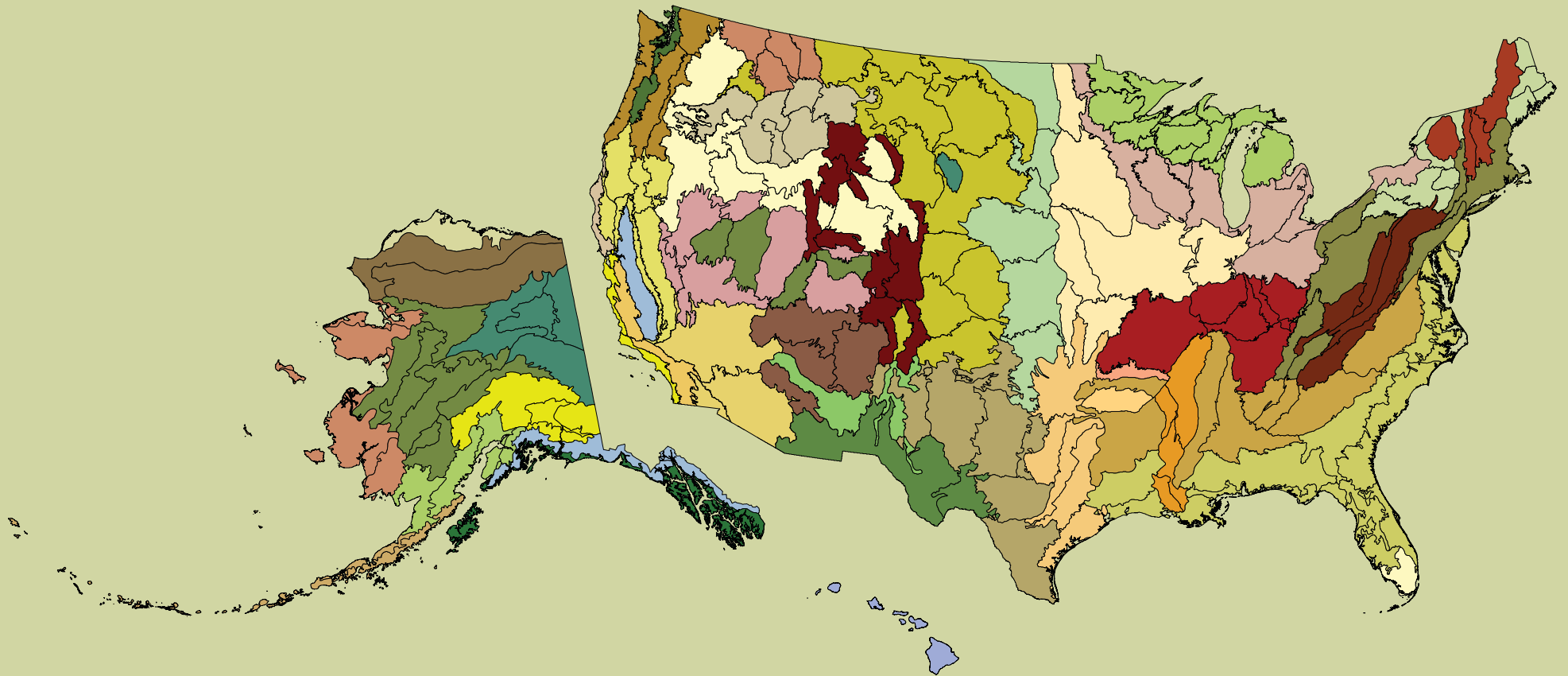




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# Forest Health Monitoring: National Status, Trends, and Analysis 2016

Editors Kevin M. Potter Barbara L. Conkling



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

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

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# Forest Health Monitoring: National Status, Trends, and Analysis 2016

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# ABSTRACT

The annual national report of the Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture, presents forest health status and trends from a national or multi-State regional perspective using a variety of sources, introduces new techniques for analyzing forest health data, and summarizes results of recently completed Evaluation Monitoring projects funded through the FHM national program. In this 16th edition in a series of annual reports, survey data are used to identify geographic patterns of insect and disease activity. Satellite data are employed to detect geographic patterns of forest fire occurrence. Recent drought and moisture surplus conditions are compared across the conterminous United States. Data collected by the Forest Inventory and Analysis (FIA) Program are employed to detect regional differences

in tree mortality. Change over time in the understory Vegetation Diversity and Structure Indicator is assessed on more than 500 FIA plots in the North Central and Northeastern States, where remeasured vegetation data are used to assess change over time in plant species diversity, occupancy and constancy. A new Regeneration Indicator, which includes a suite of tree-seedling and browse impact measurements, is described. The general magnitude of tree mortality predicted by the National Insect and Disease Risk Map is compared to FIA estimates of mortality. Six recently completed Evaluation Monitoring projects are summarized, addressing forest health concerns at smaller scales.

**Keywords**—Change detection, drought, fire, forest health, forest insects and disease, regeneration, tree mortality.

|                                |      |
|--------------------------------|------|
| LIST OF FIGURES . . . . .      | viii |
| LIST OF TABLES . . . . .       | xvi  |
| EXECUTIVE SUMMARY . . . . .    | 1    |
| LITERATURE CITED . . . . .     | 4    |
| <br><b>CHAPTER 1.</b>          |      |
| Introduction . . . . .         | 5    |
| KEVIN M. POTTER                |      |
| THE FOREST HEALTH              |      |
| MONITORING PROGRAM . . . . .   | 7    |
| DATA SOURCES. . . . .          | 11   |
| FHM REPORT PRODUCTION. . . . . | 13   |
| LITERATURE CITED . . . . .     | 13   |

## SECTION 1.

|                                                   |    |
|---------------------------------------------------|----|
| Analyses of Short-Term Forest Health Data . . . . | 19 |
|---------------------------------------------------|----|

## CHAPTER 2.

|                                                                                                                                                                                 |    |
|---------------------------------------------------------------------------------------------------------------------------------------------------------------------------------|----|
| Large-Scale Patterns of Insect and Disease<br>Activity in the Conterminous United States,<br>Alaska, and Hawai'i from the National Insect and<br>Disease Survey, 2015 . . . . . | 21 |
|---------------------------------------------------------------------------------------------------------------------------------------------------------------------------------|----|

KEVIN M. POTTER AND JEANINE L. PASCHKE

|                                 |    |
|---------------------------------|----|
| INTRODUCTION . . . . .          | 21 |
| METHODS. . . . .                | 21 |
| RESULTS AND DISCUSSION. . . . . | 26 |
| CONCLUSION . . . . .            | 40 |
| LITERATURE CITED . . . . .      | 40 |

## CHAPTER 3.

|                                                                                                                             |    |
|-----------------------------------------------------------------------------------------------------------------------------|----|
| Large-Scale Patterns of Forest Fire Occurrence<br>in the Conterminous United States, Alaska, and<br>Hawai'i, 2015 . . . . . | 43 |
|-----------------------------------------------------------------------------------------------------------------------------|----|

KEVIN M. POTTER

|                                 |    |
|---------------------------------|----|
| INTRODUCTION . . . . .          | 43 |
| METHODS. . . . .                | 44 |
| RESULTS AND DISCUSSION. . . . . | 47 |
| CONCLUSION . . . . .            | 58 |
| LITERATURE CITED . . . . .      | 60 |

# CONTENTS

Contents, cont.

