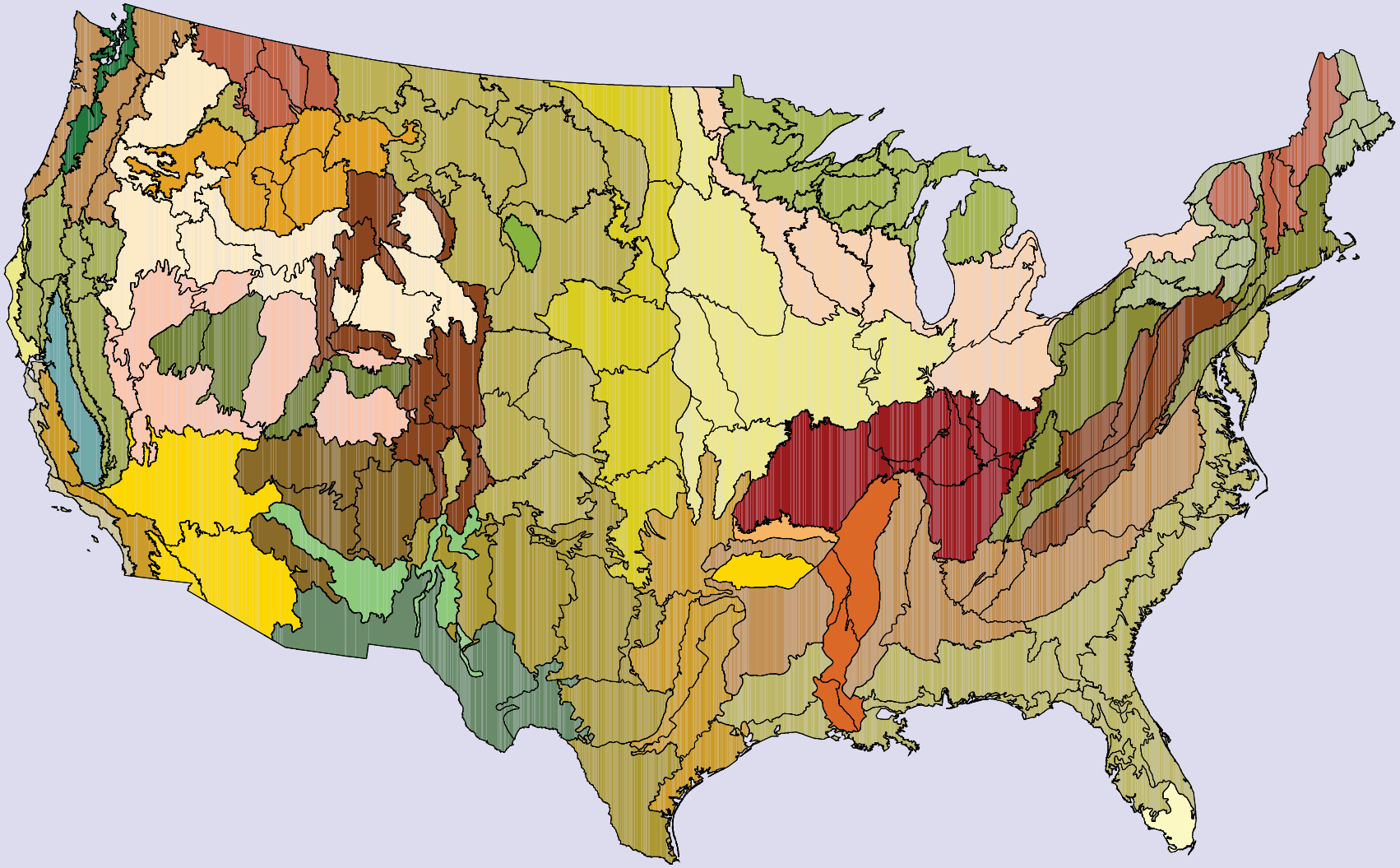
Forest Health Monitoring: 2007 National Technical Report

Barbara L. Conkling, Editor



United States
Department of
Agriculture

Forest
Service

**Southern
Research Station**

General Technical
Report SRS-147

Front cover map: Ecoregion provinces and ecoregion sections for the continental United States (Cleland and others 2005).

Back cover map: Forest land (green) backdrop derived from Advanced Very High Resolution Radiometer satellite imagery (Zhu and Evans 1994).

DISCLAIMER

The use of trade or firm names in this publication is for reader information and does not imply endorsement by the U.S. Department of Agriculture of any product or service.

December 2011

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

Forest Health Monitoring: 2007 National Technical Report

Editor

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

Abstract

The Forest Health Monitoring Program produces an annual technical report that has two main objectives. The first objective is to present information about forest health from a national perspective. The second objective is to present examples of useful techniques for analyzing forest health data new to the annual national reports and new applications of techniques formerly used. The report's organizational framework is the Criteria and Indicators for the Conservation and Sustainable Management of Temperate and Boreal Forests of the Montréal Process. Here, we present an approach to examining landscape context of forest and grassland in the United States. We explore the influence of environmental factors such as climate and air quality on a lichen species diversity indicator across the continental United States. This includes an analysis of the potential for monitoring changes in these environmental factors. We use Forest Inventory and Analysis phase 3 data to describe aspects of forest communities such as understory species composition, richness, and distribution, including discussion of invasive and introduced species. Tree mortality, which has been examined in previous Forest Health Monitoring reports, is analyzed in this report using a more intensive dataset to demonstrate the utility of Forest Inventory and Analysis phase 2 data. We explore spatial modeling of ozone injury risk, along with microscale and landscape-scale ancillary data that can be used in the modeling analyses. A discussion of redbay ambrosia beetle/laurel wilt risk includes current beetle/wilt distribution, host species distributions, climate matching, and spread modeling. Progress in monitoring and analyses of *Phytophthora ramorum* and sudden oak death is presented along with results from two different monitoring techniques.

Keywords—Air pollution, biodiversity, Criteria and Indicators, invasive insects and pathogens, landscape lichens, ozone.

Table of Contents

List of Figures v

List of Tables xi

Chapter 1. Introduction 1

Barbara L. Conkling

The Forest Health Monitoring Program ... 1
Data Sources 2
About the Report 3
Literature Cited 6

Chapter 2. The Landscape Context of Forest and Grassland in the United States 9

Kurt H. Riitters

Introduction 9
Methods 9
Results and Discussion 14
Conclusion 24
Literature Cited 24

Chapter 3. Relationship of a Lichen Species Diversity Indicator to Environmental Factors Across the Coterminous United States 25

Susan Will-Wolf, Mark J. Ambrose, and Randall S. Morin

Introduction 25
Methods 29
Results and Discussion 36
Conclusion 48
Acknowledgments 50
Literature Cited 51
Appendix 52

Chapter 4. Vegetation Diversity 65

Beth Schulz

Introduction 65
Methods 66
Results 71
Conclusions 88
Literature Cited 90
Appendix 91

Chapter 5. Tree Mortality 97

Mark J. Ambrose

Introduction 97
Data 97
Methods 97
Results and Discussion 99
Literature Cited 104

Chapter 6. Modeling Ozone Bioindicator Injury with Microscale and Landscape-Scale Explanatory Variables: A Logistic Regression Approach 107

John W. Coulston

Introduction 107
Data 108
Methods 111
Results 112
Discussion 114
Literature Cited 116

Chapter 7. Emerging Forest Pest Threat: Redbay Ambrosia Beetle and Laurel Wilt 117
Frank H. Koch and William D. Smith
Introduction 117
History of the Problem 117
Methods 119
Results and Discussion 133
Conclusions 138
Acknowledgments 139
Literature Cited 139
Appendix 141

Chapter 8. *Phytophthora ramorum* Early Detection Surveys for Forests 143
Steven Oak
Introduction 143
Methods 144
Results 145
Discussion 151
Acknowledgments 152
Literature Cited 152

Chapter 9. Summary 155
Kevin M. Potter
Literature Cited 157

Acknowledgments 158

Author Information 159

List of Figures

Figure 1.1—Ecoregion provinces and ecoregion sections for the continental United States (Cleland and others 2005). Ecoregion sections within each ecoregion province are shown in the same color 4

Figure 2.1A—The soil texture classification model identifies 12 soil textures according to the proportions of sand, silt, and clay in a soil sample. (Source: U.S. Department of Agriculture; see also http://en.wikipedia.org/wiki/Image:SoilTextureTriangle.jpg) 10

Figure 2.1B—The landcover context classification model identifies 19 landcover contexts according to the proportions of natural, developed, and agriculture landcover in a neighborhood 10

Figure 2.2A—Aggregation of 19 landcover context classes into four background classes 12

Figure 2.2B—Aggregation of 19 landcover context classes into six resource-specific classes 13

Figure 2.3—Illustration of landcover background (legend as in figure 2.2A) for all locations in the States of Colorado, Kansas, and Missouri, for neighborhood sizes of 4 ha (top), 66 ha (middle), and 5 300 ha (bottom), with inset of region immediately north of the city of Denver, CO 15

Figure 2.4—Illustration of landcover context (legend as in figure 2.2B) for forest in the States of Colorado, Kansas, and Missouri, for neighborhood sizes of 4 ha (top), 66 ha (middle), and 5 300 ha (bottom), with inset of region of the Salem Plateau in the Ozark Mountains, MO 17

Figure 2.5—Illustration of landcover context (legend as in figure 2.2B) for grassland in the States of Colorado, Kansas, and Missouri, for neighborhood sizes of 4 ha (top), 66 ha (middle), and 5 300 ha (bottom), with inset of region surrounding the city of Wichita, KS 19

Figure 2.6—The proportions of all land in each of four Resource Planning Act reporting regions (North, Pacific Coast, Rocky Mountain, South) that are within each of four landscape background classes (legend uses the same colors as shown in figure 2.2A) for neighborhood sizes of 4 ha (top left), 15 ha (top right), 66 ha (middle left), 590 ha (middle right), 5 310 ha (bottom left), and 47 800 ha (bottom right) 20

Figure 2.7—The proportions of all forest land in each of four Resource Planning Act reporting regions (North, Pacific Coast, Rocky Mountain, South) that are within each of six landscape context classes (legend uses the same colors as shown in figure 2.2B) for neighborhood sizes of 4 ha (top left), 15 ha (top right), 66 ha (middle left), 590 ha (middle right), 5 310 ha (bottom left), and 47 800 ha (bottom right) 21

Figures, cont.

Figure 2.8—The proportions of all grassland in each of four Resource Planning Act reporting regions (North, Pacific Coast, Rocky Mountain, South) that are within each of six landscape context classes (legend uses the same colors as shown in figure 2.2B) for neighborhood sizes of 4 ha (top left), 15 ha (top right), 66 ha (middle left), 590 ha (middle right), 5 310 ha (bottom left), and 47 800 ha (bottom right) 22

Figure 3.1—Summaries by State for numbers of plots (by category) included in this project. For this project West region includes all States west of the Great Plains that have lichens data (colors other than yellow) and East region includes Minnesota plus all States east of the Mississippi river that have lichens data. Both size of State and number of years sampled affect number of plots available. (Data source: U.S. Department of Agriculture, Forest Service Forest Inventory and Analysis Program) 27

Figure 3.2—Ecoregion groups, based on Bailey's ecoregion provinces (Bailey 1989; Cleland and others 2005) used for analysis (areas in gray not included). For this project West region includes all ecoregion groups west of the large gray area with no data, and East region includes all ecoregion groups east of this large gray area. Some ecoregion groups cover areas including States with no lichens data; compare with figure 3.1. Printed numbers are average for each ecoregion of Lichen S, the number of macrolichen species found per plot. Note the greater variation in average Lichen S between groups in the West. See appendix tables A3.1 and A3.3 for ecoregion provinces included in each group. (Data source: U.S. Department of Agriculture, Forest Service Forest Inventory and Analysis Program) 28

Figure 3.3—Air pollutant wet deposition (kg/ha/yr) modeled from National Atmospheric Deposition Program data. Black lines mark boundaries of ecoregion groups (see fig. 3.2). (A) NH_4^+ represents effects of agriculture; (B) NO_3^- represents urban/industrial effects. Note amounts for both are much higher in the East. (Data source: 1998-2004 average annual deposition from an interpolated 5-km^2 grid of National Atmospheric Deposition Program data [Coulston and others 2004]) 34

Figure 3.4—Scatterplots for geographic variables. (A) East – Maximum Lichen S dips at 40-43° N latitude; (B) East – Lichen S shows little relation to elevation; (C) West – Maximum Lichen S increases as latitude increases (more N); (D) West – Maximum Lichen S decreases as elevation increases. (Data sources: U.S. Department of Agriculture, Forest Service Forest Inventory and Analysis Program and the U.S. Geologic Survey) 39

Figure 3.5—Scatterplots for pollution variables. (A) East – Maximum Lichen S decreases as NO_3^- increases; (B) East – NO_3^- mostly peaks at 40-43° N latitude (A and B together explain the pattern in figure 3.4A). (C) West – Maximum Lichen S decreases as NH_4^+ increases; (D) West – Maximum NH_4^+ has a geographic pattern; it dips at longitudes corresponding to Coast, Sierra/Cascade, and Rocky Mountain ranges and peaks in lowlands. (Data sources: 1998-2004 average annual deposition from an interpolated 5-km^2 grid of National Atmospheric Deposition Program data [Coulston and others 2004], and from U.S. Department of Agriculture, Forest Service Forest Inventory and Analysis Program) 40

Figure 3.6—Scatterplots for climate variables. (A) East—Lichen S vs. precipitation; (B) East – Lichen S vs. average minimum January temperature. (C) East—Lichen S vs. average maximum July temperature. Note that in the East (A, B, C) Lichen S shows little relation to any of the climate variables. (D) West—Maximum Lichen S is limited at very low average precipitation. (E) West—Maximum Lichen S increases with average minimum January temperature; (F) West—Maximum Lichen S is lower where average maximum July temperature is either lower or higher. (Data sources: 30-year climate averages from Daly and Taylor 2000, and from U.S. Department of Agriculture, Forest Service Forest Inventory and Analysis Program) 42

Figure 4.1—Vegetation indicator plot layout ... 66

Figure 4.2—Species area and distance curves for 12 community types. Community types are identified by letters in table 4.3. Error curves represent plus and minus one standard deviation. (Data source: U.S. Forest Service Forest Inventory and Analysis Program) 74

Figure 4.3—Average species alpha for subplots in 100 percent accessible forested conditions. Plot locations are approximate. Background colors correspond to ecological provinces. States with dulled colors have not been sampled. (Data source: U.S. Forest Service Forest Inventory and Analysis Program) 77

Figure 4.4—Average plot, subplot, and quadrat alphas for 12 community types. They are average plot-black; average subplot-green; average quadrat-blue. Community types are identified by ecological province codes and forest type abbreviations shown in table 4.3. Error bars show plus and minus one standard error. (Data source: U.S. Forest Service Forest Inventory and Analysis Program) 80

Figures, cont.

Figure 4.5—Relative percent cover of introduced plant species. White dots represent plots where no introduced plants were recorded, and red dot sizes correspond to increasing levels of infestation. Plot locations are approximate. Background colors correspond to ecological provinces. States with dulled colors have not been sampled at the time of report preparation. (Data source: U.S. Forest Service Forest Inventory and Analysis Program) 83

Figure 4.6—Broad scale distributions of three invasive grass species, with percent canopy cover. Plot locations are approximate. (Data source: U.S. Forest Service Forest Inventory and Analysis Program) 85

Figure 4.7—Broad scale distributions of five introduced herb species, with percent canopy cover. Plot locations are approximate. (Data source: U.S. Forest Service Forest Inventory and Analysis Program) 86

Figure 4.8—Broad scale distributions of three introduced, invasive shrub species, with percent canopy cover. Plot locations are approximate. (Data source: U.S. Forest Service Forest Inventory and Analysis Program) 87

Figure 5.1—Forest cover in the States where mortality was analyzed. Forest cover was derived from Advanced Very High Resolution satellite imagery (Zhu and Evans 1994) 98

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 2005). (Data source: U.S. Department of Agriculture, Forest Service Forest Inventory and Analysis Program) 100

Figure 5.3—The ratio of mean dead tree diameter to mean surviving tree diameter (DDLD) on each plot at the time of its last measurement. Plot locations are approximate. (Data source: U.S. Department of Agriculture, Forest Service Forest Inventory and Analysis Program) 101

Figure 5.4—The ratio of mean dead tree diameter to mean surviving tree diameter (DDLD) on each plot at the time of its last measurement. Results are shown only for those plots on which more than 30 percent mortality (in terms of biomass) occurred. Plot locations are approximate. (Data source: U.S. Department of Agriculture, Forest Service Forest Inventory and Analysis Program) 103

Figure 6.1—Ozone biomonitoring sampling grid for the coterminous United States. The points represent the approximate center of each sampling polygon 110

Figure 6.2—The number of total ozone biosites measured and the number of ozone biosites with injury, recorded by Forest Inventory and Analysis region (IW=Interior West, NO=North, PW=Pacific Northwest, SO=South) and year. (Data source: U.S. Department of Agriculture, Forest Service Forest Inventory and Analysis Program) 112

Figure 6.3—Average area weighted SUM06 ozone exposure for each Forest Inventory and Analysis region (IW=Interior West, NO=North, PW=Pacific Northwest, SO=South) and year. Error bars represent the minimum and maximum observed SUM06 value. (Data source: U.S. Environmental Protection Agency) 113

Figure 7.1—County-level distribution of redbay (*Persea borbonia*) mortality associated with laurel wilt (*Raffaelea lauricola*) and the redbay ambrosia beetle (*Xyleborus glabratus*), by year of first detection. (Detection data sources: James Johnson, Georgia Forestry Commission; Bud Mayfield, U.S Forest Service, formerly Florida Department of Agriculture and Consumer Services, Division of Forestry; Laurie Reid, South Carolina Forestry Commission) 120

Figure 7.2—Pattern of redbay (*Persea borbonia*) mortality due to laurel wilt (*Raffaelea lauricola*) in Georgia and South Carolina based on 2006-07 survey. Plot locations are approximate. (Plot data sources: James Johnson, Georgia Forestry Commission; Laurie Reid, South Carolina Forestry Commission.) (Map created by Ed Yockey, U.S. Forest Service, Southern Research Station) 122

Figure 7.3—Number of species from the Lauraceae family occurring in U.S. counties. Distribution data: Synthesis of the North American Flora (Kartesz 2003); U.S. Department of Agriculture PLANTS Database (U.S. Department of Agriculture, Natural Resources Conservation Service 2007). Ecoregion section boundaries (Cleland and others 2007) are shown for reference. The source for background forest cover, used as a mask, was the U.S. Forest Service, Remote Sensing Applications Center 124

Figure 7.4—Redbay (*Persea borbonia*) trees per acre. Map generated through ordinary kriging of U.S. Forest Service, Forest Inventory and Analysis phase 2 plot data. Ecoregion sections (Cleland and others 2007) are shown for reference. Background forest cover source was the U.S. Forest Service, Remote Sensing Applications Center 126

Figure 7.5—Sassafras (*Sassafras albidum*) trees per acre. Map generated through ordinary kriging of U.S. Forest Service, Forest Inventory and Analysis phase 2 plot data. Ecoregion sections (Cleland and others 2007) are shown for reference. Background forest cover source was the U.S. Forest Service, Remote Sensing Applications Center 127

Figures, cont.

Figure 7.6—Potential range for redbay ambrosia beetle (*Xyleborus glabratus*) based on climate match with countries where beetle is known to be present. Matching performed in NAPPFAST software (Magarey and others 2007) using a 10-variable model (see table 7.2). Ecoregion section boundaries (Cleland and others 2007) are shown for reference. Background forest cover source was the U.S. Forest Service, Remote Sensing Applications Center, while the redbay and sassafras distribution maps were developed from U.S. Forest Service, Forest Inventory and Analysis phase 2 plot data (see figs. 7.4 and 7.5) 129

Figure 7.7—Potential extent of redbay ambrosia beetle infestation through time, assuming radial dispersion and exponential growth from three points of origin. Ecoregion section boundaries (Cleland and others 2007) are shown for reference. Background forest cover source was the U.S. Forest Service, Remote Sensing Applications Center, while the redbay and sassafras distribution maps were developed from U.S. Forest Service, Forest Inventory and Analysis phase 2 plot data (see figs. 7.4 and 7.5) 131

Figure 7.8—Complemented Weibull function used to relate host density in trees per acre to cost for the cost-weighted spread model. Black circles indicate point values that describe the step function to which this continuous function was fitted. See text for additional explanation. Residuals between step function values and predicted cost values were all within +/- 0.01 133

Figure 7A.1—Conceptual rendering of a semivariogram showing the nugget, range, and sill 141

Figure 8.1—Locations of streams surveyed in the 2007 *Phytophthora ramorum* early detection survey in forests 148

List of Tables

Table 2.1—Aggregation of the NLCD landcover legend for tripolar classification and identification of forest and grassland landcover types 11

Table 3.1—Ranges and correlations among lichen S and geographic, climate, air quality, and forest structure variables in the East region..... 31

Table 3.2—Ranges and correlations among Lichen S and geographic, climate, pollution, and forest structure variables in the West region 32

Table 3.3—Important patterns relating environmental variables to Lichen S and each other for East and West regions and ecoregion groups, summarized from tables 3.1 and 3.2, appendix tables A3.1-A3.5 with quantitative results for East groups, and appendix tables A3.6-A3.12 for West groups..... 37

Table 3.4—Summary of the potential usefulness of Lichen S as an environmental indicator for East and West regions and ecoregion groups, and suggested avenues for future research. Potential usefulness is extrapolated from results summarized in table 3.3..... 49

Table A3.1—Laurentian Forest ecoregion group in the East region..... 52

Table A3.2—Adirondacks ecoregion group in the East region 53

Table A3.3—Appalachian ecoregion group in East region..... 54

Table A3.4—Eastern Deciduous ecoregion group in the East region 55

Table A3.5—Southeastern ecoregion group in the East region 56

Table A3.6—Cascades/Coast Mountains ecoregion group in West region 57

Table A3.7—Sierras/Coast Mountains ecoregion group in the West region 58

Table A3.8—Northern Rockies ecoregion group in West region..... 59

Table A3.9—Southern Rockies/Great Basin Mountains ecoregion group in the West region..... 60

Table A3.10—Great Basin Lowlands ecoregion group in the West region 61

Table A3.11—Colorado Plateau/Southern Dry Mountains ecoregion group in the West region..... 62

Table A3.12—Arid Southwest ecoregion group in the West region..... 63

Tables, cont.

Table 4.1—Summary of datasets and type of results presented 68

Table 4.2—Identification rates and origin of species for all 915 plots 71

Table 4.3—Ecological province, forest type, identification rates, and origin of identified species for 12 community types 72

Table 4.4—Species richness and differentiation for 12 community types 79

Table 4.5—Distribution of selected native and introduced species 82

Table 4.6—Twenty-five most commonly recorded introduced species (n = 915) 84

Table A4.1—Top 5 percent most common species or genera for 12 community types 91

Table 6.1—Common and scientific names of bioindicator species 108

Table 6.2—Potential explanatory variables used in logistic regression 109

Table 6.3—Results from logistic regression model for each Forest Inventory and Analysis region 113

Table 6.4—Significance of explanatory variables for each logistic regression model and Forest Inventory and Analysis region 114

Table 7.1—Plant species from the Lauraceae family found within the conterminous United States 123

Table 7.2—Variables used in the NAPPFAST climate-matching model (Magarey and others 2007), and the minimum and maximum values for those variables found within the estimated native geographic range of the redbay ambrosia beetle 128

Table 7.3—Infestation extents calculated from the combined areas of all counties confirmed as infested by the year in question 130

Table 8.1—Summary survey statistics 2003–06 146

Table 8.2—Frequency and rank of plant genera sampled and submitted for *Phytophthora ramorum* diagnosis. Only genera ranked among the top 10 in any survey year are shown 146

Table 8.3—Summary stream baiting survey statistics, 2007 150

Table 8.4—Summary diagnostic statistics for *Phytophthora ramorum* positive streams, 2007 150

Chapter 1. Introduction

Barbara L. Conkling

The National Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture, produces an annual technical report on forest health as one of its products. The report is organized using the Criteria and Indicators for the Conservation and Sustainable Management of Temperate and Boreal Forests (Montréal Process Working Group 2007) as a framework.

This annual report has two objectives. The first is to present information about forest health from a national perspective using various indicators and ancillary data. While in depth interpretation and analysis of specific geographic or ecological regions are beyond the scope of this report, the information is presented to allow identification of areas of interest that may require a closer investigation at a smaller scale. The second objective of the report is to present examples of useful techniques for analyzing forest health data new to the national reports and new applications of techniques formerly used. Examples in this report are inchapter 5, which presents the application of techniques for assessing tree mortality using the annual inventory data collected by the Forest Inventory and Analysis (FIA) Program of the Forest Service, and chapter 6, which presents an approach for analyzing FIA phase 3 ozone bioindicator data.

The Forest Health Monitoring Program

The FHM program is a national effort to assess and report on the status and trends in forest health. The Forest Service cooperates with State forestry and agricultural agencies and other Federal agencies and universities to accomplish these tasks. The FHM Program has five major components (Forest Health Monitoring 2003):

- Detection monitoring—nationally standardized aerial and ground surveys to evaluate status and change in condition of forest ecosystems
- Evaluation monitoring—projects to determine extent, severity, and causes of undesirable changes in forest health identified through detection monitoring
- Intensive site monitoring—to enhance understanding of cause-effect relationships by linking detection monitoring to ecosystem process studies and assess specific issues, such as calcium depletion and carbon sequestration, at multiple spatial scales
- Research on monitoring techniques—to develop or improve indicators, monitoring systems, and analytical techniques,

such as urban and riparian forest health monitoring, early detection of invasive species, multivariate analyses of forest health indicators, and spatial scan statistics

- Analysis and reporting—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

The FHM program, in addition to national reporting efforts, has regional and State reporting activities. These reports may be produced with FHM's partners, both within the Forest Service and in State forestry and agricultural departments. Some examples are Keyes and others (2003), Laustsen and others (2003), Neitlich and others (2003), Steinman (2004), Snyder (2006), Lake and others (2006), Morin and others (2006), and Cumming and others (2006, 2007). The Forest Health Highlights series, available on the FHM Web site at http://fs.fed.us/foresthealth/fhm, is produced by the FHM regions in cooperation with their respective State partners. FHM and its partners also produce reports on monitoring techniques and analytical methods, such as Smith and Conkling (2004) and O'Neill and others (2005).

Data Sources

The FHM program tries to use a variety of data from sources both inside and outside the Forest Service. One major source of data is the FIA Program. The FIA Program's phase 2 is the annualized inventory measured on plots at regular intervals. FIA phase 3 plots are a subset of the phase 2 plots. Data for important ecological indicators are collected on phase 3 plots, in addition to traditional forest inventory measurements. These additional forest health indicators were measured as part of the FHM detection monitoring ground plot system prior to the 2000 survey (Palmer and others 1991).[1]

In this report, Forest Service data sources include: FHM laurel wilt survey data collected in Florida, Georgia, and South Carolina (2004–07); FHM national sudden oak death survey data (2003–06); FHM plot and lichen data (1994–99); FIA phase 3 lichen data (2000–06); FIA phase 3 vegetation diversity data (2001–04); FIA phase 3 ozone bioindicator data (2003–05); FIA annualized phase 2 survey data (1999–2006); and forest cover data developed from Moderate Resolution Imaging Spectroradiometer (MODIS) satellite imagery by the Forest Service

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

Remote Sensing Applications Center. Other data sources were: the 2001 high-resolution, national landcover map (Homer and others 2007); road map data from the Environmental Systems Research Institute (Environmental Systems Research Institute 2005); average annual rainfall, average maximum July temperature, and average minimum January temperature (Daly and Taylor 2000); annual deposition of sulfate (SO_4^{--}), nitrate (NO_3^-), and ammonium (NH_4^+) (Coulston and others 2004); national elevation dataset (U.S. Geological Survey 1999); ambient ozone data (2003–05) (U.S. Environmental Protection Agency 2004); available water capacity data (Miller and White 1998); National Oceanic and Atmospheric Administration Palmer Drought Severity Index data (2003–05) (National Climatic Data Center 1994); aspect and terrain relative moisture index data (U.S. Geological Survey 1993); population density data (U.S. Census Bureau 2004); and redbay mortality data (2006–07) (States of Georgia and South Carolina).

About the Report

We used the Santiago Declaration and accompanying Criteria and Indicators for the Conservation and Sustainable Management of Temperate and Boreal Forests that were adopted by the Forest Service as a forest sustainability assessment framework (Smith and others 2001, Forest Service, U.S. Department of Agriculture, 2004). The seven criteria as listed in the December 2007 revision (Montréal Process Working Group 2007) are:

Criterion 1—conservation of biological diversity

Criterion 2—maintenance of productive capacity of forest ecosystems

Criterion 3—maintenance of forest ecosystem health and vitality

Criterion 4—conservation and maintenance of soil and water resources

Criterion 5—maintenance of forest contribution to global carbon cycles

Criterion 6—maintenance and enhancement of long-term multiple socioeconomic benefits to meet the needs of societies

Criterion 7—legal, institutional, and economic framework for forest conservation and sustainable management

A complete evaluation of all the sustainability criteria is not appropriate in this report. Criteria 1 and 3, which are directly related to issues of forest health, are the focus.

When possible, a common ecologically based framework was used for the forest health assessments. Bailey's provinces and ecoregion sections (Bailey 1995, as revised; Cleland and others 2005, 2007) were used as the assessment units for analysis (fig. 1.1) when the spatial scale of the data and expectation of identifiable pattern in the data were appropriate for use of the ecoregion sections. This system of ecologically based units is a national hierarchical

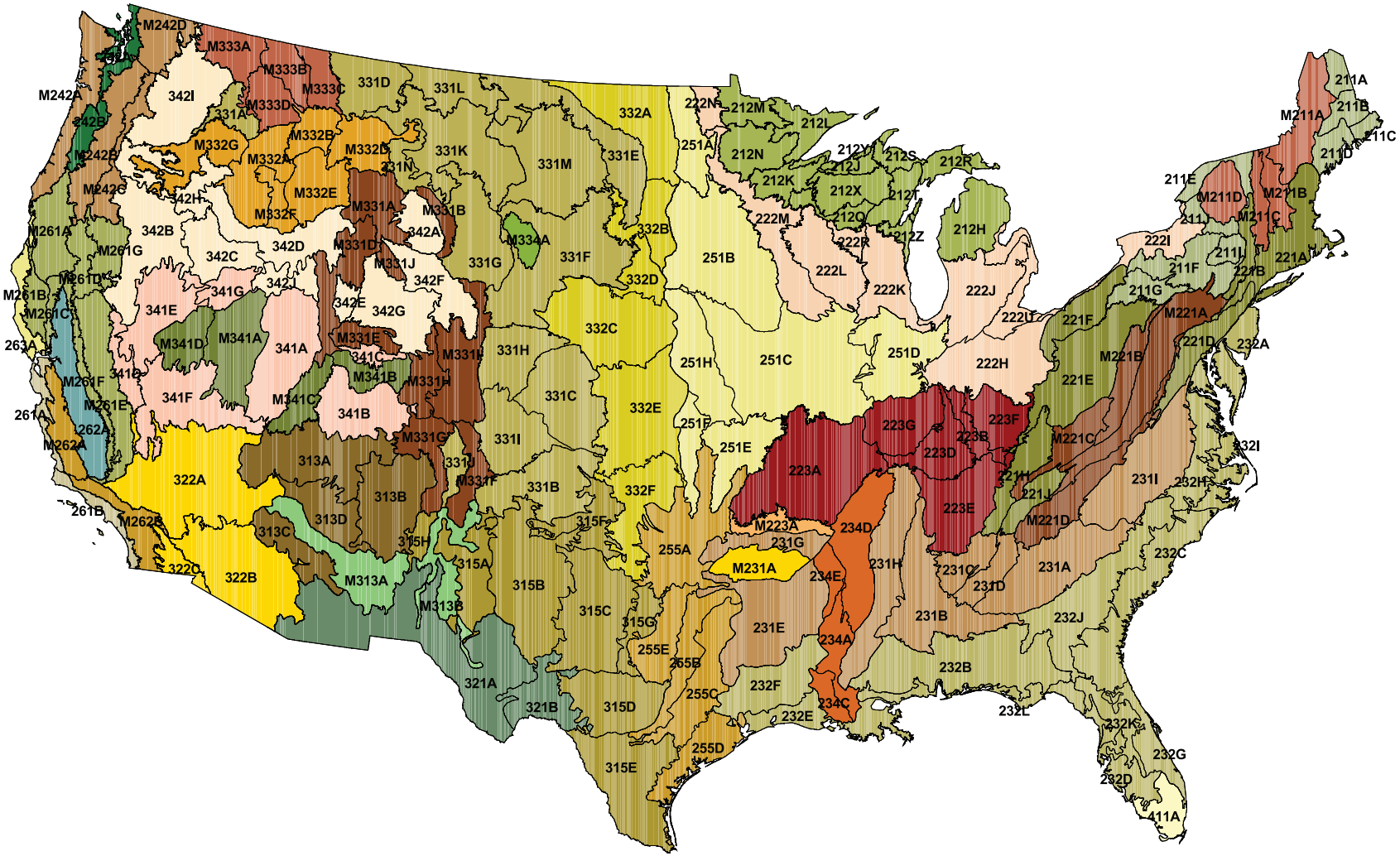


Figure 1.1—Ecoregion provinces and ecoregion sections for the continental United States (Cleland and others 2005). Ecoregion sections within each ecoregion province are shown in the same color.

Eastern ecoregion provinces

- Adirondack—New England Mixed Forest—Coniferous Forest—Alpine Meadow (M211)
- Central Appalachian Broadleaf Forest—Coniferous Forest—Meadow (M221)
- Central Interior Broadleaf Forest (223)
- Eastern Broadleaf Forest (221)
- Everglades (411)
- Laurentian Mixed Forest (212)
- Lower Mississippi Riverine Forest (234)
- Midwest Broadleaf Forest (222)
- Northeastern Mixed Forest (211)
- Ouachita Mixed Forest—Meadow (M231)
- Outer Coastal Plain Mixed Forest (232)
- Ozark Broadleaf Forest (M223)
- Prairie Parkland (Subtropical) (255)
- Prairie Parkland (Temperate) (251)
- Southeastern Mixed Forest (231)

Western ecoregion provinces

- 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 Chapparal Forest and Shrub (261)
- California Coastal Range Open Woodland—Shrub—Coniferous Forest—Meadow (M262)
- California Coastal Steppe, Mixed Forest, and Redwood Forest (263)
- California Dry Steppe (262)
- Cascade Mixed Forest—Coniferous Forest—Alpine Meadow (M242)
- Chihuahuan Semi-Desert (321)
- Colorado Plateau Semi-Desert (313)
- Great Plains—Palouse Dry Steppe (331)
- Great Plains Steppe (332)
- Intermountain Semi-Desert (342)
- Intermountain Semi-Desert and Desert (341)
- Middle Rocky Mountains Steppe—Coniferous Forest—Alpine Meadow (M332)
- Nevada—Utah Mountains—Semi-Desert—Coniferous Forest—Alpine Meadow (M341)
- Northern Rocky Mountains Forest—Steppe—Coniferous Forest—Alpine Meadow (M333)
- Pacific Lowland Mixed Forest (242)
- Sierran Steppe—Mixed Forest—Coniferous Forest—Alpine Meadow (M261)
- Southern Rocky Mountains Steppe—Open Woodland—Coniferous Forest—Alpine Meadow (M331)
- Southwest Plateau and Plains Dry Steppe and Shrub (315)

system that classifies the United States into ecoregion domains, divisions, provinces, sections, subsections, land-type associations, and land types (McNab and others 2007). Ecoregion sections may contain thousands of square miles and can be expected to have similar geology and lithology, regional climate, soils, potential natural vegetation, and potential natural communities.

Literature Cited

Bailey, R.G. 1995. Descriptions of the ecoregions of the United States. 2d ed. Misc. Publ.1391. Washington, DC: Forest Service, U.S. Department of Agriculture. 108 p. [Map scale 1:7,500,000].

Cleland, D.T.; Freeouf, J.A.; Keys, J.E. [and others]. 2007. Ecological subregions: sections and subsections for the conterminous United States. Sloan, A.M., cartog. Gen. Tech. Rep. WO-76. Washington, DC: Forest Service, U.S. Department of Agriculture. [Map, presentation scale 1:3,500,000; colored]. [Also on CD–ROM as a Geographic Information System coverage in ArcINFO format].

Cleland, D.T.; Freeouf, J.A.; Keys, J.E. [and others]. 2005. Ecological subregions: sections and subsections for the conterminous United States. June 14, 2005 version. Washington, DC: Forest Service, U.S. Department of Agriculture. [Map, presentation scale 1: 3,500,000; colored].

Coulston, J.W.; Riitters, K.H.; Smith, G.C. 2004. A preliminary assessment of Montréal process indicators of air pollution for the United States. Environmental Monitoring and Assessment. 95: 57–74.

Cumming, A.B.; Nowak, D.J.; Twardus, D.B. [and others]. 2007. Urban forests of Wisconsin: pilot monitoring project 2002. NA–FR–05–07. Newtown Square, PA: Forest Service, U.S. Department of Agriculture, Northeastern Area, State and Private Forestry. 33 p. Available online: low resolution: http://www.na.fs.fed.us/pubs/fhm/pilot/pilot_study_wisconsin_02_lr.pdf; high resolution: http://www.na.fs.fed.us/pubs/fhm/pilot/pilot_study_wisconsin2_02_hr.pdf. [Date accessed: October 19, 2007].

Cumming, A.B.; Twardus, D.B.; Smith, W.D. 2006. National forest health monitoring program, Maryland and Massachusetts street tree monitoring pilot projects. NA–FR–01–06. Newtown Square, PA: Forest Service, U.S. Department of Agriculture, Northeastern Area, State and Private Forestry. 23 p.

Daly, C.; Taylor, G. 2000. United States average monthly or annual precipitation, temperature, and relative humidity 1961-90. Arc/INFO and ArcView coverages. Corvallis, OR: Spatial Climate Analysis Service at Oregon State University (SCAS/OSU).

Environmental Systems Research Institute. 2005. ESRI data & maps: an ESRI white paper December 2005. Redlands, CA: Environmental Systems Research Institute. 41 p.

Forest Health Monitoring. 2003. Program description. Forest health monitoring fact sheet series. http://www.fhm.fs.fed.us/fact/03/prog_desc.pdf. [Date accessed: November 1, 2007].

Homer, C.; Dewitz, J.; Fry, J. [and others]. 2007. Completion of the 2001 National Land Cover Database for the conterminous United States. Photogrammetric Engineering and Remote Sensing. 73: 337–341.

Keyes, C.; Rogers, P.; LaMadeleine, L. [and others]. 2003. Utah forest health report: a baseline assessment, 1999-2001. Salt Lake City: Utah Department of Natural Resources, Division of Forestry, Fire and State Lands. 47 p.

Lake, M.; Marshall, P.; Mielke, M. [and others]. 2006. National forest health monitoring program monitoring urban forests in Indiana: pilot study 2002, part 1. Analysis of field methods and data collection. Gen. Tech. Rep. NA–FR–06–06. Newtown Square, PA: Forest Service, U.S. Department of Agriculture, Northeastern Area. http://www.fhm.fs.fed.us/pubs/ufhm/indianaforests02/indianaforests02.html. [Date accessed: November 6, 2007].

Laustsen, K.M.; Griffith, D.M.; Steinman, J.R. 2003. Fourth annual inventory report on Maine's forests. Augusta, ME: Maine Forest Service; Newtown Square, PA: Forest Service, U.S. Department of Agriculture, Northeastern Research Station and Northeastern Area. 49 p.

McNab, W.H.; Cleland, D.T.; Freeouf, J.A.; [and others] comps. 2007. Description of ecological subregions: sections of the conterminous United States [CD-ROM]. Gen. Tech. Rep. WO-76B. Washington, DC: U.S. Department of Agriculture, Forest Service. 80 p.

Miller, D.A.; White, R.A. 1998. A conterminous United States multi-layer soil characteristics data set for regional climate and hydrology modeling. Earth Interactions. 2: 1–15.

Montréal Process Working Group. 2007. The Montréal process. 3d ed. 12 p. http://www.rinya.maff.go.jp/mpci/meetings/an-6.pdf. [Date accessed: June 20, 2008].

Morin, R.S.; Liebhold, A.M.; Gottschalk, K.W.; [and others]. 2006. Analysis of forest health monitoring surveys on the Allegheny National Forest (1998-2001). Gen. Tech. Rep. NE–339. Newtown Square, PA: Forest Service, U.S. Department of Agriculture, Northeastern Research Station. 102 p. http://www.fs.fed.us/ne/newtown_square/publications/. [Date accessed: November 6, 2007].

National Climatic Data Center. 1994. Time bias corrected divisional temperature-precipitation-drought index. Doc. for dataset TD–9640. Available from: National Oceanic and Atmospheric Administration, National Climatic Data Center, Database Management Branch, Federal Building, 37 Battery Park Avenue Asheville, NC 28801–2733. 12 p. http://www.ncdc.noaa.gov/oa/climate/onlineprod/drought/readme.html. [Date accessed unknown].

Neitlich, P.; Rogers, P.; Rosentretter, R. 2003. Lichen communities indicator results from Idaho: baseline sampling. Gen. Tech. Rep. RMRS–GTR–103. Fort Collins, CO: Forest Service, U.S. Department of Agriculture, Rocky Mountain Research Station. 14 p.

O'Neill, K.P.; Amacher, M.C.; Perry, C.H. 2005. Soils as an indicator of forest health: a guide to the collection, analysis, and interpretation of soil indicator data in the forest inventory and analysis program. Gen. Tech. Rep. NC–258. St. Paul, MN: Forest Service, U.S. Department of Agriculture, North Central Research Station. 53 p.

Palmer, C.J.; Riitters, K.H.; Strickland, T. [and others]. 1991. Monitoring and research strategy for forests - environmental monitoring and assessment program (EMAP). EPA/600/4–91/012. Washington, DC: U.S. Environmental Protection Agency. 187 p.

Smith, W.B.; Vissage, J.S.; Darr, D.R.; Sheffield, R.M. 2001. Forest resources of the United States, 1997. Gen. Tech. Rep. NC–219. St. Paul, MN: Forest Service, U.S. Department of Agriculture, North Central Research Station. 191 p.

Smith, W.D.; Conkling, B.L. 2004. Analyzing forest health data. Gen. Tech. Rep. SRS–077. Asheville, NC: Forest Service, U.S. Department of Agriculture, Southern Research Station, 33 p. http://www.srs.fs.usda.gov/pubs/gtr/gtr_srs077.pdf. [Date accessed: November 6, 2007].

Snyder, C., comp. 2006. Forest health conditions in Alaska – 2005. Prot. Rep. R10–PR–5. Anchorage, AK: Forest Service, U.S. Department of Agriculture, Alaska Region; State of Alaska, Department of Natural Resources, Forestry Division. 92 p.

Steinman, J. 2004. Forest health monitoring in the Northeastern United States: disturbances and conditions during 1993-2002. NA–Tech. Pap. 01–04. Newtown Square, PA: Forest Service, U.S. Department of Agriculture, State and Private Forestry, Northeastern Area. 46 p. http://www.fhm.fs.fed.us/pubs/tp/dist_cond/dc.shtml [Date accessed: November 6, 2007].

U.S. Census Bureau. 2004. U.S. census database, 2000. Reston, VA: U.S. Geological Survey. http://www.census.gov/geo/www/gazetteer/places2k.html. [Date accessed: June 6, 2007].

Forest Service, U.S. Department of Agriculture. 2004. National report on sustainable forests – 2003. FS–766. Washington, DC. 139 p.

U.S. Environmental Protection Agency. 2004. Air quality system raw data summarization formulas. Draft version 1.1. http://www.epa.gov/ttn/airs/airsaqs/manuals/Precision%20and%20Accuracy%20Summary%20Formulas.pdf. [Date accessed: November 13, 2006].

U.S. Geological Survey. 1993. Digital elevation models—data users guide 5. Reston, VA: U.S. Geological Survey: 48.

U.S. Geological Survey. 1999. National elevation dataset. http://ned.usgs.gov/. [Date accessed: September 2006].

Chapter 2. The Landscape Context of Forest and Grassland in the United States

Kurt H. Riitters

Introduction

As development introduces competing land uses into forest and grassland landscapes, the public expresses concern for landscape patterns through headline issues such as urban sprawl and fragmentation. Resource managers need a deeper understanding of the causes and consequences of landscape patterns to know if, where, and how to take any needed actions. The spatial arrangement of the environment affects all ecological processes within that environment, and the task for resource managers is to arrange a forest or grassland in an appropriate way to provide the desired balance of biodiversity, water quality, and other amenities. National assessments of landscape patterns can help to inform those decisions by documenting the status and trends of the landscape context of natural resources.

Resource management has always considered the administrative and biophysical context within which natural resources occur. For example, forest parcels are managed according to the ecoregions in which they reside, owner objectives, road-free designations, or current human activity nearby. This chapter focuses on the landcover context of natural resources, which can indicate the type and degree of human activities. The objective is to answer two general questions about the landcover context of forest and grassland at a national scale. First, how much forest (or grassland) is contained in landscapes dominated by natural, agricultural, or urban landcover types? Second, of the forest (or grassland) contained in natural-dominated landscapes, how much resides within a potential zone of transition to more developed landscapes? High-resolution national maps provide answers to those general questions and can help local resource managers identify opportunities for protecting natural landscapes and managing future changes in transitional landscape.

Methods

Briefly, a landcover map of the conterminous United States was used to classify every 0.09-ha parcel of land according to the landcover context of its surrounding neighborhood, for several neighborhood sizes. Landcover context was classified analogously to vegetation classification, except now the labels refer to landcover types instead of species, and the classification refers to landcover in a neighborhood surrounding a land parcel instead of species contained within the parcel. The parcel-level statistics for all land, for forest only, and for grassland only were aggregated to regional summaries using the reporting units employed by the Forest Service for Resource Planning Act (RPA) assessments.

The classification model is similar to the familiar soil triangle, a tripolar chart that classifies soil texture based on the proportions of sand, silt, and clay in a soil sample (fig. 2.1A). The soil texture tripolar chart is partitioned into subspaces with labels such as sandy clay and silt loam; a soil sample that falls within a subspace is given the corresponding soil texture name. To use a tripolar chart for classifying landcover context, the proportions of three generalized

landcover types (agriculture, developed, natural) replace sand, silt, and clay along the three axes (fig. 2.1B). The new partitioning of the tripolar chart indicates the critical values of 10, 60, and 100 percent along each axis. The class labels are coded as follows: a lowercase letter (n, d, a) appears in a label if the corresponding landcover type (natural, developed, agriculture, respectively) comprises at least 10 percent, but < 60 percent of the neighborhood (see below); an uppercase letter (N, D, A) appears if that landcover type comprises at least 60 percent but < 100 percent of the neighborhood. A letter does not appear if that landcover type comprises < 10 percent of the neighborhood. The labels NN, DD, and AA indicate neighborhoods that contain exactly 100 percent of the corresponding landcover type.

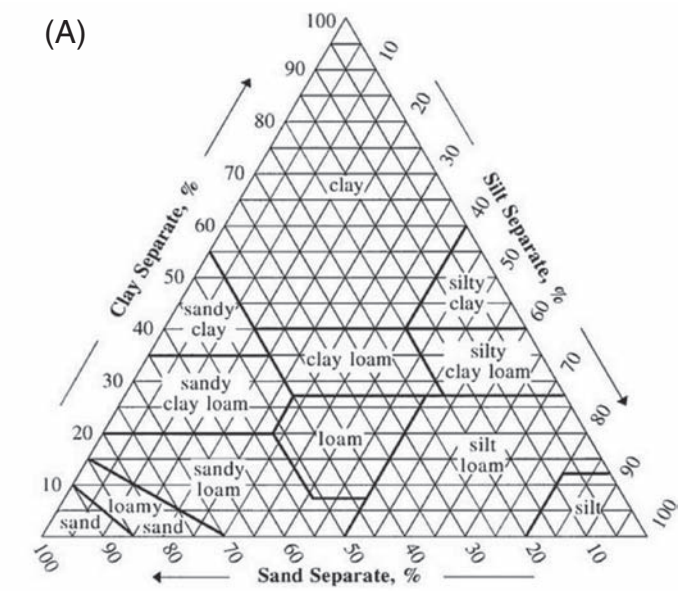


Figure 2.1A—The soil texture classification model identifies 12 soil textures according to the proportions of sand, silt, and clay in a soil sample. (Source: U.S. Department of Agriculture; see also http://en.wikipedia.org/wiki/Image:SoilTextureTriangle.jpg)

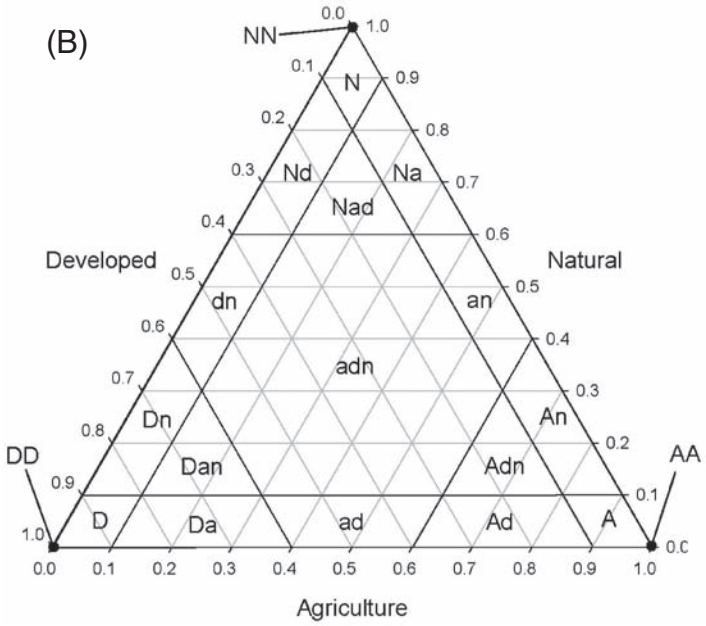


Figure 2.1B—The landcover context classification model identifies 19 landcover contexts according to the proportions of natural, developed, and agriculture landcover in a neighborhood. (See text for additional explanation.)

The model is implemented using the 2001 high-resolution, national landcover map [National Land Cover Database (NLCD)] (Homer and others 2007) of the conterminous United States. Each parcel (hereafter pixel) on the NLCD represents 0.09 ha on the ground and has one of 16 possible landcover type labels (table 2.1). Whereas the landcover label describes the **contents** of a pixel, the **context** of that pixel is described by the landcover in the surrounding neighborhood. In other words, the proportions of agriculture, developed, and natural landcover needed for the tripolar labeling of context are calculated from the landcover pixels in the surrounding neighborhood. Since the context of each pixel is based on its unique neighborhood, the new map of landcover context has a spatial resolution equivalent to the pixel size of the original landcover map. Furthermore, the original landcover map can be intersected with the map of landcover context to summarize context according to original landcover types, e.g., the context of forest pixels only.

Table 2.1—Aggregation of the NLCD landcover legend for tripolar classification and identification of forest and grassland landcover types

NLCD landcover class	Aggregation For tripolar classifications	Identification of forest and grassland landcover
Water	Natural	—
Perennial ice/snow	Natural	—
Developed, open space	Developed[a]	—
Developed, low intensity	Developed	—
Developed, medium intensity	Developed	—
Developed, high intensity	Developed	—
Barren land (rock/sand/clay)	Natural	—
Deciduous forest	Natural	Forest
Evergreen forest	Natural	Forest
Mixed forest	Natural	Forest
Shrub/scrub	Natural	—
Grassland/herbaceous	Natural	Grassland
Pasture/hay	Agriculture	—
Cultivated crops	Agriculture	—
Woody wetlands	Natural	Forest
Emergent herbaceous wetlands	Natural	—

— = neither forest nor grassland; NLCD = National Land Cover Database.
[a] The developed class also includes all pixels containing a road segment (see text).

The NLCD landcover map was modified as follows before implementing the tripolar classification. First, to better account for ecological impacts of roads, a detailed roadmap (Environmental Systems Research Institute 2005) was overlaid on the landcover map, and any pixel that contained a road segment of any size or type was changed to developed. The landcover legend was then aggregated into three generalized classes called agriculture, developed, and natural (table 2.1) for a tripolar classification model. The landcover legend was also aggregated separately to identify forest and grassland pixels (table 2.1) for summarizing context according to those two particular landcover types.

The size of a neighborhood is important because the context of any given pixel can change if more or less of the surrounding area is included in the neighborhood. In the case of a small farm embedded in a mostly forested landscape, agriculture dominates a small neighborhood in and around the farm, but is only a minor component of a larger neighborhood that is dominated by forest. As a practical matter, several neighborhood sizes were used in this analysis because there is no *a priori* reason to choose any one size. The six neighborhood sizes are 4.41 ha (7- by 7-pixel neighborhood, ~10.9 acres); 15.2 ha (13 by 13, ~37.6 acres); 65.6 ha (27 by 27, ~162 acres); 590 ha (81 by 81, ~1,460 acres); 5 310 ha (243 by 243, ~13,100 acres); and 47 800 ha (729 by 729, ~118,000 acres). Thus, the landcover context of each landcover pixel was evaluated for each of the six neighborhood sizes to produce six maps of landcover context for the conterminous United States at a spatial resolution of 0.09 ha per pixel.

To address the first question posed in the introduction to this chapter, the 19 landcover context classes (fig. 2.1B) were aggregated into four background categories called agricultural, natural, developed, or mixed (fig. 2.2A). This identifies neighborhoods dominated by, i.e., containing at least 60 percent, of one of the

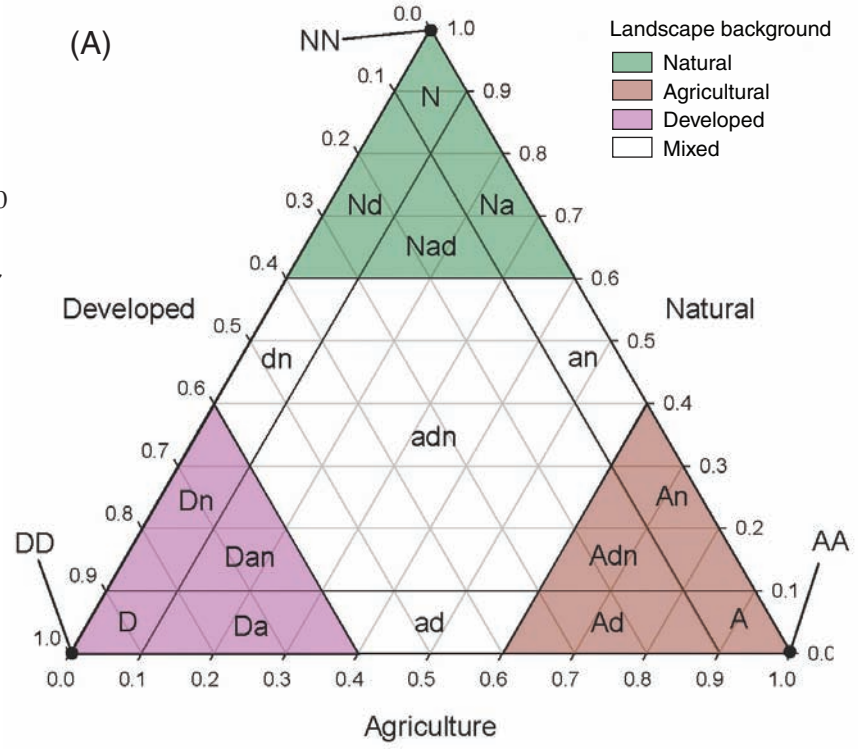


Figure 2.2A—Aggregation of 19 landcover context classes into four background classes.

three aggregated landcover types and a mixed background class for which no single landcover type dominates. To address the second question posed in the introduction, the 19 landcover context classes were aggregated in a different way (fig. 2.2B) to more specifically describe the context of forest (or grassland) that appears in a seminatural background (compare to figure 2.2A). This highlights the contexts labeled Nd (natural developed), Na (natural agricultural), and Nad (natural-agricultural-developed), which are of particular interest as potential transition zones between natural and anthropogenic landscapes, places where future landcover change is likely to be manifest as a change in the landscape background. Maps of either forest only or grassland only were overlaid on a map of landcover context to extract the context values for forest or grassland only.

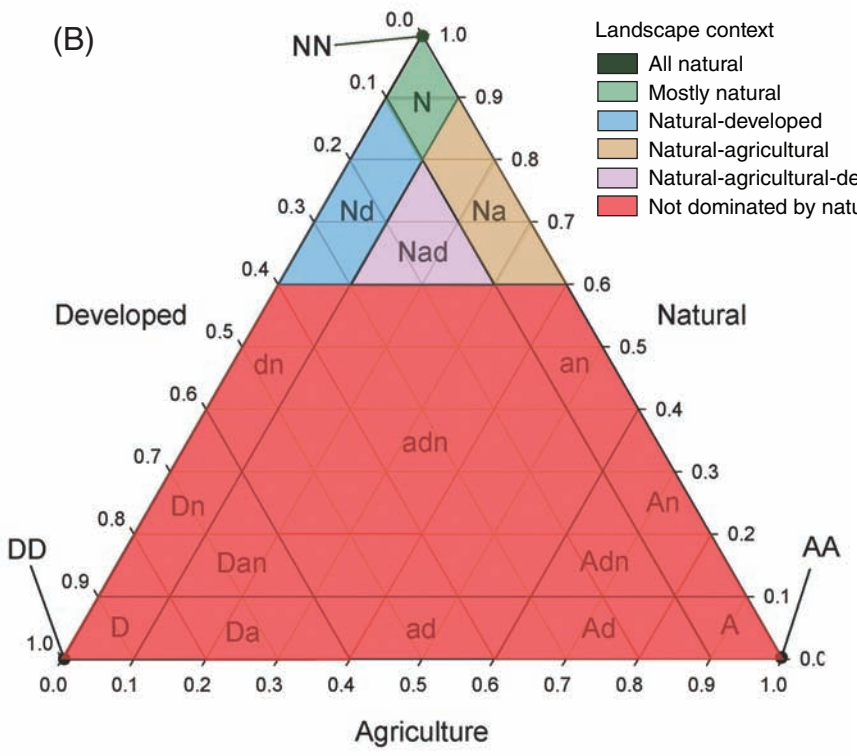


Figure 2.2B—Aggregation of 19 landcover context classes into six resource-specific classes. (See text for additional explanation.)

