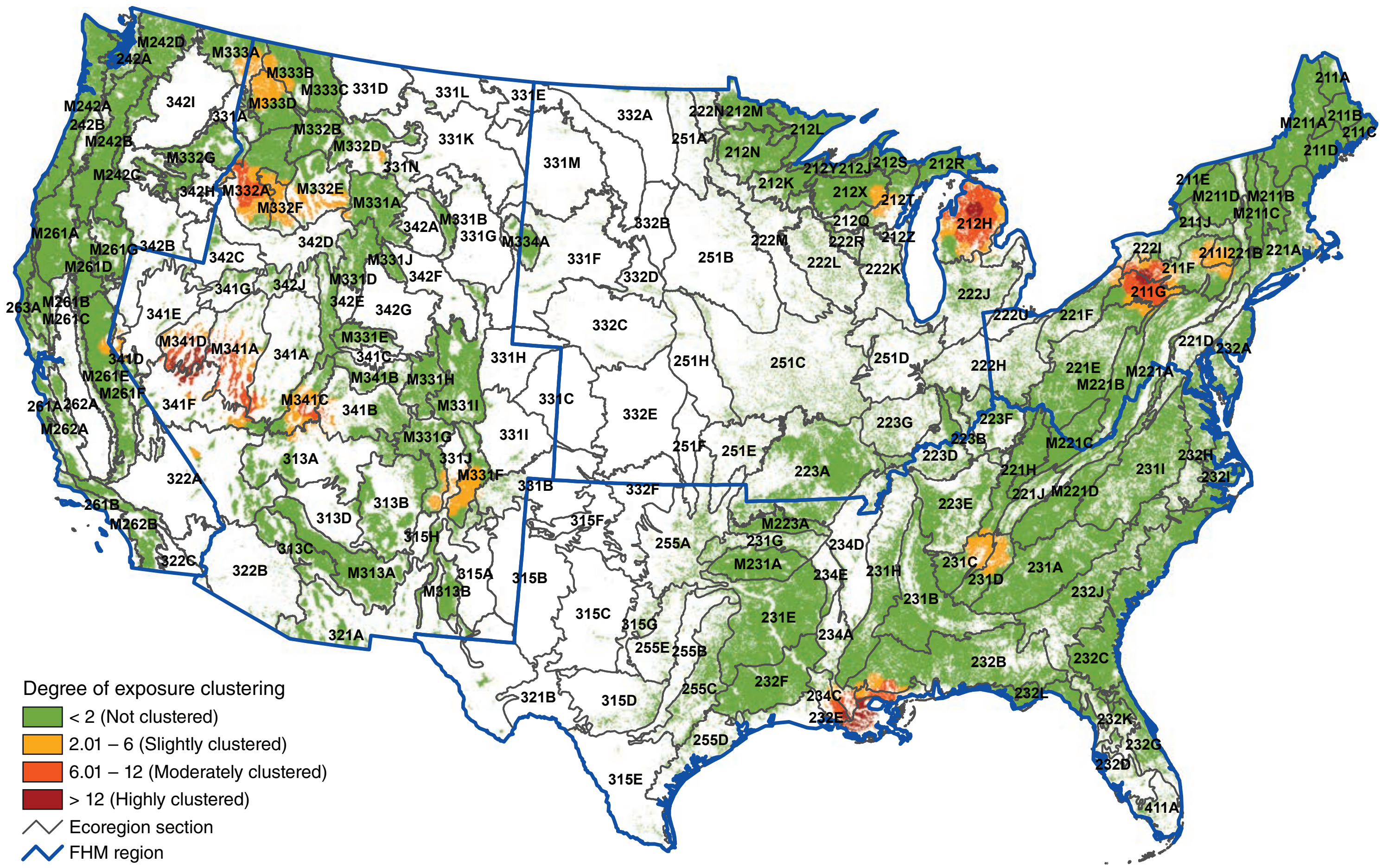

Figure 2.3—Hot spots of exposure to defoliation-causing insects and diseases in 2010. Values are Getis-Ord G_i** scores, with values greater than 2 representing significant clustering of high percentages of forest area exposed to defoliation agents. (No areas of significant clustering of low percentages of exposure, -2, were detected). The gray lines delineate ecoregion sections (Cleland and others 2007); the blue lines delineate Forest Health Monitoring regions. Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: USDA Forest Service, Forest Health Protection Program.)*

less-intense hot spot was caused by pinyon needle scale, in ecoregion section 341D-Mono at the western edge of Nevada (fig. 2.3).

Four other defoliation hot spots in the region were associated with western spruce budworm. A moderately intense hot spot occurred in Idaho, centered on ecoregion section M332A-Idaho Batholith and extending into ecoregion sections M332F-Challis Volcanics, 342H-Blue Mountain Foothills, and M332E-Beaverhead Mountains. Another moderately intense defoliation hot spot caused by western spruce budworm was located in ecoregion section M341C-Utah High Plateau in south-central Utah. Two less intense hot spots were the result of defoliation from western spruce budworm in association with another agent: one with larch needle cast (*Meria laricis*) in ecoregion sections M333D-Bitterroot Mountains, M333B-Flathead Valley, and M333A-Okanogan Highland in northern Idaho and northwestern Montana; and one with aspen defoliation in ecoregion sections M331F-Southern Parks and Rocky Mountain Range and M331G-South Central Highlands in north-central New Mexico and south-central Colorado (fig. 2.3). There were no defoliation hot spots entirely contained within the West Coast region, where western spruce budworm was also an important defoliation agent.

The most intense defoliation hot spot on hardwoods in the North East FHM region, meanwhile, was caused by forest tent caterpillar, along with eastern tent caterpillar, in ecoregion sections 211G-Northern Unglaciated Allegheny Plateau and 211F-Northern Glaciated Allegheny Plateau in north-central Pennsylvania and southwest New York (fig. 2.3). A less intense hot spot located across ecoregion sections 211I-Catskill Mountains and 211F-Northern Glaciated Allegheny Plateau, mainly in New York, were associated with forest tent caterpillar and generic defoliators. Another low-intensity hot spot in eastern Massachusetts, and in ecoregion section 221A-Lower New England, was caused by winter moth (*Operophtera brumata*), Diplodia blight (*Sphaeropsis sapinea*) on select conifer hosts, and gypsy moth.

An intense hot spot of defoliation associated mostly with forest tent caterpillar, along with a comparatively small amount of baldcypress leafroller (*Archips goyerana*), occurred in the South FHM region, in ecoregion sections 232E-Louisiana Coastal Prairie and Marshes and 234C-Atchafalaya and Red River Alluvial Plains in southern Louisiana. The other hot spot in the region was caused by the defoliation of oaks by linden looper (*Erannis tiliaria*) in ecoregion section 231C-Southern Cumberland Plateau in northeastern Alabama (fig. 2.3).

The North Central region's single high-intensity hot spot, in ecoregion section 212H-Northern Lower Peninsula Michigan, was caused largely by gypsy moth, along with some forest tent caterpillar defoliation. Similarly, a less intense hot spot in ecoregion section 212T-Northern Green Bay Lobe in northeast Wisconsin, was associated with gypsy moth with a smaller amount of defoliation by aspen blotchminer (*Lithocolletis tremuloidiella*).

In 2010, three mortality-causing agents and complexes were reported for Alaska, affecting approximately 58 000 ha (table 2.4). Alaska contains approximately 51.3 million ha of forest (Smith and others 2009).

Spruce beetle was the most widely detected mortality agent, affecting about 32 000 ha of forest, mostly in the south-central and southeastern parts of the State. Yellow-cedar (*Chamaecyparis nootkatensis*) decline was the second most widely detected mortality agent, found on about 12 000 ha in the Alaska panhandle. Northern spruce engraver beetle (*Ips perturbatus*) was detected on about 10 000 ha of forest, mostly in the central and east-central parts of the State. The ecoregion sections with the highest percentage of surveyed forest affected by mortality agents were M213A-Northern Aleutian Range and M135A-Northern Chugach Range in southern Alaska, with 1.94 percent and 1.03 percent, respectively, and M129A-Seward Mountains in east-central Alaska, with 1.36 percent (fig. 2.4).

Alaska forests were exposed to 11 defoliation agents and complexes recorded on nearly 464 000 ha (table 2.5) in 2010. Willow leaf blotchminer (*Micrurapteryx salicifoliella*) was the most widely detected defoliator, found on approximately 228 000 ha, mostly in central and east-central Alaska. The next most important defoliator in 2010 was aspen leafminer (*Phyllocnistis populiella*), present on 184 000 ha, again mostly in the eastern and east-central parts of the State. Nonspecific defoliators were detected on nearly 28 000 ha, spruce aphid (*Elatobium abietinum*) was found on about 16 000 ha, and hemlock sawfly (*Neodiprion tsugae*) was observed on approximately 4 000 ha. Twenty percent of the forest surveyed in ecoregion section 139A-Yukon Flats was affected by defoliation agents, by far the highest level of detected defoliation activity (fig. 2.5). Ecoregion

Table 2.4—The three mortality agents and complexes detected in Alaska in 2010

Agents/complexes causing mortality, 2010	Area
	ha
Spruce beetle	31 546.3
Alaska-yellow cedar decline	12 328.4
Northern spruce engraver beetle	9 622.1
Total, all agents	58 096.7

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

Table 2.5—The five leading defoliation agents and complexes detected in Alaska in 2010

Agents/complexes causing defoliation, 2010	Area
	ha
Willow leaf blotchminer	227 639.1
Aspen leafminer	183 539.4
Defoliators	27 649.5
Spruce aphid	16 231.8
Hemlock sawfly	3 680.5
Total, all agents	463 598.9

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

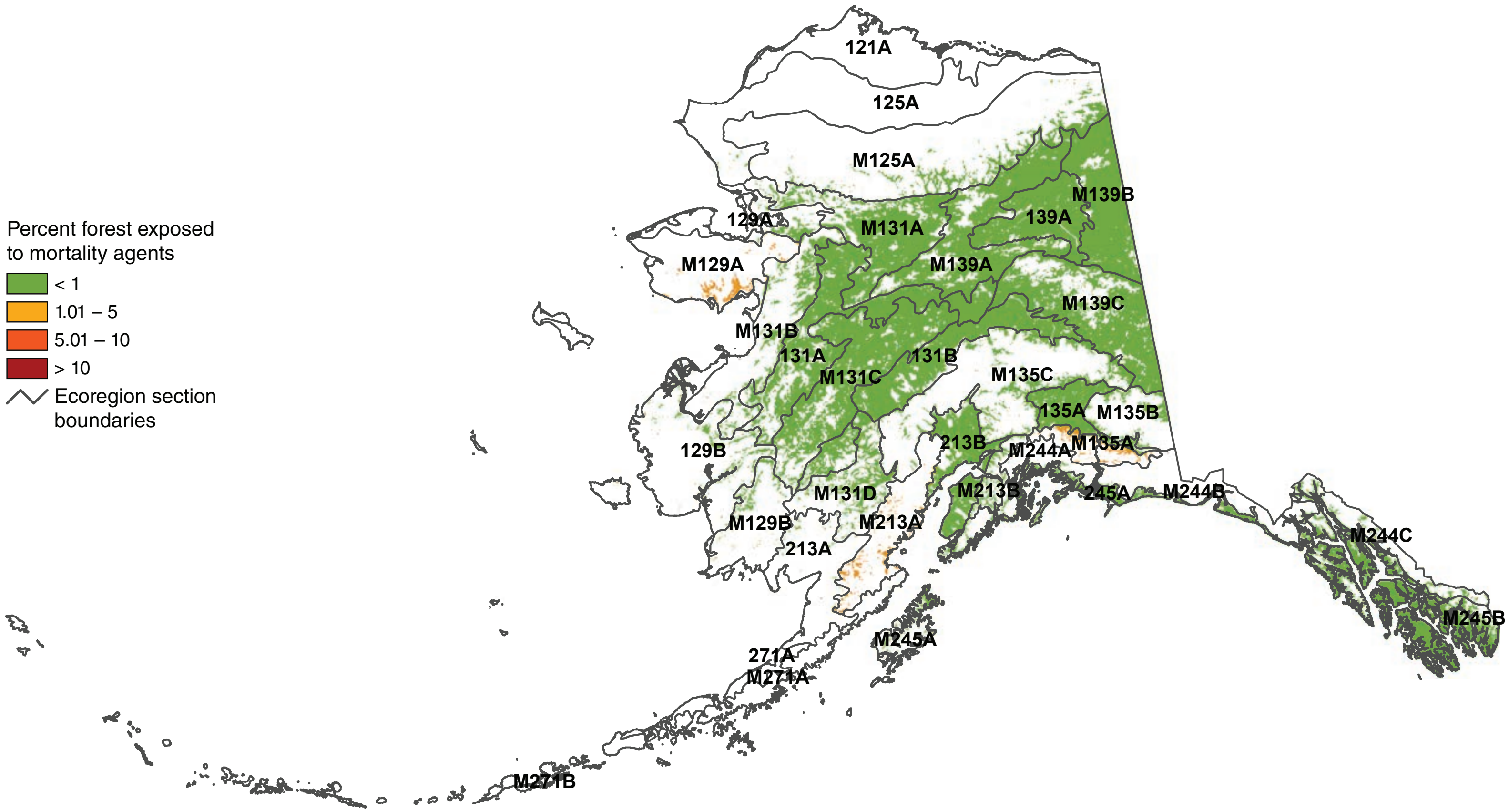

Figure 2.4—Percent of surveyed forest in Alaska ecoregion sections exposed to mortality-causing insects and diseases in 2010. The gray lines delineate ecoregion sections (Nowacki and Brock 1995). Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: USDA Forest Service, Forest Health Protection Program.)

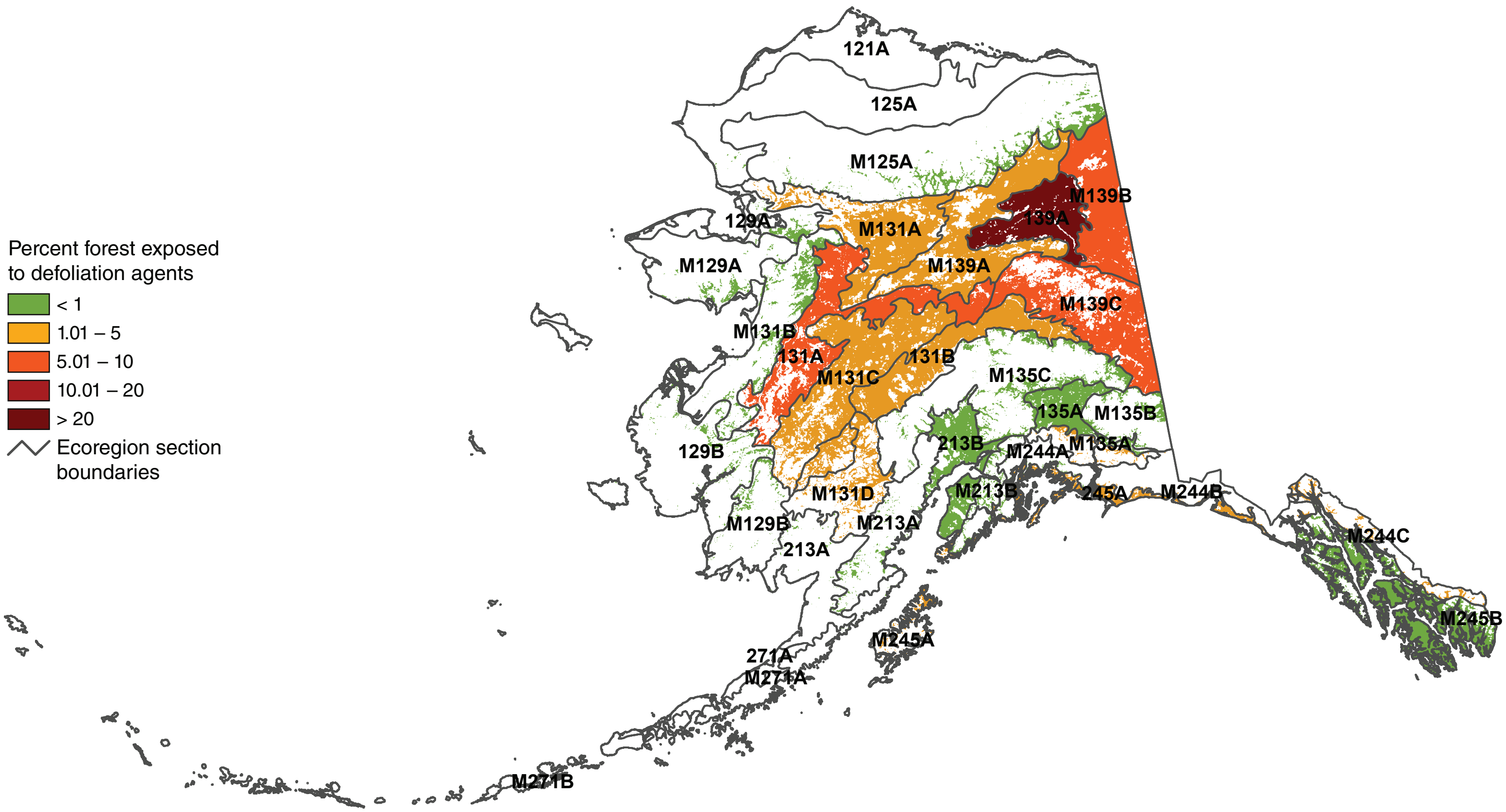

Figure 2.5—Percent of surveyed forest in Alaska ecoregion sections exposed to defoliation-causing insects and diseases in 2010. The gray lines delineate ecoregion sections (Nowacki and Brock 1995). Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: USDA Forest Service, Forest Health Protection Program.)

sections 131A-Yukon Bottomlands, M139B-Olgivie Mountains, and M139C-Dawson Range also had relatively high percentages of forest affected by detected defoliation activity.

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

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CHAPTER 3.
Large-Scale Patterns of Forest Fire Occurrence in the Conterminous United States and Alaska, 2010

Kevin M. Potter

INTRODUCTION

Free-burning fire has been a constant ecological presence on the American landscape, the expression of which has changed as new climates, peoples and land uses have become predominant (Pyne 2010). It is an important ecological mechanism that shapes the distributions of species, maintains the structure and function of fire-prone communities, and is a significant evolutionary force (Bond and Keeley 2005).

At the same time, fire outside the historic range of frequency and intensity can have extensive economic and ecological impacts. 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 in the early 1930s to about 2 million ha in the 1970s (Vinton 2004). In some regions, plant communities are undergoing rapid compositional and structural changes as a result of fire suppression (Nowacki and Abrams 2008). At the same time, fires have become larger, more intense, and more damaging because of the accumulation of fuels (Pyne 2010). Current fire regimes on more than half of the forested area in the conterminous United States have been either moderately or significantly altered from historical regimes, potentially altering key ecosystem components such as species composition, structural stage, stand age, canopy closure, and fuel loadings (Schmidt and others 2002).

Fire suppression and the introduction of nonnative plants, in particular, have dramatically altered fire regimes (Barbour and others 1999). Additionally, fire regimes altered by global climate change could cause large-scale shifts in vegetation spatial patterns (McKenzie and others 1996). Quantifying and monitoring broad-scale patterns of fire occurrence across the United States can help provide a fuller understanding of the ecological and economic impacts of fire, and of 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 useful, or where research into the ecological and socioeconomic impacts of fires may be necessary.

METHODS

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 2011) allows analysts to spatially display and summarize fire occurrences (Coulston and others 2005, Potter 2012a, Potter 2012b, Potter 2013). These are defined as the satellite detection of wildland fire in a 1-km^2 pixel for one day, in a given year. The data are derived using the MODIS Rapid Response System (Justice and others 2002) from the thermal infrared bands of imagery collected daily by two satellites at a resolution of 1 km^2, with the center of a pixel recorded as a fire occurrence when the satellites' MODIS sensors identify the presence of a fire at the time of image collection (USDA Forest Service 2011). The data represent only whether a fire was active, because the MODIS sensors do not differentiate between a hot fire in a relatively small area (0.01 km^2, for example) and a cooler fire over a larger area (1 km^2, for example). The MODIS Active Fire database does well at capturing large fires, but may underrepresent rapidly burning, small and low-intensity fires, as well as fires in areas with frequent cloud cover (Hawbaker and others 2008).

The number of fire occurrences per 100 km^2 (10 000 ha) of forested area was determined for each ecoregion section in the conterminous United States (Cleland and others 2007) and Alaska (Nowacki and Brock 1995) for 2010. This forest fire occurrence density measure was calculated after screening out wildland fires on non-forested pixels using a forest cover layer derived from MODIS imagery by the Forest Service Remote Sensing Applications Center (USDA Forest Service 2008). The total number of fire occurrences across the conterminous States and Alaska was also calculated. The same approach was used to calculate the mean number of annual fire occurrences, per 100 km^2 (10 000 ha) of forested area, by ecoregion section for the first 10 full years of MODIS Active Fire data collection (2001-10).

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 2010. The spatial units of analysis were cells of approximately 2 500 km^2 from a hexagonal lattice of the conterminous United States, intensified from hexagon coordinates for North America from the Environmental Monitoring and Assessment Program (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 non-stationarities in a data set, such as when spatial clustering is concentrated in one subregion of the data (Anselin 1992). Non-stationarities are processes whose statistical properties vary over time or space.

Briefly, G_i* sums the differences between the mean values in a local sample, determined in this case by a moving window of each hexagon and the six neighboring hexagons, and the global mean of all the forested hexagonal cells in the conterminous United States. G_i* is standardized as a *z* score with a mean of 0 and a standard deviation of 1, with values greater than 1.96 representing significant local

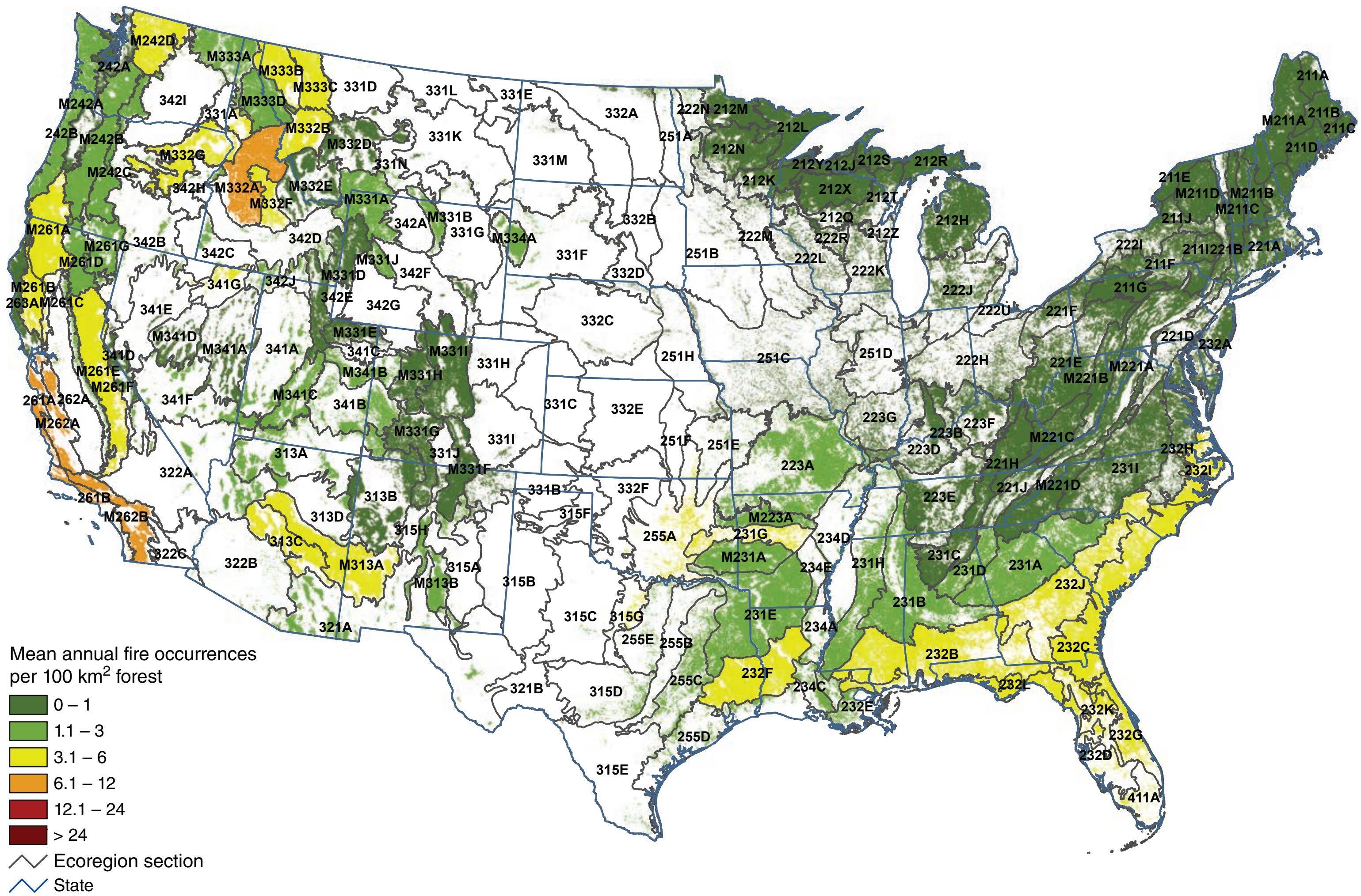

Figure 3.4—Mean number of forest fire occurrences, per 100 km^2 (10 000 ha) of forested area from 2001 to 2010 by ecoregion section in the conterminous United States. The gray lines delineate ecoregion sections (Cleland and others 2007). Forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: USDA Forest Service, Remote Sensing Application Center.)

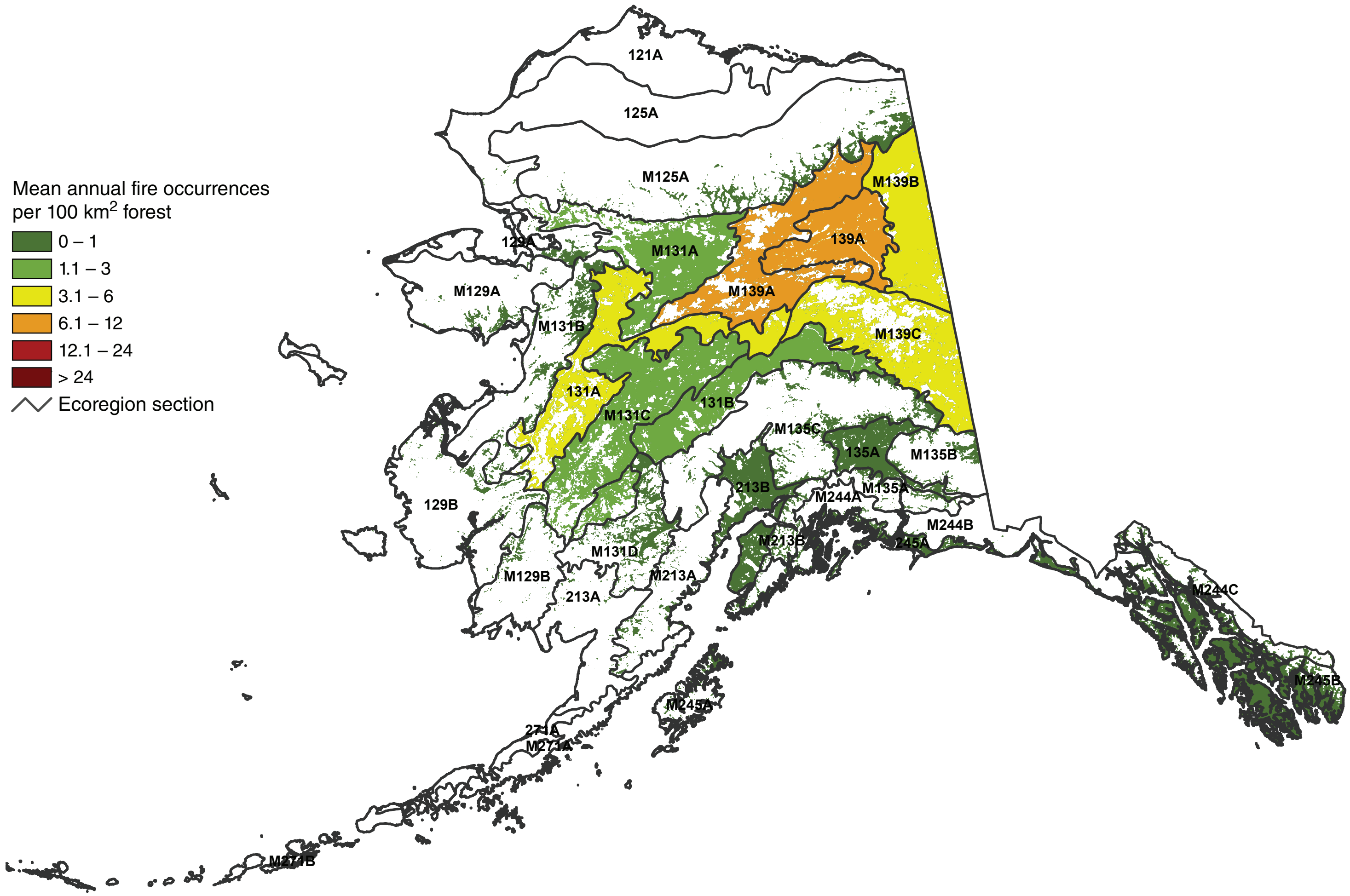

Figure 3.5—Mean number of forest fire occurrences, per 100 km^2 (10 000 ha) of forested area for 2001 to 2010, by ecoregion section within Alaska. The gray lines delineate ecoregion sections (Nowacki and Brock 1995). Forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: USDA Forest Service, Remote Sensing Application Center).

ecoregion sections each had an average of three to five fires per year per 100 km^2 of forested area: M139B-Olgivie Mountains, M139C-Dawson Range, and 131A-Yukon Bottomlands.

While summarizing fire occurrence data at the ecoregion scale allows for a comparison of fire occurrence density over time in an area, a geographical hot spot analysis can offer insights into where fire occurrences are concentrated during a given length of time. In 2010, geographical hot spots of fire occurrence within the conterminous United States were limited to several fairly large hot spots in the Southeastern Coastal Plain, four small hot spots in the central part of the country, and a handful of small hot spots across the West (fig. 3.6). This pattern of mostly small and mostly low-intensity hot spots scattered widely across the country suggests, as in 2009 (Potter 2012b), that wildland forest fires in 2010 were relatively evenly distributed across the conterminous United States, with slightly higher concentrations in a few areas.

The most intense fire hot spot was small, occurring in three ecoregions of eastern Oregon: M332G-Blue Mountains, 342B-Northwestern Basin and Range, and 342H-Blue Mountain Foothills (fig. 3.6). A moderately intense, but large, hot spot was detected in Georgia, Alabama, and Florida, centered on 232B-Gulf Coastal Plains and Flatwoods and 232J-Southern Atlantic Coastal Plains and Flatwoods.

Other low-intensity geographic hot spots of fire occurrence were detected in the following ecoregion sections:

- 232G-Florida Coastal Lowlands-Atlantic, 232D-Florida Coastal Lowlands-Gulf, 232K-Florida Coastal Plains Central Highlands, and 411A-Everglades in southern Florida
- 232C-Atlantic Coastal Flatwoods in South Carolina
- 232F-Coastal-Plains and Flatwoods-Western Gulf in Louisiana and east Texas, and 224C-Atchafalaya and Red River Alluvial Plains in Louisiana
- 234D-White and Black Alluvial Plains and 231H-Coastal Plains-Loess in southeastern Missouri, western Kentucky and Tennessee, and northeastern Arkansas
- 255A-Cross Timbers and Prairie, 251E-Osage Plains, and 251F-Flint Hills in northeastern Oklahoma and southeastern Kansas
- M313A-White Mountains-San Francisco Peaks-Mogollon Rim and 313C-Tonto Transition in central Arizona
- M341C-Utah High Plateau and 341A-Bonneville Basin in southwestern Utah
- M333D-Bitterroot Mountains, M333A-Okanogan Highland, and 331A-Palouse Prairie in northern Idaho
- M261E-Sierra Nevada, M261F-Sierra Nevada Foothills, 341D-Mono, and 341F-Southeastern Great Basin in east-central California

The results of these geographic analyses are intended to offer insights into where fire occurrences have been concentrated, but

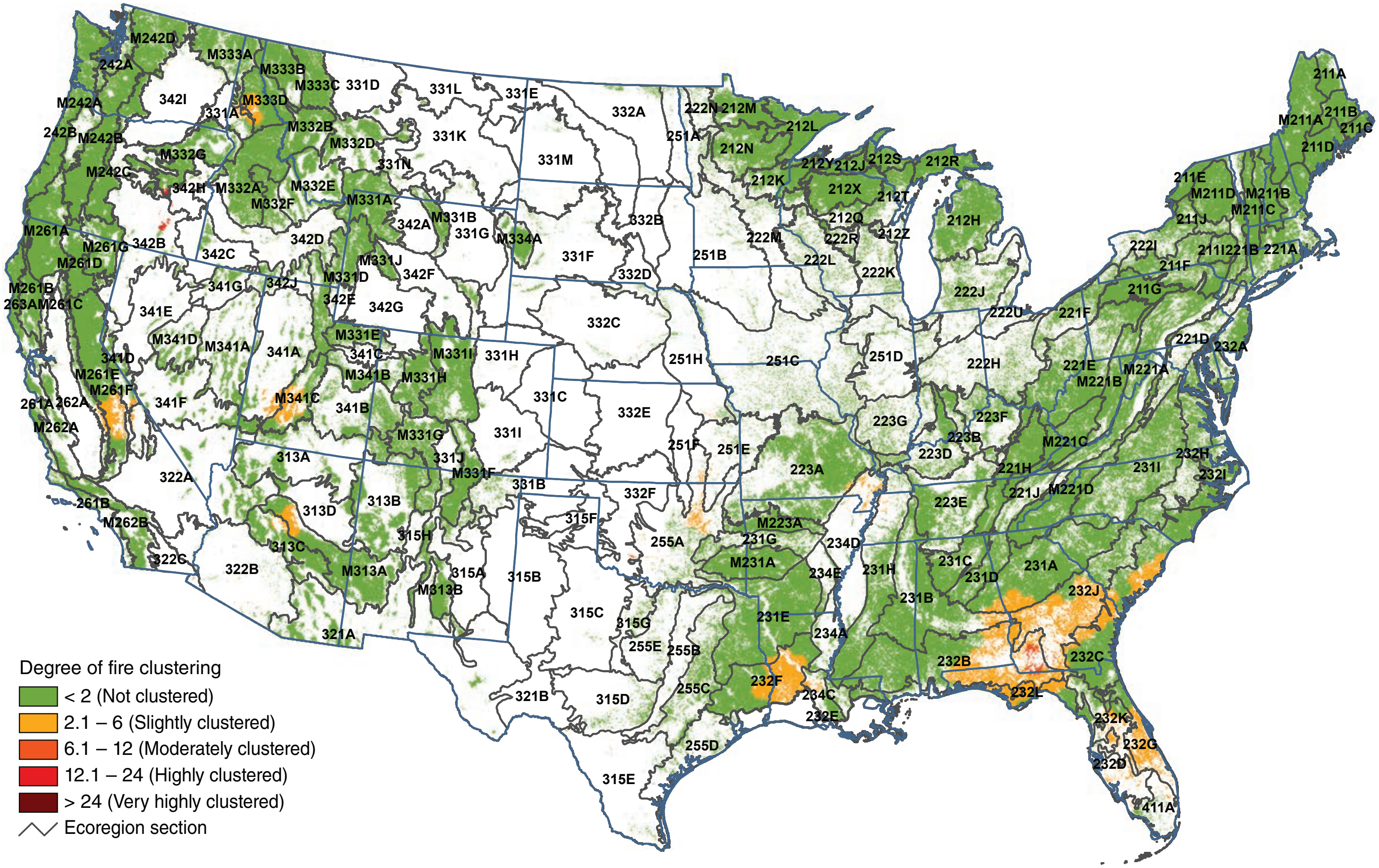

Figure 3.6—Hot spots of fire occurrence across the conterminous United States for 2010. Values are Getis-Ord G_i^* scores, with values greater than 2 representing significant clustering of high fire occurrence densities. (No areas of significant clustering of low fire occurrence densities, -2, were detected). The gray lines delineate ecoregion sections (Cleland and others 2007). Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: USDA Forest Service, Remote Sensing Application Center.)

are not intended to quantify the severity of a given fire season. Information about the concentration of fire occurrences may be useful for the identification of areas for management activities and for follow-up investigations related to the ecological and socioeconomic impacts of fires that may be outside the range of historic frequency.

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CHAPTER 4.