**CHAPTER 4.**  
Moisture Deficit and Surplus in the Conterminous  
United States for Three Time Windows: 2015,  
2013–2015, and 2011–2015. . . . . 63

FRANK H. KOCH AND JOHN W. COULSTON

INTRODUCTION . . . . . 63  
METHODS. . . . . 64  
RESULTS AND DISCUSSION. . . . . 69  
LITERATURE CITED . . . . . 78

**CHAPTER 5.**  
Tree Mortality . . . . . 81

MARK J. AMBROSE

INTRODUCTION . . . . . 81  
DATA. . . . . 81  
METHODS. . . . . 81  
RESULTS AND DISCUSSION. . . . . 83  
SUMMARY . . . . . 88  
LITERATURE CITED . . . . . 89

**SECTION 2.**

Analyses of Long-Term Forest Health Trends  
and Presentations of New Techniques. . . . . 91

**CHAPTER 6.**  
Changes on Vegetation Indicator Plots: Small  
Sample Reveals Significant Trends . . . . . 93

BETHANY SCHULZ

INTRODUCTION . . . . . 93  
METHODS. . . . . 94  
RESULTS AND DISCUSSON . . . . . 98  
CONCLUSIONS . . . . . 105  
LITERATURE CITED . . . . . 106

**CHAPTER 7.**  
Forest Regeneration Indicator Results for the  
Midwest and Northeast United States. . . . . 109

WILLIAM H. McWILLIAMS, JAMES A. WESTFALL,  
PATRICK H. BROSE, DAN C. DEY, KENNETH L.  
LAUSTSEN, SHAWN L. LEHMAN, RANDALL S. MORIN,  
TODD E. RISTAU, ALEJANDRO A. ROYO, ANDREW M.  
STOLTMAN, AND SUSAN L. STOUT

INTRODUCTION . . . . . 109  
METHODS. . . . . 110  
RESULTS AND DISCUSSON . . . . . 113  
SUMMARY . . . . . 121  
LITERATURE CITED . . . . . 122

## CHAPTER 8.

|                                                                                                                                  |     |
|----------------------------------------------------------------------------------------------------------------------------------|-----|
| Validating the National Insect and Disease Risk<br>Map using Mortality Estimates from Forest<br>Inventory and Analysis . . . . . | 125 |
|----------------------------------------------------------------------------------------------------------------------------------|-----|

FRANK J. KRIST, JR., ANDREW McMAHAN, AND  
WILLIAM B. MONAHAN

|                                  |     |
|----------------------------------|-----|
| INTRODUCTION . . . . .           | 125 |
| METHODS . . . . .                | 127 |
| RESULTS AND DISCUSSION . . . . . | 131 |
| ACKNOWLEDGMENTS . . . . .        | 143 |
| LITERATURE CITED . . . . .       | 143 |

## SECTION 3.

|                                                 |     |
|-------------------------------------------------|-----|
| Evaluation Monitoring Project Summaries . . . . | 145 |
| LITERATURE CITED . . . . .                      | 145 |

## CHAPTER 9.

|                                                                                                                                          |     |
|------------------------------------------------------------------------------------------------------------------------------------------|-----|
| Impact of Southern Pine Beetle on Forest<br>Structure and Fuel Loading in a Wildfire-prone<br>Landscape (Project NE-EM-F-13-01). . . . . | 147 |
|------------------------------------------------------------------------------------------------------------------------------------------|-----|

KENNETH L. CLARK, MATTHEW AYRES, CARISSA AOKI,  
EMILY WENGROWSKI, AND JANE PETERKEN

|                              |     |
|------------------------------|-----|
| INTRODUCTION . . . . .       | 147 |
| METHODS . . . . .            | 148 |
| RESULTS . . . . .            | 148 |
| DISCUSSION. . . . .          | 151 |
| CONCLUSIONS . . . . .        | 153 |
| CONTACT INFORMATION. . . . . | 153 |
| LITERATURE CITED . . . . .   | 153 |

*Contents, cont.*

*Contents, cont.*

## CHAPTER 10.

### Assessing Southern Pine Beetle Risk in the New Jersey Pinelands (Project NE-EM-B-12-01) . . . . . 155

CARISSA F. AOKI, LYNN FLEMING, JAMES DUNN,  
DAVID FINLEY, AND MATTHEW P. AYRES

|                                      |     |
|--------------------------------------|-----|
| INTRODUCTION . . . . .               | 155 |
| METHODS . . . . .                    | 156 |
| RESULTS . . . . .                    | 158 |
| DISCUSSION AND CONCLUSIONS . . . . . | 160 |
| CONTACT INFORMATION . . . . .        | 161 |
| LITERATURE CITED . . . . .           | 161 |

## CHAPTER 11.

### Lichen Responses to Experimental Temperature and CO<sub>2</sub> Enhancement: Implications for Climate Change Monitoring (Project NC-EM-B-14-01). . . . . 163

ROBERT J. SMITH, SARAH JOVAN, AND  
BRUCE McCUNE

|                               |     |
|-------------------------------|-----|
| INTRODUCTION . . . . .        | 163 |
| METHODS . . . . .             | 163 |
| RESULTS . . . . .             | 165 |
| DISCUSSION. . . . .           | 167 |
| ACKNOWLEDGMENTS . . . . .     | 167 |
| CONTACT INFORMATION . . . . . | 167 |
| LITERATURE CITED . . . . .    | 167 |

## CHAPTER 12.

### Interactions Among Abiotic and Biotic Factors Affecting the Health of Pine Forests in the Southeastern United States (Project SO-EM-B-12-03). . . . . 169

KAMAL J.K. GANDHI, BRITTANY F. BARNES,  
DAVID R. COYLE, KIER D. KLEPZIG, FRANK H. KOCH,  
LAWRENCE MORRIS, JOHN T. NOWAK,  
WILLIAM J. OTROSINA, AND WILLIAM D. SMITH

|                                 |     |
|---------------------------------|-----|
| INTRODUCTION . . . . .          | 169 |
| METHODS . . . . .               | 170 |
| RESULTS AND DISCUSSION. . . . . | 171 |
| ACKNOWLEDGMENTS . . . . .       | 174 |
| CONTACT INFORMATION . . . . .   | 174 |
| LITERATURE CITED . . . . .      | 174 |

## CHAPTER 13.

### Tree Mortality Estimates and Species Distribution Probabilities in Southeastern United States Forests (Project SO-EM-F-10-01). . . . . 175

MARTIN A. SPETICH, ZHAOFEI FAN, ZHEN SUI,  
MICHAEL CROSBY, HONG S. HE, STEPHEN R. SHIFLEY,  
THEODOR D. LEININGER, AND W. KEITH MOSER

|                               |     |
|-------------------------------|-----|
| INTRODUCTION . . . . .        | 175 |
| METHODS . . . . .             | 176 |
| RESULTS . . . . .             | 177 |
| DISCUSSION. . . . .           | 178 |
| CONCLUSIONS . . . . .         | 180 |
| CONTACT INFORMATION . . . . . | 180 |
| LITERATURE CITED . . . . .    | 181 |