Each map of the conterminous United States contains approximately 8.6 by 10^9, i.e., 8.6 billion, pixels representing a total area of approximately 7.4 million km^2, including approximately 2.3 and 1.2 million km^2 of forest and grassland, respectively. Note that those area estimates from NLCD will not match official area statistics for forest and grassland since the definitions of forest and grassland are not necessarily the same for NLCD as for official statistics. To avoid potential confusion when using area statistics, the results of the context analysis were expressed as percentages of NLCD area (total area, forest area only, or grassland area only) by RPA assessment region.

Results and Discussion

Figure 2.3 illustrates the results for background context (using the model shown in figure 2.2A) for neighborhood sizes of 4, 66, and 5 300 ha in the States of Colorado, Kansas, and Missouri. A comparison of results for a smaller neighborhood size (top) and a larger size (bottom) confirms the expectations that the maps are more detailed for smaller neighborhood sizes and that the context of a given location can change with neighborhood size. It is also evident (fig. 2.3) that the agricultural, developed, and natural background contexts are more likely to be observed in smaller neighborhoods, whereas the proportion of the mixed context increases with neighborhood size. The simplest explanation is that local landcover tends to be the same in a neighborhood because of local factors like soil type and road infrastructure, whereas regional landcover appears more mixed simply because it includes a variety of local situations.

Another important trend that is evident in figure 2.3 is that the signal from a locally important feature, e.g., agriculture in southwest Colorado, that is detectable in smaller neighborhoods (top) is not detectable in larger neighborhoods (bottom). That reflects the fact that different neighborhood sizes sample the landscape at different spatial frequencies; smaller neighborhoods are more sensitive to local (fine-scale) patterns, and larger neighborhoods are more sensitive to regional (coarse-scale) patterns. Furthermore, that is part of the rationale for employing several neighborhood sizes when creating maps of landcover context. A resource manager with a local orientation and ability to influence local landcover is presumably more interested in fine-scale context, e.g., the top inset map in figure 2.3, whereas a regional manager with ability to influence regional land use policy is more interested in coarse-scale context. Finally, for a particular location, the range of neighborhood sizes over which a particular background context is obtained defines the range of spatial scales over which

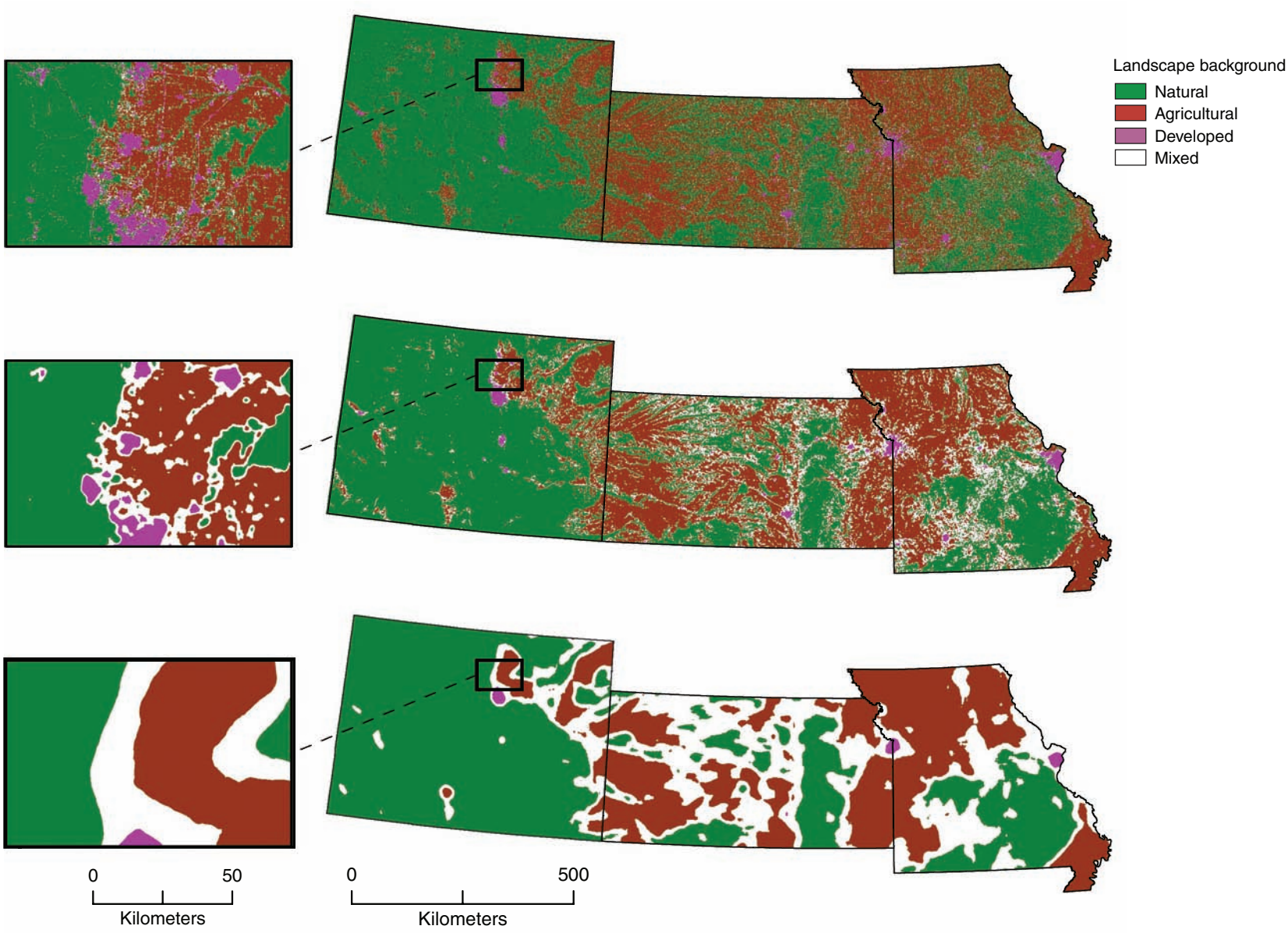


Figure 2.3—Illustration of landcover background (legend as in figure 2.2A) for all locations in the States of Colorado, Kansas, and Missouri, for neighborhood sizes of 4 ha (top), 66 ha (middle), and 5 300 ha (bottom), with inset of region immediately north of the city of Denver, CO.

particular types of landcover can be said to dominate the landscape. For example, while cities of different sizes occur throughout the three-State area, only the largest cities (Denver, Wichita, Kansas City, St. Louis) are sufficiently large to dominate the landscape at scales approximating hundreds of square kilometers (compare figure 2.3, top and bottom).

Figure 2.4 illustrates the landcover context of the forest in the three-State region. Note that the white areas in this figure are nonforest locations; the colors of the forest locations refer to the model shown in figure 2.2B. The trends in relation to increasing or decreasing neighborhood size are generally the same, with similar explanations, as were illustrated for the background context (fig. 2.3). The higher thematic detail of this version of the model is evident in the top inset map of figure 2.4, where development associated with road infrastructure is visible as a network of the natural-developed context, i.e., context class Nd. This signal is not visible for larger neighborhood sizes because roads, while pervasive and locally dense, never occupy a large percentage of total area over larger regions.

One of the more striking comparisons in figure 2.4 is the difference in forest context between the States of Colorado and Missouri over a range of neighborhood sizes. In Colorado, nearly all forest occurs in neighborhoods that are 100 percent natural, i.e., context class NN, whereas in Missouri the same is observed only on the Ozark Mountains. With increasing neighborhood size, the forest in Colorado tends to remain in mostly natural contexts (class N) while the forest in Missouri tends to occur in neighborhoods that are not dominated by natural landcover. For the largest neighborhood size, the natural-agricultural context (class Na) in Colorado appears as inclusions in regions otherwise dominated by natural landcover types, but in Missouri that same context (class Na) appears as rings or edges surrounding a core region dominated by natural landcover types.

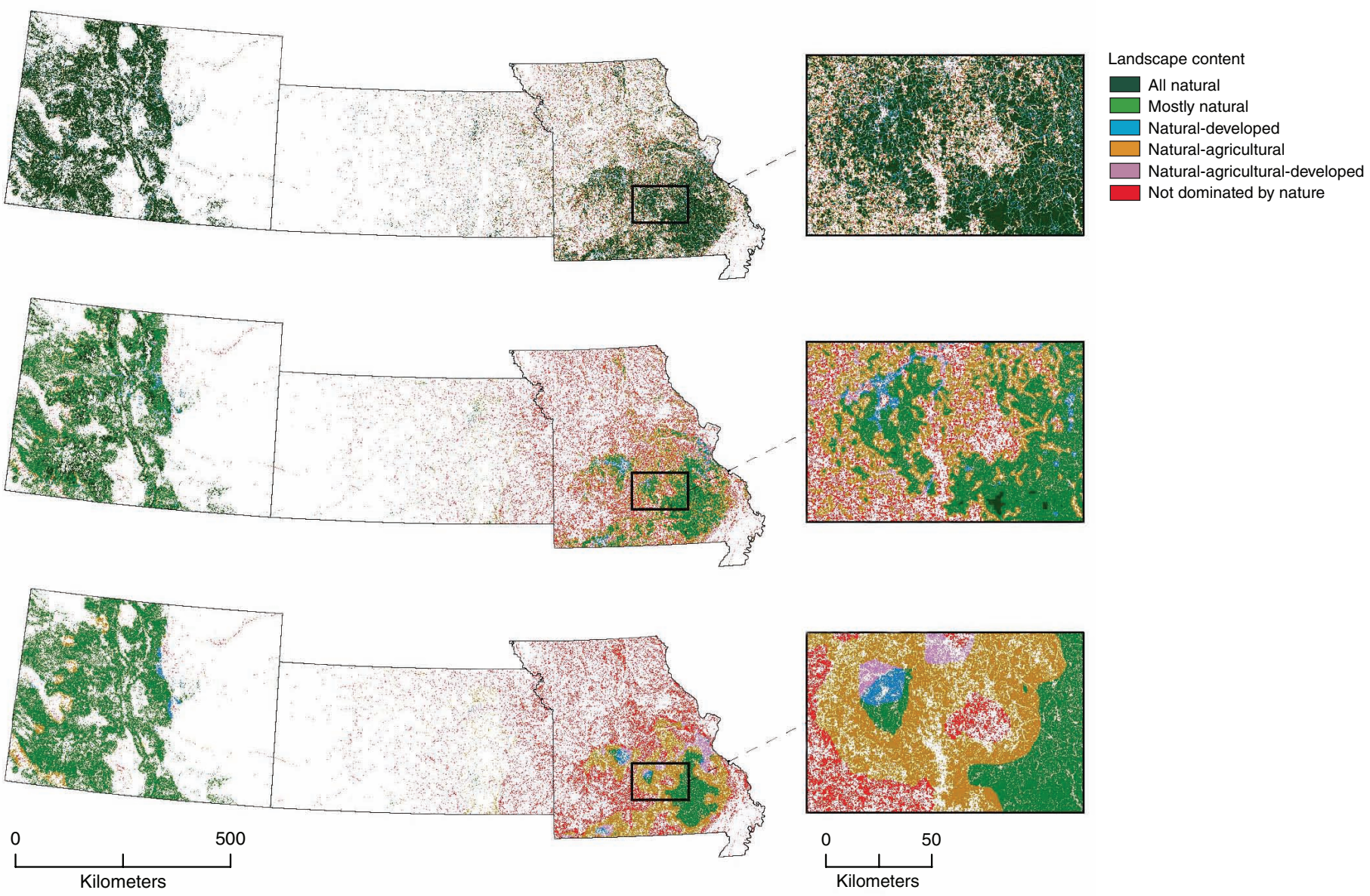


Figure 2.4—Illustration of landcover context (legend as in figure 2.2B) for forest in the States of Colorado, Kansas, and Missouri, for neighborhood sizes of 4 ha (top), 66 ha (middle), and 5 300 ha (bottom), with inset of region of the Salem Plateau in the Ozark Mountains, MO.

The examples for grassland landcover context in the three-State region (fig. 2.5) generally exhibit trends in relation to neighborhood size that are similar to the forest examples (fig. 2.4). Note that the white areas in this figure are nongrassland locations, which permits comparison of the distributions of forest and grassland across the three-State region, i.e., the presence or absence of colored pixels on the two maps. The purpose of using the model shown in figure 2.2B was to focus on the occurrence of forest and grassland in transition zones. The examples show clearly the differences between local (top) and regional (bottom) perspectives of transition zones. Local transition zones by definition are mapped in more detail, and the pixels in those zones are more likely to have a different context than their immediate neighbors, in comparison to regional transition zones.

Ultimately, the pixel-level maps of context will provide the most flexibility for both regional and local resource managers to examine contexts at different scales and in different places. Local details necessarily are ignored for the purpose of this report, for which the results are summarized within each of the four RPA reporting regions (North, Pacific Coast, Rocky Mountain, and South) and are displayed as pie charts for each region. Figure 2.6 shows the proportion of total area within each region that is within each of the four background contexts (compare to figures 2.2A and 2.3). Figure 2.7 shows the proportion of total forest area within each region that is within each of the six contexts shown in figure 2.2B (compare to figure 2.4), and figure 2.8 shows the proportion of total grassland area for the same model (fig. 2.2B, compare to figure 2.5). Note that expressing results as proportions of area (total, forest, or grassland) does not take into account difference in absolute area (total, forest, or grassland) among the reporting regions.

Considering first the regional shares of background context (fig. 2.6), several important observations can be made. In all regions

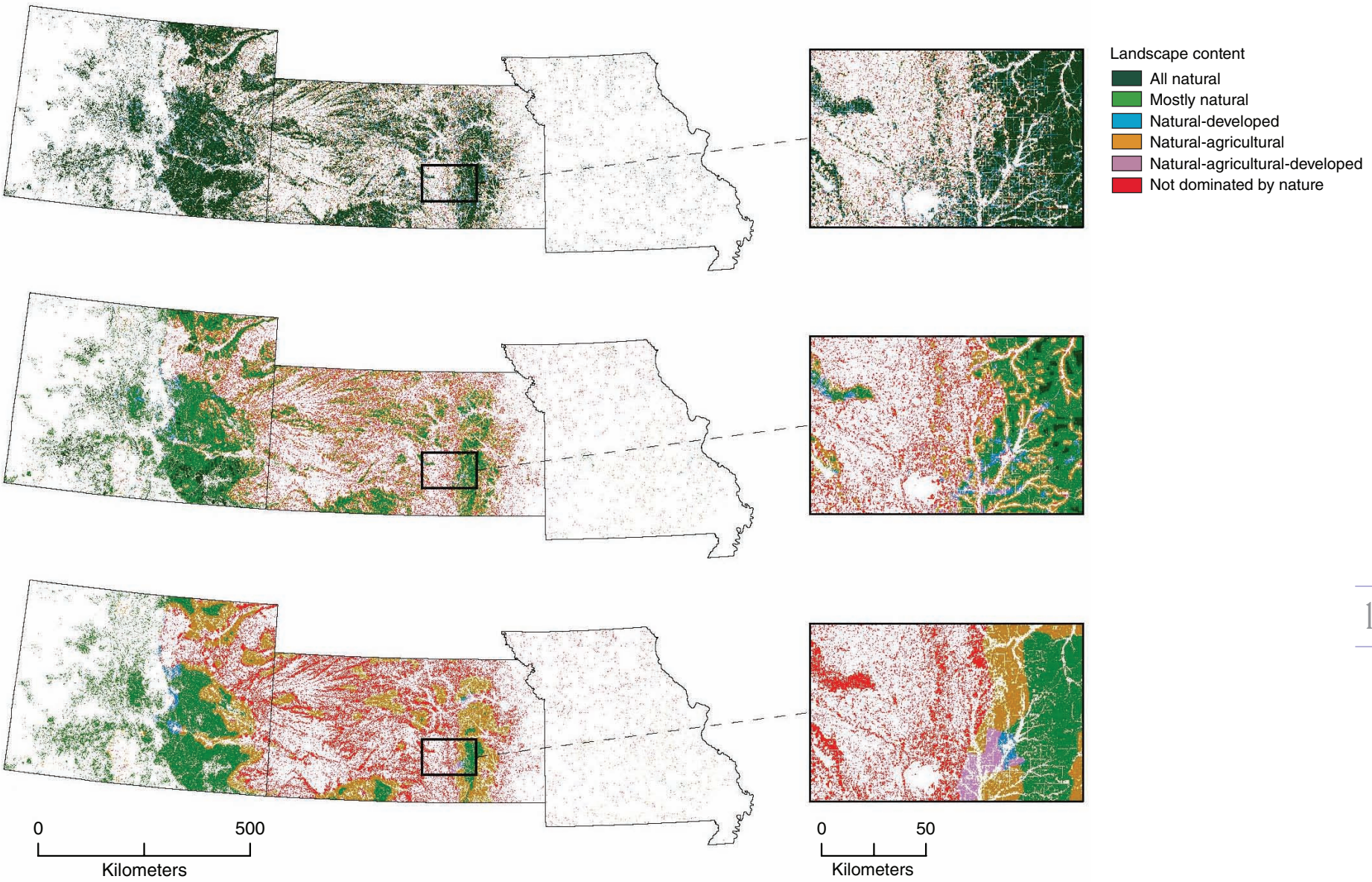


Figure 2.5—Illustration of landcover context (legend as in figure 2.2B) for grassland in the States of Colorado, Kansas, and Missouri, for neighborhood sizes of 4 ha (top), 66 ha (middle), and 5 300 ha (bottom), with inset of region surrounding the city of Wichita, KS.

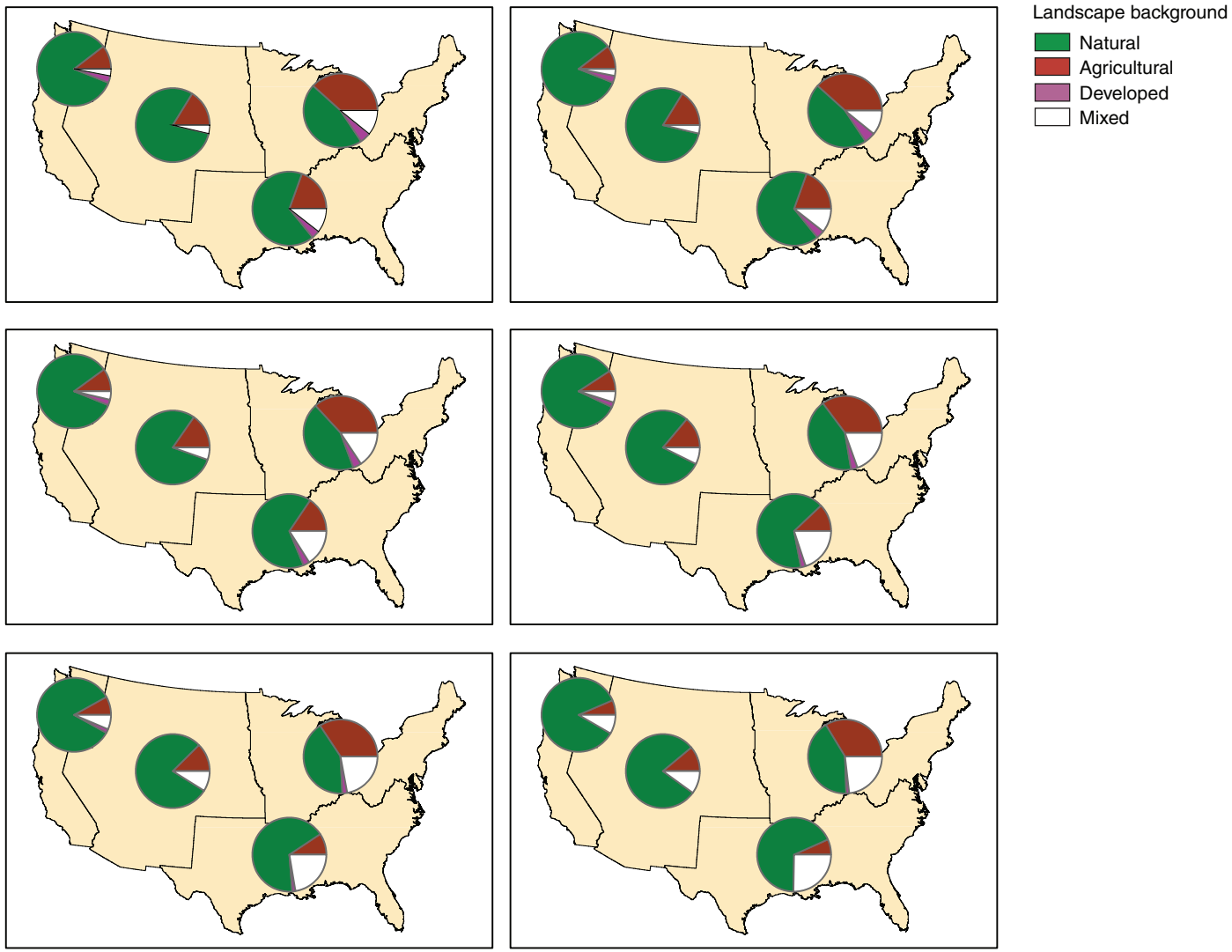


Figure 2.6—The proportions of all land in each of four Resource Planning Act reporting regions (North, Pacific Coast, Rocky Mountain, South) that are within each of four landscape background classes (legend uses the same colors as shown in figure 2.2A) for neighborhood sizes of 4 ha (top left), 15 ha (top right), 66 ha (middle left), 590 ha (middle right), 5 310 ha (bottom left), and 47 800 ha (bottom right).

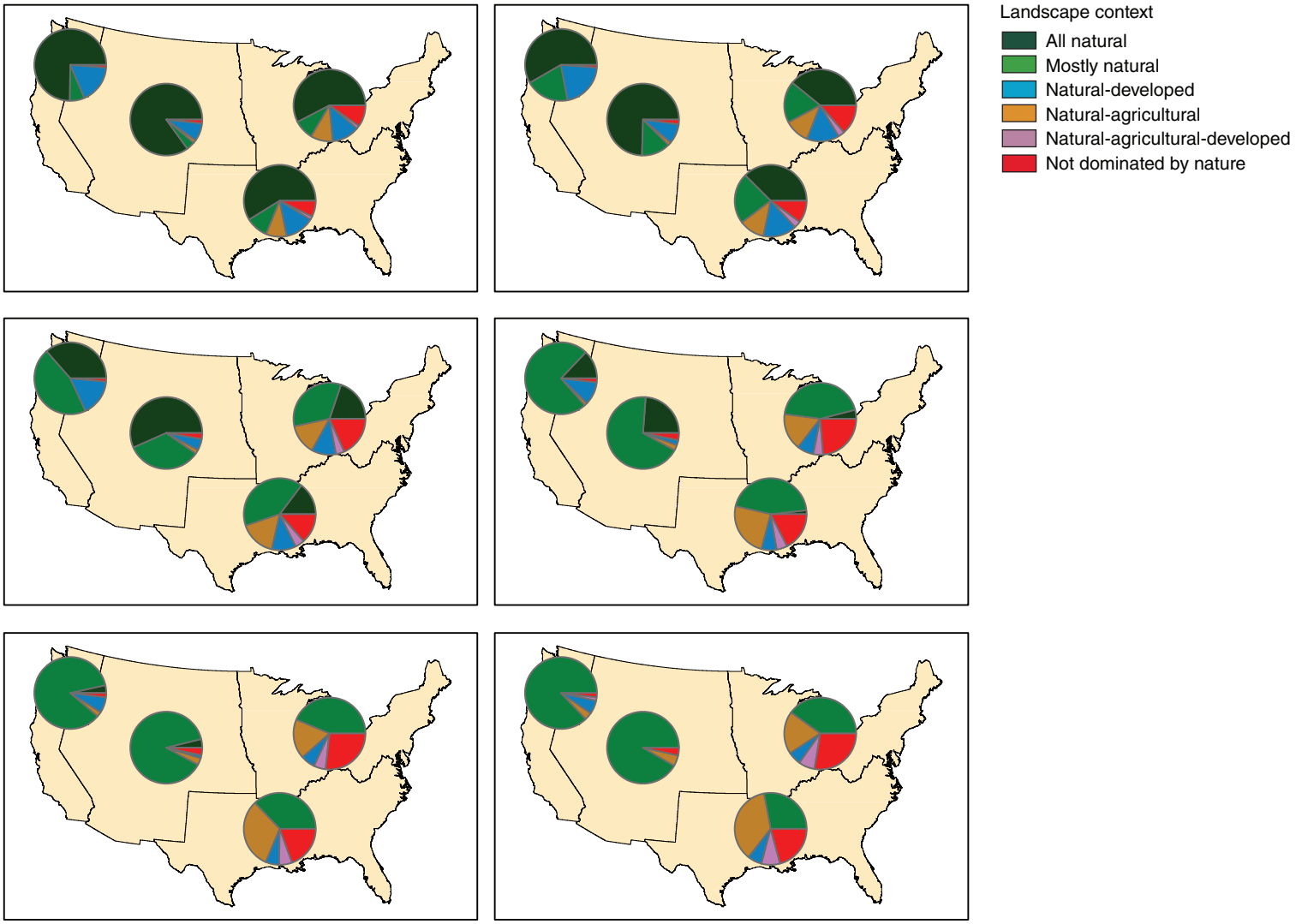


Figure 2.7—The proportions of all forest land in each of four Resource Planning Act reporting regions (North, Pacific Coast, Rocky Mountain, South) that are within each of six landscape context classes (legend uses the same colors as shown in figure 2.2B) for neighborhood sizes of 4 ha (top left), 15 ha (top right), 66 ha (middle left), 590 ha (middle right), 5 310 ha (bottom left), and 47 800 ha (bottom right).

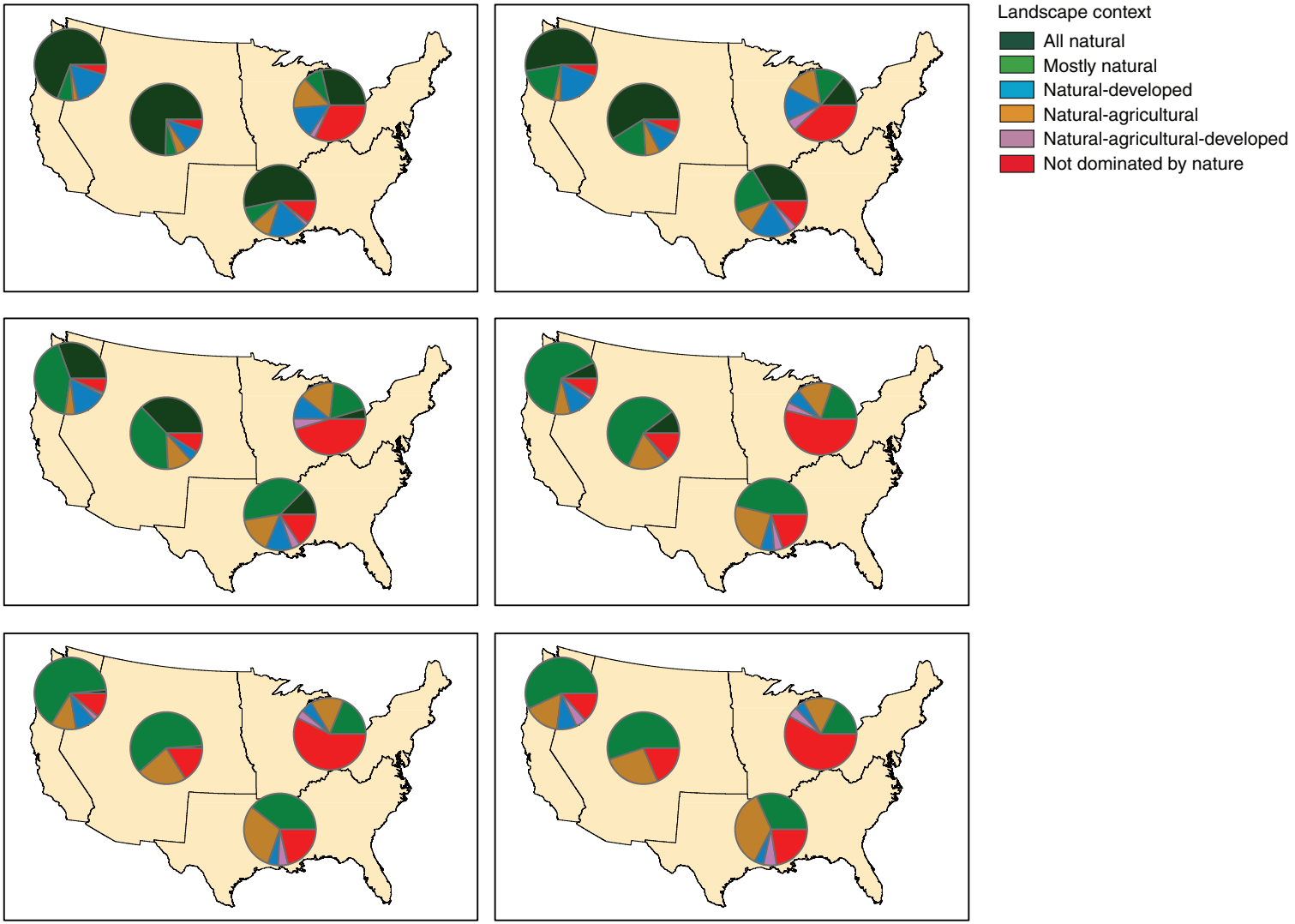


Figure 2.8—The proportions of all grassland in each of four Resource Planning Act reporting regions (North, Pacific Coast, Rocky Mountain, South) that are within each of six landscape context classes (legend uses the same colors as shown in figure 2.2B) for neighborhood sizes of 4 ha (top left), 15 ha (top right), 66 ha (middle left), 590 ha (middle right), 5 310 ha (bottom left), and 47 800 ha (bottom right).

except the North, the majority of all land occurs in a natural background context, and the regional share of agricultural background generally decreases with neighborhood size. The implication is that in those three regions, no matter how intensive agriculture might be on a local basis, it still appears as an anomaly when set against the regional backdrop of natural landscapes. In contrast, in the North region the share of the agricultural background is large compared to other regions and is less affected by neighborhood size. The latter observation, which also applies to the Rocky Mountain region, can be attributed to the presence of extensive agricultural regions such as the corn belt in the North region and the wheat belt in the Rocky Mountain region. In those cases, the natural background has been converted to an agricultural background over large areas. Finally, it is interesting that the regional share of the developed background is never large, i.e., typically < 5 percent, for any of the situations shown. That observation agrees with the proportions of developed landcover itself (not shown here), which is approximately 7 percent for the Nation (including roads) and concentrated in the more populated South and North regions.

The regional shares of forest (fig. 2.7) in different landcover contexts highlight additional aspects of landscape patterns. For example, while the regional shares of natural background (fig. 2.6) are typically 50 to 60 percent, the regional shares of forest in natural background are typically 70 to 90 percent. The implication is that forest tends to occur in natural backgrounds if it occurs at all. Of significant interest is the observation that over half of all forest, in all regions, occurs in a landscape context that is 100 percent natural on a local basis. That means simply that a large proportion of forest occurs within locally intact tracts. However, that proportion decreases rapidly with neighborhood size in all regions, implying that most of those intact tracts are not very large in total extent. A relatively small proportion (typically < 20 percent) of the forest in both the Pacific Coast and Rocky Mountain regions occurs in landscape contexts other than all natural or mostly natural. In contrast, that proportion in the North and South regions is at least 30 percent in the smallest neighborhoods and increases to 60 to 70 percent in the largest neighborhoods. The implication is that eastern forests overall are more exposed than western forests to both short- and long-range impacts from intensive human land uses such as agriculture and development.

The regional shares of grassland (fig. 2.8) in different landcover contexts are generally similar to those of forest (fig. 2.7) with two important exceptions: (a) a smaller share of grassland occurs in 100 percent natural landscapes in comparison to forest, and (b) a higher share of grassland occurs in landscapes that contain more than 10 percent agricultural or developed landcover. It should be noted that the grassland statistics for the North and South do not reflect the fact that there is relatively little grassland overall in those regions in comparison to the other two regions (results not shown here).

Conclusion

This brief summary of a multiscale, national assessment of landcover context indicates that at least half of the area of the conterminous United States can be characterized as a natural background for scales ranging from several hectares to several hundred hectares. Agricultural and developed landcover backgrounds are important locally, but at regional scale they typically occupy no more than 30 percent of total area, principally in the North and South regions and in the eastern portion of the Rocky Mountain region. Considering the questions posed in the introduction to this chapter, we conclude that at local scales, both forest and grassland tend to occur in landscape contexts dominated by forest, grassland, and other natural landcover types. For forest, the same remains true as scale increases from local to regional in the two western regions but not in the two eastern regions where much higher proportions of forest appear in landscapes that contain a significant (more than 10 percent) component of agriculture and developed landcover types. Relatively small proportions of western forest were identified as occurring in transition zones and these zones were more apparent at local scale than at regional scale. The proportions of eastern forest in transition zones were larger and more apparent at regional scale than at local scale. Trends for grassland were generally similar except that a higher proportion of grassland is found in transition zones in all regions, at both local and regional scales. In addition, the proportion of grassland found in landscapes not dominated by natural landcover types is much higher than that proportion for forests, again for all regions and at both local and regional scales.

Literature Cited

Environmental Systems Research Institute. 2005. ESRI data & maps: an ESRI white paper. Redlands, CA. 41 p.

Homer, C.; Dewitz, J.; Fry, J. [and others]. 2007. Completion of the 2001 national landcover database for the conterminous United States. Photogrammetric Engineering and Remote Sensing. 73: 337–341.

Chapter 3. Relationship of a Lichen Species Diversity Indicator to Environmental Factors Across the Coterminous United States

Susan Will-Wolf
Mark J. Ambrose
Randall S. Morin

Introduction

We have investigated relationships between one simple indicator of lichen species diversity and environmental variables in forests across the coterminous United States. We want to know whether this indicator can help quantify the influence that factors such as climate and air quality have on lichen biodiversity at large scales and whether it will be useful in monitoring changes in these factors. Since lichens are biologically distinct from the trees, shrubs, and herbaceous vegetation found on sample plots, a good indicator of effects on the lichen community complements other indicators and contributes to a more complete picture of potential effects of these causal factors on the entire forest ecosystem.

The Forest Inventory and Analysis (FIA) Program of the Forest Service, U.S. Department of Agriculture, collects lichen community data on its phase 3 (P3) plots (a subset of the full national phase 2 grid) because lichens are useful indicators of air quality impact, forest health, and forest ecosystem integrity across the United States [McCune 2000, Nimis and others 2002 (reviews), U.S. Department of Agriculture Forest Service 2007]. Collection of lichen data began in 1994 by the National Forest Health Monitoring (FHM) Program of the Forest Service, and has been continued by FIA since 2000. The standard lichen data collection protocols[1] have remained unchanged since 1994, permitting the FHM and FIA lichen data to be combined. Abundance for each macrolichen (a lichen that can be separated from its substrate) species on standing woody substrates in a 0.937-acre (0.379-ha) permanent plot is determined from samples collected by a trained and certified nonspecialist in a time-constrained (30-minute minimum to 2-hour maximum) survey (see footnote 1). A minimum certification standard (65 percent of species found by an expert) is achieved by each qualified field person. Restriction to macrolichens, use of trained nonspecialist field personnel, and time-constrained sampling make this indicator affordable. The time restriction also lessens (but does not remove) differences between field personnel and field lichen specialists (during training, audits, and quality assurance), and among field personnel with different abilities, in number of species collected on average. This increases the national comparability of data. Specimens are identified to species in the

[1] Forest Service, U.S. Department of Agriculture. 2005. Forest inventory and analysis national core field guide. Vol. 1: field data collection procedures for phase 3 plots. Version 3.0. Internal report. On file with: Forest Service, U.S. Department of Agriculture, Forest Inventory and Analysis, Rosslyn Plaza, 1620 North Kent Street, Arlington, VA 22209. [The current version is available online at http://fia.fs.fed.us/library/field-guides-methods-proc/ (Date accessed: January 2010)].

laboratory by lichen identification specialists.

Because FIA data are collected from permanent plots randomly located (one per sampling grid cell) within a regular nationwide sampling grid with no *a priori* stratification by other criteria, these data are unbiased in their representation of forest condition (Bechtold and Patterson 2005). The lichen indicator sampling protocol was designed to enable the program to monitor regional patterns in lichen diversity, community composition, and response to air quality and climate while minimizing, to the extent possible, the potentially confounding effects of local variation in tree species composition, forest type, and stand age (McCune and others 1997). With 12 years of data and most of the forested regions of the coterminous United States represented in the dataset to at least some extent, we can evaluate the success of this goal in part as we explore the potential of one lichen diversity indicator.

The FIA lichen database provides detailed information on lichen community composition. Because species distribution varies greatly across the country, lichen indicators based on lichen species composition in communities are more appropriate for smaller geographic regions. In contrast, the number of macrolichen species (lichen S) found on an FIA plot using the standard protocol is a lichen species diversity indicator consistent across the entire country. Lichen S is the simplest of several quantitative lichen indicators that can be derived from FIA lichen data. It is assumed that some less common macrolichen species are missed during sampling because of time constraints and use of nonspecialist field personnel (Will-Wolf and others 2002). If in spite of these limitations and other sources of variation, we find that lichen S is a good indicator of response to factors of interest, we have a useful tool to monitor forest ecosystems on a very broad scale.

For this study we focus on relationships between lichen S and environmental variables in forests across the coterminous United States to see how these relationships differ between Eastern States (our East region, figure 3.1) and Western States (our West region), and whether and how patterns vary among different ecoregions within the East and West (fig. 3.2). This helps us decide where lichen S can be most useful as an indicator for monitoring patterns and trends in forested ecosystems. This large-scale analysis provides a context and framework for generalizing from more intensive smaller scale analyses based on lichen community composition, of which there are already several published examples (e.g., Geiser and Neitlich 2007; Jovan 2009; Jovan and McCune 2005, 2006; McCune and others 1997; Will-Wolf and others 2006).

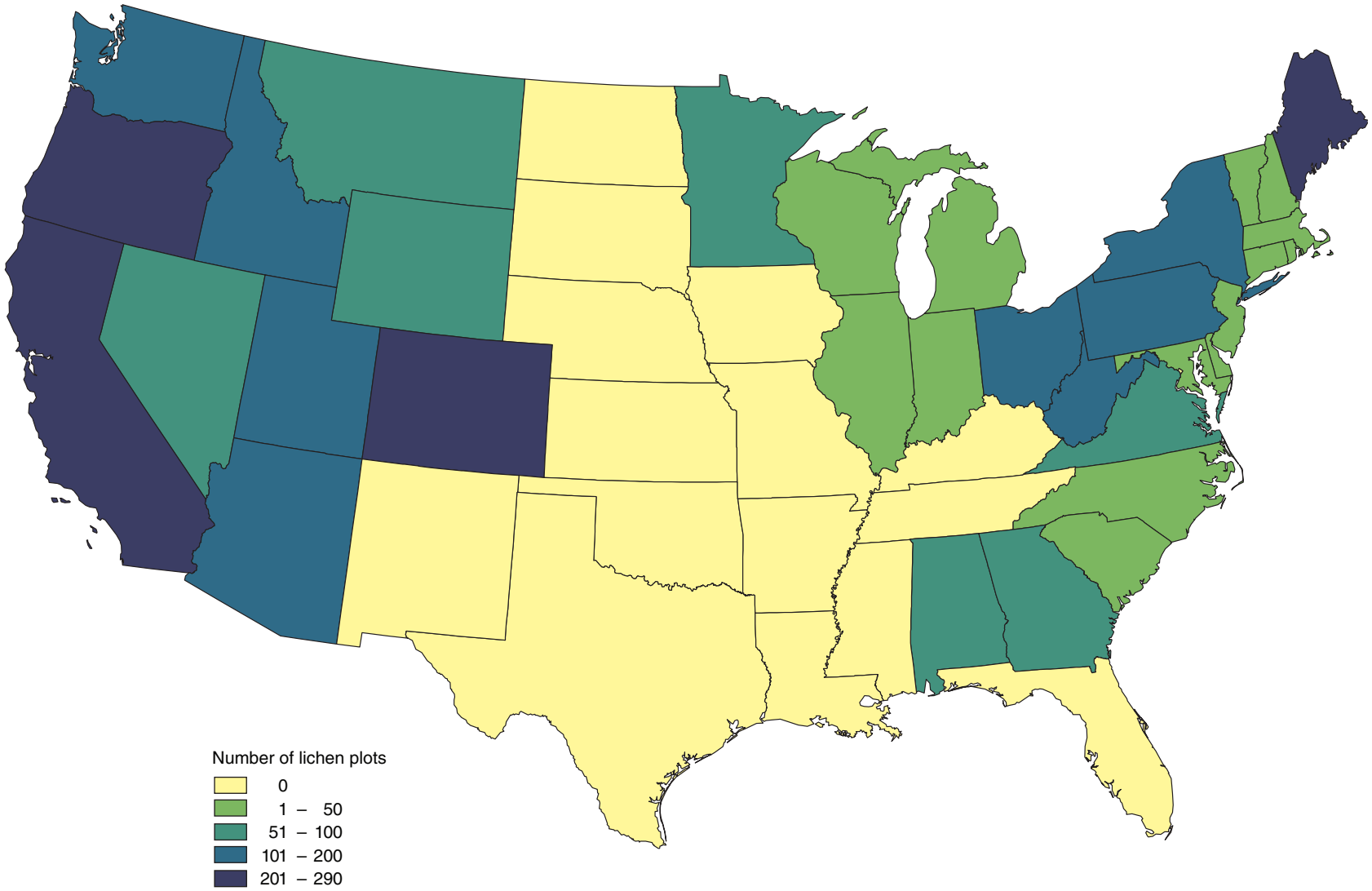


Figure 3.1—Summaries by State for numbers of plots (by category) included in this project. For this project, West region includes all States west of the Great Plains that have lichens data (colors other than yellow), and East region includes Minnesota plus all States east of the Mississippi river that have lichens data. Both size of State and number of years sampled affect number of plots available. (Data source: U.S. Department of Agriculture, Forest Service Forest Inventory and Analysis Program)

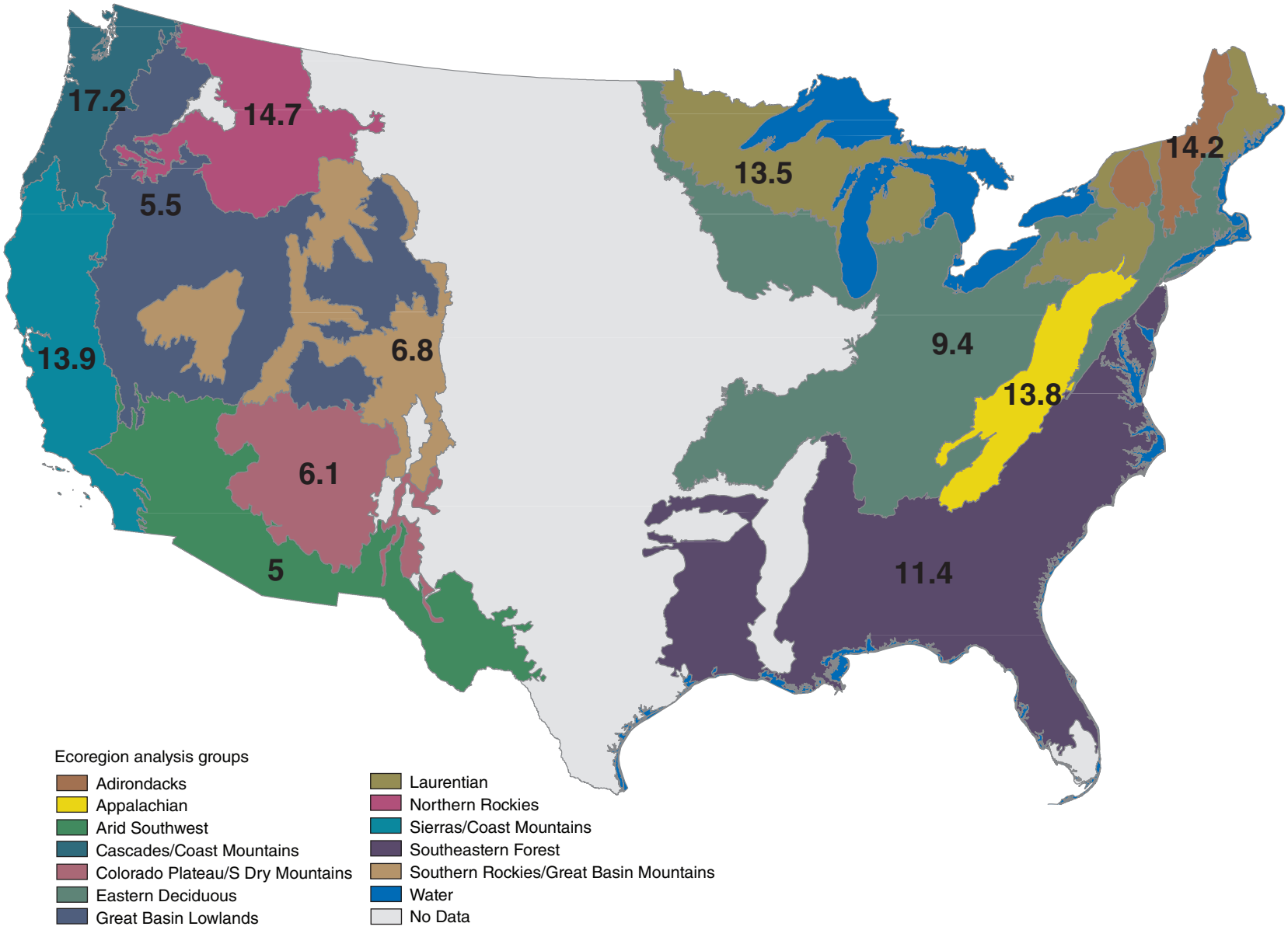


Figure 3.2—Ecoregion groups, based on Bailey's ecoregion provinces (Bailey 1989; Cleland and others 2005) used for analysis (areas in gray not included). For this project, West region includes all ecoregion groups west of the large gray area with no data, and East region includes all ecoregion groups east of this large gray area. Some ecoregion groups cover areas including States with no lichens data; compare with figure 3.1. Printed numbers are average for each ecoregion of Lichen S, the number of macrolichen species found per plot. Note the greater variation in average Lichen S between groups in the West. See appendix tables A3.1 and A3.3 for ecoregion provinces included in each group. (Data source: U.S. Department of Agriculture, Forest Service Forest Inventory and Analysis Program)

Methods

We used publicly available FIA lichen and tree data and approximate plot location[2] to develop explanatory variables and assign plots to analysis groups based on Bailey's ecoregion provinces (Bailey 1989, Cleland and others 2005). Lichen S, total live-tree basal area per acre (tree BA), and percentage of BA in softwoods (percent softwoods = conifers) were calculated for each plot. Tree BA and percent softwoods represent forest structure and composition independent of particular tree species composition and as such are appropriate to compare across the entire country and between all ecoregion groups. We chose these two tree variables because they are related to the amount and quality of lichen substrate on a plot. They were also readily available for most of the lichen plots in our data set (in contrast to other potentially usable forest structure variables). For the East (east of the Great Plains, figure 3.1), it has been demonstrated (e.g., McCune and others 1997) that very low tree BA on a plot is correlated with low lichen diversity. So, for the East data, we removed 61 plots with total live-tree BA < 21.9 square feet per acre (< 5 m^2/ha). In the West (west of the Great Plains), small trees and shrubs are known to harbor many lichens in some areas (e.g., Geiser and Neitlich 2007) and low tree BA is not correlated with low lichen diversity. So no tree BA limit was imposed for West plots. Some plots were resurveyed during the 12 years covered by our study; we included only the most recent data available for each plot. The number of plots per State (fig. 3.1) varied with both size of the State and number of years that sampling occurred. Lichen data were available for 3,276 plots, but tree data were not available for all of them. We analyzed a total of 1,399 plots in the West and 1,087 plots in the East. To maintain as large a sample as possible and to use the same data set for all analyses, we made no attempt to equalize the distribution of plots across any of the gradients represented by geographic, climate, or air quality variables.

Climate data [(1971–2000) 30-year average annual rainfall, average maximum July temperature, and average minimum January temperature] were generated on a 4-km^2 grid using the climate source model (Daly and Taylor 2000). Air quality data [(1998–2004) average annual wet deposition of SO_4^{--}, NO_3^-, and NH_4^+] were estimated from an interpolated 5-km^2 grid of National Atmospheric Deposition Program data (Coulston and others 2004).

[2] Forest Service, U.S. Department of Agriculture, 2010. Forest Inventory and Analysis Database, version 3. U.S. Department of Agriculture, Forest Service, Washington Office. Internal Report. http://fia.fs.fed.us/tools-data/. [Date accessed: January 2010].

Elevation data were acquired from the National Elevation Dataset (U.S. Geological Survey 1999) on a 1-km^2 grid. Using ArcGIS 8.3©, plot locations were intersected with the climate grids, air quality grids, and the elevation grid to generate a set of plot variables for statistical analysis. Maps were also generated to illustrate ecoregion groups and geographic variation in air quality variables.

We analyzed data separately for the East and West regions (fig. 3.1) because ecological systems are known to be very different in the Eastern and Western coterminous United States (compare column 2 of tables 3.1 and 3.2). Within the East or West, we then further subdivided the data for analysis into Bailey's ecoregion groups (fig. 3.2) to aid us in interpreting the large-scale patterns. In the East, we found grouping by ecoregion division useful and convenient (fig. 3.2, appendix tables A3.1 through A3.5). In the West, Bailey's ecoregion divisions are less geographically compact and more interspersed than in the East, so we used an alternative grouping of ecoregion provinces into analysis units that are relatively cohesive geographically (fig. 3.2, appendix tables A3.6 through A3.12). We chose this approach because lichen dispersal rate is expected to be higher within compact geographic areas and changes in lichen indicators over time (tracking this is desired for such indicators) relate to dispersal of species to plots as well as to losses of species from plots. There were few forested plots located in Great Plains ecoregions, so these ecoregions were not included in the smaller scale analyses of either eastern or western ecoregion groups.

Variables for East and West regions are summarized in tables 3.1 and 3.2; for East groups see appendix tables A3.1 through A3.5 and for West groups see appendix tables A3.6 through A3.12. Explanatory variables often represent geographic gradients of values across a region rather than random variation around a central tendency, so their values are reported as ranges. Mean and standard deviation are included as a convenience for qualitative comparison between groups. Estimated NH_4^+

Table 3.1—Ranges and correlations among lichen S and geographic, climate, air quality, and forest structure variables in the East region

		Spearman correlations (rho) (N = 1087[a], years = 1994–2005)										
Variable	Range (average ± sd)	Lichen S	Lat.	Long.	Elev.	Precip.	Max. July temp.	Min. Jan. temp.	NO_3^-	SO_4^{--}	NH_4^+	Tree BA
Lichen S	0 – 33 (12.1 ± 6.3)											
Latitude (decimal degrees, S to N)	30.64 – 49.18	0.149										
Longitude (decimal degrees, W to E)	–96.95 – –67.11	—	0.425									
Elevation (m)	0 – 1660 (310 ± 219)	—	—	—								
Precipitation (mm)	483 – 1919 (1099 ± 192)	—	**–0.550**	—	—							
Max. July temp. (°C)	21.9 – 33.9 (28.2 ± 2.5)	**–0.196**	**–0.825**	**–0.516**	**–0.556**	—						
Min. Jan. temp. (°C)	–24.3 – 5.0 (–9.7 ± 6.0)	—	**–0.966**	—	—	**0.514**	**0.866**					
NO_3^- (kg/ha/year)	5.70 – 22.51 (12.28 ± 3.02)	**–0.454**	—	—	—	—	—	—				
SO_4^{--} (kg/ha/year)	4.32 – 28.98 (15.88 ± 4.87)	—	—	—	—	—	—	—	**0.883**			
NH_4^+ (kg/ha/year)	0.70 – 4.32 (2.53 ± 0.70)	—	—	**–0.565**	—	—	—	—	**0.575**	**0.486**		
Tree BA (square foot per acre)	22.1 – 579.5 (94.7 ± 50.5)	—	0.116	—	—	—	—	—	—	—	—	
Percent BA softwoods	0 – 100 (26.2 ± 33.6)	0.184	—	—	—	—	—	—	—	–0.440	—	—
Best overall full regression model		**r^2 = 0.305, p < 0.0005: Lichen S = 23.480 –0.814 NO_3^- + 0.007 elev. –1.706 NH_4^+ (–0.008 Min. Jan.)** S larger with cleaner air, higher elevation (cooler winters)										

— = cells with dashes indicate weaker correlations; max. = maximum; min. = minimum; sd = standard deviation; lat. = latitude; long. = longitude; elev. = elevation; precip. = precipitation; temp. = temperature; NO_3^- = nitrate; SO_4^{--} = sulfate; NH_4^+ = ammonium; BA = basal area; S = south; N = north; W = west; E = east; m = meter; mm = millimeter; °C = degree Celsius; kg = kilogram; ha = hectare.

Positive correlations are in black; negative are in red. Single correlations with rho > 0.447 (in bold, all with p < 0.0005) account for at least 20 percent of variation; correlations with Lichen S stronger than this cutoff suggest potential for an indicator. The strongest correlation for each class of variables is included for comparison (if p < 0.01) even if below the cutoff. Correlations between environmental variables with rho ≥ 0.775 (at least 60 percent of variation accounted for) indicate those two variables should not be entered in the same regression model. The best overall regression model (at bottom), with contributing variables listed in descending order of significance, is only slightly better than the best minimal model; minor variables (p ≤ 0.05 but usually p ≥ 0.005) are listed in parentheses.

[a] Removed E plots with total live-tree BA < 21.8966 square feet per acre (< 5 m^2/ha).

Table 3.2—Ranges and correlations among Lichen S and geographic, climate, pollution, and forest structure variables in the West region

		Spearman correlations (*rho*) (N = 1,399 plots, years = 1994–2005)										
Variable	Range (average ± sd)	Lichen S	Lat.	Long.	Elev.	Precip.	Max. July temp.	Min. Jan. temp.	NO_3^-	SO_4^{--}	NH_4^+	Tree BA
Lichen S	0 – 45 (10.8 ± 7.8)											
Latitude (decimal degrees, S to N)	31.42 – 48.96	**0.487**										
Longitude (decimal degrees, W to E)	–124.41 – –103.1	**–0.485**	—									
Elevation (m)	0 – 3740 (1687 ± 778)	**–0.528**	—	**0.714**								
Precipitation (mm)	171 – 4731 (853 ± 624)	**0.490**	**0.499**	**–0.560**	—							
Max. July temp. (°C)	15.7 – 38.3 (26.5 ± 4.2)	—	**–0.488**	—	—	**–0.522**						
Min. Jan. temp. (°C)	–22.6 – 4.7 (-7.4 ± 5.2)	—	—	**–0.751**	**–0.773**	—	—					
NO_3^- (kg/ha/year)	0.38 – 5.90 (2.81 ± 1.01)	—	—	**0.552**	—	—	—	—				
SO_4^{--} (kg/ha/year)	0.18 – 7.53 (1.98 ± 1.06)	—	—	—	—	—	—	—	**0.727**			
NH_4^+ (kg/ha/year)	0.00 – 2.14 (0.61 ± 0.36)	–0.329	**–0.497**	—	—	—	—	—	**0.571**	—		
Tree BA (square foot per acre)	1.5 – 650.2 (105.4 ± 83.6)	0.255	—	—	—	0.379	—	—	—	—	—	
Percent BA softwoods	0 – 100 (84.2 ± 31.1)	—	—	—	—	—	—	–0.294	—	—	—	—
Best overall full regression model		**r^2 = 0.404, p < 0.0005: Lichen S = 12.563 –0.005 elev. + 0.296 lat. + 0.013 BA –0.023 Max. July (–1.706 NH_4^+)** S larger lower elevation, more north, with more/larger trees, cooler summers (cleaner air)										
Best minimal linear regression model		r^2 = 0.377, p < 0.0005: Lichen S = –30.044 –0.003 elev. + 0.515 lat. –0.216 long S larger lower elevation, more north, more west										

— = cells with dashes indicate weaker correlations; max. = maximum; min. = minimum; sd = standard deviation; lat. = latitude; long. = longitude; elev. = elevation; precip. = precipitation; temp. = temperature; NO_3^- = nitrate; SO_4^{--} = sulfate; NH_4^+ = ammonium; BA = basal area; S = south; N = north; W = west; E = east; m = meter; mm = millimeter; °C = degree Celsius; kg = kilogram; ha = hectare.

Positive correlations are in black; negative are in red. Single correlations with rho > 0.447 (in bold, all with p < 0.0005) account for at least 20 percent of variation; correlations with Lichen S stronger than this cutoff suggest potential for an indicator. The strongest correlation for each class of variables is included for comparison (if p < 0.01) even if below the cutoff. Correlations between environmental variables with rho ≥ 0.775 (at least 60 percent of variation accounted for) indicate those two variables should not be entered in the same regression model. The best overall full regression model (bottom of table), with contributing variables listed in descending order of significance, is notably better than the best minimal model (also included for comparison). Minor variables (p ≤ 0.05 but usually p ≥ 0.005) in the best overall regression model are listed in parentheses.

(fig. 3.3A) was interpreted to represent pollution from agricultural sources. Estimated NO_3^- and SO_4^{--} (NO_3^- in figure 3.3B) were interpreted to represent acidic pollution primarily from urban/industrial sources.

We investigated relationships between lichen S and environmental variables first using scatterplots and simple correlations (Sokal and Rohlf 1995). All statistical analyses were carried out in SPSS for Windows (Release 13.0.1© SPSS, Inc. 1989–2004). We report the nonparametric Spearman rank correlation coefficient *rho* in all cases since we did not test whether data met assumptions for parametric tests. With our large sample sizes, many minor correlations are statistically significant; we focus our discussion on the more important correlations that appear to account for 20 percent or more of variation in a relationship (significant at $p < 0.0005$ for any analysis with >100 plots). This subset of the statistically significant results presumably represents the more biologically important relationships that are relevant to assessing the usefulness of lichen S as an indicator.