Recent Drought Conditions in the Conterminous United States

Frank H. Koch
William D. Smith
John W. Coulston

INTRODUCTION

Droughts are common in virtually all U.S. forests, but their frequency and intensity vary widely both between and within forest ecosystems (Hanson and Weltzin 2000). Forests in the Western United States generally exhibit a pattern of annual seasonal droughts. Forests in the Eastern United States tend to exhibit one of two prevailing patterns: random occasional droughts, typical of the Appalachian Mountains and of the Northeast, or frequent late-summer droughts, typical of the Southeastern Coastal Plain and the eastern edge of the Great Plains (Hanson and Weltzin 2000). For plants, a reduction in basic growth processes (i.e., cell division and enlargement) is the most immediate response to drought; photosynthesis, which is less sensitive than these basic processes, decreases slowly at low levels of drought stress, but begins to decrease more sharply when the stress becomes moderate to severe (Kareiva and others 1993, Mattson and Haack 1987). Drought makes some forests more susceptible to infestations of tree-damaging insects and diseases (Clinton and others 1993, Mattson and Haack 1987). Furthermore, drought may increase wildland fire risk by impeding decomposition of organic matter and reducing the moisture content of downed woody materials and other potential fire fuels (Clark 1989, Keetch and Byram 1968, Schoennagel and others 2004).

Notably, forests appear to be relatively resistant to short-term drought conditions (Archaux and Wolters 2006), although individual tree species differ in their responses (Hinckley and others 1979, McDowell and others 2008). The duration of a drought event is arguably more significant than its intensity (Archaux and Wolters 2006); for example, multiple consecutive years of drought (2 to 5 years) are more likely to result in high tree mortality than a single dry year (Guarín and Taylor 2005, Millar and others 2007). This suggests that a comprehensive characterization of drought impact in forested areas should include analysis of moisture conditions in the United States over relatively long, i.e., multi-year, time windows.

In the FHM 2010 national report, we outlined a new methodology for mapping drought conditions across the conterminous United States (Koch and others 2013). As in previous work related to this topic (Koch and others 2012a, 2012b), a primary objective of this new methodology was to provide forest managers and researchers with drought-related spatial data sets that are finer-scale than products available from such sources as the National Climatic Data Center (2007) or the U.S. Drought Monitor program (Svoboda and others 2002). The primary 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). A pivotal aspect of our new methodology is a standardized drought indexing approach that allows us to directly compare, for any given location of interest, its moisture status during different time windows, regardless of their length. For example, the FHM 2010 national report includes a comparison of national drought maps for 2009, the 3-year window of 2007–09, and the 5-year window of 2005–09 (Koch and others, 2013).

One of our main goals for the current analysis was to apply the methodology devised for the FHM 2010 national report to the most recently available climate data, i.e., the monthly PRISM data through 2010, thus providing a second time step in what we anticipate to be an ongoing annual record of drought status across the conterminous United States from 2009 forward. In addition, we performed a separate national-scale analysis in which we mapped, for the 100-year period from 1911 to 2010, the frequency of 2, 3, 4, and 5 consecutive years of moderate to extreme drought conditions during the late spring-early summer "season." We focused on this late spring-early summer period because it is a time of peak emergence for certain adult forest insect pests such as the emerald ash borer, *Agrilus planipennis* (Anulewicz and others 2008, Poland and McCullough 2006). Trees that experience acute drought stress during this period may be especially attractive hosts for the newly emerged adults and also more vulnerable to attack, promoting the likelihood of pest outbreaks (Guarín and Taylor 2005, Mattson and Haack 1987). Our interest in consecutive-year frequencies was driven by the idea that any geographic area where this late spring-early summer drought pattern tends to be repeated from year to year faces an even higher outbreak risk, and so should be prioritized for pest surveillance or other management activities.

METHODS

When we performed the analyses, monthly PRISM grids for total precipitation, mean daily minimum temperature, and mean daily maximum temperature were available from the PRISM group Web site (PRISM Group 2010) for all years from 1895 to 2010. Each gridded data set covered the entire conterminous United States. The spatial resolution of these input grids was approximately 4 km (cell area = 16 km^2). However, for the purpose of future applications and 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.

Potential Evapotranspiration Maps

As in our previous work on drought (Koch and others 2012a, 2012b), 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 under ideal conditions, i.e., 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 the Thornthwaite formula (Akin 1991, Thornthwaite 1948):

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

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., 1911–2010) for the conterminous United States. We used a moisture index, *MI'*, proposed by Willmott and Feddema (1992), which has the following form:

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

where

P = precipitation

PET = potential evapotranspiration

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

This set of equations yields a dimensionless index scaled between -1 and 1. MI' can be calculated for any time period, but is commonly calculated on an annual basis using summed P and PET values (Willmott and Feddema 1992). An alternative to this summation approach is to calculate MI' from monthly precipitation and potential evapotranspiration values and then, for a given time window of interest, calculate 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 (1911–2010), we used the mean-of-months approach to calculate moisture index grids for three different time windows: 1 year (MI_1'), three years (MI_3'), and 5 years (MI_5'). Briefly, the MI_1' grids are the mean of the 12 monthly MI' grids for each year in the study period, the MI_3' grids are the mean of the 36 monthly grids from January 2 years prior through December of each year, and the MI_5' grids are the mean of the 60 consecutive monthly MI' grids from January 4 years prior to December of each year. For example, the MI_1' grid for the year 2010 is the mean of the monthly MI' grids from January to December 2010, while the MI_3' grid is the mean of grids from January 2008 to December 2010 and the MI_5' grid is the mean of the grids from January 2006 to December 2010.

Annual and Multi-Year Drought Maps

To determine degree of departure from typical moisture conditions, we first created a normal grid, $MI'_{i\ norm}$, 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_j, for each time window using these gridded data sets:

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

where

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

j = a particular target year in our 100-year study period (i.e., 1911-2010)

MDZ scores may be classified in terms of degree of moisture deficit or surplus (table 4.1). The classification scheme includes categories, e.g., severe drought, extreme drought, like those associated with the Palmer Drought Severity Index (PDSI) (Palmer 1965). Importantly, because of the standardization in equation 3, the breakpoints between categories remain the same regardless of the size of the time window of interest. For comparative analysis, we generated classified MDZ maps, based on all three time

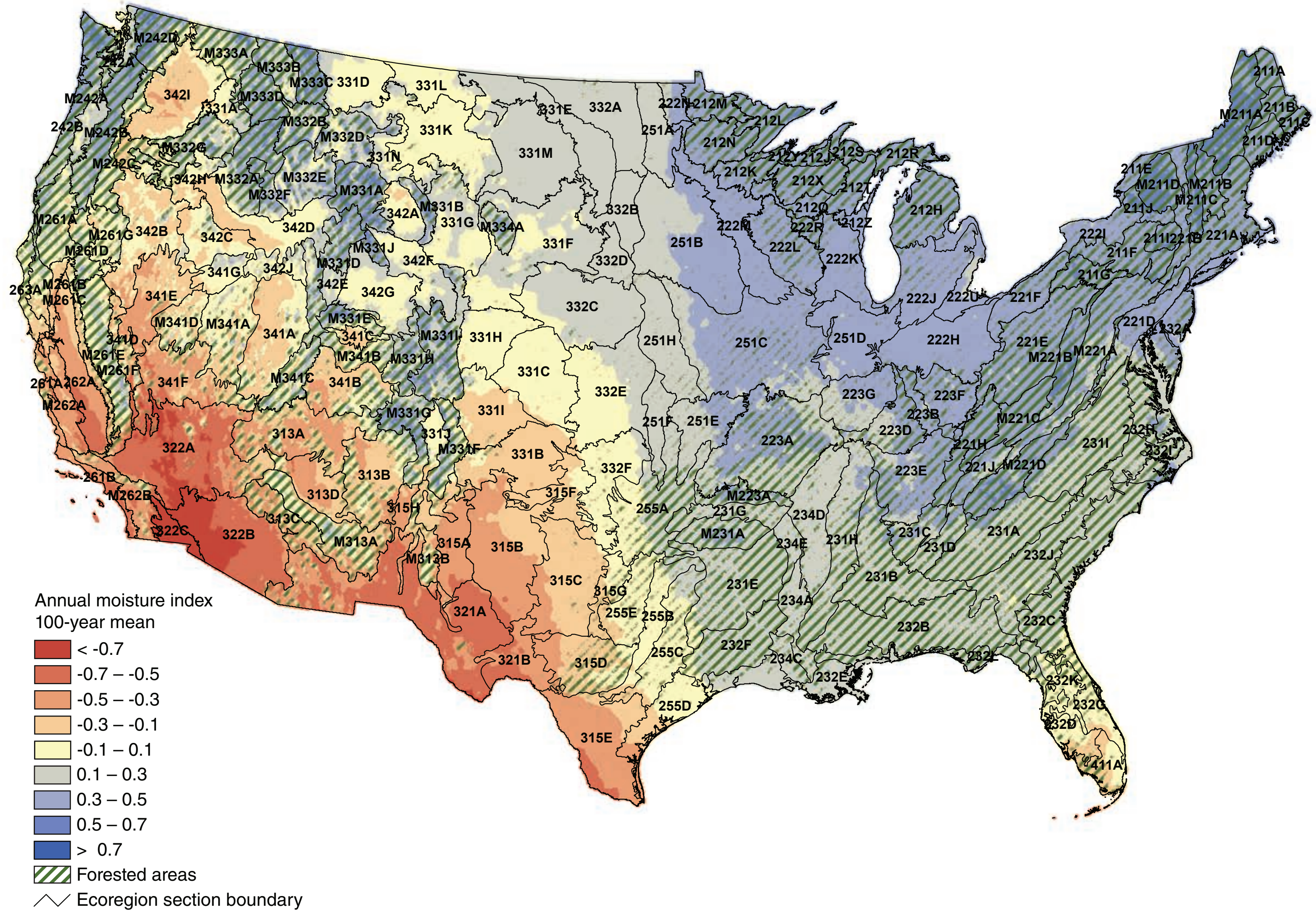

Figure 4.1—The 100-year (1911–2010) mean annual moisture index, or MI_1', for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries and labels are included for reference. Forest cover data (overlaid green hatching) derived from Moderate Resolution Imaging Spectroradiometer (MODIS) imagery by the USDA Forest Service, Remote Sensing Applications Center. (Data source: PRISM Group, Oregon State University.)

windows, for the target year 2010 (figs. 4.2–4.4). Because our analysis focused on drought (i.e., moisture deficit) conditions, we combined the four moisture surplus categories from table 4.1 into a single category for map display.

Frequency of Consecutive Years of Late Spring-Early Summer Drought

As opposed to the mean-of-months approach used in the previously described analyses, for the late spring-early summer drought analysis we calculated MI' (equation 2) based on the total P and PET values summed over a 3-month period. Notably, late spring-early summer represents a different time window depending on geographic location within the conterminous United States, i.e., depending on latitude, elevation, and climatic regime. Hence, we actually calculated nationwide MI' grids for three different 3-month windows during each year of our 1911–2010 study period: March-May, April-June, and May-July. For each of these 3-month windows, we next calculated distinct MI'_{norm} and MI'_{SD} grids based on the window's 100 individual MI' grids calculated for each year of our study period. We then applied equation 3 to generate distinct *MDZ* grids for each window in each year. (In this context, the index i in equation 3 should be interpreted as corresponding to one of the 3-month windows rather than the 1-, 3-, or 5-year windows discussed previously.)

To combine the March-May, April-June, and May-July *MDZ* grids for each year into a single nationwide grid depicting late spring-early summer moisture conditions, we first

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%
-0.5 to 0.5	Near normal conditions	38.2%
0.5 to 1	Mild moisture surplus	15%
1 to 1.5	Moderate moisture surplus	9.2%
1.5 to 2	Severe moisture surplus	4.4%
> 2	Extreme moisture surplus	2.3%

subset them using spatial data related to frost-free period. These data served to represent the approximate beginning of spring and the growing season. Briefly, we divided the conterminous United States into three geographic regions (fig. 4.5) based on the 30-year mean Julian date of the last spring freeze: Zone 1, including all areas with a mean Julian date ≤ 90, i.e., last freeze prior to April 1; Zone 2, all areas with a mean Julian date between 90 and 120, i.e., last freeze between April 1 and April 30; and Zone 3, all areas with a mean Julian date > 120, i.e., last freeze after April 30. Next, we matched each 3-month window to the most appropriate zone (fig. 4.5), and then clipped the corresponding *MDZ* grid to the zonal boundaries. Finally, we created a mosaic of these clipped grids, combining them into a single late spring-early summer grid that covers the conterminous United States.

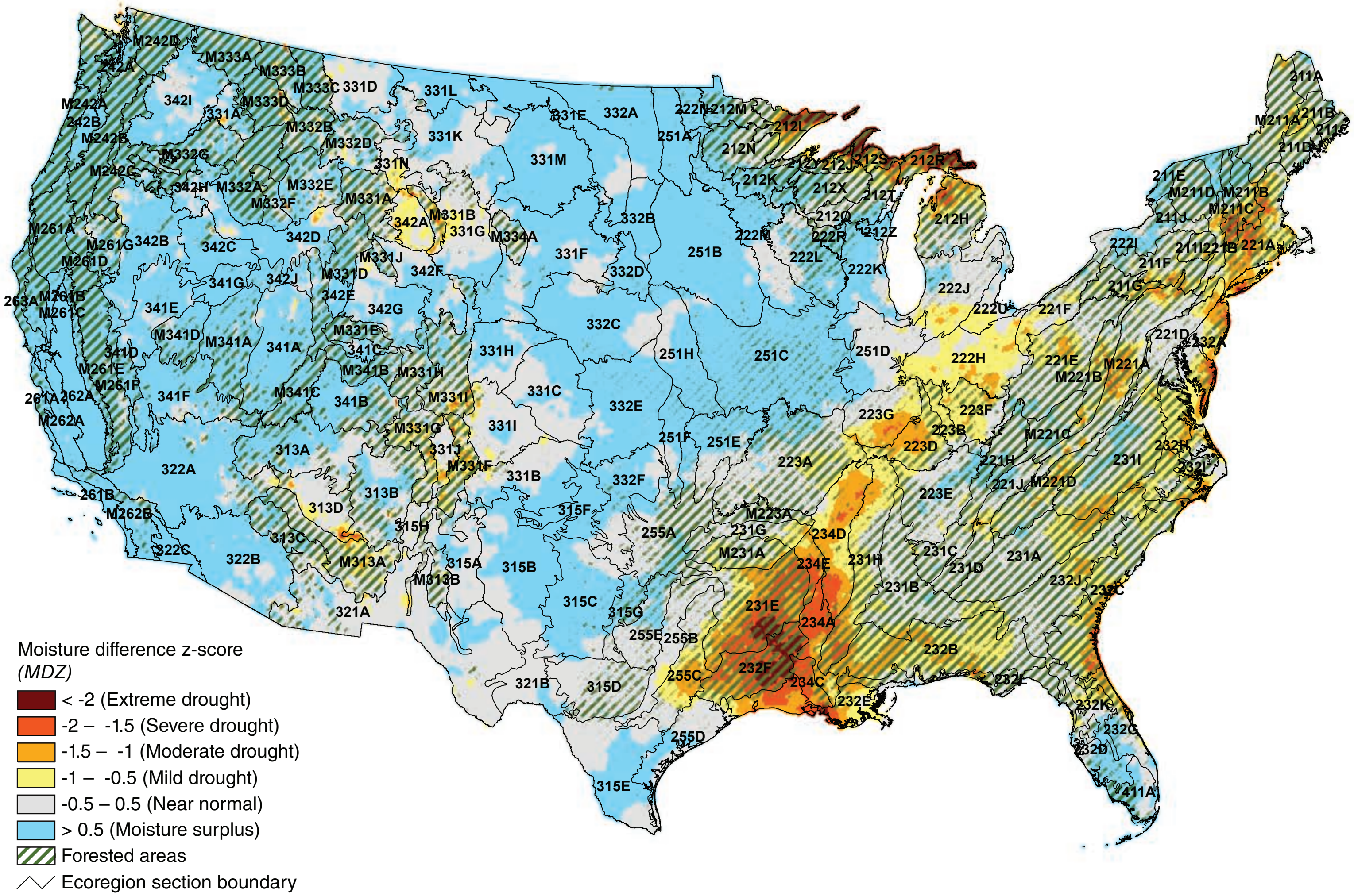

Figure 4.2—The 2010 annual (i.e., 1-year) moisture difference z-score, or MDZ, *for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries and labels are included for reference. Forest cover data (overlaid green hatching) derived from MODIS imagery by the USDA Forest Service, Remote Sensing Applications Center. (Data source: PRISM Group, Oregon State University.)*

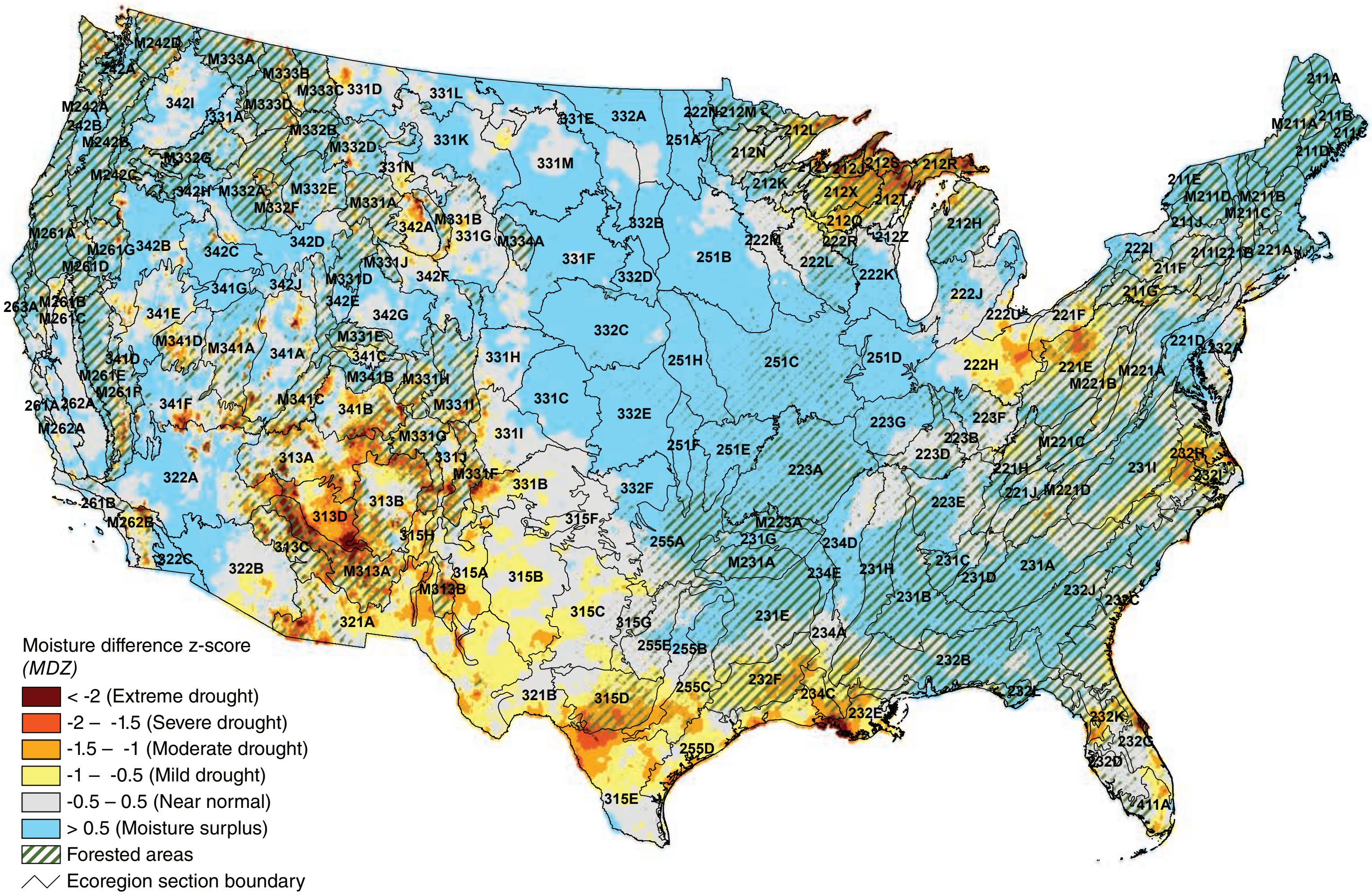

Figure 4.3—The 2008–10 (i.e., 3-year) moisture difference z-score, or MDZ, *for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries are included for reference. Forest cover data (overlaid green hatching) derived from MODIS imagery by the USDA Forest Service, Remote Sensing Applications Center. (Data source: PRISM Group, Oregon State University.)*

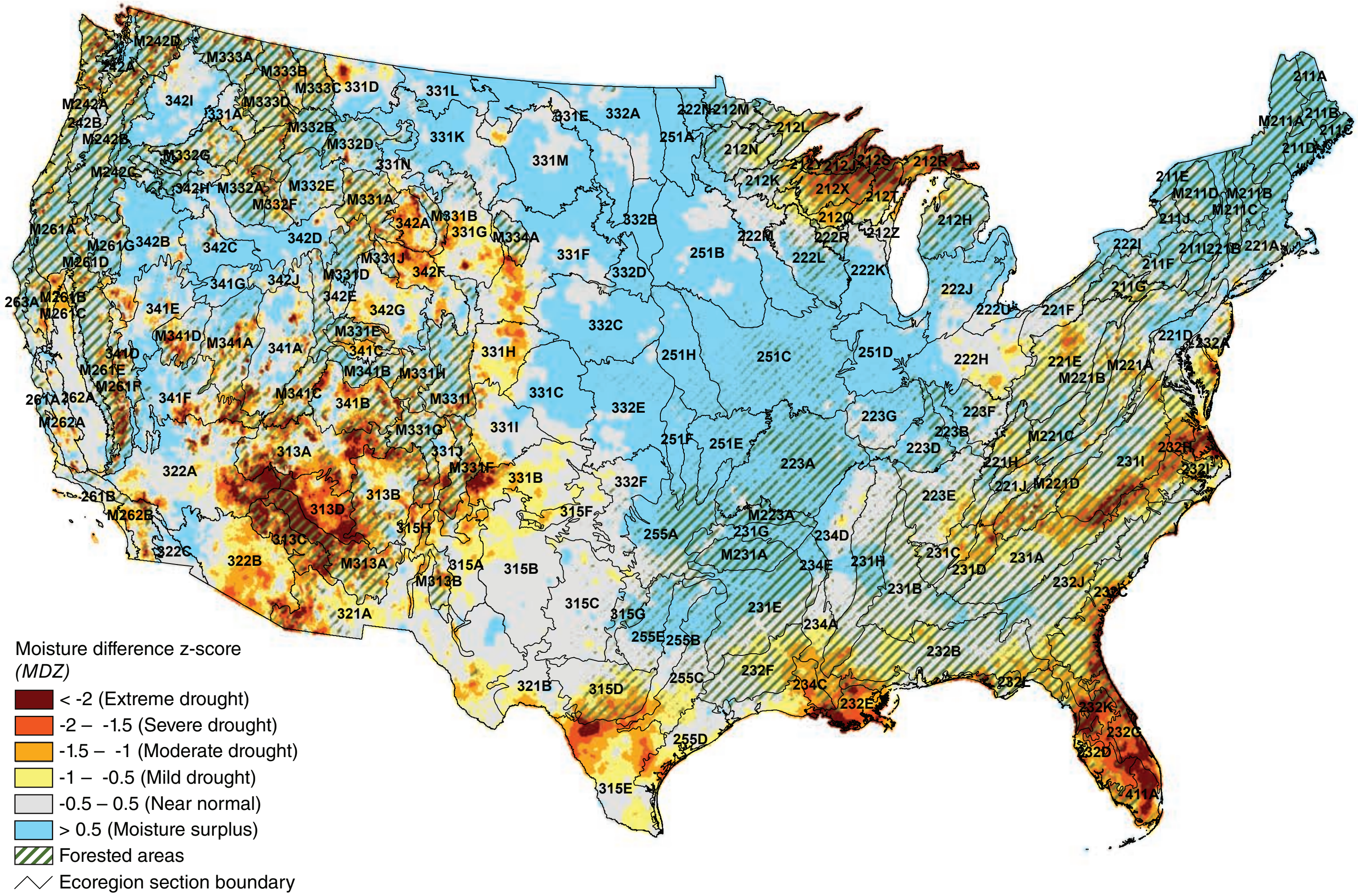

Figure 4.4—The 2006–10 (i.e., 5-year) moisture difference z-score, or MDZ, *for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries are included for reference. Forest cover data (overlaid green hatching) derived from MODIS imagery by the USDA Forest Service, Remote Sensing Applications Center. (Data source: PRISM Group, Oregon State University.)*

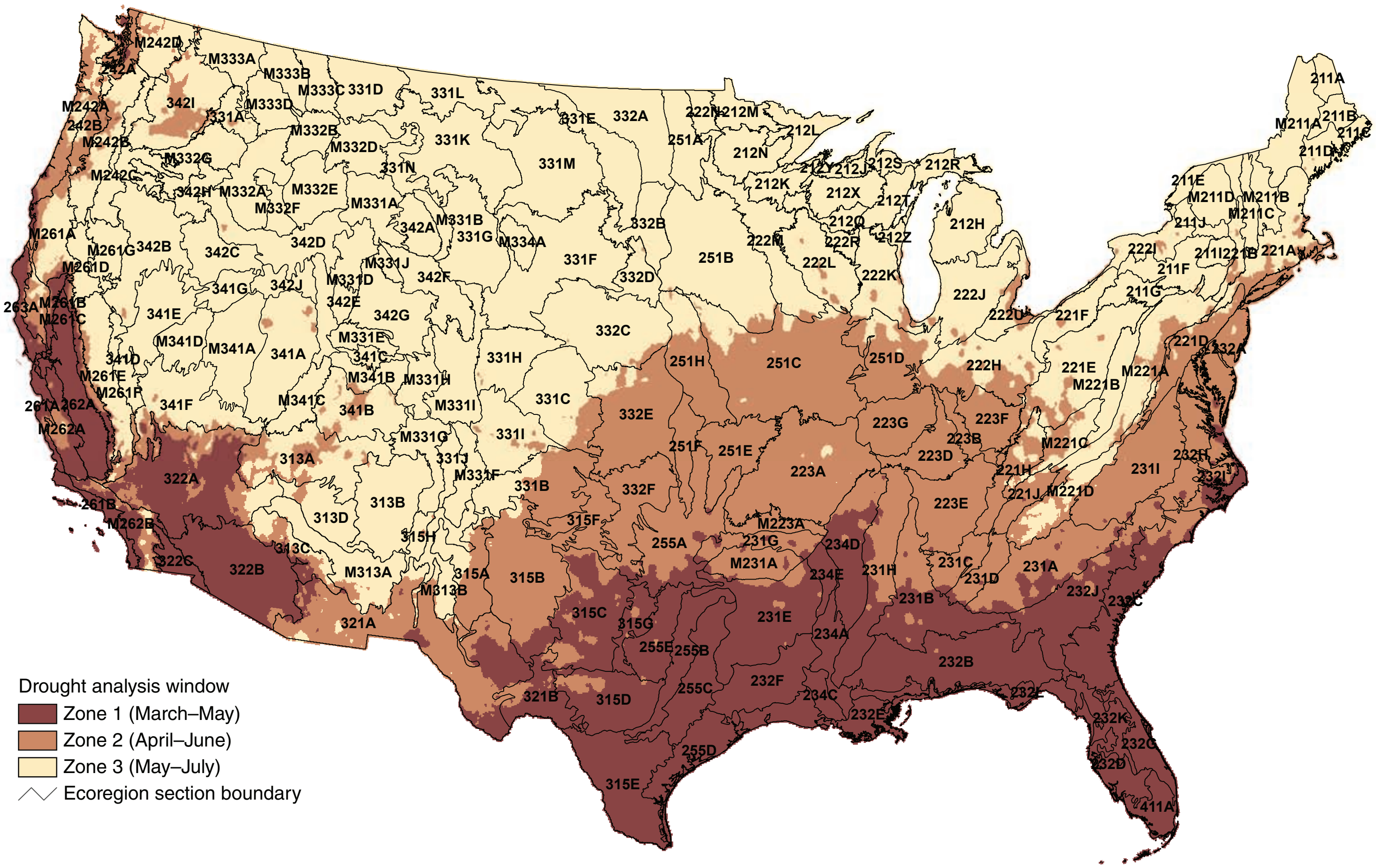

Figure 4.5—Three analysis zones, each corresponding to a particular 3-month time window used when calculating late spring-early summer drought conditions for the associated areas of the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries and labels are included for reference. Zones were developed from data describing frost-free period. (Data source: The Climate Source, LLC, Corvallis, OR.)

To estimate consecutive-year drought frequencies, we began by generating a binary grid from each late spring-early summer grid, assigning all grid cells with *MDZ* values less than -1, i.e., exhibiting moderate to extreme drought stress, a value of 1 and all other cells a value of 0. We stacked the 100 resulting binary grids in annual order, from 1911 to 2010, creating a geographically referenced, three-dimensional array; conceptually, each geographic location, i.e., each grid cell in a map of the conterminous United States, was represented by a vector, V, containing 100 temporally ordered elements (indexed by $x = 1…100$) with a value of 0 or 1. We analyzed each vector V element-by-element to tally the following frequencies (fig. 4.6): (1) the number of times that V_x and V_{x-1} were both equal to 1, indicating 2 consecutive years of moderate to extreme drought during the late spring-early summer season; (2) the number of times that V_x, V_{x-1}, and V_{x-2} were all equal to 1, indicating 3 consecutive years of moderate to extreme drought; (3) the number of times that V_x, V_{x-1}, V_{x-2}, and V_{x-3} were all equal to 1, indicating 4 consecutive years of moderate to extreme drought; and (4) the number of times that V_x, V_{x-1}, V_{x-2}, V_{x-3}, V_{x-4} were all equal to 1, indicating 5 consecutive years of moderate to extreme drought during the late spring-early summer.

RESULTS AND DISCUSSION

The 100-year (1911–2010) mean annual moisture index, or MI_1', grid (fig. 4.1) provides a general illustration of climatic regimes across the conterminous United States. (Because the 100-year mean MI_3' and MI_5' grids were only negligibly different from the mean MI_1' grid, they are not shown here.) In general, wet climates ($MI' > 0$) are characteristic through the Eastern United States, especially the Northeast. Notably, it appears that southern Florida (in particular, ecoregion sections 232C-Florida Coastal Lowlands-Atlantic, 232D-Florida Coastal Lowlands-Gulf, and 411A-Everglades) is the driest region of the Eastern United States. Although this region typically has a high level of precipitation, this is more than offset by a high level of potential evapotranspiration, resulting in negative MI' values. This explanation for the relative dryness of southern Florida, i.e., high P offset by high PET, differs from the circumstances in the driest regions of the Western United States, particularly 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 usually very low. In fact, dry climates ($MI' < 0$) are common across much of the Western United States because of generally lower precipitation than the East. However, mountainous areas in the central and northern Rocky Mountains as well as the Pacific Northwest are relatively wet, e.g., ecoregion sections M242A-Oregon and Washington Coast Ranges, M242B-Western Cascades, M331G-South-Central Highlands, and M333C-Northern Rockies. This is at least partially shaped by high levels of winter snowfall.

Figure 4.2 shows the annual (1-year) *MDZ* map for 2010 for the conterminous United

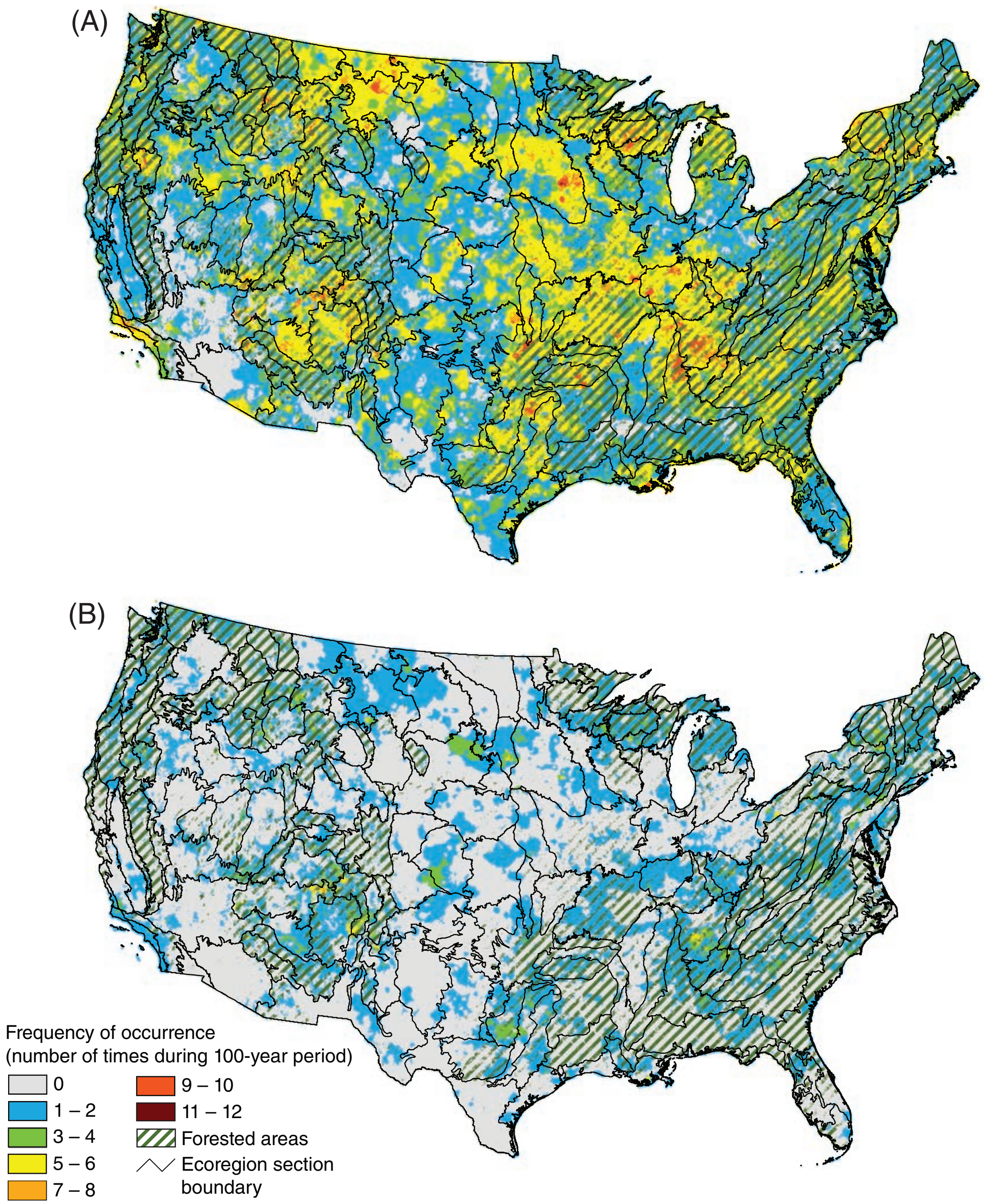

Figure 4.6—Over a 100-year period (1911–2010), the frequency of: (A) 2 consecutive years of moderate to extreme drought conditions during the late spring-early summer for the conterminous United States; (B) 3 consecutive years of moderate to extreme drought during late spring-early summer; (C) 4 consecutive years of moderate to extreme drought during late spring-early summer; and (D) 5 consecutive years of moderate to extreme drought during late spring-early summer. Ecoregion section (Cleland and others 2007) boundaries are included for reference. (Data source: PRISM Group, Oregon State University.) *(continued on next page)*

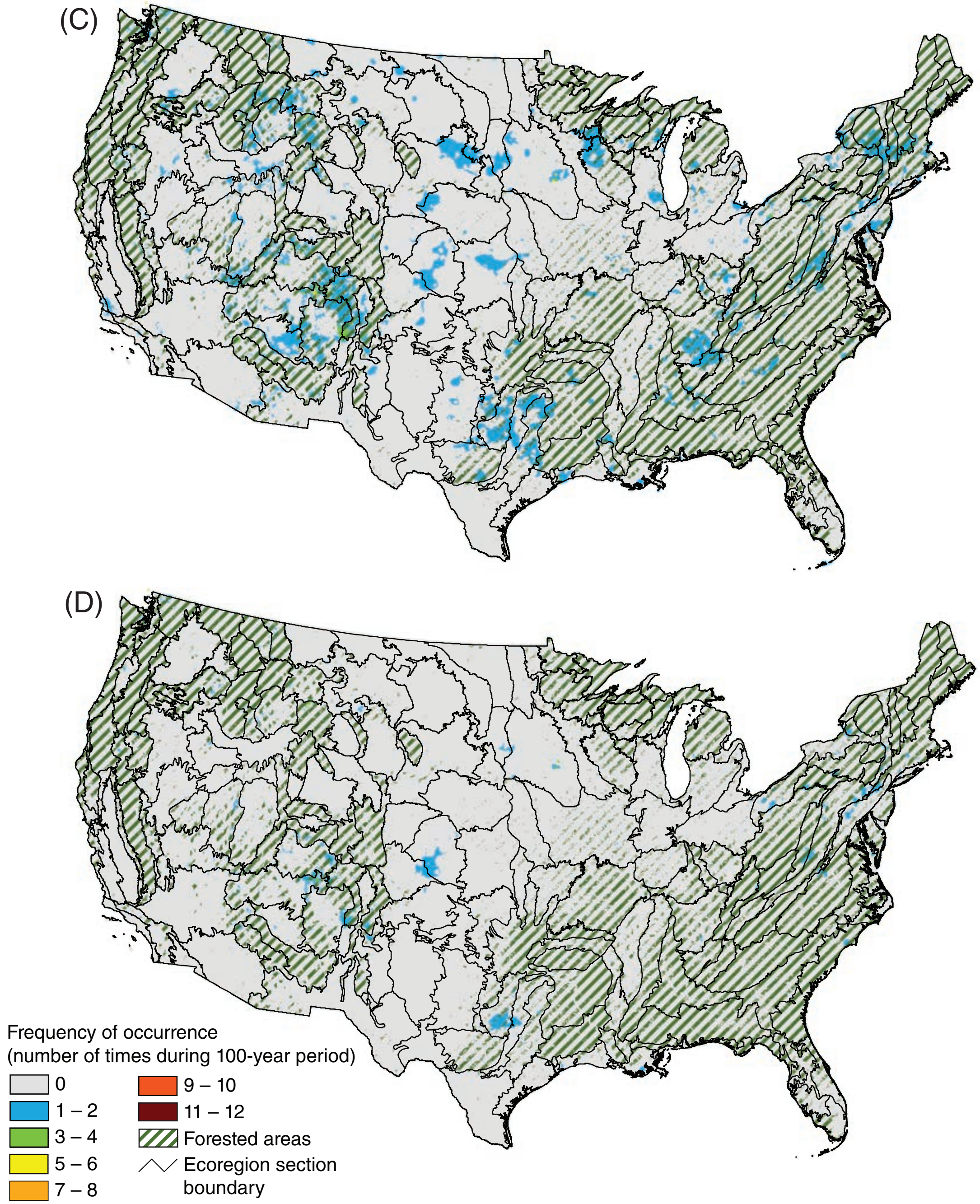

Figure 4.6 (continued)—Over a 100-year period (1911–2010), the frequency of: (A) 2 consecutive years of moderate to extreme drought conditions during the late spring-early summer for the conterminous United States; (B) 3 consecutive years of moderate to extreme drought during late spring-early summer; (C) 4 consecutive years of moderate to extreme drought during late spring-early summer; and (D) 5 consecutive years of moderate to extreme drought during late spring-early summer. Ecoregion section (Cleland and others 2007) boundaries are included for reference. (Data source: PRISM Group, Oregon State University.)