## CHAPTER 14.

Detection and Impact of an Invasive Insect-  
Disease Complex, the Polyphagous and  
Kuroshio Shot Hole Borers, *Euwallacea* spp.,  
and Their Mutualistic Fungi in California  
(Project WC-EM-F-13-02) . . . . . 183

FRANCIS NA, KAMERON SUGINO, DANNY H. WANG,  
JOSEPH D. CARRILLO, JOEY S. MAYORQUIN,  
SHANNON C. LYNCH, AND AKIF ESKALEN

INTRODUCTION . . . . . 183  
METHODS . . . . . 184  
RESULTS AND DISCUSSION . . . . . 185  
CONCLUSIONS . . . . . 188  
CONTACT INFORMATION . . . . . 189  
LITERATURE CITED . . . . . 189

ACKNOWLEDGMENTS. . . . . 190

AUTHOR INFORMATION . . . . . 191

*Contents, cont.*

# LIST OF FIGURES

*Figure 1.1*—Ecoregion provinces and sections for the conterminous United States (Cleland and others 2007) and Alaska (Nowacki and Brock 1995). Ecoregion sections within each ecoregion province are shown in the same color. Note: no equivalent ecoregion treatment exists for Hawai‘i. . . . . 8

*Figure 1.2*—The design of the Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture (FHM 2003). A fifth component, Analysis and Reporting of Results, draws from the four FHM components shown here and provides information to help support land management policies and decisions. . . . . 10

*Figure 1.3*—The Forest Inventory and Analysis mapped plot design. Subplot 1 is the center of the cluster with subplots 2, 3, and 4 located 120 feet away at azimuths of 360°, 120°, and 240°, respectively (Woudenberg and others 2010). . . . . 12

*Figure 2.1*—The extent of surveys for insect and disease activity conducted in the conterminous United States, Alaska, and Hawai‘i in 2015. Cross-hatched areas were surveyed using the new Digital Mobile Sketch Mapping (DMSM) platform, rather than the older Digital Aerial Sketch Mapping (DASM) approach, which is portrayed in green. The black lines delineate Forest Health Monitoring regions. Note: Alaska and Hawai‘i are not shown to scale with map of the conterminous United States. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection) . . . . . 23

*Figure 2.2*—Hot spots of exposure to mortality-causing insects and diseases in 2015. Values are Getis-Ord  $G_i^*$  scores, with values > 2 representing significant clustering of high percentages of forest area exposed to mortality agents. (No areas of significant clustering of low percentages of exposure, < -2, were detected.) The gray lines delineate ecoregion sections (Cleland and others 2007), and blue lines delineate Forest Health Monitoring regions. Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection) . . . . . 30

*Figure 2.3*—Hot spots of exposure to defoliation-causing insects and diseases in 2015. Values are Getis-Ord  $G_i^*$  scores, with values > 2 representing significant clustering of high percentages of forest area exposed to defoliation agents. (No areas of significant clustering of low percentages of exposure, < -2, were detected.) The gray lines delineate ecoregion sections (Cleland and others 2007), and blue lines delineate Forest Health Monitoring regions. Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection) . . . . . 34



*Figure 2.4*—Percentage of surveyed forest in Alaska ecoregion sections exposed to mortality-causing insects and diseases in 2015. The gray lines delineate ecoregion sections (Nowacki and Brock 1995). Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: U.S. Department of Agriculture, Forest Service, Forest Health Protection) . . . 37

*Figure 2.5*—Percentage of surveyed forest in Alaska ecoregion sections exposed to defoliation-causing insects and diseases in 2015. The gray lines delineate ecoregion sections (Nowacki and Brock 1995). Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: U.S. Department of Agriculture, Forest Service, Forest Health Protection) . . . 38

*Figure 2.6*—Percentage of surveyed forest on the Big Island of Hawai‘i exposed to mortality-causing rapid ‘ōhi‘a death in 2015, by county council district (State of Hawai‘i 2012). Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection) . . . 39

*Figure 3.1*—Forest fire occurrences detected by MODIS from 2001 through 2015 for the conterminous United States, Alaska, and Hawai‘i, and for the entire Nation combined. (Data source: U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center, in conjunction with the NASA MODIS Rapid Response group) . . . . 48

*Figure 3.2*—The number of forest fire occurrences, per 100 km<sup>2</sup> (10 000 hectares) of forested area, by ecoregion section within the conterminous 48 States, for 2015. The gray lines delineate ecoregion sections (Cleland and others 2007). Forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center, in conjunction with the NASA MODIS Rapid Response group) . . . . . 49

*Figure 3.3*—The number of forest fire occurrences, per 100 km<sup>2</sup> (10 000 hectares) of forested area, by ecoregion section within Alaska, for 2015. The gray lines delineate ecoregion sections (Nowacki and Brock 1995). Forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center, in conjunction with the NASA MODIS Rapid Response group) . . . . . 51

*Figures, cont.*

*Figures, cont.*

*Figure 3.4*—The number of forest fire occurrences, per 100 km<sup>2</sup> (10 000 ha) of forested area, by island in Hawai‘i, for 2015. Forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center, in conjunction with the NASA MODIS Rapid Response group) . . . . . 53

*Figure 3.5*—(A) Mean number and (B) standard deviation of forest fire occurrences per 100 km<sup>2</sup> (10 000 hectares) of forested area from 2001 through 2014, by ecoregion section within the conterminous 48 States. (C) Degree of 2015 fire occurrence density excess or deficiency by ecoregion relative to 2001–14 and accounting for variation over that time period. The dark lines delineate ecoregion sections (Cleland and others 2007). Forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center, in conjunction with the NASA MODIS Rapid Response group) . . . . . 54

*Figure 3.6*—(A) Mean number and (B) standard deviation of forest fire occurrences per 100 km<sup>2</sup> (10 000 ha) of forested area from 2001 through 2014, by ecoregion section in Alaska. (C) Degree of 2015 fire occurrence density excess or deficiency by ecoregion relative to 2001–14 and accounting for variation over that time period. The dark lines delineate ecoregion sections (Nowacki and Brock 1995). Forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center, in conjunction with the NASA MODIS Rapid Response group) . . . . . 56

*Figure 3.7*—(A) Mean number and (B) standard deviation of forest fire occurrences per 100 km<sup>2</sup> (10 000 ha) of forested area from 2001 through 2014, by island in Hawai‘i. (C) Degree of 2015 fire occurrence density excess or deficiency by island relative to 2001–14 and accounting for variation over that time period. Forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center, in conjunction with the NASA MODIS Rapid Response group) . . . . . 57

*Figure 3.8*— Hot spots of fire occurrence across the conterminous United States for 2015. Values are Getis-Ord  $G_i^*$  scores, with values  $> 2$  representing significant clustering of high numbers of fire occurrences. (No areas of significant clustering of low numbers of fire occurrences,  $< -2$ , were detected). The gray lines delineate ecoregion sections (Cleland and others 2007). Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center, in conjunction with the NASA MODIS Rapid Response group) . . . . . 59

*Figure 4.1*—The 100-year (1916–2015) mean annual moisture index, or  $MI'_{1norm}$ , for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries and labels are included for reference. Forest cover data (overlaid green hatching) derived from Moderate Resolution Imaging Spectroradiometer (MODIS) imagery by the U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center. (Data source: PRISM Climate Group, Oregon State University) . . . . . 68