We used multiple regression as a tool to identify the multiple factors that are together the best predictors for lichen S. Our models are intended only to suggest combinations of the most important explanatory variables, rather than as reliable predictive models. All independent variables were measured or estimated with notable estimation error (not fixed as assumed for regression models). This is common for ecological studies and should always be remembered when interpreting the resulting regression models. Those of our environmental variables that are strongly correlated with one another (see tables 3.1 and 3.2; appendix tables A3.1 through A3.12) were never used in the same regression model. Development of regression models employed a variety of approaches; all final-reported models used forced simultaneous entry of selected significant ($p \leq 0.05$) variables to eliminate any bias from order of entry or elimination. For each regression model we report adjusted r^2, p-value, and the full model (unstandardized B coefficients) with variables listed in descending order of significance in the model (based on t-values).

We constructed two kinds of models. Minimal regression models focused on a single variable category and only the most important variables (usually $p \leq 0.0005$) were entered. Full regression models, in contrast, equally considered variables from all categories and included minor significant variables ($0.05 \geq p \geq 0.005$). Alternate regression models with similar strength (adjusted r^2) were the norm. The best regression model of each kind had the highest adjusted r^2 of alternate forced entry models. The best full model (always had the highest r^2) was designated the best overall model.

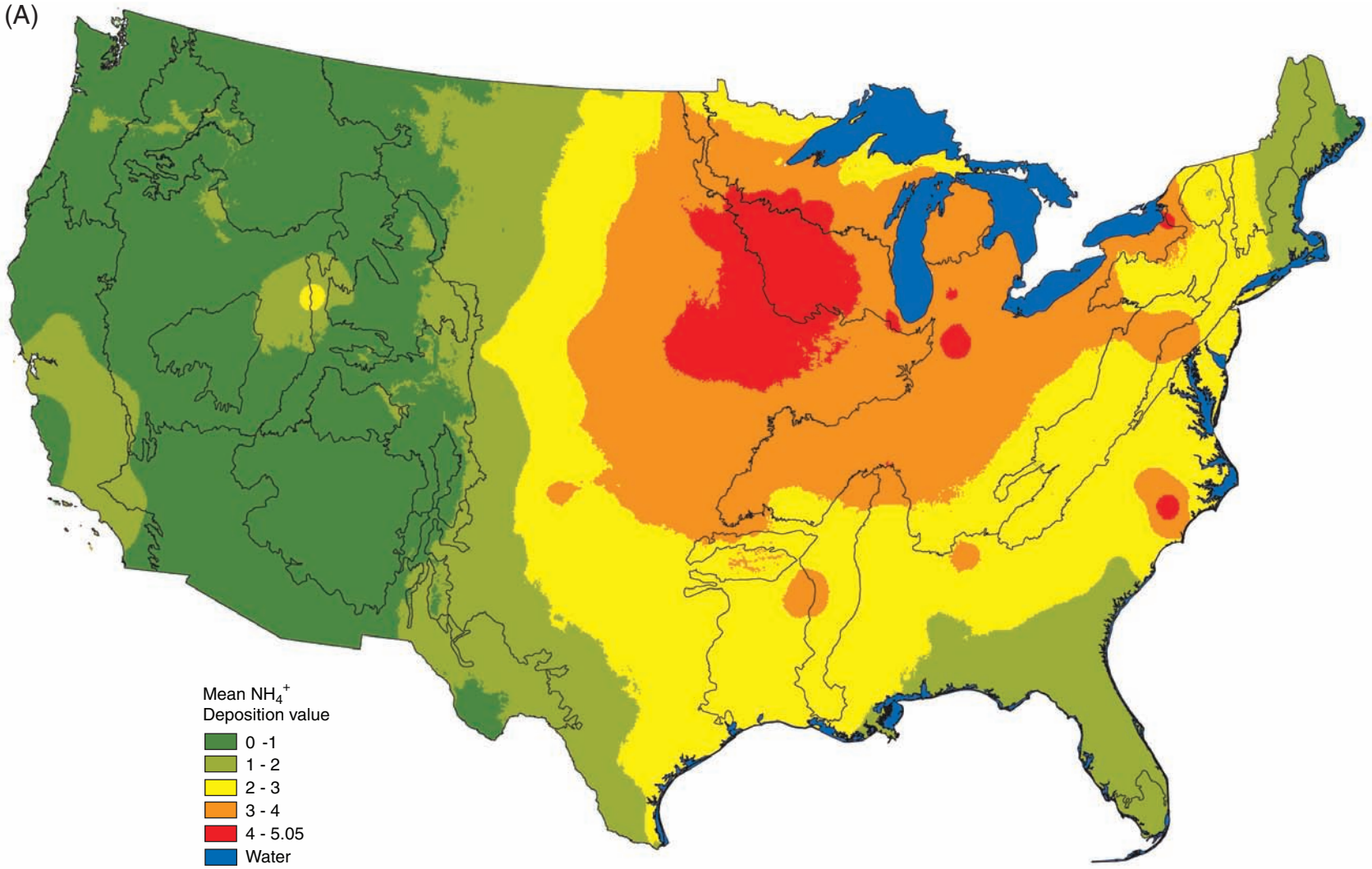


Figure 3.3—Air pollutant wet deposition (kg/ha/yr) modeled from National Atmospheric Deposition Program data. Black lines mark boundaries of ecoregion groups (see fig. 3.2). (A) NH_4^+ represents effects of agriculture; (B) NO_3^- represents urban/industrial effects. Note amounts for both are much higher in the East. (Data source: 1998-2004 average annual deposition from an interpolated 5-km^2 grid of National Atmospheric Deposition Program data [Coulston and others 2004]). *(continued on next page)*

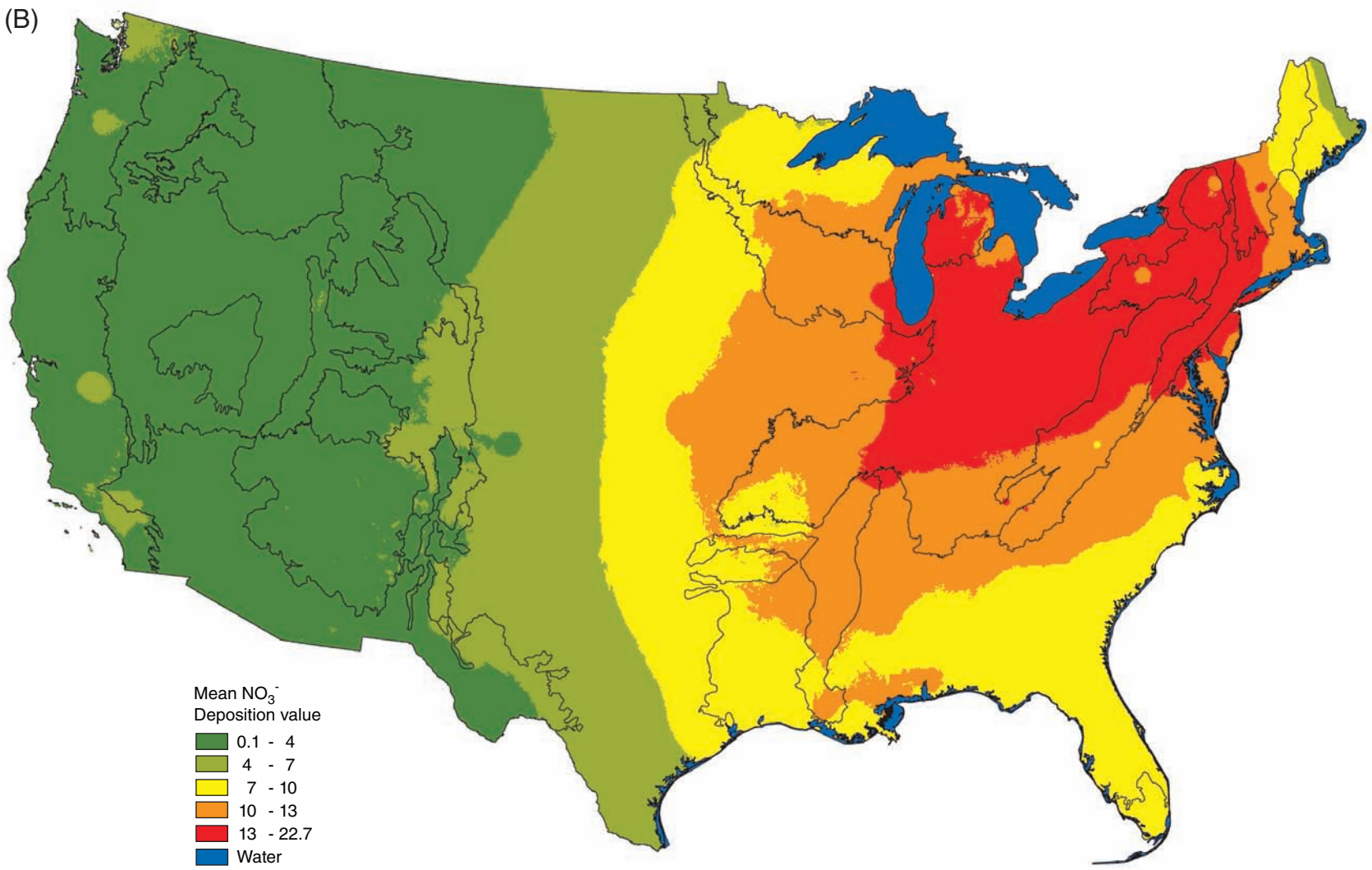


Figure 3.3 (continued)—Air pollutant wet deposition (kg/ha/yr) modeled from National Atmospheric Deposition Program data. Black lines mark boundaries of ecoregion groups (see fig. 3.2). (A) NH_4^+ represents effects of agriculture; (B) NO_3^- represents urban/industrial effects. Note amounts for both are much higher in the East. (Data source: 1998-2004 average annual deposition from an interpolated 5-km^2 grid of National Atmospheric Deposition Program data [Coulston and others 2004])

Results and Discussion

Quantitative relationships of lichen S with environmental variables for East and West regions are presented in tables 3.1 and 3.2; results for East and West groups are presented in appendix tables A3.1 through A3.12. The most important results are summarized in table 3.3. Both simple correlations and multiple regressions reflect linear relationships, while scatterplots allow us to visualize both linear and nonlinear relationships (figs. 3.4 through 3.6). Our conservative approach of reporting only the more robust relationships focuses attention on where and how lichen S may be most useful as an indicator of environmental conditions. Estimated NO_3^- and SO_4^{--} are strongly correlated with each other (Spearman $rho^2 \geq 0.50$) in both regions and all but two West ecoregion groups. Since single correlations of lichen S with NO_3^- were most often stronger than with SO_4^{--} (slightly weaker for two west ecoregion groups), we used only NO_3^- (fig. 3.3B) to represent acidic pollution.

General lichen diversity patterns—While average lichen S is higher in the East (table 3.1), overall variability and maximum lichen S are both higher in the West (table 3.2). There is of course more variation in elevation and climate in the West than in comparable geographic areas in the East. Air quality variables are more important in the East than in the West; this is probably related to the much higher levels of individual pollutants in the East (fig. 3.3, tables 3.1 and 3.2). For these large geographic scales our air pollution variables are probably correlated with other human impacts to the landscape, e.g., forest fragmentation. Therefore, we interpret relationships between lichen S and our pollution variables as potentially representing a combination of effects of air quality and other undefined human impacts on lichen S. In most cases our two forest vegetation variables, tree BA and percent softwoods, were not strong explanatory variables for our lichen diversity indicator; they show few important correlations as single variables and are only minor contributors to regression models in most cases. These findings suggest that lichen S is indeed relatively insensitive to important aspects of local variation in forest structure and is, in most areas, more responsive to large-scale variables such as climate and air quality.

Scatterplots supplement linear correlations and multiple regression results. The selected scatterplots we include (figs. 3.4 through 3.6) show that the relationships of lichen S with environmental variables often limit relationships. That is, the explanatory variable appears mostly to impose an upper limit on lichen S but no lower limit.

Table 3.3—Important patterns relating environmental variables to Lichen S and each other for East and West regions and ecoregion groups, summarized from tables 3.1 and 3.2, appendix tables A3.1-A3.5 with quantitative results for East groups, and appendix tables A3.6-A3.12 for West groups

EAST REGION:	Lichen S is higher with cleaner air, but is not linked to other variables. Summer and winter temperatures are strongly positively correlated, Regression model is weak.						
Ecoregion	Laurentian Forest	Adirondacks	Appalachian	Eastern Deciduous	Southeastern		
Average Lichen S	13.5	14.2	13.8	9.4	11.4		
Patterns	Lichen S is higher north, with cleaner air. Region has cooler temperatures, cleaner air. Regression model is strong.	Lichen S is higher north and east, with cleaner air. Region has cooler temperatures, cleaner air, all pollutants correlated. Regression model is strong.	Lichen S is higher south and west, at higher elevations, with cleaner air. Region has cool temperatures, poorer air quality, all pollutants correlated. Regression model is intermediate.	No strong patterns with Lichen S. Region has warm temperatures, poorer air quality. Regression model is weak.	No strong patterns with Lichen S. Region has warmer temperatures, cleaner air, all pollutants correlated. Regression model is very weak.		

WEST REGION:	Lichen S is higher to the north, to the west, at lower elevations, wetter, and with cleaner air. Summer and winter temperatures are weakly correlated. Regression model is intermediate.						
Ecoregion	Cascades/Coast Mountains	Sierras/Coast Mountains	Northern Rockies	Southern Rockies/ Great Basin Mountains	Great Basin Lowlands	Colorado Plateau/ Southern Dry Mountains	Arid Southwest
Average Lichen S	17.2	13.9	14.7	6.8	5.5	6.1	5.0
Patterns	No strong patterns with Lichen S. Summer & winter temperatures are weakly correlated. Regression model is very weak.	Lichen S is higher west, at lower elevations. Summer & winter temperatures are moderately correlated, N pollutants correlated. Regression model is weak.	Lichen S is higher north, west, at lower elevations, with poorer air quality. Summer & winter temperatures are weakly correlated. Regression model is intermediate.	Lichen S is higher east. Summer & winter temperatures are moderately correlated. Regression model is weak.	Lichen S is higher north. Summer & winter temperatures are weakly correlated, pollutants correlated. Regression model is intermediate.	Lichen S is higher at higher elevations, wetter, with cleaner air. Summer & winter temperatures are moderately correlated, Regression model is intermediate.	Lichen S is higher at higher elevations, wetter, cooler summer temps, with more trees. Summer & winter temperatures are weakly correlated, Regression model is strong.

Instead of a central tendency well represented by an average response of lichen S at a particular level of the variable, the distribution of lichen S is solid because many additional factors may also limit lichen S. This pattern is common in studies of organism-environment relationships in natural communities because organisms are affected simultaneously by more than the single displayed factor (Thomson and others 1996). Such a limited relationship can be observed in a scatterplot but is much more difficult to express statistically and is often not well represented by a linear correlation or regression model.

We found that in most cases the relation of lichen S to the environment differed among ecoregion groups within region and between groups and the region as a whole. This suggests that it may be necessary to have different response models for different ecoregion groups, even though this may cause difficulties estimating response for plots located near boundaries between ecoregion groups.

East Lichen Diversity Patterns—*East Region*—In the East, the strongest linear relationship is between lichen S and NO_3^-, accounting for about 20 percent of variation in lichen S (table 3.1). Maximum lichen S drops as NO_3^- rises (fig. 3.5A), and a lower proportion of plots with very low lichen S occurs where NO_3^- is below 10 kg/ha/year. The few plots with NO_3^- above 18 kg/ha/year limit our ability to make strong assertions about areas with pollution this high. A comparison of the scatterplots of lichen S vs. latitude (fig. 3.4A) with the plots of NO_3^- vs. latitude (fig. 3.5B) shows that maximum lichen S dips at the same latitudes as NO_3^- peaks (~40 to 43° N.; coincident with eastern urban/industrial corridors). Because these latter two relatively strong relationships are not linear, neither lichen S nor NO_3^- is strongly correlated with latitude (table 3.1). The stronger response of lichen S to NO_3^- than to NH_4^+ suggests that acidic pollution and urban industrial effects have so far contributed more than agricultural effects to reduced lichen S in the East.

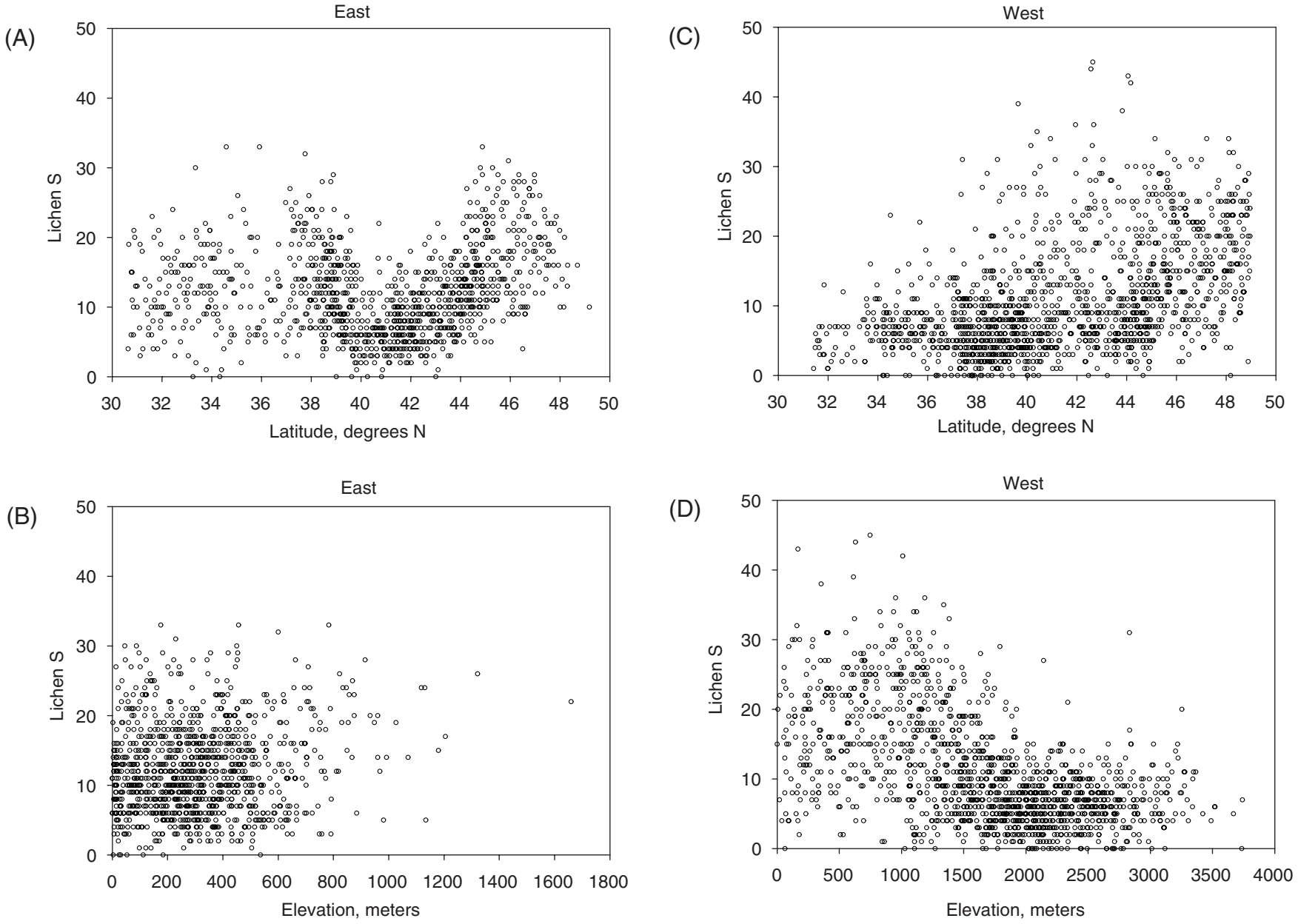


Figure 3.4—Scatterplots for geographic variables. (A) East – Maximum Lichen S dips at 40-43˚ N latitude; (B) East – Lichen S shows little relation to elevation. (C) West – Maximum Lichen S increases as latitude increases (more N); (D) West – Maximum Lichen S decreases as elevation increases. (Data sources: U.S. Department of Agriculture, Forest Service Forest Inventory and Analysis Program and the U.S. Geologic Survey)

Scatterplots indicate that for the East, elevation and climate variables are not strong single predictors of lichen S even as limiting factors (figs. 3.4B, 3.6A, 3.6B, and 3.6C); this is confirmed by weak simple correlations (table 3.1). One reason for the lack of climate pattern may be the interrelations of climate variables in the East. Summer and winter temperatures are tightly correlated across the East region (table 3.1) and in all but one East group (appendix tables A3.1 through A3.5) representing a single gradient of warm to cool. For the East region as a whole, precipitation decreases to the north as does temperature (table 3.1). This pattern suggests that higher precipitation may alleviate the drying effect of higher temperatures in the East and may lead to a narrower range of moisture conditions for plants (potential evapotranspiration) than appears from temperature or precipitation alone. The suite of East climate variables thus appears to effectively collapse for our data set to a single major (and

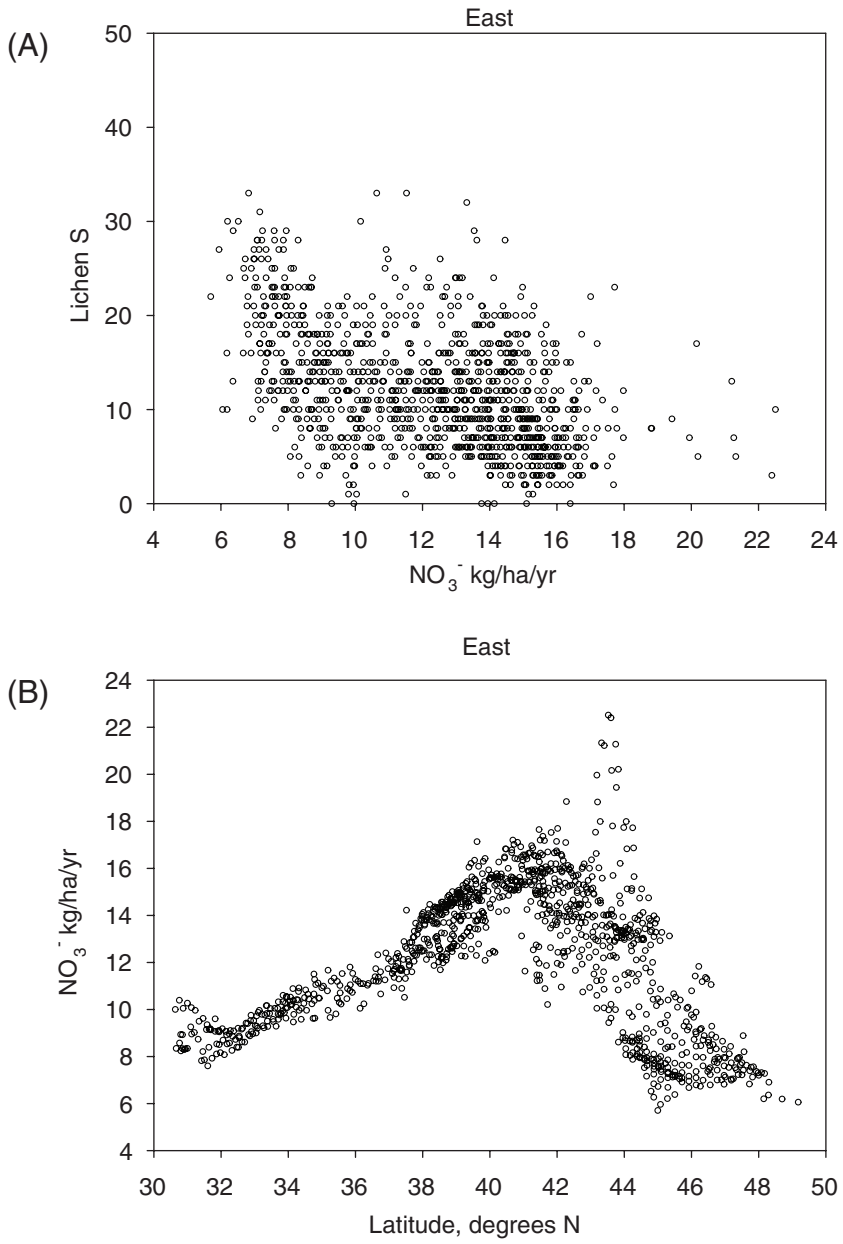
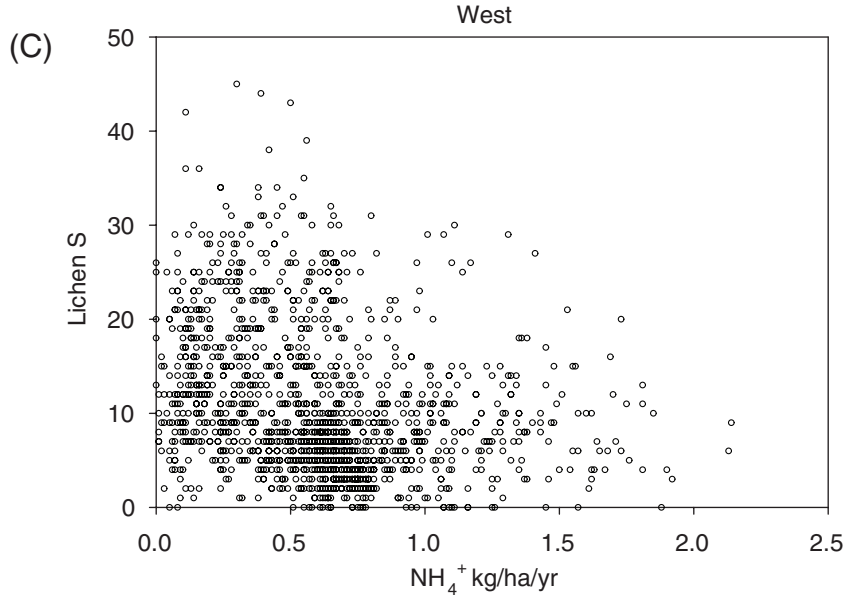
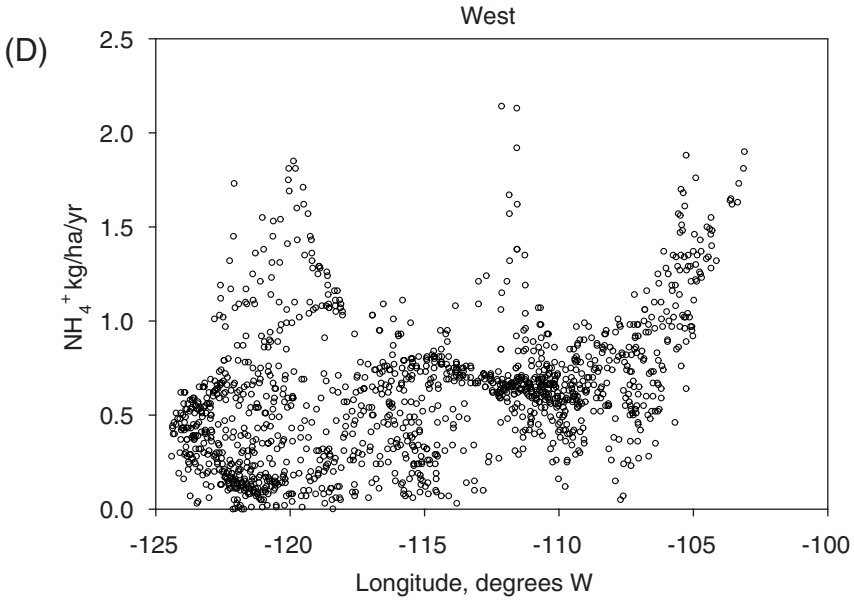


Figure 3.5—Scatterplots for pollution variables. (A) East – Maximum Lichen S decreases as NO_3^- increases; (B) East – NO_3^- mostly peaks at 40-43° N latitude (A and B together explain the pattern in figure 3.4A). (Data sources: 1998-2004 average annual deposition from an interpolated 5-km^2 grid of National Atmospheric Deposition Program data [Coulston and others 2004], and from U.S. Department of Agriculture, Forest Service Forest Inventory and Analysis Program)
(continued on next page)





relatively shallow) gradient of warm to cool climates with moderate variation in moisture. Also, all three climate variables have narrower ranges than in the West region.

For the East region, only multiple regression models including NO_3^- were reasonably informative. The minimal regression models are fairly weak. The overall best full regression model including four variables (table 3.1) explains a little more than 30 percent of variation in lichen S, and is not much better than the best minimal model ($r^2 = 0.300$, model not shown). Lichen S is "predicted" to be larger where pollution is lower, at higher elevations, and where winters are cooler.

East Ecoregion Groups—Average lichen S is higher in the mountainous Adirondacks and Appalachian ecoregion groups and in the northern Laurentian Forest group than it is in the other two groups (fig. 3.2, table 3.3; differences among groups were not evaluated

Figure 3.5 (continued)—Scatterplots for pollution variables. (C) West – Maximum Lichen S decreases as NH_4^+ increases; (D) West – Maximum NH_4^+ has a geographic pattern; it dips at longitudes corresponding to Coast, Sierra/Cascade, and Rocky Mountain ranges and peaks in lowlands. (Data sources: 1998-2004 average annual deposition from an interpolated 5-km^2 grid of National Atmospheric Deposition Program data [Coulston and others 2004], and from U.S. Department of Agriculture, Forest Service Forest Inventory and Analysis Program)

statistically). Within these three groups higher lichen S occurs at higher elevations and/or farther north, strengthening an interpretation that in the East, lichen S is higher when temperatures are cooler. The rank order of average lichen S for ecoregion groups does not match the order of average air quality for the five ecoregion groups, while it does match the order of average temperatures for them, lending further support to a temperature explanation. However, in all three cooler ecoregion groups, warmer conditions (lower elevations or more southern location) are also correlated with higher NO_3^-, because that is where urban areas and industry are concentrated. With our data, it thus remains difficult to tease apart impacts of temperature and air quality on lichen S within and even between ecoregion groups (table 3.3). Two intensive studies (McCune and others 1997, Will-Wolf and others 2006) demonstrate for Southeastern and Mid-Atlantic States (including most of the Appalachian ecoregion group) that lichen S is larger at higher elevations and/or cooler areas independent of air quality effect.

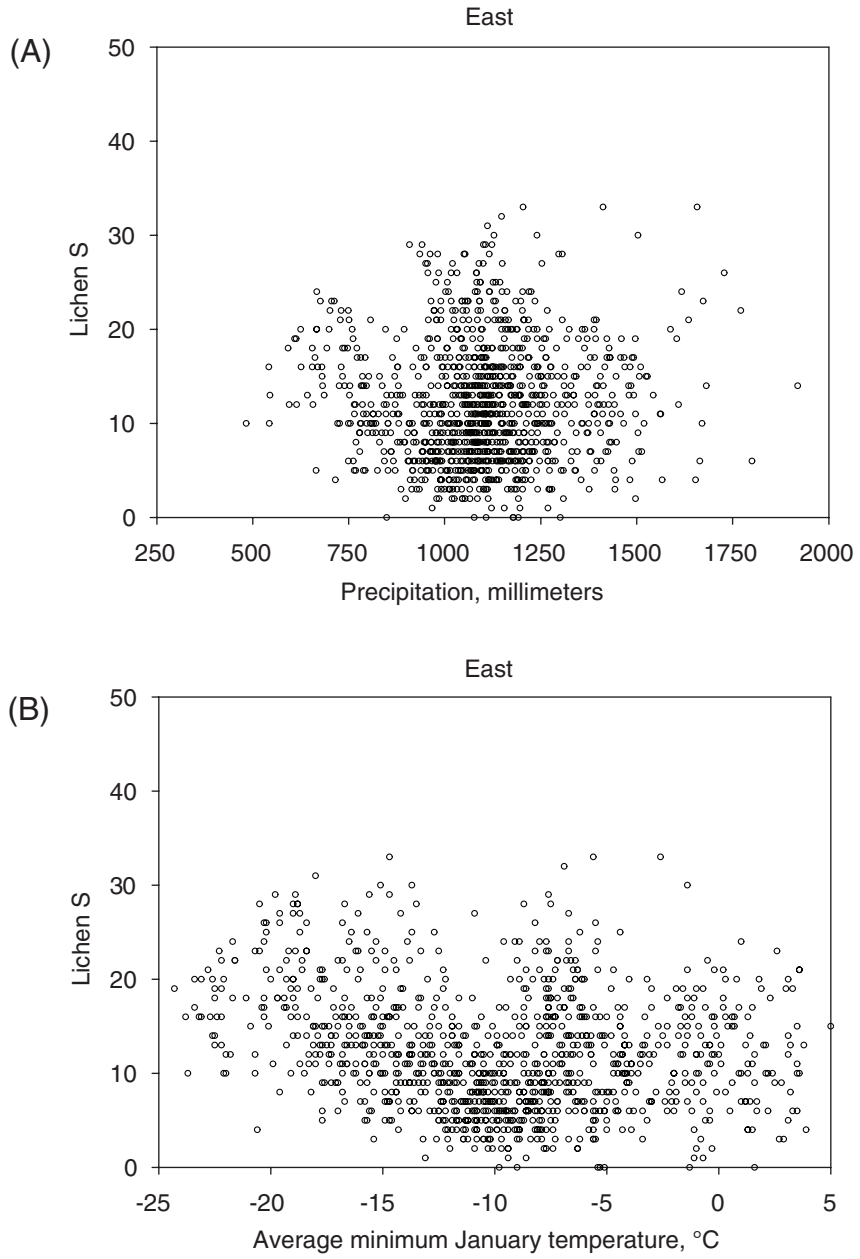


Figure 3.6—Scatterplots for climate variables. (A) East—Lichen S vs. precipitation; (B) East – Lichen S vs. average minimum January temperature. (Data sources: 30-year climate averages from Daly and Taylor 2000, and from U.S. Department of Agriculture, Forest Service Forest Inventory and Analysis Program) *(continued on next page)*

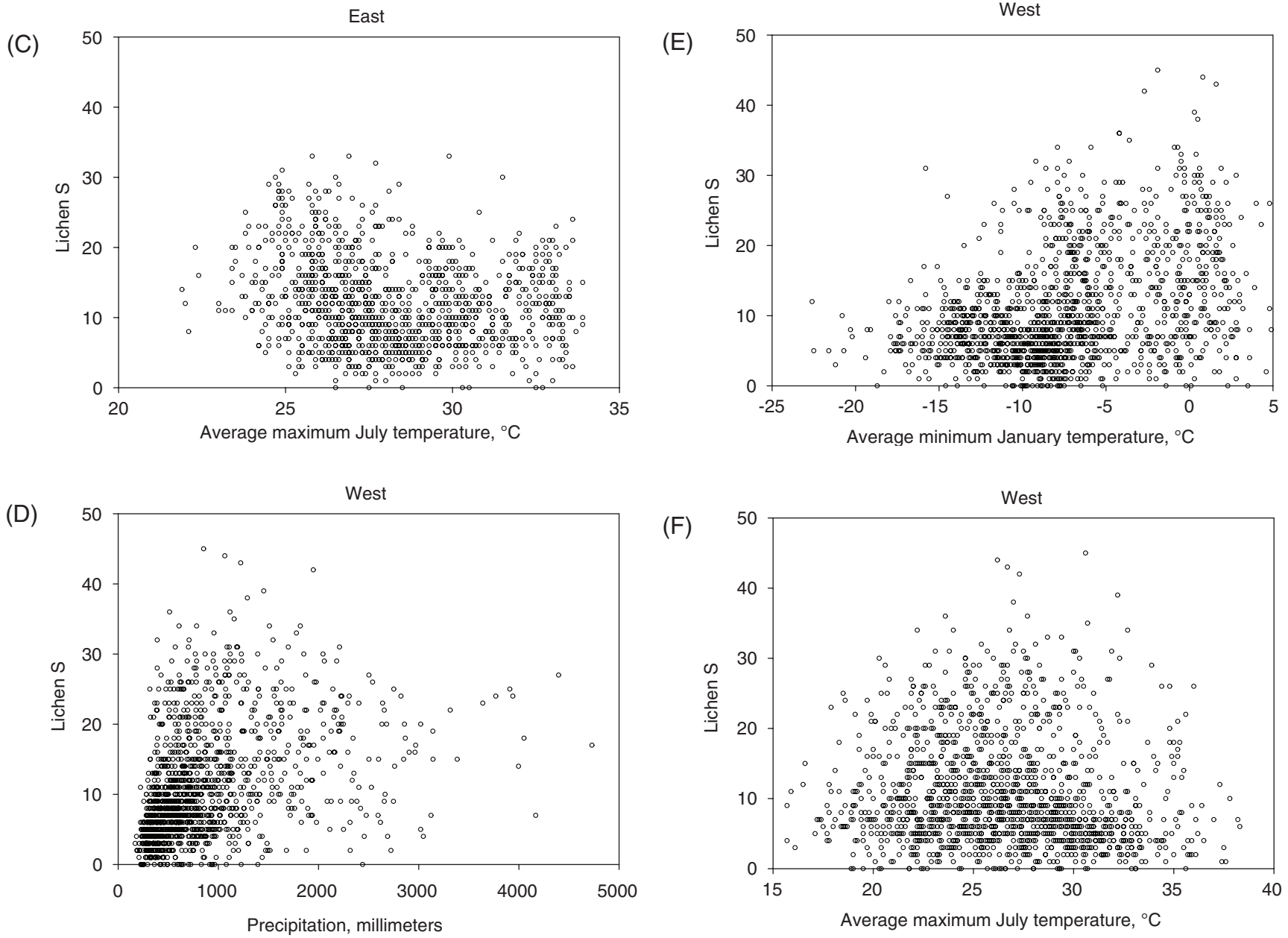


Figure 3.6 (continued)—Scatterplots for climate variables. (C) East—Lichen S vs. average maximum July temperature. Note that in the East (A, B, C) Lichen S shows little relation to any of the climate variables. (D) West—Maximum Lichen S is limited at very low average precipitation. (E) West—Maximum Lichen S increases with average minimum January temperature; (F) West—Maximum Lichen S is lower where average maximum July temperature is either lower or higher. (Data sources: 30-year climate averages from Daly and Taylor 2000, and from U.S. Department of Agriculture, Forest Service Forest Inventory and Analysis Program)

If lichen diversity is indeed naturally lower where it is warmer in several parts of the East, the higher levels of pollutants (associated with lower lichen S) in many of those same warmer areas may have further reduced lichen diversity there. The combined effect of air quality and temperature would thus effectively increase the range of higher to lower lichen biodiversity across the East region.

Air quality is lower to the south within the Laurentian Forest, Adirondacks, and Eastern Deciduous ecoregion groups (appendix tables A3.1, A3.2, and A3.4) and is lower to the north within the Appalachian and southeastern ecoregion groups (appendix tables A3.3 and A3.5). All five of these patterns within ecoregions reflect increased midlatitude pollution (in either the Ohio Valley urban/industrial corridor or the midlatitude east coast urban/industrial region) that produces the pattern on the scatterplot of NO_3^- vs. latitude in figure 3.5B.

The Eastern Deciduous and Southeastern ecoregion groups show weak relationships of all environmental variables with lichen S, and the best regression models explain small proportions of the variation in lichen S (table 3.3). Both are quite large geographic areas (fig. 3.2) that have low within-group variation in climate and pollution as compared with other East ecoregion groups. A reasonable interpretation is that relationships of lichens to environment are more subtle, probably reflected as variation in species composition rather than in lichen S for these two ecoregion groups.

West Lichen Diversity Patterns—*West Region*—In contrast to the East, scatterplots in the West region showed strong responses of lichen S to the environment, and this was confirmed by correlations (table 3.2). Maximum lichen S increases linearly with latitude (fig. 3.4C) and decreases linearly with elevation (fig. 3.4D), both supported by strong correlations (table 3.2). Lichen S is also significantly higher to the West. Very low precipitation coincides with low lichen S (fig. 3.6D), while maximum lichen S increases steeply as precipitation increases for sites near the drier end of the precipitation gradient, reflected in a moderate positive correlation (table 3.2). At precipitation levels above 800 to 1000 mm/year, maximum

lichen S no longer increases with precipitation, suggesting precipitation limits lichen S only below 1000 mm/year. Winter temperatures may also affect lichen S in the West; the upper limit of lichen S increases as January temperatures increase (fig. 3.6E). It appears that maximum July temperature might limit lichen S at both ends; the upper limit of lichen S is highest at midrange, declining with departure to both lower and higher average maximum temperatures (fig. 3.6F). Such essentially nonlinear temperature patterns are not reflected in strong correlations (table 3.2).

Even though in the West region maximum NH_4^+ levels are low (table 3.1), NH_4^+ deposition still appears to limit maximum lichen S (fig. 3.5C). This is the strongest relationship between lichen S and air quality for the West region and is confirmed by a moderate negative correlation (table 3.2). The scatterplot of NH_4^+ vs. longitude (fig. 3.5D) suggests strong geographic variation in levels of NH_4^+, explaining perhaps why the negative correlation of lichen S with NH_4^+ is not stronger (table 3.2). The longitudes of peaks for NH_4^+ levels in this scatterplot correspond roughly to lowlands where most agriculture occurs, while the longitudes of troughs correspond to north-south mountain ranges.

Minimal regression models are generally stronger for the West than for the East. The overall best full regression model including five variables is notably better than the best minimal model (both shown in table 3.2), explaining about 40 percent of variation in lichen S. Two variables without important single correlations contribute significantly to the best model: total plot BA and average maximum July temperature.

Climate variables have wide ranges for the West region and strong differences between West ecoregion groups, resulting in strong and varied correlations between climate and geography (table 3.2, appendix tables A3.6 through A3.12). Summer and winter temperatures are not strongly correlated with one another in the West region nor in several of the West groups. Precipitation is either negatively correlated with temperature (West region and some groups) or is independent of temperature in some ecoregion groups. For the former groups, lower precipitation where temperatures are warmer exaggerates the drying effect of higher temperatures, giving a wider range of moisture conditions (potential evapotranspiration) for plants than indicated from either precipitation or temperature alone across the same area. Thus, there is no single climate gradient for the

West and for West regions. All three climate variables must be considered independently, and they relate differently to each other and to geographic variables in the various ecoregion groups. Geographic variables representing integrated climate conditions often have stronger patterns with lichen S than do the direct climate variables (tables 3.2 and 3.3) in our data sets.

In the West region and several ecoregion groups, lichen S is higher in lower elevation areas where air quality also is often lower. Thus, the impact of air quality in these West areas would be to reduce lichen S in higher diversity areas and decrease the overall range of lichen S (as opposed to increasing its range in some East ecoregion groups).

West Ecoregion Groups—For four of the West ecoregion groups, the best full regression model supports further use of lichen S (strong or intermediate in table 3.3), predicting almost 40 percent or more of variation in lichen S (appendix tables A3.8 and A3.10 through A3.12). Precipitation is the strongest contributor to regression models for the Arid Southwest and Colorado Plateau/Southern Dry Mountains groups that have the driest climates and the lowest maximum lichen S. For the Northern Rockies and Great Basin Lowlands ecoregion groups, geographic variables contribute more than do climate variables to best regression models (see discussion above under West Region). The wide variation between ecoregion groups in correlations between environmental variables is probably the reason relations of lichen S to explanatory variables are so varied between regions (table 3.3). It seems clear that better predictive models for different regions will need to be developed independently.

For three West ecoregion groups the best regression model explains relatively little variation in lichen S. Regression models for the Sierras/Coast Mountains and the Southern Rockies/Great Basin Mountains ecoregion groups explain 20 to 30 percent of variation in lichen S; each region also has a strong geographic variable (table 3.3, appendix tables A3.9 and A3.12). Jovan and McCune (2005, 2006) and Jovan (2008) found strong relationships of lichen species composition and some response of lichen S with similar explanatory variables for the Sierras/Coast Mountains ecoregion in more intensive studies covering smaller areas. The Cascades/Coast Mountains ecoregion group, with the highest average lichen S in our entire study (fig. 3.2, table 3.3), shows extremely weak correlations of our environmental variables with lichen S. Tree BA, the strongest, is weakly correlated with lichen S, and the best full regression model, including tree BA and NH_4+ pollution, predicts

only 11 percent of the variation in lichen S (appendix table A3.6). Intensive studies of lichen communities in this ecoregion group using FIA data (Geiser and Neitlich 2007, Jovan 2008, Will-Wolf and others 2006) confirm that here lichen S does not vary with environmental variables; one must examine species composition to explore response to environment.

East-West Comparisons—If pollution and other human activities increase, they will likely occur more at lower elevations in both the East and West regions, but they may well have quite different broad impacts on lichen diversity in the two regions. The occurrence of higher lichen S at higher elevations in mountainous East ecoregion groups means that the more species-rich areas will be less affected by increased human activities. In contrast, higher lichen S at lower elevations in most mountainous West ecoregion groups means that the more species-rich and vulnerable areas will also be more affected by increased human activities, perhaps resulting in much greater reduction in regional biodiversity for lichens in the West than in the East.

Some notable differences in climate between West and East may have important implications for developing indicators of response to climate. In the East, where summer and winter temperatures are positively correlated in most areas and links between climate and geography are similar between areas, if lichen S can be shown to be correlated with some new climate variable such as evapotranspiration, it can be expected to be useful across scales and between ecoregion groups. Calibration of lichen S response between areas will be straightforward.

For the West, in contrast, the lack of a single pattern of correlations between climate variables and the varied links between climate and geography variables pose distinct challenges for developing lichen S as a climate indicator. Geographic variables that presumably integrate the effects of climate are currently the strongest predictors of lichen S in the West region and most ecoregion groups. However, the very different combinations of how climate variables are linked to geographic variables in the West region versus ecoregion groups (between scales) and between ecoregion groups suggest that as climate changes, the relation between climate and geography may also change in unexpected ways. For example, in one ecoregion the predictive power of elevation for lichen S might currently relate most to variation in winter temperatures, while in an adjacent ecoregion the predictive power of elevation might relate most to summer temperatures that are weakly correlated with winter temperatures. In the first region, after some period of climate change, its own original relationships might be replaced by

those for the adjacent region. Use of elevation as the predictor for lichen S indicating climate response in the first ecoregion would need to be developed with such possibilities in mind, so as to avoid misinterpretations when using lichen S for trend analysis.

Further Studies—Followup studies for particular regions and ecoregion groups should address a number of issues to improve the use of lichen S as an indicator. Suggestions for each region are summarized in table 3.4. For any region, additional climate variables such as potential evapotranspiration should be explored to see if stronger climate signals emerge. Variables that represent other human impacts, such as intensity of agricultural and reduction in percent forest near plots, can be developed and tested. Variables that represent other aspects of forest structure, such as number of tree species, average tree size, or proportion of early versus late-successional tree species, can be developed from FIA tree data and tested to further explore relationships with local plot conditions. More complicated models including covariates, nonlinear terms, and interaction terms can also be explored for ecoregion groups. For more in-depth analyses of ecoregion groups, data sets should perhaps have more equal representation of plots across the geographic area under study. For the most promising regions, cross-indicator analyses should also be considered. After final models for adjacent ecoregion groups have been completed, between-group studies should explore how to best interpret response at plots near boundaries.

For three of five East ecoregion groups, lichen S has potential to be a useful indicator for response to both pollution and climate if variables representing these factors and also independent of each other can be developed. Developing such variables requires more intensive studies tailored to the particular region. Explanatory climate variables seem to behave similarly enough between East ecoregion groups that approaches to improving predictive models for one East ecoregion group may be applicable to at least some other groups.

Conclusion

Lichen S, the number of lichen species found at an FIA plot, shows a number of strong relationships to environmental variables across the coterminous United States. Our findings confirm that forest structure has relatively little impact on lichen S at such large geographic scales. For some ecoregion groups within the East and West, lichen S is correlated with few of our variables, and the best regression model is weak. Lichen S alone currently shows no potential as a useful indicator there, probably because important relationships in those regions must be linked to relative abundance of different lichen species.

Table 3.4—Summary of the potential usefulness of Lichen S as an environmental indicator for East and West regions and ecoregion groups, and suggested avenues for future research. Potential usefulness is extrapolated from results summarized in table 3.3

EAST REGION:	Lichen S is potentially useful as an indicator of response to air quality but not to other environmental factors as represented by current variables.						
Next steps for research	Derive a potential evapotranspiration variable; investigate its relationship with Lichen S. Evaluate relationships between air pollution variables and other variables representing human impacts such as forest fragmentation, agriculture, and urbanization to improve interpretation of Lichen S response. Explore other local forest structure variables.						
East Ecoregions:	Laurentian Forest	Adirondacks	Appalachian	Eastern Deciduous	Southeastern		
Usefulness of Lichen S	Moderate – response to air quality & temperature	Moderate – response to air quality & temperature	Moderate – response to air quality & temperature	Low	Low		
Next steps for research	Evaluate response to air quality vs climate	Evaluate response to air quality vs climate	Evaluate response to air quality vs climate	Explore evapotranspiration variable, other human impact variables	Explore evapotranspiration variable, other human impact variables		
WEST REGION:	Lichen S is potentially useful as an indicator of climate and of response to air quality after accounting for climate.						
Next steps for research	Investigate combining Lichen S with other region-wide biological indicators of response to climate for a cross-indicator analysis. Derive a potential evapotranspiration variable; investigate its relationship with Lichen S. Explore other local forest structure variables.						
West Ecoregions:	Cascades/Coast Mountains	Sierras/Coast Mountains	Northern Rockies	Southern Rockies/Great Basin Mountains	Great Basin Lowlands	Colorado Plateau/ Southern Dry Mountains	Arid Southwest
Usefulness of Lichen S	Low	Low	Moderate – response to winter temperatures	Low	Moderate – response to air quality & latitude	Moderate – response to longitude	High – response to precipitation & summer temperatures
Next steps for research	Not needed; other research available	Not needed; other research available	Investigate cross-indicator analyses	Explore evapotranspiration variable	See region next steps. Evaluate response to air quality vs climate	See region next steps. Evaluate response to air quality vs climate.	See region next steps

The fact that pollution and not environment has the strongest correlation with lichen S in the East region suggests that air pollution representing human impact has already had a measurable negative impact on lichen biodiversity across the East. For the East as a whole, it appears lichen S may be a useful indicator for response to pollution and other human impacts but not for response to changes in climate. For three of five East ecoregion groups, lichen S may be a useful indicator for response to both pollution and climate, but separating the two factors is difficult.

The lack of strong correlations between lichen S and pollution variables in the West region suggest that human activities are not yet having a major impact on lichen biodiversity across the region as a whole, though some relationships between lichen S and pollution suggest vulnerability. If pollution and other human impacts increase in the West, increases will likely occur more at lower elevations where lichen S is usually higher. Thus, they may impact the most diverse lichen communities and have a stronger than expected impact on regional biodiversity. For the West as a whole and for four of seven West ecoregion groups, lichen S has potential as a useful indicator for response to climate and possibly for response to pollution as well.

Some notable differences in climate between the West and East have important implications for developing indicators of response to climate. In the East, with a more uniform climate, an improved climate variable can be anticipated to be broadly useful to link with lichen S. For the West, in contrast, the lack of a single pattern of correlations between climate variables and the varied links between climate and geography variables pose distinct challenges for developing lichen S as a climate indicator. Different models for each ecoregion will likely be needed, and calibrating between regions may be difficult.

Strong response patterns have emerged even though lichen S is affected by many sources of variation in addition to those of interest to us. It appears that despite these limitations and probably because the FIA lichen sample protocol is designed to average across much local variation, this relatively imprecise lichen diversity indicator does indeed have potential for tracking response to pollution and/or environment. The variation we found shows each area to be studied needs its own separate model for how lichen S responds to environmental factors. Our analyses provide useful baselines for comparison with resurveys and trend analysis for the future and highlight logical next steps for developing better models.

Acknowledgments

Borys Tkacz, Forest Health Monitoring National Program Manager, and Sarah Jovan, FIA lichen indicator coadvisor, made many helpful comments and critiques on drafts of this chapter, as did two anonymous reviewers. The final result is the responsibility of the authors.

Literature Cited

Bailey, R.G. 1989. Explanatory supplement to ecoregions map of the continents. Environmental Conservation. 16: 307–309.

Bechtold, William A.; Patterson, Paul L., eds. 2005. The enhanced forest inventory and analysis program - national sampling design and estimation procedures. Gen. Tech. Rep. SRS–80. Asheville, NC: Forest Service, U.S. Department of Agriculture, Southern Research Station. 85 p.

Cleland, D.T.; Freeouf, J.A.; Keys, J.E. [and others]. 2005. Ecological subregions: sections and subsections for the conterminous United States. June 14, 2005 version. Washington, DC: Forest Service, U.S. Department of Agriculture. [Map, presentation scale 1: 3,500,000; colored].

Coulston, J.W.; Riitters, K.H.; Smith, G.C. 2004. A preliminary assessment of Montréal process indicators of air pollution for the United States. Environmental Monitoring and Assessment. 95: 57–74.

Daly, C.; Taylor, G. 2000. United States average monthly or annual precipitation, temperature, and relative humidity 1961–90. Arc/INFO and ArcView coverages. Corvallis, OR: Spatial Climate Analysis Service, Oregon State University.

Geiser, L.H.; Neitlich, P. 2007. Air pollution and climate gradients in western Oregon and Washington indicated by epiphytic macrolichens. Environmental Pollution. 145: 203–218.

Jovan, S. 2008. Lichen bioindication of biodiversity, air quality, and climate:baseline results from monitoring in Washington, Oregon, and California. Gen. Tech. Rep. PNW–GTR–737. Portland, OR: Forest Service, U.S. Department of Agriculture, Pacific Northwest Research Station. 115 p.

Jovan, S. 2009. Baseline results from the lichen community indicator in the Pacific Northwest: air quality patterns and evidence of a nitrogen problem. In: Ambrose, M.J.; Conkling, B.L., eds. Forest health monitoring 2006 national technical report. Gen. Tech. Rep. SRS–117. Asheville, NC: Forest Service, U.S. Department of Agriculture, Southern Research Station: 35–46. Chapter 5.

Jovan, S.; McCune, B. 2005. Air-quality bioindication in the greater central valley of California, with epiphytic macrolichen communities. Ecological Applications. 15: 1712–1726.

Jovan, S.; McCune, B. 2006. Using epiphytic macrolichen communities for biomonitoring ammonia in forests of the greater Sierra Nevada, California. Water, Air, and Soil Pollution. 170: 69–93.

McCune, B. 2000. Lichen communities as indicators of forest health. Bryologist. 103: 353–356.

McCune, B.; Dey, J.; Peck, J. [and others]. 1997. Regional gradients in lichen communities of the Southeast United States. Bryologist. 100: 145–158.

Nimis, P.L.; Scheidegger, C.; Wolseley, P. 2002. Monitoring with lichens – monitoring lichens. NATO Sci. Ser. The Hague, The Netherlands: Kluwer Academic Publishers. 408 p.

Sokal, R.R.; Rohlf, F.J. 1995. Biometry. 3^{d} ed. New York: W.H. Freeman. 887 p.

Thomson, J.D.; Weiblen, G.; Thomson, B.A. [and others]. 1996. Untangling multiple factors in spatial distributions: lilies, gophers, and rocks. Ecology. 77: 1698–1715.

Forest Service, U.S. Department of Agriculture. 2007. Forest inventory and analysis national program library. Overview and forest health indicator factsheets. http://fia.fs.fed.us/library/fact-sheets/. [Date accessed: June].

U.S. Geological Survey. 1999. National elevation dataset. http://ned.usgs.gov/. [Date accessed: September 2006].

Will-Wolf, S.; Esseen, P.-A.; Neitlich, P. 2002. Monitoring biodiversity and ecosystem function: forests. In: Nimis, P.L.; Scheidegger, C.; Wolseley, P.A., eds. Monitoring with lichens-monitoring lichens. Dordrecht, Netherlands: Kluwer Academic Publishers: 203–222.

Will-Wolf, S.; Geiser, L.H.; Neitlich, P.; Reis, A. 2006. Forest lichen communities and environment - how consistent are relationships across scales? Journal of Vegetation Science. 17: 171–184.

Appendix

Table A3.1—Laurentian Forest ecoregion group in the East region

Variable	Range (average ± sd)	Spearman correlations (*rho*) (Bailey's province codes = 211, 212; N = 278[a]; years = 1994–2004)										
		Lichen S	Lat.	Long.	Elev.	Precip.	Max. July temp.	Min. Jan. temp.	NO_3^-	SO_4^{--}	NH_4^+	Tree BA
Lichen S	2 – 33 (average 13.5 ± 6.8 sd)											
Latitude (decimal degrees, S to N)	41.14 – 49.18	**0.597**										
Longitude (decimal degrees, W to E)	–95.31 – –67.11	—	—									
Elevation (m)	6 – 818 (336 ± 178)	—	—	**–0.562**								
Precipitation (mm)	542 – 1609 (967 ± 179)	—	**–0.507**	**0.765**	—							
Max. July temp. (°C)	22.0 – 29.4 (26.4 ± 1.00)	—	—	—	—	—						
Min. Jan. temp. (°C)	–8.9 – –24.3 (–14.8 ± 3.83)	**–0.546**	**–0.903**	—	—	—	—					
NO_3^- (kg/ha/year)	5.70 – 22.40 (11.67 ± 3.99)	**–0.687**	**–0.771**	—	**0.477**	—	—	**0.656**				
SO_4^{--} (kg/ha/year)	4.32 – 25.31 (13.22 ± 5.71)	**–0.633**	**–0.940**	—	—	**0.502**	—	**0.845**	**0.869**			
NH_4^+ (kg/ha/year)	0.70 – 4.23 (2.52 ± 0.90)	—	—	**–0.790**	—	**–0.512**	—	—	**0.477**	—		
Tree BA (square foot per acre)	22.1 – 330.8 (96.3 ± 48.5)	—	—	—	—	—	—	—	—	—	—	
Percent BA softwoods	0 – 100 percent (30.9 ± 35.4)	—	—	—	—	—	—	—	**–0.481**	—	—	—
Best overall full regression model		**r^2 = 0.527, p < 0.0005: Lichen S = 32.913 –0.717 NO_3^- + 1.359 lat. (+ 0.146 long. + 0.006 precip.)** Lichen S larger with less acidic air pollution, more north (more east, wetter).										