States. Most of the Western United States experienced a moisture surplus in 2010, although there were scattered pockets of moderate to extreme drought, largely limited to ecoregion sections (Cleland and others 2007) in the Rocky Mountain region such as M331B-Bighorn Mountains, M331F-Southern Parks and Rocky Mountain Range, M331G-South Central Highlands, and M331I-Northern Parks and Ranges (as well as the southeastern tip of 313D-Painted Desert, an area that is largely non-forested). This pattern of general moisture surplus in the West is a significant departure from a trend of intense and prolonged region-wide drought during most of the last decade (Groisman and Knight 2008, Mueller and others 2005, NOAA 2010, 2011, O'Driscoll 2007). In contrast, there were fairly extensive areas of drought in the Eastern United States during 2010. Two areas are particularly noteworthy. The first is a large "hot spot" of drought in the Southeastern United States along the central coast of the Gulf of Mexico. This hot spot is centered on the heavily forested sections 231E-Mid Coastal Plains-Western and 232F-Coastal Plains and Flatwoods-Western Gulf, each of which had large areas of severe to extreme drought during 2010. The adjacent (and less heavily forested) sections 232E-Louisiana Coastal Prairie and Marshes, 234A-Southern Mississippi Alluvial Plain, 234C-Atchafalaya and Red River Alluvial Plains, and 234E-Arkansas River Alluvial Plain also contained sizeable areas of severe drought. By way of an explanation, this geographic region had near-record dry conditions throughout the spring and summer of 2010, which was further amplified by record high summer temperatures (NOAA 2011). These conditions have been linked to a marked increase in *Ips* bark beetle damage in this region, resulting in scattered mortality of thousands of trees and, occasionally, high mortality in individual forest stands (Louisiana Department of Agriculture and Forestry 2011). The second hot spot of note is the western Great Lakes region, particularly the heavily forested sections 212L-Northern Superior Uplands, 212R-Eastern Upper Peninsula, and 212S-Northern Upper Peninsula, all of which contained large areas of severe to extreme drought. This portion of the Great Lakes region experienced record dryness during the spring of 2010 (NOAA 2011).

Besides these two prominent drought hot spots, there were numerous pockets of drought distributed across the Eastern United States in 2010 (fig. 4.2). Foremost is a distinctive pattern of moderate to extreme drought along much of the Atlantic Coast, especially in the forested ecoregion sections 221A-Lower New England, 232A-Northern Atlantic Coastal Plain, 232C-Atlantic Coastal Flatwoods, 232H-Middle Atlantic Coastal Plains and Flatwoods, and 232I-Northern Atlantic Coastal Flatwoods. This pattern appears to have been influenced by hot, dry weather that occurred in the region from July to September 2010 (NOAA 2011, NDMC 2011).

When combined with the annual (i.e., single-year) *MDZ* map in figure 4.2, the 3-year (fig. 4.3) and 5-year (fig. 4.4) *MDZ* maps provide an overview of the recent chronology of moisture conditions in the conterminous United States. For instance, the persistent drought conditions that affected much of the Western United States, and especially the Desert Southwest region, during the last decade (Groisman and Knight 2008; Mueller and others 2005; NOAA 2010, 2011; O'Driscoll 2007) are partially captured by the 3-year and 5-year *MDZ* maps. (These two maps contrast strongly with the annual *MDZ* map, which supports the notion that the observed pattern of moisture surplus throughout most of the West in 2010 represents a substantial departure from the region's recent history.)

Additionally, the drought hot spot that appeared in the Great Lakes region during 2010 (see fig. 4.2) is also reflected in the 3-year and 5-year *MDZ* maps, suggesting that drought stress may be a persistent problem for forests in this region. This may similarly be true regarding the previously described hot spot on the central Gulf Coast. It is worth mentioning that in these geographic regions as well as others (e.g., central to southern Florida) the 5-year *MDZ* map (fig. 4.4) appears to show more extensive and/or severe drought conditions than the 3-year *MDZ* map (fig. 4.3). This discrepancy between maps may indicate temporally variable, yet fundamentally persistent, drought conditions in a region of interest, as is the case for the Western United States. However, it may instead be explained by the occurrence of markedly bad drought conditions at some point during the first 2 years of the 5-year *MDZ* window, i.e., 2006–07 for the current analysis. For example, a portion of the Southeastern United States, i.e., parts of sections 231I-Central Appalachian Piedmont, 232H-Middle Atlantic Coastal Plain and Flatwoods, and 232I-Northern Atlantic Coastal Plain and Flatwoods, showed substantially worse drought conditions in the 5-year *MDZ* map than in the 3-year map; a historically exceptional drought that occurred during 2007 (O'Driscoll 2007) is probably the major factor behind this difference. Thus, while the 1-year, 3-year, and 5-year *MDZ* maps together provide a fairly comprehensive short-term overview, it may be additionally important to consider a particular region's longer-term drought history when evaluating the current health level of the region's forests.

With respect to the late spring-early summer drought frequency maps (fig. 4.6), no especially strong geographic pattern emerges, although some parts of the conterminous United States may benefit from further investigation. For example, figure 4.6A highlights a number of areas where two consecutive years of moderate or worse late spring-early summer drought occurred nine or more times between 1911 and 2010; because this represents a fairly large proportion of our 100-year study period, it seems reasonable to assume these highlighted areas face an elevated risk of outbreaks of certain forest pests. Two geographic regions contain the largest clusters of high-frequency areas and may therefore deserve additional attention: the south-central United States (particularly the forested ecoregion sections 223E-Interior Low Plateau-Highland Rim and 231B-Coastal Plains-Middle) and the western

Great Lakes region (particularly the forested sections 212Q-North Central Wisconsin Uplands and 212X-Northern Highlands).

Despite lacking a strong pattern, the moderate level of spatial variability in the 2-consecutive-year drought map suggests that it might serve well as an input to future pest risk mapping projects, i.e., as an additional discriminatory layer to complement data on host distribution, pathways of introduction, and the pest's environmental constraints. In contrast, perhaps the most important thing demonstrated by the 3-, 4- and 5-consecutive-year drought maps (figs. 4.6B–4.6D) is that very little of the country is likely to see a protracted pattern of repeated late spring-early summer droughts. A few ecoregion sections did have small areas where there were multiple, i.e., five or more, occurrences of 3 consecutive years of late spring-early summer drought during our study period (fig. 4.6B), such as the aforementioned section 223E in the south-central United States, and in the West, sections 313A-Grand Canyon and M331G-South Central Highlands. However, < 8 percent of the conterminous United States saw 4 consecutive years of late spring-early summer occur at least once during our 100-year study period, and only 0.2 percent saw this happen more than twice. Furthermore, just over 1 percent of the country experienced 5 consecutive years of late spring-early summer drought at any point during the study period.

A similar set of consecutive-year frequency maps could be produced for any season deemed relevant to a particular forest health issue, e.g., to test drought-related hypotheses pertaining to the issue. In addition to this type of on-demand product, and assuming the spatial data, i.e., the high-resolution maps of precipitation and temperature, underlying these analyses continue to be available for public use, we expect to produce our 1-year, 3-year, and 5-year *MDZ* maps in the future as a standard component of national-scale forest health reporting. Nevertheless, it is important for users to interpret and compare the *MDZ* drought maps cautiously. Although the maps use a standardized index scale that applies regardless of the size of the time window, it should also be understood that, for instance, an extreme drought, i.e., where *MDZ* < -2, that persists over a 5-year period has substantially different forest health implications than an extreme drought over a 1-year period. In future work, we hope to provide forest managers and other decisionmakers with better quantitative evidence regarding some of these relationships between drought 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 also 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.

In early national reports (2001–04) of the Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture, mortality was analyzed using phase 3 data from the FHM and Forest Inventory and Analysis (FIA) programs of the Forest Service. Those data spanned a relatively long time period (for some States, up to 12 years), but the sample was not spatially intense (approximately 1 plot per 96,000 acres). In the 2008 FHM national report (Ambrose 2012), the same method was applied to FIA phase 2 data, which were more spatially intense (approximately 1 plot per 6,000 acres) but came from the relatively small number of States in the Eastern United States where repeated plot measurements had been taken. In the 2009 and 2010 FHM reports, the method was applied to larger areas, using increasing numbers of plots. For this report, the repeated phase 2 data cover much of the Central and Eastern United States, and we can begin to use data from a third cycle of measurements, i.e., a third measurement of the plots.

The mission of the 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 1994). Thus, the aim of this mortality analysis contrasts with how mortality might be approached in other reports, such as FIA State reports or State Forest Health Highlights. The approach to mortality presented here seeks to detect 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.

A mortality baseline is still being established for most of the United States. To discern trends in mortality rates, at least three complete cycles of FIA data are required. With the data currently available, it is only possible to do a spatial comparison of ecoregion sections and identify regions of higher than average mortality (relative to growth) for further study.

DATA

FIA phase 2 inventory data are collected using a rotating panel sample design (Bechtold and Patterson 2005). Field plots are divided into spatially balanced panels, with one panel measured each year. A single cycle of measurements consists of measuring all panels. This "annualized" method of inventory was adopted, State by State, in 1999. An 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.

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

Table 5.1 shows the 36 States from which consistent, repeated phase 2 measurements were available, the time period spanned by

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

Time period	States	Effective sample intensity	Proportion of plots measured a third time
1999–2010	Indiana	1 plot: 6,000 acres	1/5
1999–2010	Wisconsin	1 plot: 3,000 acres[a]	1/5
1999–2009	Maine	1 plot: 6,000 acres	1/5
1999–2009	Minnesota	1 plot: 3,000 acres[a]	0
1999–2009	Missouri	1 plot: 6,000 acres[b]	0
2000–2009	Arkansas, Iowa, Pennsylvania	1 plot: 6,000 acres	0
2000–2010	Michigan	1 plot: 2,000 acres[c]	1/5
2000–2010	Virginia	1 plot: 7,500 acres	0
2001–2009	Illinois, Kansas, Nebraska, South Dakota	1 plot: 7,500 acres	0
2001–2009	Ohio	1 plot: 10,000 acres	0
2001–2010	Alabama	1 plot: 8,400 acres	0
2001–2010	Georgia, North Dakota, Tennessee	1 plot: 6,000 acres	0
2001–2010	Texas[d]	1 plot: 6,000 acres	2/5
2002–2009	Florida	1 plot: 30,000 acres	0
2002–2009	Kentucky	1 plot: 10,000 acres	0
2002–2009	New York	1 plot: 15,000 acres	0
2002–2010	New Hampshire	1 plot: 10,000 acres	0
2002–2010	South Carolina	1 plot: 7,500 acres	0
2003–2009	Massachusetts, Rhode Island	1 plot: 15,000 acres	0
2003–2010	North Carolina	1 plot: 21,000 acres	0
2003–2010	Connecticut, Vermont	1 plot: 10,000 acres	0
2004–2009	Delaware, Maryland, New Jersey, West Virginia	1 plot: 30,000 acres	0

[a] In Minnesota and Wisconsin, the phase 2 inventory was done at twice the standard FIA sample intensity, approximately one plot per 3,000 acres.
[b] In Missouri the phase 2 inventory was done at twice the standard FIA sample intensity, approximately one plot per 3,000 acres on national forest lands, and at the standard intensity on all other lands.
[c] In Michigan the phase 2 inventory was done at triple the standard FIA sample intensity, approximately one plot per 2,000 acres.
[d] Annualized growth and mortality data were only available for eastern Texas.

the data, and the number of panels of data available. Additional measurements of any plot, beyond the minimum of two required for a single mortality estimate, improves the mortality estimate. At present, third plot measurements have been taken in some States (table 5.1). The States included in this analysis, as well as the forest cover within those States, are shown in figure 5.1.

METHODS

FIA phase 2 tree and sapling data were used to estimate average annual tree mortality in terms of tons of biomass per acre. The biomass represented by each tree in tons was calculated by FIA and provided in the FIA Database-version 4.0 (USDA Forest Service 2010). To compare mortality rates across forest types and climate zones, the ratio of annual mortality to gross growth (MRATIO) is used as a standardized mortality indicator (Coulston and others 2005a). Gross growth rate and mortality rate, in terms of tons of biomass per acre, were independently calculated for each ecoregion section (Cleland and others 2007, McNab and others 2007) using a mixed modeling procedure where plot to plot variability is considered a random effect and time is a fixed effect. The mixed modeling approach has been shown to be particularly efficient for estimation using data where not all plots have been measured over identical time intervals (Gregoire and others 1995). In the estimation procedure, within plot temporal correlation was based on a covariance matrix 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 both the 2001 and the 2003 FHM national reports (Coulston and others 2005b, Coulston and others 2005c).

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

In addition, the ratio of average dead tree diameter to average surviving live tree diameter (DDLD ratio) was calculated for each plot where mortality occurred. Low DDLD ratios (much less than 1) usually indicate competition-induced mortality typical of young, vigorous stands, while high ratios (much greater than 1) indicate mortality associated with senescence or some external factors such as insects, disease, or severe drought stress (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.

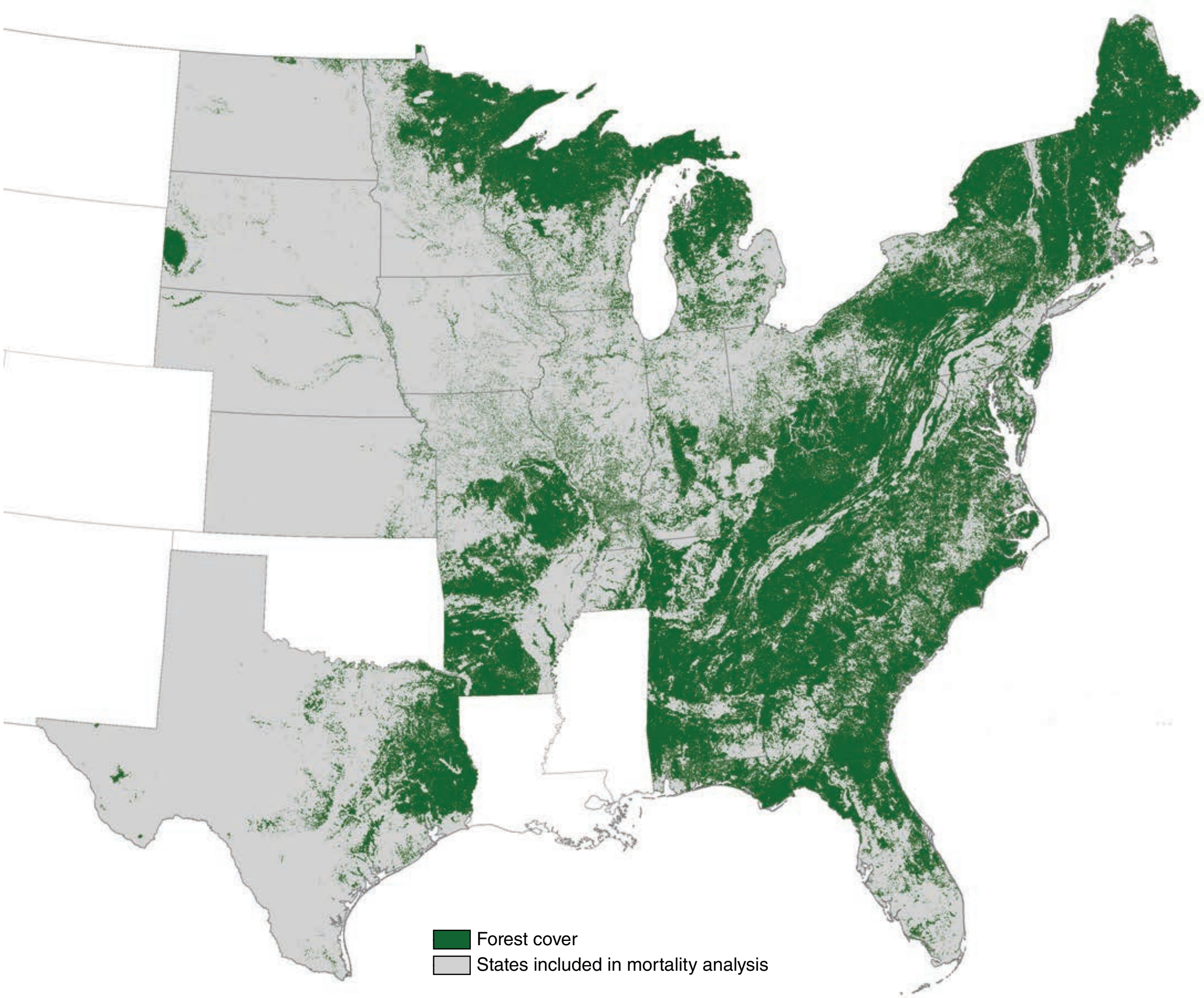

Figure 5.1—Forest cover in the States where mortality was analyzed. Forest cover was derived from Advanced Very High Resolution Radiometer satellite imagery (Zhu and Evans 1994).

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 section. 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 the observed mortality is beyond the scope of this report, often there are well-known insects and pathogens that are "likely suspects" once the affected tree species are identified.

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

RESULTS AND DISCUSSION

The MRATIO values are shown in figure 5.2. Table 5.2 shows the tree species experiencing the greatest mortality in ecoregion sections having MRATIOs of 0.6 or greater.

The highest MRATIO occurred in ecoregion section 331F-Western Great Plains (MRATIO = 1.98) in South Dakota and Nebraska, where mortality actually exceeded growth. Other areas of high mortality relative to growth were ecoregion sections 332D-North-Central Great Plains, also in South Dakota and Nebraska, (MRATIO = 0.82), 232D-Florida Coastal Lowlands (MRATIO = 0.72), 255D-Central Gulf Prairie and Marshes in eastern Texas (MRATIO = 0.70), and 251B-North Central Glaciated Plains, which stretch from southeastern North Dakota to central Iowa (MRATIO = 0.62).

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 we 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. Interestingly, the same pattern of mean and median DDLD close to one and some high DDLD values was observed in nearly all ecoregion sections, regardless of the overall mortality level. So the DDLD analyzed at the ecoregion scale is not very revealing.

In three of the ecoregion sections exhibiting highest mortality relative to growth (331F-Western Great Plains, 332D-North-Central Great Plains, and 251B-North Central Glaciated Plains), the predominant vegetation is grassland, and there were few forested plots measured. Tree growth rates in these regions (especially in ecoregion section 331F) are quite low, so

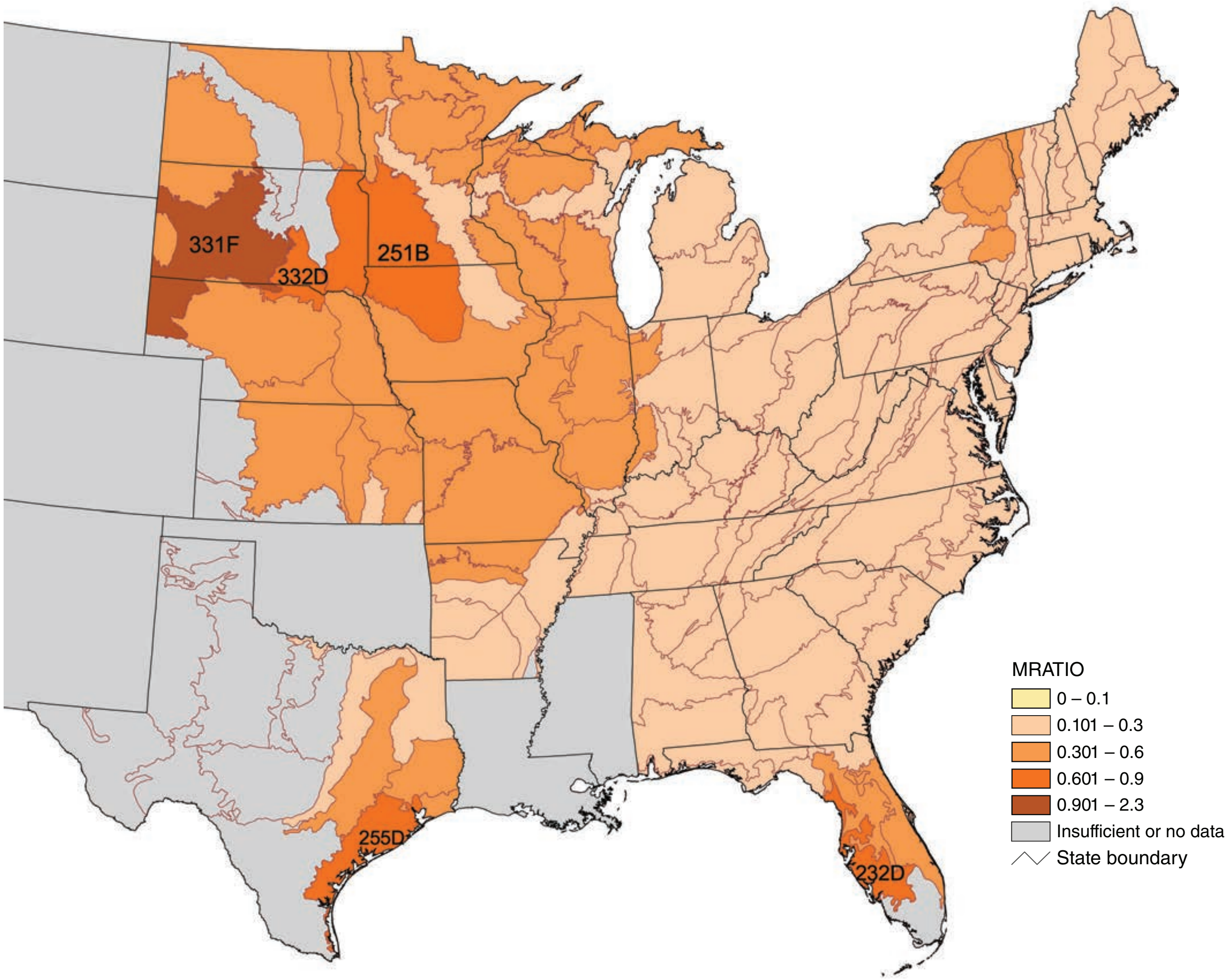

Figure 5.2—Tree mortality expressed as the ratio of annual mortality of woody biomass to gross annual growth in woody biomass (MRATIO) by ecoregion section (Cleland and others 2007). (Data source: USDA Forest Service, Forest Inventory and Analysis Program.)

Table 5.2—Tree species responsible for at least 10 percent of the mortality (in terms of biomass) for ecoregions where the MRATIO was 0.60 or greater

Ecoregion section	MRATIO	Tree species	Percent of total ecoregion mortality biomass	Mean age of dead trees[a]	Species percent mortality (biomass)	Species percent mortality (stems)
331F-Western Great Plains	1.98	Ponderosa pine (*Pinus ponderosa*)	51.21	76	5.51	12.02
		Green ash (*F. pennsylvanica*)	25.48	42	21.80	22.48
332D-North-Central Great Plains	0.82	Bur oak (*Quercus macrocarpa*)	29.35	74	4.99	5.33
		Hackberry (*Celtis occidentalis*)	19.33	60	11.98	6.25
		Green ash (*F. pennsylvanica*)	15.26	77	13.21	19.68
		Ponderosa pine (*P. ponderosa*)	10.91	59	8.18	43.48
232D-Florida Coastal Lowlands	0.72	Live oak (*Quercus virginiana*)	12.44	56	14.42	15.19
		Slash pine (*Pinus elliotii*)	12.25	39	7.61	13.79
255D-Central Gulf Prairie and Marshes	0.70	Loblolly pine (*Pinus taeda*)	26.70	66	8.23	6.80
		Pecan (*Carya illinoensis*)	23.54	60	48.30	28.08
		Water oak (*Quercus nigra*)	21.37	48	21.32	21.48
251B-North Central Glaciated Plains	0.62	American elm (*Ulmus americana*)	35.43	54	32.24	26.12
		Bur oak (*Q. macrocarpa*)	12.30	106	4.65	4.30

[a] Ages are estimated from the stand age as determined by the FIA field crew. It is possible that the age of individual trees that died differed significantly from the stand age, especially in mixed-species stands.

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

Ecoregion section	Mean DDLD	Maximum DDLD	Median DDLD	Minimum DDLD	MRATIO
255D-Central Gulf Prairies and Marshes	1.29	3.16	1.16	0.28	0.70
232D-Florida Coastal Lowlands	1.13	7.66	0.90	0.22	0.72
251B-North Central Glaciated Plains	1.00	4.44	0.74	0.12	0.62
331F-Western Great Plains	0.98	3.29	0.91	0.22	1.98
332D-North-Central Great Plains	0.89	1.83	0.96	0.29	0.82

the high MRATIOSs are due to a combination of low growth and high mortality. Most of the forest in these sections is riparian forest, and, indeed, 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 typically a part of the plains ecosystem, so one suspects that the pine mortality is occurring on plots close to ecoregion section M334A-Black Hills (perhaps on plots actually in the Black Hills but included in ecoregion section 331F-Western Great Plains due to mapping error).

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 fairly low, as would be the case when remnant larger trees die, leaving young, vigorous stands behind. Tree growth is generally slow in these ecoregion sections because of naturally dry conditions. Where the number of sample plots is small and tree growth is slow, care must be taken in interpreting mortality relative to growth over short time intervals.

In ecoregion section 331F-Western Great Plains, where the MRATIO was highest (MRATIO = 1.98), by far the largest amount of biomass that died was ponderosa pine (table 5.2); however, this represented a relatively small proportion of the ponderosa pine in the ecorgion. Green ash, which made up only half as much of the ecoregion mortality as ponderosa pine, suffered a much larger proportional loss of the total ash stock (about 22 percent of both biomass and stems). This suggests that ash may be suffering from much more serious forest health issues than pine in this ecoregion.

In ecoregion section 332D-North-Central Great Plains, four species experienced the highest total mortality in terms of biomass and together represent about 75 percent of the mortality in the ecoregion: bur oak, hackberry, green ash, and ponderosa pine. Of these, hackberry and green ash suffered the greatest proportional loss of biomass (11.98 and 13.21 percent, respectively). The relatively high mean age of the dead trees suggests that the mortality is at least partially due to senescence of older stands.

One might be tempted to suspect the invasive insect, the emerald ash borer as the cause of the ash mortality in ecoregion sections 331F-Western Great Plains and 332D-North-Central Great Plains. However, this pest had not yet been reported in or near these regions as of the time that the mortality data were collected or the time of this writing (USDA Forest Service and others 2011, N.d.). More likely possible causes of the ash mortality include ash yellows (Pokorny and Sinclair 1994), environmental conditions, or simply senescence of older stands.

In ecoregion section 232D-Florida Coastal Lowlands, live oak and slash pine each represented about 12 percent of the mortality. The causes are unclear. Researchers in Florida are investigating pests that effect slash pine (southern pine beetle) and oak (variable oakleaf caterpillar). However, these research and monitoring efforts are focused in northern Florida, not in most of the area experiencing high mortality (Florida Department of Agriculture and Consumer, Division of Forestry 2009).

In ecoregion section 255D-Central Gulf Prairie and Marshes in eastern Texas, most of the mortality occurred in loblolly pine, pecan, and water oak. Of these, pecan suffered the largest proportional loss (48.3 percent of biomass and 28.08 percent of stems). The causes of this mortality are not readily apparent. In the case of water oak, one might suspect oak wilt, which is a major problem in much of Texas. However, oak wilt has not been confirmed in much of this ecoregion (Appel and others 2008).

In ecoregion section 251B-North Central Glaciated Plains, by far the largest amount of biomass that died was American elm. Elm also suffered the largest proportional loss, in terms of both biomass (32.24 percent) and number of stems (26.12 percent). Dutch elm disease is the suspected cause. The pathogen which causes it is known to occur throughout the Midwest, including every county of Iowa since 2002 (Feeley 2010). Dutch elm disease has severely affected riparian forests in North Dakota (North Dakota Forest Service 2007). The disease is also reported to be a problem in Minnesota (Minnesota DNR 2009) and nearby Illinois (Illinois DNR 2009).

The mortality pattern shown in these analyses do not immediately suggest large-scale forest health issues. Mortality is rather low in most of the areas for which data are available. The areas of highest mortality occur in the mostly riparian forests of several plains ecoregions. Further study of the health of these forests may be warranted. Further investigation may also be warranted into the causes of mortality in the Gulf Coast ecoregions of Florida and Texas.

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SECTION 2.

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

CHAPTER 6.

Fragmentation of Eastern United States Forest Types

Kurt H. Riitters
John W. Coulston

INTRODUCTION

Fragmentation is a continuing threat to the sustainability of forests in the Eastern United States, where land use changes supporting a growing human population are the primary driver of forest fragmentation (Stein and others 2009). While once mostly forested, approximately 40 percent of the original forest area has been converted to other land uses, and most of the remainder is not original forest (Smith and others 2009). The direct loss of forest land is an obvious threat; less obvious are the threats posed by isolation and edge which encompass a wide range of negative biotic and abiotic influences on remnant forest (e.g., Forman and Alexander 1998, Harper and others 2005, Laurance 2008, Murcia 1995, Ries and others 2004). Landcover data from 1992 indicated that forest tended to be dominant and well-connected where it occurred, but also that fragmentation was so pervasive that only 10 percent of the eastern forest area was not fragmented at a landscape scale of 66 ha, and that at least 40 percent of forest area was within 90 m of forest edge (Riitters and others 2002, 2004). Between 1992 and 2001, there was a net loss of interior forest in the east, and landscapes once dominated by forest are now dominated by other land uses (Wickham and others 2007, 2008). In 16 of the 31 Eastern States, the wildland-urban interface now encompasses more than 25 percent of total land area (Radeloff and others 2005), and one-third of the eastern forest exists within neighborhoods that also contain at least 10 percent agricultural landcover (Riitters 2011).

The objective of this section is to demonstrate an approach to improve national assessments of forest fragmentation by incorporating information about the specific forest types that are fragmented. National assessments are appropriately based on high resolution, wall-to-wall landcover maps (Heinz Center 2008), but the current generation of those maps does not describe in much detail the forest types that are fragmented. Such information could improve land management and policy by identifying forest types of special concern for conservation or remediation, especially if fragmentation is related to specific ecological services like wildlife habitat or water quality (e.g., Burkhard and others 2009; Kienast and others 2009). The approach demonstrated here combines landcover data from the 2001 National Land Cover Data (NLCD) landcover map (Homer and others 2007) with field plot information from the Forest Inventory and Analysis (FIA) Program of the Forest Service, U.S. Department of Agriculture (USDA Forest Service 2010). We evaluate the fragmentation status of forest types in the Eastern United States (fig. 6.1) and estimate the area of intact forest by forest type.

Figure 6.1—The study area includes 31 Eastern States.

METHODS

Bechtold and Patterson (2005) provide a detailed description of the FIA inventory which may be summarized as follows. The FIA inventory uses a permanent, national, grid-based, equal probability sample design across all land. Each sample location is determined to be either a forest land use or a non-forest land use. For those locations determined to be a forest land use, a field inventory plot is installed to collect additional information. A variety of site and vegetation measurements are taken on a cluster of four fixed-area subplots spanning approximately 0.4 ha, which may extend into more than one forest type. FIA uses a post-stratified estimator, which accounts for different sampling intensities that arise because of intentional increases in sample size or unintentionally as a result of survey nonresponse. In effect, each plot has a weight factor that accounts for those differences. In addition, each within-plot forest type is weighted by its relative area on the field plot. The area estimates that we report were derived by combining the two weight factors (Bechtold and Patterson 2005). We used data from 152,804 plot locations across the study area, using the most recent measurement for measurement years 2000 to 2008. Forest types were defined by FIA protocols (USDA Forest Service 2010). We selected 75 of the 92 forest types in the FIA database by excluding nonstocked forest land and the forest types which occupied less than 70 000 ha each.

Fragmentation was measured using the 2001 NLCD landcover map (Homer and others 2007). The NLCD map identifies 16 landcover types at a spatial resolution of 0.09 ha per pixel and a minimum mapping unit of 0.45 ha. The 16 NLCD landcover types were combined into two generalized landcover types called forest (including the NLCD deciduous forest, evergreen forest, mixed forest, and woody wetlands classes) and non-forest (including all other NLCD classes). Forest area density (Pf), defined as the proportion of a fixed-area neighborhood that has forest landcover, was measured within a 4.41 ha (7 pixel X 7 pixel) neighborhood centered on each inventory plot location (Riitters and others 2002). That neighborhood size was large enough to reliably estimate Pf yet small enough to characterize fragmentation in the immediate vicinity of a field plot. Pf was converted to a categorical variable (Pf class) with seven classes labeled as intact ($Pf = 1.0$), interior ($0.9 \leq Pf < 1.0$), dominant ($0.6 \leq Pf < 0.9$), transitional ($0.4 \leq Pf < 0.6$), patchy ($0.1 \leq Pf < 0.4$), rare ($0.0 < Pf < 0.1$), and none ($Pf = 0.0$). The class “none” was included because it was possible for inventory plots to occur in neighborhoods containing no forest landcover. The Pf class was then treated as a new plot-level attribute when using the FIA weight factors to summarize Pf classes by forest types.