*Figure 4.2*—The 2015 annual (i.e., 1-year) moisture difference z-score, or MDZ, for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries and labels are included for reference. Forest cover data (overlaid green hatching) derived from MODIS imagery by the U.S. Department of Agriculture, Forest Service,

Remote Sensing Applications Center. (Data source: PRISM Climate Group, Oregon State University) . . . . . 70

*Figure 4.3*—The 2014 annual (i.e., 1-year) moisture difference z-score, or MDZ, for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries and labels are included for reference. Forest cover data (overlaid green hatching) derived from MODIS imagery by the U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center. (Data source: PRISM Climate Group, Oregon State University) . . . . . 72

*Figure 4.4*—The 2013–2015 (i.e., 3-year) moisture difference z-score (MDZ) for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries are included for reference. Forest cover data (overlaid green hatching) derived from MODIS imagery by the U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center. (Data source: PRISM Climate Group, Oregon State University). . . 74

*Figure 4.5*—The 2011–2015 (i.e., 5-year) moisture difference z-score (MDZ) for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries are included for reference. Forest cover data (overlaid green hatching) derived from MODIS imagery by the U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center. (Data source: PRISM Climate Group, Oregon State University). . . 75

*Figures, cont.*

*Figures, cont.*

*Figure 5.1*—Forest cover in the States where mortality was analyzed by ecoregion section (Cleland and others 2007). Forest cover was derived from MODIS satellite imagery (USDA Forest Service 2008). . . . . 82

*Figure 5.2*—Tree mortality expressed as the ratio of annual mortality volume to gross annual volume growth (MRATIO) by ecoregion section. Ecoregions with high MRATIOS are identified by section number (Cleland and others 2007). (Data source: USDA, Forest Service Forest Inventory and Analysis Program). . . . . 84

*Figure 6.1*—Approximate locations where the suite of Vegetation Indicator measurements were collected twice in the conterminous United States. . . . . 95

*Figure 6.2*—Constancy of introduced species by condition. . . . . 104

*Figure 7.1*—Distribution of forest land and dominant forest-type groups for the Midwest and Northeast United States, 2008 (Homer and others 2012, Ruefenacht and others 2008). . . . . 111

*Figure 7.2*—Distribution of Forest Inventory and Analysis (FIA) sample locations, Midwest and Northeast United States from 2012 to 2014, and Pennsylvania from 2009 to 2014. (All locations are approximate). . . . 114

*Figure 7.3*—(A) Number of seedlings by height class, (B) number of seedlings for species with at least 1 percent of the total number of seedlings, ranked by species, and (C) distribution of forested phase 2-plus

samples by the average number of seedlings per acre in West Virginia, 2012–2013. Error bars represent 1 standard error or a 68-percent confidence interval . . . . . 115

*Figure 7.4*—Percent of total number of seedlings and dominant/co-dominant live saplings and growing-stock trees on forest land for select species by species group and size in Missouri. Seedling estimates are for 2012–2013. Sapling and tree estimates are for 2009–2013. . . . . 116

*Figure 7.5*—(A) Percent of regeneration samples adequately stocked with advance regeneration for stands from 40- to 75-percent stocked with live trees by canopy species group in Pennsylvania for 2004, 2009, and 2014; and (B) percentage of regeneration samples adequately stocked with advance regeneration for stands from 40- to 75-percent stocked with live trees by commercial species group, also in Pennsylvania for 2004, 2009, and 2014. . . . 117

*Figure 7.6*—(A) Probability of exceeding low browse impact and (B) probability of exceeding moderate browse impact for Pennsylvania in 2014. . . . . 119

*Figure 7.7*—Distribution of forested phase 2-plus samples (A) by browse impact and ecological province (Cleland and others 2007), in Delaware (B), Maryland (C), New Jersey (D), from 2012 through 2014, and in Pennsylvania (E), from 2010 through 2014. Error bars represent 1 standard error or a 68-percent confidence interval. . . . . 120

Figure 8.1—Progression of the National Insect and Disease Risk Map (NIDRM) in recent history. . . . . 126

Figure 8.2—State-level National Insect and Disease Risk Map (NIDRM) (y-axis) versus Forest Inventory and Analysis (FIA) (x-axis) plots and linear regressions for (A) live basal area (BA) and (B-D) mortality BA scenario. The x-axis values for the three FIA mortality scenarios represent mortality from: (B) insect and disease only (scenario 1); (C) insect and disease and all of unknown/other (scenario 2); and (D) insect and disease and one-half of unknown/other (scenario 3). A few State points are labeled for reference. See table 8.2 for regression parameters. . . . 134

Figure 8.3—National Insect and Disease Risk Map (y-axis) versus Forest Inventory and Analysis scenario 3 (x-axis) plots for mortality basal area by (A) State, (B) section, (C) county, and (D) subsection. Dotted line is the linear regression fit. Solid lines bracket the + 50-percent (upper line) and -50-percent (lower line) residual zone. See table 8.2 for regression parameters.. . . . 135

Figure 8.4—Residuals (as a percent) from the National Insect and Disease Risk Map (NIDRM)– Forest Inventory and Analysis (FIA) mortality linear regressions by (A) State, (B) county, (C) ecosection, and (D) subsection. Maps depict areas where mortality predicted by NIDRM approximately agrees, over-estimates, or under-estimates relative to FIA.. . . . 137

Figure 8.5—Percentage of total basal area that is modeled as potential host for insects or diseases in the National Insect and Disease Risk Map (NIDRM), by (A) State, (B) county, (C) ecosection, and (D) subsection. Analysis restricted to FIA forest land. . . . . 138

Figure 8.6—Number of forest pests modeled in the National Insect and Disease Risk Map (NIDRM) by (A) State, (B) county, (C) ecosection, and (D) subsection. . . . . 140

Figure 9.1—Basal area of live trees separated into pines (*Pinus rigida*, *P. echinata*, *P. virginiana*), oaks (*Quercus alba*, *Q. prinus*, *Q. marlandica*, *Q. velutina*, *Q. stellata*, *Q. falcata*, *Q. bicolor*), and other hardwoods (*Acer rubra*, *Nyssa sylvatica*, *Carya glabra*, *Magnolia virginiana*, *Sassafras albidum*) in control, natural, and treated plots.. . . . 149

Figure 9.2—Basal area of live saplings separated into pines (*Pinus rigida*, *P. echinata*, *P. virginiana*), oaks (*Quercus alba*, *Q. prinus*, *Q. marlandica*, *Q. velutina*, *Q. stellata*, *Q. falcata*, *Q. bicolor*), and other hardwoods (*Acer rubra*, *Nyssa sylvatica*, *Carya glabra*, *Magnolia virginiana*, *Sassafras albidum*) in control, natural, and treated plots.. . . . 151

Figure 10.1—(A) Comparison of land area in each forest type with number of spots in each forest type; data from southern Pinelands only. (B) Log odds of number of spots per forest type, based on land area per forest type. . . . . 159