— = cells with dashes indicate correlations weaker than the cutoff; max. = maximum; min. = minimum; sd = standard deviation; lat. = latitude; long. = longitude; elev. = elevation; precip. = precipitation; temp. = temperature; NO_3^- = nitrate; SO_4^{--} = sulfate; NH_4^+ = ammonium; BA = basal area; S = south; N = north; W = west; E = east; m = meter; mm = millimeter; °C = degree Celsius; kg = kilogram; ha = hectare. Range and intercorrelations of Lichen S plus geography, climate, pollution, and forest structure variables; the best regression model is given at bottom. Positive correlations are in black; negative are in red. Single correlations with *rho* > 0.447 (in bold, all with p < 0.0005) account for at least 20 percent of variation; correlations with Lichen S stronger than this cutoff suggest potential for an indicator. Correlations between environmental variables with *rho* ≥ 0.775 (at least 60 percent of variation accounted for) indicate those two variables should not be entered into the same regression model. The best overall regression model is included with contributing variables listed in descending order of significance; minor variables ($p \leq 0.05$ but usually $p \geq 0.005$) are listed in parentheses.

[a] Removed E plots with total live-tree BA < 21.8966 square feet per acre (< 5 m^2/ha).

Table A3.2—Adirondacks ecoregion group in the East region

Variable	Range (average ± sd)	Spearman correlations (*rho*) (Bailey's province code = M211; N = 152[a]; years = 1994–2004)										
		Lichen S	Lat.	Long.	Elev.	Precip.	Max. July temp.	Min. Jan. temp.	NO_3^-	SO_4^{--}	NH_4^+	Tree BA
Lichen S	3 – 31 (average 14.2 ± 5.8 sd)											
Latitude (decimal degrees, S to N)	42.07 – 47.40	**0.534**										
Longitude (decimal degrees, W to E)	–75.30 – –68.69	**0.574**	**0.712**									
Elevation (m)	155 – 960 (429 ± 152)	—	—	**–0.601**								
Precipitation (mm)	812 – 1681 (1118 ± 147)	—	—	—	**0.504**							
Max. July temp. (°C)	21.9 – 28.0 (25.3 ± 1.28)	—	**–0.616**	—	—	—						
Min. Jan. temp. (°C)	–10.9 – –20.4 (–16.1 ± 2.13)	**–0.486**	**–0.877**	—	—	—	**0.720**					
NO_3^- (kg/ha/year)	7.16 – 17.79 (11.58 ± 2.66)	**–0.544**	**–0.795**	**–0.951**	—	—	—	**0.668**				
SO_4^{--} (kg/ha/year)	7.81 – 18.82 (12.64 ± 3.04)	**–0.534**	**–0.884**	**–0.870**	—	—	**0.564**	**0.784**	**0.940**			
NH_4^+ (kg/ha/year)	1.12 – 3.22 (1.93 ± 0.48)	**–0.497**	**–0.625**	**–0.940**	—	—	—	**0.561**	**0.925**	**0.871**		
Tree BA (square foot per acre)	24.1 – 283.0 (104.0 ± 52.1)	—	—	—	—	—	—	—	—	—	—	
Percent BA softwoods	0 – 100 percent (39.1 ± 33.5)	—	—	—	—	—	—	—	—	—	—	—
Best overall full regression model		**r^2 = 0.429, p < 0.0005: Lichen S = 38.491 + 1.261 long. (+ 1.587 lat.)** Lichen S larger more east, (more north).										

— = cells with dashes indicate correlations weaker than the cutoff; max. = maximum; min. = minimum; sd = standard deviation; lat. = latitude; long. = longitude; elev. = elevation; precip. = precipitation; temp. = temperature; NO_3^- = nitrate; SO_4^{--} = sulfate; NH_4^+ = ammonium; BA = basal area; S = south; N = north; W = west; E = east; m = meter; mm = millimeter; °C = degree Celsius; kg = kilogram; ha = hectare.

Range and intercorrelations of Lichen S plus geography, climate, pollution, and forest structure variables; the best regression model is given at bottom. Positive correlations are in black; negative are in red. Single correlations with *rho* > 0.447 (in bold, all with p < 0.0005) account for at least 20 percent of variation; correlations with Lichen S stronger than this cutoff suggest potential for an indicator. Correlations between environmental variables with *rho* ≥ 0.775 (at least 60 percent of variation accounted for) indicate those two variables should not be entered into the same regression model. The best overall regression model is included with contributing variables listed in descending order of significance; minor variables (p ≤ 0.05 but usually p ≥ 0.005) are listed in parentheses.

[a] Removed E plots with total live-tree BA < 21.8966 square feet per acre (< 5 m^2/ha).

Table A3.3—Appalachian ecoregion group in East region

Variable	Range (average ± sd)	Spearman correlations (*rho*) (Bailey's province code = M221; N = 140[a]; years = 1994–2005)										
		Lichen S	Lat.	Long.	Elev.	Precip.	Max. July temp.	Min. Jan. temp.	NO_3^-	SO_4^{--}	NH_4^+	Tree BA
Lichen S	1 – 33 (average 13.8 ± 7.7 sd)											
Latitude (decimal degrees, S to N)	34.77 – 41.24	–0.506										
Longitude (decimal degrees, W to E)	–84.14 – –75.75	–0.480	**0.907**									
Elevation (m)	152 – 1660 (598 ± 263)	**0.457**	–0.539	–0.542								
Precipitation (mm)	894 – 1919 (1154 ± 188)	—	—	–0.499	**0.513**							
Max. July temp. (°C)	24.1 – 30.4 (27.9 ± 1.47)	—	—	—	–0.647	—						
Min. Jan. temp. (°C)	–3.4 – –10.5 (–7.5 ± 1.50)	—	–0.758	–0.657	—	—	**0.555**					
NO_3^- (kg/ha/year)	10.48 – 17.20 (13.80 ± 1.67)	–0.498	**0.933**	**0.749**	–0.531	—	—	–0.681				
SO_4^{--} (kg/ha/year)	16.03 – 24.33 (20.08 ± 1.90)	—	**0.837**	**0.614**	—	—	—	–0.656	**0.926**			
NH_4^+ (kg/ha/year)	2.09 – 3.42 (2.67 ± 0.26)	–0.518	**0.845**	**0.750**	–0.675	—	—	–0.460	**0.851**	**0.768**		
Tree BA (square foot per acre)	23.0 – 500.7 (103.2 ± 58.9)	—	—	—	—	—	—	—	—	—	—	
Percent BA softwoods	0 – 95.48 percent (7.8 ± 18.1)	—	—	—	—	—	—	—	—	—	—	—
Best overall full regression model		**r^2 = 0.373, p < 0.0005: Lichen S = 207.782 –5.226 lat. (– 0.293 Min. Jan. – 0.012 precip.)** Lichen S larger more south, (cooler winters, drier).										

— = cells with dashes indicate correlations weaker than the cutoff; max. = maximum; min. = minimum; sd = standard deviation; lat. = latitude; long. = longitude; elev. = elevation; precip. = precipitation; temp. = temperature; NO_3^- = nitrate; SO_4^{--} = sulfate; NH_4^+ = ammonium; BA = basal area; S = south; N = north; W = west; E = east; m = meter; mm = millimeter; °C = degree Celsius; kg = kilogram; ha = hectare. Range and intercorrelations of Lichen S plus geography, climate, pollution, and forest structure variables; the best regression model is given at bottom. Positive correlations are in black; negative are in red. Single correlations with *rho* > 0.447 (in bold, all with p < 0.0005) account for at least 20 percent of variation; correlations with Lichen S stronger than this cutoff suggest potential for an indicator. Correlations between environmental variables with *rho* ≥ 0.775 (at least 60 percent of variation accounted for) indicate those two variables should not be entered into the same regression model. The best overall regression model is included with contributing variables listed in descending order of significance; minor variables (p ≤ 0.05 but usually p ≥ 0.005) are listed in parentheses.

[a] Removed E plots with total live-tree BA < 21.8966 square feet per acre (< 5 m^2/ha).

Table A3.4—Eastern Deciduous ecoregion group in the East region

Variable	Range (average ± sd)	Spearman correlations (*rho*) (Bailey's province codes = 221, 222, 223; N = 266[a]; years = 1994–2005)										
		Lichen S	Lat.	Long.	Elev.	Precip.	Max. July temp.	Min. Jan. temp.	NO_3^-	SO_4^{--}	NH_4^+	Tree BA
Lichen S	0 – 21 (average 9.4 ± 4.5 sd)											
Latitude (decimal degrees, S to N)	34.79 – 48.30	—										
Longitude (decimal degrees, W to E)	–96.05 – –70.42	—	**0.450**									
Elevation (m)	7 – 584 (250 ± 111)	—	—	—								
Precipitation (mm)	545 – 1378 (1064 ± 136)	—	—	**0.533**	—							
Max. July temp. (°C)	26.0 – 32.4 (28.6 ± 1.20)	—	**–0.821**	**–0.516**	—	—						
Min. Jan. temp. (°C)	–1.9 – –22.5 –9.3 ± 2.92	—	**–0.881**	**–0.657**	—	—	**0.728**					
NO_3^- (kg/ha/year)	6.36 – 22.51 (14.1 ± 2.18)	–0.253	**–0.601**	—	**0.450**	—	—	—				
SO_4^{--} (kg/ha/year)	4.88 – 28.98 (18.97 ± 4.17)	—	—	—	**0.550**	—	—	**0.494**	**0.844**			
NH_4^+ (kg/ha/year)	1.42 – 4.32 (2.87 ± 0.65)	–0.268	—	**–0.749**	—	**–0.607**	—	—	—	—		
Tree BA (square foot per acre)	22.2 – 579.5 (87.1 ± 50.7)	—	—	—	—	—	—	—	—	—	—	
Percent BA softwoods	0 – 100 percent (11.5 ± 22.7)	—	—	—	—	—	—	—	—	—	—	—
Best overall full regression model		**r^2 = 0.257, p < 0.0005: lichen S = 32.644 –6.949 NH_4^+ – 0.702 long. (+ 0.432 NO_3^-)** Lichen S larger with less ammonia pollution, more west, (more acidic air pollution).										

— = cells with dashes indicate correlations weaker than the cutoff; max. = maximum; min. = minimum; sd = standard deviation; lat. = latitude; long. = longitude; elev. = elevation; precip. = precipitation; temp. = temperature; NO_3^- = nitrate; SO_4^{--} = sulfate; NH_4^+ = ammonium; BA = basal area; S = south; N = north; W = west; E = east; m = meter; mm = millimeter; °C = degree Celsius; kg = kilogram; ha = hectare. Range and intercorrelations of Lichen S plus geography, climate, pollution, and forest structure variables; the best regression model is given at bottom. Positive correlations are in black; negative are in red. Single correlations with *rho* > 0.447 (in bold, all with p < 0.0005) account for at least 20 percent of variation; Correlations weaker than the cutoff are included for Lichen S (p < 0.0009) because they were the two highest correlations found. Correlations between environmental variables with *rho* ≥ 0.775 (at least 60 percent of variation accounted for) indicate those two variables should not be entered into the same regression model. The best overall regression model is included with contributing variables listed in descending order of significance; minor variables (p ≤ 0.05 but usually p ≥ 0.005) are listed in parentheses.

[a] Removed E plots with total live-tree BA <21.8966 square feet per acre (< 5 m^2/ha).

Table A3.5—Southeastern ecoregion group in the East region

Variable	Range (average ± sd)	Spearman correlations (*rho*) (Bailey's province codes = 231, 232; N = 239[a]; years = 1994–99)										
		Lichen S	Lat.	Long.	Elev.	Precip.	Max. July temp.	Min. Jan. temp.	NO_3^-	SO_4^{--}	NH_4^+	Tree BA
Lichen S	0 – 33 (average 11.4 ± 5.5 sd)											
Latitude (decimal degrees, S to N)	30.64 – 40.26	—										
Longitude (decimal degrees, W to E)	-88.34 – -74.03	–0.246	**0.834**									
Elevation (m)	0 – 476 (112 ± 104)	0.274	—	**–0.570**								
Precipitation (mm)	1038 – 1670 (1259 ± 152)	—	**–0.738**	**–0.857**	—							
Max. July temp. (°C)	28.5 – 33.9 (31.9 ± 1.14)	—	**–0.894**	**–0.687**	—	**0.543**						
Min. Jan. temp. (°C)	5.0 – -6.3 (-1.5 ± 2.73)	—	**–0.950**	**–0.691**	—	**0.649**	**0.910**					
NO_3^- (kg/ha/year)	7.60 – 15.03 (10.55 ± 1.62)	—	**0.927**	**0.678**	—	**–0.605**	**–0.888**	**–0.939**				
SO_4^{--} (kg/ha/year)	9.63 – 20.00 (15.29 ± 2.33)	—	**0.909**	**0.648**	—	**–0.595**	**–0.896**	**–0.935**	**0.978**			
NH_4^+ (kg/ha/year)	1.35 – 3.60 (2.40 ± 0.41)	—	**0.745**	**0.475**	—	—	**–0.702**	**–0.730**	**0.717**	**0.732**		
Tree BA (square foot per acre)	22.6 – 309.2 (91.4 ± 45.6)	—	—	—	—	—	—	—	—	—	—	
Percent BA softwoods	0 – 100 percent (40.8 ± 37.36)	—	—	—	—	—	—	—	—	—	—	—
Best overall full regression model		**r^2 = 0.134, p < 0.0005: Lichen S = 15.571 (+ 0.018 elev. – 2.593 NH_4^+)** Lichen S larger (higher, with less ammonia air pollution).										

— = cells with dashes indicate correlations weaker than the cutoff; max. = maximum; min. = minimum; sd = standard deviation; lat. = latitude; long. = longitude; elev. = elevation; precip. = precipitation; temp. = temperature; NO_3^- = nitrate; SO_4^{--} = sulfate; NH_4^+ = ammonium; BA = basal area; S = south; N = north; W = west; E = east; m = meter; mm = millimeter; °C = degree Celsius; kg = kilogram; ha = hectare. Range and intercorrelations of Lichen S plus geography, climate, pollution, and forest structure variables; the best regression model is given at bottom. Positive correlations are in black; negative are in red. Single correlations with *rho* > 0.447 (in bold, all with p < 0.0005) account for at least 20 percent of variation; Correlations weaker than the cutoff are included for Lichen S (p < 0.0009) because they were the two highest correlations found. Correlations between environmental variables with *rho* ≥ 0.775 (at least 60 percent of variation accounted for) indicate those two variables should not be entered into the same regression model. The best overall regression model is included with contributing variables listed in descending order of significance; minor variables (p ≤ 0.05 but usually p ≥ 0.005) are listed in parentheses.

[a] Removed E plots with total live-tree BA <21.8966 square feet per acre (< 5 m^2/ha).

Table A3.6—Cascades/Coast Mountains ecoregion group in West region

Variable	Range (average ± sd)	Spearman correlations (*rho*) (Bailey's province codes = 242, M242; N = 213; years = 1998–2001)										
		Lichen S	Lat.	Long.	Elev.	Precip.	Max. July temp.	Min. Jan. temp.	NO_3^-	SO_4^{--}	NH_4^+	Tree BA
Lichen S	0 – 43 (average 17.2 ± 8.3 sd)											
Latitude (decimal degrees, S to N)	42.12 – 48.96	—										
Longitude (decimal degrees, W to E)	–124.41 – –119.87	—	—									
Elevation (m)	0 – 2110 (853 ± 581)	—	—	**0.754**								
Precipitation (mm)	294 – 4731 (1733 ± 831)	—	—	**–0.506**	—							
Max. July temp. (°C)	16.5 – 28.6 (23.5 ± 2.52)	—	**–0.538**	—	—	—						
Min. Jan. temp. (°C)	–12.3 – 4.8 (–2.54 ± 3.57)	—	—	**–0.892**	**–0.849**	**0.482**	—					
NO_3^- (kg/ha/year)	0.74 – 5.27 (2.79 ± 1.05)	—	**0.832**	—	—	—	—	—				
SO_4^{--} (kg/ha/year)	0.32 – 7.53 (3.54 ± 1.53)	—	**0.678**	**–0.702**	—	**0.558**	—	**0.501**	**0.706**			
NH_4^+ (kg/ha/year)	0.00 – 1.32 (0.33 ± 0.25)	0.191	—	**–0.514**	**–0.751**	—	—	**0.609**	—	**0.531**		
Tree BA (square foot per acre)	1.8 – 650.2 (164.3 ± 112.8)	0.313	—	—	—	—	—	—	—	—	—	
Percent BA softwoods	0 – 100 percent (88.8 ± 24.3)	—	—	**0.498**	**0.581**	—	—	**–0.567**	—	—	—	—
Best overall full regression model		**r^2 = 0.113, p < 0.0005: Lichen S = 11.945 + 0.023 tree BA (+ –4.461 NH_4^+)** Lichen S larger with more/bigger trees, (with more ammonia air pollution).										

— = cells with dashes indicate correlations weaker than the cutoff; max. = maximum; min. = minimum; sd = standard deviation; lat. = latitude; long. = longitude; elev. = elevation; precip. = precipitation; temp. = temperature; NO_3^- = nitrate; SO_4^{--} = sulfate; NH_4^+ = ammonium; BA = basal area; S = south; N = north; W = west; E = east; m = meter; mm = millimeter; °C = degree Celsius; kg = kilogram; ha = hectare. Range and intercorrelations of Lichen S plus geography, climate, pollution, and forest structure variables; the best regression model is given at bottom. Positive correlations are in black; negative are in red. Single correlations with *rho* > 0.447 (in bold, all with p < 0.0005) account for at least 20 percent of variation; correlations weaker than the cutoff are included for Lichen S (p < 0.006) because they were the two highest correlations found. Correlations between environmental variables with *rho* ≥ 0.775 (at least 60 percent of variation accounted for) indicate those two variables should not be entered into the same regression model. The best overall regression model is included with contributing variables listed in descending order of significance; minor variables (p ≤ 0.05 but usually p ≥ 0.005) are listed in parentheses.

Table A3.7—Sierras/Coast Mountains ecoregion group in the West region

Variable	Range (average ± sd)	Spearman correlations (*rho*) (Bailey's province codes = 261, M261, 262, M262, 263; N = 254; years = 1998–2003)										
		Lichen S	Lat.	Long.	Elev.	Precip.	Max. July temp.	Min. Jan. temp.	NO_3^-	SO_4^{--}	NH_4^+	Tree BA
Lichen S	0 – 45 (average 13.9 ± 8.5 sd)											
Latitude (decimal degrees, S to N)	33.71 – 43.68	—										
Longitude (decimal degrees, W to E)	–124.10 – –116.66	**–0.469**	**–0.659**									
Elevation (m)	37 – 3118 (1240 ± 698)	**–0.474**	—	**0.550**								
Precipitation (mm)	224 – 4052 (1048 ± 587)	—	—	**–0.554**	—							
Max. July temp. (°C)	18.9 – 37.3 (28.9 ± 3.86)	—	—	—	**–0.646**	—						
Min. Jan. temp. (°C)	–12.3 – 4.9 (–2.24 ± 3.78)	—	—	—	**–0.821**	—	**0.559**					
NO_3^- (kg/ha/year)	0.69 – 5.08 (2.21 ± 0.96)	—	**–0.880**	**0.612**	—	—	—	—				
SO_4^{--} (kg/ha/year)	0.18 – 3.33 (1.34 ± 0.57)	—	—	—	**–0.516**	—	—	**0.675**	**0.604**			
NH_4^+ (kg/ha/year)	0.11 – 1.85 (0.70 ± 0.42)	—	**–0.889**	**0.597**	—	—	—	—	**0.948**	**0.540**		
Tree BA (square foot per acre)	1.9 – 568.4 (122.3 ± 94.2)	—	—	—	—	—	—	—	—	—	—	
Percent BA softwoods	0 – 100 percent (68.2 ± 38.9)	—	—	—	**0.694**	—	**–0.544**	**–0.713**	—	**–0.504**	**0.694**	—
Best overall full regression model		**r^2 = 0.293, p < 0.0005: lichen S = 24.938 – 0.067 elev. –5.445 NH_4^+** Lichen S larger lower, with less ammonia air pollution.										

— = cells with dashes indicate correlations weaker than the cutoff; max. = maximum; min. = minimum; sd = standard deviation; lat. = latitude; long. = longitude; elev. = elevation; precip. = precipitation; temp. = temperature; NO_3^- = nitrate; SO_4^{--} = sulfate; NH_4^+ = ammonium; BA = basal area; S = south; N = north; W = west; E = east; m = meter; mm = millimeter; °C = degree Celsius; kg = kilogram; ha = hectare. Range and intercorrelations of Lichen S plus geography, climate, pollution, and forest structure variables; the best regression model is given at bottom. Positive correlations are in black; negative are in red. Single correlations with *rho* > 0.447 (in bold, all with p < 0.0005) account for at least 20 percent of variation; correlations with Lichen S stronger than this cutoff suggest potential for an indicator. Correlations between environmental variables with *rho* ≥ 0.775 (at least 60 percent of variation accounted for) indicate those two variables should not be entered into the same regression model. The best overall regression model is included with contributing variables listed in descending order of significance; minor variables (p ≤ 0.05 but usually p ≥ 0.005) are listed in parentheses.

Table A3.8—Northern Rockies ecoregion group in West region

Variable	Range (average ± sd)	Spearman correlations (*rho*) (Bailey's province codes = M332, M333; N = 235; years = 1996–2005)										
		Lichen S	Lat.	Long.	Elev.	Precip.	Max. July temp.	Min. Jan. temp.	NO_3^-	SO_4^{--}	NH_4^+	Tree BA
Lichen S	2 – 34 (average 14.7 ± 7.4 sd)											
Latitude (decimal degrees, S to N)	43.42 – 48.96	**0.531**										
Longitude (decimal degrees, W to E)	–120.84 – –109.48	**–0.533**	—									
Elevation (m)	497 – 2953 (1525 ± 527)	**–0.657**	**–0.652**	**0.545**								
Precipitation (mm)	316 – 2153 (850 ± 311)	—	—	—	—							
Max. July temp. (°C)	17.8 – 30.5 (24.7 ± 2.47)	—	—	—	**–0.649**	**–0.515**						
Min. Jan. temp. (°C)	–15.4 – –2.8 (–9.3 ± 2.68)	**0.450**	—	**–0.620**	**–0.636**	—	—					
NO_3^- (kg/ha/year)	0.38 – 3.49 (2.08 ± 0.62)	**0.487**	**0.773**	—	**–0.738**	—	—	—				
SO_4^{--} (kg/ha/year)	0.21 – 3.25 (1.32 ± 0.69)	**0.568**	**0.687**	**–0.499**	**–0.846**	—	**0.568**	**0.508**	**0.904**			
NH_4^+ (kg/ha/year)	0.00 – 1.09 (0.37 ± 0.23)	—	—	—	—	—	**0.464**	—	**0.601**	**0.503**		
Tree BA (square foot per acre)	4.0 – 340.2 (98.4 ± 64.3)	—	—	—	—	—	—	—	—	—	—	
Percent BA softwoods	0 – 100 percent (97.2 ± 12.5)	—	—	—	—	—	—	—	—	—	—	—
Best overall full regression model		**r^2 = 0.463, p < 0.0005: lichen S = –167.956 –1.124 long. –0.004 elev. + 1.274 lat.** Lichen S larger more west, lower, more north.										

— = cells with dashes indicate correlations weaker than the cutoff; max. = maximum; min. = minimum; sd = standard deviation; lat. = latitude; long. = longitude; elev. = elevation; precip. = precipitation; temp. = temperature; NO_3^- = nitrate; SO_4^{--} = sulfate; NH_4^+ = ammonium; BA = basal area; S = south; N = north; W = west; E = east; m = meter; mm = millimeter; °C = degree Celsius; kg = kilogram; ha = hectare. Range and intercorrelations of Lichen S plus geography, climate, pollution, and forest structure variables; the best regression model is given at bottom. Positive correlations are in black; negative are in red. Single correlations with *rho* > 0.447 (in bold, all with p < 0.0005) account for at least 20 percent of variation; correlations with Lichen S stronger than this cutoff suggest potential for an indicator. Correlations between environmental variables with *rho* ≥ 0.775 (at least 60 percent of variation accounted for) indicate those two variables should not be entered into the same regression model. The best overall regression model is included with contributing variables listed in descending order of significance; minor variables (p ≤ 0.05 but usually p ≥ 0.005) are listed in parentheses.

Table A3.9—Southern Rockies/Great Basin Mountains ecoregion group in the West region

		Spearman correlations (*rho*) (Bailey's province codes = M331, M341; N = 324; years = 1994–2005)										
Variable	Range (average ± sd)	Lichen S	Lat.	Long.	Elev.	Precip.	Max. July temp.	Min. Jan. temp.	NO_3^-	SO_4^{--}	NH_4^+	Tree BA
Lichen S	0 – 31 (average 6.8 ± 3.7 sd)											
Latitude (decimal degrees, S to N)	37.01 – 45.42	—										
Longitude (decimal degrees, W to E)	–116.86 – –104.44	**0.473**	—									
Elevation (m)	1585 – 3740 (2552 ± 426)	—	—	—								
Precipitation (mm)	230 – 1495 (616 ± 239)	—	—	—	**0.478**							
Max. July temp. (°C)	15.7 – 33.2 (24.1 ± 3.58)	—	—	—	**–0.800**	**–0.681**						
Min. Jan. temp. (°C)	–15.4 – –2.8 (–9.29 ± 2.68)	—	—	—	**–0.546**	**–0.488**	**0.611**					
NO_3^- (kg/ha/year)	1.87 – 5.90 (3.56 ± 0.90)	—	**–0.467**	**0.746**	**0.473**	—	—	—				
SO_4^{--} (kg/ha/year)	1.14 – 3.04 (2.00 ± 0.47)	0.414	—	**0.805**	**0.448**	—	—	—	**0.949**			
NH_4^+ (kg/ha/year)	0.05 – 2.14 (0.77 ± 0.32)	—	—	—	—	—	—	—	—	—		
Tree BA (square foot per acre)	1.5 – 361.5 (94.0 ± 63.4)	—	—	—	—	—	—	—	—	—	—	
Percent BA softwoods	0 – 100 percent (84.8 ± 31.1)	—	—	—	—	—	—	—	—	—	—	—
Best overall full regression model		**r^2 = 0.223, p < 0.0005: lichen S = 52.959 + 0.443 long. (+ 0.614 NO_3^-)** Lichen S larger more east (with more acidic air pollution).										

— = cells with dashes indicate correlations weaker than the cutoff; max. = maximum; min. = minimum; sd = standard deviation; lat. = latitude; long. = longitude; elev. = elevation; precip. = precipitation; temp. = temperature; NO_3^- = nitrate; SO_4^{--} = sulfate; NH_4^+ = ammonium; BA = basal area; S = south; N = north; W = west; E = east; m = meter; mm = millimeter; °C = degree Celsius; kg = kilogram; ha = hectare. Range and intercorrelations of Lichen S plus geography, climate, pollution, and forest structure variables; the best regression model is given at bottom. Positive correlations are in black; negative are in red. Single correlations with *rho* > 0.447 (in bold, all with p < 0.0005) account for at least 20 percent of variation; one correlation weaker than the cutoff is included for lichen S (p < 0.0009) because it is the second highest correlation found. Correlations between environmental variables with *rho* ≥ 0.775 (at least 60 percent of variation accounted for) indicate those two variables should not be entered into the same regression model. The best overall regression model is included with contributing variables listed in descending order of significance; minor variables (p ≤ 0.05 but usually p ≥ 0.005) are listed in parentheses.

Table A3.10—Great Basin Lowlands ecoregion group in the West region

Variable	Range (average ± sd)	Spearman correlations (*rho*) (Bailey's province codes = 341, 342; N = 143; years = 1994–2005)										
		Lichen S	Lat.	Long.	Elev.	Precip.	Max. July temp.	Min. Jan. temp.	NO_3^-	SO_4^{--}	NH_4^+	Tree BA
Lichen S	0 – 27 (average 5.5 ± 4.5 sd)											
Latitude (decimal degrees, S to N)	36.96 – 48.16	**0.541**										
Longitude (decimal degrees, W to E)	–121.40 – –106.32	—	—									
Elevation (m)	528 – 2823 (1867 ± 462)	–0.405	**–0.628**	**0.462**								
Precipitation (mm)	171 – 961 (391 ± 139)	—	—	—	—							
Max. July temp. (°C)	19.6 – 34.3 (28.5 ± 2.54)	—	—	—	—	**–0.576**						
Min. Jan. temp. (°C)	–16.2 – –3.5 (–8.88 ± 2.48)	—	—	**–0.733**	—	—	—					
NO_3^- (kg/ha/year)	1.04 – 4.16 (1.57 ± 0.55)	—	**–0.499**	**0.723**	**0.519**	—	—	**–0.453**				
SO_4^{--} (kg/ha/year)	0.21 – 3.31 (1.57 ± 0.55)	—	—	**0.541**	—	—	—	—	**0.757**			
NH_4^+ (kg/ha/year)	0.02 – 1.32 (0.64 ± 0.28)	—	—	**0.524**	—	—	—	—	**0.597**	—		
Tree BA (square foot per acre)	1.8 – 308.6 (70.6 ± 58.7)	—	—	—	—	—	—	—	—	—	—	
Percent BA softwoods	0 – 100 percent (88.5 ± 28.3)	—	—	—	—	—	—	—	—	—	—	—
Best overall full regression model		**$r^2 = 0.468$, $p < 0.0005$: Lichen S = –38.103 + 1.049 lat. –5.484 NH_4^+ + 1.783 NO_3^-** Lichen S larger more north, with less ammonia air pollution, with more acidic air pollution.										

— = cells with dashes indicate correlations weaker than the cutoff; max. = maximum; min. = minimum; sd = standard deviation; lat. = latitude; long. = longitude; elev. = elevation; precip. = precipitation; temp. = temperature; NO_3^- = nitrate; SO_4^{--} = sulfate; NH_4^+ = ammonium; BA = basal area; S = south; N = north; W = west; E = east; m = meter; mm = millimeter; °C = degree Celsius; kg = kilogram; ha = hectare. Range and intercorrelations of Lichen S plus geography, climate, pollution, and forest structure variables; the best regression model is given at bottom. Positive correlations are in black; negative are in red. Single correlations with *rho* > 0.447 (in bold, all with $p < 0.0005$) account for at least 20 percent of variation; one correlation weaker than the cutoff is included for Lichen S ($p < 0.0009$) because it is the second highest correlation found. Correlations between environmental variables with *rho* ≥ 0.775 (at least 60 percent of variation accounted for) indicate those two variables should not be entered into the same regression model. The best overall regression model is included with contributing variables listed in descending order of significance; minor variables ($p \leq 0.05$ but usually $p \geq 0.005$) are listed in parentheses.

Table A3.11—Colorado Plateau/Southern Dry Mountains ecoregion group in the West region

Variable	Range (average ± sd)	Spearman correlations (*rho*) (Bailey's province codes = 313, M313; N = 151; years = 1994–2005) Lichen S	Lat.	Long.	Elev.	Precip.	Max. July temp.	Min. Jan. temp.	NO_3^-	SO_4^{--}	NH_4^+	Tree BA
Lichen S	0 – 14 (average 6.1 ± 2.7 sd)											
Latitude (decimal degrees, S to N)	33.08 – 37.99	—										
Longitude (decimal degrees, W to E)	–113.81 – –106.95	—	—									
Elevation (m)	1088 – 2800 (1954 ± 356)	**0.460**	—	—								
Precipitation (mm)	183 – 826 (451 ± 152)	**0.545**	**–0.532**	—	—							
Max. July temp. (°C)	23.5 – 38.3 (30.3 ± 3.04)	**–0.495**	—	—	**–0.927**	**0.578**						
Min. Jan. temp. (°C)	–14.3 – 1.5 (–7.54 ± 2.93)	—	—	**–0.540**	**–0.750**	—	**0.650**					
NO_3^- (kg/ha/year)	2.41 – 4.40 (3.22 ± 0.36)	**0.513**	—	**0.492**	**0.861**	—	**–0.808**	**–0.712**				
SO_4^{--} (kg/ha/year)	1.39 – 2.87 (2.03 ± 0.29)	**0.489**	—	**0.697**	**0.629**	—	**–0.569**	**–0.564**	**0.849**			
NH_4^+ (kg/ha/year)	0.29 – 0.86 (0.63 ± 0.10)	—	—	—	—	—	—	—	—	—		
Tree BA (square foot per acre)	2.3 – 255.9 (83.0 ± 56.0)	—	—	—	—	—	—	—	—	—	—	
Percent BA softwoods	0 – 100 percent (88.4 ± 25.4)	—	—	—	—	**–0.543**	—	—	—	—	—	—
Best overall full regression model		**r^2 = 0.356, p < 0.0005: lichen S = 42.393 + 0.007 precip. + 0.403 long. + 1.678 NO_3^-** Lichen S larger wetter, more east, with more acidic air pollution.										

— = cells with dashes indicate correlations weaker than the cutoff; max. = maximum; min. = minimum; sd = standard deviation; lat. = latitude; long. = longitude; elev. = elevation; precip. = precipitation; temp. = temperature; NO_3^- = nitrate; SO_4^{--} = sulfate; NH_4^+ = ammonium; BA = basal area; S = south; N = north; W = west; E = east; m = meter; mm = millimeter; °C = degree Celsius; kg = kilogram; ha = hectare. Range and intercorrelations of Lichen S plus geography, climate, pollution, and forest structure variables; the best regression model is given at bottom. Positive correlations are in black; negative are in red. Single correlations with *rho* > 0.447 (in bold, all with p < 0.0005) account for at least 20 percent of variation; correlations with Lichen S stronger than this cutoff suggest potential for an indicator. Correlations between environmental variables with *rho* ≥ 0.775 (at least 60 percent of variation accounted for) indicate those two variables should not be entered into the same regression model. The best overall regression model is included with contributing variables listed in descending order of significance; minor variables (p ≤ 0.05 but usually p ≥ 0.005) are listed in parentheses.

Table A3.12—Arid Southwest ecoregion group in the West region

Variable	Range (average ± sd)	Spearman correlations (*rho*) (Bailey's province codes = 321, 322; N = 39; years = 1999–2005)										
		Lichen S	Lat.	Long.	Elev.	Precip.	Max. July temp.	Min. Jan. temp.	NO_3^-	SO_4^{--}	NH_4^+	Tree BA
Lichen S	0 – 13 (average 5.0 ± 2.7 sd)											
Latitude (decimal degrees, S to N)	31.42 – 36.70	—										
Longitude (decimal degrees, W to E)	–115.67 – –109.22	—	**–0.571**									
Elevation (m)	844 – 2448 (1544 ± 398)	**0.607**	—	—								
Precipitation (mm)	257 – 669 (445 ± 116)	**0.555**	**–0.467**	—	**0.499**							
Max. July temp. (°C)	27.1 – 37.6 (32.6 ± 2.40)	**–0.591**	—	—	**–0.891**	**–0.630**						
Min. Jan. temp. (°C)	–7.1 – 2.8 (–1.95 ± 2.46)	—	**–0.758**	—	—	—	—					
NO_3^- (kg/ha/year)	2.07 – 3.94 (3.04 ± 0.42)	—	—	—	**0.631**	**0.618**	**–0.643**	—				
SO_4^{--} (kg/ha/year)	1.41 – 2.75 (2.07 ± 0.38)	—	**–0.876**	**0.781**	—	**0.527**	—	—	**0.643**			
NH_4^+ (kg/ha/year)	0.42 – 0.90 (0.69 ± 0.09)	—	—	—	—	—	—	—	**0.449**	—		
Tree BA (square foot per acre)	4.7 – 160.0 (54.1 ± 40.4)	**0.555**	—	—	**0.609**	**0.519**	**–0.603**	—	—	—	—	
Percent BA softwoods	0 – 100 percent (42.8 ± 44.3)	—	**0.720**	**–0.533**	—	—	—	**–0.464**	—	**–0.627**	—	—
Best overall full regression model		**r^2 = 0.593, p < 0.0005: lichen S = 25.626 + 0.020 precip. + 1.364 lat. (+ 0.415 long.)** Lichen S larger wetter, more north, (more east).										

— = cells with dashes indicate correlations weaker than the cutoff; max. = maximum; min. = minimum; sd = standard deviation; lat. = latitude; long. = longitude; elev. = elevation; precip. = precipitation; temp. = temperature; NO_3^- = nitrate; SO_4^{--} = sulfate; NH_4^+ = ammonium; BA = basal area; S = south; N = north; W = west; E = east; m = meter; mm = millimeter; °C = degree Celsius; kg = kilogram; ha = hectare. Range and intercorrelations of Lichen S plus geography, climate, pollution, and forest structure variables; the best regression model is given at bottom. Positive correlations are in black; negative are in red. However correlations reported here with *rho* > 0.447 (in bold) have p < 0.005 because there are so few plots in this group; correlations with Lichen S stronger than this cutoff suggest potential for an indicator. Correlations between environmental variables with *rho* ≥ 0.775 (at least 60 percent of variation accounted for) indicate those two variables should not be entered into the same regression model. The best overall regression model is included with contributing variables listed in descending order of significance; minor variables (p ≤ 0.05 but usually p ≥ 0.005) are listed in parentheses.

Chapter 4. Vegetation Diversity

Beth Schulz

Introduction

Vegetation is the source of primary production and a fundamental determinant of habitat and wildfire fuel profile characterization. As such, changes in vegetation can have cascading effects through an ecosystem. While individual species can be important indicators of a site's potential productivity, economic value, and wildlife forage and shelter, changes in the composition and spatial arrangement of vascular plants in a forest may indicate the presence of chronic stresses such as discrete site degradation, climate change, and pollution. These stresses can lead to decline or local eradication of sensitive species, as well as the increase and dominance of opportunistic species, such as many invasive nonnative plants. Changes in species diversity, composition, and the abundance of nonnative species are national concerns, as reflected in international criteria for assessing sustainability of forestry practices (Forest Service, U.S. Department of Agriculture, 2004). Monitoring for change requires establishing an understanding of current conditions that can be reassessed in the future.

The vegetation diversity and structure indicator from the Forest Inventory and Analysis (FIA) Program of the Forest Service, U.S. Department of Agriculture, provides baseline data on forest vascular plant composition. The inventory is conducted with consistent methods across the vast array of forest types in the Nation. Permanent plots are established on a systematic grid in forests across the country and will be revisited in the future, providing a powerful means for change detection through time. The FIA phase 3 (P3) grid is geographically extensive with one plot every 96,000 acres. Therefore, while individual plot data characterize the changing conditions of sampled forest stands, population estimates are appropriate at regional and national scales.

This report focuses on attributes derived from the vascular plant composition data to describe the conditions of forest plant communities across the United States. Species composition data allow for analysis of diversity and individual species' distribution and abundance at several scales, from measured forest stands to large geographic areas, including assessments of the impacts of nonnative species on native plant communities.

Methods

The FIA program collects vegetation data on the systematically located P3 plots (fig. 4.1). All vascular plants rooted in or hanging over the four subplots are identified (see footnote 2, page 29, chapter 3). Plant identifications are recorded using plant symbols defined by the Natural Resource Conservation Service PLANTS Database (U.S. Department of Agriculture, Natural Resource Conservation Service 2000). For each species on the subplot, total percent canopy cover and canopy cover within three horizontal layers are estimated and recorded. Species rooted in or overhanging each of three permanently positioned 1-m^2 quadrats on each subplot are also recorded. Unknown species are collected near the plot, then pressed, dried, labeled, and identified later by an FIA vegetation specialist or submitted to a qualified herbarium.

The FIA sampling design is focused on accessible forested lands; this results in some plots with less area sampled than the four full subplots. These plots provide 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 are not 100 percent within accessible forest lands.

Ecological provinces, as described by Bailey (1995) and Cleland and others (2007), define areas with similar physical and biological features that influence vegetation and are useful for defining populations. Forest cover type is often highly correlated with understory species composition and is used as the primary grouping class within defined geographic areas. By grouping plots by ecological province and forest type, population estimate variability may be reduced. Forest type is assigned by mensuration field crews, and also derived by algorithm based on live-tree stocking of softwoods and

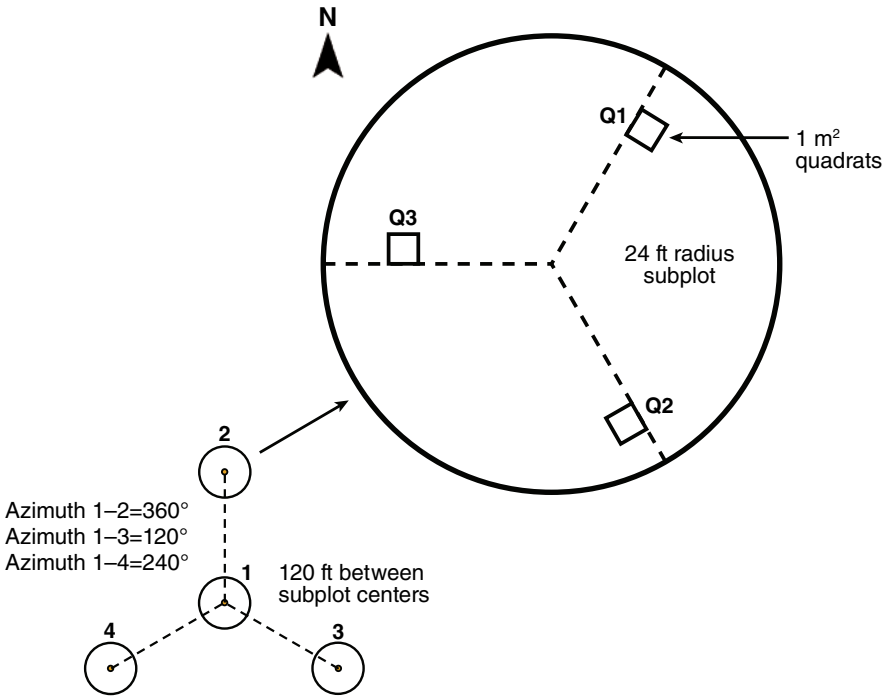


Figure 4.1—Vegetation indicator plot layout.

hardwoods (Forest Service, U.S. Department of Agriculture, 2006). In this report, a forest type within an ecological province will be referred to as a community type when summarizing and comparing population estimates.

The dataset includes 915 plots on the P3 grid collected from 2001 through 2004 from the States of Oregon, Washington, Utah, South Dakota, North Dakota, Kansas, Nebraska, Missouri, Minnesota, Wisconsin, Michigan, Illinois, Indiana, Ohio, and Pennsylvania; other States either do not collect vegetation data or the data are not available. Of those plots, 862 have at least one known 100-percent forested subplot, 583 plots are fully forested (four subplots), and 504 plots are fully forested in a single condition class. Condition class is determined by a number of attributes assigned to the plot such as forest type, stand-size class, and ownership. The type of results that can be presented vary by data subset (table 4.1). On the 504 fully forested, single-condition plots, there are 127 community types. Twelve of these community types are represented by 10 or more plots and are used in this report to demonstrate basic population estimates and comparisons. The 12 types are concentrated in areas with the longest history of vegetation indicator data collection.

Identification level rates and plant origin are determined by comparing the final vegetation species codes, after all identifications of unknown plants are complete, with the PLANTS Database. Species' nativities (native or introduced) are assigned based on information from the PLANTS Database. Note that not all introduced plants are invasive, and some native plants are invasive and/or noxious.

Species richness (alpha α) is a count of the number of unique plants recorded in each fully sampled (100-percent accessible forest land) plot. Gamma (γ) represents the number of unique species recorded across all samples in the population and represents the lower bound of the true number of species in a population. Any inventory that does not sample the entire population is expected to miss species (Colwell and others 2004, Heltshe and Forrester 1983). Given the expansive geographic range and low sample density of the P3 grid, it is assumed that many species are missed. To estimate the true number of species to be expected in the sampled community, nonparametric jackknife estimates are calculated (Heltshe and Forrester 1983, McCune and Grace 2002). Nonparametric estimates avoid assumptions about species distributions and discovery rates; most natural communities demonstrate patchiness and clumping of species. Plot-based samples are not random samples of individuals but random samples of space. These estimates are based on the number of species that are recorded on one or two plots across the sample. Jackknife estimates are calculated using the formulas:

Table 4.1—Summary of datasets and type of results presented

Number of plots (n)	Dataset characteristics	Results presented
915	Matching FIA P2 data available (including approximate plot locations)	Table 4.2—Identification rates and origin of species for all 915 plots Table 4.5—Distribution of selected native and introduced species Table 4.6—Top 25 most commonly recorded introduced species in all 915 plots Figure 4.5—Percentage of the number of and relative proportion cover of introduced plant species Figure 4.6—Distribution of three invasive grasses Figure 4.7—Distribution of five introduced herbs Figure 4.8—Distribution of three introduced shrubs
862	Plots with at least one known fully forested subplot	Figure 4.3—Average species richness for subplots in 100-percent accessible forested conditions
230	Fully forested, single-condition plots belonging to an ecological province/ forest-type pair represented by at least 10 plots	Table 4.3—Ecological province, forest type, identification rates, and origin of identified species for 12 community types Table 4.4—Species richness and differentiation for 12 community types Figure 4.2—Species area and distance curves for 12 community types Figure 4.4—Average plot, subplot, and quadrat alphas for 12 community types Appendix table A4.1—Top 5 percent most common species or genera for 12 community types

FIA = Forest Inventory and Analysis; P2 = phase 2.

1st order:

$$Jack_1 = S + \frac{r'(n-1)}{n}$$

where

S = the number of observed species

r' = the number of species occurring in only one sample unit

n = the number of sample units (plots)

2nd order:

$$Jack_2 = S + \frac{r'(2n-3)}{n} - \frac{r''(n-2)^2}{n(n-1)}$$

where

r″ = the number of species occurring in exactly two sample units

To quantify the sample's ability to characterize the species composition, species area and distance curves are used. Species area curves assessing sample adequacy are based on repeated subsampling of a fixed sample using species presence/absence (P/A). Distance curves describe the Sørensen distance (dissimilarity) between the species composition of a subsample of plots and the entire sample of plots, as a function of sample size (McCune and Grace 2002). Distance curves incorporate abundance information for each species. This is useful to determine a sample size (number of plots) that provides a consistent species composition as opposed to a consistent species list. These are produced using PC–ORD™ software (McCune and Medford 1999). Given the extent of the P3 grid and the concept of community type (forest type within an ecological province), it is expected that community types with high species richness require more samples to characterize their species composition.

Differential diversity, in its most basic sense, describes the extent to which species composition and abundance change from one place to another (Scheiner 2003, Whittaker and others 2001). It can be calculated a number of ways; in this report, it is simply:

$$\beta = \frac{\gamma}{\overline{\alpha}}$$

For community type beta, gamma is the total number of species for the community type and average alpha is the average number of species per plot. For plot-level beta, gamma is the total number of species observed on the plot and average alpha is the average number of species per subplot. Distribution of individual species are assessed using P/A data, which describes how the observed species are dispersed over an area of interest. This is useful in large-scale projects where vegetation composition changes dramatically across the area of interest (McCune and Grace 2002). The P/A data can be expressed as frequency and constancy; both are ratios of the number of sample or subsample units where a species was recorded to the number of total sample or subsample units visited. In this report, constancy is the percentage of the total number

of plots where the species has been observed, without the restriction of equal forested area plots. Frequency is used at subsample levels and is restricted to sample units of equal area, such as fully forested subplots or the standard 1-m^2 quadrats. Subplot and quadrat frequencies are valuable to describe fine-scale species distribution.

Quadrat frequency is compiled as:

$$F_s = \frac{\sum_{i}^{n}\sum_{j}^{4}\sum_{q}^{3}\delta_{ijq}\mathrm{I}_{ijqs}}{\sum_{i}^{n}\sum_{j}^{4}\sum_{q}^{3}\delta_{ijq}}$$

where

F_s = the frequency of species s on a quadrat basis for the population of interest

δ_{ijq} = a zero-one domain indicator function, which is 1 if quadrat q of subplot j of plot i is in accessible forest, 0 otherwise

I_{ijqs} = a zero-one species indicator function, which is 1 if species s is present on quadrat q of subplot j of plot i, 0 otherwise

Subplot frequency is computed as:

$$F_s' = \frac{\sum_{i}^{n}\sum_{j}^{4}\delta_{ij}I_{ijs}}{\sum_{i}^{n}\sum_{j}^{4}\delta_{ij}}$$

where

F_s' = the subplot frequency of species s for the population

δ_{ij} = a zero-one domain indicator function, which is 1 if subplot j of plot i is in 100-percent accessible forest, 0 otherwise

I_{ijs} = a zero-one species indicator function, which is 1 if species s is present on subplot j of plot i, 0 otherwise

The percentage number and relative proportion of cover of introduced species are calculated at the plot level; results are presented as mapped products. Percentage number of introduced species per plot is calculated by dividing the number of introduced species by the total number of unique species recorded on the plot (including plants not identified to species). Relative proportion of cover of introduced species is calculated by dividing the sum of all introduced plant species by the sum of all plant covers.

$$Rc_i = \frac{\sum_{j}^{4}\sum_{s'}c_{ijs'}}{\sum_{j}^{4}\sum_{s}c_{ijs}}$$

where

Rc_i = relative cover of all introduced species on plot i

$c_{ijs'}$ = canopy cover of introduced species s' on subplot j of plot i

c_{ijs} = canopy cover of species s on subplot j of plot i

s' = index for introduced species, a subset of s

s = index for any species

j = index for subplots

Results

Species Composition—Basic summaries of identification rates and species origin for the 915 plots provide insights to the nature of vascular plant composition in the forests of the 15 States that were sampled (table 4.2). Of all individual plant observations, 82 percent were identified to species. Of those identified to species, 93 percent are native. Of the unique plant codes recorded (γ), 89 percent of those identified to species are native. The origin of plants identified to species for each of the 12 community types are shown in table 4.3. In general, conifer community types have a higher percentage of native plants. However, the number of plants not identified to species (genus codes plus unknown codes) exceeds the number of identified introduced species for all groups in tables 4.2 and 4.3, and so the true extent of the number of introduced species remains uncertain.

The numbers of unknown codes underestimate the true number of species represented by plants recorded as such; there are numerous potential species for each genus and each symbol for unknown plants listed. The origin of plants identified to genus were estimated by evaluating all species of each recorded genus to determine what proportion of all species within the genus is native or introduced, with the assumption that species identified only to the genus had the same

Table 4.2—Identification rates and origin of species for all 915 plots

		Level of identification			Species by origin	
Type of record	n	Unknowns	Genus	Species	Native	Introduced
Individual plant observations (total)	35,980	1,624	4,830	29,491	27,418	2,108
Unique plant codes recorded (gamma)	2,975	33	345	2,595	2,318	279

Table 4.3—Ecological province, forest type, identification rates, and origin of identified species for 12 community types

Community type					Plants identified to species		
Ecological province (code)[a] and forest type (abbreviation)[b]	ID[c]	Plots (n)	Gamma[d]	Species	Native	Introduced	Unknowns[e]
			number of unique species codes				
Laurentian Mixed Forest (212)							
Northern white-cedar (Cedar)	A	13	270	231	221	10	39
Sugar maple/beech/birch (SMaple)	B	28	335	281	256	25	54
Hard maple/basswood (HMaple)	C	14	271	216	203	13	55
Aspen (Aspen)	D	33	438	364	325	39	74
Eastern Broadleaf Forest (221)							
White oak/red oak/hickory (Oak-Hic)	E	10	250	201	178	23	49
Sugar maple/beech/birch (SMaple)	F	11	201	149	135	14	52
Midwest Broadleaf Forest (222)							
White oak/red oak/hickory (Oak-Hic)	G	10	218	181	161	20	37
Central Interior Broadleaf Forest (223)							
White oak/red oak/hickory (Oak-Hic)	H	42	470	376	352	24	94
White oak (WOak)	I	10	262	210	199	11	52
Colorado Plateau Semi-Desert (313)							
Pinyon/juniper woodland (PJ)	J	12	98	81	76	5	17
Interior Semi-Desert and Desert (341)							
Pinyon/juniper woodland (PJ)	K	34	248	203	194	9	45
Nevada-Utah Mountain Semi-Desert Coniferous Forest (M341)							
Pinyon/juniper woodland (PJ)	L	13	100	79	78	1	21

[a] Used in figure 4.4 to simplify labels, as described by Cleland and others (2007).
[b] Used in figure 4.4 to simplify labels.
[c] Used in figure 4.2 to simplify labels.
[d] Gamma represents the total number of unique plant codes recorded in a community, including codes for plants identified to species, genus, and symbols for plants that cannot be identified to genus.
[e] Unknowns include both genus codes and symbols for unknown plants.

origin rate. For the 4,830 plant observations recorded as genera, origin is estimated to be 88 percent native and 12 percent introduced. Of the 12 communities, the highest percentage of introduced species, based on all plants identified to species, is also 12 percent (Eastern Broadleaf Forest—White oak/red oak/hickory).

Distance curves describing the Sørensen distance (dissimilarity) between subsample species composition and the entire sample species composition for each community type indicate how well the sample captured the overall species composition of the community types (fig. 4.2). Converging error curves representing plus or minus one standard deviation on graphs of community types identified in table 4.3 with more sampled plots indicate better representation of species composition. However, note that several types with smaller sample sizes are also fairly well represented (A. northern white cedar in the Laurentian Mixed Forest ecoprovince, and L. pinyon/juniper woodland in the Nevada-Utah Mountain Semi-Desert Coniferous Forest). There is no evidence that we have oversampled any community type; species area curves do not level off in any case. This is not surprising, given the extent of sampling design.

The approximate top 5 percent most frequently recorded species or genera codes recorded in each of the 12 community types are listed in appendix table A4.1. Constancy of 100 percent indicates the species was recorded on all plots representing that community. The abundance of most common species, listed in appendix table A4.1 as average cover, reveal an important characteristic in plant community composition; some species are extremely common (high constancy) but are present in low abundance (cover). Note that Canada mayflower (*Maianthemum canadense* Desf.) occurs often in all four forest types in the Laurentian Mixed Forest ecological province but with an average cover of < 1 percent. In the larger dataset of 915 plots, 56 percent of all species abundance records are equal to 1 percent canopy cover. Only three community types include an introduced species among the top 5 percent most common species.

Species Richness—Geographic patterns of species richness are shown in figure 4.3 as average subplot alpha ($\overline{\alpha}$). Mapped results include the 862 plots with at least 1 known fully forested subplot. The mesic forests of the Great Lakes States have higher subplot species richness, in general, than the arid semi-desert woodlands.