RESULTS

The percentage of each forest type’s total area that is in each of the seven Pf classes is shown in figure 6.2. The forest types are sorted in

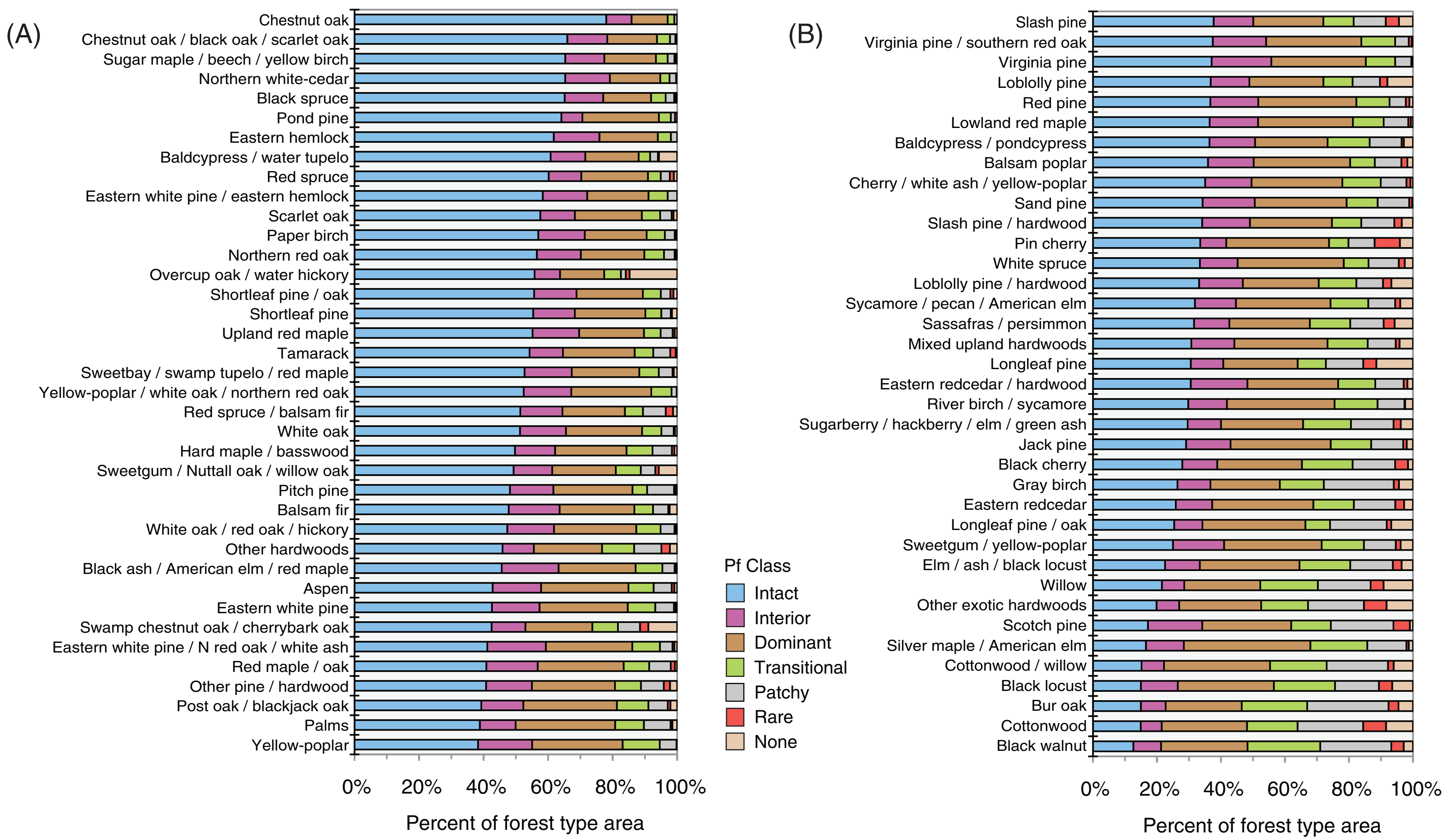

Figure 6.2—The percentage of total forest type area in each of seven forest area density (Pf) classes, sorted descending by percentage in the intact area density class. Forest type nomenclature is from appendix F of USDA Forest Service (2010).

descending order by percentage of intact forest landcover, such that the forest type with the highest percentage (chestnut oak) is at the top of figure 6.2A and that with the lowest percentage (black walnut) is at the bottom of figure 6.2B. In figure 6.3, the estimated area of intact forest landcover is shown for each forest type sorted in descending order. Note the scale change on the x-axis between figure 6.3A and figure 6.3B.

DISCUSSION

Over all forest types, approximately 81 percent of forest area was contained in a neighborhood that consisted of at least 60 percent forest landcover (Pf classes dominant, interior, and intact), and approximately 45 percent was contained in a neighborhood with intact forest landcover. While these results apply to forest land area as defined by the FIA inventory, they are generally consistent with earlier estimates of dominant and intact eastern forest that were made for forest landcover in general (Riitters and others 2002, Wickham and others 2008). The high percentage (81 percent) of area with sufficient forest landcover to qualify as dominant indicates that forest landcover tends to be dominant where forest occurs, and the low percentage (45 percent) of intact forest indicates that fragmentation is pervasive.

The percentage area in the intact forest area density class varied from 13 percent to 78 percent among individual forest types. Fragmentation would be considered a natural attribute of many of the forest types that exhibited low percentages of intact forest. For example, cottonwood and willow are typical of narrow riparian forests in the semi-arid western part of the study area, and intactness is lost from fragmentation by water. Bur oak is an example of naturally fragmented forest in savannah regions where fragmentation by grass-shrub landcover is a natural condition. Forest types exhibiting the largest percentages of intact forest are partly explained by (lack of) accessibility due to steep slopes, e.g., chestnut oak, or hydric soils, e.g., northern white cedar, black spruce, pond pine. Perhaps the best evidence for the pervasiveness of fragmentation is between those extremes, for the forest types that are not naturally fragmented and that occur in relatively accessible locations; typically less than half of the area of those forest types qualified as intact forest in a modest 4.41 ha neighborhood. Except for "natural" fragmentation by water or grassland, the majority of that fragmentation is associated with anthropogenic land uses such as agriculture, housing, and infrastructure (Riitters and Coulston 2005, Wade and others 2003).

The regional supply of intact forest is driven more by total area than by the characteristics of individual forest types. A large share of the total area of intact forest was contributed by the sugar maple/beech/yellow birch forest type (fig. 6.2A), which exhibited the second-largest percentage of intact forest on a per-forest type basis and which occupied a large share of total forest area. In contrast, large shares of total intact forest area were also contributed by three forest types (mixed upland hardwoods, loblolly pine, white oak/red oak/hickory) that individually

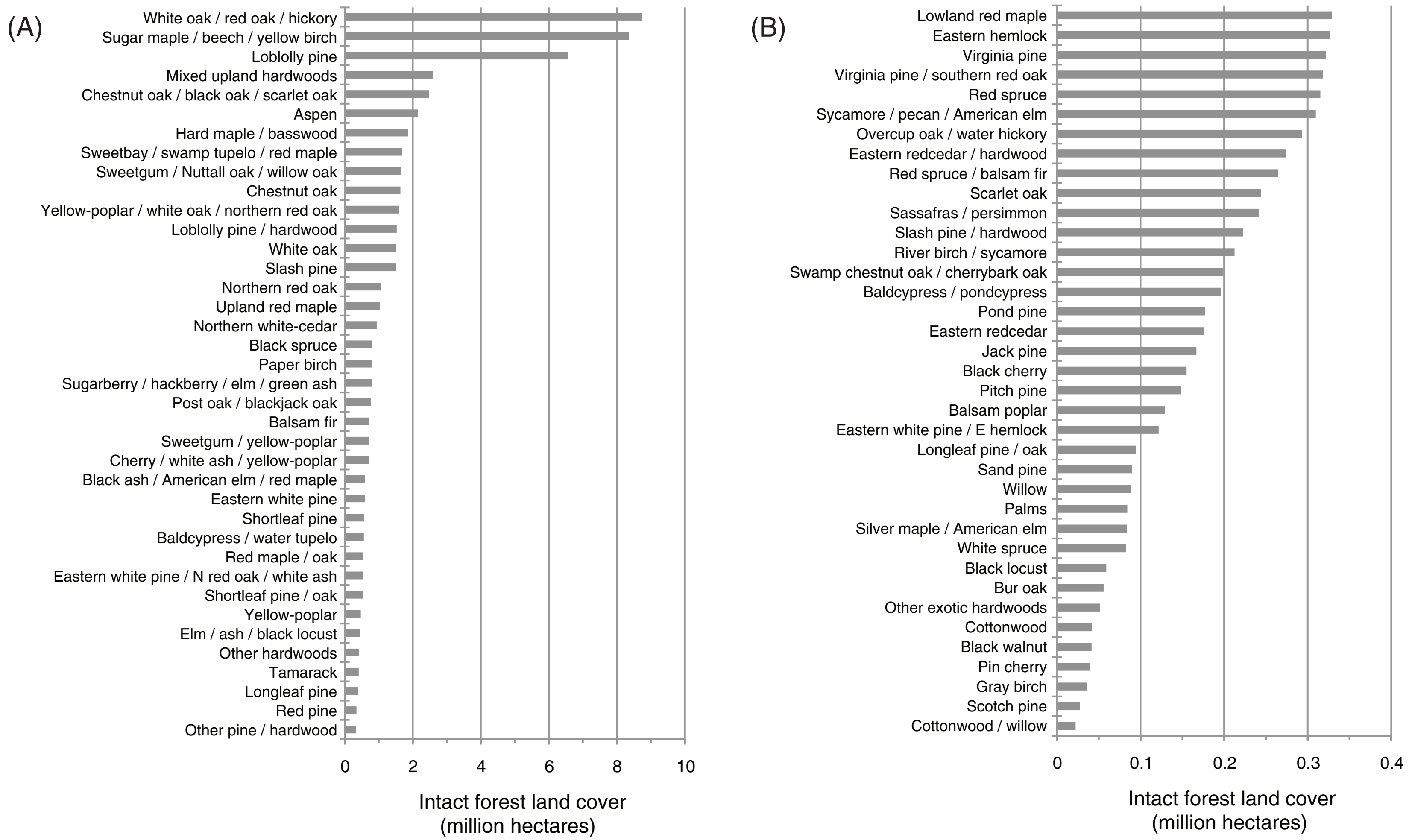

Figure 6.3—Estimated total area of intact forest landcover, by forest type, sorted descending by area. Note the scale change between (A) and (B). Forest type nomenclature is from appendix F of USDA Forest Service (2010).

exhibited moderate to low percentages of intact forest but that occupied a large share of total forest area. Approximately 36 percent of intact forest area was concentrated in only three forest types—white oak/red oak/hickory, sugar maple/beech/yellow birch, and loblolly pine—and the 37 forest types with the least individual intact areas together comprised only 9 percent of total intact forest area. Mitigation of fragmentation and conservation of intact forest may be desired to improve the sustainability of ecological services obtained from specific forest types. If so, land management plans should be specifically directed at those types because plans aimed generally at conserving intact forest would be directed disproportionally to the most common forest types.

In summary, previous national assessments of forest fragmentation did not account for potential differences among forest types because the landcover maps which portray fragmentation did not identify forest types (USDA Forest Service 2001, 2004). This section demonstrated an approach to estimating the degree and area of fragmentation by forest type by combining landcover maps with field inventory data. The statistical features of the field inventory system permit forest types to be compared in terms of the fragmentation that they experience, and permit estimation of fragmented landcover area in a way that is consistent with national forest inventory. In principle, fragmentation data may be summarized by other plot attributes such as ownership by using the methods demonstrated here.

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CHAPTER 7.

Regional Distribution of Introduced Plant Species in the Forests of the Northeastern United States

Beth Schulz
W. Keith Moser
Cassandra Olson
Katherine Johnson

INTRODUCTION

Many plant species have been introduced to the United States by humans since European settlement, sometimes deliberately and sometimes inadvertently, such as in contaminated crop seed or soil. Some species have successfully escaped cultivation and become invasive, spreading and establishing new populations distant from original population centers. Indeed, introduced plant species have forever changed the vegetative landscape of North America.

Not every plant that arrives on the scene becomes established and not every established plant becomes a problem invasive. A specific pattern of site and timing is generally needed for an exotic to take hold in an ecosystem. However, while many introduced plants do not exhibit invasive qualities for long periods after introduction, some reach a point of naturalization when they become invasive where they had previously been benign (Mack 2003). Once established, invasive plants can threaten the sustainability of native forest community composition, structure, function, and resource productivity (Webster and others 2006). Native forest ecosystems that developed over centuries were (and are) limited in their ability to compete against these invaders.

There is an economic cost attributable to the control or management of invasive plants in forest ecosystems. Some authors have put the cost nationwide of all invasive species in the billions (Pimentel and others 2005); certainly the cost to Upper Midwest and Northeastern forests is substantial.

Today, introduced plants are expanding their distributions across this region. These plants occur in all the major life forms found in forest ecosystems: trees, shrubs, vines, herbs/forbs, and grasses. As forests are more and more impacted by fragmentation and other forest health stressors, they become more susceptible to trans-regional and trans-national plant invasion, often at the expense of the indigenous species. Generally, pathways that contribute to the spread of introduced plants, contribute to the spread of more than one species or life form.

Fragmentation is a process of site disturbance whereby intact pieces of forest land are broken up either by active human-influenced processes, like roads and urban development, or by parcelization of ownerships, which introduces more subtle, but still significant, management changes. Fragmentation is important because it is generally recognized that introduced species are more common on forests edges than in the interior of undisturbed forests (Kuhman and others 2010, Moser and others 2009, Vilà and Ibàñez 2011).

Yet, other factors can influence the pace and impact of plant invasions in forests. Some have linked the number of introduced species to overall species richness (Stohlgren and others 1999). Others have shown that absolute or temporal availability of resources is important; invasive species are known to thrive on higher productivity sites (Richardson and Pyšek 2006). Spatial scale is important when considering basic predictors of where introduced species are likely to be found (Kuhman and others 2010, Stohlgren and others 1999).

In addition to local surveys and studies, a regional perspective is central to understanding the factors influencing introduced plant distribution. A regional perspective may assist land managers tasked with minimizing the spread of non-native plants by helping them to prioritize the use of limited resources. One goal of this report is to examine factors important in determining the regional distribution of invasive plants in the upper Midwestern and Northeastern United States.

The Forest Health Vegetation Indicator (VEG) species data include a census of all vascular plants on a subset of the plots maintained by the Forest inventory and Analysis (FIA) Program of the Forest Service, U.S. Department of Agriculture, and are appropriate for regional- or national-scale reporting (Schulz and others 2009). Forest Health Indicators are collected on a one-sixteenth subset (phase 3) of FIA phase 2 plots, or about one plot to every 96,000 acres (Bechtold and Patterson 2005). VEG data have been collected discontinuously since 2001; the FIA unit managed by the Northern Research Station of the Forest Service has collected VEG data more consistently across broader areas than other FIA regions. The data can be used to examine introduced species as a group and by growth habits in addition to measurements of individual species distribution. Overall occupancy of nonnative plants in forests can be estimated as percentage and relative cover of introduced species, as suggested by Noss (1999) and anticipated by the Heinz Center (2006). Ecological provinces are defined by climatic, broad vegetation classes (Cleland and others 2005) and are useful for distinguishing populations at regional scales. They are especially well-suited for reporting forest health indicator results because they are large enough to encompass the sparse FIA phase 3 grid to provide adequate sample sizes, while designating areas that provide similar climatic influences on vegetation.

Our objective is to examine the presence and abundance of introduced species across the forests of the Northeastern United States to determine what broad-scale factors can be used to predict their distribution. Specifically, we look at introduced species distribution over the entire region that falls under the purview of the Northern Research Station FIA unit, by a coarse measure of forest fragmentation (forest intactness), using ecoregion provinces as subpopulations. We examine introduced species as a group, by growth habits, and a selected list of individual species.

METHODS

The Northern Research Station FIA unit collects forest-related data throughout a 24-State region in the Northeastern United States. Standard forest inventory data were collected on phase 2 plots; additional variables related to forest health were collected on phase 3 plots (Bechtold and Patterson 2005), including VEG.

All vascular plants rooted in or hanging over the four subplots (chapter 1, fig. 1.2) were identified. Plant identifications were recorded using plant symbols defined by the Natural Resource Conservation Service (NRCS) PLANTS database (USDA NRCS 2000). For each species on the subplot, total percent canopy cover was estimated and recorded. Species rooted in or overhanging each of three permanently positioned 1m^2 quadrats on each subplot were also recorded. Unknown species were collected near the plot and identified later by an FIA vegetation specialist or submitted to a qualified herbarium.

Each phase 3 plot is also a phase 2 plot. All phase 2 data were available for each plot. The phase 2 data included detailed tree and forest stand data, along with physical site information. We examined initial data from 1,305 plot visits where vegetation data were collected; this represented about three-fifths of the total phase 3 grid for the region.

The FIA sampling design was focused on accessible forested lands; this resulted in some plots with less area sampled than the four full subplots, i.e., some portion of subplot area was non-forested. These plots provided valuable information, but plot summaries and population estimations must be calculated and presented appropriately. Calculations for attributes that are dependent on fixed area measurements exclude sample units that were not 100 percent within accessible forest lands.

Introduced species were designated using NRCS PLANTS database and refined with local knowledge. As the distribution of introduced species was evaluated, it is important to note that many plants observed were never identified to species due to their phenological stage at the time of plot visits. We assumed that the proportion of introduced species among the unidentified plants to be similar to their proportion of all plants identified to species.

For each plot, species richness and the number of introduced species were compiled. We then calculated the percentage of number of introduced species and relative cover of introduced species. The percentage of number of introduced species is simply the sum of introduced species divided by the number of all species identified to species per plot, multiplied by 100. The relative cover of introduced species is the sum of subplot cover of all introduced species divided by the sum of subplot cover by all taxa (species, genera, or unidentified plants) for each plot. Estimates and variances for population level summaries were computed using methods described in Schulz and others (2009) and results were compiled for each ecological province with at least 20 intact plots. The student's *t*-test was used to test for significant differences.

Condition type was derived from FIA phase 2 condition classifications, as a coarse measure of intactness. Conditions were designated by virtue of the following criteria: forest type, stand-size class, land use, regeneration status, reserved status, ownership, and tree density (Bechtold and Patterson 2005). Each plot was designated to one of three condition types based on the number and types of condition classes assigned. If the plot was 100 percent forest and was determined to be a single condition, it was designated as an "intact" stand. Plots that were 100 percent forest but had more than one condition assigned, were designated as a "multiple condition." Plots that were less than 100 percent forest were designated as "forest edge."

Plants identified to species were assigned growth habits based their primary designations in the NRCS PLANTS database, and then compiled into four basic forms: forbs, graminoids (grass-like), shrubs, and trees. Species designated as herbaceous vines were included as forbs, species designated as subshrubs and woody vines were included as shrubs. The chi-square test of independence was used to determine if the categories "origin" and "growth habit" were independent; that is, if distribution of introduced species by growth habits was the same as the distribution of native species by growth habit within the same ecological province.

Individual species selected for distribution analysis were chosen for several reasons; all were among the most common species encountered, some were listed as species of concern for phase 2 plot sampling, and some were so naturalized that many people do not recognize them as nonnative species. We also included species from a variety of growth habits and geographic ranges.

RESULTS

A total of 2,570 taxa (unique species, genera, and unknown codes) were recorded, with 2,210 identified to species. Of the 2,210 species, 303 were considered to be introduced in the NRCS PLANTS database. We included two additional grass species, reed canary grass (*Phalaris arundinacea* L.) and common reed (*Phragmites australis* (Cav.) Trin. ex Steud.) with invasive populations that are of concern in the region as introduced species, bringing the total number of species considered as introduced to 305. The appendix to this chapter lists the introduced species in order of highest constancy (percentage of plots where observed). Of the 1,302 plots included in analysis, 864, or about 66 percent, had at least one introduced species present.

The northeastern corner of the United States, where the Northern Research Station FIA unit conducts forest inventory, encompasses, in total or in part, 14 ecoregion provinces.

Forest Intactness

Plots were summarized by condition type to compare the occupancy of introduced species to the level of forest stand intactness. Plots located on the forest edges have the greatest percentage of plots with introduced species. Compared to the 66.4 percent of all 1,302 plots, 58.75 percent of the 720 intact forest plots, 68 percent of the 120 multiple condition plots, and 77.7 percent of the 462 plots with some non-forest had at least one introduced species. On the 864 plots where at least one introduced species was recorded, the percentage of identified species that are introduced is least in intact stands and greatest on plots with some non-forest. This same trend is observed for the relative cover of introduced species (fig. 7.1). Each condition type is significantly different from the others for both measures ($\alpha < 0.05$).

Populations Defined by Ecoregion Provinces

The 14 ecoregion provinces are listed in table 7.1, along with the percentage of plots with at least one introduced species (PPWI)

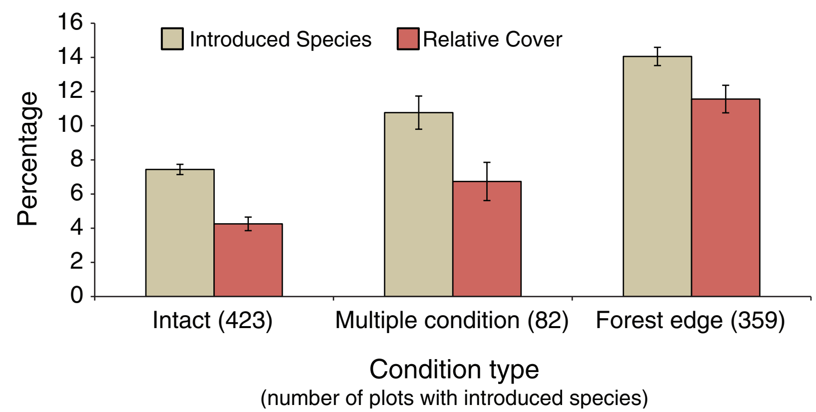

Figure 7.1—Average percentage of introduced species and relative cover by condition type for plots with introduced species. Error bars represent "plus one" and "minus one" standard error.

Table 7.1—Proportion of plots with at least one introduced species (proportion of plots with introduced, or PPWI) and the number of plots in each condition type by ecological province

Ecological province		PPWI	Condition type		
			Intact	Multiple condition	Forest edge
Code	*Name*	*Percentage*	*Number of plots*		
211	Northeastern Mixed Forest	62.7	80	12	34
M211	Adirondack-New England Mixed Forest – Coniferous Forest – Alpine Meadow	41.6	65	12	12
212	Laurentian Mixed Forest	45.5	207	40	76
221	Eastern Broadleaf Forest	87.5	85	18	65
M221	Central Appalachian Broadleaf Forest – Coniferous Forest – Meadow	57.7	50	3	18
222	Midwest Broadleaf Forest	87.2	49	11	81
223	Central Interior Broadleaf Forest	70.2	112	14	65
231	Southeastern Mixed Forest	80	5	0	0
232	Outer Coastal Plain Mixed Forest	64.6	28	4	33
251	Prairie Parkland (Temperate)	85.5	24	4	55
255	Prairie Parkland (Subtropical)	100	3	1	0
331	Great Plains – Palouse Dry Steppe	100	3	0	11
332	Great Plains Steppe	100	5	1	9
M334	Black Hills Coniferous Forest	85.7	4	0	3

and number of plots in each condition type. The five ecoregion provinces with fewer than 20 intact plots are excluded from analyses that compare plot species richness to PPWI. Average species richness at the quadrat, subplot, and plot level (fig. 7.2) and occupancy by introduced species (fig. 7.3) varied across the nine ecoregion provinces.

The values of PPWI for each ecoregion province with more than 20 intact plots were strongly related with the proportion of forest edge plots (fig. 7.4A) and with the average plot species richness (fig. 7.4B). At the broad regional scale, the proportion of forest edge plots explains a greater proportion of variation of PPWI than average species richness.

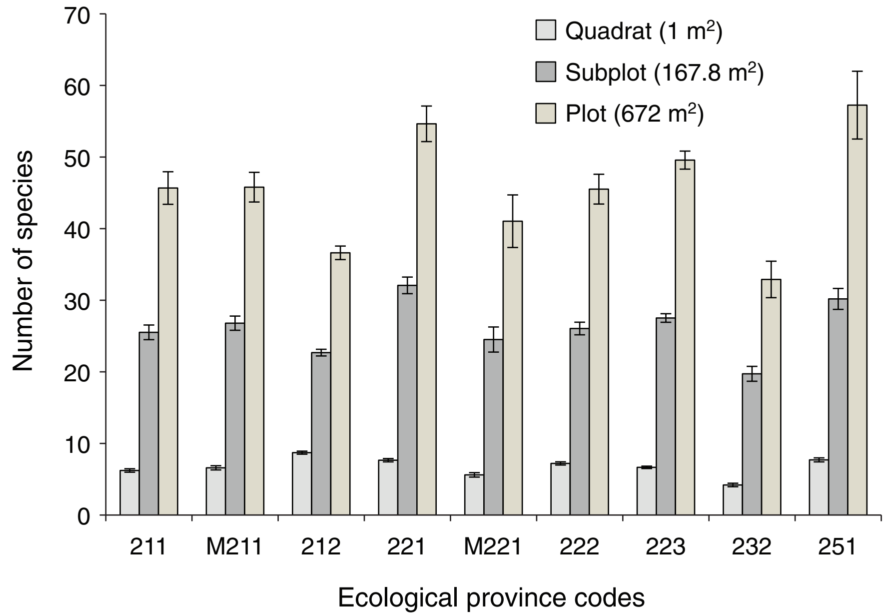

Figure 7.2—Average species richness for quadrats, subplots, and plots of 100-percent forested by ecoregion province. Error bars represent "plus one" and "minus one" standard error.

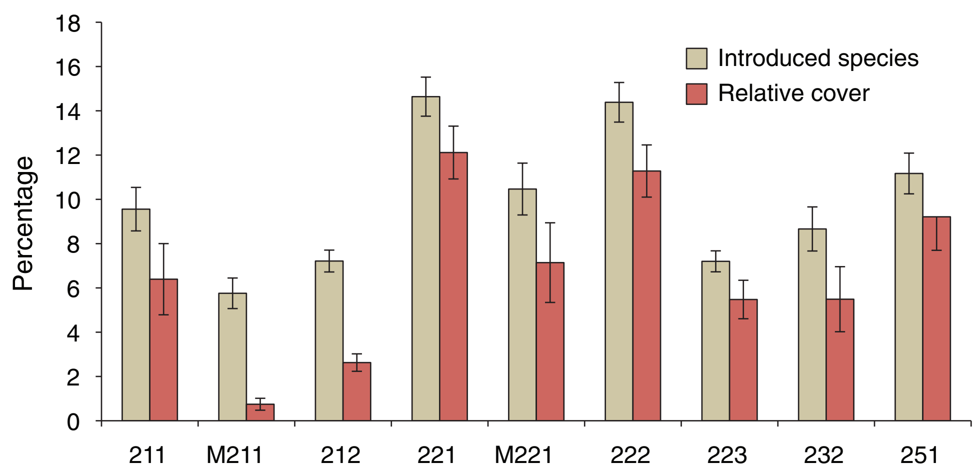

Figure 7.3—Occupancy of introduced species by ecological province, expressed as a percentage of total species richness and cover. Error bars represent "plus one" and "minus one" standard error.

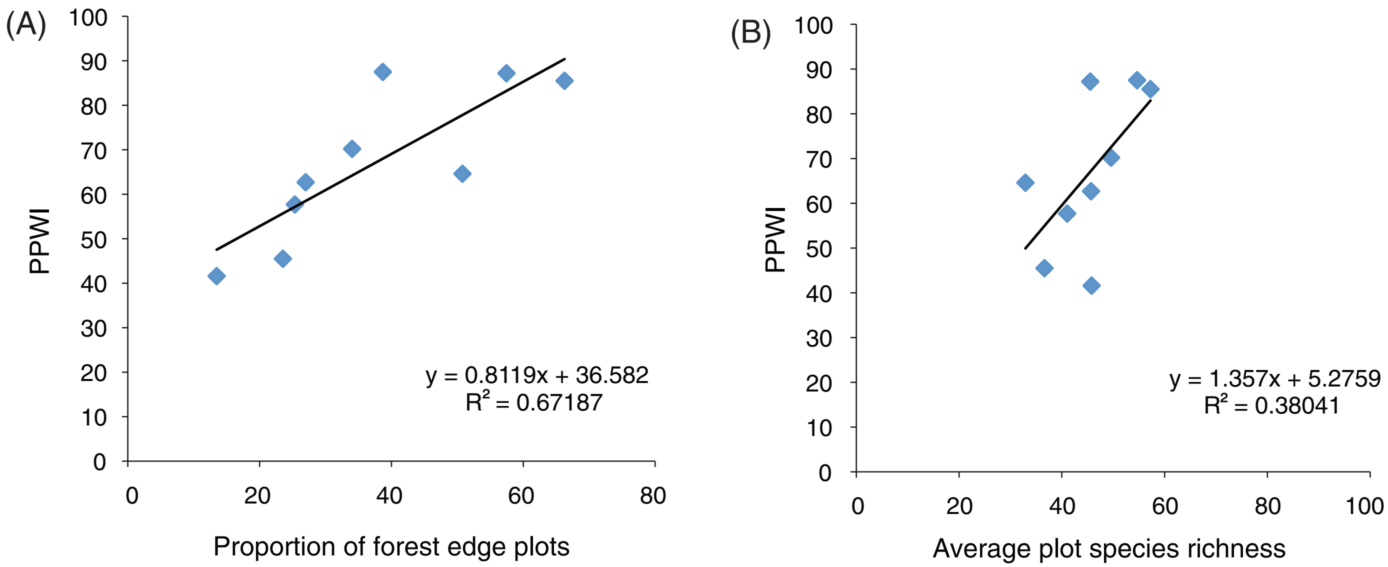

Figure 7.4—Relationship between proportion of plots with introduced species (PPWI) and proportion of forest edge plots (A) and average plot species richness (B) across ecological provinces.

Average plot species richness compilations do not include forest edge plots because species richness is related to area sampled, and forest edge plots are not fully forested. However, percentage of introduced species and relative cover are not area-sensitive metrics, unlike direct assessments of species richness, so all plots within each ecoregion province are included. Each plot does have at least one fully forested subplot, however. The relationship between subplot species richness and PPWI was not as strong (r^2 = 0.32), as that for plot species richness and PPWI.

The ecoregion provinces also varied from one another by the most common introduced species recorded (table 7.2). The constancies for this short list of species drops quickly in those ecoregion provinces with lower introduced species occupancy measures (M211, 212).

Table 7.2—The five most common introduced species per ecological province

Ecological province (number of plots) Scientific name	Common name	Growth habit	Constancy
211-Northeastern Mixed Forest (n = 126)			
Rosa multiflora	Multiflora rose	Vine, Shrub	11.90
Anthoxanthum odoratum	Sweet vernalgrass	Graminoid	10.32
Epipactis helleborine	Helleborine	Forb/herb	10.32
Lonicera morrowii	Morrow's honeysuckle	Shrub	9.52
Hieracium caespitosum	Meadow hawkweed	Forb/herb	7.94
M211-Adirondack-New England Mixed Forest – Coniferous Forest – Alpine Meadow (n = 89)			
Epipactis helleborine	Helleborine	Forb/herb	12.36
Phleum pratense	Timothy	Graminoid	6.74
Leucanthemum vulgare	Oxeye daisy	Forb/herb	5.62
Vicia cracca	Bird vetch	Vine, Forb/herb	5.62
Trifolium aureum	Golden clover	Forb/herb	4.49
212-Laurentian Mixed Forest (n = 323)			
Hieracium aurantiacum	Orange hawkweed	Forb/herb	10.84
Rumex acetosella	Common sheep sorrel	Forb/herb	4.64
Polygonum convolvulus	Black bindweed	Vine, Forb/herb	4.33
Solanum dulcamara	Climbing nightshade	Forb/herb	4.33
Phleum pratense	Timothy	Graminoid	3.72
221-Eastern Broadleaf Forest (n = 168)			
Rosa multiflora	Multiflora rose	Vine, Shrub	70.24
Lonicera japonica	Japanese honeysuckle	Vine	23.21
Berberis thunbergii	Japanese barberry	Shrub	19.05
Polygonum persicaria	Spotted ladysthumb	Forb/herb	18.45
Alliaria petiolata	Garlic mustard	Forb/herb	17.86
M221-Central Appalachian Broadleaf Forest – Coniferous Forest – Meadow (n = 71)			
Rosa multiflora	Multiflora rose	Vine, Shrub	29.58
Microstegium vimineum	Nepalese browntop	Graminoid	15.49
Elaeagnus umbellata	Autumn olive	Shrub	15.49
Berberis thunbergii	Japanese barberry	Shrub	12.68
Alliaria petiolata	Garlic mustard	Forb/herb	12.68

continued

Table 7.2 (continued)—The five most common introduced species per ecological province

Ecological province (number of plots) Scientific name	Common name	Growth habit	Constancy
222-Midwest Broadleaf Forest (n = 141)			
Rosa multiflora	Multiflora rose	Vine, Shrub	45.39
Alliaria petiolata	Garlic mustard	Forb/herb	24.11
Glechoma hederacea	Ground ivy	Forb/herb	15.60
Phalaris arundinacea	Reed canarygrass	Graminoid	13.48
Phleum pratense	Timothy	Graminoid	12.06
223-Central Interior Broadleaf Forest (n = 191)			
Rosa multiflora	Multiflora rose	Vine, Shrub	41.36
Lonicera japonica	Japanese honeysuckle	Vine	19.89
Daucus carota	Queen Anne's lace	Forb/herb	10.99
Elaeagnus umbellata	Autumn olive	Shrub	5.76
Lolium pratense	Meadow ryegrass	Graminoid	5.24
231-Southeastern Mixed Forest (n = 5)			
Lonicera japonica	Japanese honeysuckle	Vine	60.00
Microstegium vimineum	Nepalese browntop	Graminoid	60.00
Rosa multiflora	Multiflora rose	Vine, Shrub	40.00
Rhamnus cathartica	Common buckthorn	Tree, Shrub	20.00
Commelina communis	Asiatic dayflower	Forb/herb	20.00
232-Outer Coastal Plain Mixed Forest (n = 65)			
Lonicera japonica	Japanese honeysuckle	Vine	47.69
Rosa multiflora	Multiflora rose	Vine, Shrub	15.38
Polygonum hydropiper	Marshpepper knotweed	Forb/herb	6.15
Hypericum perforatum	Common St. Johnswort	Forb/herb	4.62
Microstegium vimineum	Nepalese browntop	Graminoid	4.62
251-Prairie Parkland (Temperate) (n = 83)			
Rosa multiflora	Multiflora rose	Vine, Shrub	46.99
Polygonum convolvulus	Black bindweed	Vine, Forb/herb	16.87
Phalaris arundinacea	Reed canarygrass	Graminoid	15.66
Morus alba	White mulberry	Tree, Shrub	15.66
Lolium arundinaceum	Tall fescue	Graminoid	14.46

continued

Table 7.2 (continued)—The five most common introduced species per ecological province

Ecological province (number of plots) Scientific name	Common name	Growth habit	Constancy
255-Prairie Parkland (Subtropical) (n = 4)			
Rosa multiflora	Multiflora rose	Vine, Shrub	50.00
Morus alba	White mulberry	Tree, Shrub	50.00
Torilis arvensis	Spreading hedgeparsley	Forb/herb	50.00
Lespedeza cuneata	Chinese lespedeza	Subshrub, Shrub, Forb	50.00
Lolium arundinaceum	Tall fescue	Graminoid	25.00
331-Great Plains – Palouse Dry Steppe (n = 14)			
Tragopogon dubius	Yellow salsify	Forb/herb	35.71
Melilotus officinalis	Yellow sweetclover	Forb/herb	28.57
Bromus japonicus	Japanese brome	Graminoid	28.57
Nepeta cataria	Catnip	Forb/herb	21.43
Lactuca serriola	Prickly lettuce	Forb/herb	21.43
332-Great Plains Steppe (n = 15)			
Morus alba	White mulberry	Tree, Shrub	40.00
Medicago lupulina	Black medick	Forb/herb	26.67
Melilotus officinalis	Yellow sweetclover	Forb/herb	20.00
Trifolium repens	White clover	Forb/herb	20.00
Verbascum thapsus	Common mullein	Forb/herb	20.00
M334-Black Hills Coniferous Forest (n = 7)			
Poa compressa	Canada bluegrass	Graminoid	28.57
Cirsium arvense	Canada thistle	Forb/herb	28.57
Artemisia absinthium	Absinthium	Subshrub, Shrub, Forb	28.57
Agropyron cristatum	Crested wheatgrass	Graminoid	14.29
Tragopogon dubius	Yellow salsify	Forb/herb	14.29

Growth Habits of Introduced Species by Ecoregion Provinces

Examination of the distribution of introduced species by growth habits across ecoregion provinces reveals some interesting trends (table 7.3). Overall, forbs made up the largest proportion of both native and introduced species, ranging from about 45 percent of native species in ecoregion province 232 to over 72 percent of introduced species in ecoregion province 212. The proportion of graminoids (grass and grass-like plants) ranged from 10 percent for introduced species in ecoregion province 212 to about 24 percent for introduced species in ecoregion province 251. Proportion of shrubs ranged from 9 percent for introduced species in ecoregion province M211 to 30 percent for introduced species in ecoregion province 232. Tree species made up less than 5 percent of introduced species in ecoregion provinces 232 and 251, but accounted for over 23 percent of native species in ecoregion province 232.

Results of the chi-square test for independence show that in over half of the ecoregion provinces, the distribution of native and introduced species by growth habit were significantly different. The greatest difference was in ecoregion province 212, and there were no significant differences ($\alpha \geq 0.05$) in ecoregion provinces M211, 221, 232, and 251.

Table 7.3—Percentage of native and introduced species by growth habit and results of chi-squared test of independence (degrees of freedom =3) to determine if species origin and growth habits were independent within each ecological province

Ecological province	Species origin	Growth habit				Chi-square	Significance level (alpha)
		Forb	Graminoid	Shrub	Tree		
		- - - - - - - - - - - - - *Percentage*			- - - - - - - - - - -		
211	Native	46.86	20.93	15.14	17.07		
211	Introduced	59.81	14.02	16.82	9.35	8.9	0.030
M211	Native	46.21	19.19	17.30	17.30		
M211	Introduced	63.64	20.45	9.09	6.82	6.7	0.076
212	Native	47.00	19.07	19.62	14.31		
212	Introduced	72.73	10.00	11.82	5.45	25.1	> 0.001
221	Native	49.49	17.75	14.86	17.89		
221	Introduced	54.74	16.79	17.52	10.95	4.5	0.212
M221	Native	52.21	13.97	13.24	20.59		
M221	Introduced	50.91	16.36	25.45	7.27	9.7	0.022
222	Native	52.72	14.43	15.91	16.94		
222	Introduced	64.79	14.79	11.97	8.45	9.8	0.020
223	Native	58.01	12.78	13.34	15.87		
223	Introduced	60.67	19.10	12.36	7.87	5.9	0.117
232	Native	44.98	14.53	16.96	23.53		
232	Introduced	47.62	16.67	30.95	4.76	10.1	0.018
251	Native	55.18	17.93	12.30	14.59		
251	Introduced	61.90	23.81	9.52	4.76	7.7	0.052

Note: Significant differences are greater for larger chi-square values and indicate that species origin and growth habit are dependent, i.e., native and introduced species have different distributions across growth habits.