*Figures, cont.*

|                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                        |     |
|--------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------|-----|
| <i>Figures, cont.</i>                                                                                                                                                                                                                                                                                                                                                                                                                                                                                  |     |
| <i>Figure 10.2</i> —Means and standard errors of northern and southern region uninfested forest structure plots and southern region infested forest structure plots, in principal component space. Directions of major variables in relation to each component axis are also indicated. . . . .                                                                                                                                                                                                        | 159 |
| <i>Figure 10.3</i> —(A) Mean resin flow with standard error for each of six soil types. PD = poorly drained, MD = moderately well-drained, WD = well-drained. (B) Mean resin flow with standard error comparing upland vs. wetland sites. . . . .                                                                                                                                                                                                                                                      | 160 |
| <i>Figure 11.1</i> —Lichen index scores (“NMS 1” and “NMS 2”) and species richness for each Forest Inventory and Analysis (FIA) site in the Great Lakes and Midwest regions. Similar NMS scores among sites indicate similar lichen community compositions. Linking existing regional FIA plots to climate change effects at the Spruce and Peatland Responses Under Climatic and Environmental change (SPRUCE) experiment is a key goal. . . . .                                                      | 164 |
| <i>Figure 11.2</i> —Changes in epiphytic lichen species richness, Shannon diversity ( $H'$ ), and growth of <i>Evernia</i> transplants over the 2-year period prior to treatments at the Spruce and Peatland Responses Under Climatic and Environmental change (SPRUCE) experimental site. Means $\pm$ 95 percent CI for each treatment (units are $^{\circ}\text{C}$ above ambient). Shannon diversity is an index incorporating species evenness based on proportional abundance of species. . . . . | 166 |
| <i>Figure 12.1</i> —Typical dieback and mortality in a pine stand in Georgia. (Photo by: Kamal Gandhi, University of Georgia) . . . .                                                                                                                                                                                                                                                                                                                                                                  | 169 |
| <i>Figure 12.2</i> —Remeasured Forest Inventory and Analysis (FIA) plots in the loblolly pine and loblolly pine/hardwood forest types where (A) plots exhibiting negative net pine growth, labeled according to disturbance agent, and (B) locations of plots with negative net pine growth versus all other pine plots (adapted from Coyle and others 2015). Plot locations are approximate. . . . .                                                                                                  | 172 |
| <i>Figure 12.3</i> —Soil levels of calcium (Ca), potassium (K), magnesium (Mg), and manganese (Mn) in healthy and unhealthy (symptomatic) pine stands in Georgia (GA) and Alabama (AL). . . . .                                                                                                                                                                                                                                                                                                        | 173 |
| <i>Figure 12.4</i> —Basal area in healthy and unhealthy pine stands in Alabama and Georgia. . . . .                                                                                                                                                                                                                                                                                                                                                                                                    | 173 |
| <i>Figure 13.1</i> —The 1999–2010 proportion of dead basal area for the non-oak group, white oak group, and the red oak group. Vertical lines are standard error for dead basal area of each species group and year. Based on 6,997 Forest Inventory and Analysis plots across the Ozark Highlands. . . . .                                                                                                                                                                                            | 177 |
| <i>Figure 13.2</i> —Distribution probabilities for southern red oak for the three climate envelope models: BIOCLIM, GLM, and MaxEnt. . . . .                                                                                                                                                                                                                                                                                                                                                           | 178 |
| <i>Figure 13.3</i> —Distribution probabilities for water oak for the three climate envelope models: BIOCLIM, GLM, and MaxEnt. . . .                                                                                                                                                                                                                                                                                                                                                                    | 178 |

*Figure 13.4*—Distribution probabilities for post oak for the three climate envelope models: BIOCLIM, GLM, and MaxEnt. . . . 179

*Figure 13.5*—Distribution probabilities for red maple for the three climate envelope models; BIOCLIM, GLM and MaxEnt. . . . 179

*Figure 14.1*—Distribution of polyphagous shot hole borers (PSHB) and Kuroshio shot hole borers (KSHB) infestation in California (A) as of 2012, (B) as of 2013, (C) as of 2014, (D) as of 2015, and (E) as of 2016. Red circles indicate infestation locations of PSHB, and blue circles indicate infestation locations of KSHB. Painted areas (F) indicate the possible infestation zones of P/KSHB in California. . . 186

*Figure 14.2*—Percent inhibition of *Fusarium* spp. by endophytic bacterial isolates obtained from sycamore. . . . 187

*Figure 14.3*—Beetle mortality over time after spray application with *Beauveria bassiana*, *Metarhizium brunneum*, or 0.01 percent Triton-X100 solution (control) in laboratory conditions. . . . 188

*Figure 14.4*—Mean new beetle galleries over time in branch collars of spray-treated trees. Pruning wounds were applied with *Bacillus subtilis*, *Beauveria bassiana*, vegetable oil, water, or no treatment. . . . 188

*Figures, cont.*



# LIST OF TABLES

|                                                                                                                                                                                                            |    |
|------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------|----|
| <i>Table 2.1</i> —Mortality agents and complexes affecting more than 5000 ha in the conterminous United States during 2015 . . .                                                                           | 27 |
| <i>Table 2.2</i> — Beetle taxa included in the “western bark beetle” group. . . . .                                                                                                                        | 27 |
| <i>Table 2.3</i> —The top five mortality agents or complexes for each Forest Health Monitoring region, and for Alaska and Hawai‘i, in 2015 . . . . .                                                       | 28 |
| <i>Table 2.4</i> —Defoliation agents and complexes affecting more than 5000 ha in the conterminous United States in 2015 . . . . .                                                                         | 32 |
| <i>Table 2.5</i> —The top five defoliation agents or complexes for each Forest Health Monitoring region and for Alaska in 2015 . . . . .                                                                   | 33 |
| <i>Table 4.1</i> —Moisture difference z-score (MDZ) value ranges for nine wetness and drought categories, along with each category’s approximate theoretical frequency of occurrence . . . . .             | 69 |
| <i>Table 5.1</i> — Ecoregion sections having the highest mortality relative to growth (MRATIO); annual growth, mortality, and removal rates; and disturbances associated with areas of mortality . . . . . | 86 |

|                                                                                                                                                                                                                                                                               |     |
|-------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------|-----|
| <i>Table 6.1</i> —Plot pair numbers by State, types of changes recorded, and numbers of revisits within the recommended calendar window for 514 revisited VEG plots . . . . .                                                                                                 | 99  |
| <i>Table 6.2</i> —Number of plot pairs with two visits by ecological division, with total number of species identified, number of species recorded only once, and number of genus and unknown codes used for each visit period . . . . .                                      | 100 |
| <i>Table 6.3</i> —Number of intact plots with two visits by ecological division and average plot and subplot species richness for intact plots for each visit, with the Student’s t-test probability of average subplot species richness being equal at both visits . . . . . | 101 |
| <i>Table 6.4</i> —Scientific and common names of 23 species with probability of plot frequency being equal at both visits, as determined with McNemar’s test in at least one of the compared datasets, in two ecological divisions . . . . .                                  | 102 |
| <i>Table 6.5</i> —The 24 species with no change in frequency . . . . .                                                                                                                                                                                                        | 103 |



*Table 6.6*—Relative richness and cover of introduced species by fragmentation class for each visit period, and the 2-tailed probability of equal means, as determined by the paired Student’s t-test . . . . .104

*Table 8.1*—Forest pest and host tree species combinations modeled for this analysis in the Eastern United States . . . . .130

*Table 8.2*—Linear regression parameters for each area of interest (AOI), metric, and mortality scenario . . . . .133