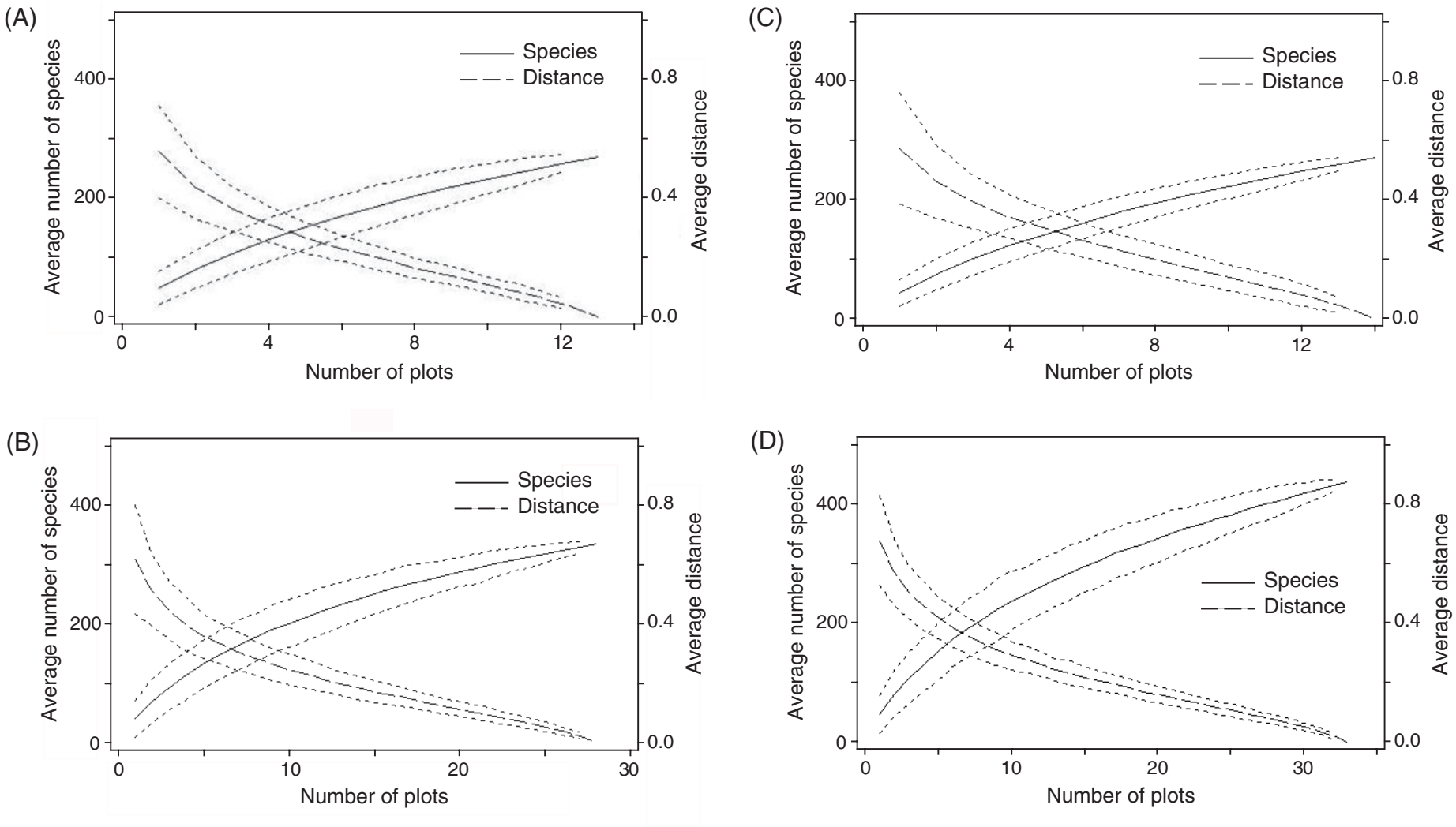


Figure 4.2—Species area and distance curves for 12 community types. Community types are identified by letters in table 4.3. Error curves represent plus and minus one standard deviation. (Data source: U.S. Forest Service Forest Inventory and Analysis Program) *(continued on next page)*

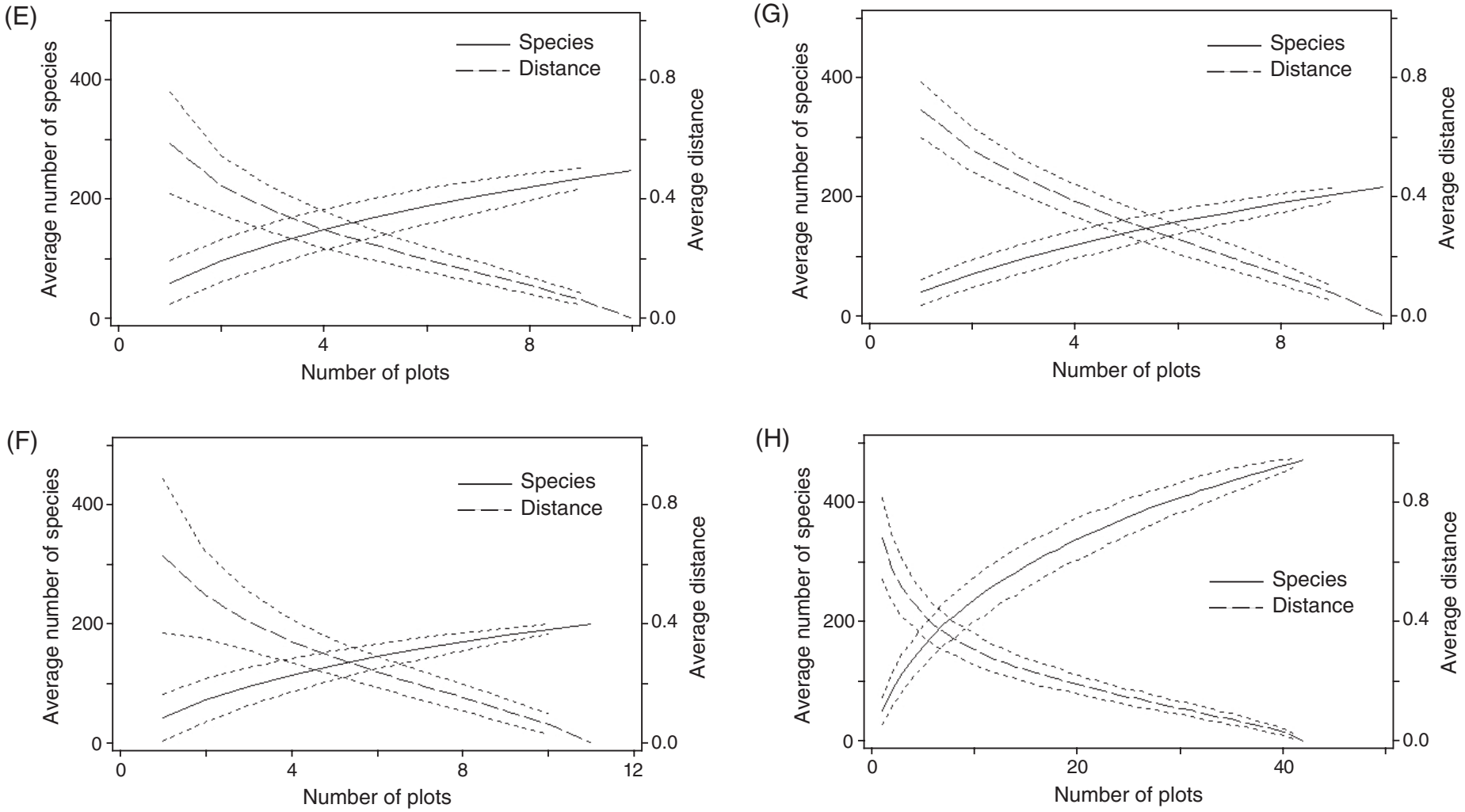


Figure 4.2 (continued)—Species area and distance curves for 12 community types. Community types are identified by letters in table 4.3. Error curves represent plus and minus one standard deviation. (Data source: U.S. Forest Service Forest Inventory and Analysis Program) *(continued on next page)*

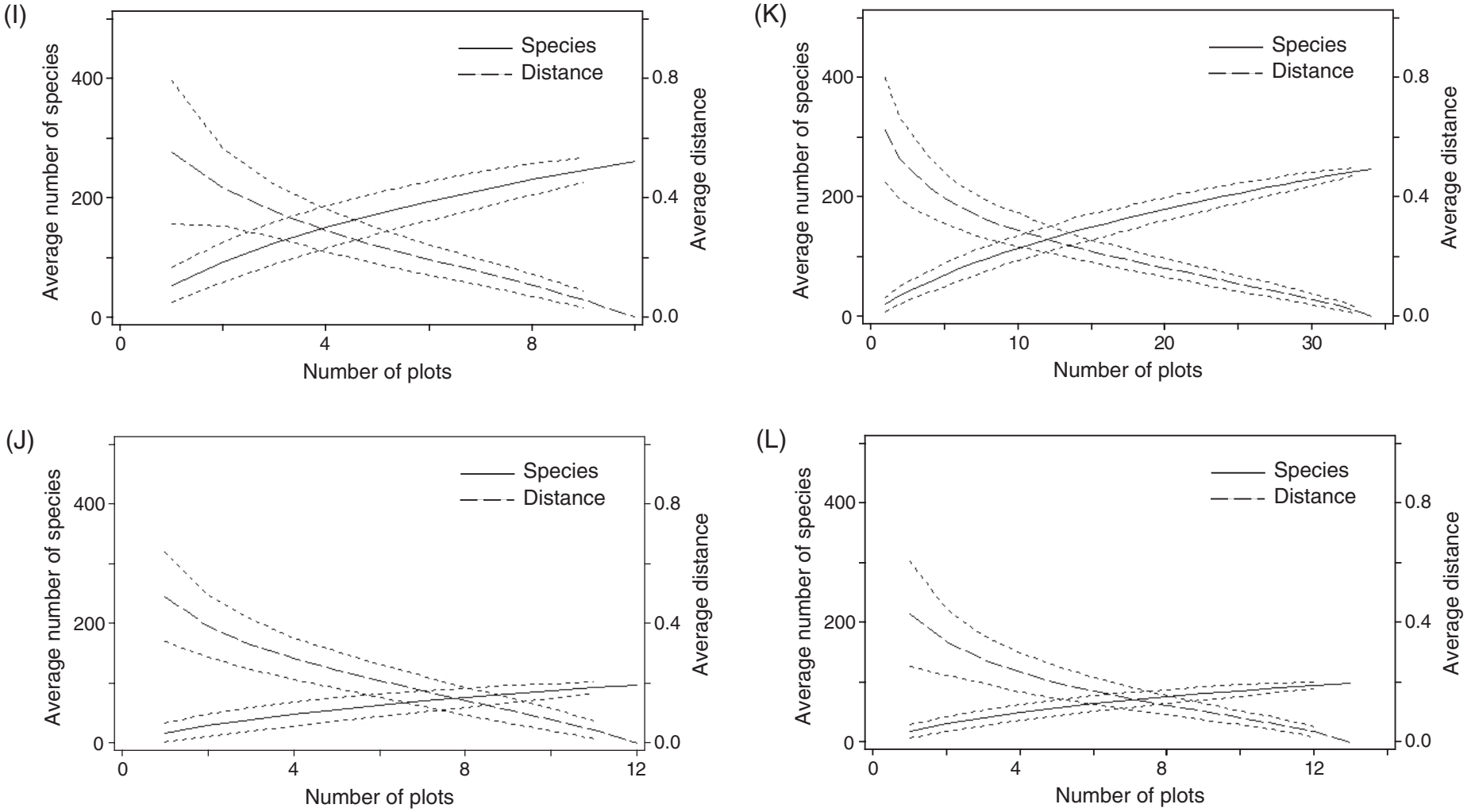


Figure 4.2 (continued)—Species area and distance curves for 12 community types. Community types are identified by letters in table 4.3. Error curves represent plus and minus one standard deviation. (Data source: U.S. Forest Service Forest Inventory and Analysis Program)

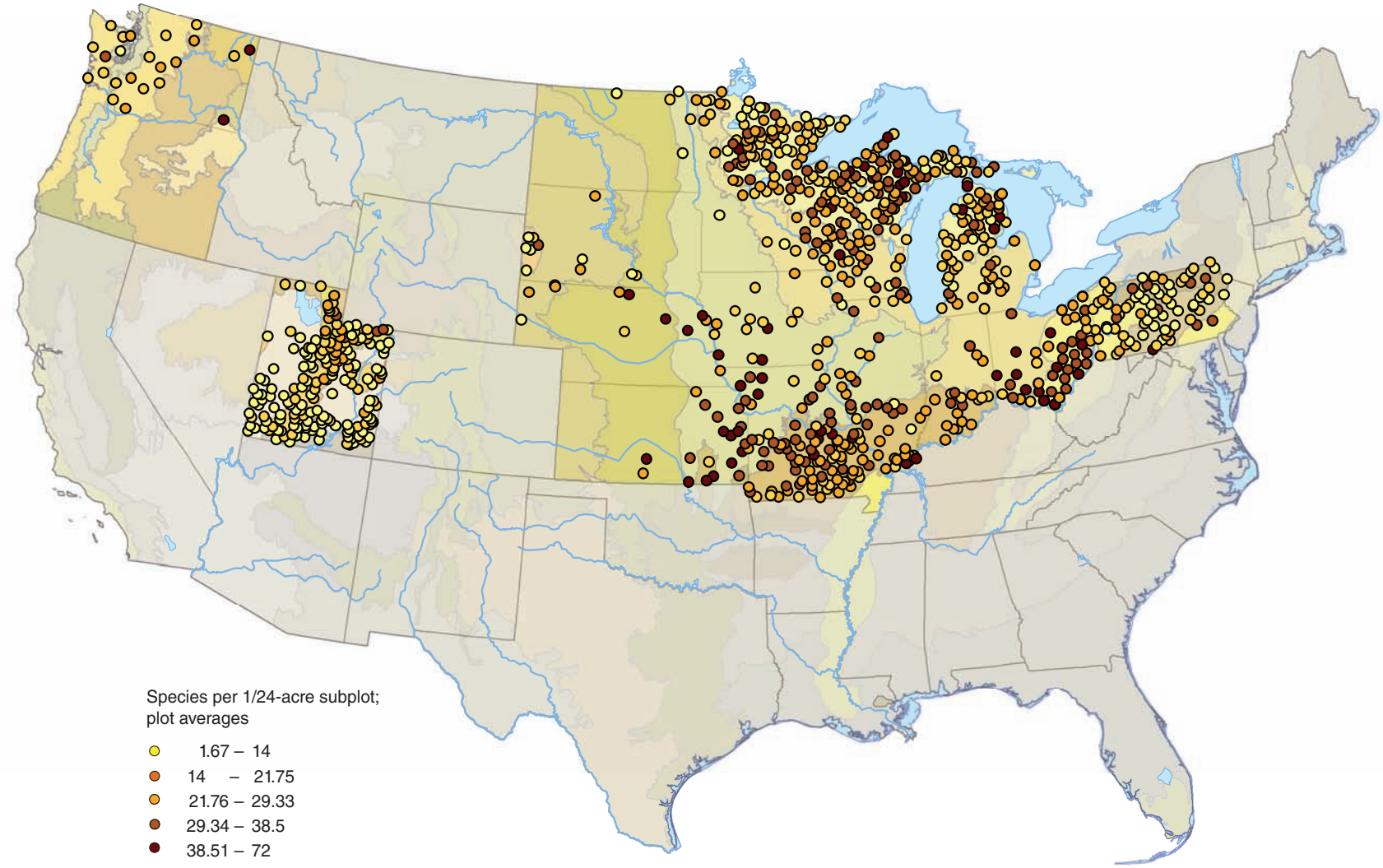


Figure 4.3—Average species alpha for subplots in 100 percent accessible forested conditions. Plot locations are approximate. Background colors correspond to ecological provinces. States with dulled colors have not been sampled. (Data source: U.S. Forest Service Forest Inventory and Analysis Program)

Population estimates of overall species richness by community type were calculated from fully forested plots in single-condition types, limited to 12 community types represented by at least 10 plots. Species area curves (fig. 4.2) indicate no community type is fully represented with the sample sizes available (species area curves do not level off). Some basic trends are supported, though. Most community types detect at least 200 species with 10 plots; the exceptions are the semi-desert, pinyon-juniper woodlands that only capture about 100 species with 10 plots (figs. 4.2J, 4.2K, and 4.2L). Population estimates for the 12 community types are summarized in table 4.4, including jackknife predictors of the true number of species. Most estimates included are highly dependent on sample size and should not be directly compared.

Population estimates of community average plot, subplot, and quadrat alphas reveal that the pinyon-juniper woodlands in the desert and semi-desert provinces (three community types to the far right-hand side of graph) support fewer species per plot than the other communities support on a single subplot [one-quarter of the area of the entire plot (fig. 4.4)]. The figure orders communities by average plot alpha descending order; note that average number of species per subplot and quadrat are not strictly proportional to the number of species observed at the plot level.

Direct comparison of communities should be limited to those with the same sample size. The northern white cedar (*Thuja occidentalis* L.) forest type in the Laurentian Mixed Forest province and the pinyon-juniper woodland in the Nevada-Utah Mountain Semi-Desert Coniferous Forest province are both represented by 13 plots (table 4.4). Independent sample t-tests of average plot, subplot, and quadrat alphas are significantly different (p value ≤ 0.00001 in all cases). The cool, moist northern white cedar forests support more species per unit area than the hot, dry pinyon-juniper forests. The higher species richness also includes a higher proportion of introduced species (3.7 percent) compared to the high desert pinyon-juniper (1 percent) (table 4.3). Forty-six percent of the northern white cedar plots had at least one introduced species, where only 23 percent of the pinyon-juniper plots had an introduced species. Differential diversity (β) at both the population level and the plots level were similar (table 4.4).

Table 4.4—Species richness and differentiation for 12 community types

Community type					Estimates of "true" gamma			
Ecological province and forest type	n	Average plot alpha[a]	(SE)	Beta	Gamma	1st order jackknife	2nd order jackknife	Average plot beta[b]
					number of species			
Laurentian Mixed Forest (212)								
Northern white-cedar (Cedar)	13	50.0	(4.53)	5.6	270	408.5	491.2	1.62
Sugar maple/beech/birch (SMaple)	28	42.8	(3.49)	8.3	335	475.8	558.6	1.74
Hard maple/basswood (HMaple)	14	44.9	(2.99)	6.3	271	416	517	1.84
Aspen (Aspen)	33	48.8	(2.91)	9.4	438	639.7	775	1.78
Eastern Broadleaf Forest (221)								
White oak/red oak/hickory (Oak-Hic)	10	62.6	(6.25)	4.1	250	375.1	458.6	1.82
Sugar maple/beech/birch (SMaple)	11	43.5	(6.37)	4.7	201	292.8	344.5	1.77
Midwest Broadleaf Forest (222)								
White oak/red oak/hickory (Oak-Hic)	10	41.8	(3.61)	5.4	218	339.5	421.2	1.85
Central Interior Broadleaf Forest (223)								
White oak/red oak/hickory (Oak-Hic)	42	53.3	(1.78)	9.4	470	657	753.9	1.93
White oak (WOak)	10	59.6	(5.08)	4.8	262	395	481	2.05
Colorado Plateau Semi-Desert (313)								
Pinyon/juniper woodland (PJ)	12	17.9	(2.36)	5.5	98	153	191	1.72
Interior Semi-Desert and Desert (341)								
Pinyon/juniper woodland (PJ)	34	20.5	(1.06)	12.3	248	387.8	485.9	1.74
Nevada-Utah Mountain Semi-Desert Coniferous Forest (M341)								
Pinyon/juniper woodland (PJ)	13	18.5	(1.54)	5.5	100	148.9	179.8	1.55

[a] Average number of species per 0.16-acre plot (four 0.04-acre subplots).

[b] Average plot beta $= \frac{\sum_{n}^{p=1} \frac{\gamma_{plot}}{\overline{\alpha}_{s}}}{n}$ and signified the average amount of species composition change across four subplots in a "homogenous" forest type.

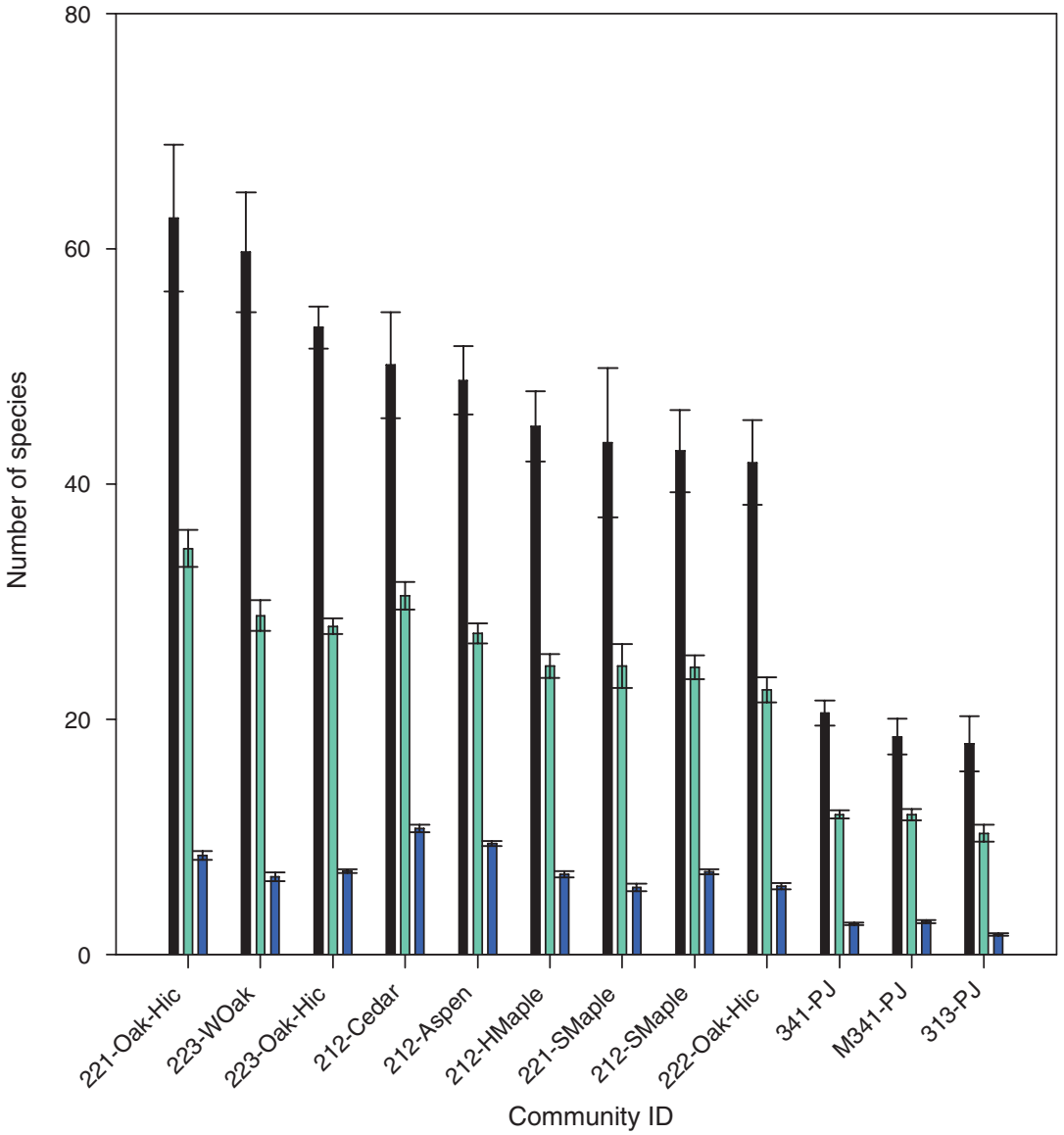


Figure 4.4—Average plot, subplot, and quadrat alphas for 12 community types. They are average plot-black; average subplot-green; average quadrat-blue. Community types are identified by ecological province codes and forest type abbreviations shown in table 4.3. Error bars show plus and minus one standard error. (Data source: U.S. Forest Service Forest Inventory and Analysis Program)

Species Distributions—The distribution of individual species at broad scales is described by constancy. The most common species by occurrence is Virginia creeper [*Parthenocissus quinquefolia* (L.) Planch.] (table 4.5). Other species shown were selected based on how common they appeared in the entire database for the 915 plots, and to provide some comparison of introduced and native species (table 4.5, I and N) with similar growth habits and constancies.

Species dominance is indicated by frequency, the proportion of standard area sample units where the species was recorded, and its abundance, recorded as canopy cover. Note that the native American fly honeysuckle (*Lonicera canadensis* Bart. ex Marsh.) was encountered on more plots than the introduced Japanese honeysuckle (*L. japonica* Thunb.), but Japanese honeysuckle has a much higher average cover and quadrat frequency on the plots where it does occur, indicating invasive plant tendencies.

Multiflora rose (*Rosa multiflora* Thunb.) and cheatgrass (*Bromus tectorum* L.) are the two most common introduced species found in the gamma analyses for all plots. They also are the only two introduced species listed in the top 5 percent most common species in the 12 plant communities (table A4.1). Both are considered to be invasive.

Invasive and Introduced Plant Species—Of the 915 plots included, 67 percent had at least 1 known introduced species identified. Of the 301 plots with no introduced species recorded at the time they were initially visited, 42 had all plants identified to species, with no unknowns. The percentage of introduced species with respect to the total number of species recorded and the relative proportion of introduced plant species cover for the included 915 plots is shown in figure 4.5. The top 10 most frequently recorded introduced species (table 4.6) are either considered to be noxious or invasive.

The broadscale geographic distribution of invasive species of interest, grouped by growth habit (grasses, forbs, and shrubs), are displayed in figures 4.6, 4.7, and 4.8. Reed canarygrass (*Phalaris arundinacea* L.) is a native species that is considered invasive; once widely used to restore roadbed cuts, this species has demonstrated the ability to quickly overtake wetland systems. Garlic mustard [*Alliaria petiolata* (M. Bieb.) Cavara & Grande] is an introduced forb that is highly shade tolerant, readily spreads in undisturbed forests, and has allelopathic properties to inhibit other plant growth. It has been recorded on many plots in the Great Lakes States, and its average cover and frequencies on subplots and quadrats (table 4.5) indicate its invasive tendencies. Invasive shrubs are well established in the North Central States; multiflora rose was handed out to the public free of charge for wildlife enhancement in the 1960s.

Table 4.5—Distribution of selected native and introduced species

Growth habit	Scientific name	Common name	Origin	All plots (n = 915) constancy	Plots where present		
					Average cover	Subplot frequency	Quadrat frequency
Vine	*Parthenocissus quinquefolia*	Virginia creeper	N	41.09	6.60	0.79	0.36
Tree	*Acer rubrum*	Red maple	N	35.41	15.15	0.77	0.33
Shrub, vine	*Toxicodendron radicans*	Eastern poison ivy	N	32.46	7.98	0.63	0.21
Tree	*Quercus rubra*	Northern red oak	N	29.95	10.50	0.59	0.09
Vine, shrub	*Rosa multiflora*	Multiflora rose	I	23.50	7.90	0.62	0.16
Tree, shrub	*Sassafras albidum*	Sassafras	N	17.38	8.98	0.72	0.21
Graminoid	*Achnatherum hymenoides*	Indian ricegrass	N	8.42	1.53	0.64	0.11
Graminoid	*Bromus tectorum*	Cheatgrass	I	8.42	4.47	0.72	0.33
Shrub	*Lonicera canadensis*	American fly honeysuckle	N	8.20	2.63	0.60	0.11
Subshrub, shrub	*Mahonia repens*	Creeping barberry	N	5.68	2.79	0.62	0.25
Vine	*Lonicera japonica*	Japanese honeysuckle	I	5.57	16.90	0.64	0.34
Forb/herb	*Daucus carota*	Queen Anne's lace	I	5.36	3.89	0.50	0.18
Forb/herb	*Phytolacca americana*	American pokeweed	N	5.36	3.57	0.46	0.07
Forb/herb	*Alliaria petiolata*	Garlic mustard	I	4.48	8.90	0.67	0.33
Forb/herb	*Apocynum androsaemifolium*	Spreading dogbane	N	4.48	1.71	0.50	0.16
Forb/herb	*Circaea lutetiana*	Broadleaf enchanter's nightshade	N	4.48	2.37	0.62	0.22
Forb/herb	*Leucanthemum vulgare*	Oxeye daisy	I	4.48	2.14	0.47	0.11
Forb/herb	*Sanguinaria canadensis*	Bloodroot	N	4.48	2.22	0.49	0.10

N = native species; I = introduced species.

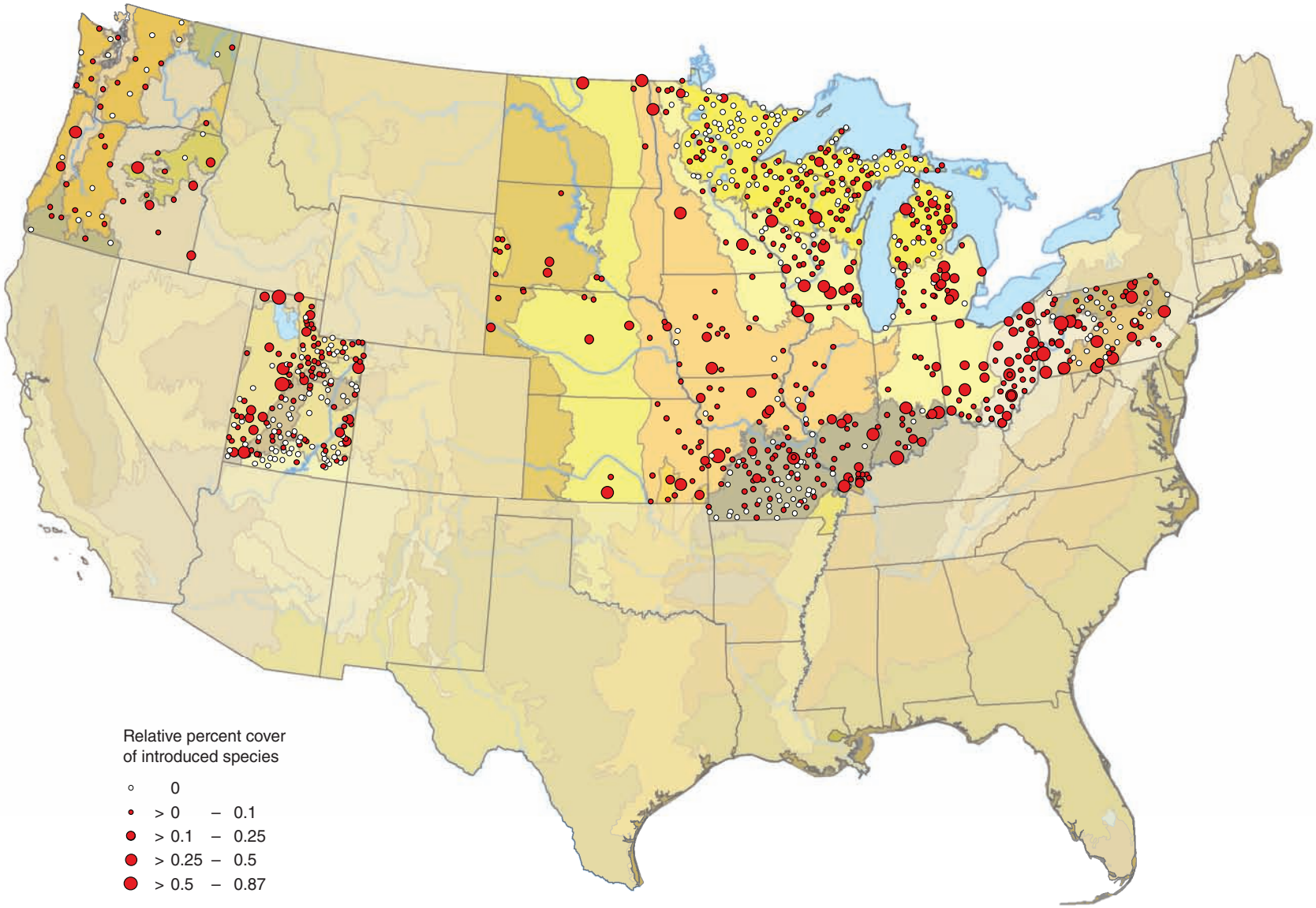


Figure 4.5—Relative percent cover of introduced plant species. White dots represent plots where no introduced plants were recorded, and red dot sizes correspond to increasing levels of infestation. Plot locations are approximate. Background colors correspond to ecological provinces. States with dulled colors have not been sampled at the time of report preparation. (Data source: U.S. Forest Service Forest Inventory and Analysis Program)

Table 4.6—Twenty-five most commonly recorded introduced species (n = 915)

Growth habit	Scientific name	Common name	Constancy
Vine, shrub	*Rosa multiflora*	Multiflora rose	23.50
Graminoid	*Bromus tectorum*	Cheatgrass	8.42
Vine	*Lonicera japonica*	Japanese honeysuckle	5.57
Forb/herb	*Daucus carota*	Queen Anne's lace	5.36
Forb/herb	*Hieracium aurantiacum*	Orange hawkweed	5.14
Vine, forb/herb	*Polygonum convolvulus*	Black bindweed	5.03
Forb/herb	*Leucanthemum vulgare*	Oxeye daisy	4.48
Forb/herb	*Alliaria petiolata*	Garlic mustard	4.48
Forb/herb	*Tragopogon dubius*	Yellow salsify	4.26
Graminoid	*Phleum pratense*	Timothy	4.04
Forb/herb	*Trifolium repens*	White clover	3.93
Forb/herb	*Polygonum persicaria*	Spotted ladysthumb	3.72
Forb/herb	*T. pratense*	Red clover	3.72
Forb/herb	*Glechoma hederacea*	Ground ivy	3.50
Graminoid	*Dactylis glomerata*	Orchardgrass	3.39
Subshrub, forb/herb	*Solanum dulcamara*	Climbing nightshade	3.28
Forb/herb	*Hypericum perforatum*	Common St. Johnswort	3.28
Forb/herb	*Arctium minus*	Lesser burdock	3.17
Forb/herb	*Cirsium arvense*	Canada thistle	3.17
Forb/herb	*Rumex acetosella*	Common sheep sorrel	2.95
Graminoid	*Poa compressa*	Canada bluegrass	2.95
Graminoid	*Lolium pratense*	Meadow fescue	2.84
Forb/herb	*Melilotus officinalis*	Yellow sweetclover	2.84
Tree, shrub	*Rhamnus cathartica*	Common buckthorn	2.84
Tree, shrub	*Morus alba*	White mulberry	2.84

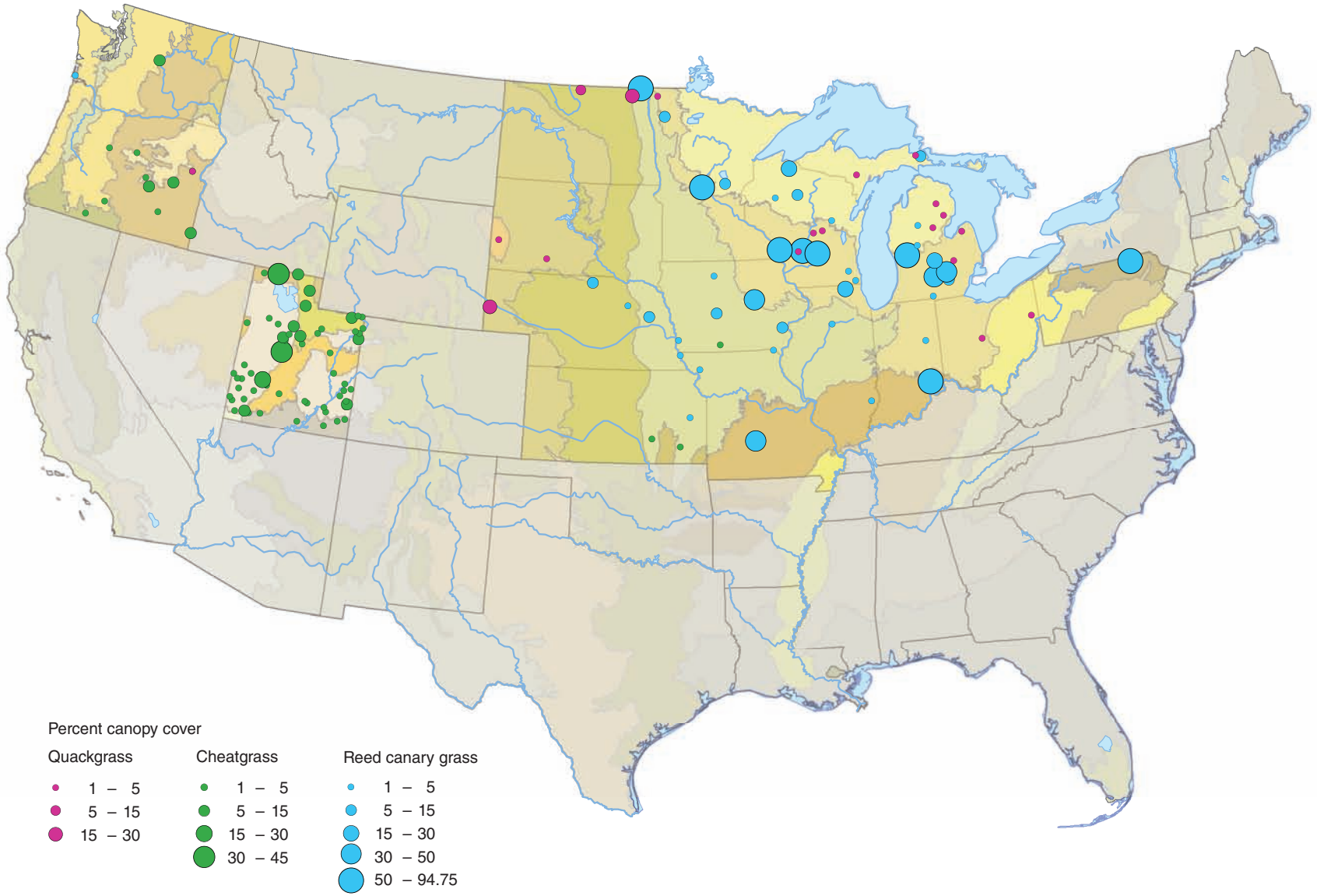


Figure 4.6—Broad scale distributions of three invasive grass species, with percent canopy cover. Plot locations are approximate. (Data source: U.S. Forest Service Forest Inventory and Analysis Program)

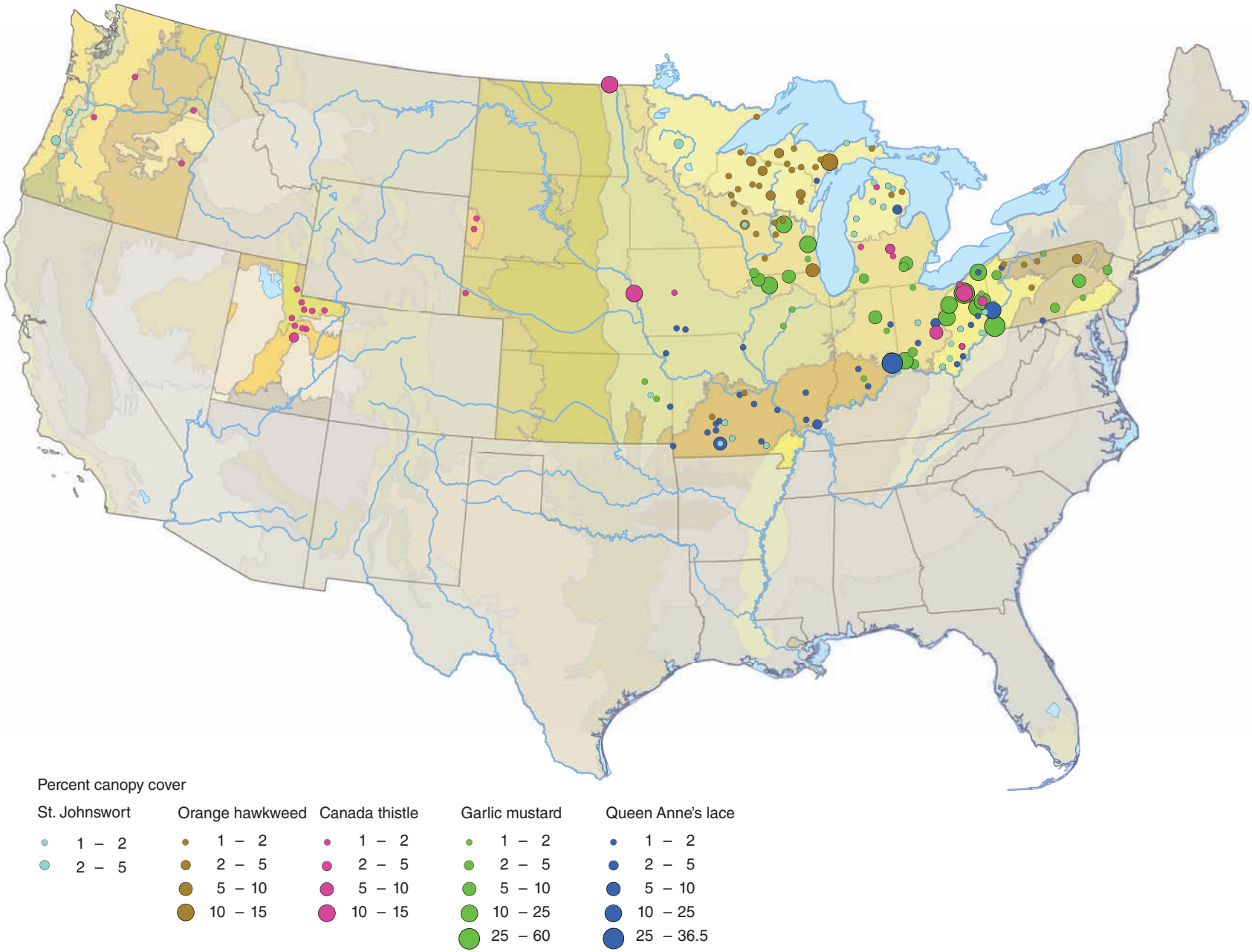


Figure 4.7—Broad scale distributions of five introduced herb species, with percent canopy cover. Plot locations are approximate. (Data source: U.S. Forest Service Forest Inventory and Analysis Program)

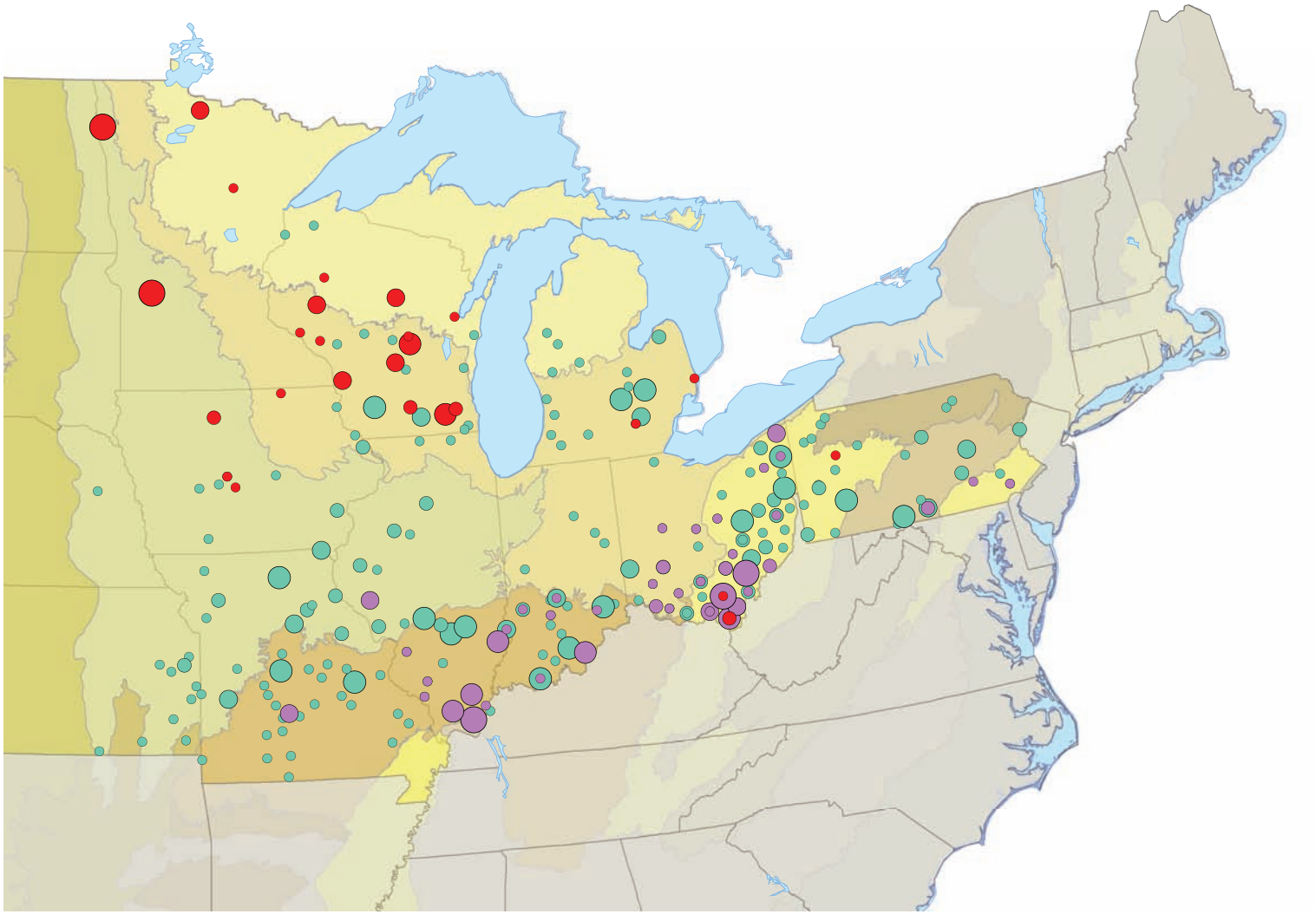


Figure 4.8—Broad scale distributions of three introduced, invasive shrub species, with percent canopy cover. Plot locations are approximate. (Data source: U.S. Forest Service Forest Inventory and Analysis Program)

Conclusions

Over the widely dispersed 915 plots, 2,975 unique plant codes were recorded, with 2,595 identified to species. The three most commonly recorded plant species are Virginia creeper, red maple (*Acer rubrum* L.), and poison ivy [*Toxicodendron radicans* (L.) Kuntze]. Across all 915 plots, only 7 percent of all individual plant observations recorded to species are introduced. Although 17 percent of all individual plant observations were not identified to species, examination of the genera identified suggest that the proportion of introduced species among the unidentified plants raises the estimate of percentage of introduced species by about 1 percent.

When assessing such an extensively dispersed plot network, many species are expected to be missed. Species composition data is more informative and less variable when grouped by ecological provinces and forest type to define communities. Community types defined with larger sample sizes are more thoroughly characterized. Some of the most commonly observed species in these communities are present in low abundance. Inventories that are limited to the most abundant species would have missed these species the majority of the time.

Number of species is a fundamental and easily interpretable measure of diversity. Species richness in itself is not an indicator of forest health; it is a technical characteristic of plant communities. However, when sample unit areas are standard, species richness trends over space and time can be assessed. Results from varied regions of the country quantify the intuitive impressions that forests in dry, hot climates support fewer species than cool, moist forests. An entire plot in the pinyon-juniper woodlands supports fewer species than found on a single subplot (one-fourth the area of a plot) in any of the other community types summarized.

Change or lack of change in species numbers does not reveal the entire story. Changes in species present on a site over time will reflect normal succession or changing conditions brought about by natural disturbance or management practices. Plant species composition data, along with information on site conditions

and disturbance history, allow researchers to hypothesize what forces are driving changes and predict future trends.

Of the 915 plots, 67 percent had at least 1 introduced plant species. The most commonly encountered introduced species is multiflora rose, with a constancy of 23.5 percent. The next most common introduced species recorded were cheatgrass and Japanese honeysuckle, with constancies of 8.42 and 5.57 percent, respectively. Each region of the country has plots with either no introduced species or a high percentage of introduced species numbers with high relative cover.

Species frequency and abundance together provide a clearer description of dominance across a sample unit than either alone. A species with a small growth habit may not contribute much in the way of canopy cover but may be widespread across a plot or population and have a high quadrat frequency. Conversely, a single large tree may provide the majority of canopy cover on a subplot but not be rooted in or overhanging any quadrats. Plants considered to be invasive often have higher frequencies than native species of the same growth habit.

Due to the extensive spatial scale of the P3 grid, the vegetation indicator cannot be expected to capture all species in any community type sampled. However, the indicator does allow for broadscale point-in-time estimates and will be a powerful tool for assessing change when plots are revisited. Because the vegetation indicator focuses on all vascular plants on the plot, all introduced species, not just a list of crisis species of the day, are monitored (Rudis and others 2006). Species that may become serious threats in the future—as well as today's species of concern that may become less important as invaders in the future—will be tracked through time.

Literature Cited

Bailey, R.G. 1995. Description of the ecoregions of the United States. 2d ed. Misc. Publ. 1391.Washington, DC: Forest Service, U.S. Department of Agriculture. 108 p. [Map scale 1:7,500,000].

Cleland, D.T.; Freeouf, J.A.; Keys, J.E. [and others]. 2007. Ecological subregions: sections and subsections for the conterminous United States. Sloan, A.M., cartog. Gen. Tech. Rep. WO-76. Washington, DC: Forest Service, U.S. Department of Agriculture. [Map, presentation scale 1: 3,500,000; colored]. [Also on CD–ROM as a Geographic Information System coverage in ArcINFO format].

Colwell, R.K.; Mao, C. Xuan; Chang, J. 2004. Interpolating, extrapolating, and comparing incidence-based species accumulation curves. Ecology. 85: 2717–2727.

Heltshe, J.F.; Forrester, N.E. 1983. Estimating species richness using the jackknife procedure. Biometrics. 39: 1–11.

McCune, B.; Grace, J.B. 2002. Analysis of ecological communities. Gleneden Beach, OR: MjM Software Design. 300 p.

McCune, B.; Mefford, M.J. 1999. PC–ORD. Multivariate analysis of ecological data, Version 4. Gleneden Beach, OR: MjM Software Design.

Rudis, V.A.; Gray, A.; McWilliams, W. [and others]. 2006. Regional monitoring of nonnative plant invasions with the forest inventory and analysis program. In: McRoberts, R.E. [and others], eds. Proceedings of the sixth annual FIA symposium. Gen. Tech. Rep. WO–70. Washington, DC: Forest Service, U.S. Department of Agriculture: 49–64.

Scheiner, S.M. 2003. Six types of species-area curves. Global Ecology and Biogeography. 12: 441–447.

Forest Service, U.S. Department of Agriculture. 2004. National report on sustainable forests – 2003. FS–766. Washington, DC. 139 p. http://www.fs.fed.us/research/sustain/2003SustainabilityReport/. [Date accessed: Unknown].

Forest Service, U.S. Department of Agriculture. 2006. The forest inventory and analysis database: database description and users guide. Version 2.1. Research Triangle Park, NC. 197 p.

U.S. Department of Agriculture, Natural Resource Conservation Service. 2000. The PLANTS Database. Baton Rouge, LA: National Plant Data Center. http://npdc.usda.gov/. [Date accessed: March 3, 2000].

Whittaker, R.J.; Willis, K.J.; Field, R. 2001. Scale and species richness: towards a general, hierarchical theory of species diversity. Journal of Biogeography. 28: 453–470.

Appendix

Table A4.1—Top 5 percent most common species or genera for 12 community types

Community type	Scientific name	Common name	Origin	Constancy	Cover[a]
Laurentian Mixed Forest					
Northern white-cedar (n = 13)	*Thuja occidentalis*	Northern white cedar	N	100	28
	Viola	Violet	–	100	2
	Trientalis borealis	Starflower	N	100	1
	Abies balsamea	Balsam fir	N	92	13
	Coptis trifolia	Threeleaf goldthread	N	92	2
	Maianthemum canadense	Canada mayflower	N	92	1.2
	Cornus canadensis	Bunchberry dogwood	N	85	1.3
	Carex	Sedge	–	77	14
	Betula papyrifera	Paper birch	N	77	4.3
	Alnus incana ssp. *rugosa*	Speckled alder	N	69	25
	Fraxinus nigra	Black ash	N	69	8
	Rubus pubescens	Dwarf red blackberry	N	69	3.6
	Clintonia borealis	Bluebead	N	69	0.8
Sugar maple/beech/birch (n = 28)	*Maianthemum canadense*	Canada mayflower	N	93	1.7
	Acer saccharum	Sugar maple	N	89	45
	Abies balsamea	Balsam fir	N	86	8.7
	Trientalis borealis	Starflower	N	86	1.5
	Acer rubrum	Red maple	N	79	9
	Betula alleghaniensis	Yellow birch	N	68	8.3
	Carex	Sedge	–	68	6.8
	Lonicera canadensis	American fly honeysuckle	N	64	1.5
	Viola	Violet	–	64	1.45
	Dryopteris carthusiana	Spinulose woodfern	N	57	3.7
	Prunus serotina	Black cherry	N	57	2.4
	Pteridium aquilinum	Western brackenfern	N	46	17
	Tsuga canadensis	Eastern hemlock	N	46	6.5
	Populus tremuloides	Quaking aspen	N	46	3
	B. papyrifera	Paper birch	N	46	2.8
	Maianthemum racemosum ssp. *racemosum*	Feathery false lily of the valley	N	46	0.5
	Eurybia macrophylla	Bigleaf aster	N	43	10
	Picea glauca	White spruce	N	43	2.8
	Aralia nudicaulis	Wild sarsaparilla	N	43	1.5
	C. intumescens	Greater bladder sedge	N	43	1

continued

Table A4.1—Top 5 percent most common species or genera for 12 community types (continued)

Community type	Scientific name	Common name	Origin	Constancy	Cover[a]
Laurentian Mixed Forest (continued)					
Hard maple/basswood (n = 14)	*Tilia americana*	American basswood	N	100	11.6
	Ostrya virginiana	Hophornbeam	N	93	8.3
	Acer saccharum	Sugar maple	N	86	56
	Fraxinus americana	White ash	N	79	62
	Maianthemum canadense	Canada mayflower	N	79	1
	Arisaema triphyllum	Jack in the pulpit	N	79	0.6
	Aralia nudicaulis	Wild sarsaparilla	N	71	5.25
	Carex	Sedge	–	64	4.5
	Osmorhiza claytonii	Clayton's sweetroot	N	64	1.9
	Viola	Violet	–	64	1.2
	Eurybia macrophylla	Bigleaf aster	N	64	1
	Caulophyllum thalictroides	Blue cohosh	N	57	0.6
	F. nigra	Black ash	N	50	55
	Dryopteris carthusiana	Spinulose woodfern	N	50	2.25
	Athyrium filix-femina	Common ladyfern	N	50	1.75
	Trillium grandiflorum	White trillium	N	50	1.1
Aspen (n = 33)	*Maianthemum canadense*	Canada mayflower	N	97	1.5
	Populus tremuloides	Quaking aspen	N	94	28
	Eurybia macrophylla	Bigleaf aster	N	88	6.6
	Aralia nudicaulis	Wild sarsaparilla	N	76	4.6
	Trientalis borealis	Starflower	N	76	1
	Pteridium aquilinum	Western brackenfern	N	73	18.5
	Corylus cornuta	Beaked hazelnut	N	67	10.7
	Viola	Violet	–	67	1
	Clintonia borealis	Bluebead	N	64	2
	Abies balsamea	Balsam fir	N	61	8.3
	Solidago	Goldenrod	–	61	4
	Cornus canadensis	Bunchberry dogwood	N	61	3
	Rubus pubescens	Dwarf red blackberry	N	61	2
	Acer rubrum	Red maple	N	58	9
	Carex	Sedge	–	58	7.4
	Betula papyrifera	Paper birch	N	52	4.5
	R. idaeus ssp. *strigosus*	Grayleaf red raspberry	N	52	2
	Fragaria virginiana	Virginia strawberry	N	52	1.25

continued

Table A4.1—Top 5 percent most common species or genera for 12 community types (continued)

Community type	Scientific name	Common name	Origin	Constancy	Cover[a]
Laurentian Mixed Forest (continued)	*Prunus serotina*	Black cherry	N	45	6.7
Aspen (n = 33) (continued)	*Anemone quinquefolia*	Nightcaps	N	42	1
	P. virginiana	Chokecherry	N	39	1.8
	Thalictrum dioicum	Early meadow-rue	N	36	3
Eastern Broadleaf Forest					
White oak/red oak/hickory (n = 10)	*Quercus velutina*	Black oak	N	100	9
	Prunus serotina	Black cherry	N	100	7
	Parthenocissus quinquefolia	Virginia creeper	N	100	5.9
	Toxicodendron radicans	Eastern poison ivy	N	100	5.5
	Viola	Violet	–	100	1.65
	Rosa multiflora	Multiflora rose	I	90	8.3
	Q. alba	White oak	N	90	6.9
	Cornus florida	Flowering dogwood	N	90	6.75
	Sassafras albidum	Sassafras	N	90	6
	Fraxinus americana	White ash	N	90	4.3
	Acer rubrum	Red maple	N	80	18.75
	Ulmus americana	American elm	N	80	5.5
	Polystichum acrostichoides	Christmas fern	N	80	4.4
	Rubus occidentalis	Black raspberry	N	80	1.3
Sugar maple/beech/birch (n = 11)	*Acer rubrum*	Red maple	N	82	14.75
	Prunus serotina	Black cherry	N	82	8.8
	A. saccharum	Sugar maple	N	73	34.3
	Fagus grandifolia	American beech	N	73	21.2
	Fraxinus americana	White ash	N	73	7.4
	Parthenocissus quinquefolia	Virginia creeper	N	73	2.75
	Toxicodendron radicans	Eastern poison ivy	N	73	1.4
	Rosa multiflora	Multiflora rose	I	64	8.7
	Lindera benzoin	Northern spicebush	N	64	6.7
	Ulmus americana	American elm	N	64	5.5
	Carya cordiformis	Bitternut hickory	N	64	3.2
	Carex	Sedge	–	64	1.9
	Polystichum acrostichoides	Christmas fern	N	64	1.7
	Smilax rotundifolia	Roundleaf greenbrier	N	64	1
	Viola	Violet	–	64	1

continued

Table A4.1—Top 5 percent most common species or genera for 12 community types (continued)

Community type	Scientific name	Common name	Origin	Constancy	Cover[a]
Midwest Broadleaf Forest					
White oak/red oak/hickory (n = 10)	*Parthenocissus quinquefolia*	Virginia creeper	N	100	4.2
	Prunus serotina	Black cherry	N	90	10.87
	Quercus alba	White oak	N	90	8
	Ulmus americana	American elm	N	70	18
	Q. rubra	Northern red oak	N	70	13.1
	Populus tremuloides	Quaking aspen	N	70	4.7
	Ribes cynosbati	Eastern prickly gooseberry	N	70	4.6
	Carya ovata	Shagbark hickory	N	60	9.8
	C. cordiformis	Bitternut hickory	N	60	6.5
	Betula papyrifera	Paper birch	N	50	8.2
	Corylus americana	American hazelnut	N	50	3.1
	Rubus allegheniensis	Allegheny blackberry	N	50	2
	Circaea lutetiana	Broadleaf enchanter's nightshade	N	50	1.5
	Toxicodendron radicans	Eastern poison ivy	N	50	1
Central Interior Broadleaf Forest					
White oak/red oak/hickory (n = 42)	*Parthenocissus quinquefolia*	Virginia creeper	N	100	9.9
	Quercus alba	White oak	N	95	25.5
	Q. velutina	Black oak	N	81	17.1
	Sassafras albidum	Sassafras	N	81	8.7
	Toxicodendron radicans	Eastern poison ivy	N	71	8.7
	Prunus serotina	Black cherry	N	71	2
	Cornus florida	Flowering dogwood	N	67	17
	Symphoricarpos orbiculatus	Coralberry	N	64	3.5
	Desmodium nudiflorum	Nakedflower ticktrefoil	N	62	6.9
	Q. rubra	Northern red oak	N	60	7
	Vitis aestivalis	Summer grape	N	60	3
	Rhus aromatica	Fragrant sumac	N	52	2.7
	Carex	Sedge	–	50	2.9
	Q. stellata	Post oak	N	48	15.3
	Juniperus virginiana	Eastern redcedar	N	48	4
	Viola	Violet	–	48	1.4
	Acer rubrum	Red maple	N	45	11.25
	Nyssa sylvatica	Blackgum	N	43	8
	Vaccinium pallidum	Blue Ridge blueberry	N	43	5

continued

Table A4.1—Top 5 percent most common species or genera for 12 community types (continued)

Community type	Scientific name	Common name	Origin	Constancy	Cover[a]
Central Interior Broadleaf Forest (continued)	*Potentilla simplex*	Common cinquefoil	N	43	0.5
White oak/red oak/hickory (n = 42) (cont)	*Ulmus rubra*	Slippery elm	N	40	9.26
	Carya alba	Mockernut hickory	N	40	7.8
	Rubus	Blackberry	–	40	1.6
	Galium circaezans	Licorice bedstraw	N	40	1
White oak (n=10)	*Quercus alba*	White oak	N	100	39.25
	Sassafras albidum	Sassafras	N	100	4.5
	Parthenocissus quinquefolia	Virginia creeper	N	100	2.6
	Cornus florida	Flowering dogwood	N	90	9.5
	Q. velutina	Black oak	N	80	15.8
	Prunus serotina	Black cherry	N	80	1.4
	Antennaria plantaginifolia	Woman's tobacco	N	70	0.5
	Juniperus virginiana	Eastern redcedar	N	70	0.5
	Acer rubrum	Red maple	N	60	8.9
	Vaccinium pallidum	Blue Ridge blueberry	N	60	4.5
	Toxicodendron radicans	Eastern poison ivy	N	60	4.2
	Carex	Sedge	–	60	3.4
	Carya texana	Black hickory	N	60	3.29
	Vitis	Grape	–	60	1.4
	Rhus aromatica	Fragrant sumac	N	60	0.24
	C. cordiformis	Bitternut hickory	N	50	1.3
	Lespedeza virginica	Slender lespedeza	N	50	0.5
Colorado Plateau Semi-Desert	*Juniperus osteosperma*	Utah juniper	N	100	10.8
Pinyon/juniper woodland (n = 12)	*Pinus edulis*	Twoneedle pinyon	N	100	7.4
	Ephedra viridis	Mormon tea	N	75	1.3
	Gutierrezia sarothrae	Broom snakeweed	N	75	0.7
	Artemisia tridentata	Big sagebrush	N	58	1.9
	Amelanchier utahensis	Utah serviceberry	N	50	1.8
	Cercocarpus montanus	Alderleaf mountain mahogany	N	42	9.35
	Shepherdia rotundifolia	Roundleaf buffaloberry	N	42	4.5
	Opuntia polyacantha	Plains pricklypear	N	42	1.35
	Poa secunda	Sandberg bluegrass	N	42	1.35
	Achnatherum hymenoides	Indian ricegrass	N	42	0.6
	Penstemon	Beardtongue	–	42	0.4

continued

Table A4.1—Top 5 percent most common species or genera for 12 community types (continued)

Community type	Scientific name	Common name	Origin	Constancy	Cover[a]
Interior Semi-Desert and Desert					
Pinyon/juniper woodland (n = 34)	*Juniperus osteosperma*	Utah juniper	N	100	8.4
	Artemisia tridentata	Big sagebrush	N	71	2.6
	Opuntia polyacantha	Plains pricklypear	N	71	0.84
	Pinus edulis	Twoneedle pinyon	N	68	6.4
	Gutierrezia sarothrae	Broom snakeweed	N	68	1.1
	Ephedra viridis	Mormon tea	N	65	1.2
	Bromus tectorum	Cheatgrass	I	62	1.3
	Achnatherum hymenoides	Indian ricegrass	N	62	0.8
	Poa secunda	Sandberg bluegrass	N	38	1.4
	Pinus monophylla	Singleleaf pinyon	N	35	9
	Petradoria pumila	Grass-leaved rock goldenrod	N	32	0.9
	Cryptantha	Cryptantha	–	29	0.4
	Elymus elymoides	Squirreltail	N	26	0.72
	Purshia tridentata	Antelope bitterbrush	N	24	1.1
	Penstemon	Beardtongue	–	24	0.46
Nevada-Utah Mountain Semi-Desert Coniferous Forest					
Pinyon/juniper woodland (n = 13)	*Juniperus osteosperma*	Utah juniper	N	100	9.4
	Pinus edulis	Twoneedle pinyon	N	92	13.8
	Achnatherum hymenoides	Indian ricegrass	N	85	0.7
	Cercocarpus montanus	Alderleaf mountain mahogany	N	54	3.9
	Gutierrezia sarothrae	Broom snakeweed	N	54	0.9
	Poa secunda	Sandberg bluegrass	N	54	0.9
	Phlox austromontana	Mountain phlox	N	54	0.8
	Opuntia polyacantha	Plains pricklypear	N	46	1
	Artemisia tridentata	Big sagebrush	N	38	2.25
	Purshia tridentata	Antelope bitterbrush	N	38	1.6
	Ephedra viridis	Mormon tea	N	38	1.35
	Petradoria pumila	Grass-leaved rock goldenrod	N	38	1.1
	Cryptantha	Cryptantha	–	38	0.6

– = designations are not included for taxa identified as genera (blank cells); some species within genera may be either native or introduced; N = native species; I = introduced species.