Species Distribution by Condition Type

We plotted the constancy of the 23 selected species in each condition type to see if they followed the trend of being more commonly recorded on forest edge plots (fig 7.5). Most species do follow this trend, but several do not. Note how prevalent multiflora rose (*Rosa multiflora*) is across the region.

Regional Distribution of Selected Species

Figures 7.6-7.10 display the regional distribution of selected forbs, grasses, woody shrubs and vines, and trees. Background shadings on maps represent ecoregion provinces (Cleland and others 2005). Some species are widespread throughout the northeastern forests, while others are concentrated in particular ecoregion provinces (table 7.4).

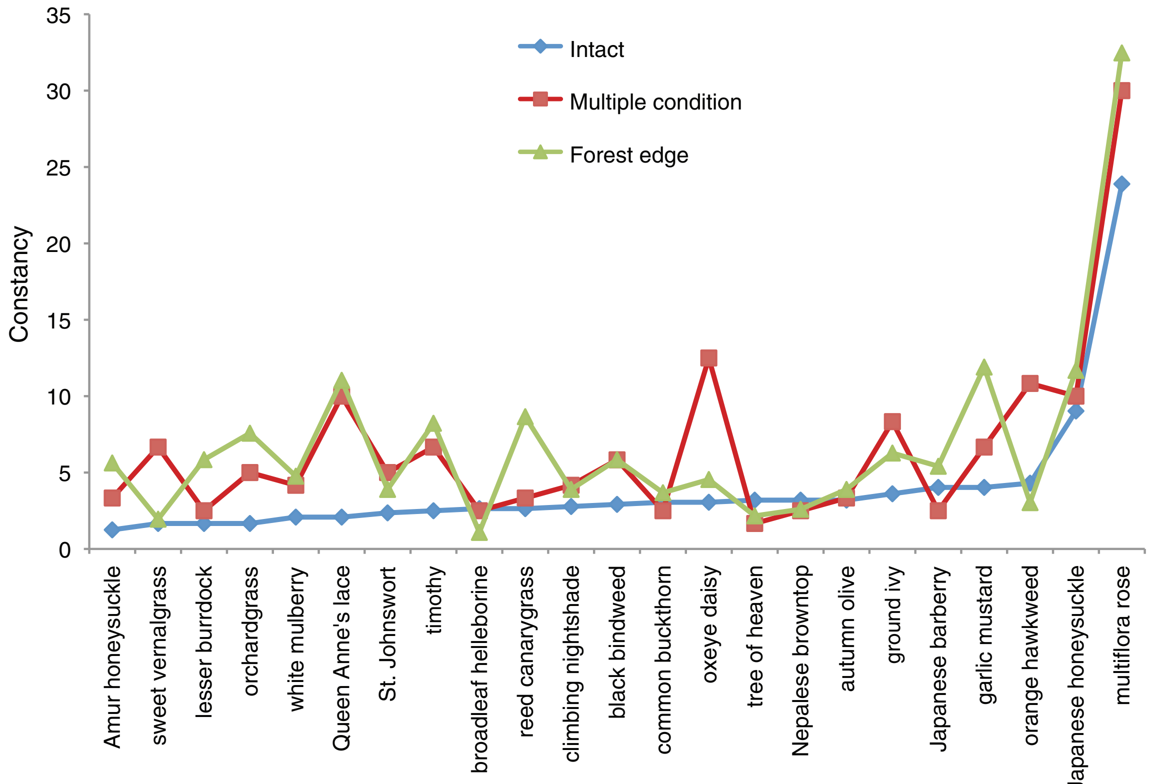

Figure 7.5—Constancy (proportion of plots where recorded) of selected species for condition types, intact forest, multiple condition, and forest edge.

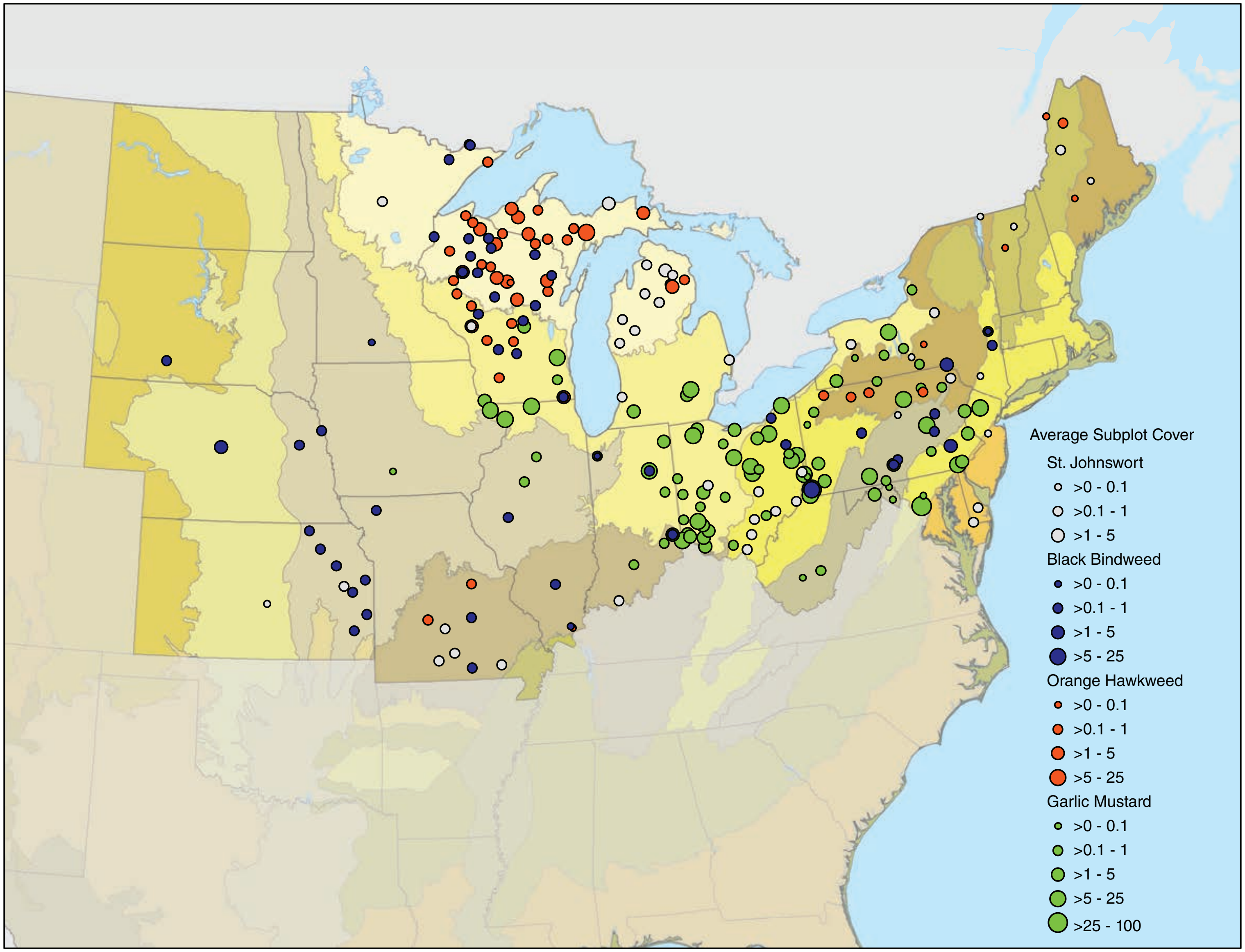

Figure 7.6—Distribution and average subplot cover of four selected forb species: St. Johnswort, black bindweed, orange hawkweed, and garlic mustard. Plot locations are approximate.

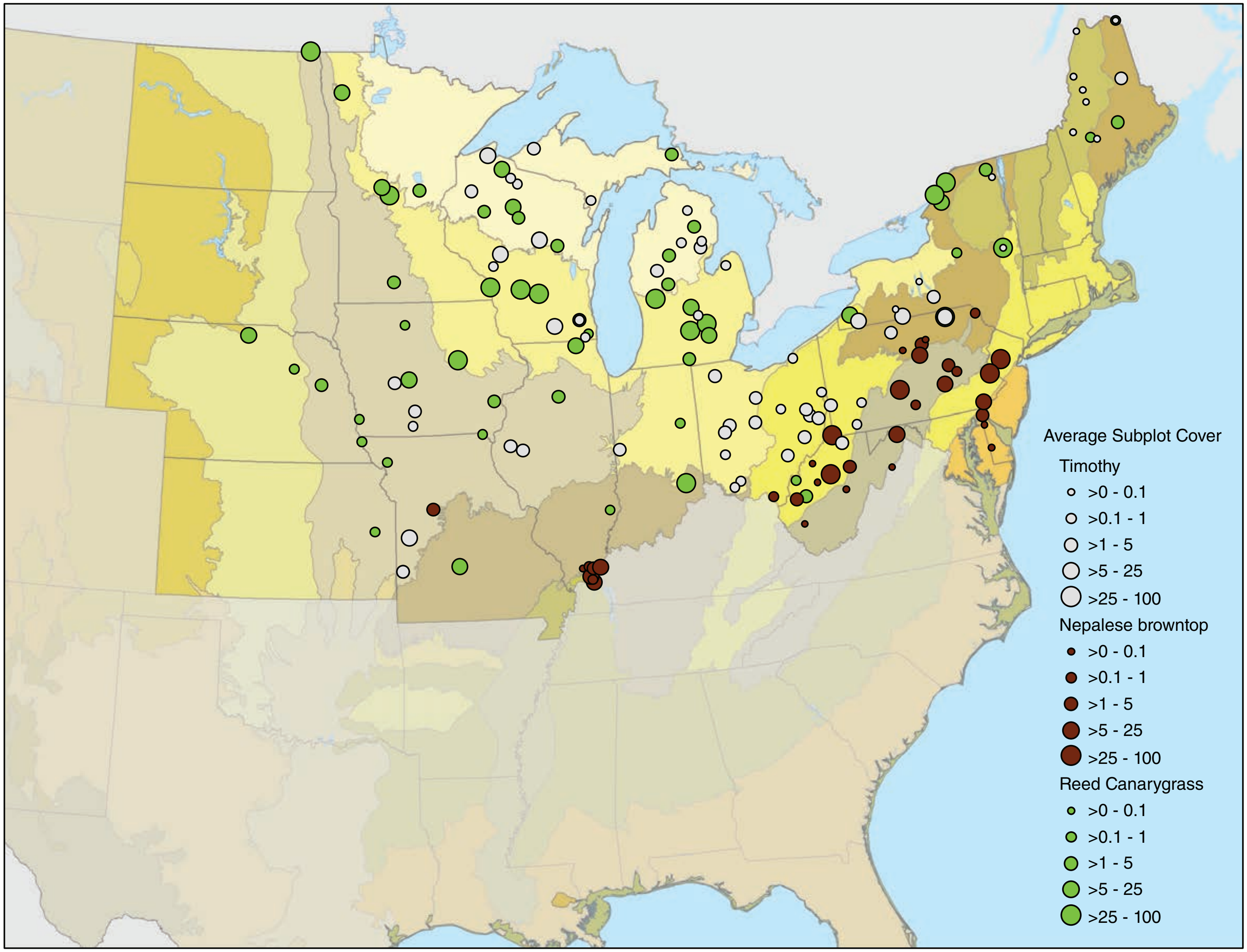

Figure 7.7—Distribution and average subplot cover of three selected grass species: timothy grass, Nepalese browntop, and reed canarygrass. Plot locations are approximate.

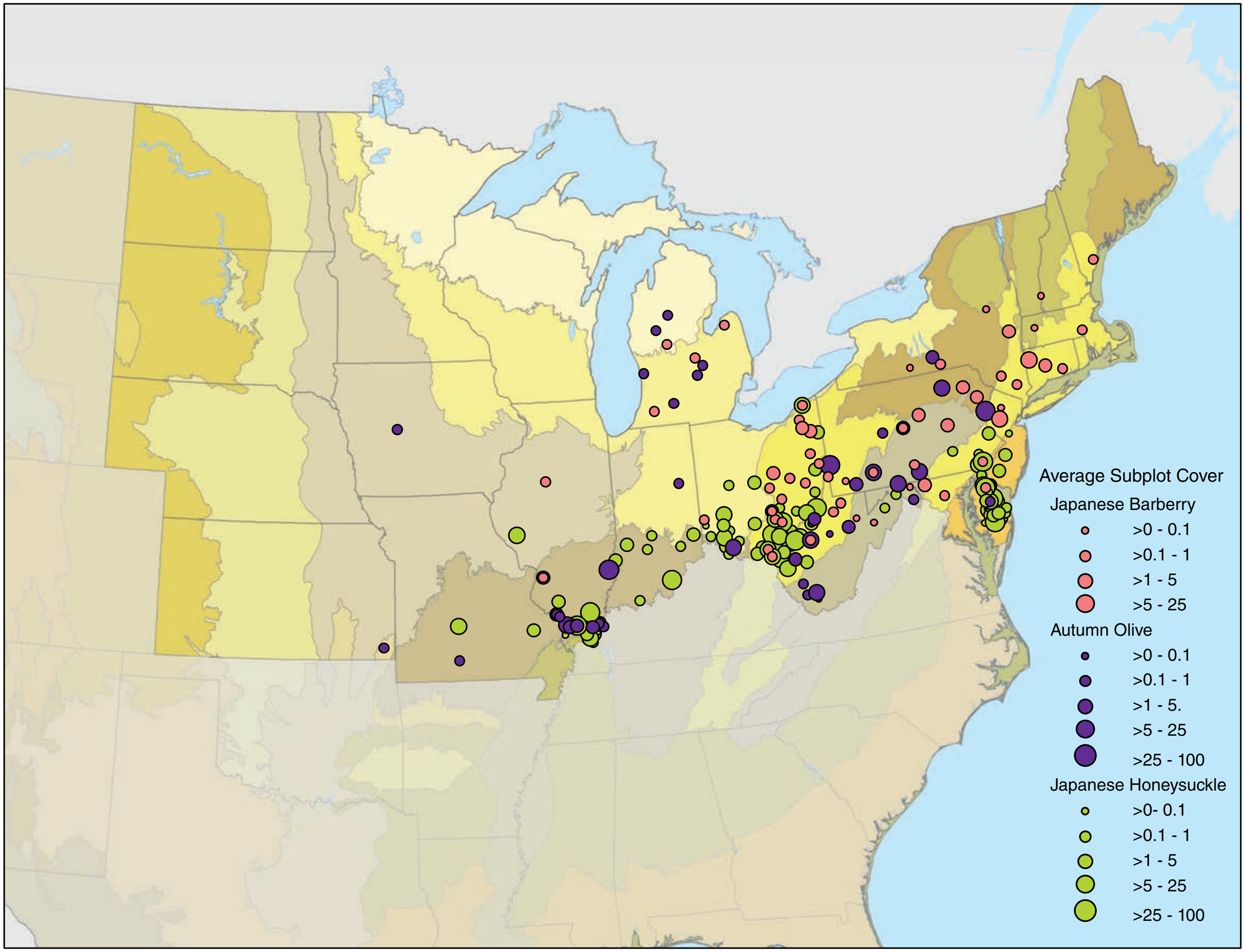

Figure 7.8—Distribution and average subplot cover of three selected introduced non-tree woody plant species: Japanese barberry, autumn olive, and Japanese honeysuckle. Plot locations are approximate.

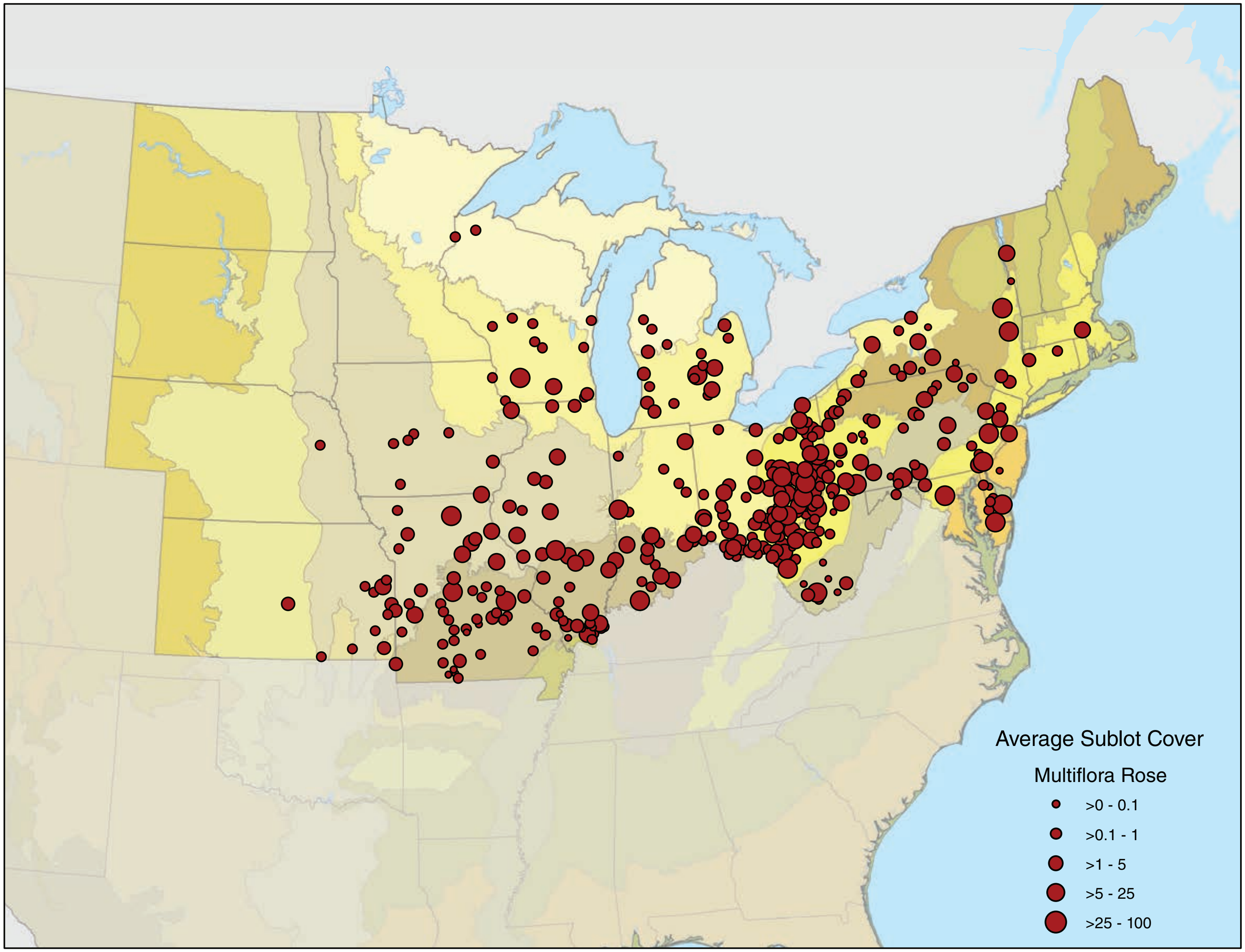

Figure 7.9—Distribution and average subplot cover of multiflora rose, the most commonly reported introduced species in the Region. Plot locations are approximate.

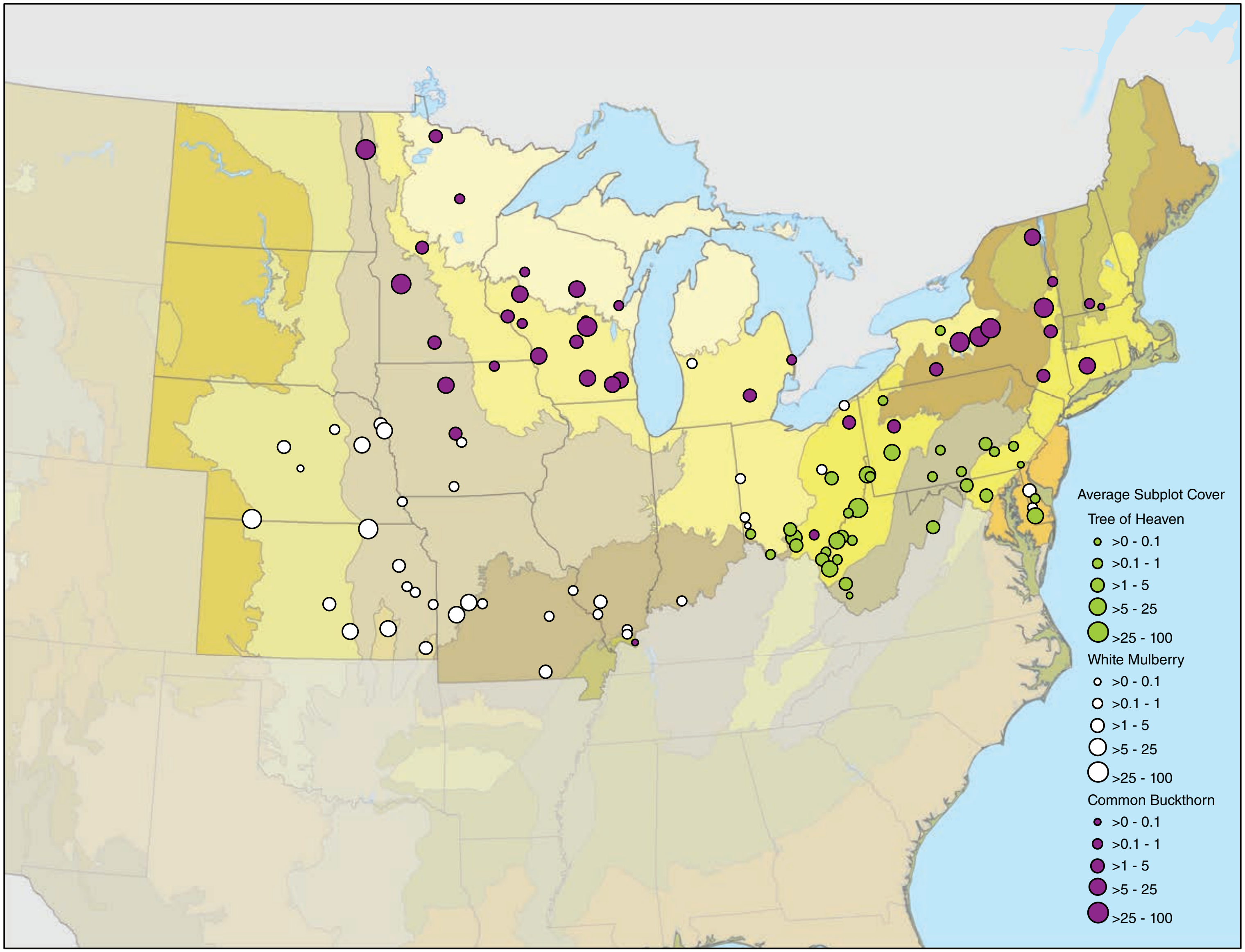

Figure 7.10—Distribution and average subplot cover of three selected introduced tree species: tree of heaven, white mulberry, and common buckthorn. In some areas buckthorn is more of a shrub than a small tree. Plot locations are approximate.

Table 7.4—Constancy of selected introduced species for the region and by ecological province

		Ecological province codes (number of plots)								
	Region	211	M211	212	221	M221	222	223	232	251
Common name	(1302)	(126)	(89)	(323)	(168)	(71)	(141)	(191)	(65)	(83)
	Percentage of plots where species was recorded									
Forbs										
Garlic mustard	7.3	6.35	0.00	0.00	17.86	12.68	24.11	2.62	0.00	7.23
Lesser burdock	3.3	5.56	0.00	0.62	3.57	0.00	8.51	1.57	0.00	9.64
Queen Anne's lace	6.1	3.17	0.00	1.24	11.31	5.63	12.06	10.99	0.00	10.84
Broadleaf helleborine	2.1	10.32	12.36	0.31	0.60	0.00	0.71	0.00	0.00	0.00
Ground ivy	4.99	3.17	0.00	0.62	16.67	2.82	15.60	0.52	1.54	4.82
Orange hawkweed	4.5	3.97	3.37	10.84	1.19	0.00	7.09	1.57	0.00	0.00
St. Johnswort	3.15	3.17	2.25	3.10	4.76	1.41	4.26	2.62	4.62	1.20
Oxeye daisy	4.6	4.76	5.62	3.10	11.31	4.23	5.67	1.57	1.54	3.61
Climbing nightshade	3.38	7.14	1.12	4.33	1.19	0.00	11.35	0.52	0.00	0.00
Black bindweed	4.22	0.79	0.00	4.33	4.76	5.63	4.96	2.62	0.00	16.87
Grasses										
Sweet vernalgrass	2.23	10.32	0.00	0.00	7.14	2.82	0.71	0.00	1.54	0.00
Orchardgrass	4.15	3.17	0.00	0.62	11.90	5.63	7.09	2.09	0.00	9.64
Nepalese browntop	3.0	3.17	0.00	0.00	5.36	15.49	0.71	3.14	4.62	1.20
Reed canarygrass	4.99	5.56	1.12	3.41	4.17	0.00	13.48	1.57	0.00	15.66
Timothy	4.9	6.35	6.74	3.72	8.33	0.00	12.06	0.00	0.00	8.43
Shrubs										
Japanese barberry	4.38	3.97	2.25	0.31	19.05	12.68	3.55	0.52	1.54	1.20
Autumn olive	3.5	1.59	0.00	0.62	5.95	15.49	4.26	5.76	1.54	1.20
Amur honeysuckle	3.0	0.79	0.00	0.00	2.38	1.41	9.22	5.24	3.08	8.43
Vines										
Japanese honeysuckle	10.29	0.00	0.00	0.00	23.21	7.04	9.93	19.90	47.69	1.20
Multiflora rose	27.65	11.90	0.00	2.17	70.24	29.58	45.39	41.36	15.38	46.99
Trees										
Tree of heaven	2.76	0.00	0.00	0.00	12.50	11.27	2.13	0.52	3.08	0.00
White mulberry	3.3	0.00	0.00	0.00	2.98	0.00	3.55	4.71	3.08	15.66
Common buckthorn	3.23	2.38	3.37	1.55	5.36	0.00	10.64	0.00	0.00	7.23

DISCUSSION

Estimates of occupancy by introduced species by condition type, a coarse measure of forest fragmentation, suggest that introduced species are more abundant on forest edges. An examination of the adjacent non-forest conditions in this data set revealed that the vast majority of non-forest lands on forest edge plots are either developed, agricultural, or range, indicating a close proximity to human activities. This trend was followed at the ecoregion province level; provinces with a higher percentage of forest edge plots had higher occupancy of introduced species. At finer scales, other predictors for introduced species may prove to be more useful. Fortunately, because data are collected and stored at the subplot level, it will be possible to do further analyses.

The ecoregion province summaries showed different rates of introduced species occupancy. Provinces M211, 212, and 223 had the lowest occupancy as measured by percentage of introduced species and relative cover (fig. 7.3); together, these provinces include about 48 percent of the plots in this study. Province 223 did have a high rate of PPWI, however (table 7.1).

The full census of vascular plants on each plot allowed us to examine trends of introduced species in terms of growth habits; surveys limited to short lists of species can only assess those species on the list, and are not likely to give a clear picture of overall trends. In an earlier analysis of invasive species in the upper Midwest (Moser and others 2009), it was speculated that herbaceous plants are less likely to invade northern forests. Using the full species lists, we found that the proportion of introduced forbs is greater than the proportion of native forb species in ecoregion province 212, which encompasses the northern portions of Michigan, Wisconsin, and Minnesota.

Multiflora rose is by far the most common introduced species in the Region, with an overall constancy of 27.65 percent, and as high as 70 percent in province 221. However, it was not recorded in province M211 (table 7.4). Although it probably does occur within the province, multiflora rose was not recorded on any of the 89 forested plots, illustrating that it is much less prominent there. Originally introduced to the United States as root stock for ornamental roses, mutiflora rose was widely promoted, starting in the 1930s, as a natural fence row to contain livestock, and then as a wildlife forage species and crash barrier in highway medians (Swearingen and others 2010). Today it is widespread.

We can look at the distribution of any individual species recorded in the inventory. Although it was not practical to examine every introduced species for this report, we examined the distribution of several species of high interest (figs. 7.5–7.10, table 7.4). The species highlighted in figures 7.6–7.10 show a variety of ranges. Orange hawkweed (*Hieracium aurantiacum*) was most concentrated in ecoregion province 212 where intact forest plots are more common than forest edge plots (fig. 7.6 and table 7.1). Timothy

(*Phleum pretense*), a grass so common it is often mistaken for a native species, was widespread, while Nepalese browntop (*Microstegium vimineum*), was limited to the southern portion of the Northern region.

Most of the selected species followed the trend of being recorded more often in forest edge or multiple forest condition plots but a few did not (fig. 7.5). Shade-tolerant species are troubling because they can survive in closed canopy forests, potentially far from traffic corridors where they may have originally been introduced.

Tree of heaven (*Ailianthus altissima*) is a short-lived, pollution tolerant tree. It grows fast, up to one to two meters per year in its first few years. Although it grows best in full sunlight, it is able to take advantage of gaps in the forest canopy and quickly fill them (Knapp and Canham 2000). It also produces an allelopathic chemical that inhibits most other nearby plant growth (Mergen 1959). One species that is not affected by this chemical is white ash (*Fraxinus americana*) (Mergen 1959). Indeed, white ash was present on 26 of the 39 plots where tree of heaven was recorded.

Nepalese browntop, also known as Japanese stiltgrass, is problematic in more southern climates, but has been found as far north as Massachusetts and New York. It reproduces vegetatively and by seed and is prevalent along river corridors. Seed dispersal is facilitated along waterways by flooding where spread of seeds increases (Swearingen and others 2010; Warren and others 2011). A recent investigation revealed that undisturbed leaf litter and understory canopy shade can limit the establishment of Nepalese browntop, but once disturbed by moving water or large animals (including humans), sites with normal amounts of leaf litter can become prone to invasion when seeds come in contact with mineral soil (Schramm and Ehrenfeld 2010). In addition, removal of understory by herbivory by deer or silvicultural thinning is likely to facilitate the establishment of this species. Kuhman and others (2010) found that in the Southern Appalachian Mountains, this grass was positively correlated with forest canopy cover, unlike the other species in their study. Twenty six of 39 plots with stiltgrass were intact forest, with 13 different forest types represented. However, nine of those plots were in the white oak/red oak/hickory type. As the floods of 2011 recede, we may see an increase of this invasive grass.

Broadleaf helleborine (*Epipactis helleborine*) was found slightly more often in intact forests than forest edges plots (fig. 7.5). It was relatively common (ranked 34 among 305 species), but concentrated in provinces 211 and M211 (table 7.4). Little information could be found on this particular species. However, Swearingen and others (2010) list it as a plant “to watch” in the mid-Atlantic States as it becomes more widespread in dry, gravely soils in forests and woodland edges. Because VEG data collection included all vascular species, we were able to provide information on the distribution of up-and-coming species of concern.

Common buckthorn and Japanese barberry (*Berberis thunbergii*) are found nearly as often in intact forests as in forest edge or multiple condition plots (fig. 7.5). Buckthorn is a good example of a cultivated plant that survived for many years and then became naturalized and spread into natural areas. While best growth is in full light, it produces an abundance of seed that can germinate in partial light conditions and are borne in berries that are spread by birds (Swearingen and others 2010). Japanese barberry was promoted as an ornamental plant in the late 1800s; it rapidly spread into abandoned agricultural fields and open areas. DeGasperis and Motzkin (2007) studied the current distribution of this species and found it occurred more often in forests that re-established after agriculture abandonment in the early 20th century, after barberry had been introduced. More modern disturbances did not result in additional spread if seed sources were not immediately available and although barberry may be present in areas that were wooded in the early 20th century, it occurs in a smaller proportion of these stands.

One species we expected to see more often on intact plots was garlic mustard (*Alliaria petiolata*), known for its shade tolerance. Although it was wide spread in the southeastern portion of the region (fig. 7.6), it was recorded most often on forest edge plots (fig. 7.5).

It is often preferable to summarize data by ecoregion province and forest type in reports focused on FIA data. In this data set, there are 212 ecoregion province/forest type pairs, 89 of which are represented by 1 plot, and a total of 166 have 5 or fewer plots. There are 27 pairs with at least 10 plots, and 13 of these ecoregion province/forest type pairs were designated as either multiple conditions or forest edge condition types. This data set is only about 60 percent of the total phase 3 grid for the region; more thorough analyses should be conducted with a complete set of FIA phase 3 plots.

CONCLUSION

The FIA phase 3 VEG data allow for estimation of the occupancy of introduced species in terms of percent number of introduced species and relative cover. Results indicate a strong influence of forest fragmentation on the regional distribution of introduced species. Occupancy of introduced species varied across ecoregion provinces; ecoregion provinces with a higher proportion of forest edge plots had the highest occupancy by introduced species.

Although the proportion of introduced species by growth habits was different from the proportion of native species in each growth habit for more than half of the provinces, forb species dominated both native and introduced growth habits in all ecoregion provinces. The two provinces with the lowest occupancy of introduced species (M211 and 212) had higher proportions of introduced forb species compared to their proportion of native forb species.

The distribution of individual species varied across ecoregion provinces and by condition type. Multiflora rose was by far the most common introduced species, but varied in constancy from 0 percent (M211) to 70 percent (221). Of the selected species, most were recorded in forest edge or multiple condition plots, but a few were found more often in intact forest stands. One of the more commonly recorded forb species, broadleaf helleborine, was found more often on intact forest plots, and has only recently become a species to watch for invasive tendencies. We are able to report on the distribution of this species because of the full vascular plant species inventories available from plots where VEG data has been recorded.

Our findings highlight the importance of efforts to manage roadside and trailhead vegetation to minimize the spread of introduced and potentially invasive plant species into intact forests. This region-wide analysis of the distribution of introduced species established in the forests of the Northeastern United States is just a beginning. Further examination of distribution and abundance within each ecoregion province are possible with these data. However, with the additional plot data collected in 2009 and 2010, more ecoregion province/ forest type pairs and some revisited plots will be available, providing for new ways to examine trends and report indications of changing species distributions.