*Table 9.1*—Structural characteristics of canopy and understory in control, natural infested, and treated plots . . . . .150

*Table 11.1*—Epiphytic lichen species abundance and frequency among 72 transects over a 2-year period prior to treatments at SPRUCE experimental site . . .165

*List of Tables, cont.*



# EXECUTIVE SUMMARY

**H**ealthy ecosystems are those that are stable and sustainable, able to maintain their organization and autonomy over time while remaining resilient to stress (Costanza 1992). Healthy forests are vital to our future (Edmonds and others 2011), and consistent, large-scale, and long-term monitoring of key indicators of forest health status, change, and trends is necessary to identify forest resources deteriorating across large regions (Riitters and Tkacz 2004). The Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture, with cooperating researchers within and outside the Forest Service and with State partners, quantifies status and trends in the health of U.S. forests (chapter 1). The analyses and results outlined in sections 1 and 2 of this FHM annual national report offer a snapshot of the current condition of U.S. forests from a national or multi-State regional perspective, incorporating baseline investigations of forest ecosystem health, examinations of change over time in forest health metrics, and assessments of developing threats to forest stability and sustainability. For datasets collected on an annual basis, analyses are presented from 2015 data. For datasets collected over several years, analyses are presented at a longer temporal scale. Finally, section 3 of this report presents summaries of results from recently completed Evaluation Monitoring (EM) projects that have been funded through the FHM national program to determine the extent, severity, and/or causes of specific forest health problems (FHM 2016).

Monitoring the occurrence of forest pest and pathogen outbreaks is important at regional scales because of the significant impact insects and disease can have on forest health across landscapes (chapter 2). National Insect and Disease Survey data collected in 2015 by the Forest Health Protection Program of the Forest Service and partners in State agencies identified 70 different mortality-causing agents and complexes on 2.12 million ha in the conterminous United States, and 58 defoliating agents and complexes on approximately 4.85 million ha. Geographic hot spots of forest mortality were associated with bark beetle infestations (mostly pinyon ips, fir engraver, western pine beetle, mountain pine beetle, and spruce beetle) in the West, and with emerald ash borer in the East. Hot spots of defoliation were associated with western spruce budworm in the West, and with yellow poplar weevil, loblolly pine sawfly, gypsy moth, winter moth, and spruce budworm. Mortality was recorded on a very small proportion of the surveyed area in Alaska. The most important defoliation agents in Alaska were aspen leafminer and willow leaf blotchminer. In Hawai'i, approximately 18 000 ha of mortality caused by rapid 'ōhi'a death was detected on the Big Island.

Forest fire occurrence outside the historic range of frequency and intensity can result in extensive economic and ecological impacts. The detection of regional patterns of fire occurrence density can allow for the identification of areas at greatest risk of significant impact and for

the selection of locations for more intensive analysis (chapter 3). In 2015, the number of satellite-detected forest fire occurrences recorded for the conterminous States was the lowest since 2011. Ecoregions in the interior of the Pacific Northwest States and near the coast of northwestern California and southeastern Oregon experienced the most fires per 100 km<sup>2</sup> of forested area. Geographic hot spots of high fire occurrence density were detected in these same areas. Ecoregions in the Pacific Northwest, the Great Lakes States, and the Northeast experienced greater fire occurrence density than normal compared to the 12-year mean and accounting for variability over time. Alaska experienced moderately high fire occurrence densities in the interior of the State, but low densities elsewhere. The Big Island of Hawai'i experienced very high fire occurrence density as a result of an ongoing volcanic eruption.

Most U.S. forests experience droughts, with varying degrees of intensity and duration between and within forest ecosystems. Arguably, the duration of a drought event is more critical than its intensity. A standardized drought and moisture surplus indexing approach was applied to monthly climate data from 2015 to map drought conditions and surplus moisture availability across the conterminous United States at a fine scale (chapter 4). Much of the Eastern and Central United States experienced near-normal or moderate surplus conditions in 2015, while the Pacific Coast and much of the northern Rocky Mountain region experienced at least moderate drought conditions with some areas suffering extreme drought, especially

northern California and northern Washington, Idaho, and Montana. Analyses of longer term (3-year and 5-year) conditions show a strong dichotomy between areas west of the Rocky Mountains, where most areas—and notably California—experienced drought conditions, and the remainder of the country, which generally experienced at least mild moisture surplus conditions.

Mortality is a natural process in all forested ecosystems, but high levels of mortality at large scales may indicate that the health of forests is declining. Phase 2 data collected by the Forest Inventory and Analysis (FIA) Program of the Forest Service offer tree mortality information on a relatively spatially intense basis of approximately 1 plot per 6,000 acres (chapter 5). An analysis of FIA plots from all the Central and Eastern States found that the highest ratios of annual mortality to gross growth occurred in ecoregion sections located in western South Dakota and Nebraska, in northwest Ohio and southeast Michigan, and in east Texas. In the first of these regions, much mortality was of ponderosa pine and of species in riparian areas such as eastern cottonwood, bur oak, and green ash. High ratios of mortality to growth in Ohio and Michigan were most likely associated with the loss of ash to emerald ash borer and possibly with the effects of oak wilt. In east Texas, mortality was high in oak-hickory and loblolly-shortleaf pine forests, where recent drought may have increased susceptibility to insects and diseases such as pine engraver beetles and oak wilt.

The Vegetation Diversity and Structure Indicator (VEG) was among the indicators designed by the FHM Program to provide data to describe current conditions of forest ecosystems and detect changes and trends over time. It entails a complete survey of vascular plant species on each forested subplot, including species identification and percentage of total canopy cover as a measure of abundance (chapter 6). VEG data were collected twice over time on more than 500 FIA plots in the North Central and Northeastern States, providing an opportunity to explore what has changed over the time between plot visits. Specifically, these data were used to describe the changes in physical setting or forest stand conditions recorded between visits on plots that may have influenced vegetation composition and growth. The number and types of changes that have occurred on the plots demonstrate the dynamic nature of the forests in the region. Despite the relatively limited sample size and early data collection challenges, the data indicate increasing species richness, increasing frequency of about half the species, and increasing occupancy and constancy of introduced species.

Young forests are pivotal in the forest life cycle because they set the future compositional and structural trajectories following stand initiation disturbance, and hence, the potential for sustaining future forest values (chapter 7). Young forests of the Midwest and Northeast United States have emerged as a policy and management issue because they are rare and difficult to regenerate. Ongoing concern over

an aging forest, dwindling young forest habitat, and regeneration problems resulted in the implementation of a Regeneration Indicator (RI), which was applied across 24 States by the Northern Research Station (NRS) Forest Inventory and Analysis (FIA) Program. The RI protocols include a suite of tree-seedling and browse impact measurements. This chapter presents examples that utilize State-level analyses that have been used in NRS-FIA comprehensive reports to summarize findings and issues. Future research needs are discussed and conclusions are drawn concerning how to best fill knowledge gaps on the status and condition of regeneration in the Midwest and Northeast United States.