[a] Cover is average cover over plots where species occurs. To determine average cover over entire community, multiply cover by (constancy/100 percent).

Chapter 5. Tree Mortality

Mark J. Ambrose

Introduction

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

In previous national technical reports of the National Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture, mortality was analyzed using FHM data and data from the phase 3 (P3) grid from the Forest Inventory and Analysis (FIA) Program of the Forest Service. Those data spanned a relatively long time period, but the sample was not spatially intense (approximately one plot: 96,000 acres). In this report, a similar method is applied to FIA phase 2 (P2) data as a demonstration of how the more intensive P2 dataset can be used in forest health analyses.

Data

Mortality analysis was possible for areas where data were available from repeated plot measurements using consistent sampling protocols. Repeated annualized P2 plot measurements were available from Iowa, Indiana, Michigan, Minnesota, Missouri, and Wisconsin. Initial plot measurements occurred in 1999 and 2000, and plots were revisited in 2004 and 2005. Two P2 panels of remeasurement data were available for all States except Michigan and Wisconsin, where only one panel had been remeasured. In Minnesota, Michigan, Wisconsin, and parts of Missouri the P2 plot network had been intensified two to three times, so even with only one or two panels of data, the sample was quite large. Remeasurement data were available for all six States from at least one plot per 15,000 acres. The States included in this analysis, as well as the forest cover within those States, are shown in figure 5.1.

Methods

The FIA P2 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 was calculated by FIA and provided in the FIA database (version 2.0). 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 2005d). Exactly two measurements of each plot were available in the dataset. The gross growth and the mortality over the interval between each pair of plot measurements, in tons of biomass per acre, were calculated for each plot. Then, average growth and mortality rates were independently estimated for each ecoregion section (Cleland and others 2005) using simple linear regression.[1] MRATIOs

[1] In previous FHM reports (Coulston and others 2005a, 2005b, 2005c, 2005d) growth and mortality rates were estimated using a more complex mixed modeling procedure (Smith and Conkling 2004). The mixed model was most useful at efficient estimation using data where not all plots had been measured at the same time intervals (Gregoire and others 1995). Because the FIA P2 data used here had all plots measured on a 5-year cycle with exactly two measurements of each plot and 80 percent overlap of the time intervals between measurements, a simpler linear regression model was used.

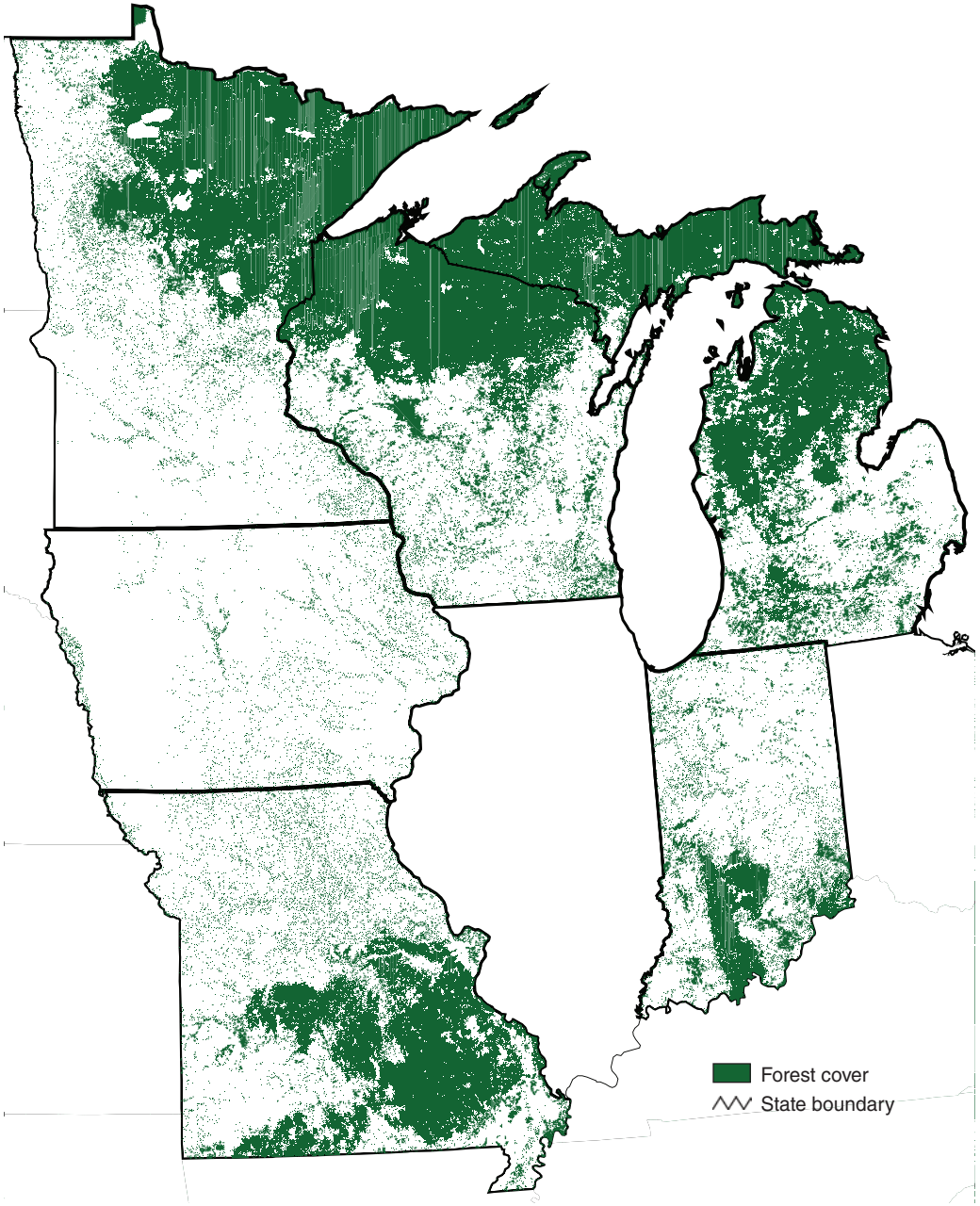


Figure 5.1—Forest cover in the States where mortality was analyzed. Forest cover was derived from Advanced Very High Resolution satellite imagery (Zhu and Evans 1994).

were then calculated from the growth and mortality rates.

The value of the MRATIO is that it normalizes mortality by growth rate, allowing a comparison of mortality rates across diverse regions of the United States. For analyses of smaller areas of the country within which growth rates are similar, it may be more useful in general to consider absolute measures of mortality. The MRATIO was used as the mortality indicator for this report even though the available data were limited to a relatively small region (fig. 5.1) as a demonstration of how this method may be applied using FIA P2 data.

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

In addition, the ratio of average dead-tree diameter to average live-tree diameter (DDLD ratio) was calculated for each plot where mortality occurred. Low DDLD ratios (much < 1) usually indicate competition-induced mortality typical of young, vigorous stands, while high ratios (much > 1) indicate mortality associated with senescence or some external factors such as insects or disease (Smith and Conkling 2004). Intermediate DDLD ratios can be hard to interpret because a variety of stand conditions can produce such values. The DDLD ratio is most useful for analyzing mortality within regions that 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, had produced a deceptively high DDLD.

To further analyze tree mortality, the number of stems and total biomass of trees that had died were calculated by species within each ecoregion. Identifying the tree species experiencing high mortality in an ecoregion is a first step in identifying what forest health issue may be affecting the forests. Although determining particular causal agents associated with observed mortality with certainty is beyond the scope of this report, often there are well-known insects and pathogens that are "likely suspects" once the affected species are identified.

Results and Discussion

The MRATIO values are shown in figure 5.2. The highest MRATIO (1.04) occurred in ecoregion section 251G—Missouri Loess Hills (previously named 251G—Central Loess Plains; NcNab and Avers 1994). Other areas of high mortality relative to growth occurred in northern Minnesota, in ecoregion sections 212L—Northern Superior Uplands, 212M—Northern Minnesota and Ontario, and 222N—Lake Agassiz, Aspen Parklands.

Results of the analysis of the relative sizes of trees that died, the DDLD ratio, is shown in figure 5.3. DDLD values vary widely within any

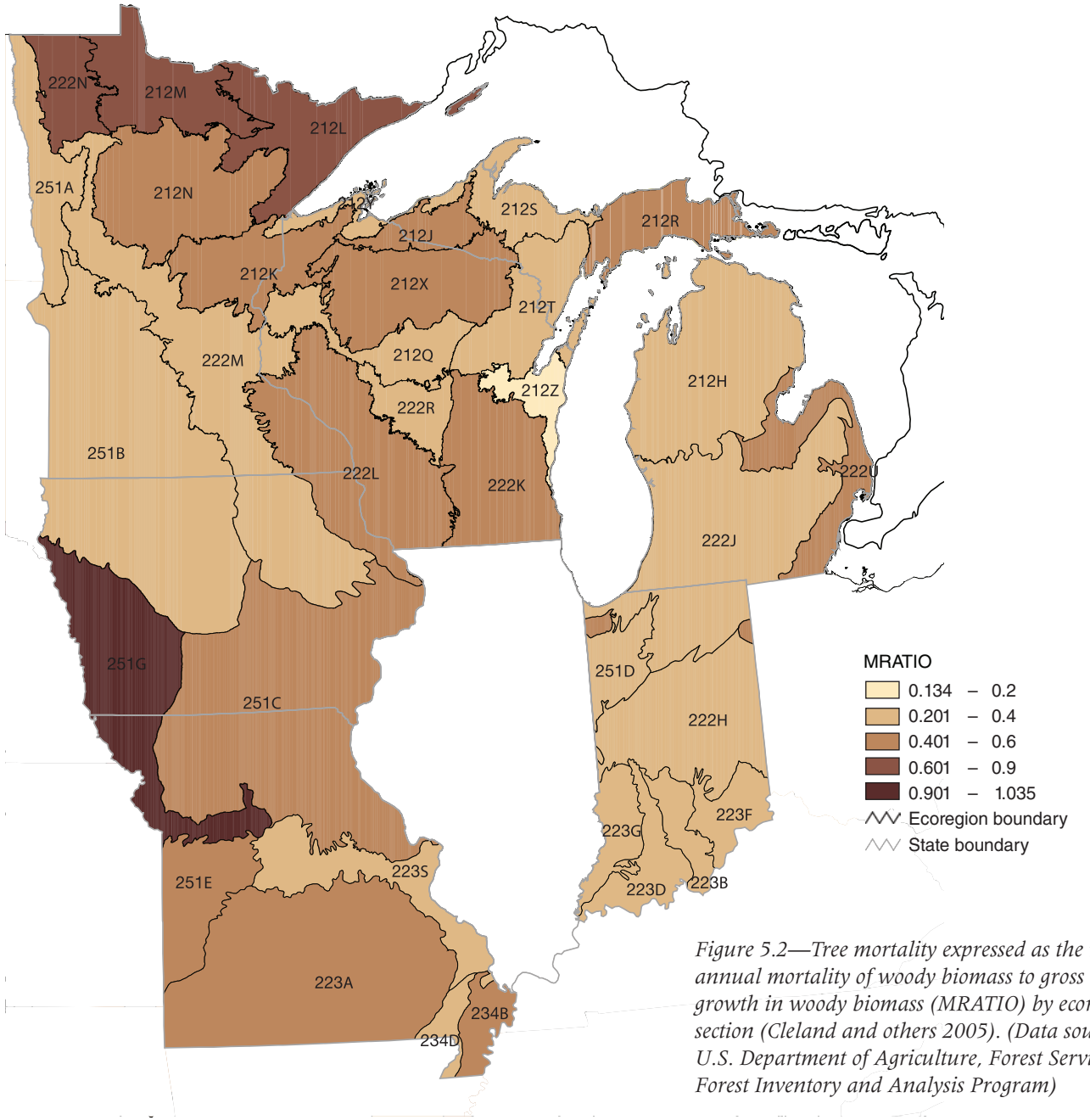


Figure 5.2—Tree mortality expressed as the ratio of annual mortality of woody biomass to gross annual growth in woody biomass (MRATIO) by ecoregion section (Cleland and others 2005). (Data source: U.S. Department of Agriculture, Forest Service Forest Inventory and Analysis Program)

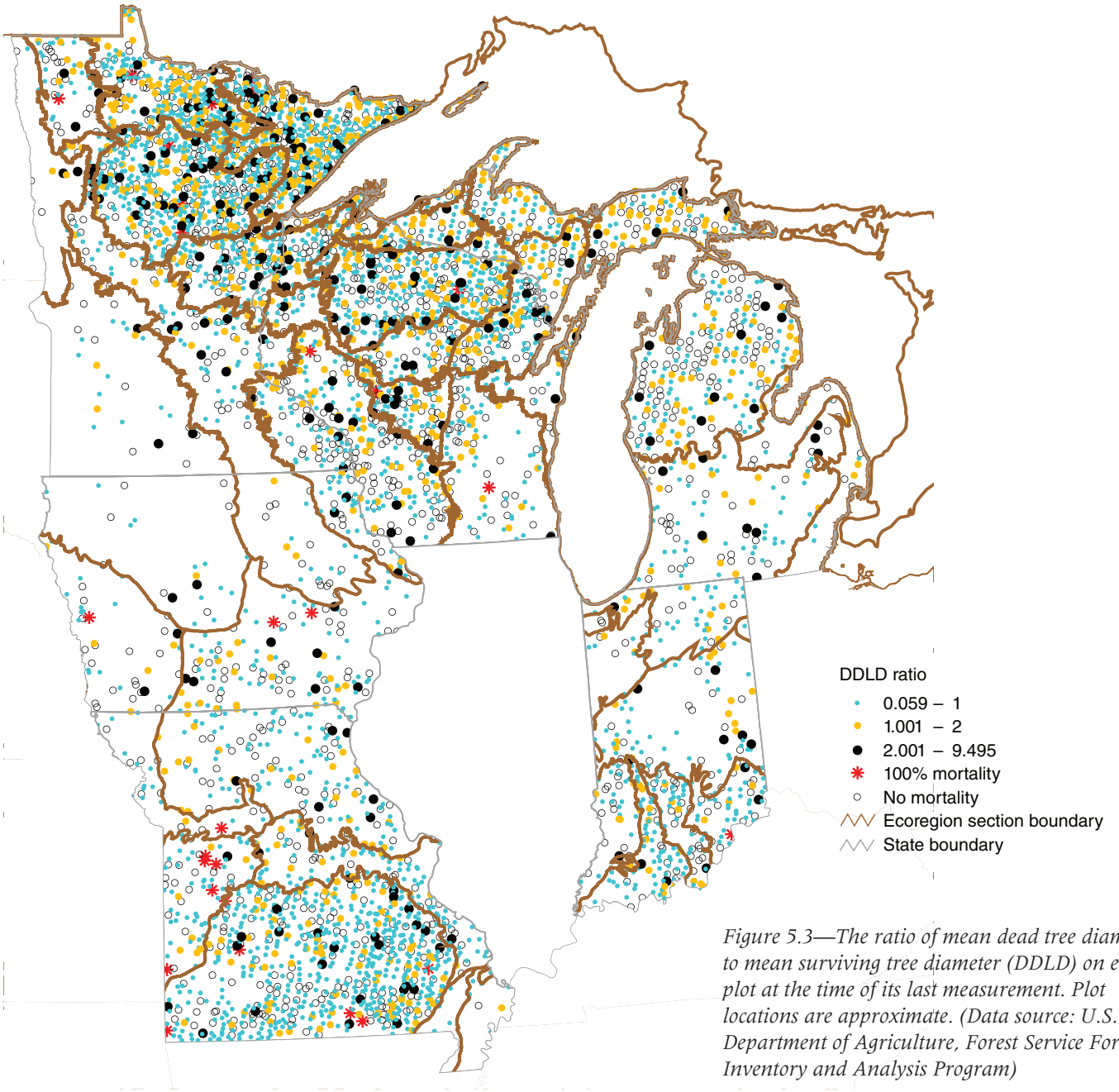


Figure 5.3—The ratio of mean dead tree diameter to mean surviving tree diameter (DDLD) on each plot at the time of its last measurement. Plot locations are approximate. (Data source: U.S. Department of Agriculture, Forest Service Forest Inventory and Analysis Program)

given ecoregion section. Although many plots have high DDLDs, often times the actual level of mortality is very low, as would be the case when remnant larger trees die, leaving young, vigorous stands behind. To focus attention on those plots where mortality was high and where mostly large trees were dying, the proportion of the plot biomass that died over the measurement cycle was calculated for each plot. Figure 5.4 shows the DDLDs for only those plots where more than 30 percent of the biomass died.

In both areas having high MRATIOs, ecoregion section 251G—Missouri Loess Hills, and sections 212L—Northern Superior Uplands, 212M—Northern Minnesota and Ontario, and 222N—Lake Agassiz, Aspen Parklands (fig. 5.2), there were several plots where either all the trees died or the DDLD was very high (> 1.5), while on many other plots, the DDLD was in the intermediate range (0.751 to 1.5), where dying trees were about the same size as survivors. These DDLD values suggest that the observed mortality was generally not all competition induced but rather associated with some acute cause(s), (e.g., insects, pathogens, extreme weather) or stand senescence.

Ecoregion section 251G—Missouri Loess Hills, where the MRATIO was highest, does not contain very much forest (figs. 5.1 and 5.2). In that ecoregion the largest amount of biomass that died was American elm (*Ulmus americana*). About 40 percent of the elms (in terms of both number of stems and biomass) died over the measurement cycle. American elm was also the tree that had died most frequently in adjacent ecoregion sections 251B—North-Central Glaciated Plains and 251C—Central Dissected Till Plains. Elm mortality was likely due to Dutch elm disease.

The other areas of high mortality relative to growth occurred in northern Minnesota, in ecoregion sections 212L—Northern Superior Uplands, 212M—Northern Minnesota and Ontario, and 222N—Lake Agassiz, Aspen Parklands. In those ecoregions, quaking aspen (*Populus tremuloides*) was the species that died most frequently. Aspen was also the species exhibiting highest mortality (in terms of biomass) in nearby areas of Minnesota, Wisconsin, and Michigan [fig. 5.2; ecoregion sections 212H—Northern Lower Peninsula (of Michigan), 212Q—North Central Wisconsin Uplands, 212T—Northern Green Bay Lobe, 212N—Northern Minnesota Drift and Lake Plains, 212K—Western Superior Uplands, 212X—Northern Highlands, 222K—Southwestern Great Lakes Morainal, 222M—Minnesota and Northeast Iowa Morainal-Oak Savannah, 222R—Wisconsin Central Sands, and 251A—Red River Valley].

Recent "Forest Health Highlights for Minnesota and Wisconsin" describe drought (Minnesota Department of Natural Resources, Forest Service, U.S. Department of Agriculture, 2004) and major outbreaks of forest tent caterpillar (Minnesota Department of Natural Resources, Forest Service, U.S. Department of Agriculture, 2002, 2004; Wisconsin Department

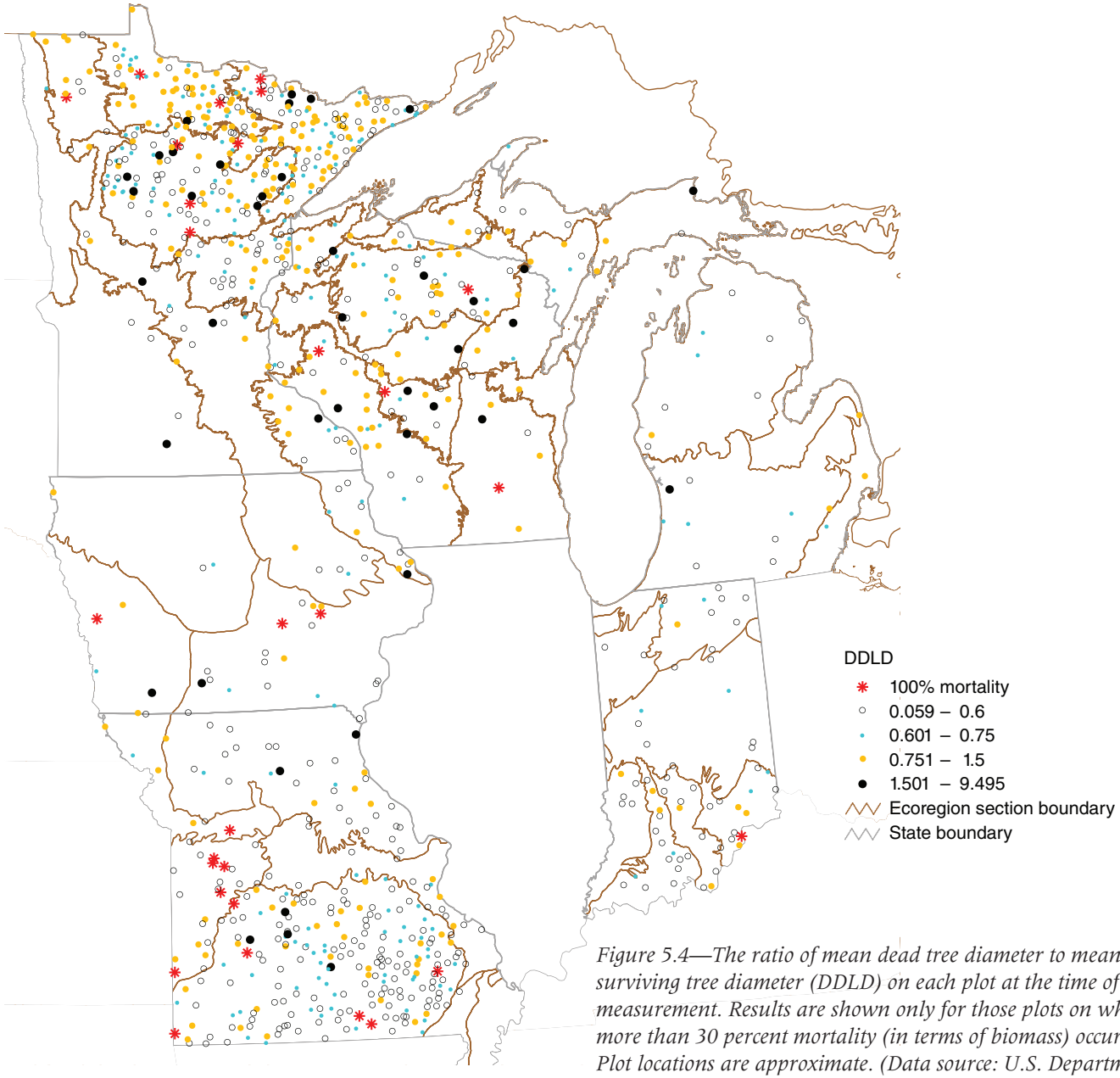


Figure 5.4—The ratio of mean dead tree diameter to mean surviving tree diameter (DDLD) on each plot at the time of its last measurement. Results are shown only for those plots on which more than 30 percent mortality (in terms of biomass) occurred. Plot locations are approximate. (Data source: U.S. Department of Agriculture, Forest Service Forest Inventory and Analysis Program)

of Natural Resources, U.S. Department of Agriculture, 2001, 2002) as causal agents that led to widespread aspen and birch mortality in the region. Gypsy moth also is present in Michigan and eastern Wisconsin (Michigan Department of Natural Resources 2004; Wisconsin Department of Natural Resources, Forest Service, U.S. Department of Agriculture, 2002) and may be contributing to aspen mortality. Mature aspen stands can experience sudden deterioration and heavy mortality over very short-time periods (Frey and others 2004). This phenomenon is not thoroughly understood, but factors that predispose aspen stands to deterioration include climate (Hogg and Hurdle 1995), age (Brandt and others 2003), and stand structure (Mueller-Dombois and others 1983), while inciting factors include drought (Hogg and others 2002) and insect defoliation (Candau and others 2002, Hogg and others 2002).

In future years, as more FIA P2 data are collected, these mortality analyses will be expanded to larger areas of the United States. The MRATIO and DDLD indicators should prove more useful as area of mortality analysis includes a greater variety of ecological regions and forest types.

Literature Cited

Brandt, J.P.; Cerezke, H.F.; Mallet, K.I. [and others]. 2003. Factors affecting trembling aspen (*Populus tremuloides* Michx.) health in Alberta, Saskatchewan, and Manitoba. Canada. Forest Ecology and Management. 178: 287–300.

Candau, J.-N.; Abt, V.; Keatley, L. 2002. Bioclimatic analysis of declining aspen stands in northeastern Ontario. Ontario For. Res. Rep. 154. Sault Ste. Marie, Ontario, Canada: Ontario Ministry of Natural Resources, Ontario Forest Research Institute. 16 p.

Cleland, D.T.; Freeouf, J.A.; Keys, J.E. [and others]. 2005. Ecological subregions: sections and subsections for the conterminous United States. June 14, 2005 version. Washington, DC: Forest Service, U.S. Department of Agriculture. [Map, presentation scale 1: 3,500,000; colored]. [Also on CD–ROM as a Geographic Information System coverage in ArcINFO format].

Coulston, J.W.; Ambrose, M.J.; Riitters, K.H.; Conkling, B.L. 2005a. Forest health monitoring 2002 national technical report. Gen. Tech. Rep. SRS–84. Asheville, NC: Forest Service, U.S. Department of Agriculture, Southern Research Station. 97 p.

Coulston, J.W.; Ambrose, M.J.; Riitters, K.H. [and others]. 2005b. Forest health monitoring 2003 national technical report. Gen. Tech. Rep. SRS–85. Asheville, NC: Forest Service, U.S. Department of Agriculture, Southern Research Station. 97 p.

Coulston, J.W.; Ambrose, M.J.; Riitters, K.H.; Conkling, B.L. 2005c. Forest health monitoring 2004 national technical report. Gen. Tech. Rep. SRS–90. Asheville, NC: Forest Service, U.S. Department of Agriculture, Southern Research Station. 81 p.

Coulston, J.W.; Ambrose, M.J.; Stolte. K.S. [and others]. 2005d. Criterion 3—health and vitality. In: Conkling, B.L.; Coulston, J.W.; Ambrose, M.J., eds. Forest health monitoring 2001 national technical report. Gen. Tech. Rep. SRS–81. Asheville, NC: Forest Service, U.S. Department of Agriculture, Southern Research Station: 39-82.

Frey, B.R.; Lieffers, V.J.; Hogg, E.H.; Landhäusser, S.M. 2004. Predicting landscape patterns of aspen dieback: mechanisms and knowledge gaps. Canadian Journal of Forest Research. 34: 1379–1390.

Gregoire, T.G.; Schabenberger, O.; Barret, J.P. 1995. Linear modeling of irregularly spaced, unbalanced, longitudinal data from permanentplot measurements. Canadian Journal of Forest Research. 25: 137–156.

Hogg, E.H.; Brandt, J.P.; Kochtubajda, B. 2002. Growth and dieback of aspen forest in northwestern Alberta, Canada in relation to climate and insects. Canadian Journal of Forest Research. 32: 823–832.

Hogg, E.H.; Hurdle, P.A. 1995. The aspen parkland in western Canada: a dry-climate analogue for the future boreal forest? Water, Air, and Soil Pollution. 82: 391–400.

McNab, W.H.; Avers, P.E., comp. 1994. Ecological subregions of the United States: section descriptions, WO-WSA-5. Washington, DC: U.S. Department of Agriculture, Forest Service. 267 p.

Michigan Department of Natural Resources. 2004. Michigan 2004 forest health highlights. Lansing, MI: Michigan Department of Natural Resources, Forest, Mineral, and Fire Management Division. 8 p.

Minnesota Department of Natural Resources; Forest Service, U.S. Department of Agriculture. 2002. Minnesota forest health highlights 2002. St. Paul, MN: Minnesota Department of Natural Resources, Division of Forestry; and Forest Service, U.S. Department of Agriculture, Forest Health Protection, Northeastern Area. [Not paged]. http://fhm.fs.fed.us/fhh/fhh-02/mn/mn_02.htm. [Date accessed: August 16, 2007].

Minnesota Department of Natural Resources; Forest Service, U.S. Department of Agriculture. 2004. 2004 Minnesota forest health highlights. St. Paul, MN: Minnesota Department of Natural Resources, Division of Forestry; and Forest Service, U.S. Department of Agriculture, Forest Health Protection, Northeastern Area. 18 p. http://fhm.fs.fed.us/fhh/fhh-04/mn/mn_04.pdf. [Date accessed: August 16, 2007].

Mueller-Dombois, D.; Canfield, J.E.; Holt, R.A.; Buelow, G.P. 1983. Tree-group death in North American forests: a pathological problem or a new problem for vegetation ecology? Phytocoenologia. 11: 117–137.

Smith, W.D.; Conkling, B.L. 2004. Analyzing forest health data. Gen. Tech. Rep. SRS–77. Asheville, NC: Forest Service, U.S. Department of Agriculture, Southern Research Station. 35 p.

Wisconsin Department of Natural Resources; Forest Service, U.S. Department of Agriculture. 2001. Wisconsin forest health highlights 2001. Madison, WI: Wisconsin Department of Natural Resources, Division of Forestry. [Not paged]. http://fhm.fs.fed.us/fhh/fhh-01/wi/wi_01.htm. [Date accessed: August 16, 2007].

Wisconsin Department of Natural Resources; Forest Service, U.S. Department of Agriculture. 2002. Wisconsin forest health highlights 2002. Madison, WI: Wisconsin Department of Natural Resources, Division of Forestry. [Not paged]. http://fhm.fs.fed.us/fhh/fhh-02/wi/wi_02.htm. [Date accessed: August 16, 2007].

Zhu, Z.; Evans, D.L. 1994. U.S. forest types and predicted percent forest cover from AVHRR data. Photogrammetric Engineering and Remote Sensing. 60: 525–531.

Chapter 6. Modeling Ozone Bioindicator Injury with Microscale and Landscape-Scale Explanatory Variables: A Logistic Regression Approach

John W. Coulston

Introduction

Tropospheric ozone occurs at phytotoxic levels in the United States (Lefohn and Pinkerton 1988). Several plant species, including commercially important timber species, are sensitive to elevated ozone levels. Exposure to elevated ozone can cause growth reduction and foliar injury and make trees more susceptible to secondary stressors such as insects and pathogens (Chappelka and Samuelson 1998). In response to this threat, the Forest Service, U.S. Department of Agriculture, maintains a national ozone biomonitoring program.

The goal of ozone biomonitoring is to identify geographic areas where the risk of ozone injury is high and the forest community is sensitive. These areas may then become candidate areas for followup investigation through the evaluation monitoring tier of the National Forest Health Monitoring (FHM) Program of the Forest Service (see the definition of evaluation monitoring in chapter 1). Information about plant injury from ozone is collected at biomonitoring plots by examining bioindicator species. In general, biomonitoring plots are located in relatively open areas within or near to forests, and biomonitoring species are both tree and nontree species (table 6.1). To achieve the goal of the biomonitoring program (to identify geographic areas with sensitive forest communities and high risk of ozone injury), spatial models, e.g., kriging, are used to predict the likelihood of ozone injury (based on biomonitoring data) at plot locations from the Forest Inventory and Analysis (FIA) Program of the Forest Service. The plot data are then used to identify the sensitivity of the forest community. Smith and others (2007) provide guidance on how to select an appropriate spatial interpolation model, but they also note that future research will attempt to improve the precision of the estimates from the spatial models. The objective of this chapter is to identify appropriate ancillary data and the appropriate spatial scale of those data for use in spatial modeling of risk of ozone injury.

Ozone injury to plants is a function of the sensitivity of the plant species, the ambient ozone concentration, and environmental conditions (McCool 1998). Here, we examine the importance of microscale variables recorded on the biomonitoring plots and landscape-scale variables available as Geographic Information System maps for predicting the likelihood of ozone injury to biomonitoring plants. The microscale variables examined are aspect, terrain position, soil depth, soil wetness, and soil drainage. The landscape-scale variables are SUM06 ozone, Palmer Drought Severity Index (PDSI), aspect, available water capacity, terrain relative moisture index, and population density (a more complete description of each variable is presented in the following section) (table 6.2). A logistic regression model is used to identify the significant variables for predicting the likelihood of ozone injury on biomonitoring plots.

Table 6.1—Common and scientific names of bioindicator species

Common name	Scientific name
Blackberry	*Rubus allegheniensis*
Black cherry	*Prunus serotina*
Common and tall milkweed	*Asclepias spp.*
Yellow poplar	*Liriodendron tulipifera*
White ash	*Fraxinus americana*
Sassafras	*Sassafras albidum*
Spreading dogbane	*Apocynum androsaemifolium*
Big leaf aster	*Aster macrophylum*
Sweetgum	*Liquidambar styraciflua*
Pin cherry	*Prunus pensylvanica*
Ponderosa pine	*Pinus ponderosa*
Jeffrey pine	*Pinus jeffreyi*
Blue elderberry	*Sambucus cerulea*
Quaking aspen	*Populus tremuloides*
Scouler's willow	*Salix scouleriana*
Red alder	*Alnus rubra*
Skunk bush	*Rhus trilobata*
Ninebark	*Physocarpus malvaceus*
Mountain snowberry	*Symphoricarpos oreaphilus*
Western wormwood	*Artemesia ludoviciana*
Red elderberry	*Sambucus racemosa*
Huckleberry	*Vaccinium membranaceum*
Evening primrose	*Oenothera elata*
Mugwort	*Artemesia douglasiana*
California black oak	*Quercus kellogii*
Pacific ninebark	*Physiocarpus capitatus*

Data

The data used for model development were acquired from several sources. The FIA program collects information on bioindicator plant injury at biomonitoring sites (fig. 6.1). At each ozone biomonitoring plot, the amount and severity of ozone injury on bioindicator species was collected. Bioindicator species include ozone-sensitive species such as black cherry (*Prunus serotina*) in the Eastern United States and Scouler's willow (*Salix scouleriana*) in the Western United States (table 6.1). We used the biomonitoring plot data as the binary response variable in our model (0 = no injury detected, 1 = injury detected) for each year 2003 through 2005. We also used microscale variables collected on biomonitoring sites during the 2003 through 2005 field seasons. Landscape-scale variables were obtained from the U.S. Environmental Protection Agency, U.S. Geological Survey, and other sources (table 6.2).

Table 6.2—Potential explanatory variables used in logistic regression

Explanatory variable	Type	Categories	Variable name	Spatial scale	Reference
Microscale					
Aspect	Continuous		AspP[a]	Plot level	U.S. Department of Agriculture Forest Service 2002[b]
Terrain position	Categorical		TerrPos	Plot level	U.S. Department of Agriculture Forest Service 2002[b]
	1	Ridgetop or upperslope			
	2	Bench or level area along slope			
	3	Lower slope			
	4	Flat and unrelated to slope			
	5	Bottomland with occasional flooding			
Soil depth	Binary		SoilDpt	Plot level	U.S. Department of Agriculture Forest Service 2002[b]
	1	Bedrock is not exposed			U.S. Department of Agriculture Forest Service 2002[b]
	2	Bedrock is exposed. Soil generally shallow			
Soil drainage (Eastern United States)	Categorical		SoilDrn	Plot level	U.S. Department of Agriculture Forest Service 2002[b]
	1	Well drained			
	2	Generally wet			
	3	Excessively dry			
Soil wetness (Western United States)	Categorical		SoilWt	Plot level	U.S. Department of Agriculture Forest Service 2002[b]
	1	Wet			
	2	Moderately dry			
	3	Very dry			
Landscape scale					
Ambient ozone	Continuous		SUM06[c]	5-Km raster cells	U.S. Environmental Protection Agency 2004
Available water capacity	Continuous		Awc	1: 250,000	Miller and White 1998
Palmer drought severity	Continuous		PDSI[d]	U.S. Climate Division	National Climate Data Center 1994
Aspect	Continuous		AspG[a]	3 arc-second raster cells	U.S. Geological Survey 1993
Terrain relative moisture index	Continuous		TRMI[e]	3 arc-second raster cells	U.S. Geological Survey 1993
Population density	Continuous		Pden	U.S. counties	U.S. Census Bureau 2004

[a] Aspect was rescaled to a continuous variable denoting northernness scaled 0 to 2 by [cos(aspect)+1] to account for the fact that, for example, aspects of 15° and 345° have the same northernness.
[b] Forest Service, U.S. Department of Agriculture, 2002. Forest inventory and analysis national core field guide: field data collection procedures for phase 3 plots. Version 1.7. Internal report. Vol. 2. [Not paged]. On file with: Forest Service, U.S. Department of Agriculture, Forest Inventory and Analysis, Rosslyn Plaza, 1620 North Kent Street, Arlington, VA 22209.
[c] For each U.S. Environmental Protection Agency ozone monitoring station the sum of all hourly ozone concentration > 0.06 parts per million (ppm) were summarized from 8 a.m. to 8 p.m. for June, July, and August for 2003, 2004, and 2005. SUM06 (ppm-hours) ozone values were assigned to each biomonitoring plot by inverse distance weighting interpolation.
[d] Average June, July, and August Palmer Drought Severity Index (PDSI) was calculated for each climate division in the coterminous United States for each year (2003, 2004, 2005). PDSI is scaled from ~-7 to 7 where negative values indicate drought stress.
[e] TRMI is generated using a digital elevation model. The algorithm identifies topographic position, e.g., ridgetops and valley bottoms, to assign a moisture index scale from 0 (dry) to 60 (wet).

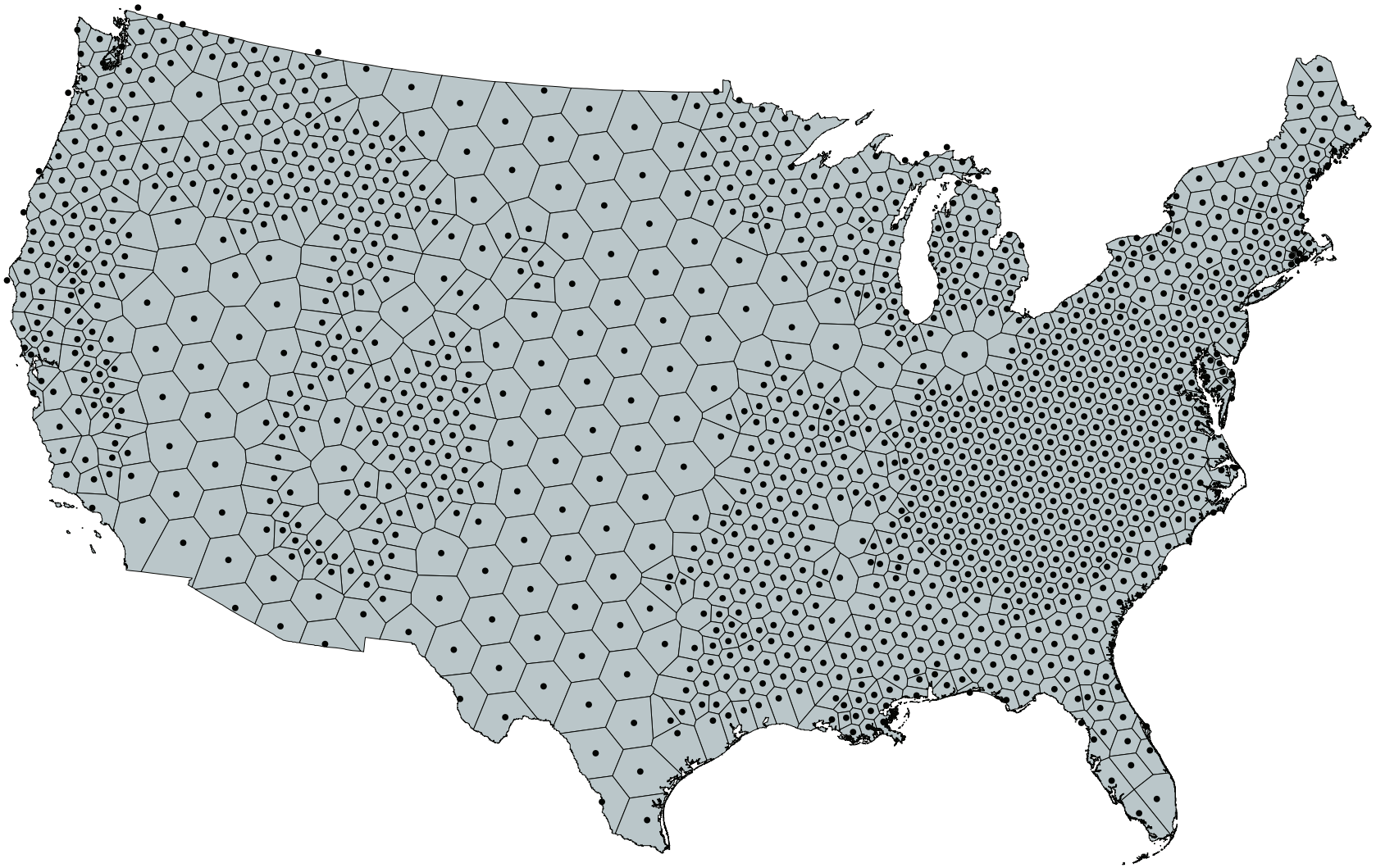


Figure 6.1—Ozone biomonitoring sampling grid for the coterminous United States. The points represent the approximate center of each sampling polygon.

Methods

Logistic regression (SAS 2004) was used to examine the relationship among biosite ozone injury, microscale variables, and landscape-scale variables. Because of different sampling intensities, each biosite was weighted by the area it represents (fig. 6.1). The general form of the linear logistic model is:

$$Log(\pi / (1 - \pi) = a + B`x$$

where

Log = natural logarithm of $(\cdot)$

π = the probability that the response equals 1 (ozone injury recorded) given the vector of explanatory variables x

a = intercept

B = vector of parameter estimates

The explanatory variables in x can be binary, categorical, ordinal, or continuous; and interactions among variables can also be examined. Because of potential regional differences in environmental conditions and ambient ozone concentrations, each FIA region [North, South, Interior West, and Pacific Northwest (which includes California)] was examined independently. Also, the North and South FIA regions collect information on soil drainage on biomonitoring plots while the Interior West and Pacific Northwest FIA regions collect information on soil wetness (table 6.2). Specific interactions were selected to examine the potential relationship among ozone injury, ambient ozone concentrations, terrain position, and moisture based on landscape and plot-level variables. In the East (North and South FIA regions), the full set of explanatory variables tested is denoted x_e and in the West (Interior West and Pacific Northwest FIA regions) the full set of explanatory variables tested is denoted x_w

where

$$x_e = \begin{bmatrix} AspP \\ TerrPos \\ SoilDpt \\ SoilDrn \\ SUM06 \\ Awc \\ PDSI \\ AspG \\ TRMI \\ Pden \\ SUM06 * TerrPos \\ SUM06 * SoilDrn \\ SUM06 * PDSI \\ SUM06 * TRMI \end{bmatrix} \text{ and } x_w = \begin{bmatrix} AspP \\ TerrPos \\ SoilDpt \\ SoilWt \\ SUM06 \\ Awc \\ PDSI \\ AspG \\ TRMI \\ Pden \\ SUM06 * TerrPos \\ SUM06 * SoilWt \\ SUM06 * PDSI \\ SUM06 * TRMI \end{bmatrix}$$

The null hypothesis for the full model (H_o: $B = 0$) was tested and each variable was examined. Variables that were not significant at the $p = 0.10$ level were removed from the models. The final model selected for each region was the model where $B \neq 0$, each variable was significant at $p < 0.10$, and the minimum Akaike Information Criterion value was minimized (SAS Institute 2004). The generalized coefficient of determination (pseudo-R^2) was used to examine the predictive power of each final model.

Results

From 2003 through 2005, ozone injury to bioindicator plants was recorded in every FIA region except for the Interior West (fig. 6.2). Injury occurrence was relatively constant in the North and Pacific Northwest FIA regions (2003 through 2005). However, in the South FIA region, the number of biomonitoring plots with injury tended to decrease even though the total number of plots examined increased between 2003 and 2005 (fig. 6.2). The Interior West FIA region was not examined using the logistic regression approach because of the lack of recorded ozone injury, but ambient ozone concentrations from 2003 through 2005 in the Interior West were comparable to the other FIA regions (fig. 6.3).

Statistically significant logistic regression models were developed for the North, Pacific Northwest, and South FIA regions (table 6.3).

The models for the three regions all contained SUM06 as a significant explanatory variable. Most of the interaction terms examined were not statistically significant. However, in the South FIA region, the SUM06*PDSI interaction was significant (table 6.4). TerrPos was the only microscale variable selected for the final model and only was used in the North FIA

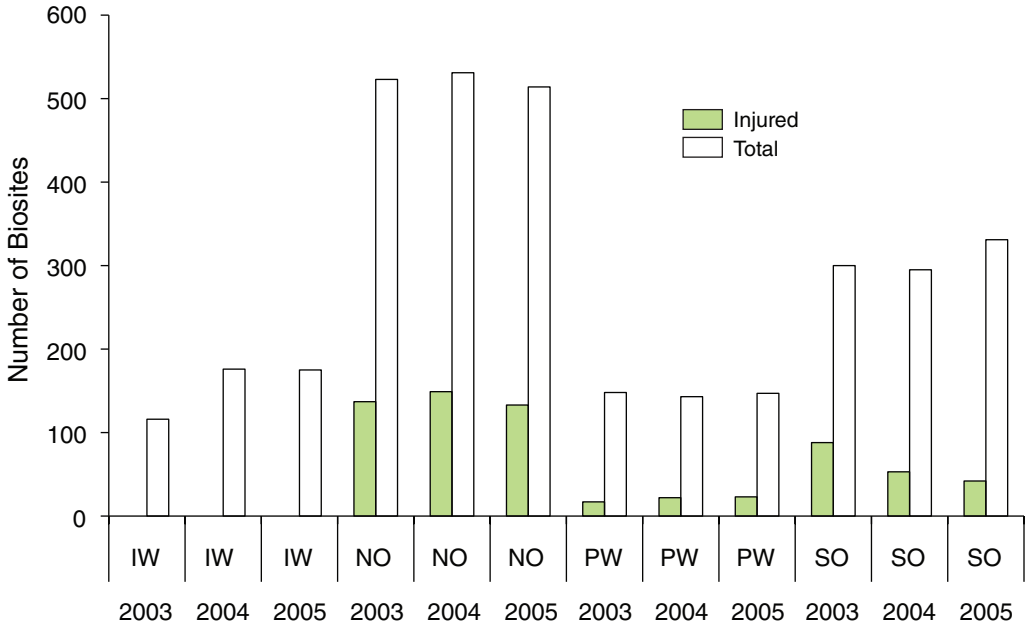


Figure 6.2—The number of total ozone biosites measured and the number of ozone biosites with injury, recorded by Forest Inventory and Analysis region (IW=Interior West, NO=North, PW=Pacific Northwest, SO=South) and year. (Data source: U.S. Department of Agriculture, Forest Service Forest Inventory and Analysis Program)

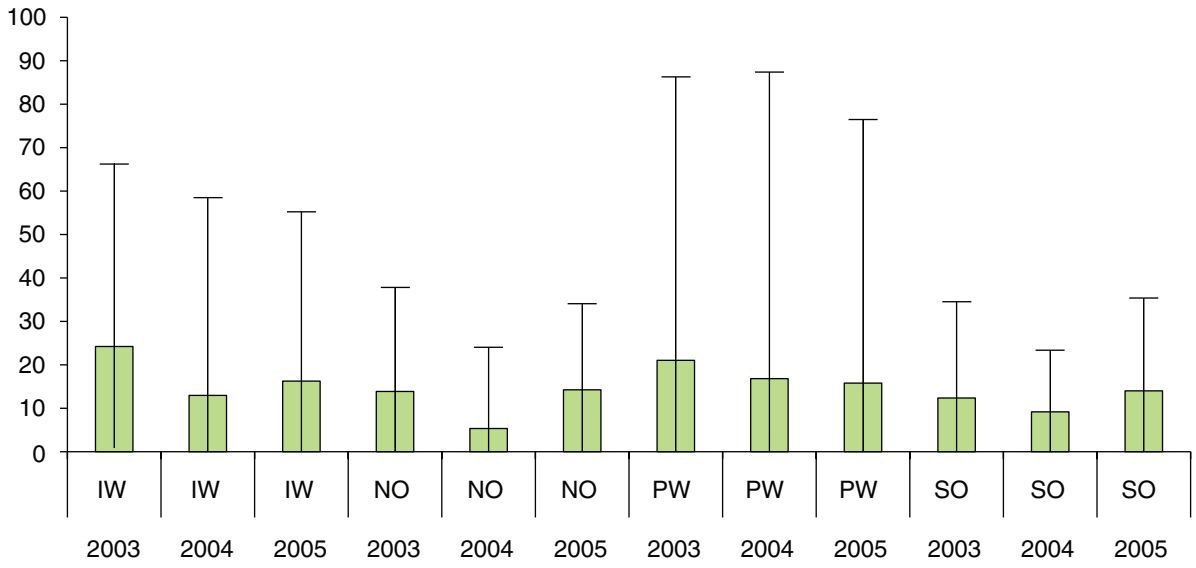


Figure 6.3—Average area weighted SUM06 ozone exposure for each Forest Inventory and Analysis region (IW=Interior West, NO=North, PW=Pacific Northwest, SO=South) and year. Error bars represent the minimum and maximum observed SUM06 value. (Data source: U.S. Environmental Protection Agency)

Table 6.3—Results from logistic regression model for each Forest Inventory and Analysis region

Region	Explanatory variables (x)	Chi-square	p-value	R^2	Percent of concordant pairs
					percent
North	SUM06, TerrPos, Pden	103.31	0.0001	0.0638	66.6
Pacific Northwest	PDSI, SUM06, Pden	94.96	0.0001	0.1949	84.1
South	PDSI, PDSI*SUM06	89.78	0.0001	0.0924	73.7

region model. Terrain position 1 (ridgetop or upperslope) and terrain position 4 (flatland unrelated to slope) had lower p-values (table 6.4) than the other terrain position categories. The best model for predicting the probability of ozone injury to bioindicator plants, based on R^2 and the percent of concordant pairs, was the Pacific Northwest model. The models for the South and North FIA regions had R^2 of 0.092 and 0.064, respectively.

Table 6.4—Significance of explanatory variables for each logistic regression model and Forest Inventory and Analysis region

Region	Explanatory variables (x)	Parameter estimate (B)	p-value
North	Intercept (a)	–5.55	0.0001
	SUM06	0.024	0.0033
	TerrPos		0.0032
	1	0.241	0.09
	2	–0.234	0.2404
	3	0.201	0.2519
	4	0.421	0.0002
	Pden	0.359	0.0001
Pacific Northwest	Intercept (a)	–6.326	0.0001
	PDSI	0.124	0.329
	SUM06	0.053	0.001
	Pden	0.292	0.0044
South	Intercept (a)	–1.79	0.0001
	PDSI	–0.325	0.0016
	PDSI*SUM06	0.045	0.0001

Discussion

Ozone injury to plants is related to the ambient ozone concentration, plant species' sensitivity to ozone, soil moisture, and light, all of which influence ozone uptake by plants. The purpose of this analysis was to examine the relationship between ozone injury to bioindicator plants and microscale and landscape-scale explanatory variables. Generally, only landscape-scale explanatory variables were selected for the final model for each FIA region with terrain position as the exception in the North FIA region. However, the terrain relative moisture index landscape-scale variable can be used in place of the terrain position variable with minimal impact to the predictive power of the model. Smith and others (2007) suggested using either inverse distance weighting interpolation or kriging to predict the likelihood of ozone injury to bioindicator plants at unmeasured locations but also encouraged the development of spatially explicit models that include explanatory variables such as ambient ozone concentrations and moisture conditions. The results presented here indicate that landscape-scale variables were statistically significant most frequently, i.e., explained significant amounts of the variation in ozone injury to bioindicator plants.

The significance of SUM06 ozone and PDSI from this analysis corresponds with results from Smith and others (2003) and Davis and Orendovici (2006). Davis and Orendovici (2006) found a statistically significant relationship between the incidences of ozone symptoms on vegetation in the Edwin B. Forsythe National Wildlife Refuge (New Jersey) and plant species, PDSI, and the interaction of two different ambient ozone statistics. Smith and others (2003) found that SUM06 ozone and PDSI were significant explanatory variables in a linear regression to predict a composite ozone bioindicator variable in the North FIA region. The statistical significance of county-level population density was not tested in the studies described above. In fact, population density is not a causal mechanism of ozone-induced foliar injury, and the correlation between population density and SUM06 ozone was < 0.23 in the North, South, and Pacific Northwest FIA regions. However, population density may serve as a surrogate for other explanatory variables not included in this analysis.

The results presented in this chapter identify key landscape-level variables that account for statistically significant amounts of the variance of ozone-induced plant injury from ozone. While statistically significant logistic models were developed for each FIA region where ozone injury occurred on biomonitoring sites, logistic regression provided little improvement, in respect to predictive power, over standard spatial interpolation techniques such as kriging and inverse distance weighting. However, the results from our analysis provide direction for future research:

1. Future modeling efforts should focus on using landscape-scale variables rather than microscale (plot-level) variables.
2. Predictive models vary regionally and perhaps subregionally. Future modeling efforts should examine the importance of using subregional areas for model development.
3. The SUM06 index was used to represent ambient ozone exposure. Based on suggestions from Davis and Orendovici (2006), other ambient ozone indices such as N100 (number of hours that ambient ozone is ≥ 100 parts per billion) may be more appropriate explanatory variables.
4. PDSI is derived by climate division, which may be too coarse for this kind of modeling. Other variables such as the ratio of precipitation to evapotranspiration should be examined as a potential fine-scale surrogate for PDSI.

Literature Cited

Chappelka, A.H.; Samuelson, L.J. 1998. Ambient ozone effects on forest trees of the Eastern United States: a review. New Phytologist. 139: 91–108.

Davis, D.D.; Orendovici, T. 2006. Incidence of ozone symptoms on vegetation within a national wildlife refuge in New Jersey, USA. Environmental Pollution. 143: 555–564.

Lefohn, A.S.; Pinkerton, J.E. 1988. High resolution characterization of ozone data for sites located in forested areas of the United States. Journal of the Air Pollution Control Association. 38: 1504–1511.

McCool, P.M. 1998. Introduction. In: Flager, R.B., ed. Recognition of air pollution injury to vegetation: a pictorial atlas. Pittsburgh: Air and Waste Management Association: 1.1–1.2.

Miller, D.A.; White, R.A. 1998. A conterminous United States multi-layer soil characteristics data set for regional climate and hydrology modeling. Earth Interactions. 2: 1–15.