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Appendix 7.1—Introduced species recorded on 1,302 plots in Northeastern United States, in order of number of plots where recorded

Common name	Scientific name	Number of plots
Multiflora rose	*Rosa multiflora* Thunb. ex Murr.	360
Japanese honeysuckle	*Lonicera japonica* Thunb.	134
Garlic mustard	*Alliaria petiolata* (Bieb.) Cavara & Grande	95
Queen Anne's lace	*Daucus carota* L.	80
Reed canarygrass	*Phalaris arundinacea* L.	65
Ground ivy	*Glechoma hederacea* L.	65
Timothy	*Phleum pratense* L.	64
Oxeye daisy	*Leucanthemum vulgare* Lam.	60
Orange hawkweed	*Hieracium aurantiacum* L.	59
Japanese barberry	*Berberis thunbergii* DC.	57
Black bindweed	*Polygonum convolvulus* L.	55
Orchardgrass	*Dactylis glomerata* L.	54
Spotted ladysthumb	*Polygonum persicaria* L.	54
Red clover	*Trifolium pratense* L.	51
White clover	*Trifolium repens* L.	49
Autumn olive	*Elaeagnus umbellata* Thunb.	46
Climbing nightshade	*Solanum dulcamara* L.	44
White mulberry	*Morus alba* L.	43
Lesser burdock	*Arctium minus* Bernh.	43
Common buckthorn	*Rhamnus cathartica* L.	42
Common St. Johnswort	*Hypericum perforatum* L.	41
Nepalese browntop	*Microstegium vimineum* (Trin.) A. Camus	39
Amur honeysuckle	*Lonicera maackii* (Rupr.) Herder	39
Tree of heaven	*Ailanthus altissima* (P. Mill.) Swingle	36
Common sheep sorrel	*Rumex acetosella* L.	36
Meadow ryegrass	*Lolium pratense* (Huds.) S.J. Darbyshire	33
Canada bluegrass	*Poa compressa* L.	32
Black medick	*Medicago lupulina* L.	30
Sweet vernalgrass	*Anthoxanthum odoratum* L.	29
Yellow sweetclover	*Melilotus officinalis* (L.) Lam.	29
Sulphur cinquefoil	*Potentilla recta* L.	29

continued

Appendix 7.1 (continued)—Introduced species recorded on 1,302 plots in Northeastern United States, in order of number of plots where recorded

Common name	Scientific name	Number of plots
Redtop	*Agrostis gigantea* Roth	28
Morrow's honeysuckle	*Lonicera morrowii* Gray	28
Broad-leaved helleborine	*Epipactis helleborine* (L.) Crantz	27
Glossy buckthorn	*Frangula alnus* P. Mill.	26
Quackgrass	*Elymus repens* (L.) Gould	25
Asian bittersweet	*Celastrus orbiculata* Thunb.	25
Narrowleaf plantain	*Plantago lanceolata* L.	25
Creeping jenny	*Lysimachia nummularia* L.	25
Curly dock	*Rumex crispus* L.	25
Common mullein	*Verbascum thapsus* L.	25
Tatarian honeysuckle	*Lonicera tatarica* L.	24
Canada thistle	*Cirsium arvense* (L.) Scop.	23
Bull thistle	*Cirsium vulgare* (Savi) Ten.	23
Rough bluegrass	*Poa trivialis* L.	22
Tall fescue	*Lolium arundinaceum* (Schreb.) S.J. Darbyshire	21
European privet	*Ligustrum vulgare* L.	20
Sweet cherry	*Prunus avium* (L.) L.	20
Spreading hedgeparsley	*Torilis arvensis* (Huds.) Link	19
Deptford pink	*Dianthus armeria* L.	18
Bird vetch	*Vicia cracca* L.	18
Scotch pine	*Pinus sylvestris* L.	17
Prickly lettuce	*Lactuca serriola* L.	17
Meadow hawkweed	*Hieracium caespitosum* Dumort.	17
Coltsfoot	*Tussilago farfara* L.	17
Yellow salsify	Tragopogon dubius Scop.	16
Asiatic dayflower	*Commelina communis* L.	15
Common velvetgrass	*Holcus lanatus* L.	14
Catnip	*Nepeta cataria* L.	14
Common hawkweed	*Hieracium lachenalii* K.C. Gmel.	14
Bitter dock	*Rumex obtusifolius* L.	14
Golden clover	*Trifolium aureum* Pollich	14

continued

Appendix 7.1 (continued)—Introduced species recorded on 1,302 plots in Northeastern United States, in order of number of plots where recorded

Common name	Scientific name	Number of plots
Wine raspberry	*Rubus phoenicolasius* Maxim.	13
Japanese brome	*Bromus japonicus* Thunb. ex Murr.	11
Annual bluegrass	*Poa annua* L.	11
Big chickweed	*Cerastium fontanum* ssp. *vulgare* Baumg.	11
Marshpepper knotweed	*Polygonum hydropiper* L.	11
Dames rocket	*Hesperis matronalis* L.	11
Common motherwort	*Leonurus cardiaca* L.	11
Chinese lespedeza	*Lespedeza cuneata* (Dum.-Cours.) G. Don	11
Tall morning-glory	*Ipomoea purpurea* (L.) Roth	11
Common chickweed	*Stellaria media* (L.) Vill.	11
Common sowthistle	*Sonchus oleraceus* L.	11
Purple crownvetch	*Coronilla varia* L.	10
Indian strawberry	*Duchesnea indica* (Andr.) Focke	10
Russian olive	*Elaeagnus angustifolia* L.	10
Spotted knapweed	*Centaurea biebersteinii* DC.	10
Fly honeysuckle	*Lonicera x xylosteoides* Tausch	10
Oriental ladysthumb	*Polygonum cespitosum* Blume	10
Butter and eggs	*Linaria vulgaris* P. Mill.	10
Birdfoot deervetch	*Lotus corniculatus* L.	10
Garden yellowrocket	*Barbarea vulgaris* Ait. f.	10
Alsike clover	*Trifolium hybridum* L.	10
Common reed	*Phragmites australis* (Cav.) Trin. ex Steud.	9
Common mouse-ear Chickweed	*Cerastium fontanum* Baumg.	9
Norway spruce	*Picea abies* (L.) Karst.	9
False baby's breath	*Galium mollugo* L.	9
Spotted snapweed	*Impatiens balsamina* L.	9
Nodding plumeless thistle	*Carduus nutans* L.	8
Paradise apple	*Malus pumila* P. Mill.	8
Norway maple	*Acer platanoides* L.	8
Common tansy	*Tanacetum vulgare* L.	8

continued

Appendix 7.1 (continued)—Introduced species recorded on 1,302 plots in Northeastern United States, in order of number of plots where recorded

Common name	Scientific name	Number of plots
Narrowleaf cattail	*Typha angustifolia* L.	8
Field bindweed	*Convolvulus arvensis* L.	7
Brittlestem hempnettle	*Galeopsis tetrahit* L.	7
Alfalfa	*Medicago sativa* L.	7
Greater burdock	*Arctium lappa* L.	7
Mouseear hawkweed	*Hieracium pilosella* L.	7
Green bristlegrass	*Setaria viridis* (L.) Beauv.	7
Field clover	*Trifolium campestre* Schreb.	7
Cheatgrass	*Bromus tectorum* L.	6
Poison hemlock	*Conium maculatum* L.	6
Fuller's teasel	*Dipsacus fullonum ssp. sylvestris* L.	6
Marijuana	*Cannabis sativa* L.	6
Colonial bentgrass	*Agrostis capillaris* L.	6
Love-lies-bleeding	*Amaranthus caudatus* L.	6
Wild garlic	*Allium vineale* L.	6
White sweetclover	*Melilotus alba* Medikus	6
Corn speedwell	*Veronica arvensis* L.	6
Johnsongrass	*Sorghum halepense* (L.) Pers.	6
Jack-go-to-bed-at-noon	*Tragopogon pratensis* L.	6
Siberian elm	*Ulmus pumila* L.	6
Wild parsnip	*Pastinaca sativa* L.	5
Barnyardgrass	*Echinochloa crus-galli* (L.) Beauv.	5
Winged burning bush	*Euonymus alata* (Thunb.) Sieb.	5
Chicory	*Cichorium intybus* L.	5
Dwarf honeysuckle	*Lonicera xylosteum* L.	5
Common barberry	*Berberis vulgaris* L.	5
Japanese knotweed	*Polygonum cuspidatum* Sieb. & Zucc.	5
Celandine	*Chelidonium majus* L.	5
Purple loosestrife	*Lythrum salicaria* L.	5
Maidenstears	*Silene vulgaris* (Moench) Garcke	5
Field sowthistle	*Sonchus arvensis* L.	5
Princesstree	*Paulownia tomentosa* (Thunb.) Sieb. & Zucc. ex Steud.	4

continued

Appendix 7.1 (continued)—Introduced species recorded on 1,302 plots in Northeastern United States, in order of number of plots where recorded

Common name	Scientific name	Number of plots
Creeping buttercup	*Ranunculus repens* L.	4
Burnweed	*Erechtites hieracifolia* (L.) Raf. ex DC.	4
Marsh thistle	*Cirsium palustre* (L.) Scop.	4
Absinthium	*Artemisia absinthium* L.	4
Ornamental jewelweed	*Impatiens glandulifera* Royle	4
Hairy catsear	*Hypochaeris radicata* L.	4
Henbit deadnettle	*Lamium amplexicaule* L.	4
Korean clover	*Kummerowia stipulacea* (Maxim.) Makino	4
Japanese clover	*Kummerowia striata* (Thunb.) Schindl.	4
Tall hawkweed	*Hieracium piloselloides* Vill.	4
Rugosa rose	*Rosa rugosa* Thunb.	4
Prickly Russian thistle	*Salsola tragus* L.	4
Black nightshade	*Solanum nigrum* L.	4
Grasslike starwort	*Stellaria graminea* L.	4
Corn gromwell	*Buglossoides arvensis* (L.) I.M. Johnston	3
Leafy spurge	*Euphorbia esula* L.	3
Winter creeper	*Euonymus fortunei* (Turcz.) Hand.-Maz.	3
Silver cinquefoil	*Potentilla argentea* L.	3
Hoary false madwort	*Berteroa incana* (L.) DC.	3
Austrian pine	*Pinus nigra* Arnold	3
Musk mallow	*Malva moschata* L.	3
Common mallow	*Malva neglecta* Wallr.	3
Dwarf snapdragon	*Chaenorhinum minus* (L.) Lange	3
Hedge false bindweed	*Calystegia sepium* ssp. *sepium* (L.) R. Br.	3
Oriental bittersweet	*Celastrus orbiculatus* Thunb.	3
Rampion bellflower	*Campanula rapunculoides* L.	3
Peppermint	*Mentha x piperita* L. (pro sp.)	3
True forget-me-not	*Myosotis scorpioides* L.	3
Purple deadnettle	*Lamium purpureum* L.	3
Redstar	*Ipomoea coccinea* L.	3
Corn	*Zea mays* L.	3

continued

Appendix 7.1 (continued)—Introduced species recorded on 1,302 plots in Northeastern United States, in order of number of plots where recorded

Common name	Scientific name	Number of plots
Witch's moneybags	*Hylotelephium telephium ssp. telephium* (L.) H. Ohba.	3
Germander speedwell	*Veronica chamaedrys* L.	3
Common wheat	*Triticum aestivum* L.	3
Beefsteakplant	*Perilla frutescens* (L.) Britt.	2
Shepherd's purse	*Capsella bursa-pastoris* (L.) Medik.	2
Meadow brome	*Bromus commutatus* Schrad.	2
Lesser pond sedge	*Carex acutiformis* Ehrh.	2
India mustard	*Brassica juncea* (L.) Czern.	2
Field mustard	*Brassica rapa* L.	2
Smooth hawksbeard	*Crepis capillaris* (L.) Wallr.	2
Scarlet pimpernel	*Anagallis arvensis* L.	2
Chinese yam	*Dioscorea oppositifolia* L.	2
Codlins and cream	*Epilobium hirsutum* L.	2
Wormseed wallflower	*Erysimum cheiranthoides* L.	2
Stinkgrass	*Eragrostis cilianensis* (All.) Vign. ex Janchen	2
Smooth crabgrass	*Digitaria ischaemum* (Schreb.) Schreb. ex Muhl.	2
David's spurge	*Euphorbia davidii* Subils	2
Thymeleaf sandwort	*Arenaria serpyllifolia* L.	2
Spearmint	*Mentha spicata* L.	2
Meadow foxtail	*Alopecurus pratensis* L.	2
Redroot amaranth	*Amaranthus retroflexus* L.	2
Velvetleaf	*Abutilon theophrasti* Medik.	2
Yellow Spring bedstraw	*Galium verum* L.	2
Gallant-soldier	*Galinsoga parviflora* Cav.	2
Tall yellow sweetclover	*Melilotus altissimus* Thuill.	2
Field pepperweed	*Lepidium campestre* (L.) Ait. f.	2
Ivyleaf morning-glory	*Ipomoea hederacea* Jacq.	2
Wild oat	*Avena fatua* L.	2
Woolly burdock	*Arctium tomentosum* P. Mill.	2
Bermudagrass	*Cynodon dactylon* (L.) Pers.	2
Rose of Sharon	*Hibiscus syriacus* L.	2

continued

Appendix 7.1 (continued)—Introduced species recorded on 1,302 plots in Northeastern United States, in order of number of plots where recorded

Common name	Scientific name	Number of plots
Fig buttercup	*Ranunculus ficaria* L.	2
Cereal rye	*Secale cereale* L.	2
White willow	*Salix alba* L.	2
Japanese bristlegrass	*Setaria faberi* Herrm.	2
Spiny sowthistle	*Sonchus asper* (L.) Hill	2
Yellow bristlegrass	*Setaria pumila* (Poir.) Roemer & J.A. Schultes	2
Hedgemustard	*Sisymbrium officinale* (L.) Scop.	2
Bladder campion	*Silene latifolia* ssp. *alba* Poir.	2
Suckling clover	*Trifolium dubium* Sibthorp	2
Garden vetch	*Vicia sativa* L.	2
Common comfrey	*Symphytum officinale* L.	2
Common periwinkle	*Vinca minor* L.	2
Garden valerian	*Valeriana officinalis* L.	2
Field pennycress	*Thlaspi arvense* L.	2
Rabbitfoot clover	*Trifolium arvense* L.	2
Erect hedgeparsley	*Torilis japonica* (Houtt.) DC.	2
Birdeye speedwell	*Veronica persica* Poir.	2
Stinging nettle	*Urtica dioica* ssp. *dioica* L.	2
Pearl millet	*Pennisetum glaucum* (L.) R. Br.	1
Japanese pachysandra	*Pachysandra terminalis* Sieb. & Zucc.	1
Erect brome	*Bromus erectus* Huds.	1
Wild radish	*Raphanus raphanistrum* L.	1
Black mustard	*Brassica nigra* (L.) W.D.J. Koch	1
Siberian peashrub	*Caragana arborescens* Lam.	1
Caucasian bluestem	*Bothriochloa bladhii* (Retz.) S.T. Blake	1
Smooth brome	*Bromus inermis* ssp. *inermis var. inermis* Leyss.	1
Whitetop	*Cardaria draba* (L.) Desv.	1
Caraway	*Carum carvi* L.	1
Bald brome	*Bromus racemosus* L.	1
Kenilworth ivy	*Cymbalaria muralis* P.G. Gaertn., B. Mey. & Scherb.	1

continued

Appendix 7.1 (continued)—Introduced species recorded on 1,302 plots in Northeastern United States, in order of number of plots where recorded

Common name	Scientific name	Number of plots
Rye brome	*Bromus secalinus* L.	1
Corn brome	*Bromus squarrosus* L.	1
Tidalmarsh flatsedge	*Cyperus serotinus* Rottb.	1
Splitlip hempnettle	*Galeopsis bifida* Boenn.	1
Acacia	*Acacia sophorae* (Labill.) R.Br.	1
Indian teasel	*Dipsacus sativus* (L.) Honckeny	1
Tall oatgrass	*Arrhenatherum elatius* (L.) Beauv. ex J.& K. Presl	1
Birthwort	*Aristolochia clematitis* L.	1
Garden chervil	*Anthriscus cerefolium* (L.) Hoffmann	1
Corn chamomile	*Anthemis arvensis* L.	1
Annual vernalgrass	*Anthoxanthum aristatum* Boiss.	1
Annual wallrocket	*Diplotaxis muralis* (L.) DC.	1
Violet crabgrass	*Digitaria violascens* Link	1
Blessed thistle	*Cnicus benedictus* L.	1
Weeping lovegrass	*Eragrostis curvula* (Schrad.) Nees	1
Hairy cupgrass	*Eriochloa villosa* (Thunb.) Kunth	1
Doubtful knight's-spur	*Consolida ajacis* (L.) Schur	1
Buckwheat	*Fagopyrum esculentum* Moench	1
Sweet autumn virginsbower	*Clematis terniflora* DC.	1
European spindletree	*Euonymus europaea* L.	1
Blue flax	*Linum perenne* L.	1
Rose campion	*Lychnis coronaria* (L.) Desr.	1
Black bindweed	*Polygonum convolvulus* var. *convolvulus* L.	1
Border privet	*Ligustrum obtusifolium* Sieb. & Zucc.	1
European stoneseed	*Lithospermum officinale* L.	1
Chinese privet	*Ligustrum sinense* Lour.	1
White poplar	*Populus alba* L.	1
Oval-leaf knotweed	*Polygonum arenastrum* Jord. ex Boreau	1
Gold-of-pleasure	*Camelina sativa* (L.) Crantz	1
Oakleaf goosefoot	*Chenopodium glaucum* L.	1
Sticky chickweed	*Cerastium glomeratum* Thuill.	1

continued

Appendix 7.1 (continued)—Introduced species recorded on 1,302 plots in Northeastern United States, in order of number of plots where recorded

Common name	Scientific name	Number of plots
Sneezeweed	*Achillea ptarmica* L.	1
Amur peppervine	*Ampelopsis brevipedunculata* (Maxim.) Trautv.	1
Crested wheatgrass	*Agropyron cristatum* (L.) Gaertn.	1
Wild chives	*Allium schoenoprasum* L.	1
Broadleaf wild leek	*Allium ampeloprasum* L.	1
Amur maple	*Acer ginnala* Maxim.	1
Common yarrow	*Achillea millefolium* var. *millefolium* L.	1
European columbine	*Aquilegia vulgaris* L.	1
Broadleaf Solomon's seal	*Polygonatum hirsutum* (Bosc ex Poir.) Pursh	1
Common corncockle	*Agrostemma githago* L.	1
Monkshoodvine	*Ampelopsis aconitifolia* Bunge	1
Common bugle	*Ajuga reptans* L.	1
Bishop's goutweed	*Aegopodium podagraria* L.	1
Orange daylily	*Hemerocallis fulva* (L.) L.	1
Weeping forsythia	*Forsythia suspensa* (Thunb.) Vahl	1
Dovefoot geranium	*Geranium molle* L.	1
Roundfruit rush	*Juncus compressus* Jacq.	1
Common barley	*Hordeum vulgare* L.	1
Plume poppy	*Macleaya cordata* (Willd.) R. Br.	1
Disc mayweed	*Matricaria discoidea* DC.	1
Spotted henbit	*Lamium maculatum* L.	1
Italian ryegrass	*Lolium perenne* ssp. *multiflorum* L.	1
Field cottonrose	*Logfia arvensis* (L.) Holub	1
European stickseed	*Lappula squarrosa* (Retz.) Dumort.	1
Bell's honeysuckle	*Lonicera x bella* Zabel	1
Chinese ginseng	*Panax ginseng* C. Meyer	1
Garden asparagus	*Asparagus officinalis* L.	1
Field scabiosa	*Knautia arvensis* (L.) Coult.	1
Hyssop	*Hyssopus officinalis* L.	1
White deadnettle	*Lamium album* L.	1
Jimsonweed	*Datura stramonium* L.	1

continued

Appendix 7.1 (continued)—Introduced species recorded on 1,302 plots in Northeastern United States, in order of number of plots where recorded

Common name	Scientific name	Number of plots
Hibiscus	*Hibiscus lunariifolius* Willd.	1
Dwarf iris	Iris pumila L.	1
Orchardgrass	*Dactylis glomerata* ssp. *glomerata* L.	1
European meadow rush	*Juncus inflexus* L.	1
Flower of an hour	*Hibiscus trionum* L.	1
Asiatic tearthumb	*Polygonum perfoliatum* L.	1
European gooseberry	*Ribes uva-crispa* var. *sativum* L.	1
Sweetbriar rose	*Rosa eglanteria* L.	1
European black currant	*Ribes nigrum* L.	1
St. Anthony's turnip	*Ranunculus bulbosus* L.	1
Cultivated currant	*Ribes rubrum* L.	1
Common pear	*Pyrus communis* L.	1
Laurel willow	*Salix pentandra* L.	1
Cutleaf blackberry	*Rubus laciniatus* Willd.	1
Bouncingbet	*Saponaria officinalis* L.	1
Old-man-in-the-Spring	*Senecio vulgaris* L.	1
Grain sorghum	*Sorghum bicolor* ssp. *bicolor* (L.) Moench	1
Yellow bristlegrass	*Setaria pumila* ssp. *pallidifusca* (Poir.) Roemer & J.A. Schultes	1
Japanese meadowsweet	*Spiraea japonica* L. f.	1
Bladder campion	*Silene latifolia* Poir.	1
Small tumbleweed mustard	*Sisymbrium loeselii* L.	1
Garden vetch	*Vicia sativa* ssp. *nigra* L.	1
Alexanders	*Smyrnium olusatrum* L.	1
Lewiston cornsalad	*Valerianella locusta* (L.) Lat.	1
Common lilac	*Syringa vulgaris* L.	1
Nightflowering silene	*Silene noctiflora* L.	1
Bigleaf periwinkle	*Vinca major* L.	1
Small-leaf spiderwort	*Tradescantia fluminensis* Vell.	1
European cranberrybush	*Viburnum opulus* var. *opulus* L.	1
Threadstalk speedwell	*Veronica filiformis* Sm.	1

CHAPTER 8.

National Trends in Ozone Injury to Forest Plants: 16 Years of Biomonitoring

Gretchen Smith

INTRODUCTION

The ozone indicator, an important research component of the Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture, was developed and implemented to address specific concerns about the negative effects of ground-level ozone pollution on forest health and productivity. Ozone is a highly toxic air contaminant that has been shown repeatedly to damage tree growth and cause significant disturbance to forest ecosystems. Ozone also causes distinct foliar injury symptoms to certain species (bioindicator plants) that can be used to detect and monitor ozone stress (biomonitoring) in the forest environment.

Biomonitoring surveys, begun in 1994 in the Eastern United States and 1998 in the Western United States, provide important regional information on ozone air quality, and a field-based measure of ozone injury and probable impact unavailable from any other data source (Coulston and others 2003, Smith and others 2007). Currently, the national biomonitoring network consists of over 1,005 field sites in 40 States. At every site, the amount and severity of injury to the foliage of ozone-sensitive plants is used to formulate a plot-level injury index referred to as the ozone biosite index or BI (Smith and others 2007). BI values can be used to identify forested areas at risk of ozone impact (Coulston and others 2003) and to describe relative ozone air quality. This report does not address risk, per se, but does examine how emerging long term trends in the BI in different regions of the country may be informing the risk assessment process.

The Forest Inventory and Analysis (FIA) Program of the Forest Service took over implementation of the biomonitoring program in 2002. Data collection, documentation, and reporting are coordinated out of three regional FIA units, each belonging respectively to the Northern Research Station, the Southern Research Station, and the Pacific Northwest Research Station of the Forest Service, with the Northern Research Station FIA unit having the longest record of biomonitoring data, extending from 1994 to 2010. At the regional level, the ozone indicator was designed to assess if plant-damaging concentrations of ozone are present in U.S. forests, where ozone stress is highest and most frequent, and whether or not ozone stress is increasing or decreasing over time. The purpose of this report is to address these issues of forest health assessment with a summary review of the major findings of the ozone surveys for each region. The broad relationship between injury (BI) and ozone exposure is also discussed.

METHODS

Sample Area

Ozone sampling occurs on a unique national grid (Smith and others 2007, White and others 1992) that consists of a single panel of ozone biomonitoring sites that are measured every year (fig. 8.1). The field sites vary in size and do not have set boundaries. They are defined by the presence of ozone sensitive bioindicator species indigenous to each FIA region. The Northern Research Station study area covers 24 States, typically divided into the Northeast and North Central sub-regions; the Southern Research Station study area covers 13 States from Virginia to eastern Texas, with data available starting

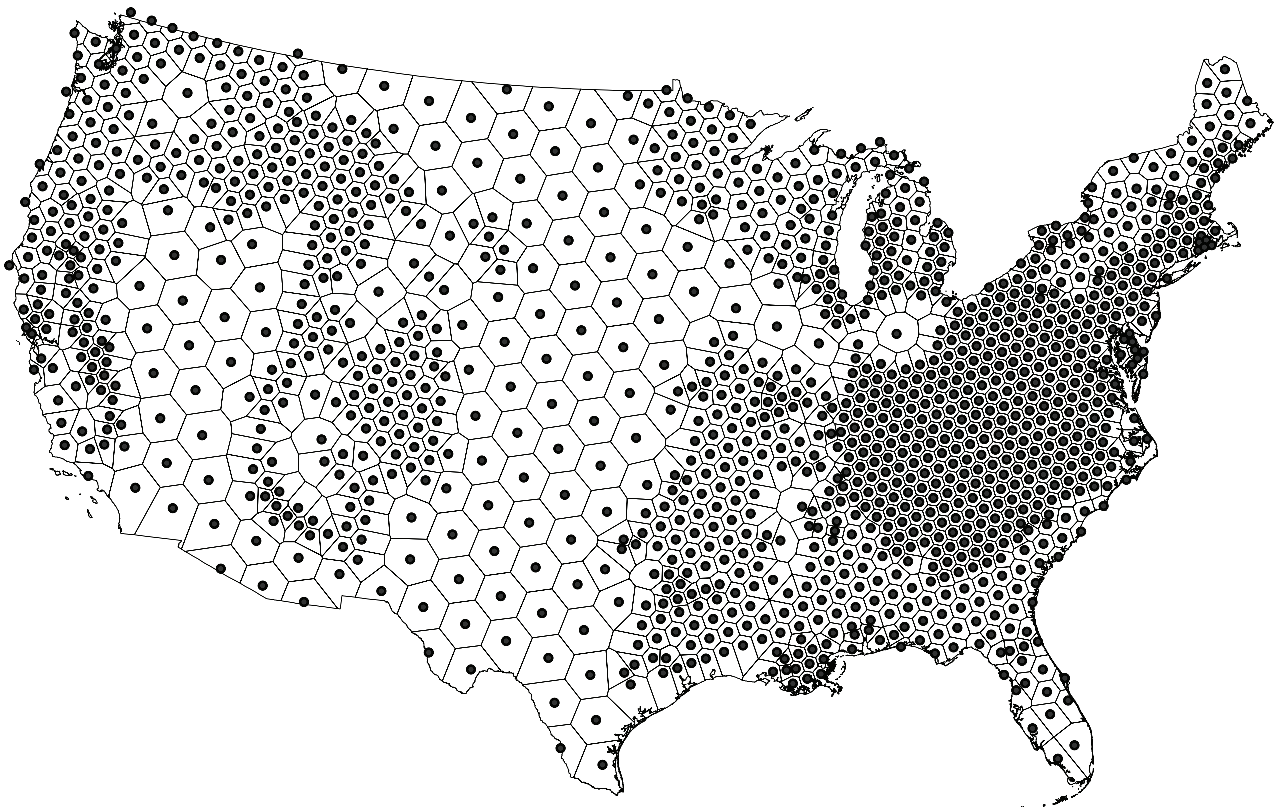

Figure 8.1—Forest Inventory and Analysis ozone biomonitoring grid developed from the Environmental Monitoring and Assessment Program (EMAP) base grid (White and others 1992). The grid has four sampling intensities based on sensitive species and ambient ozone concentrations.

in 1997; and the Pacific Northwest Research Station study area covers three States—California, Oregon, and Washington—with data from 2000. Procedures for biomonitoring are standardized nationally (USDA Forest Service 2006, Woodall and others 2010), using a defined temporal evaluation window to minimize variability associated with the seasonality of plant response to ozone exposure.

Foliar Injury

Crews return to the same sites and evaluate the same species and general population of plants every year. Trained and certified in ozone injury recognition every year, the crews submit injured leaf vouchers to regional experts to validate the field results. The site-level biosite index (BI) is derived from the amount, severity, and incidence of ozone-induced foliar injury to ozone-sensitive bioindicator plants at each biosite (Smith and others 2007). The site-level values describe a gradation of plant injury response that quantifies the degree of ozone injury conditions[1] on the biomonitoring plots.

Ozone Exposure

SUM06 and N100 are two cumulative ozone exposure indices that are used to characterize ambient ozone exposures. Hourly ozone data obtained from the U.S. Environmental Protection Agency (EPA) (http://www.epa.gov/air/data/index.html) were used to interpolate an ozone exposure surface across the landscape and assign an average growing season (June, July, and August) SUM06 (the sum of all hourly average ozone concentrations ≥ 0.06 ppm) value to each biosite and year. The same database was used to assign an N100 (the number of hours of ozone ≥ 100 ppb) value to each biosite and year. The SUM06 metric provides an indication of chronic ozone stress for the growing season, and N100 an indication of peak ozone concentrations.

Analysis

Descriptive statistics presented here include the percent plots with validated ozone injury by year and region. Calculations of the average BI by year and region were also made. Additional plot-level estimates of ozone exposure and site moisture were obtained for the Northern region only to determine if fluctuations and trends in foliar injury over the 16-year period from 1994 to 2009 are correlated with trends in ozone exposure.

RESULTS AND DISCUSSION

Field Implementation

National field implementation began in New England in 1994 and spread south to the mid-Atlantic States, and west to the North Central States; new States entering the program every year. In the Northern Research Station study area, the number of years of biomonitoring varies from 9 to 17 depending on the start year

[1]As defined by Smith and others (2008): visible symptoms on bioindicator plants indicate that O_3 is present at concentrations that cause injury and that predisposing conditions (e.g., adequate site moisture) are coincident.

for each State (table 8.1) and an occasional year when sampling was interrupted. By 2002, the ozone grid was complete and all 24 States in the Northern Research Station area were participating in the biomonitoring program.

In the Southern Research Station area, the first sites were established in Alabama, Georgia, and Virginia in 1997 and 1998, with nine new States added between 1999 and 2002. Two States, Mississippi and Oklahoma, have been participating since 2009. Sites are largely absent from the coastal areas of the more Southern States due to an absence of bioindicator species in these areas, and the generally wet conditions. The Pacific Northwest Research Station initiated pilot studies in 1998 and 1999, but considers 2000 the official start year for the ozone surveys. Pacific Northwest Research Station FIA crews have been sampling without interruption for the last 10 years.

For all three FIA units, the number of biosites evaluated every year tended to stabilize in 2002, when FIA took over implementation on an improved ozone grid (table 8.2).

Table 8.1—Number of years of biomonitoring, number of years with ozone injury, and year biomonitoring was started, by region and State, 1994–2010

Region and year	Number of years		Start year[a]
	Biomonitoring	Ozone injury detected	
Northeast States			
Connecticut	17	17	1994
Delaware	15	14	1995
Maine	17	9	1994
Maryland	17	17	1994
Massachusetts	17	17	1994
New Hampshire	17	15	1994
New Jersey	17	15	1994
New York	12	12	1999
Ohio	14	14	1997
Pennsylvania	14	14	1995
Rhode Island	17	17	1994
Vermont	17	17	1994
West Virginia	16	16	1995
North Central States			
Illinois	14	14	1997
Indiana	15	15	1996
Iowa	11	6	2000
Kansas	9	5	2002
Michigan	17	16	1994
Minnesota	17	6	1994
Missouri	11	9	2000
Nebraska	9	1	2002
North Dakota	9	0	2002
South Dakota	9	3	2002
Wisconsin	17	17	1994

continued

Table 8.1 (continued)—Number of years of biomonitoring, number of years with ozone injury, and year biomonitoring was started, by region and State, 1994–2010

Region and year	Number of years: Biomonitoring	Number of years: Ozone injury detected	Start year[a]
Southern States			
Alabama	13	2	1998
Arkansas	10	3	2001
Florida	9	1	2002
Georgia	14	13	1997
Kentucky	11	11	2000
Louisiana	8	3	2001
Mississippi	2	1	2009
North Carolina	12	10	1999
Oklahoma	2	2	2009
South Carolina	12	12	1999
Tennessee	11	10	2000
Texas	9	4	2002
Virginia	14	11	1997
West Coast States			
California	10	10	2000
Oregon	10	0	2000
Washington	10	7	2000

[a] Some States are missing interim years between start date and current year.

Table 8.2—Number of evaluated biosites by year and by Forest Inventory and Analysis region

Region and year	Number of biosites evaluated: Northern	Southern	Pacific Northwest
1994	118	-	-
1995	284	-	-
1996	229	-	-
1997	274	19	-
1998	465	22	-
1999	560	90	-
2000	559	178	70
2001	574	248	77
2002	490	316	125
2003	498	320	134
2004	494	351	130
2005	472	359	136
2006	470	335	138
2007	463	314	132
2008	457	314	129
2009	467	382	134
2010	470	401	134

– = No biosites evaluated.

Air Quality and the Ozone Grid

Differences in maximum and mean ozone exposure statistics help to define the FIA regions and States in terms of ozone air quality during the growing season (table 8.3). Relatively clean air States are found in Northern New England (Maine, New Hampshire, and Vermont), the Northern Plains (Nebraska, South Dakota, and North Dakota), and the Northwest (Oregon and Washington); while moderate air quality States are found in southern New England (Massachusetts, Connecticut, and Rhode Island), the East North Central region (Illinois, Indiana, and Ohio), and the South (Georgia, South Carolina, North Carolina, Tennessee, and Kentucky). States with unhealthy air quality are in the mid-Atlantic region (Virginia, West Virginia, Maryland, Delaware, Pennsylvania, New Jersey, and New York), and the Southwest (California). Additional States, e.g., Kansas, Iowa, and the Great Lakes States (Minnesota, Wisconsin, and Michigan), tend to fall into an intermediate air quality category or have a wide range of ozone exposures from clean to moderate depending on proximity to population centers within each State.

The exposure characteristics of a given State or sub-region do not always line up with the results of the ozone survey in terms of how

Table 8.3—Regional differences in maximum and mean ozone exposure data, 1994–2005

Region[a]	Range of maximum ozone exposure values (SUM06)[b] 1994-2005	Mean value 1994-2005	Ozone exposure category[c]
Northern New England	8.3 - 29.2	6.2	Clean
Southern New England	14.9 - 34.7	18.0	Moderate
Mid-Atlantic States	22.2 – 71.2	25.9	Unhealthy
Northern Plains	7.7 – 24.2	6.1	Clean
East North Central	17.1 – 52.3	20.7	Moderate
South	20.9 – 92.8	16.9	Moderate
Northwest	6.5 – 25.1	5.9	Clean
Southwest	76.8 - 117.3	28.7	Unhealthy

[a] Regions are defined as follows. Northern New England: Maine, New Hampshire, Vermont; Southern New England: Massachusetts, Connecticut, Rhode Island; Mid-Atlantic: Delaware, Maryland, New Jersey, New York, Pennsylvania, Virginia, West Virginia; Northern Plains: Nebraska, North Dakota, South Dakota; East North Central: Illinois, Indiana, Ohio; South: Alabama, Georgia, Kentucky, North Carolina, South Carolina, Tennessee; Northwest: Oregon, Washington; and Southwest: California.

[b] SUM06 = Sum of hourly ozone concentrations ≥0.06 ppm. Maximum and mean values are calculated by State and year and then averaged for each region.

[c] Descriptive ozone exposure categories are based on mean values. Clean: SUM06 <10 ppm-hr; Moderate: SUM06 10-25 ppm-hr; Unhealthy: SUM06 >25 ppm-hr.

often ozone-induced foliar injury is detected on the ozone grid (table 8.1). For example, over the 1994 to 2010 time period, ozone injury was detected every year in almost every State in the Eastern United States, from Maine (clean) south to Georgia (moderate), and from Ohio (moderate) west to Kansas and north to Wisconsin; and in the Western United States, in Washington (clean) as well as California (unhealthy). The only States with no ozone injury or very few years with injury detected are North Dakota, South Dakota, Nebraska, Oregon, Alabama, Arkansas, Florida, Louisiana, and east Texas.

It is noteworthy from a monitoring perspective when ozone injury is detected in a State or region previously thought to be free of ozone stress, even if the injury occurs on only a small number of the bioindicator plants or sites. This is the case in the State of Washington, where the repeated detection of ozone injury at a single location is, at least in part, explained by the soil moisture conditions at the biosite, and in northern portions of Vermont, which may be influenced by polluted air masses moving north from the mid-Atlantic region or by the absence of other pollutants which react with O_3 and effectively remove it from the air. FIA ozone surveys also detected injury for the first time at several locations in the more northern portions of California (Campbell and others 2007) starting in 2005. In the many States where ozone injury is detected every year, or almost every year, the survey results underscore that a large area of forest land in this country, in both the East and the West, is subject to levels of ozone pollution that may negatively affect the forest ecosystem.

Every year, the EPA publishes an ozone exceedance map (http://www.asl-associates.com/revised_8-hr_075.htm), which highlights the counties in each State that are out of compliance with the National Ambient Air Quality Standard (NAAQS) for O_3 set to protect vegetation from harmful effects. In preparation for the 2007 review of the ozone standard, the EPA overlaid Forest Service biomonitoring data with the exceedance map and found that there were many counties in compliance with the existing O_3 standard where FIA field crews were routinely documenting ozone injury to sensitive plants (U.S. EPA 2007). This study was one of several that served as evidence that the secondary ozone standard needed to be strengthened, a recommendation that was adopted by the EPA Clean Air Scientific Advisory Committee and the EPA Administrator, and that became law in 2008. Currently, there is a new proposal by the EPA to establish a distinct cumulative, seasonal "secondary" standard, referred to as the W126 index, which is designed to protect sensitive vegetation and ecosystems, including forests, parks, wildlife refuges, and wilderness areas (http://www.epa.gov/air/ozonepollution/actions.html#jan10s). The multi-year findings of the FIA field-based biomonitoring program suggest that this protective action has scientific merit.

Trend Data

Air—EPA reports that ground-level O_3 concentrations are 10 percent lower in 2008 than in 2001 across the Nation with the most notable decline occurring after 2002 (U.S. EPA 2010[2]). Still, there are localized areas such as parts of the Los Angeles air basin, and in or near Atlanta, where ground-level O_3 is increasing. There are also growing concerns about ozone air quality in parts of the Interior United States (e.g., Wyoming, Utah, and Idaho), where increased activity associated with the natural gas industry is contributing to previously undocumented peaks in localized O_3 concentrations. A comparison of trend data from California versus the Eastern United States shows that the majority of ozone improvement in recent years occurred in the East as a result of successfully implemented pollution control measures leading to large reductions in NO_x emissions (ozone precursor pollutants) beginning in 2003.

Injury—The percent of injured biosites indicates how widespread ozone injury conditions are for the Northern Research Station, Southern Research Station, and Pacific Northwest Research Station regions by year from 1994 to 2010 (fig. 8.2). Percent of injured sites for the North fluctuated from one year to the next showing an overall downward trend over 17 years of biomonitoring. The highest percent injury occurred in 1994 (55.9 percent), 1998 (48.4 percent), and 2000 (47.6 percent), the lowest in 2008 (23.2 percent) and 2009 (19.9 percent). In the years prior to 2003, percent injured plots averaged above 30 percent for 7 of the 9 years, and for only 3 of the 8 years from 2003 on. Although the overall trend was downward, percent injured plots was back up above 30 percent in 2010, perhaps signaling a change in injury conditions.