The 2012 National Insect and Disease Risk Map (NIDRM) estimates potential tree mortality from insects and diseases over a 15-year future time period across most treed areas of the United States, and supports prevention and suppression activities (chapter 8). It is derived from science-based, transparent geospatial methods, with input from forest health specialists, but has not been formally validated over large areas. NIDRM models were re-run using forest parameter input datasets characterizing the forest conditions of 2002, thereby providing an estimate of mortality for 2002–2016. These results were compared to FIA estimates of mortality for a similar period. A key finding was that NIDRM is reasonably accurate at predicting tree mortality due to insects and diseases at landscape scales, validating its use in a variety of management and planning activities. NIDRM under-prediction

often occurred in areas with either low tree density or few forest pests modeled. Meanwhile, NDRIM predicted mortality at rates greater than FIA often in areas affected by forest pests with periodic or stochastic outbreaks.

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

- Addressed how southern pine beetle damage and management responses alter forest structure, species composition, and canopy fuel loading in New Jersey (chapter 9);
- Investigated the attributes of pitch pine defenses and stand structure relative to an ongoing southern pine beetle outbreak, also in New Jersey (chapter 10);
- Linked experimental changes in epiphytic lichen growth and diversity to observed lichen diversity across the forests of the Great Lakes and Midwest regions, to aid in the development of scientifically defensible climate change indicators (chapter 11);
- Assessed the extent and severity of pine health issues in the Southeast while investigating the main factors affecting pine health and what kinds of management practices can prevent and mitigate damage (chapter 12);
- Evaluated the relationship between drought and oak mortality in the Ozark Highlands of Arkansas and Missouri, and between drought and red oak mortality across 10 Southeastern

States, and examined the likely future distribution of six major coastal tree species (chapter 13); and

- Addressed the detection, impact, and control of an invasive insect-disease complex, the polyphagous and Kuroshio shot hole borers and their mutualistic fungi, in southern California (chapter 14).

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

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Forests cover a vast area of the United States, 304 million ha or approximately one-third of the Nation's land area (Smith and others 2009). These forests possess the capacity to provide a broad range of goods and services to current and future generations, to safeguard biological diversity, and to contribute to the resilience of ecosystems, societies, and economies (USDA Forest Service 2011). Their ecological roles include supplying large and consistent quantities of clean water, preventing soil erosion, and providing habitat for a broad diversity of plant and animal species. Their socioeconomic benefits include wood products, nontimber goods, recreational opportunities, and pleasing natural beauty. Both the ecological integrity and the continued capacity of these forests to provide ecological and economic goods and services are of concern, however, in the face of a long list of threats, including insect and disease infestation, fragmentation, catastrophic fire, invasive species, and the effects of climate change.

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

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

Although the concept of a healthy forest has universal appeal, forest ecologists and managers have struggled with how exactly to define forest health (Teale and Castello 2011), and there is no universally accepted definition. Most definitions of forest health can be categorized as representing either an ecological or a utilitarian perspective (Kolb and others 1994). From an ecological perspective, the current understanding of ecosystem dynamics suggests that healthy ecosystems are those that are able to maintain their organization and autonomy over time while remaining resilient to stress (Costanza 1992), and that evaluations of forest health should emphasize factors that affect the inherent processes and resilience of forests (Edmonds and others 2011, Kolb and others 1994, Raffa and others 2009). On the other hand, the utilitarian perspective holds that a forest is healthy if management objectives are met, and that a forest is unhealthy if these objectives are not met (Kolb and others 1994). Although this definition may be appropriate when a single, unambiguous

# CHAPTER 1.

## Introduction

KEVIN M. POTTER

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

This national report, the 16th in an annual series sponsored by the Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture, attempts to quantify the status of, changes to, and trends in a wide variety of broadly defined indicators of forest health. The indicators described in this report encompass forest insect and disease activity, wildland fire occurrence, drought, tree mortality, understory vegetation, and regeneration, among others. The previous reports in this series are Ambrose and Conkling (2007, 2009), Conkling (2011), Conkling and others (2005), Coulston and others (2005a, 2005b, 2005c), and Potter and Conkling (2012a, 2012b; 2013a, 2013b; 2014; 2015a, 2015b, 2016).

This report has three specific objectives. The first is to present information about forest health from a national perspective, or from a multi-State regional perspective when appropriate,

using data collected by the Forest Health Protection (FHP) and Forest Inventory and Analysis (FIA) programs of the Forest Service, as well as from other sources available at a wide extent. The chapters that present analyses at a national-scale, or multi-State regional scale, are divided between section 1 and section 2 of the report. Section 1 presents results from the analyses of forest health data that are available on an annual basis. Such repeated analyses of regularly collected indicator measurements allow for the detection of trends over time and help establish a baseline for future comparisons (Riitters and Tkacz 2004). Section 2 presents longer-term forest health trends, in addition to describing new techniques for analyzing forest health data at national or regional scales (the second objective of the report). While in-depth interpretation and analysis of specific geographic or ecological regions are beyond the scope of these parts of the report, the chapters in sections 1 and 2 present information that can be used to identify areas that may require investigation at a finer scale.

The second objective of the report is to present new techniques for analyzing forest health data as well as new applications of established techniques, presented in selected chapters of section 2. Examples in this report are chapters 6, 7, and 8, which, respectively, assess change over time in understory vegetation measurements, describe a new regeneration indicator that includes a suite of tree-seedling and browse impact measurements, and compares the general magnitude of tree



mortality predicted by the National Insect and Disease Risk Map with FIA estimates of mortality.

The third objective of the report is to present results of recently completed Evaluation Monitoring (EM) projects funded through the FHM national program. These project summaries, presented in section 3, determine the extent, severity, and/or cause of forest health problems (FHM 2016), generally at a finer scale than that addressed by the analyses in sections 1 and 2. Each of the six chapters in section 3 contains an overview of an EM project, key results, and contacts for more information.

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

## THE FOREST HEALTH MONITORING PROGRAM

The national FHM Program is designed to determine the status, changes, and trends in indicators of forest condition on an annual basis and covers all forested lands through a partnership encompassing the Forest Service, State foresters, and other State and Federal agencies and academic groups (FHM 2016). The FHM Program utilizes data from a wide variety of data sources, both inside and outside the Forest Service, and develops analytical approaches for addressing forest health issues that affect the sustainability of forest ecosystems. The FHM Program has five major components (fig. 1.2):