National Climate Data Center. 1994. Time bias corrected divisional temperature-precipitation-drought index. Documentation for dataset TD–9640. 12 p. On file with: Database Management Branch, National Climate Data Center, National Oceanic and Atmospheric Administration, Federal Building, 37 Battery Park Avenue, Asheville, NC 28801–2733.

SAS Institute Inc. 2004. SAS/STAT 9.1 user's guide. Cary, NC. SAS Institute Inc. 5,121 p.

Smith, G.; Coulston, J.; Jepsen, J.; Prichard, T. 2003. A national ozone biomonitoring program: results from field surveys of ozone sensitive plants in northeastern forest (1994–2000). Environmental Monitoring and Assessment. 87: 271–291.

Smith, G.C.; Smith, W.D.; Coulston, J.W. 2007. Ozone biomonitoring sampling and estimation. Gen. Tech. Rep. NRS–20. Newtown Square, PA: Forest Service, U.S. Department of Agriculture, Northern Research Station. 34 p.

U.S. Census Bureau. 2004. U.S. census database, 2000. Reston, VA: U.S. Geological Survey. http://www.census.gov/geo/www/gazetteer/places2k.html,. [Date accessed: June 6, 2007].

U.S. Environmental Protection Agency. 2004. Air quality system raw data summarization formulas. Draft version 1.1. http://www.epa.gov/ttn/airs/airsaqs/manuals/Precision%20and%20Accuracy%20Summary%20Formulas.pdf. [Date accessed: November 13, 2006].

U.S. Geological Survey. 1993. Digital elevation models—data users guide 5. Reston, VA: 48.

Chapter 7. Emerging Forest Pest Threat: Redbay Ambrosia Beetle and Laurel Wilt

Frank H. Koch

William D. Smith

Introduction

Accidentally introduced from Asia, the redbay ambrosia beetle (*Xyleborus glabratus*) carries a symbiotic fungus (*Raffaelea lauricola*) that has been linked to mortality of trees and shrubs from the Lauraceae family (Fraedrich and others 2007, Harrington and others 2008). The disease caused by the fungus, subsequently named laurel wilt, has severely impacted redbay (*Persea borbonia*) in coastal areas of Florida, Georgia, and South Carolina and has also been linked to mortality of sassafras (*Sassafras albidum*) in this region. As a recently discovered invader, little is known about the biology and behavior of the beetle or its associated fungus. A number of studies are ongoing, including research in Florida related to potential control via fungicides (Mayfield 2007). To provide a basic foundation for additional research or management decisionmaking, the objectives of this chapter are to (1) provide information on the current range of the redbay ambrosia beetle and laurel wilt in the United States, (2) describe the relative distributions of known and potential host species, and (3) employ spatial models to assess climatic factors affecting the beetle's range expansion and to examine the spatial pattern of spread. Further details regarding interpolated host maps as well as the climate and spread analyses are provided in Koch and Smith (2008).

History of the Problem

In 2003, redbay trees in the Hilton Head Island, SC, area began exhibiting high levels of mortality. By the following year, an estimated 75 to 80 percent of redbay trees on the island had been killed, with the exact cause or causes unknown at the time (Fraedrich and others 2007). The species was not known to have any significant pests (Coder 2006b), although insects and precipitation trends during the previous several years, i.e., periods of drought followed by elevated precipitation, or the interaction of the two, were suggested as possible explanations (Fraedrich and others 2007). The mortality on Hilton Head was ultimately linked to the recently introduced redbay ambrosia beetle, which had first been detected in 2002 using a U.S. Department of Agriculture early detection and monitoring trap in Port Wentworth, GA (Fraedrich and others 2007, Rabaglia 2003). More specifically, the redbay trees were killed by the laurel wilt fungus associated with the

beetle (Fraedrich and others 2007). During the past few years, redbay mortality attributed to the beetle and its associated fungus has been reported extensively in coastal areas of Georgia, South Carolina, and Florida. Anecdotal estimates of the rate of spread of the beetle have been reported to range from 15 to 17.5 miles per year in Georgia.[1] However, natural spread appears to be compounded by accidental human transport, as suggested by detection of the beetle in Indian River County, FL, in 2006, more than 130 miles from the closest other county known to be infested at the time. Human-aided transport, possibly via wood chips and firewood, may also have occurred in at least two Georgia counties (see footnote 1), albeit at shorter distances than the apparent jump in Florida.

The redbay ambrosia beetle is one of 10 ambrosia beetle species first detected in the United States between 1985 and 2005 (Haack 2006). Like many other wood-boring beetles, the redbay ambrosia beetle was likely introduced to the country via solid wood packing materials (Fraedrich and others 2007, Haack 2006). Most ambrosia beetles, including the redbay ambrosia beetle, are members of the tribe Xyleborina; species in this tribe are typically difficult to detect, infest a broad suite of host species, and exhibit polygamous, sib-mating behavior, facilitating their establishment in newly invaded areas (Rabaglia and others 2006). The term ambrosia generally refers to symbiotic fungi typically carried by female ambrosia beetles in specialized storage structures (mycangia). These fungi are introduced into host trees or shrubs when the beetles bore galleries into the xylem of the hosts (Furniss and Carolin 1977). Both adult ambrosia beetles and larvae feed on the introduced fungi, rather than on the wood of the trees they infest (Mayfield and Thomas 2006). The laurel wilt fungus was probably introduced from Asia along with the redbay ambrosia beetle, and so far the beetle is the only known vector (Fraedrich and others 2007).

The redbay ambrosia beetle has been documented in India (the States of Assam and West Bengal), Bangladesh, Myanmar, Taiwan, and southern Japan, including the Bonin Islands (Holistic Insect Systematics Laboratory 2005, Rabaglia and others 2006). In its native range, the beetle is typically associated with host plant species from the Lauraceae family, such as Asian spicebush (*Lindera latifolia*) and yellow litsea (*Litsea elongata*) (Rabaglia and others 2006). The beetle is considered of minor importance in Asia, primarily infesting weakened trees as part of a complex of insects (Rabaglia 2003). Nonetheless, it seems to follow the pattern of several other

[1] Cameron, R.S.; Bates, C.; Johnson, J. 2008. Distribution and spread of laurel wilt disease in Georgia: 2006–2008 survey and field observations. 29 p. Unpublished report. Georgia Forestry Commission. [Location unknown]. http://www.fs.fed.us/r8/foresthealth/laurelwilt/resources/pubs/georgia_laurel_wilt_report_2006-08.pdf. [Date accessed: October 3, 2008.]

nonnative ambrosia beetle species that, upon introduction in the United States, have begun to attack ostensibly healthy hosts (Mayfield and Thomas 2006). In addition to its documented impact on redbay and sassafras, the laurel wilt fungus has been found on two imperiled shrub species from the Lauraceae family: pondberry (*Lindera melissifolia*), which is on the Federal endangered species list, and pondspice (*Litsea aestivalis*), which is listed as endangered in Florida and Maryland and threatened in Georgia (Fraedrich 2007; U.S. Department of Agriculture, Natural Resources Conservation Service 2007). Laboratory tests of pathogenicity are underway for a number of tree and shrub species; Lauraceae species such as swamp bay (*Persea palustris*), northern spicebush (*Lindera benzoin*), and California laurel (*Umbellularia californica*) have proven to be susceptible to the fungus (Fraedrich 2007, Forest Service, U.S. Department of Agriculture, 2007). The only commercially important Lauraceae species growing in the United States is the avocado (*Persea americana*), which appears to be susceptible to laurel wilt but may be more resistant than other species (Forest Service, U.S. Department of Agriculture, 2007). Redbay and sassafras, while not commercially important, serve as key food sources for wildlife (Coder 2006a, Griggs 1990), and redbay is often an understory dominant in barrier island forests of the Southeast (Helm and others 1991).

Methods

Current Beetle/Wilt Distribution—Forest health specialists in Florida, Georgia, and South Carolina provided data, from 2004 to 2007, indicating which counties in each State contain redbay trees with symptoms of laurel wilt and/or confirmed occurrence of the redbay ambrosia beetle. In the resulting map (fig. 7.1), affected counties have been labeled according to the year of first-confirmed detection. Additionally, in 2006–07, the Georgia and South Carolina Forestry Commissions completed surveys of redbay mortality in their coastal counties. The States followed similar protocols in their surveys, with 1/10-acre linear plots placed according to a systematic grid. In South Carolina, survey plots were established on a 10-minute longitude by 10-minute latitude grid across the range of redbay. This plot network was intensified to 5-minute longitude by 5-minute latitude in areas around the perceived edge of the infested zone.[2] In Georgia, survey plots were established on a 16.1 km by 16.1 km (10-mile by 10-mile) grid for most areas, including counties where the beetle and wilt had not been detected at the time, and on an 8-km by 8-km (5-mile by 5-mile) grid for areas with

[2] Boone, A.J. 2007. A survey of redbay mortality in South Carolina. 17 p. Unpublished report to the South Carolina Forestry Commission. On file with: U.S. Forest Service National Forest Health Monitoring Research Team, 3041 Cornwallis Road, Research Triangle Park, NC 27709.

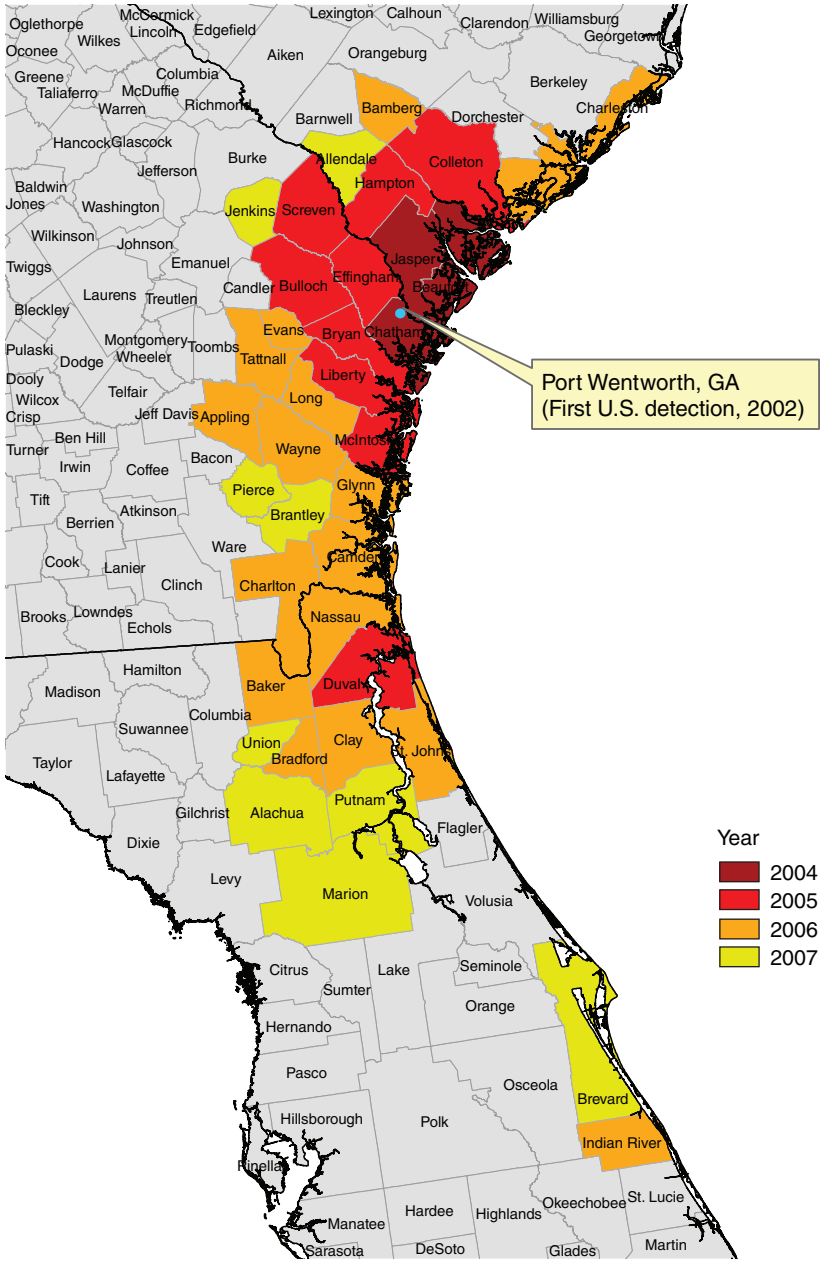


Figure 7.1—County-level distribution of redbay (Persea borbonia) *mortality associated with laurel wilt* (Raffaelea lauricola) *and the redbay ambrosia beetle* (Xyleborus glabratus)*, by year of first detection. (Detection data sources: James Johnson, Georgia Forestry Commission; Bud Mayfield, U.S Forest Service, formerly Florida Department of Agriculture and Consumer Services, Division of Forestry; Laurie Reid, South Carolina Forestry Commission)*

localized infestations.[3] All redbay trees in each plot were examined to derive total percent mortality, percent flagging, and percent healthy estimates for the plot (see footnotes 3 and 2). When laurel wilt was suspected to be present on a plot outside the confirmed infestation zone, affected trees were inspected for characteristic black staining of the xylem tissue, or instead, tissue samples were sent to Forest Service pathologists for verification (see footnotes 3 and 2). The approximate plot locations in both States, labeled according to each plot's percent redbay mortality estimate, were combined into a single map (fig. 7.2) in order to depict the current spatial pattern and extent of infestation. Field personnel in both States also attempted to record sassafras mortality but noted the difficulty of doing so during the dormant season when the surveys were performed, so these data were not included in the map analysis.

Host Species Distributions—Because it has been shown that the redbay ambrosia beetle will attack several different species from the Lauraceae family, county-level distributions of all Lauraceae species found in the conterminous United States (table 7.1), whether native or nonnative, were mapped using the Floristic Synthesis of North America and PLANTS phytogeographic databases (Kartesz 2003; U.S. Department of Agriculture, Natural Resources Conservation Service 2007). The species distribution data in these databases are based on voucher specimens, published reports, or other documentation of species occurrence. National distribution maps for each unique Lauraceae species or variety were combined into a single map depicting the diversity of potential host species per county (fig. 7.3). All Lauraceae species found in the United States are associated with forested ecosystems, so a map of forest cover developed from MODIS satellite imagery by the Forest Service Remote Sensing Applications Center was used to mask nonforest areas from the county-level map.

Maps of trees per acre for the two most prominent host species, redbay and sassafras, were generated through ordinary kriging (see appendix 7.1) of Forest Service FIA phase 2 plot data. For both species, separate interpolations were performed for each U.S. ecoregion section (Cleland and others 2007) containing plots in which the species occurred. Interpolated values for unknown locations were based on the 30 closest neighboring plots or, if the

[3] Beck, M.J. 2007. Georgia redbay survey summary. 3 p. Unpublished report to the Georgia Forestry Commission. On file with: U.S. Forest Service National Forest Health Monitoring Research Team, 3041 Cornwallis Road, Research Triangle Park, NC 27709.

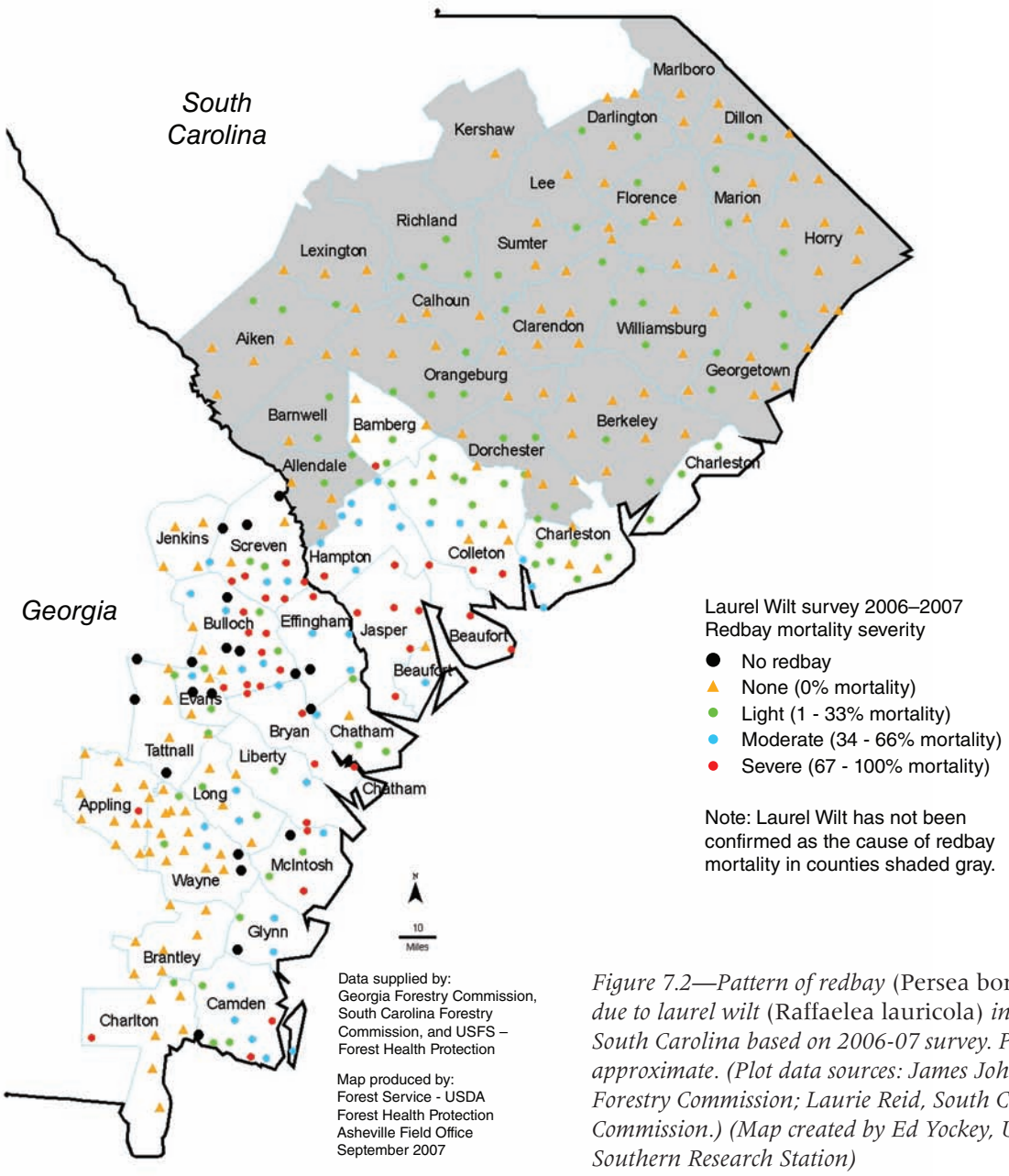


Figure 7.2—Pattern of redbay (Persea borbonia) *mortality due to laurel wilt* (Raffaelea lauricola) *in Georgia and South Carolina based on 2006-07 survey. Plot locations are approximate. (Plot data sources: James Johnson, Georgia Forestry Commission; Laurie Reid, South Carolina Forestry Commission.) (Map created by Ed Yockey, U.S. Forest Service, Southern Research Station)*

Table 7.1—Plant species from the Lauraceae family found within the conterminous United States

Species	Common name(s)	Growth form	Native?	Distribution
Cassytha filiformis	Love vine	Vine	Yes	FL, TX
Cinnamomum camphora	Camphor tree	Tree	No	CA and Southeastern United States
Laurus nobilis	Bay laurel; sweet bay	Tree	No	CA
Licaria triandra	Pepperleaf sweetwood	Tree	Yes	Rare; limited to one FL county
Lindera benzoin	Northern spicebush	Shrub; tree	Yes	Common throughout Eastern United States
Lindera melissifolia	Southern spicebush; pondberry	Shrub; tree	Yes	Rare; endangered (five Southeastern States)
Lindera subcoriacea	Bog spicebush	Shrub; tree	Yes	Rare; endangered (FL, NC)
Litsea aestivalis	Pondspice	Shrub	Yes	Rare; endangered (FL, MD) / threatened (GA)
Nectandra coriacea	Lancewood	Shrub; tree	Yes	FL
Persea americana	Avocado	Tree	No	FL (commercially grown in southern part of state and in CA)
Persea borbonia	Redbay	Shrub; tree	Yes	Southeastern United States
Persea humilis	Silk bay	Shrub; tree	Yes	Rare; endemic to FL
Persea palustris	Swamp bay	Shrub; tree	Yes	Southeastern United States, distribution similar to redbay
Sassafras albidum	Sassafras	Shrub; tree	Yes	Common throughout Eastern United States
Umbellularia californica var. *californica* var. *fresnensis*	California bay laurel	Shrub; tree	Yes	var. *californica*: CA, OR, WA; var. *fresnensis*: limited to Fresno County, CA

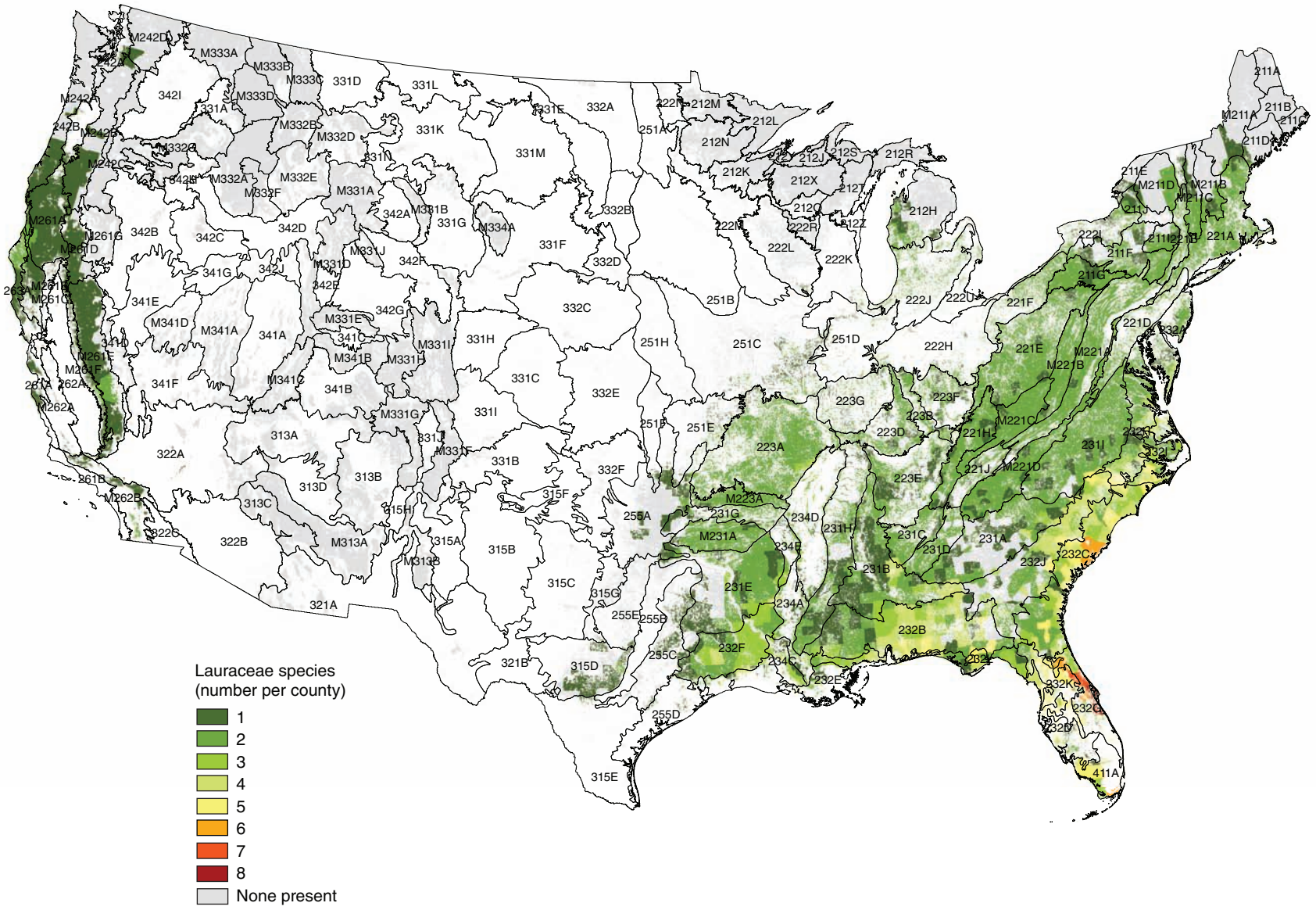


Figure 7.3—Number of species from the Lauraceae family occurring in U.S. counties. Distribution data: Synthesis of the North American Flora (Kartesz 2003); U.S. Department of Agriculture PLANTS Database (U.S. Department of Agriculture, Natural Resources Conservation Service 2007). Ecoregion section boundaries (Cleland and others 2007) are shown for reference. The source for background forest cover, used as a mask, was the U.S. Forest Service, Remote Sensing Applications Center.

number of plots within a 60-km radius of an unknown location was < 30, then all plots within this distance threshold were used. The resulting kriged surfaces for each section, with a 1-km^2 spatial resolution, were merged into comprehensive trees-per-acre maps for each species (figs. 7.4 and 7.5), which were masked using the MODIS forest-cover map developed by the Remote Sensing Applications Center.

Climate Matching—A comparison of climatic conditions between the conterminous United States and the other countries where the redbay ambrosia beetle is known to occur was performed using NCSU/APHIS Plant Pest Forecast (NAPPFAST), a Web-based software application for modeling the potential distributions of pest species (Magarey and others 2007; U.S. Department of Agriculture, Animal and Plant Health Inspection Service 2007). NAPPFAST employs global datasets for a large suite of climatic and environmental variables and includes a module for performing broad-scale climate matching. Based on a user-selected subset of input variables and a set of defined locations representing a pest's known geographic range, NAPPFAST generates gridded maps (approximately 10-km^2 spatial resolution) of all areas on the globe with conditions matching those in the defined range. The matching process follows a simple bioclimatic envelope approach. For each input variable, minimum and maximum possible values are defined according to the values found in the pest's known geographic range. Then, for a new region of interest, e.g., the conterminous United States, geographic areas are identified that fall between these minimum and maximum values. A set of geographic areas is defined for each input variable in this manner, and in a subsequent overlay process, only those areas indicated as suitable, i.e., between the minimum and maximum values, for all variables are retained in the final climate match for the region of interest.

A climate-matching surface for the redbay ambrosia beetle was generated by first delineating the countries of the beetle's native range. Only the States of Assam and Bengal were included from India, and northern Japan was excluded because none of the beetle's known host species grows in that portion of the country. Species distributions are seldom defined by a single variable; temperature is often the most important limiting factor, but moisture affects both insect and host life cycles (Baker and others 2000, Peacock and others 2006). Hence, a climate model was applied that included three annual variables (growing degree days,

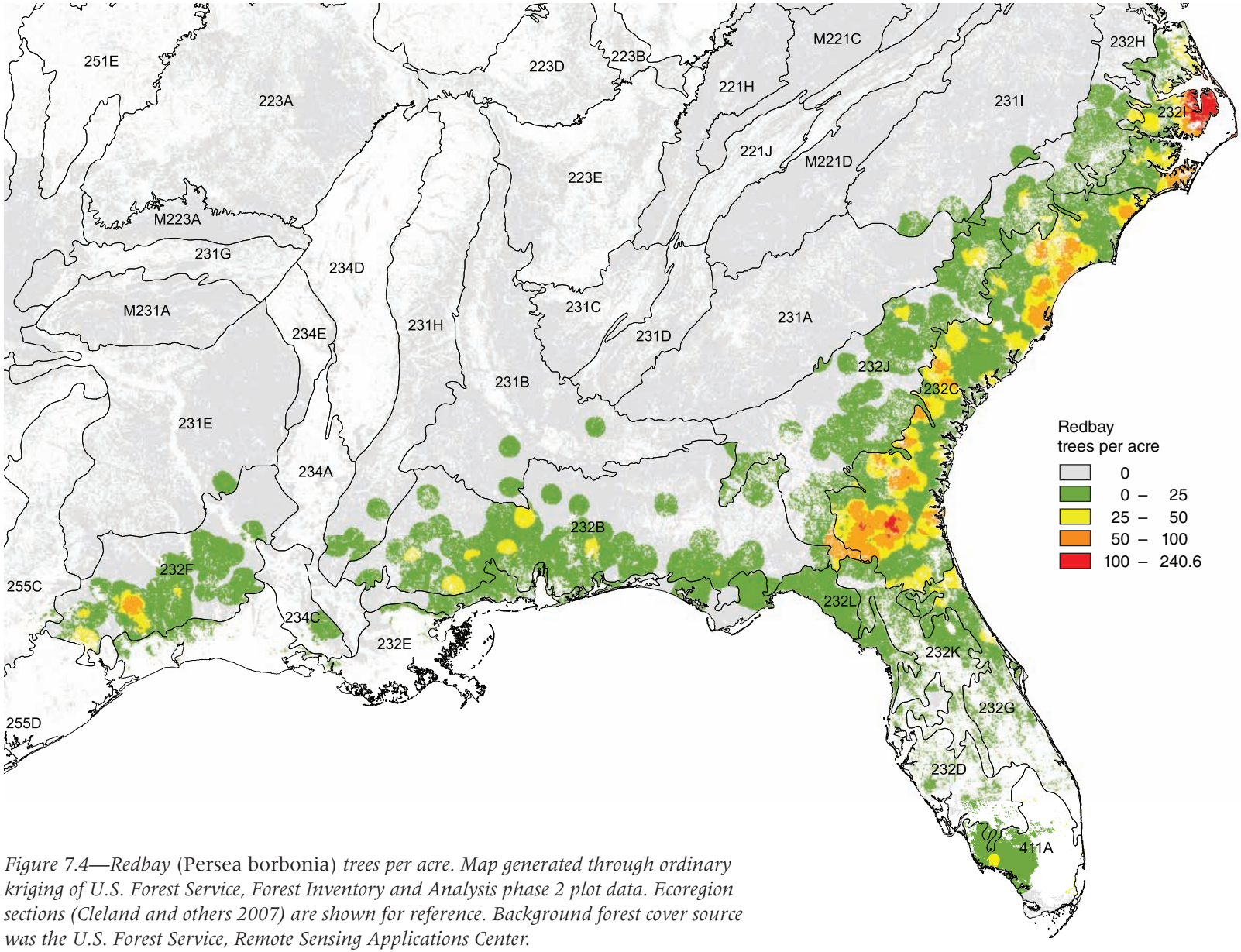


Figure 7.4—Redbay (Persea borbonia) *trees per acre. Map generated through ordinary kriging of U.S. Forest Service, Forest Inventory and Analysis phase 2 plot data. Ecoregion sections (Cleland and others 2007) are shown for reference. Background forest cover source was the U.S. Forest Service, Remote Sensing Applications Center.*

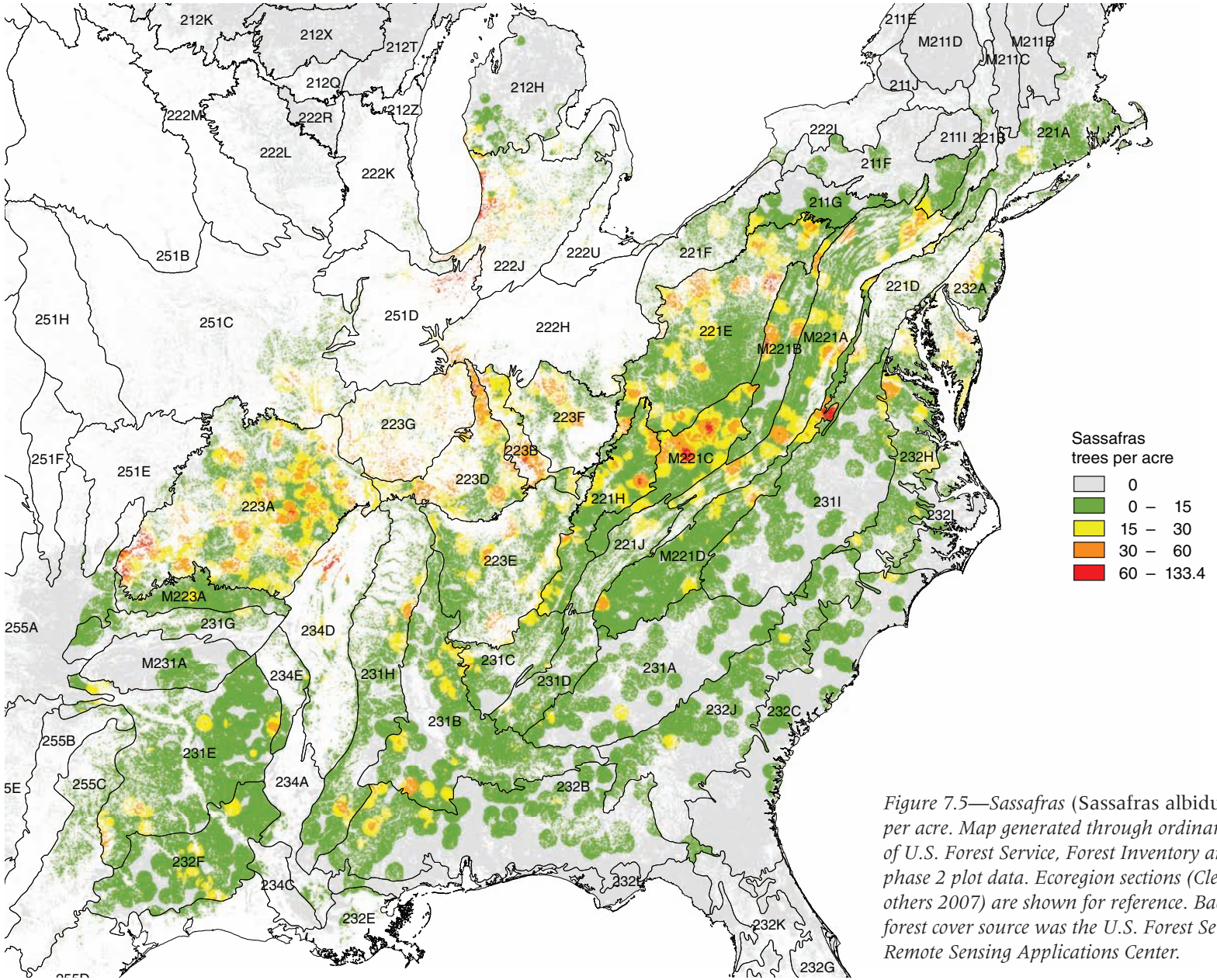


Figure 7.5—Sassafras (Sassafras albidum) *trees per acre. Map generated through ordinary kriging of U.S. Forest Service, Forest Inventory and Analysis phase 2 plot data. Ecoregion sections (Cleland and others 2007) are shown for reference. Background forest cover source was the U.S. Forest Service, Remote Sensing Applications Center.*

growing season moisture percentage, and 10-year extreme minimum temperature) to describe general climatic conditions. In addition, the model included variables representing seasonal effects, i.e., the amount of rainfall in each month of the approximate growing season, April through October (table 7.2). For the purpose of comparison, the model results were overlaid on the distributions of redbay and sassafras as mapped from FIA data (fig. 7.6).

Spread Modeling—A simple estimate of the rate of spread for the redbay ambrosia beetle and its associated fungus, assuming radial dispersion with exponential population growth, was developed from the county-level distribution data. In general, a spread rate estimated from such coarse-scale spatial data will not differ substantially from the rate that might be estimated with a more intensive, but costly, network of monitoring plots (Tobin and others 2007). Following procedures outlined by Banks (1994), the velocity of the expanding invasion front, i.e., the annual rate of spread, was calculated based on the infestation extents in 2004 through 2007, which were estimated as the total area (in km^2) of the counties infested as of

Table 7.2—Variables used in the NAPPFAST climate-matching model (Magarey and others 2007), and the minimum and maximum values for those variables found within the estimated native geographic range of the redbay ambrosia beetle

Variable	Minimum value	Maximum value
10-year mean extreme minimum cold temperature (°F)	-16	60
30-year mean growing degree days (baseline temperature 50 °F)[a]	1,645	13,036
30-year mean growing season moisture (percent)[b]	61	482
30-year mean April precipitation (inches)	0.08	11.87
30-year mean May precipitation (inches)	1.75	24.89
30-year mean June precipitation (inches)	3.15	37.38
30-year mean July precipitation (inches)	5.49	36.15
30-year mean August precipitation (inches)	4.15	36.07
30-year mean September precipitation (inches)	3.76	22.8
30-year mean October precipitation (inches)	1.83	11.83

NAPPFAST = NCSU/APHIS Plant Pest Forecast.
[a] Measure of accumulated warmth supporting the growth of organisms. For any given day in a year, its contribution to the accumulated annual growing degree days is based on the difference between the mean of the minimum and maximum daily temperature values and the baseline temperature.
[b] Ratio of precipitation to potential evapotranspiration.

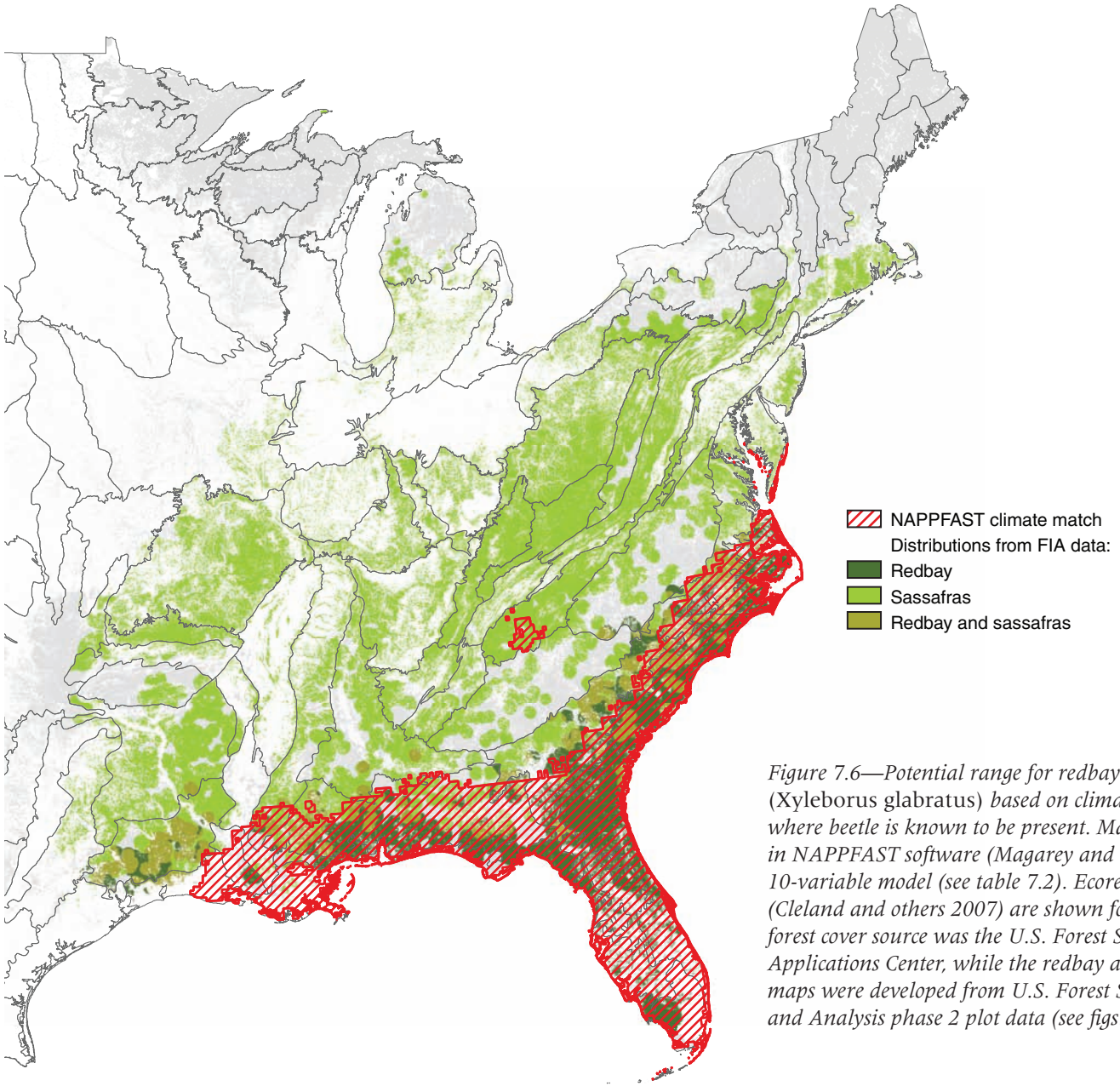


Figure 7.6—Potential range for redbay ambrosia beetle (Xyleborus glabratus) *based on climate match with countries where beetle is known to be present. Matching performed in NAPPFAST software (Magarey and others 2007) using a 10-variable model (see table 7.2). Ecoregion section boundaries (Cleland and others 2007) are shown for reference. Background forest cover source was the U.S. Forest Service, Remote Sensing Applications Center, while the redbay and sassafras distribution maps were developed from U.S. Forest Service, Forest Inventory and Analysis phase 2 plot data (see figs. 7.4 and 7.5).*

the year in question (table 7.3). If the dispersion geometry is assumed to be semicircular, i.e., in cases where dispersion is partially prohibited by a lengthy geographic barrier such as a coastline, the infestation extent A at time t may be approximated as

$$A = \frac{\pi R^2}{2} = 2\pi a D t^2$$

where

R = the radius of the semicircular invasion extent

a = the net growth rate

D = the dispersion coefficient

This equation may be simplified for t by taking the square root of both sides:

$$\sqrt{A} = \sqrt{2\pi a D}t$$

In short, the square root of the infested area is directly proportional to time t. The data from table 7.3 were applied in a least squares regression of $\sqrt{A}$ on year (t) to derive an estimate of $\sqrt{2\pi aD}$. Using this parameter estimate, the velocity of the expanding invasion front was subsequently calculated as

$$v = \frac{\delta R}{\delta t} = 2\sqrt{aD}$$

where

v = the velocity of expansion, or annual rate of spread, in km per year (Banks 1994).

Spatial spread of the redbay ambrosia beetle and laurel wilt through time (fig. 7.7) was modeled in a gridded environment using a cost-weighted distance function. Rather than calculating the shortest Euclidean distance between a pixel in a map grid and the nearest origin or source location, this function weights the distance vector according to an underlying cost grid. Cost grid values > 1 increase the cost of moving along the vector, while values < 1 decrease the cost of moving along the vector. The result is an accumulated cost, in distance units, of traveling along the vector; this accumulated cost may be more or less than the vector's actual Euclidean length.

Table 7.3—Infestation extents calculated from the combined areas of all counties confirmed as infested by the year in question

Year	Infestation extent
	km^2
2004	4 398
2005	18 965
2006	41 480
2007	57 723

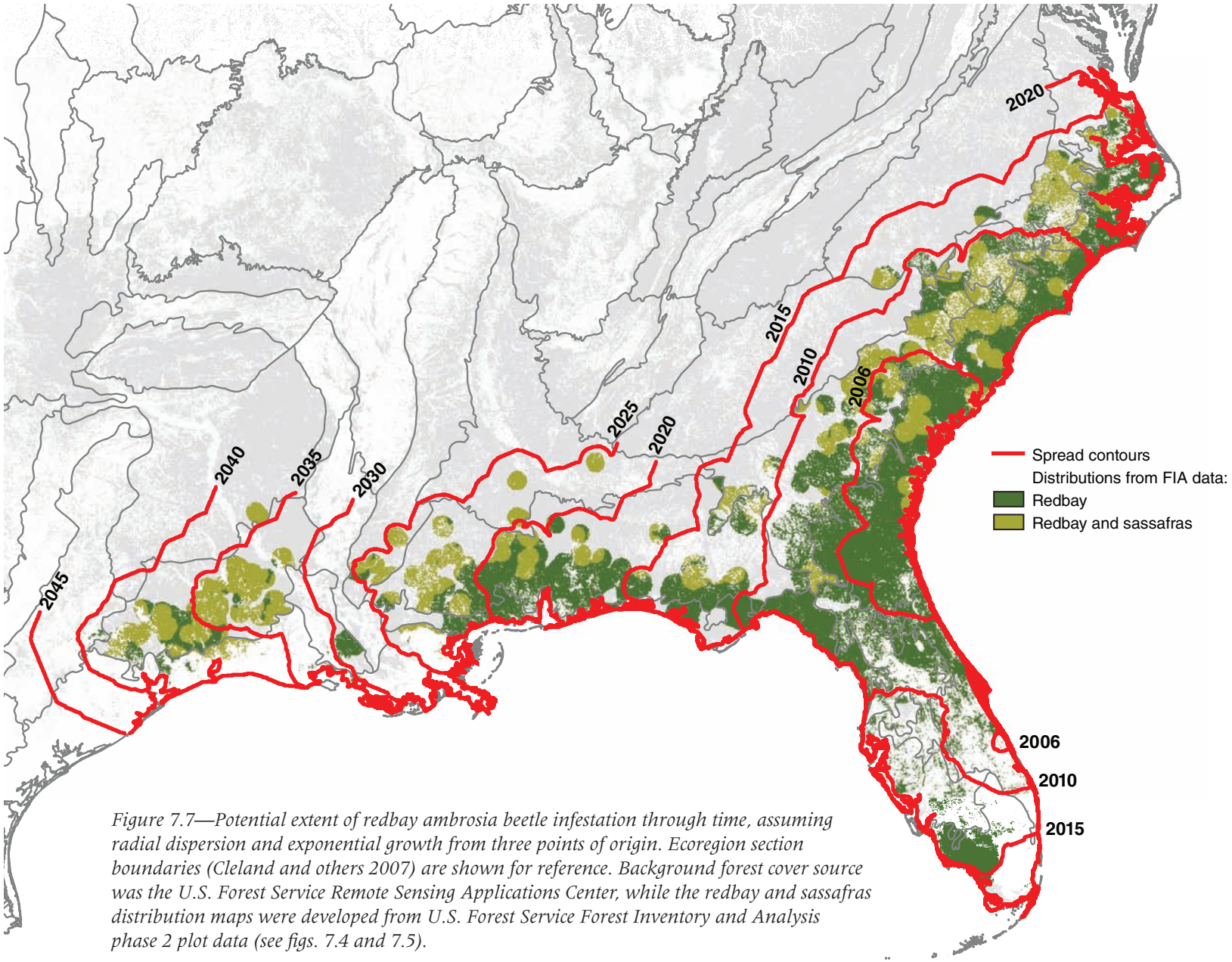


Figure 7.7—Potential extent of redbay ambrosia beetle infestation through time, assuming radial dispersion and exponential growth from three points of origin. Ecoregion section boundaries (Cleland and others 2007) are shown for reference. Background forest cover source was the U.S. Forest Service Remote Sensing Applications Center, while the redbay and sassafras distribution maps were developed from U.S. Forest Service Forest Inventory and Analysis phase 2 plot data (see figs. 7.4 and 7.5).

A nationwide cost grid was developed primarily from the interpolated map of redbay trees per acre, under the assumption that high host densities would decrease the travel cost for dispersing redbay ambrosia beetles, while a lack of hosts would greatly increase the travel cost, i.e., travel cost is inversely related to host availability. Although the beetle does attack sassafras, these attacks have so far occurred only in areas where redbay is common, so it is unclear whether the beetle is attracted to and will infest sassafras when redbay is absent. Therefore, we only added sassafras trees per acre to the total value for grid cells where the redbay trees per acre value was also greater than zero. Typically, the addition of sassafras resulted in a negligible increase in the total trees per acre value. A series of simple step functions, relating trees per acre (x) to cost (y), was developed by varying minimum and maximum possible cost values, as well as the threshold values between steps. The step functions were applied to the total trees per acre grid in order to make preliminary cost-weighted spread maps, which were then compared to the county-level infestation data. The step function that yielded the best approximation of the infestation pattern through time had a maximum cost value of 3 when no host was present and a minimum cost value of 0.25 at a host density of 240 trees per acre. Subsequently, a continuous function was estimated by fitting the selected step function with the complemented Weibull equation (Haefner 2005):

$$y = a * \exp\left(-\left[\frac{x}{b}\right]^{c}\right)$$

where

x = the trees per acre value

y = the cost value

a = a parameter that scales the maximum value

b = a parameter that controls the point on the x-axis at which the function is approximately zero

c = a parameter specifying whether the function is convex or concave

Parameters *a*, *b*, and *c* were estimated using the NLIN procedure for fitting nonlinear models in SAS (SAS Institute 2004); the parameterized function, as well as the step function to which it was fitted, is shown in figure 7.8. The function was applied to the total trees per acre map to create the cost grid, which was then used to create a map of cost-weighted distance from points of origin in Beaufort County, SC, as well as Duval and Indian River Counties in Florida.

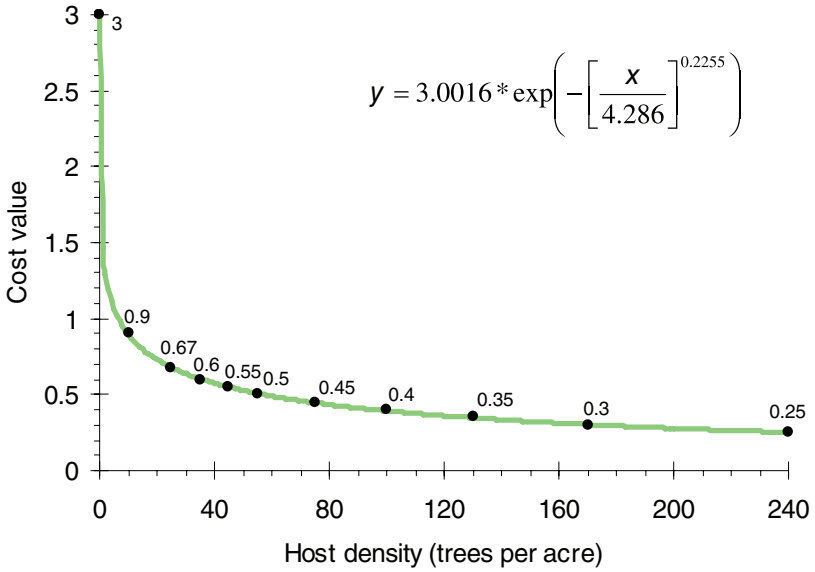


Figure 7.8—Complemented Weibull function used to relate host density in trees per acre to cost for the cost-weighted spread model. Black circles indicate point values that describe the step function to which this continuous function was fitted. See text for additional explanation. Residuals between step function values and predicted cost values were all within +/- 0.01.

The latter two points of origin were assumed to represent initially isolated infestations that had been caused by accidental long-distance transport of the redbay ambrosia beetle from locations to the north. All three originating infestations were assumed to have started 1 year prior to the confirmed presence of the beetle and/or fungus at each location. The cost-weighted distance map was reclassified into an increasing series of equal intervals based on the previously calculated invasion front velocity, such that each interval represented the amount of area added to the infestation extent with one additional year of infestation.

Results and Discussion

Current Beetle/Wilt Distribution—In interpreting the county-level distribution map for the redbay ambrosia beetle and laurel wilt (fig. 7.1), it is important to acknowledge the coarseness of the data and the fact that difficulties in positively confirming any pest's presence may result in a timelag in reporting. According to these data, in the 5 years since the ambrosia beetle was first detected in Port Wentworth, 18 Georgia counties, comprising the majority of the State's Lower Coastal Plain province (Hodler and Schretter 1986), have

been impacted by laurel wilt (fig. 7.1). In South Carolina, the seven counties closest to Port Wentworth have been impacted. Based on the chronology depicted by these data, it seems that the northward expansion of the beetle and fungus along the coast of South Carolina has been slower than the southward expansion in Georgia. However, the current extent of the invasion in Georgia may also have been influenced by northward movement of the beetle and fungus from locations in Florida. In northeastern Florida, the addition of four impacted counties in 2007 suggests rapid inland and westward expansion in addition to spreading south along the Atlantic coast. In addition, the recent detection of laurel wilt in Brevard County, FL, seems to indicate ongoing northward expansion from the previously isolated occurrence in Indian River County.

The Georgia and South Carolina redbay mortality survey results (fig. 7.2) provide some specific details on geographic pattern. In South Carolina, approximately 21 percent of the plots contained redbay trees affected by laurel wilt. The highest concentration of mortality was in the two coastal counties—Jasper and Beaufort—that are closest to Port Wentworth. However, Hampton, Bamberg, and Colleton Counties also had plots exhibiting severe (> 67 percent) redbay mortality. The leading edge of the infestation in the State appears to run along the border between Allendale and Hampton Counties, and then through southern Bamberg, Colleton, and southern Charleston Counties, basically mirroring the map of confirmed positive counties (fig. 7.1). Complicating the matter somewhat is light (< 33 percent) redbay mortality recorded in many plots throughout South Carolina. A lack of evidence of the laurel wilt fungus in these plots suggests the mortality was not precipitated by the redbay ambrosia beetle but instead represents a level of background mortality due to other insects, diseases, or undetermined environmental factors (see footnote 2).

The pattern of moderate (34 to 66 percent) to severe redbay mortality is more diffuse in Georgia. While there appears to be a large cluster of heavy mortality spanning Screven and Bulloch Counties, it should be noted that these two counties were intensively sampled; substantial redbay mortality was also recorded at plots in all of the less-intensively sampled counties directly along the coast, particularly Bryan County. Several plots in Camden County, just north of the Florida border, also exhibited moderate to severe mortality, supporting the notion of northward spread of the beetle and its associated fungus. Clear identification of

the leading front of infestation in Georgia is more difficult than for South Carolina because relatively isolated inland plots in Charlton and Appling Counties exhibited severe redbay mortality while neighboring plots displayed no mortality. As in South Carolina, the leading front appears to mirror the furthest extent of infestation as depicted in the county-level map. The Georgia survey also noted mortality of sassafras on one plot each in MacIntosh and Glynn Counties (see footnote 3), and mortality of sassafras due to laurel wilt has since been confirmed in Beaufort County, SC[4]. As previously noted, there is no evidence that the redbay ambrosia beetle is preferentially attracted to sassafras, so it appears likely the attacks on sassafras in these locations were merely opportunistic.

Host Species Distributions—As with fig. 7.1, the distributions of Lauraceae species are represented coarsely, i.e., at the county level, in fig. 7.3, so the amount and connectivity of potential host habitat may be less than the map suggests. Moreover, it is important to consider the lack of evidence that any Lauraceae species besides redbay, despite their demonstrated susceptibility to the laurel wilt fungus, is an equally attractive host for the redbay ambrosia beetle. Keeping these points in mind, with the exception of the Rocky Mountain region and at northern latitudes (above roughly 45° N.), most forested areas of the conterminous United States have at least one species from the Lauraceae family. California bay laurel is a common understory species in the forests of the Pacific coast; notably, this species has played a major role in the epidemiology of *Phytophthora ramorum*, the pathogen that causes sudden oak death (Rizzo and Garbelotto 2003). Northern spicebush and sassafras are both widespread throughout the East. Nevertheless, coastal areas of the Southeastern United States exhibit the greatest diversity of Lauraceae species. As noted in table 7.1, many of these species are rare and thus only sporadically distributed. The nonnative camphor tree (*Cinnamomum camphora*) is also sporadically distributed across the Southeast and the Pacific coast, while the bay laurel (*Laurus nobilis*) is limited to a few counties in northern California. Avocado trees are grown in commercial orchards in south Florida but can be found in a few sites elsewhere in the State. The avocado industry is actually far more extensive in southern California, but occurrence of the species outside a commercial setting in the State is undocumented.

[4] Personal communication. 2007. Laurie Reid, Forest Health Program Coordinator, South Carolina Forestry Commission, P.O. Box 21707, Columbia, SC 29212.

Redbay (fig. 7.4) is widespread across most of the ecoregion sections of the southeastern Coastal Plain at low densities (< 25 trees per acre), reaching as far south as the Everglades (section 411A). In particular, section 232C—Atlantic Coastal Flatwoods, which encompasses much of the area currently affected by redbay ambrosia beetle and laurel wilt, contains numerous redbay clusters of moderate to very high (> 100 trees per acre) density. Nonetheless, the greatest concentration of redbay by far is found on the Albemarle Peninsula of eastern North Carolina (in section 232I—Northern Atlantic Coastal Flatwoods). Along the gulf coast, there appears to be a break in the distribution of redbay at section 234A—Southern Mississippi Alluvial, which may be due in part to a lack of forested area in this section. West of this break, the species persists into the forests of eastern Texas.

Sassafras (fig. 7.5) is widely distributed at low levels (< 15 trees per acre) throughout the Southeast. Notably, the species reaches very high (> 60 trees per acre) density levels in the westernmost portions of section 223A—Ozark Highlands; this section serves as the western end of a swath of moderate to very high sassafras density extending eastward to sections 221E—Southern Unglaciated Allegheny Plateau and M221C— Northern Cumberland Mountains. Sassafras is present at low to moderate levels at least as far north as Michigan (section 212H—Northern Lower Peninsula and section 221A—Lower New England). In short, with the exception of some apparent gaps in sections 231A—Southern Appalachian Piedmont and 231I—Central Appalachian Piedmont, there appears to be a near-continuous distribution of sassafras from the upper edge of the southeastern Coastal Plain, although it is worth noting the great variation in topography and other environmental characteristics across the species' range.

Climate Matching—The range of values for each variable used in the climate matching is fairly large (table 7.2). This may be partially attributed to the scale at which the native geographic distribution of the redbay ambrosia beetle was defined during the modeling process. Without detailed distribution information, it was typically necessary to include entire countries or large portions of those countries in the delineated extent. More specific observational data on the beetle's occurrence would allow refinement of this distribution and probably decrease the range in variable values. Regardless, the resulting map (fig. 7.6) illustrates some points relevant to the risk of the beetle's spread in the United States. Most importantly, the climatically suitable area is largely constrained to the Coastal Plain of the

Southeastern United States, or essentially the range of redbay; with the exception of a small area in the southern portion of the Blue Ridge Mountains (ecoregion section M221D), no other part of the conterminous United States appears suitable under the model. Functionally, the model does not define meaningful upper limits for monthly rainfall since the maximum values for these variables under the model are generally far beyond the maximums found in the conterminous United States. However, the model does define minimum acceptable levels of monthly rainfall during the growing season, and these minimums, especially for August and September, exclude much of the area deemed suitable under the model, i.e., only the Southeastern United States receives adequate rainfall across all months of the growing season. The ecological importance of this seasonal effect is uncertain, but it is worth noting that the fungi associated with ambrosia beetles typically have exacting moisture requirements: if there is too much moisture, the beetles and larvae will drown in an overabundance of fungi, and if there is too little moisture, the fungi will die, leaving no food source for the beetles (Furniss and Carolin 1977). At the least, the climate matching results suggest that coastal areas of the Southeast are far more susceptible than elsewhere in the United States, and that this may limit the spread of redbay ambrosia beetle and laurel wilt into forests of the U.S. interior.