For the first seven sample years in the South (1997 to 2003), the percent injured biosites was often similar to those in the North and the overall trend was downward. Values were

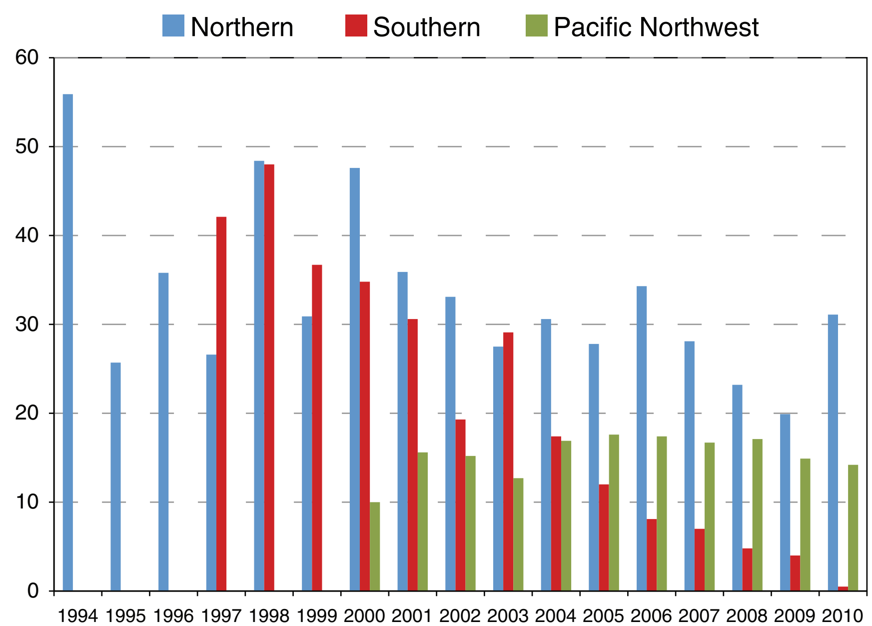

Figure 8.2—Ozone injury to forest plants in the United States by Forest Inventory and Analysis region: percent injured plots by year, 1994 to 2010.

[2] http://www.epa.gov/airtrends/2010/.

relatively high (>30 percent) in 1997 through 2001, dropping to 19.3 percent in 2002, increasing again to 29.1 percent in 2003 before dropping sharply to <10 percent in 2006, and continuing to decline to a minimum percent injured plots (<1 percent) in 2010. In contrast, the percent injured plots in the Pacific Northwest fluctuated between 10 and 17 percent for all 11 years of biomonitoring showing, if anything, a slight increasing trend in percent injured biosites over the 2000 to 2010 time period.

Even more than percent injured biosites, the BI values are expected to fluctuate from one year to the next in response to variable ozone exposure levels and other factors that influence ozone flux. BI values provide a comparative measure of injury severity with increasing values indicating an increased risk of probable ozone impacts to sensitive trees and ecosystems (Smith and others 2007). Site-level BI data for the Northern Research Station are presented with estimated SUM06 and N100 data (fig. 8.3) to

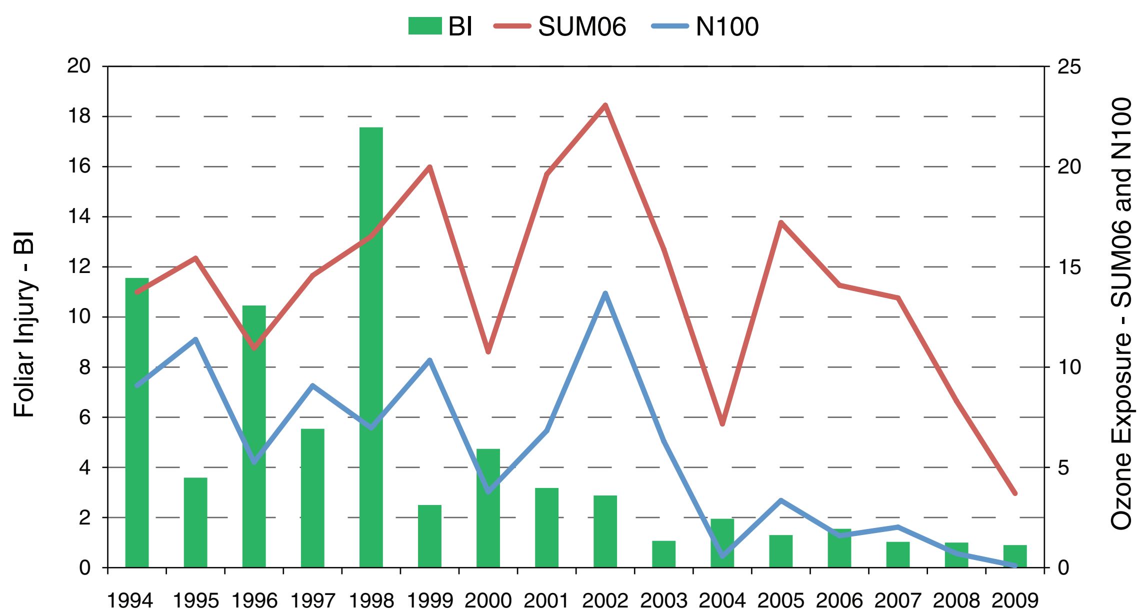

Figure 8.3—Trends in ozone-induced foliar injury (BI) and ozone exposure (SUM06 and N100) in the Forest Inventory and Analysis Northern Region from 1994 to 2009.

show possible associations between foliar injury and ozone exposure over the 1994 to 2009 time period.[3]

Mean regional BI values are relatively high in 1994 but fluctuate down and up again over the next 4 years reaching a maximum value of 17.6 in 1998 before dropping off sharply in 1999. BI increases in 2000 but then starts a downward trend to relatively low values suggesting very little risk of ozone impact on a regional scale from 2003 on. Both SUM06 and N100 also fluctuate from one year to the next, but there is little direct association between injury and exposure. In 1995 and 1999, for example, BI values drop off from the previous year even though ozone exposure values are increasing. This can be explained by the fact that 1995 and 1999 were two of the driest years over the 1994 to 2009 survey period especially in the high ozone areas of the Northern Research Station such as the mid-Atlantic States (Smith and others 2008, Smith 2009). Dry conditions caused plant stomata to close, thus preventing ozone uptake and subsequent injury. This result demonstrates the biological relevance of the biomonitoring data since the BI values reflect how much ozone gets inside the plant rather than what can be measured in the ambient air.

Focusing on trends, it is clear that injury severity and the implied risk of ozone impact as described by the BI data have been steadily decreasing in recent years. As suggested earlier, ground-level ozone concentrations have also been decreasing in the East since 2002, peak concentrations (N100) much more so than the more moderate concentrations captured in the SUM06 statistic. In this sense, the trend in BI and percent injured plots for the North region mirrors the ozone exposure data showing an overall declining trend from 1994 through 2009.

BI data from the South are not available for the most recent years (2008 and 2009), but for the 1997 to 2007 time period there is no obvious decline in BI values (fig. 8.4). In contrast, the

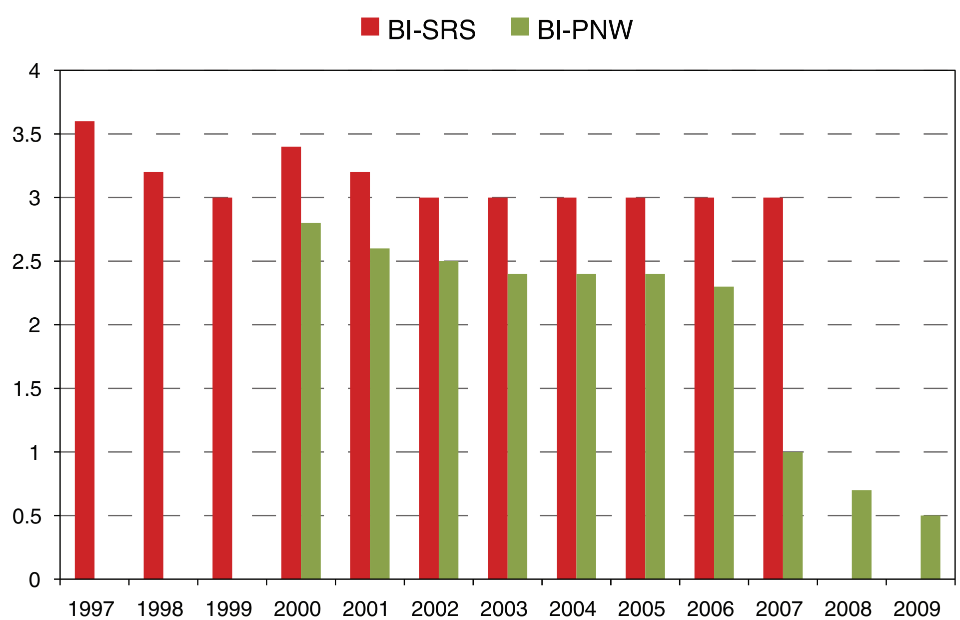

Figure 8.4—Trends in ozone-induced foliar injury (BI) in the Southern Research Station Forest Inventory and Analysis and Pacific Northwest Research Station Forest Inventory and Analysis regions from 1997 to 2009.

[3]Air quality data for 2010 were not available from the EPA for inclusion in this report.

BI data for the Pacific Northwest does suggest a decreasing trend in foliar injury severity and probable ozone impact, especially for California, where the majority of sites and plants with injury are located.

Injury and Exposure—Regional Summaries

Pacific Northwest Research Station Study Area— Reporting on results for 2000–05, Campbell and others (2007) noted that ozone injury occurs frequently on biosites in California demonstrating that ozone is present at toxic levels. Injury generally correlated with ambient ozone concentrations such that areas with the highest SUM06 values had the highest percent injured sites and highest mean BI. Although the highest percentage of biosites with injury occurred in southern California, new areas of previously unreported injury were detected in northern parts of the State. This early study did not discern any trends in ozone injury between 2000 and 2005. However, with the additional data from 2006 through 2010, we can now suggest that even though the percent injured biosites has not changed much, there is a discernible downward trend in injury severity.

Southern Research Station Study Area—Rose and others (2009) examined biomonitoring results for 2002–06 and concluded that even though ambient ozone concentrations were reportedly on the decline in the South during that time period, ozone-induced foliar injury was still occurring every year, particularly in Georgia and South Carolina where BI values were highest. However, a 5-year average of the BI data suggested that most of the forest land in the South is at low risk of ozone impact. The authors suggest that a prolonged region-wide drought may have served to protect the southern forests from ambient ozone concentrations and lower the regional mean BI values. Examining relationships between injury and exposure, they were able to demonstrate that the difference between sites with and without injury had more to do with site moisture conditions, or the combination of site moisture and ozone exposure, than with ozone exposure alone. No trend data were reported. However, the findings reported here suggest that the percent injured biosites has been declining steadily since 2003.

Northern Research Station Study Area—Differences in calculated mean values for percent injured biosites and average BI suggest that ozone stress is highest in the mid-Atlantic States, similarly moderate in the east North Central States and southern New England, and relatively low in northern New England and the Northern Plains States. Region-wide trends suggest that ozone injury is declining possibly in response to a decline in peak ozone concentrations in normally high ozone areas. Biosites with injury occur at all SUM06 and N100 exposures, but when SUM06 and N100 are relatively low, the percentage of uninjured sites (BI=0) is much greater than the percentage of injured sites (BI>0); and at all SUM06 and N100 exposures, when site moisture is limiting, the percentage of uninjured sites (BI=0) is much greater than the percentage of injured sites (BI>0). These findings are in accordance with results reported by Campbell and others (2007) for western forests. They reported a general association of injury and exposure, but found that when looking at individual biosites, high levels of injury can occur in areas of low ozone exposure and low levels of injury can occur in areas of high ozone exposure.

SUMMARY AND CONCLUSIONS

Ozone has long been considered one of the most widespread and damaging air pollutants to forest health (Percy and others 2003). In addition, it acts as a greenhouse gas contributing significantly to atmospheric warming on a global scale. Campbell and others (2007) make the point that although air quality is improving in the United States as result of emission reductions, ozone standards meant to protect plant health are still being exceeded in many areas. Regionally, there are increased sources of ozone pollution as populations increase and more ozone precursor pollutants are moving into the United States from Asia via long-range transport. Ozone precursor pollutants are also expected to increase with the regional expansion of the oil and gas industry both in the Interior States and in the Northeastern United States. On a global scale, as the climate continues to warm, we can expect ground-level ozone concentrations to increase in all areas due to the fact that O_3 formation is driven, in large part, by high sunlight intensity and warm temperatures.

In this report, the ozone indicator data establish the fact that plant-damaging concentrations of O_3 are present in U.S. forests, occurring frequently, if not every year, in most States in the Northern Research Station, Southern Research Station, and Pacific Northwest Research Station regions. Region specific studies have demonstrated a general association between injury and exposure such that areas with the highest SUM06 values have the highest percent injured sites and mean BI. In Eastern forests, annual fluctuations in injury are strongly influenced by both exposure and site moisture conditions. Years of extreme drought result in a sharply reduced BI despite high ozone exposures. Trend data suggest that ozone stress is decreasing over time in all regions particularly in recent years possibly due to a national declining trend in peak ozone concentrations. This trend may reverse with the combined pressure of increasing population, increasing ozone precursor pollutants, and rising temperatures during the growing season.

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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 research projects supported by the FHM program and include enough information for readers to pursue specific interests. Three project summaries are included in this report.

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CHAPTER 9.

Oak Savanna Restoration in Central Iowa: Assessing Indicators of Forest Health for Ecological Monitoring

(PROJECT NC-F-04-02)

Heidi Asbjornsen[1]

Lars Brudvig[2]

INTRODUCTION

Savanna ecosystems were once a dominant feature of the Midwestern Corn Belt Plains ecoregion, occurring within the dynamic boundary between prairies to the west and forests to the east, and maintained in the landscape by complex interactions between fire, climate, topography, and human activities (Anderson 1998). Characterized by their continuous understory layer and widely scattered overstory trees, primarily oak species, Midwestern savannas are today extremely rare, largely converted to agricultural or transitioned to woodlands following changes to disturbance regime. Today, less than 1 percent of the original extent of savanna vegetation remains (Nuzzo 1986), mostly in a highly degraded state due to fire suppression, overgrazing, habitat fragmentation, and subsequent woody encroachment and invasion by non-savanna understory and overstory species (Anderson 1998, Gobster and others 2000).

The health of Midwestern oak savannas is of regional concern due to low rates of oak regeneration and increasing domination of the understory by shade tolerant species, both of which alter the quality, composition, structure, and ecological functions of these forested systems. Restoring native oak savanna ecosystems generally involves overstory thinning and reintroduction of fire (McCarty 1998). However, little is known about the impacts of such restoration activities on biotic and abiotic ecosystem attributes and on achieving restoration goals, and about the extent to which standard monitoring protocols, e.g., those established by the Forest Inventory and Analysis Program of the Forest Service, U.S. Department of Agriculture, are sensitive to these changes. Long-term monitoring and evaluation is necessary to better understand current forest conditions and the effects of restoration treatments to guide future management decisions.

Our research involved a replicated landscape scale experiment to restore oak savanna ecosystems at a site in central Iowa that had been encroached by shade tolerant species and transitioned into woodland vegetation. The restoration process included mechanical removal of encroaching vegetation and prescribed fire. The overall goal of our research was to complement monitoring of the health of oak savanna ecosystems by the Forest Health Montioring (FHM) Program of the Forest Service; the goal was to achieve our monitoring through the collection of process-level ecosystem indicators of restored and degraded savannas to identify sensitive indicators for long-term monitoring. The planned objectives of our project included:

- Objective 1: Assess the effects of savanna restoration on stand structure, growth, and productivity of remnant savanna oak trees.
- Objective 2: Determine patterns and success of oak seedling recruitment in response to restoration treatments.

[1]Associate Professor, University of New Hampshire, Department of Natural Resources and the Environment, 268 James Hall, Durham, NH 03824

[2]Assistant Professor, Michigan State University, Department of Plant Biology, 166 Plant Biology Building, East Lansing, MI 48824

- Objective 3: Document the response of the understory herbaceous layer to restoration treatments, particularly in terms of species composition and diversity.
- Objective 4: Assess the effects of restoration treatments on biophysical variables (e.g., light, soil moisture, and soil properties).
- Objective 5: Establish FHM Detection Monitoring plots at the savanna restoration site for comparison with process-based data.

METHODS, RESULTS, AND DISCUSSION

Site Description and Study Design

The study was conducted on eight white oak (*Quercus alba*) savanna remnants near Saylorville Lake in Des Moines, IA, ranging in size from 1.5 to 3.3 ha. Following several decades of fire suppression, these sites were encroached by non-savanna tree species (e.g., *Fraxinus americana*, *Ulmus* sp., *Ostrya virginiana*), leading to canopy gap closure. Encroaching woody vegetation was removed by mechanical treatment in 2002–04 from four randomly selected remnants. One transect was established along the length of each study site (100–200 m) for sampling of vegetation (Asbjornsen and others 2005, Brudvig and Asbjornsen 2007, Karnitz and Asbjornsen 2006). Concurrently, four FIA phase 2 plots were established at each site. Phase 2 plots were arranged linearly, to coincide with sampling transects and to fit within savanna site boundaries. FIA-based data are detailed in objective 5.

Objective 1

Assess the effects of savanna restoration on stand structure and growth and productivity of remnant savanna oak trees.

Methods—Along each 100–200 m transect, we conducted annual vegetation surveys from 2002 to 2006. We recorded species and diameter at breast height (d.b.h.) for trees and species of all samplings and shrubs. Percent cover by understory vegetation, leaf litter, bare ground, and down woody material was also recorded (Brudvig and Asbjornsen 2007). In brief, along each transect we sampled trees in contiguous 10x10-m plots, saplings in contiguous 10x4-m plots, shrubs in 3-m² circular plots located every 10 m along the transects and understory data in 1x1-m plots located every 10 m along the transects. Full sampling details are described in Brudvig and Asbjornsen (2007). To assess the growth response of the remnant savanna trees, tree cores were extracted using an increment borer from large oak trees growing in restored and encroached sites in 2009. The cores were assessed for width and year, and results from the sites compared to assess change in annual mean increment growth and change in basal area across sites (Brudvig and others 2011). We used analyses of variance (ANOVA) to determine the impacts of the removal treatment on stand structure and overstory tree response. We ran a separate ANOVA for each response variable, e.g., tree density, sapling density, etc., using site-level means in our analysis and restoration treatment as the independent variable (n=4/treatment).

Results and discussion—The savanna restoration treatment resulted in the reestablishment of the savanna structure comprised of overstory oak trees at relatively low density, reducing canopy cover from 84 to 89 percent to 8 to 52 percent, while stem densities for smaller size classes (<40 cm) were also reduced. Nevertheless, the understory was dominated by advanced regeneration of shade tolerant tree species, suggesting that encroached savannas represent an alternative stable state. Thus, in addition to understory removal, management interventions including prescribed fire will likely be needed to establish the understory herbaceous layer (Brudvig and Asbjornsen 2007). The dendrochronology assessment of growth response revealed that basal area of overstory oak trees increased by 59 percent following removal of encroaching vegetation. These results suggest that encroaching trees compete directly with savanna trees for key resources thereby reducing growth rates, but that even after long periods of suppressed growth, these savanna oaks have the potential to respond to release from competition through accelerated growth (Brudvig and others 2011).

Conclusion—Removal of encroaching vegetation from degraded savanna ecosystems is an effective approach for restoring savanna overstory structure and promoting growth of mature savanna oak trees. However, restoration of the understory herbaceous structure and composition will require additional restoration interventions such as prescribed fire. Without such interventions, these savannas will likely transition back to the alternative stable woodland state consisting of intercanopy gaps filled with non-savanna woody vegetation.

Objective 2

Determine patterns and success of oak seedling recruitment in response to restoration treatments.

Methods—Along each transect, we annually surveyed from 2002 to 2006 all saplings within 4-m wide belts, all shrubs within 3-m^2 plots every 10 m along the transect, and all seedlings in 1-m^2 plots every 10 m along the transect (Brudvig and Asbjornsen 2007). In addition, 10 “canopy” and 10 “canopy-gap” plots were established within each of the 8 study sites, but outside the main 100–200-m sampling transects. All canopy and canopy-gap plots were annually surveyed from 2002 to 2006 for *Q. alba* seedlings in the year before and for 3 subsequent years after restoration, by recording height, basal diameter, and number of leaves, after the removal treatment (Brudvig and Asbjornsen 2008). Finally, we transplanted *Q. alba* seedlings every meter along transects radiating from overstory *Q. alba* trees toward inter-canopy gaps (5-6 seedlings/transect), as well as seedlings in inter-canopy gaps. For each seedling, we collected data on basal diameter, height, number of leaves, herbivory, and survival over a 2-year period (Brudvig and Asbjornsen 2009a).

Results and discussion—Following the removal treatment, seedlings of *Q. alba* exhibited a gradual increase in abundance over the 3-year post-treatment measurement periods. In contrast, seedlings of other species (e.g., *Ostrya virginiana, Fagus americana, Ulmus americana, Prunus serotina, U. rubra*) did not vary in abundance after 3 years. We also observed a recruitment pulse in shrub density 2 years and sapling density 3 years after removal of encroaching vegetation, primary attributed to vigorous stump sprouting. Thus, regeneration is dominated by encroaching species shortly after removal treatments, providing evidence for the existence of an alternative woodland stable state resulting from the savanna encroachment process (Brudvig and Asbjornsen 2007). However, *Q. alba* seedlings growing in canopy and canopy-gap locations exhibited clear differences, with canopy-gap seedlings displaying greater survival, as well as increases in height, basal diameter, and number of leaves relative to canopy (control) sites. These findings suggest that removal of woody encroachment can have positive impact on promoting regeneration of *Q. alba*, a critical component of ensuring the recruitment of young oaks into the canopy over longer time scales (Brudvig and Asbjornsen 2008). Growth and survival of transplanted seedlings increased with distance from overstory trees and were greatest in the gap areas of restored sites (Brudvig and Asbjornsen, 2009a).

Conclusion—Removal of encroaching vegetation from degraded savannas leads to rapid growth response in understory shade-tolerant (non-savanna species) shrubs and saplings, while at the same time creating gap environments that are more favorable to the establishment and growth of desirable *Q. alba* seedlings. Further work with prescribed fire and/or grazing may elucidate to what extent tree-herbaceous understory dynamics may be restored through restoration interventions in Midwestern oak savannas.

Objective 3

Document the response of the understory herbaceous layer to restoration treatments, particularly in terms of species composition and diversity.

Methods—We annually surveyed understory species composition and abundance in 1-m^2 plots located every 10 m along each transect from 2002 to 2006 (Brudvig 2010). With these data, we calculated species richness (number of species), Simpson's diversity, and species evenness using standard protocols (Magurran 2004) at the local (1x1m) and site (sum of 1x1-m plots/site) scales. We subsequently calculated beta richness and Simpson's diversity as the difference between site and local scale values (Brudvig 2010).

Results and discussion—Following the removal treatment, understory species richness and Simpson's diversity increased at local and site scales. Species evenness and beta diversity and richness (indicators of spatial turnover) were unaffected. These changes were due to increased richness and cover of graminoids and woody species following encroachment removal. Restoration promoted savanna indicator species, as well as non-savanna species, including exotic species, at local and site scales.

Conclusion—Restoration by woody encroachment removal resulted in establishment and proliferation of savanna and non-savanna understory species. Future work might investigate the long-term effects of reintroduction of characteristic savanna understory species (not colonizing naturally following restoration) and prescribed understory fire on richness and cover of woody, exotic, and other non-savanna understory species.

Objective 4

Assess the effects of restoration treatments on biophysical variables, e.g., light, soil moisture, soil properties.

Methods—At each site, we randomly selected five large, open-grown *Q. alba* trees, and established a randomly oriented transect radiating from the bole to 1.5 times the distance to the canopy edge. Along each transect, we established five to six 1x1-m "understory" plots. Similarly, 5-6 "gap" plots were established at three times the distance to the canopy edge. Between July 2004 and August 2006, we sampled the plots for vegetation, light (hemispherical photography), soil physical (texture) and chemical (pH, percent organic matter, concentrations of nitrate N, total P, and K) properties, and soil moisture.

Results and discussion—The restoration treatment of removing encroaching vegetation significantly altered biophysical gradients relative to the control sites. Restored sites exhibited a strong relationship between light and distance from overstory trees. Restored sites also had greater variability in soil moisture due to both higher levels immediately after rain and greater drying rates. With restoration, a positive relationship occurred between understory vegetation cover and distance from overstory trees, while species richness increased with distance from overstory trees in the final year. In contrast, there was little evidence for spatial patterns of soil nutrients, and more long-term monitoring may be needed to fully understand restoration impacts on savanna soil resource patterns. Common understory species were correlated with gradients of canopy cover and soil moisture associated with restoration plots, as well as with gradients of soil texture and N associated with both restoration and control plots. These findings suggest that an important consequence of removal of encroaching vegetation is the conversion of a homogenized biophysical environment common to encroached savannas to more diverse patterns of environmental gradients typical of intact healthy savannas (Brudvig and Asbjornsen 2009b).

Conclusion—Despite decades of degradation as a result of fire suppression and understory encroachment, Midwestern oak savannas maintain high resiliency that enables them to respond positively to restoration inventions.

The reestablishment of biophysical gradients in the understory environment, particularly related to light and moisture during the initial years following removal of encroaching vegetation, is a key aspect of promoting diversity and composition of understory plant species as part of the savanna restoration process.

Objective 5

Establish FHM Detection Monitoring plots at the savanna restoration site for comparison with process-based data.

Methods—FIA phase 2 plots were surveyed in 2002 and 2004 at two control and two savanna restoration sites, and four FIA plots were surveyed in 2006 and 2008 at four control and four savanna restoration sites, (two additional FIA plots were established in 2006). In each year, woody species were recorded using standard FIA methodology for seedling, sapling, and tree size classes. We analyzed these data with repeated measures ANOVA, and these results were compared to data derived from the transect-based sampling methodology (Brudvig and Asbjornsen 2007; described above in objectives 1 and 2).

Results and discussion—FIA sampling conducted over the course of the study (2002–08) illustrated patterns of reduced sapling and overstory tree densities and increased tree seedling densities following woody encroachment removal; however, replication was too low (n=2) to resolve these differences statistically (fig. 9.1). Conversely,

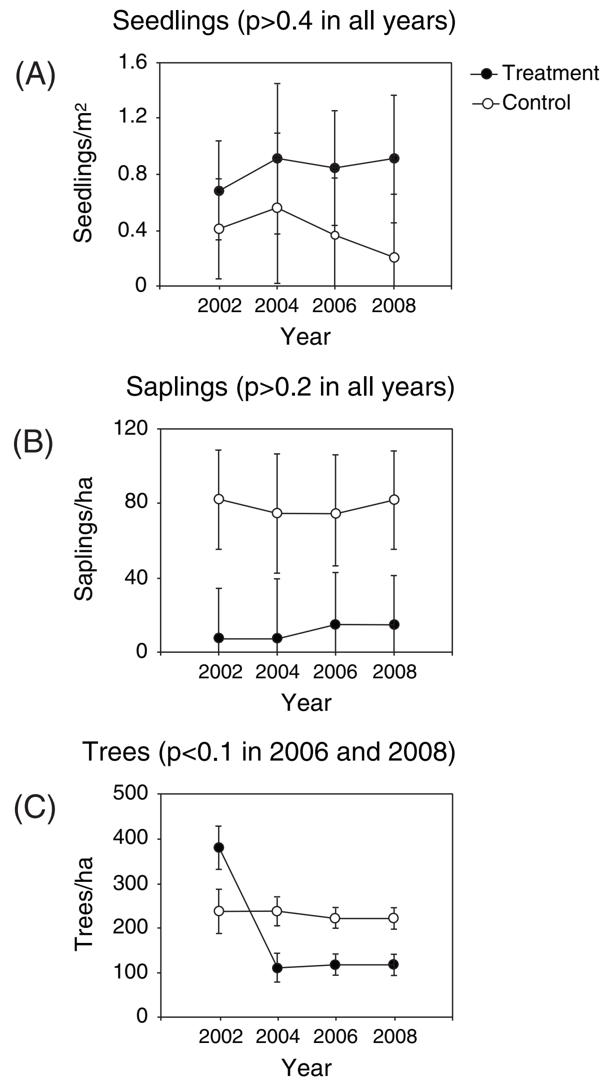

Figure 9.1—Effects of restoration (mechanical woody encroachment removal, conducted during 2002–04) on oak savanna stand structure: density of (A) woody species seedlings, (B) saplings, and (C) overstory trees. Data were collected using four Forest Inventory and Analysis phase 2 plots/site (n=2 sites/treatment). Replication was too low to statistically resolve patterns. Values are mean ±SE.

FIA sampling during 2006 and 2008, with increased replication (n=4) was able to resolve these differences: increased seedling density and reduced sapling and tree density following restoration by woody encroachment removal (fig. 9.2). In general, these FIA derived data mirrored results of data derived from transect-based sampling, though it is difficult to draw any strong conclusions regarding the sensitivity of FIA phase 2 plots to temporal change, due to low sample size. For example, with n=2 phase 2 plots sampled every other year, we were unable to resolve the sapling recruitment pulse that was evident through the transect-based data.

Conclusion—Data from FIA phase 2 plots effectively documented coarse patterns in stand structure following oak savanna restoration, e.g., major reduction in overstory density, but were ineffective at resolving finer scale changes in stand structure following restoration, e.g., temporal changes and sapling recruitment pulse. This was likely due to low replication and it is possible that these changes would have been resolved with annual sampling at full (n=4) replication. Finally, standard FIA plot layout was not useful for our study sites, as sites were not wide enough to accommodate normal phase 2 plot arrangement. As such, rearrangement of subplots to fit within our sites was necessary.

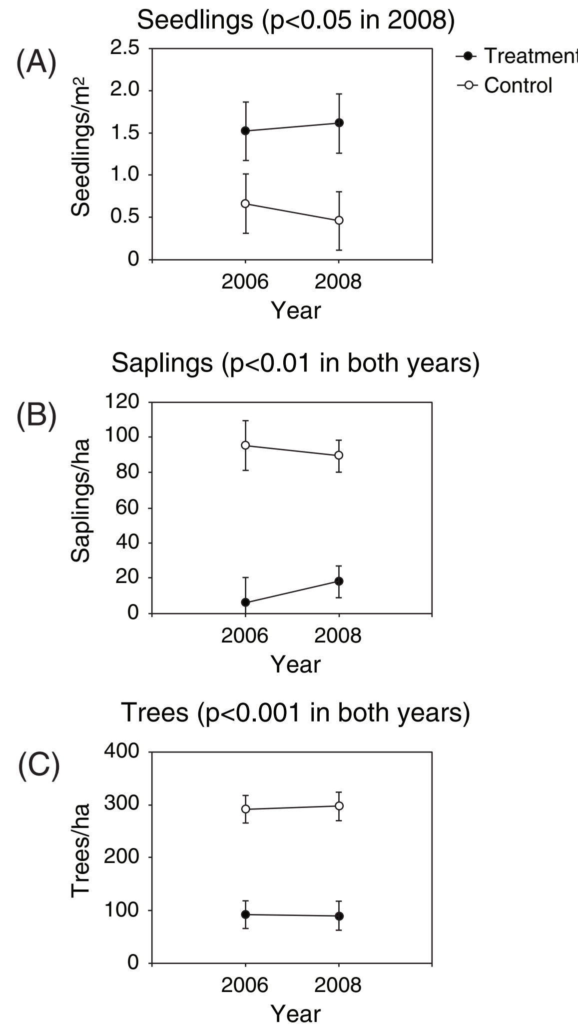

Figure 9.2—Effects of restoration (mechanical woody encroachment removal, conducted during 2002–04) on oak savanna stand structure: density of (A) woody species seedlings, (B) saplings, and (C) overstory trees. Data were collected using four Forest Inventory and Analysis phase 2 plots/site (n=4 sites/treatment). This level of replication was sufficient for resolving differences between treatment groups for all strata in 2008 (p<0.05). Values are mean ±SE.

CONTACT INFORMATION

Heidi Asbjornsen: Email: heidi.asbjornsen@unh.edu; Telephone: 603-862-1011.

Lars Brudvig: Email: brudvig@msu.edu; Telephone: 517-355-8262.

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CHAPTER 10.

Bacterial Leaf Scorch Distribution and Isothermal Lines

(PROJECT NC-EM-08-02)

Gerard C. Adams[1]
Mursel Catall[1]
James Walla[2]
Ann B. Gould[3]

INTRODUCTION

Bacterial leaf scorch (BLS) of shade trees is the common name for a disease caused by *Xylella fastidiosa*, a xylem-inhabiting bacterium that has fastidious nutritional requirements and is difficult to culture or verify by culturing. Forest trees including oak, sycamore, elm, planetree, sweetgum, mulberry and maple are species susceptible to *Xylella* infection (McElrone and others 1999) throughout the Eastern and Southeastern United States. It is not yet known how common and widespread BLS is in trees in the North Central States (Iowa, Illinois, Indiana, Michigan, Minnesota, Missouri, and Wisconsin) and Plains States (Kansas, Nebraska, North Dakota, and South Dakota). In New Jersey, BLS was first detected in populations of trees in the red oak group in several western counties 20 years ago and since has spread throughout the State, affecting as many as 44 percent of susceptible oaks in some communities (New Jersey Forest Service 2002). Population increases of *X. fastidiosa*, production of unidentified toxins (Hayward and Mariano 1997), xanthan-like gums, and biofilms in vessel elements lead to water stress symptoms (Simpson and others 2000), especially chlorosis followed by necrosis of leaf margins and interveinal areas, leaf curling, decreased seed production, delayed budbreak, early autumn dormancy, decline, dieback, and sometimes mortality (Barnard and others 1988, Lashomb and others 2002). Increasing incidence and distribution of BLS combined with drought will increase decline and mortality in susceptible hardwoods. Moisture stress increases the expression of symptoms of BLS. *Xylella* is vectored by various insects in the Homoptera family including sharpshooter leafhoppers and spittlebugs (Pooler and others 1997). Introduction of new vectors that are more efficient in transmitting the pathogen can increase the economic damage caused by the disease as occurred in California when the glassy-winged sharpshooter increased the incidence of the *X. fastidiosa*-induced disease, Pierce's disease, which has been threatening the grape crop. *X. fastidiosa* occurs in numerous strains which have only recently been well distinguished (Qin and others 2001). One strain causes citrus variegated chlorosis (CVC), a disease infecting citrus trees. Currently in Brazil about 5 million diseased trees are destroyed yearly, causing approximately $50 million in losses. Quarantines are in force in the United States to prevent introduction of the citrus strain. The regional strains of *X. fastidiosa* in forests and amenity shade trees of the North Central and Plains States do not appear to cause severe disease symptoms like those infecting grape and citrus.

Leaf scorch in trees can be caused by numerous unidentified abiotic causes as well as by the bacterial pathogen. A regional survey using detection of the bacterial pathogen provides a worthwhile evaluation of the proportion of scorch that can be attributed to the pathogen and the relative proportion attributable to environmental or unknown causes. This information is important to

[1] Associate Professor and Ph.D. (Plant Pathology), Dept. Plant Pathology, Michigan State University, East Lansing, MI 48824-1311.

[2] Research Scientist, Dept. Plant Pathology, North Dakota State University, Fargo, ND 58105-5075.

[3] Associate Extension Specialist, Dept. Plant Biology and Pathology, Rutgers University, New Brunswick, NJ 08901-8520

improving understanding of the causes of stress and decline in trees, particularly those stresses caused by unsuitable planting practices, and problems in urban soils and sites.

Having a measure of the incidence and distribution of BLS in the North Central and Plains States is worthwhile for establishing a baseline of current conditions. With a record of conditions, the influence of new or more effective vectors or of changes in climate warming can be documented more accurately. Because BLS is a factor in decline of trees, changes in distribution and incidence will impact forest health. If the changes are due to warming climate, decline in important forest species such as oaks and maples could be modeled and future trends in forest health could be predicted. Additionally, the host range of BLS in hardwoods and other woody plants is not yet well known, or known only for limited regions of the United States (McElrone and others 1999).

The planned objectives of this 2-year project included:

Objective 1: Determine the incidence of BLS in species of *Quercus, Acer, Platanus, Ulmus, Morus, Tilia* and other hardwoods in the 11 North Central and Plains States.

Objective 2: Determine the distribution of BLS in the 11 North Central and Plains States.

Objective 3: Relate the occurrence of BLS to mapped landscape-scale physiographic, edaphic, and climatic data.

METHODS

Organization of the project began with conference calls and a Web site initiated by a specialist[4] in the Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture. Conference calls included State foresters who had previously worked in forest health monitoring. Once procedures were agreed upon, the methods were posted on the Web site with description of the project and information on the etiology of the disease and illustrations of the symptoms. Additional volunteers were solicited by direct communication and by outreach and Extension articles. Volunteers included State Department of Agriculture employees, University Extension agents, State Department of Natural Resources employees, and private arborists, landscapers, and master gardeners. Cooperators and volunteers were advised to locate trees in their region of the 11 States showing leaf scorch during late July through October in 2008 and 2009. It was planned to sample urban and rural trees, and trees in forest stands. Sampling design was to sample from every symptomatic tree seen by the individual collector with a goal of obtaining 30 trees per year for each State by the total collectors in the State. Because of the scarcity of symptomatic trees, sampling was random for habitat, size, species and number. In some instances leaf scorch was evident by May,

[4] Manfred Mielke, Plant Pathologist and Forest Health Monitoring Northcentral Region Program Manager, 1992 Folwell Ave., St. Paul, MN 55108; mmielke@fs.fed.us.

but samples were collected usually in August to allow the titer of the pathogenic bacteria to increase in the samples ("titer" refers to the detectable concentration of the pathogen in the host tissue). The sampling protocol was to collect two samples of scorched leaves per tree, old scorched leaves from one side of the tree and younger scorched leaves from the opposite side. Each sample was to include at least two to five leaves with symptoms attached to a shoot (approximately pencil width). After review of the number of samples and diversity of species collected in each State, the first year, successive year shoots and leaves were to be double-bagged in a self-addressed stamped Tyvek® envelope for shipment. Approximate tree locations were recorded by GPS coordinates or street addresses. Diameter at breast height was approximated or measured and percentage of crown showing scorch was recorded. Information was recorded on or in the envelopes and the exterior of the envelopes was pre-stamped with the categories of information requested from the collector. The requested information included: name and address of the collector, collection date, State, county, city, and street address location information, GPS coordinates indicating datum and format used, genus and species of the host, stem diameter size class (d.b.h.) estimated categories of sapling <5 inches (12.5 cm), pole 5-11 inches (12.5–27.5 cm), or large >11 inches (27.5 cm). Crown symptoms of percent of foliage affected by scorch symptoms, and percent dieback were recorded. For dieback, three approximated categories were used: low (<5 percent), moderate (5-20 percent), and high (>20 percent).

Envelopes were shipped the same day, or stored in a refrigerator at approximately 4 °C until shipment. Many leaf samples were photographed to record differences in scorch symptoms. Then, samples were processed to extract DNA from petiole xylem tissues. Quality of DNA, and presence and quantity of *X. fastidiosa* DNA was determined using machinery and reagents of the quantitative polymerase chain reaction DNA amplification methodology (real-time PCR or qPCR). Two standard protocols were used: the SYBR® Green protocol (Applied BioSystems; used in the Adams laboratory) or the TaqMan® protocol (Applied BioSystems; used in the Gould laboratory) with *X. fastidiosa* specific primers (Schaad and others 2002). For several samples, presence of *X. fastidiosa* also was determined with commercial enzyme linked immunosorbent assay (ELISA) technology (Agdia, Inc.) and also verified by qPCR. Three replicates of each sample were tested per assay. Detection sensitivity of qPCR methods ranged from 13.2 to 13,200,000 cells/mL. Quality assurance was checked and verified by sending a dozen or more samples between the two laboratories for comparison in double-blind assays (samples labeled and results matched to samples by a noninvolved student worker).

Positive and negative trees were mapped for distribution using MapSource® software (Garmin Ltd.), Google Maps®, and Google earth®. Climatic and physiographic isotherm lines were obtained from USDA Plant Hardiness zone maps. MapSource® distribution patterns were overlaid on black and white diagrams of the Forest Service divisions, the North Central and Plains States, and the 1990 and 2006 hardiness zone border lines were superimposed individually and in combination on the final maps constructed in Adobe Illustrator®, and correlations examined and discussed.

RESULTS

During 2008 to 2009, approximately 471 trees were sampled that exhibited typical symptoms of leaf scorch. The volunteers collecting the samples were skilled at distinguishing scorch symptoms from insect damage, salt burn, nutrient deficiencies, anthracnose leaf diseases, leaf spots and other problems that can be confused with leaf scorch. Most of the host species collected were *Quercus* species, primarily *Q. rubra* with 89 samples submitted. Thirty samples of *Q. palustris*, *Q. macrocarpa*, and *Vaccinium corymbosum* were collected also. In total, 69 species of trees, shrubs and vines were ultimately submitted (table 10.1). A total of 106 collections of *Acer* spp. were also submitted. Many species of maple were collected in 2009 in Michigan following an unusual abiotic event where leaf scorch suddenly appeared in many trees in some counties without apparent relationship to weather or site conditions. The plant species that were determined to be infected with *X. fastidiosa* included 11 species, *Q. imbricaria*, *Q. macrocarpa*, *Q. palustris*, *Q. rubra*, *Q. bicolor* an unidentified *Quercus* sp., unidentified *Acer* sp., *Ulmus davidiana* var. *japonica, Morus rubra, Aesculus* sp., and *Fraxinus americana* 'Rosehill.' Fifteen collections of *Tilia* spp. were submitted but none were positive for BLS.

Table 10.1—Collections (2008–09) of plants with leaf scorch symptoms assayed by real-time qPCR

Species and cultivars	Number of samples	Positive assays[a]
Acer fremanii 'Autumn Blaze'	1	No
Acer ginnala (Amur maple)	8	No
Acer negundo (Box elder)	9	No
Acer platanoides (Norway maple, inc., 'Crimson King', 'Variagated')	21	No
Acer rubrum (Red maple)	10	No
Acer saccharinum (Silver maple)	3	No
Acer saccharum (Sugar maple)	16	No
Acer tataricum (Tatarian maple)	1	No
Acer sp. unidentified	37	Yes (2)
Aesculus sp.	1	Yes (1)
Aesculus sp. Hybrid	1	No
Aesculus x carnea (Red buckeye, 'Briotii')	1	No
Aesculus glabra (Ohio buckeye)	8	No
Aesculus hippocastanum (Horse chestnut)	4	No
Aesculus octardra (Yellow buckeye)	1	No
Amelanchier alnifolia (Serviceberry)	1	No
Caragana arborescens (Siberian peashrub)	1	No
Carpinus caroliniana (American hornbeam)	1	No
Catalpa speciosa (Northern catalpa)	1	No
Celtis occidentalis (Common hackberry)	1	No
Cercis occidentalis (Western redbud)	1	No
Gymnocladus dioicus (Kentucky coffee tree)	1	No
Fraxinus americana (White ash, inc. 'Rosehill')	4	Yes (2)
Fraxinus mandshurica (Manchurian ash)	3	No

continued

Table 10.1 (continued)—Collections (2008–09) of plants with leaf scorch symptoms assayed by real-time qPCR

Species and cultivars	Number of samples	Positive assays[a]
Fraxinus pennsylvanica (Green ash)	17	No
Juglans nigra (Black walnut)	1	No
Liquidambar styraciflua (American sweetgum)	1	No
Malus spp. (Flowering crabapple)	3	No
Malus domestica	1	No
Morus rubra (Mulberry)	4	Yes (1)
Parthenocissus quinquefolia (Virginia creeper)	1	No
Philliodendron amurense (Amur corktree)	1	No
Platanus x acerifolia (London planetree)	1	No
Platanus occidentalis (Sycamore)	1	No
Populus deltoids (Eastern cottonwood)	1	No
Populus sp. Hybrid	1	No
Populus tremula 'Erecta' (Columnar poplar)	2	No
Populus tremuloides (Quaking aspen)	3	No
Prunus serotina (Cherry)	1	No
Prunus virginiana (Chokecherry)	1	No
Pyrus sp. (Pear)	2	No
Pyrus ussuriensis 'Prairie gem'	4	No
Quercus alba (White oak)	25	No
Quercus acutissima (Sawtooth oak)	1	No
Quercus bicolor (Swamp white oak)	7	Yes (1)
Quercus coccinea (Scarlet oak)	1	No
Quercus ellipsoidalis (Northern pin oak)	1	No
Quercus imbricaria (Shingle oak)	3	Yes (1)

continued

Table 10.1 (continued)—Collections (2008–09) of plants with leaf scorch symptoms assayed by real-time qPCR

Species and cultivars	Number of samples	Positive assays[a]
Quercus macrocarpa (Bur oak)	30	Yes (2)
Quercus palustris (Pin oak)	30	Yes (8)
Quercus robur Hybrid (*Q. robur* 'Fastigiata' x *Q. bicolor* 'Regal Prince')	1	No
Quercus rober Hybrid (*Q. robur* x *Q. macrocarpa*)	1	No
Quercus rubra (Northern red oak)	89	Yes (2)
Quercus velutina (Black oak)	4	No
Quercus sp. unidentified	16	Yes (3)
Salix pentandra (Laurel leaf willow)	1	No
Sorbus aucuparia (European mountain-ash)	2	No
Syringa meyeri (Korean dwarf lilac)	1	No
Syringa reticulata (Japanese tree lilac)	3	No
Syringa villosa (Late or Villous lilac)	2	No
Syringa vulgaris (Common lilac)	3	No
Tilia americana (American linden, basswood)	12	No
Tilia cordata (Little-leaf linden)	3	No
Ulmus americana (American elm)	13	No
Ulmus davidiana var. *japonica* (Japanese elm)	4	Yes (1)
Ulmus parvifolia (Lacebark elm)	1	No
Vaccinium corymbosum (highbush blueberry)	29	No
Viburnum sp. (Viburnum)	1	No
Vitis sp. (Grape)	1	No

[a] 24 BLS (*Xylella fastidiosa*) positive assays, replicated three times, from 471 scorch samples, or 5 percent positive.

Sample collection in each State depended greatly on the interest and enthusiasm of volunteers. In 2008, the volunteers were primarily foresters in the Department of Natural Resources and tree enthusiasts in the Department of Agriculture in each State. In 2009, volunteers from the landscape industry, university extension service, and master gardeners also participated. Samples were received from 14 States. Four States were outside the North Central States and the Plains States, with one sample each from Colorado and Montana, two from Oklahoma, and nine from Utah. Of the 11 planned States, the following collections numbers were received: Illinois 18, Indiana 45, Iowa 0, Kansas 29, Michigan 143, Minnesota 18, Missouri 40, Nebraska 4, North Dakota 118, South Dakota 5, and Wisconsin 45. A remarkable diversity of species was collected in North Dakota. States that had BLS affected trees included Illinois, Indiana, Kansas, Michigan, Missouri (and Oklahoma). There were insufficient samples to verify whether BLS occurred in Iowa, Nebraska, and South Dakota. However, there is a record of BLS in Nebraska on mulberry (Sinclair and Lyon 2005). BLS-positive trees were not encountered in Minnesota or North Dakota. With 118 scorch samples from North Dakota, it is unlikely that BLS occurs there. The overall mean incidence of BLS-positive trees among 471 trees and woody shrubs exhibiting typical leaf scorch symptoms in late summer or fall was 5 percent (24 plants).

In figures 10.1 and 10.2, the distribution of the collected samples is illustrated by the dots within the boundaries of each State and the occurrence and location of BLS-positive trees are illustrated by the X markers within the States. The distribution of *X. fastidiosa* has been studied primarily in regards to occurrence

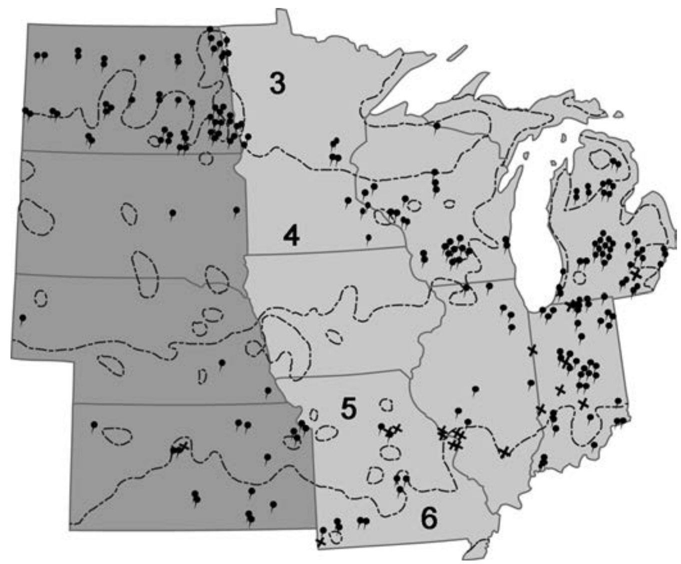

Figure 10.1—U.S. Department of Agriculture plant hardiness zone map for 1990, with dashed horizontal lines (isotherms) illustrating the boundaries of growth zones 3-7 of expected average annual minimum temperatures: Zone 3 = -35 to -40 ˚C, zone 4 = -29 to -35 ˚C, zone 5 = -23 to -29 ˚C, and zone 6 = -18 to -23 ˚C. Zones are constructed from records of lowest winter temperatures in the area in the preceding fifteen years (approximately). The locations of our scorch samples are represented as circular push pins for bacteria leaf scorch-negative trees and X-marks for BLS-positive trees. The geographical regions of the collections are shown, with Plain States (darker shading) and North Central States (lighter shading).

of Pierce's disease of grapes which is common throughout Southeastern North America and rare north of Tennesee (Anas and others 2008), although recently it has become a problem further north in Oklahoma (Smith and Dominiak-Olson 2009). BLS of hardwood trees has been commonly reported in Southern

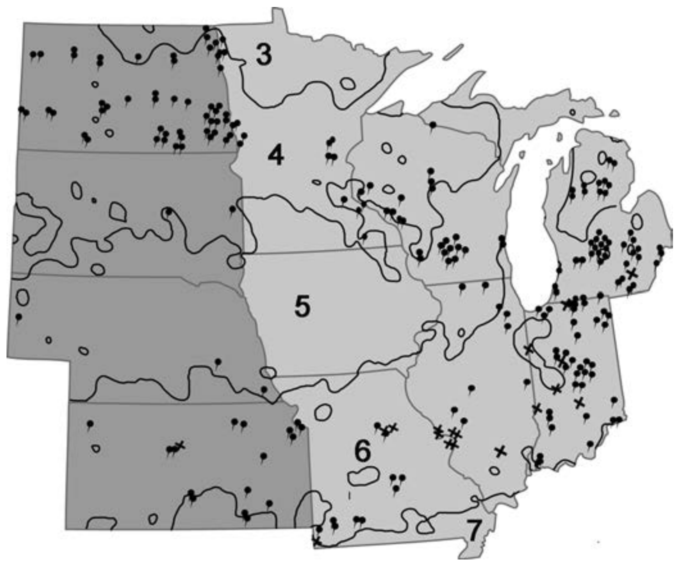

Figure 10.2—Newest U.S. Department of Agriculture plant hardiness zone map from 2006 with solid horizontal lines (isotherms) illustrating the boundaries of growth zones 3-7 of expected average annual minimum temperatures (zone 7 = -12 to -23 ˚C). Lowest winter temperatures are warming since the 1990 map; for example, the 1990 isotherms place the lower half of Michigan in zone 5, while the 2006 isotherms place the lower half in zone 6. This is a change of approximately 5 ˚C warmer. We hypothesize that bacteria leaf scorch (BLS) incidence will be increasing northward with the warming climate

and mid-Atlantic States (Purcell and Hopkins 1996). Winter temperatures with 2-3 days below -12 ˚C reduce risk of Pierce's disease (Anas and others 2008) and are believed to be detrimental to survival of *X. fastidiosa* in hardwoods. High numbers of samples and zero to low numbers of BLS-positive trees in North Dakota (0-positive), Michigan (1-positive), and Wisconsin (2-positive), may be the result of unfavorable winter weather as the northernmost States show lower incidence of BLS than the States south of Michigan. Incidence does appear to increase for more Southern States, but sample numbers, experience of collectors, and number of collectors in the field, are undoubtedly interacting with data on determining frequency.

Maps with isotherms, representing the mean lowest winter temperatures over 30 years, are not available for the States we were studying. However, a good determination of isotherms over 10-24 years is the Plant Hardiness Zones Map prepared for the U.S. Department of Agriculture (USDA). Each isotherm represents a 10 ˚F difference in the average annual minimum temperature. The isotherms from the current USDA map prepared in 2006 (The National Arbor Day Foundation 2006) are shown in figure 10.1 as solid lines and the 1990 Map isotherms as dashed lines both overlaid onto the occurrence and incidence markers for the collected plants of the 11 States. The BLS-positive trees occur in zone 6 (0 to -10 ˚F), except two trees from Wisconsin that occur in zone 5 (-10 to -20 ˚F). The warming of winter over the past decades is dramatically illustrated by the differences between the solid and hatched lines from the two zone maps on the occurrence and incidence data. An additional worthwhile comparison to construct would be to overlay the ecoregions map (Vogel and others 2005) on the occurrence and incidence data.

DISCUSSION

The survey and research were relatively successful in developing a distribution and incidence database, and a host range, as baselines for future studies. As the diagnostic tests used for detecting BLS become more frequently and widely used by the national plant diagnostic network, then geographical outliers and an expanding host range should begin to accumulate. Our knowledge of the pathogen, the diseases it causes, and the symptoms expressed in woody plants should increase considerably as the detection technology advances.

In this study, 5 percent of trees showing scorched leaves were BLS-positive out of 471 samples and 69 species. Plant Hardiness Zone 5 is the northernmost limit for BLS in this study and Nebraska (positive report in Sinclair and Lyon [2005]) is included in zone 5, as well. Zone 6 is the region where 92 percent of BLS-positive samples originated in this study. Zone 7 (10 °F to 0 °F) includes the South Central States, such as Oklahoma, where *X. fastidiosa* (Pierce's disease) occurs in grapevines. We are certain that BLS is unlikely to be present in North Dakota (zones 4 and 3) at a titer sufficient for the qPCR detection, as no BLS-positive samples were found out of 118 trees exhibiting scorch. We assume this is due to the winter cold affecting either the vectors or the trees. We are not certain, due to sample size, whether BLS occurs in Minnesota where winter temperature may also exclude it. Minnesota has territory in three hardiness zones (zones 5, 4, and 3), the warmest being zone 5. Since Wisconsin has BLS-positive trees in zone 5, it is possible that the lower third of Minnesota also may have some BLS-positive trees. The new Plant Hardiness Zones Map (The National Arbor Day Foundation 2006) readily demonstrates the gradual warming of the continent over the past few decades and shows that the plants' northern ranges are extending. We hypothesize that this extension is, and will be, effecting the distribution and incidence of BLS. Additionally, the incidence and distribution of BLS might be affected by the local variations of moisture, soil type, microclimates, winds, and other conditions affecting plant growth and health.

Improvement in the methodology for detection of *Xylella* is needed for trees with low titers of the bacterium, particularly trees in the northern States. Collections during 2008-2009 found two red oak trees in Wisconsin that were BLS-positive, however, detection was erratic sometimes giving positive assays and other times negative assays. Double-blind tests were conducted with Rutgers University on these samples using petioles (unprocessed) and DNA (processed) samples so that precision in extraction methods and assay sensitivity could independently be compared. The double-blind tests revealed that the samples (unprocessed or processed) had titers at the limit of detection by the current technology. The limit of detection is at approximately one cell of the pathogen in the volume of processed extraction applied to the assay. The results revealed that the quality control was successful. The results also uncovered the possibility that the current assays may be failing to detect infections in the northern regions due to problems of low titer. One hypothesis needing testing is that many northern hardwoods may harbor the pathogen at titers below current detection. If this is the case, then increasing incidence of BLS with increasingly warm climate will be due to a buildup of titer, rather than advancing movement of vectors.

To improve our understanding of the epidemiology of BLS, more accurate detection methods should be developed. Because qPCR detection methods are already as sensitive as one cell per sample, increasing sensitivity by concentrating samples to increase pathogen cell numbers is a reasonable approach. Higher titers may exist in roots in northern climates and further research is warranted to verify this issue. Processing larger samples would yield more target for the pathogen but also increase competing host material which may or may not inhibit or mask detection. Two approaches to this potential problem would be separating pathogen target from host material or selectively increasing pathogen target while decreasing or subtracting host material. Processing greater masses of petiole tissue can be readily accomplished with reasonable numbers of samples. To separate pathogen target DNA from host DNA, pulse field gel electrophoresis (PFGE) (Qin and others 2001) or cesium chloride density centrifugation (Tran-Nguyen and Gibb 2007) have been successfully employed. A method of increasing the pathogen target while decreasing the amount of host competing target known as suppression subtraction hybridization (SSH) (Cimerman and others 2006) has been developed and successfully employed in similar pathogen-host extractions.

Diagnosis and detection by ELISA is quick and relatively sensitive, and development of qPCR has improved sensitivity tenfold. However, to economically process samples for qPCR assays, numerous (10 to 20) samples need to be loaded on the machine at one time. Waiting for sufficient samples to be received, however, delays diagnosis, which aggravates cooperators and discourages public participation. The solution is to combine the two assays, utilizing the less sensitive ELISA technique for expediting diagnosis and utilizing the more sensitive qPCR technique for increased accuracy.

CONCLUSIONS

In this study, 5 percent of trees showing scorched leaves were BLS-positive out of 471 samples and 69 species. Plant Hardiness Zone 5 is the northernmost isotherm for BLS in this study, and Nebraska [positive report in Sinclair and Lyon 2005)] is included in zone 5, as well. The isotherm lines delimiting zone 6 encompassed 92 percent of BLS-positive samples. Zone 7 (10 ˚F to 0 ˚F) includes the southern central States, such as Oklahoma, where *X. fastidiosa* occurs in grapevines.

CONTACT INFORMATION

Gerard C. Adams: e-mail gadams@msu.edu

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CHAPTER 11.

A Test of the Forest Inventory and Analysis Program's Down Woody Material Indicator for Regional Fuel Estimation in the Southern Appalachian Mountains

(PROJECT SO-F-06-01)

G. Geoff Wang[1]

Zhi-Ping Wang[1]

Aaron D. Stottlemyer[2]

Thomas A. Waldrop[3]

INTRODUCTION

Both the National Fire Plan (http://199.134.225.50/nwcc/t2_wa4/pdf/RuralAssistance.pdf) and the Healthy Forest Initiative (http://www.fs.fed.us/projects/hfi/2003/august/documents/hfi-fact-sheet.pdf) call for reduction of hazardous fuels. Consequently, estimations of forest fuel loading at various scales become necessary. The Forest Inventory and Analysis (FIA) Program of the Forest Service, U.S. Department of Agriculture, is currently sampling down woody materials (DWM) at its phase 3 plots at the intensity of one plot every 96,000 acres. In this study, DWM is defined as a collection of fine woody material (FWM) (i.e., 1-hour, 10-hour, and 100-hour fuels), coarse woody material (CWM) (i.e., 1,000-hour fuel), litter, and duff. Because multiple fuel complexes may exist at a much smaller scale (fig. 11.1), it is not clear if the FIA's program current DWM sampling intensity would produce reasonable estimations of regional fuel loading.

OBJECTIVES

The objective of our study was to test whether the fuel estimations derived from the FIA phase 3 plots capture multiple and distinct fuel complexes in the Southern Appalachian Mountains. Based on the test, the minimum sampling intensity for obtaining an adequate regional DWM estimation was suggested for the Southern Appalachian Mountains.

METHODS

The study area in the Southern Appalachian Mountains involved three national forests (Chattahoochee National Forest in northeastern Georgia, Nantahala National Forest in western North Carolina, and Sumter National Forest in northwestern South Carolina) and one national park (Great Smoky Mountains National Park in southeastern Tennessee).

Data were collected from three different sources: FIA phase 3 data, data collected by Intensive Sampling Data (Brudnak and others 2007), and new data collected in this study (New Data). The most recent FIA phase 3 plots in the studied national forests/park were acquired, with the year of sampling ranging from 2001 to 2005. Using a stratified random sampling method, Brudnak and others (2007) intensively sampled one subjectively selected 10-square-mile area at each studied national forest/park by installing 193 to 297 plots (50 × 44 feet in size) and referencing slope location and aspect. In addition to the two sources of available data described above, we conducted additional sampling in fall 2007, with 20 plots in each national forest/park. Those plots were randomly selected within each forest and park, but subject to restriction of road access. CWM, FWM, litter, duff, and shrub and herb loadings were measured in all plots using the FIA phase 3 method. Estimates of various DWM components were calculated using the equations in Chojnacky and others (2004).

[1] Professor and Former Visiting Professor, Department Forestry and Natural Resources, Clemson University, Clemson, SC 29634

[2] Instructor, Wildlife Technology Department, Pennsylvania State University DuBois, DuBois , PA 15801

[3] Supervisory Research Forester, USDA Forest Service, Southern Research Station, Clemson, SC 29634

Figure 11.1—Five common fuel complexes in Southern Appalachian Mountains: A = undisturbed stands; B = burned stand; C = harvested stand; D = southern pine beetle damaged stand; E = wind damaged stands. (Photo: USDA Forest Service.)

Biomass of DWM components is summarized using descriptive statistics. In order to determine an adequate sampling density, we calculated running averages of DWM estimates from plots that were sampled in the study. The change of DWM estimates with the increase of sampling size was visually inspected and a minimum sample size was interpreted when the estimates approached a stable value.

RESULTS AND DISCUSSION

The amounts of total DWM estimated using the FIA phase 3 data were generally much less, by 47 to 73 percent depending on study area, than those estimated using the intensively sampled data (table 11.1). When individual DWM components were compared, 1,000-hour fuel was estimated consistently and considerably lower, by 81 to 98 percent, based on the FIA phase 3 data when compared to the intensively sampled data. These discrepancies in 1,000-hour fuel or CWD appeared extremely large, which was the main reason why the intensively sampled data resulted in much higher estimates of total DWM in each forest/park. The large discrepancies in CWM, however, could not be simply attributed to low sampling intensity of FIA phase 3 plots.

Table 11.1—A comparison of the down woody material (DWM) estimates derived from Forest Inventory and Analysis phase 3 plots and the intensive sampling plots

Sampled area[a]	Method	N	1-hr	10-hr	100-hr	1000-hr	Litter	Duff	Total
			tons per acre						
Georgia	FIA[b]	6	0.22	0.79	1.50	1.08	2.06	2.17	7.81
	Intensive[c]	297	0.27	0.92	3.78	14.80	2.76	6.40	28.93
	Percent[d]		-19.99	-14.78	-60.20	-92.69	-25.54	-66.11	-73.00
North Carolina	FIA	7	0.14	0.97	1.91	5.37	4.02	7.04	19.45
	Intensive	250	0.34	0.95	3.65	28.52	2.92	6.23	42.63
	Percent		-59.45	1.37	-47.58	-81.18	37.49	12.96	-54.38
South Carolina	FIA	1	0.55	3.88	5.63	0.32	0.90	2.03	13.30
	Intensive	275	0.24	1.05	3.95	13.86	2.63	3.54	25.27
	Percent		130.55	268.64	42.54	-97.66	-65.83	-42.70	-47.35
Tennessee	FIA	9	0.28	0.86	1.48	2.16	2.45	4.18	11.41
	Intensive	193	0.38	0.90	3.77	19.48	3.20	5.05	32.78
	Percent		-26.32	-4.14	-60.78	-88.90	-23.56	-17.24	-65.20

[a] Sampled area indicates the national forest/park found in these States.
[b] FIA method can be found in Woodall and Williams (2005).
[c] Intensive sampled method can be found in Waldrop and others (2007).
[d] Percent = 100 x (FIA-Intensive)/Intensive, where the estimates using intensive data are assumed as criteria.

We could not find other apparent reasons responsible for these discrepancies. However, it is possible that each 10 square mile area selected for intensive sampling may have higher CWM than each forest/park.

In each national forest/park, the change of the running average of the total DWM with the number of plots diminished and approached a stable value before sampling size reached about 12 plots (fig. 11.2A). When considered over a large area (i.e., with the three national forests and the one national park combined), the running average of the total DWM approached a stable value with the number of sampling plots increased to about 30 plots (fig. 11.2B).

CONCLUSIONS

We found a large discrepancy between the FIA phase 3 estimates and those derived using the intensive sampling data of Brudnak and others (2007). These discrepancies are attributed to the extremely large difference in CWM estimates between the two methods, which could not be explained satisfactorily. FIA phase 3 sampling intensity (approximately one plot per

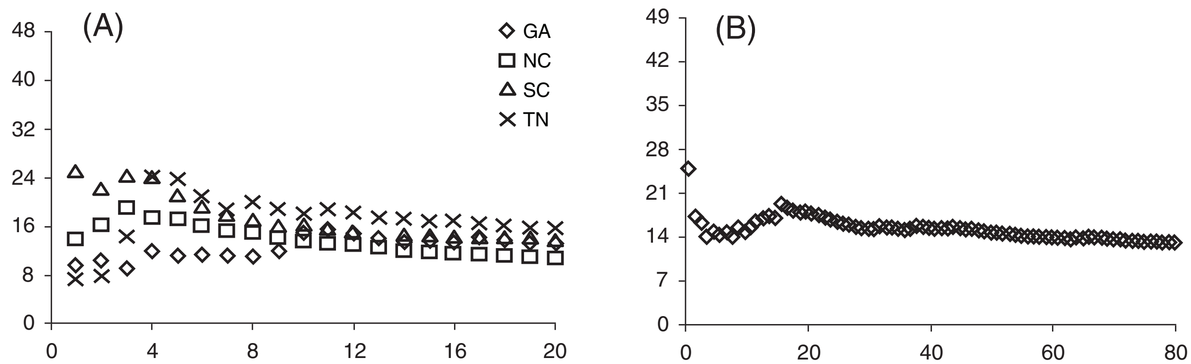

Figure 11.2—The changes in mean down woody material (DWM) weight (y-axes) with increasing number of sampling plots (x-axes) in each national forest/park (A) (diamond = Chattahooche National Forest; square = Nantahala National Forest; triangle = Sumter National Forest; cross = Great Smoky National Park) or four areas combined (B).

96,000 acres) is appropriate at a regional scale when fuel loading is averaged over a large area (>2 million acres). At a smaller scale (i.e., at individual county or individual national forest/ park scale), the FIA phase 3 sampling intensity would likely be too sparse to generate reliable fuel loading estimates.

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AUTHOR INFORMATION

SECTIONS 1 AND 2, FOREST HEALTH MONITORING RESEARCH

MARK J. AMBROSE, Research Assistant, North Carolina State University, Department of Forestry and Environmental Resources, Raleigh, NC 27695.

JOHN W. COULSTON, Research Scientist, U.S. Department of Agriculture Forest Service, Southern Research Station, Knoxville, TN 37919.

KATHERINE JOHNSON, Biologist, U.S. Department of Agriculture Forest Service, Northern Research Station, Newtown Square, PA 19073.

FRANK H. KOCH, Research Ecologist, U.S. Department of Agriculture Forest Service, Southern Research Station, Research Triangle Park, NC 27709 (formerly Research Assistant Professor, North Carolina State University, Department of Forestry and Environmental Resources, Raleigh, NC 27695).

W. KEITH MOSER, Research Forester, U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station, Flagstaff, AZ 86001.

CASSANDRA OLSON, Ecologist, U.S. Department of Agriculture Forest Service, Northern Research Station, St. Paul, MN 55108.

JEANINE L. PASCHKE, Geographic Information Systems Analyst, Sanborn, Fort Collins, CO 80526.

KEVIN M. POTTER, Research Assistant Professor, North Carolina State University, Department of Forestry and Environmental Resources, Raleigh, NC 27695.

KURT H. RIITTERS, Research Scientist, U.S. Department of Agriculture Forest Service, Southern Research Station, Eastern Forest Environmental Threat Assessment Center, Research Triangle Park, NC 27709.

BETH SCHULZ, Research Ecologist, U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station, Anchorage, AK 99503.

GRETCHEN C. SMITH, Forest Inventory and Analysis National Ozone Field Advisor, University of Massachusetts, Department of Environmental Conservation, Amherst, MA 01003.

WILLIAM D. SMITH, Research Scientist, U.S. Department of Agriculture Forest Service, Southern Research Station, Eastern Forest Environmental Threat Assessment Center, Research Triangle Park, NC 27709.

SECTION 3, EVALUATION MONITORING PROJECT SUMMARIES

Gerald C. Adams, Associate Professor, Michigan State University, Department of Plant Pathology, East Lansing, MI 48824.

Heidi Asbjornsen, Associate Professor, University of New Hampshire, Department of Natural Resources and the Environment, Durham, NH 03824.

Lars A. Brundvig, Assistant Professor, Michigan State University, Department of Plant Biology, East Lansing, MI 48824.

Mursel Catal, Ph.D. (Plant Pathology), Michigan State University, Department of Plant Pathology, East Lansing, MI 48824.

Ann B. Gould, Associate Extension Specialist, Rutgers University, Department of Plant Biology and Pathology, New Brunswick, NJ 08901.

Aaron D. Stottlemeyer, Instructor, Pennsylvania State University DuBois, Wildlife Technology Department, DuBois, PA 15801.

Thomas A. Waldrop, Supervisory Research Forester, U.S. Department of Agriculture Forest Service, Southern Research Station, Clemson, SC 29634.

James A. Walla, Research Scientist, North Dakota State University, Department of Plant Pathology, Fargo, ND 58105.

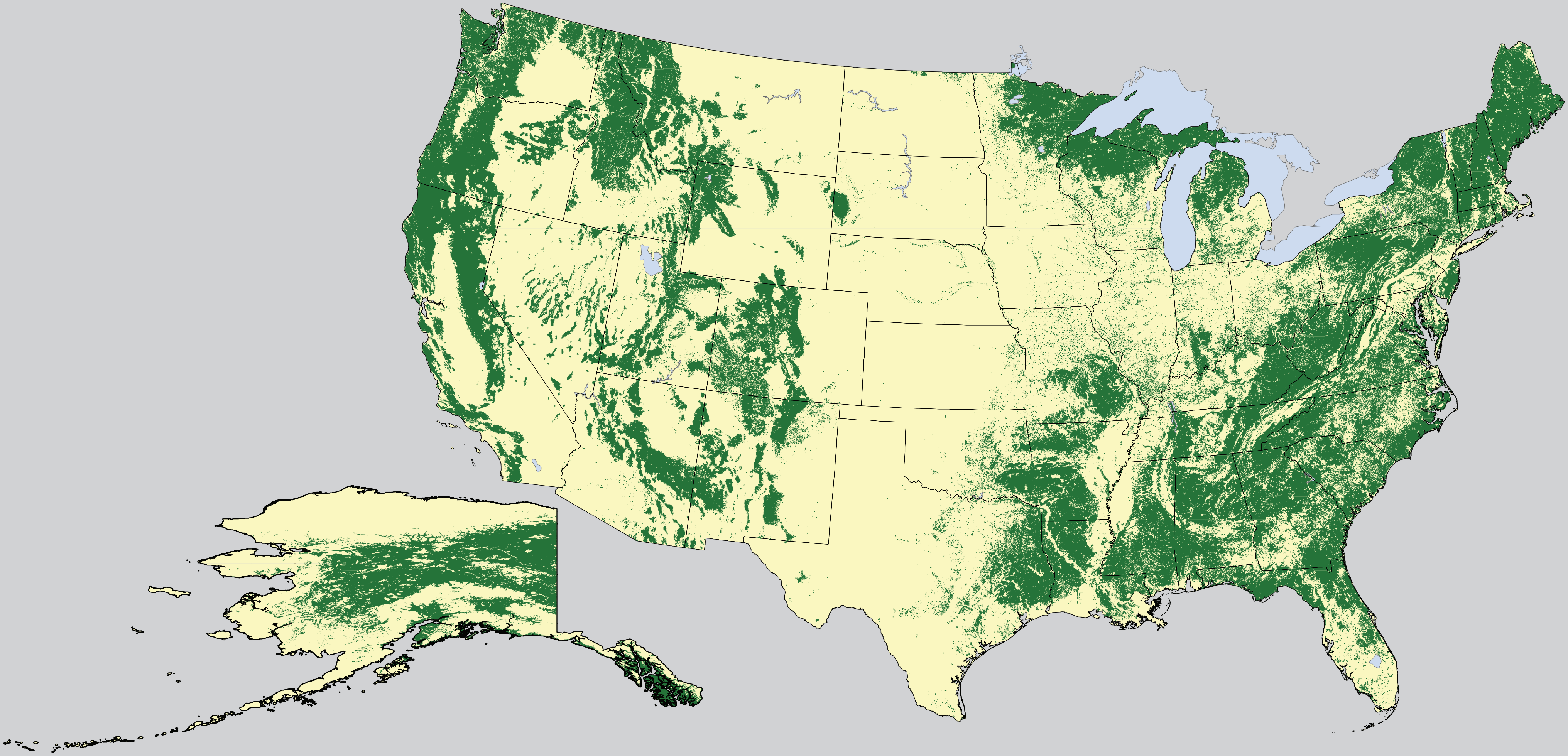
G. Geoff Wang, Professor, Clemson University, Department of Forestry and Natural Resources, Clemson, SC 29634.

Zhi-Ping Wang, former Visiting Professor, Clemson University, Department of Forestry and Natural Resources, Clemson, SC 29634.

Potter, Kevin M., Conkling, Barbara L., eds. 2013. Forest health monitoring: national status, trends, and analysis 2011. Gen. Tech. Rep. SRS-185. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 149 p.

The annual national report of the Forest Health Monitoring Program of the Forest Service, U.S. Department of Agriculture, presents forest health status and trends from a national or multi-State regional perspective using a variety of sources, introduces new techniques for analyzing forest health data, and summarizes results of recently completed Evaluation Monitoring projects funded through the national Forest Health Monitoring program. Survey data are used to identify geographic patterns of insect and disease activity. Satellite data are employed to detect geographic clusters of forest fire occurrence. Data collected by the Forest Inventory and Analysis Program of the Forest Service are employed to detect regional differences in tree mortality. Fragmentation status of forest types in the Eastern United States is evaluated and the area of intact forest is estimated by forest type. The presence and abundance of introduced plant species in the Northeastern United States are examined to determine what broad-scale factors might predict their distribution. Results from 16 years of ozone damage biomonitoring are presented, demonstrating overall declines in damage over time. Three recently completed Evaluation Monitoring projects are summarized, addressing forest health concerns at smaller scales.

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



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