Spread Modeling—The calculated invasion front velocity, i.e., the annual rate of spread, for the beetle and its associated fungus was 46.9 km (29.1 miles) per year, which falls in the range of anecdotal estimates. However, our assumption of simple radial diffusion does not account for potential variability in the rate of spread between geographically distinct areas; there is some evidence suggesting the actual rate of spread is much higher in Florida than in South Carolina,[5] where northward expansion has been relatively slow despite a relative abundance of redbay (see footnote 2). The rate also does not account for the possibility of long-distance dispersal events and so may be seen as a conservative estimate of the spread rate in this regard. Broadly, pest invasions may occasionally accelerate or decelerate through time, particularly in response to anthropogenic influences (Liebhold and others 1992). A more realistic model might incorporate specific population parameters or allow for long-distance dispersal beyond the main infestation front. Furthermore, it is important to consider that the rate of spread was estimated using only a few years of data and for a pest that was

[5] Mayfield, A.E. 2007. RE: redbay wilt/ambrosia beetle. fkoch@fs.fed.us. (22 May, 2007).

only recently identified in the United States. The limited data make the estimate especially sensitive to the possibility that some of the apparently new infestations discovered in any given year are actually just late detections of infestations that had existed for some time previously. As a result, the invasion timeline may be falsely accelerated and the actual rate of spread may be overestimated. To alleviate this problem, the rate of spread estimate may be refined as additional years of data become available; indeed, the estimate presented here has decreased from a spread rate of approximately 55 km per year calculated using only the distribution data from 2004 to 2006 (Koch and Smith 2008). This decrease, based on an additional year of data, emphasizes that the estimate should only be interpreted as preliminary.

The possible timelag in detections also has bearing for the cost function and corresponding cost grid used in modeling spread, since we fitted the function to the county-level distribution data as they are currently depicted. Beyond this issue, it must also be acknowledged that the cost-weighted spread model assumes a fairly straightforward relationship between host density and the dispersal capability of the redbay ambrosia beetle. When viewed at a broad scale, i.e., at the metapopulation level, it seems reasonable that the distance the main invasion front advances in a given year depends on the availability of hosts to sustain population growth near the front. However, this approach does not account for individual beetles that may venture beyond the main front in search of hosts. Still, if nothing else, this simple model illustrates the scope of the threat represented by the beetle and its associated fungus (fig. 7.7). Assuming a mean rate of spread of approximately 47 km/year and assuming spread is influenced by the density of available hosts, the redbay ambrosia beetle could reach both the northern and southern limits of redbay by 2020, with nearly the entire range of redbay infested by 2040. Even if spread is reduced by 16 km (10 miles) per year, e.g., as a result of suppression efforts, the entire range of redbay may be infested in just over 50 years.

Conclusions

There is little reason to doubt that the current redbay ambrosia beetle infestation, if unchecked, will expand north and south along the Atlantic coast of the United States and west along the gulf coast. Redbay is present throughout these regions along with other susceptible and potentially susceptible hosts such as sassafras that, while not necessarily attractive, may still be utilized by the beetle. There are few geographic barriers aside from the Mississippi River, and a fairly restrictive model of climatic suitability puts nearly the entire range of redbay at approximately equal risk of infestation. While the climate-matching model

does seem to suggest that the western edge of the redbay range may be unsuitable, this should be viewed cautiously given the limitations of the analysis. In any case, it seems unlikely the redbay ambrosia beetle and its associated fungus will spread significantly into inland forests based on the apparent climatic and host constraints. Moreover, the Appalachians or related mountain chains may represent an additional geographic barrier to broad northward expansion. Further research is needed on both the beetle and fungus to ascertain a more certain picture of the potential risks.

Acknowledgments

Laurie Reid, South Carolina Forestry Commission; Albert "Bud" Mayfield, Forest Service, formerly Florida Department of Agriculture and Consumer Services, Division of Forestry; James Johnson, Georgia Forestry Commission; Steve Fraedrich, Forest Service; and Don Duerr, Forest Service provided a wealth of background information and key datasets used in writing this chapter.

Literature Cited

Baker, R.H.A.; Sansford, C.E.; Jarvis, C.H. [and others]. 2000. The role of climatic mapping in predicting the potential geographic distribution of non-indigenous pests under current and future climates. Agriculture, Ecosystems and Environment. 82: 57–71.

Banks, R.B. 1994. Growth and diffusion phenomena: mathematical frameworks and applications. Berlin, Germany: Springer-Verlag. 449 p.

Cleland, D.T.; Freeouf, J.A.; Keys, J.E. [and others]. 2007. Ecological subregions: sections and subsections for the conterminous United States. Sloan, A.M., cartog. Gen. Tech. Rep. WO-76. Washington, DC: Forest Service, U.S. Department of Agriculture. [Map, presentation scale 1: 3,500,000; colored]. [Also on CD–ROM as a Geographic Information System coverage in ArcINFO format].

Coder, K.D. 2006a. Identifying characteristics of redbay (*Persea borbonia*). Outreach Publ. SFNR06–4. Athens, GA: University of Georgia, Warnell School of Forestry and Natural Resources. 5 p.

Coder, K.D. 2006b. Stress, pests, and injury in redbay (*Persea borbonia*). Outreach Publ. SFNR06–5. Athens, GA: University of Georgia, Warnell School of Forestry and Natural Resources. 5 p.

Fraedrich, S.W. 2007. Laurel wilt: hosts and pathogenicity tests. Forest Service, U.S. Department of Agriculture, Forest Health Protection, Southern Region. [Online document]. http://www.fs.fed.us/r8/foresthealth/laurelwilt/resources/presentations/additional_information/fraedrich_hosts_patho_tests.pdf. [Date accessed: July 23].

Fraedrich, S.W.; Harrington, T.C.; Rabaglia, R.J. 2007. Laurel wilt: a new and devastating disease of redbay caused by a fungal symbiont of the exotic redbay ambrosia beetle. Newsletter of the Michigan Entomological Society. 52(1&2): 15–16.

Furniss, R.L.; Carolin, V.M. 1977. Western forest insects. Misc. Publ. 1389. Washington, DC: Forest Service, U.S. Department of Agriculture. 654 p.

Griggs, M.M. 1990. *Sassafras albidum* (Nutt.) Nees—sassafras. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America. 2. Hardwoods. Agric. Handb. 654. Washington, DC: Forest Service, U.S. Department of Agriculture: 773–777.

Haack, R.A. 2006. Exotic bark- and wood-boring Coleoptera in the United States: recent establishments and interceptions. Canadian Journal of Forest Research. 36: 269–288.

Haefner, J.W. 2005. Modeling biological systems. New York: Springer Science+Business Media. 475 p.

Harrington, T.C.; Fraedrich, S.W.; Aghayeva, D.N. 2008. *Raffaelea lauricola*, a new ambrosia beetle symbiont and pathogen on the Lauraceae. Mycotaxon. 104: 399–404.

Helm, A.C.; Nicholas, N.S.; Zedaker, S.M.; Young, S.T. 1991. Maritime forests on Bull Island, Cape Romain, South Carolina. Bulletin of the Torrey Botanical Club. 118: 170–175.

Hodler, T.W.; Schretter, H.A. 1986. The atlas of Georgia. Athens, GA: University of Georgia Press. 273 p.

Holistic Insect Systematics Laboratory. 2005. Xyleborini taxonomy and literature database. East Lansing, MI: Michigan State University, Department of Entomology. http://xyleborini.tamu.edu/. [Date accessed: June 17, 2007].

Kartesz, J.T. 2003. Atlas with biological attributes for the vascular flora of the United States, Canada, and Greenland [CD–ROM]. 2d ed. In: Kartesz, J.T. Synthesis of the North American flora. Version 2.0. Chapel Hill, NC: North Carolina Botanical Garden, Biota of North America Program.

Koch, F.H.; Smith, W.D. 2008. Spatio-temporal analysis of *Xyleborus glabratus* (Coleoptera: Circulionidae: Scolytinae) invasion in Eastern U.S. forests. Environmental Entomology. 37(2): 442–452.

Liebhold, A.M.; Halverson, J.A.; Elmes, G.A. 1992. Gypsy moth invasion in North America: a quantitative analysis. Journal of Biogeography. 19(5): 513–520.

Magarey, R.D.; Fowler, G.A.; Borchert, D.M. [and others]. 2007. NAPPFAST: an Internet system for the weather-based mapping of plant pathogens. Plant Disease. 91(4): 336–345.

Mayfield, A.E. 2007. Evaluation of systemic fungicides for preventing laurel wilt. Forest Service, U.S. Department of Agriculture, Forest Health Protection, Southern Region. [Online document]. http://www.fs.fed.us/r8/foresthealth/laurelwilt/resources/presentations/jekyll_island_jan2007/mayfield_pres_fungicide_redbay_revised.pdf. [Date accessed: July 23].

Mayfield, A.E.; Thomas, M.C. 2006. Pest alert: redbay ambrosia beetle, *Xyleborus glabratus* Eichhoff (Scolytinae: Curculionidae). Tallahassee, FL: Florida Department of Agriculture and Human Services, Division of Plant Industry. 4 p. http://www.doacs.state.fl.us/pi/enpp/ento/x.glabratus.html. [Date accessed: March 12, 2007].

Peacock, L.; Worner, S.; Sedcole, R. 2006. Climate variables and their role in site discrimination of invasive insect species distributions. Environmental Entomology. 35(4): 958–963.

Rabaglia, R.J. 2003. Pest report: *Xyleborus glabratus*. Fort Collins, CO: North American Forest Commission, Exotic Forest Pest Information System. 4 p. http://spfnic.fs.fed.us/exfor/data/pestreports.cfm?pestidval=148&langdisplay=english. [Date accessed: March 12, 2007].

Rabaglia, R.J.; Dole, S.A.; Cognato, A.I. 2006. Review of American Xyleborina (Coleoptera: Curculionidae: Scolytinae) occurring north of Mexico, with an illustrated key. Annals of the Entomological Society of America. 99(6): 1034–1056.

Rizzo, D.M.; Garbelotto, M. 2003. Sudden oak death: endangering California and Oregon forest ecosystems. Frontiers in Ecology and Environment. 1(5): 197–204.

SAS Institute. 2004. SAS/STAT user's guide. Version 9.1. Cary, NC: SAS Institute Inc. 5,121 p.

Tobin, P.C.; Liebhold, A.M.; Roberts, E.A. 2007. Comparison of methods for estimating the spread of a non-indigenous species. Journal of Biogeography. 34: 305–312.

U.S. Department of Agriculture, Animal and Plant Health Inspection Service. 2007. NCSU-APHIS plant pest forecasting system (NAPPFAST). http://www.nappfast.org. [Date accessed: December 7].

Forest Service, U.S. Department of Agriculture. 2007. Laurel wilt: plant susceptibility. http://www.fs.fed.us/r8/foresthealth/laurelwilt/plant_susceptibility.shtml. [Date accessed: July 23].

U.S. Department of Agriculture, Natural Resources Conservation Service. 2007. The PLANTS database. Baton Rouge, LA: National Plant Data Center. http://plants.usda.gov. [Date accessed: June 17].

Appendix

Kriging is a geostatistical technique for spatially interpolating values of a continuous variable at unknown points from the values of neighboring points. The underlying spatial structure, i.e., spatial dependence or autocorrelation, of a variable may be quantified using an empirical semivariogram constructed from pairs of the known points comprising a sample of the variable. For a set of points, the semivariance at a given distance h is calculated as:

$$\gamma(h) = \frac{1}{2N(h)} \sum_{N(h)} (Z(x_i) - Z(x_j))^2$$

where

x_i and x_j = known point locations separated by distance h

$Z(x_i)$ and $Z(x_j)$ = variable values at the known points

$N(h)$ = the number of point pairs separated by distance h (Cressie 1993)

Briefly, the semivariance at distance h equals one-half the mean squared difference in variable values between all point pairs at distance h. When plotted, an empirical semivariogram depicts the relationship between semivariance and distance (fig. 7A.1). Generally, it is impractical to plot the semivariance values at all possible values of h, so the values are grouped into lag distance bins, e.g., all point pairs between 10 km and 20 km apart, for simplicity. A semivariogram has three key parameters. The nugget is the semivariance when the distance equals zero and is often equated to measurement error. The sill is the semivariance, and the range is the distance at which the plot flattens out, i.e., where the sample locations are no longer spatially autocorrelated (Environmental Systems Research Institute 2003). The partial sill is the difference between the nugget and sill values.

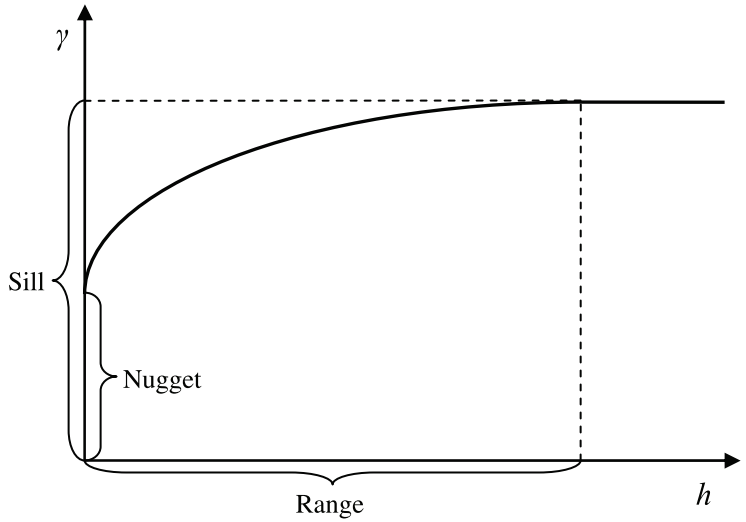


Figure 7A.1—Conceptual rendering of a semivariogram showing the nugget, range, and sill.

An empirical semivariogram may be estimated using one of numerous model semivariograms. For the redbay and sassafras trees-per-acre surfaces, spherical semivariogram models were fitted to empirical semivariograms constructed using all FIA plots in the ecoregion sections containing each species. For each, the lag distance was set to 30 000 m, and weighted least squares (Cressie 1993) was used to estimate the model semivariogram parameters: for redbay, range = 355,600 m, partial sill = 467.79, and nugget = 1544.7; for sassafras, range = 212,480 m, partial sill = 323.11, and nugget = 1333.5. These estimated spherical models were applied when performing the bysection interpolations for each species.

Ordinary kriging is a commonly used form of kriging that assumes a constant mean. With respect to spatial prediction, kriging is similar to inverse distance weighting and other methods that estimate values for unknown points based on a weighted average of known neighboring points. The variable value Z at unknown location s_0 is estimated as

$$\hat{Z}(s_0) = \sum_{i=1}^{n} \lambda_i Z(s_i)$$

where

$Z(s_i)$ = the known variable value at neighboring location s_i

n = the number of neighboring known points used to estimate the value at s_o

λ_i = the kriging weight for location s_i, such that $\sum_{i=1}^{n} \lambda_i = 1$

The kriging weight for a given neighboring point s_i is calculated using the model semivariogram, such that the weight reflects not just distance but also the degree of spatial autocorrelation indicated by the model.

Literature Cited

Cressie, N.A.C. 1993. Statistics for spatial data. New York: John Wiley. 900 p.

Environmental Systems Research Institute. 2003. Using ArcGIS geostatistical analyst. Redlands, CA: Environmental Systems Research Institute. 300 p.

Chapter 8. *Phytophthora ramorum* Early Detection Surveys for Forests

Steven Oak

Introduction

Diseases caused by *Phytophthora ramorum* Werres, De Cock, & Man in't Veld including ramorum canker (also known as sudden oak death), ramorum blight, and ramorum dieback were first detected in U.S. forest landscapes in Marin County, CA, in the mid-1990s. The range in the United States has expanded since then, but in 2007 it was still limited to 14 central coastal California counties and a small area in Curry County, OR (U.S. Department of Agriculture, Animal and Plant Health Inspection Service 2007). Despite the presently limited disease distribution in forests and measures to limit artificial movement of the pathogen by State and Federal regulatory agencies through quarantine and restrictions on the movement of plant materials, other areas are at risk. This is especially true for oak-dominated forest ecosystems in the Eastern United States. In these areas, susceptibility of many closely related native Eastern United States trees and shrubs have been demonstrated in greenhouse inoculation trials (Tooley and Kyde 2007) and via natural infection in Europe (Brasier and others 2004). Climate parameters over much of the area are at least seasonally similar to those prevailing in *P. ramorum* endemic areas on the West coast, and brisk trade in many susceptible woody ornamental hosts grown there make accidental introduction an ever-present danger.

The risk of establishment of the pathogen outside the regulated areas and the potential for severe ecosystem damage prompted Federal and State forest management agencies in seven Eastern States to join in pilot tests of early detection terrestrial vegetation survey methods for forests. These surveys began in 2003, were expanded to more States, and continued annually through 2006 when aquatic survey methods were pilot tested concurrent with the terrestrial vegetation surveys in 11 States. Results of terrestrial vegetation surveys (Oak and others 2008a) and the 2006 pilot stream survey (Oak and others 2008b) have been presented. The purpose of this chapter is to present the combined results of the national *P. ramorum* early detection survey for U.S. forests using both terrestrial vegetation (2003 through 2006) (Oak and others 2008a) and stream baiting (2006 and 2007) survey protocols.

The occurrence and distribution of *Phytophthora* species in waterways in various environmental settings worldwide has been studied using filtration and plant tissue baits from a wide variety of plant species (Klotz and others 1959, McIntosh 1966, Robertson 1975, Thompson and Allen 1974, von Broembsen 1984). Recently, monitoring of *P. ramorum* in forest streams of California and Oregon has been shown to be effective using pear fruits and foliage of tanoak (*Lithocarpus densiflorus*)

and rhododendron (*Rhododendron* spp.) as bait (Hansen and Sutton 2006, Tjosvold and others 2006). The success of these methods for detecting *P. ramorum* in water, even before host symptoms are visible in aerial surveys, has resulted in early detection and treatment of previously unknown infested sites (Hansen and Sutton 2006, Murphy and others 2006). Given that previously unknown infection centers have been detected up to 8 km downstream from the inoculum source, monitoring forest streams for *P. ramorum* in high-risk regions should afford an opportunity to survey larger land areas with greater efficiency and at lower cost than is possible with current terrestrial vegetation survey techniques.

The 2006 pilot survey was highly successful in detecting multiple *Phytophthora* species in surveyed waterways (Oak and others 2008b). *Phytophthora* species were detected in over 80 percent of the bait sets diagnosed from 90 watersheds surveyed in 11 States. *P. ramorum* was detected in all streams known by previous sampling to be positive and for the first time in one Washington stream not previously known to harbor the pathogen. Because of these results, stream baiting protocols replaced terrestrial vegetation surveys in the 2007 national *P. ramorum* early detection survey for U.S. forests.

Methods

Terrestrial Vegetation Surveys—Terrestrial survey methods have been described in detail by Oak and others (2006). Briefly, the survey is guided by projected risk based on abundance and distribution of putatively susceptible hosts, climatic match with endemic West coast areas, and potential pathways of introduction via woody ornamental nursery stock. Up to 30 survey locations per cooperating State are used in two types of field settings—wooded nursery perimeter and general forest area. Four 100-m transects of variable width (dependent upon host type density) are used to survey each location. Symptomatic host or associated host plant (HAP) species (U.S. Department of Agriculture, Animal and Plant Health Inspection Service 2007) are targeted where available, while species belonging to HAP genera are targeted outside the endemic States of California and Oregon. The presence of *P. ramorum* in or on host tissues is verified by nested or realtime Polymerase Chain Reaction (PCR) according to U.S. Department of Agriculture, Animal and Plant Health Inspection Service (USDA–APHIS) approved protocols (DeVries and Levy 2006, Levy and Mavrodieva 2004). Replicates of each sample are tested at both local and regional laboratories in order to

minimize the possibility of unnecessary action based on false positive or uncertain results from a single diagnostic procedure.

Stream Surveys—Protocols closely followed those established for the 2006 pilot survey (Oak and others 2008b). Each stream is baited with four detached but otherwise unwounded *Rhododendron* spp. leaves contained in a mesh bag made of plastic window screen. Bags are tethered to streambanks and floated in the current for 1 to 2 weeks (exposure period depended on symptom development). After retrieval, bait leaves are wrapped in a paper towel moistened with streamwater, sealed in a plastic bag, and placed on ice for transport to diagnostic laboratories.

Bait leaves are washed under running tap water to remove silt and organic debris and blotted dry. Symptomatic leaf pieces are then excised without surface disinfestation. The only significant departure from the 2006 protocol is the mandatory use of isolation on selective medium in local diagnostic laboratories, while regional laboratories use nested or realtime PCR.

Results

Terrestrial Vegetation Surveys—*P. ramorum* forest surveys were conducted in at least 1 year between 2003 and 2006 in a total of 39 States (table 8.1). Of the 25 States that confirmed *P. ramorum* in woody ornamental nurseries during this period (California Oak Mortality Task Force 2009) only Arizona, Colorado, and New Mexico did not conduct surveys. Projected risk in these States is low. Nursery perimeters represented 62 percent of the 3,570 locations, reflecting our judgment that these settings were most likely to show symptoms the earliest, given the large shipments of potentially infected woody ornamental nursery stock in 2004 and subsequent, smaller shipments in each year thereafter. Local and regional laboratories diagnosed a combined 12,699 symptomatic tissue samples, and only 2 were found positive for *P. ramorum*. Both of these were *Quercus agrifolia* bark samples collected in San Francisco County, CA.

Symptomatic tissues from 44 identified genera were submitted for molecular diagnostics. Samples from species of *Acer* and *Lonicera* were the most abundant (table 8.2), while samples

Table 8.1—Summary survey statistics 2003–06

Survey characteristic	Year				Overall
	2003	2004	2005	2006	
Cooperating States	7	36	39	36	39
Locations surveyed					
Nursery perimeter	44	881	682	607	2,214
General forest	128	304	487	437	1,356
Total	172	1,185	1,169	1,044	3,570
Samples diagnosed	1,092	4,263	3,328	4,016	12,699
Phytophthora ramorum positive diagnosis	0	2	0	0	2

Table 8.2—Frequency and rank of plant genera sampled and submitted for *Phytophthora ramorum* diagnosis. Only genera ranked among the top 10 in any survey year are shown

Sample genus	Overall		Year							
			2003		2004		2005		2006	
	No.	Rank	No.	Rank	No.	Rank	No.	Rank	No.	Rank
Acer	2,642	1	11	6	970	1	829	1	832	1
Lonicera	2,068	2	17	5	712	3	639	2	700	2
Unidentified	1,637	3	284	2	850	2	208	7	295	4
Kalmia	1,318	4	363	1	437	4	277	4	241	5
Quercus	1,204	5	175	4	295	5	416	3	318	3
Rhododendron	933	6	231	3	257	6	244	5	201	6
Vaccinium	655	7	0	–[a]	234	7	218	6	203	7
Viburnum	366	8	3	8	108	8	130	8	125	10
Rubus	237	9	0	–[a]	37	13	4	22	196	8
Prunus	214	10	0	–[a]	64	10	20	17	130	9
Hamamelis	213	11	0	–[a]	83	9	71	9	59	16
Aesculus	136	12	0	–[a]	43	11	46	10	47	17
Castanea	48	20	8	7	27	14	5	21	8	25
Total	12,699		1,092		4,263		3,328		4,016	

[a] Rank is undefined due to absence of samples.

from unidentified genera ranked third in abundance. Symptomatic leaves were by far the most common tissue type, accounting for 94 percent of all samples. Bark from stem hosts represented < 5 percent of samples.

Stream Surveys—Overall, in 2007, 152 streams were surveyed in 28 cooperating States for the requisite 5 baiting periods or more (fig. 8.1). The extra bait deployments resulted in a possible 871 bait sets (table 8.3). Some bait sets were partially or completely lost due to high storm flows, vandalism, or other causes. In the case of partial losses, diagnosis by isolation only was performed. Overall, < 4 percent of bait sets were lost.

Previous surveys showed that *P. ramorum* was present in 11 streams surveyed in California and Oregon. These served as positive checks. The pathogen was detected in baits from all of these check streams at least once in five baiting periods by either direct isolation, PCR, or both diagnostic methods. In addition, *P. ramorum* was detected in two streams that were not known to harbor the pathogen prior to the 2007 survey. One stream was in Washington and was different from the stream found positive in the 2006 pilot survey (Oak and others 2008b), while the other was in Mississippi. The watersheds drained by both streams contained one or more confirmed positive woody ornamental nurseries. *P. ramorum* negative diagnoses for other streams are supported by the fact that *Phytophthora* spp. were detected by isolation in over 80 percent of all bait sets diagnosed nationally, including recovery of multiple species in *P. ramorum* positive streams.

PCR was more sensitive than isolation for detecting *P. ramorum* in known positive streams (table 8.4). However, the need for replicate diagnostics was demonstrated by the fact that the pathogen was detected by isolation, but not PCR, in baits from the Washington stream, while the reverse was true for baits from the Mississippi stream. Further, single positive PCR results were obtained from baits deployed in several California and Oregon streams not known to harbor the pathogen. This also occurred in the 2006 pilot survey (Oak and others 2008b). The small number of these cases combined with the location far from known inoculum sources and the lack of repeat positives through time by either diagnostic method suggests the possibility of false positive PCR diagnosis.

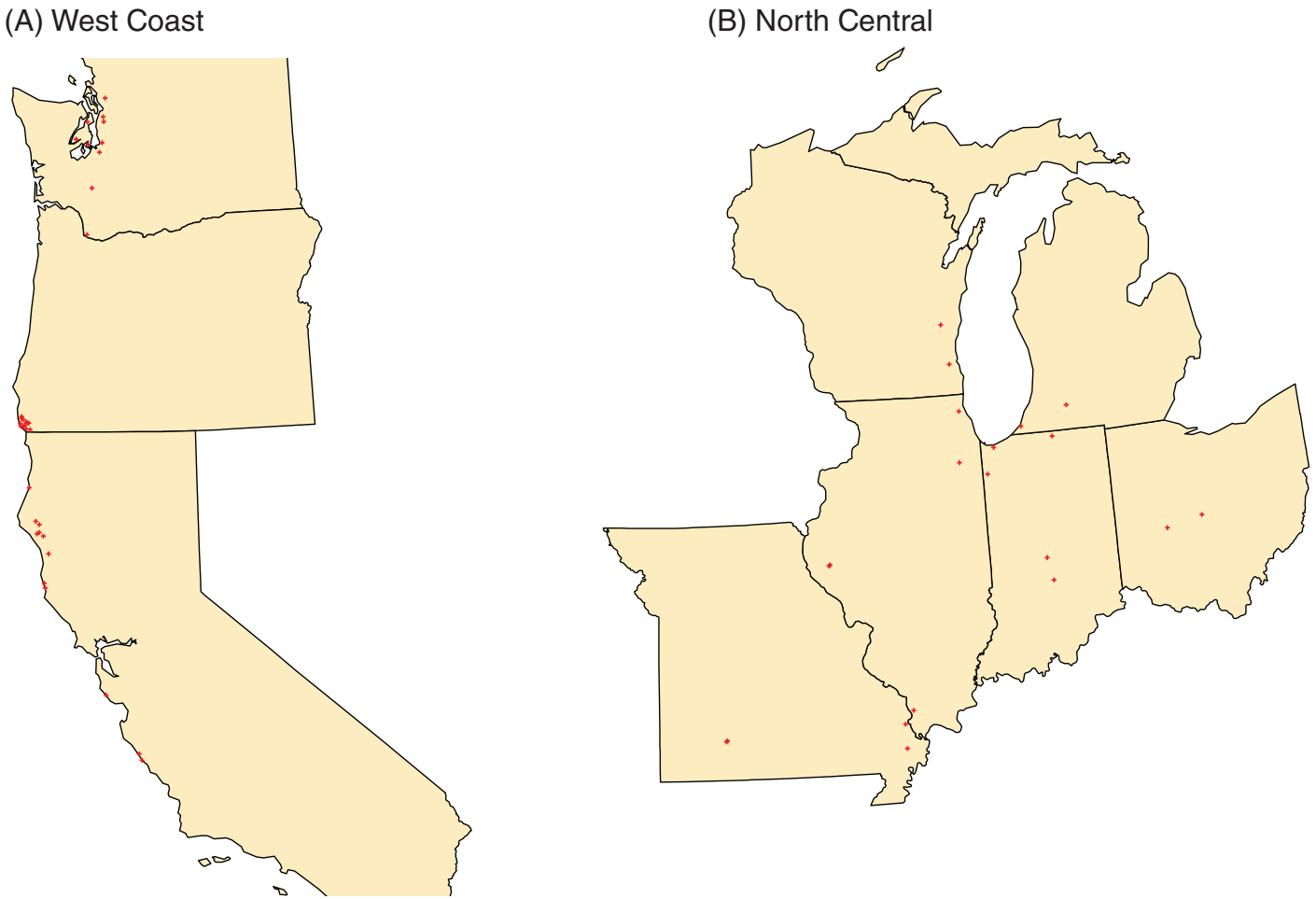


Figure 8.1—Locations of streams surveyed in the 2007 Phytophthora ramorum *early detection survey in forests. Note: Maps in this figure are not to scale in relation to one another.*

(continued on next page)

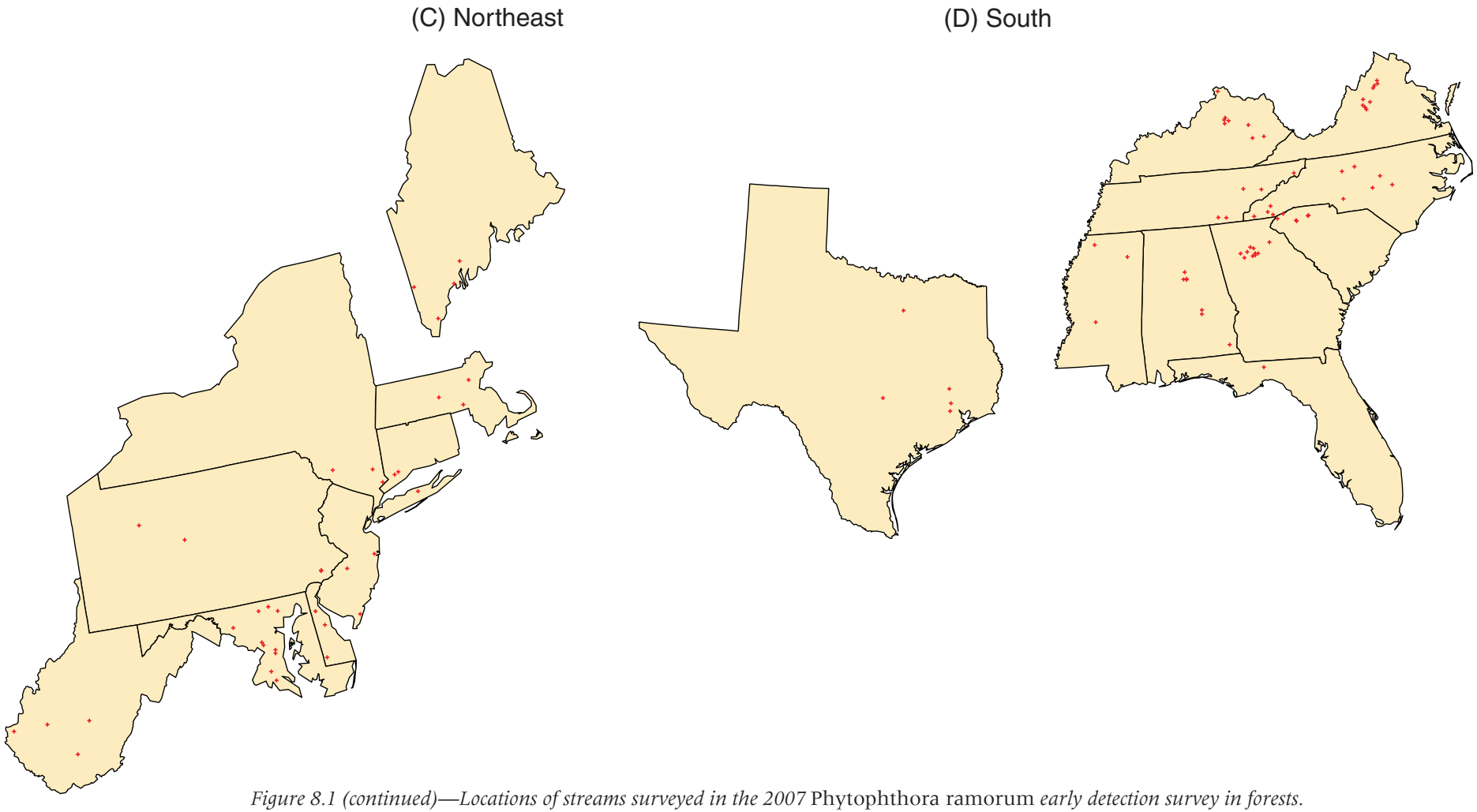


Figure 8.1 (continued)—Locations of streams surveyed in the 2007 Phytophthora ramorum *early detection survey in forests.*

Table 8.3—Summary stream baiting survey statistics, 2007

Survey characteristic	Region				Total
	West Coast	North Central	Northeast	South	
Cooperating States	3	6	9	10	28
Streams surveyed					
Phytophthora ramorum positive	12[a]	0	0	1	13
P. ramorum negative	22	18	36	63	139
Total	34	18	36	64	152
Bait sets diagnosed					
Isolation subtotal	259	90	180	317	846
P. ramorum positive	22	0	0	0	22
Other *Phytophthora* spp. positive	166	88	139	265	658
All *Phytophthora* spp. positive (percent)	188 (72.6)	88 (97.8)	139 (77.2)	265 (83.6)	680 (80.4)
PCR subtotal	257	88	181	305	831
P. ramorum positive	53	0	0	1	54

PCR = Polymerase Chain Reaction.
[a] *Phytophthora ramorum* was known to be present in 11 West coast streams prior to survey. These streams served as positive checks.

Table 8.4—Summary diagnostic statistics for *Phytophthora ramorum* positive streams, 2007

State	Streams	*Phytophthora ramorum* diagnosis					
		Isolation			Polymerase chain reaction		
		Attempts	Positive	Percent	Attempts	Positive	Percent
California	9	43	9	20.9	84	33	39.3
Oregon	2	26	12	46.2	26	19	73.1
Washington	1	9	1	11.1	9	0	0
Mississippi	1	5	0	0	5	1	20
All	13	83	22	26.5	124	53	42.7

Discussion

Terrestrial surveys conducted from 2003 through 2006 resulted in the addition of San Francisco County, CA, to the regulated area, consolidating the central coastal California distribution of the pathogen to include 14 counties. No other detections were made by sampling symptomatic plants from HAP genera in 4 years of terrestrial survey. However, in only 2 years of rhododendron leaf baiting (the first year being a limited pilot survey) the pathogen was detected in three streams in locations far removed from endemic areas. Two of these streams were in western Washington while the third was in Mississippi. The positive detections by stream baiting surveys exceeded those made by terrestrial vegetation surveys in half the time from 20 percent of the samples and 4 percent of the locations. This demonstrates the distinct advantage of stream baiting over terrestrial vegetation surveys for *P. ramorum* early detection.

While the pathogen was detected in streams outside regulated areas on the West coast, the source of inoculum could not be identified with certainty. Intensive surveys failed to detect infections in streamside plants. The proximity of confirmed positive woody ornamental nurseries to the positive streams suggests them as sources, but some detections were made after infected plants were removed and destroyed during the implementation of the USDA–APHIS confirmed nursery protocol. Possible inoculum sources include infected, but as yet undetected plants in the suspect nurseries, including crop as well as noncrop plants; soil-borne inoculum that was not eradicated during implementation of the confirmed nursery protocol; infected plants sold and planted in ornamental landscapes; and infected plants in or near the surveyed streams that escaped detection, perhaps even plants not yet known to be hosts. Monitoring of *P. ramorum*-positive streams and environs will continue until inoculum sources are identified and eradicated.

Overall, these results support the conclusion that *P. ramorum* is not widely established in U.S. forests, even in proximity to nurseries where the pathogen has been confirmed in infected woody ornamental stock located in high-risk areas where host type and climate are conducive to disease development. However, not all infected plants were intercepted at the nurseries, and it is certain that some are planted in residential and commercial landscapes in high-risk areas. Introductions from infested West coast nurseries have continued annually from 2003 through 2007. The length of latent periods between introduction, establishment in native vegetation, and detection are unknown. These facts dictate continued early detection efforts to maximize the efficacy of eradication of new outbreaks.

Acknowledgments

We extend thanks to the forest health specialists in State forestry agencies for execution of the field survey; to State departments of agriculture and universities for local diagnostic laboratory services; and to regional forest health monitoring coordinators James Steinman, Manfred Mielke, Jeri Lynn Harris, James Brown, and Alison Nelson for logistical support of training and reporting. Special thanks are extended to Susan Diehl, Mississippi State University, for eastern regional laboratory diagnostics and Melodie Putnam, Oregon State University, for West coast regional laboratory diagnostics.

Literature Cited

Brasier, C.M.; Denman, S.; Rose, J. [and others]. 2004. First report of ramorum bleeding canker on *Quercus falcata*, caused by *Phytophthora ramorum*. Plant Pathology. 53: 804.

California Oak Mortality Task Force. 2009. Nursery chronology. http://www.suddenoakdeath.org/html/chronology.html. [Date accessed: Unknown].

DeVries, R.M.; Levy, L. 2006. Quantitative multiplex real-time PCR (qPCR) for detection of *Phytophthora ramorum* using a TaqMan System on the Cepheid SmartCycler® and the ABI 7900/7000. USDA–APHIS plant health; pest information; *Phytophthora ramorum*; real-time PCR protocol. Rev. 4, April 3, 2006. 11 p. http://www.aphis.usda.gov/plant_health/plant_pest_info/pram/downloads/pdf_files/WI-B-T-1-6Rev4.pdf. [Date accessed: Unknown].

Hansen, E.M; Sutton, W. 2006. Persistence of *Phytophthora ramorum* after eradication efforts in Oregon tanoak forests [Poster]. In: Brasier, Clive; Jung, Thomas; Oßwald, W., eds. Proceedings of the 3rd international IUFRO working party 07.02.09 meeting. Farnham, UK: Forest Research. P. 178.

Klotz, L.J.; Wong, P.P.; DeWolfe, T.A. 1959. Survey of irrigation water for the presence of *Phytophthora* spp. pathogenic to citrus. Plant Disease Reporter. 43: 830–832.

Levy, L.; Mavrodieva, V. 2004. PCR detection and dna isolation methods for use in the *Phytophthora ramorum* national program. USDA–APHIS. plant health; pest information; *Phytophthora ramorum*; nested PCR protocol. 11 p. http://www.aphis.usda.gov/plant_health/plant_pest_info/pram/downloads/pdf_files/pcrprotocol4.pdf. [Date accessed: Unknown].

McIntosh, D.L. 1966. The occurrence of *Phytophthora* spp. in irrigation systems in British Columbia. Canadian Journal of Botany. 44: 1591–1596.

Murphy, S.K.; Lee, C.; Bienapfl, J. [and others]. 2006 Monitoring *Phytophthora ramorum* distribution in streams within coastal California watersheds [Poster]. In: Frankel, S.J.; Shea, P.J.; Haverty, M.I., tech. coords. Proceedings of the sudden oak death second science symposium: the state of our knowledge. Gen. Tech. Rep. PSW–GTR–196. Albany, CA: Forest Service, U.S. Department of Agriculture, Pacific Southwest Research Station: 531.

Oak, S.W.; Smith, W.D.; Tkacz, B.M. 2006. *Phytophthora ramorum* detection surveys for forests in the United States. In: Brasier, Clive; Jung, Thomas; Oßwald, W., eds. Proceedings of the 3rd international IUFRO working party 07.02.09 meeting. Farnham, UK: Forest Research: 28–30.

Oak, S.W.; Elledge, A.H.; Yockey, E.K. [and others] 2008a. *Phytophthora ramorum* early detection surveys for forests in the United States, 2003–2006. In: Frankel, Susan J.; Kliejunas, John T.; Palmieri, Katherine M., tech. coords. Proceedings of the sudden oak death third science symposium. Gen. Tech. Rep. PSW–GTR–214. Albany, CA: Forest Service, U.S. Department of Agriculture, Pacific Southwest Research Station: 413–416.

Oak, S.W.; Hwang, J.; Jeffers, S.N.; Tkacz, B.M. 2008b. 2006 pilot survey for *Phytophthora ramorum* in forest streams in the USA. In: Frankel, Susan J.; Kliejunas, John T.; Palmieri, Katherine M., tech. coords. Proceedings of the sudden oak death third science symposium. Gen. Tech. Rep. PSW–GTR–214. Albany, CA: Forest Service, U.S. Department of Agriculture, Pacific Southwest Research Station: 59–64.

Robertson, G.I. 1975. A paper disc technique for the recovery of *Pythium* spp. from soil or water. New Zealand Journal of Agricultural Research. 18: 409–410.

Thompson, S.V.; Allen, R.M. 1974. Occurrence of *Phytophthora* species and other potential plant pathogens in recycled irrigation water. Plant Disease Reporter. 57: 391–393.

Tjosvold, S.A.; Chambers, D.L.; Koike, S.; Fichtner, E. 2006. Epidemiology of *Phytophthora ramorum* infecting rhododendrons under simulated nursery conditions. In: Frankel, Susan J.; Shea, Patrick J.; Haverty, Michael I., tech. coords. Proceedings of the sudden oak death second science symposium: the state of our knowledge. Gen. Tech. Rep. PSW–GTR–196. Albany, CA: Forest Service, U.S. Department of Agriculture, Pacific Southwest Research Station: 459–461.

Tooley, P.W.; Kyde, K.L. 2007. Susceptibility of some eastern forest species to *Phytophthora ramorum*. Plant Disease. 91: 435–438.

U.S. Department of Agriculture, Animal and Plant Health Inspection Service. 2007. Plant health; pest information; *Phytophthora ramorum*; http://www.aphis.usda.gov/plant_health/plant_pest_info/pram/index.shtml. [Date accessed: Unknown].

von Broembsen, S.L. 1984. Distribution of *Phytophthora cinnamomi* in rivers of the Southwestern Cape Province. Phytophylactica. 16: 227–229.

Chapter 9. Summary

Kevin M. Potter

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 National Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture, with its cooperating partners within and outside the Forest Service, quantifies the health of U.S. forests within the context of the sustainable forest management criteria and indicators outlined in the Santiago Declaration on the conservation and sustainable management of temporal and boreal forests (Anon. 1995). The analyses and results outlined in this report offer a snapshot of the current condition of U.S. forests, incorporating baseline investigations of forest ecosystem health (chapters 2, 3, and 4), examination of change over time in forest health metrics (chapters 5 and 6), and the assessment of developing threats to forest stability and sustainability (chapters 7 and 8).

Understanding the landscape-scale impacts of forest fragmentation requires quantifying the context in which natural landcover exists (chapter 2). A newly developed landcover context model classifies every 0.09-ha parcel of land in the conterminous United States based on the surrounding natural, developed, and agricultural landcover. Using this approach, at least half of the area of the conterminous United States can be characterized as a natural background for scales ranging from several hectares to several hundred hectares. Agricultural and developed landcover backgrounds are important locally but typically occupy < 30 percent of the total area at regional scales. At local scales, forest and grassland tend to occur in contexts dominated by natural landcover types. The same is the case at regional scales in the West but not in the East, where more forest occurs within agricultural and developed contexts.

Lichen species diversity provides a promising metric for quantifying the effects of air pollution and other environmental variables on forest ecosystems (chapter 3). Lichen community data, collected since 1994, are relatively insensitive to local variation in forest structure and more related to large-scale variables in most areas. Specifically, lichen diversity in the East appears to be negatively influenced by deposition of NO_3^- and thus may be a useful indicator of forest response to air pollution and other human impacts, but it may not be a useful indicator for response to changes in climate in the East. In the West, lichen diversity is correlated with environmental variables, such as precipitation, latitude, and elevation. If interpreted carefully, however, lichen diversity may also serve as a useful indicator for pollution and climate response in the West.

Biodiversity is a central component of forest health. Like the lichen community data, data for the vegetation diversity and structure indicator from the Forest Inventory and Analysis (FIA) Program of the Forest Service are collected on phase 3 (P3) plots, providing important

baseline information on the composition of forest vascular plants, including the presence of exotic species (chapter 4). In general, forests in cool and moist areas support more plant species than those in hotter and drier climates. Sixty-seven percent of the plots contain at least one introduced plant with multiflora rose, cheatgrass, and Japanese honeysuckle being the most common exotics. The P3 vegetation data will be a powerful tool for assessing change when plots are revisited.

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. FIA phase 2 data offer tree mortality information at a more spatially intense sample than the P3 data used in past forest health annual technical reports (chapter 5). A test of this approach for six Midwestern States found that the areas of highest mortality relative to growth were in the Loess Hills of Missouri and in northern Minnesota. Much of the mortality in Missouri may be the result of American elm loss to Dutch elm disease, while the high mortality of quaking aspen in northern Minnesota may be attributable to a number of causes.

Several plant species are sensitive to elevated ozone levels, which can injure trees and make them more susceptible to pest and pathogen infestation. The Forest Service national ozone biomonitoring program aims to identify areas where the risk of ozone is high and the forest community is sensitive (chapter 6). Between 2003 and 2005, ozone injury to bioindicator plants was recorded in every FIA region except the Interior West, with injury occurrence relatively constant in the North and Pacific Northwest but decreasing in the South. While statistically significant logistic models were developed for each FIA region where ozone injury occurred on biomonitoring sites, logistic regression provided little improvement, in respect to predictive power, over standard spatial interpolation techniques. However, the results from analysis provide direction for future research.

A developing threat to forest health is laurel wilt, a disease causing widespread mortality in southeastern coastal redbay stands (chapter 7). Laurel wilt is caused by a fungus associated with the exotic redbay ambrosia beetle, which has spread to coastal areas of Georgia, South

Carolina, and Florida after its initial detection in 2002. The climatically suitable area for the beetle is largely constrained to the coastal plain of the southeastern United States. Based on host species distributions, climate matching, and spread modeling, the infestation is likely to expand north and south along the Atlantic coast and west along the gulf coast, with the entire range of redbay at approximately equal risk of infestation.

Sudden oak death, a more established forest tree disease first detected in the mid-1990s, remains limited to northern California and Oregon but may pose a considerable threat to other regions, especially the oak-dominated forests of the Eastern United States (chapter 8). Terrestrial vegetation survey early detection efforts in 39 States between 2003 and 2006 documented only one previously unknown occurrence of *Phytophthora ramorum* in California, suggesting that it is not yet widely established in U.S. forests. However, it is likely that infected landscape plants have been planted in high-risk areas. Stream baiting surveys conducted in 2007 in 152 streams across 28 States detected the pathogen in two streams not previously known to harbor the pathogen, near woody plant nurseries in Washington and Mississippi. Stream baiting for the pathogen appears to be an efficient approach for early detection surveys.

This report presents results from the ongoing national scale detection monitoring efforts from FHM and its cooperators using a wide variety of regional-scale data and analysis techniques. 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 http://fhm.fs.fed.us. This FHM national technical report is produced by the National Forest Health Monitoring Research Team, which is a part of the Eastern Forest Environmental Threat Assessment Center established under the Healthy Forest Restoration Act as part of a nationwide network of early warning activities about threats to forest health. For more information about the research team and about threats to U.S. forests, please visit www.forestthreats.org/about.

Literature Cited

Anon. 1995. Criteria and indicators for the conservation and sustainable management of temperate and boreal forests. In: Sustaining the world's forests, the Santiago Declaration. www.fs.fed.us/global/pub/links/santiago.htm. [Date accessed: August 28, 2007].

Costanza, R. 1992. Toward an operational definition of ecosystem health. In: Costanza, R.; Norton, B.G.; Haskell, B.D., eds. Ecosystem health: new goals for environmental management. Washington, DC: Island Press: 239–256.

Acknowledgments

This research was supported in part through the project "Forest Health Assessment" of cooperative agreement SRS 05–CA–11330146–180, project "Forest Health Monitoring, Analysis and Assessment" of research joint venture agreement 06–JV–11330146–124, project "Modeling Uncertainty in Forest Ecosystem Risk Assessments" of research joint venture agreement 06–JV–11330146–123, project "Forest Health Monitoring and Assessment" of research joint venture agreement 07–JV–11330146–134, project "Forest Health Monitoring and Assessment" of research joint venture agreement 08–JV–11330146–078, project "Forest Health Monitoring, Assessment and Analysis" of research joint venture agreement 09–JV–11330146–087, and project "Forest Health Monitoring, Analysis, and Assessment of joint research venture 10-JV-11330146-064 between North Carolina State University [in accordance with Federal law and U.S. Department of Agriculture policy, this institution is prohibited from discriminating on the basis of race, color, national origin, sex, age, or disability. (Not all prohibited bases apply to all programs.) To file a complaint of discrimination, write USDA, Director, Office of Civil Rights, Room 326–W, Whitten Building, 1400 Independence Avenue, SW, Washington, DC 20250–9410 or call 202–720–5964 (voice and TDD)]. USDA is an equal opportunity provider and employer. This research was supported in part by cooperative agreement SRS 06–CA–11220145–079 with the University of Wisconsin-Madison [in accordance with Federal law and U.S. Department of Agriculture policy, this institution is prohibited from discriminating on the basis of race, color, national origin, sex, age, or disability. (Not all prohibited bases apply to all programs.) To file a complaint of discrimination, write USDA, Director, Office of Civil Rights, Room 326–W, Whitten Building, 1400 Independence Avenue, SW, Washington, DC 20250–9410 or call 202–720–5964 (voice and TDD)]. USDA is an equal opportunity provider and employer. This research was supported by funds provided by the Forest Service, U.S. Department of Agriculture, Southern Research Station, Asheville, NC.

The authors thank the following for their constructive comments: Bruce Moltzen, Robert Rabaglia, Dale Starkey, Borys Tkacz, Stanley Zarnoch, and an additional anonymous reviewer. Additional thanks to Mark Ambrose for his general assistance with preparation of this report.

Author Information

Mark J. Ambrose, Research Assistant, 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.

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

Frank H. Koch, Research Ecologist, Forest Service, U.S. Department of Agriculture, 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).

Randall S. Morin, Research Forester, Forest Service, U.S. Department of Agriculture, Northern Research Station, Newtown Square, PA 19073.

Steven Oak, Pathologist, Forest Service, U.S. Department of Agriculture, Region 8, Asheville, NC 28804.

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

Kurt H. Riitters, Research Scientist, Forest Service, U.S. Department of Agriculture, Southern Research Station, Research Triangle Park, NC 27709.

Bethany Schulz, Research Ecologist/National Vegetation Indicator Advisor, Forest Service, U.S. Department of Agriculture, Pacific Northwest Research Station, Anchorage, AK 99503.

William D. Smith, Research Scientist, Forest Service, U.S. Department of Agriculture, Southern Research Station, Research Triangle Park, NC 27709.

Susan Will-Wolf, Senior Scientist, University of Wisconsin, Department of Botany, Madison, WI 53706-1381.

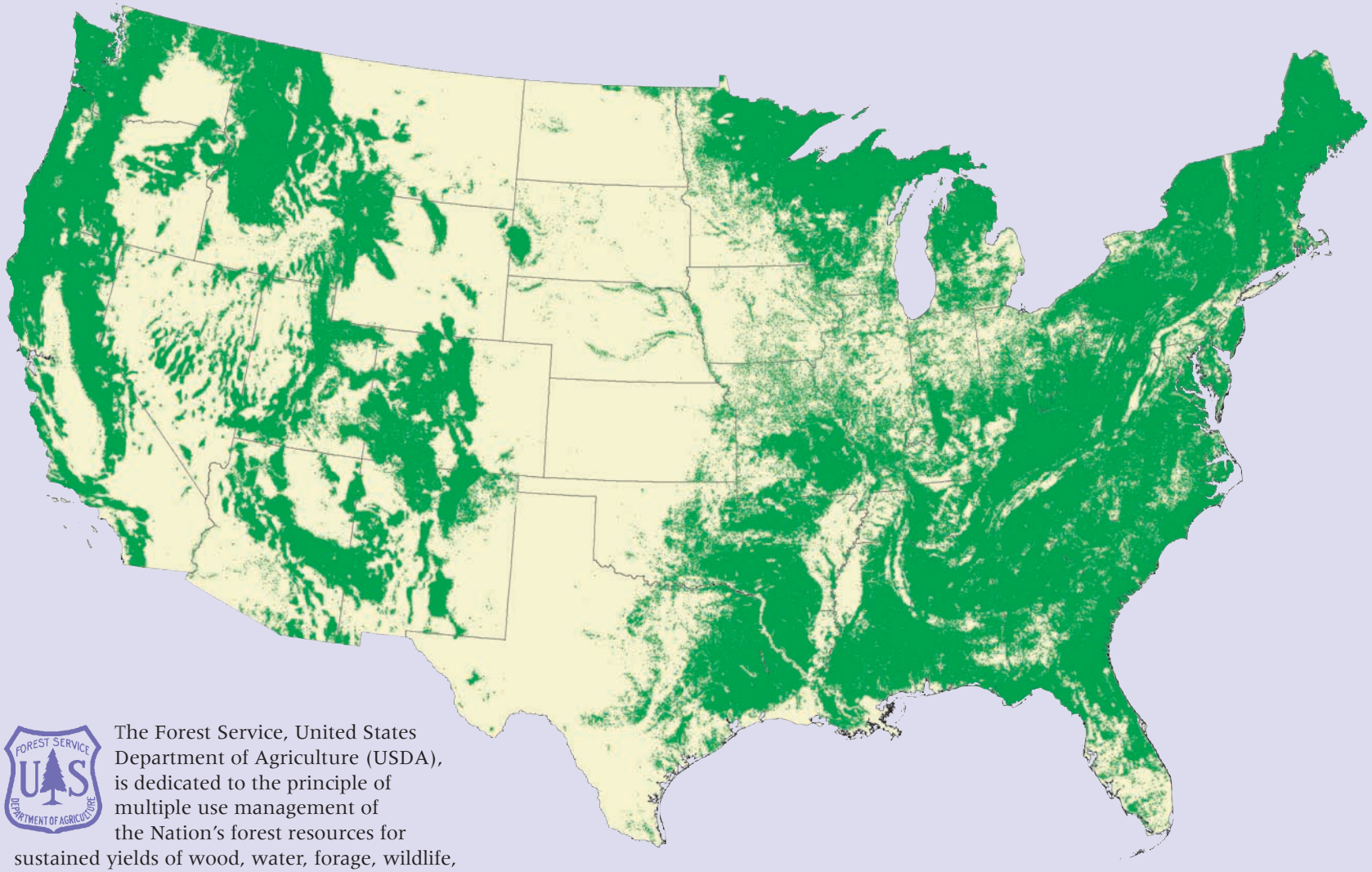
Conkling, Barbara L., ed. 2011. Forest health monitoring: 2007 national technical report. Gen. Tech. Rep. SRS-147. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 159 p.

The Forest Health Monitoring Program produces an annual technical report that has two main objectives. The first objective is to present information about forest health from a national perspective. The second objective is to present examples of useful techniques for analyzing forest health data new to the annual national reports and new applications of techniques formerly used. The report's organizational framework is the Criteria and Indicators for the Conservation and Sustainable Management of Temperate and Boreal Forests of the Montreal Process. Here, we present an approach to examining landscape context of forest and grassland in the United States. We explore the influence of environmental factors such as climate and air quality on a lichen species diversity indicator across the continental United States. This includes an analysis of the potential for monitoring changes in these environmental factors. We use Forest Inventory and Analysis phase 3 data to describe aspects of forest communities such as understory species composition, richness, and distribution, including discussion of invasive and introduced species. Tree mortality, which has been examined in previous Forest Health Monitoring reports, is analyzed in this report using a more intensive dataset to demonstrate the utility of Forest Inventory and Analysis phase 2 data. We explore spatial modeling of ozone injury risk, along with microscale and landscape-scale ancillary data that can be used in the modeling analyses. A discussion of redbay ambrosia beetle/laurel wilt risk includes current beetle/wilt distribution, host species distributions, climate matching, and spread modeling. Progress in monitoring and analyses of *Phytophthora ramorum* and sudden oak death is presented along with results from two different monitoring techniques.

Keywords—Air pollution, biodiversity, Criteria and Indicators, invasive insects and pathogens, landscape lichens, ozone.

DISCLAIMER

The use of trade or firm names in this publication is for reader information and does not imply endorsement by the U.S. Department of Agriculture of any product or service.



The Forest Service, United States Department of Agriculture (USDA), is dedicated to the principle of multiple use management of the Nation's forest resources for sustained yields of wood, water, forage, wildlife, and recreation. Through forestry research, cooperation with the States and private forest owners, and management of the National Forests and National Grasslands, it strives—as directed by Congress—to provide increasingly greater service to a growing Nation.

The USDA prohibits discrimination in all its programs and activities on the basis of race, color, national origin, age, disability, and where applicable, sex, marital status, familial status, parental status, religion, sexual orientation, genetic information, political beliefs, reprisal, or because all or part of an individual's income is derived from any public assistance program. (Not all prohibited bases apply to all programs.) Persons with disabilities who require alternative means for communication of program information (Braille, large print, audiotape, etc.) should contact USDA's TARGET Center at (202) 720-2600 (voice and TDD).

To file a complaint of discrimination, write to USDA, Director, Office of Civil Rights, 1400 Independence Avenue, SW, Washington, D.C. 20250-9410, or call (800) 795-3272 (voice) or (202) 720-6382 (TDD). USDA is an equal opportunity provider and employer.