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

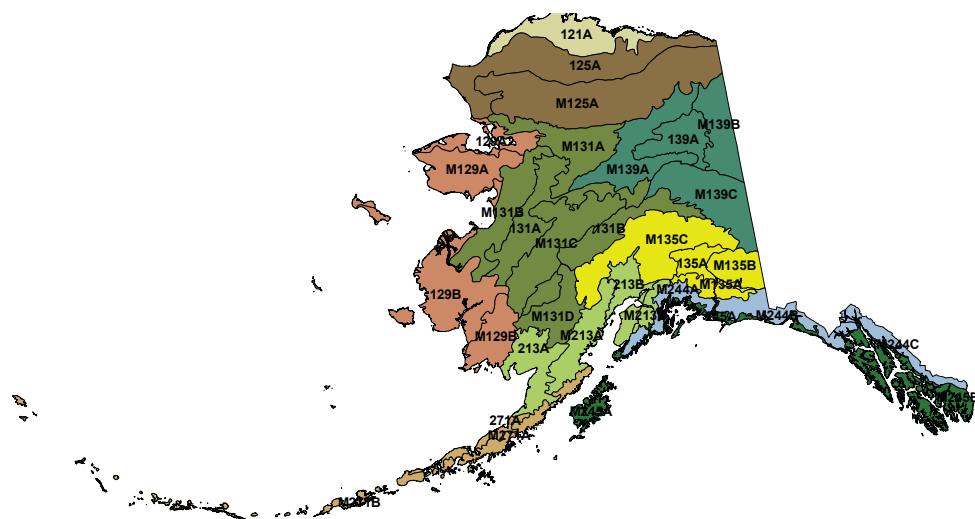
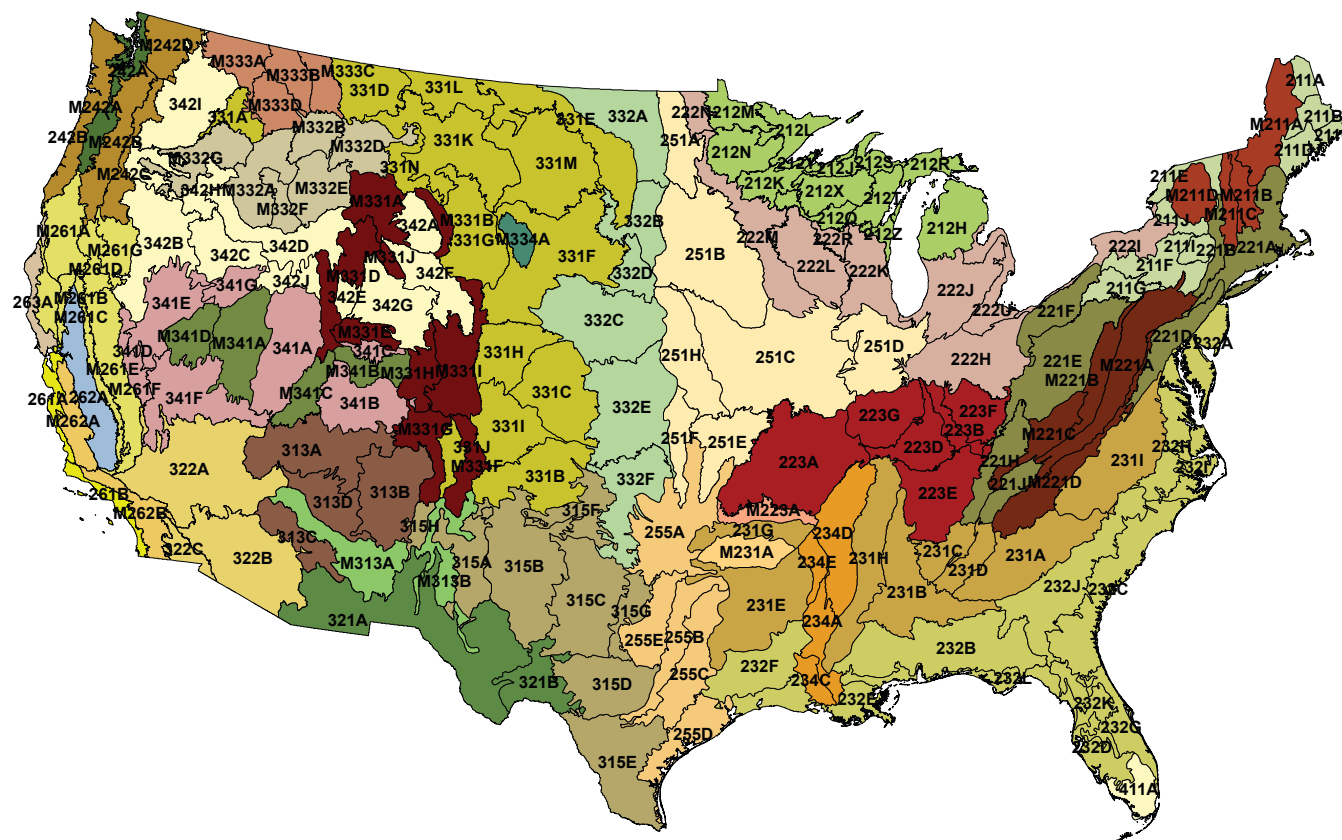


Figure 1.1—Ecoregion provinces and sections for the conterminous United States (Cleland and others 2007) and Alaska (Nowacki and Brock 1995). Ecoregion sections within each ecoregion province are shown in the same color. Note: no equivalent ecoregion treatment exists for Hawai'i.

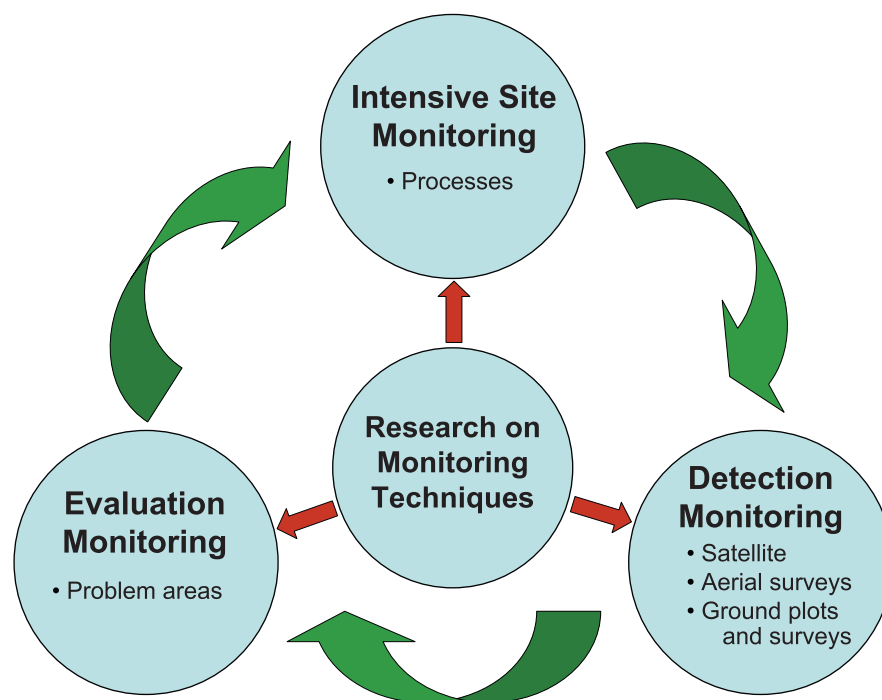


#### Alaska Ecoregion Provinces

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

#### Conterminous States Ecoregion Provinces

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



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

- **Research on Monitoring Techniques**—work to develop or improve indicators, monitoring systems, and analytical techniques, such as urban and riparian forest health monitoring, early detection of invasive species, multivariate analyses of forest health indicators, and spatial scan statistics (section 2 of this report).
- **Analysis and Reporting**—synthesis of information from various data sources within and external to the Forest Service to produce issue-driven reports on status and change in forest health at national, regional, and State levels (sections 1, 2, and 3 of this report).

The FHM Program, in addition to national reporting, generates regional and State reports, often in cooperation with FHM partners, both

within the Forest Service and in State forestry and agricultural departments. For example, the FHM regions cooperate with their respective State partners to produce the annual Forest Health Highlights report series, available on the FHM Web site at [www.fs.fed.us/foresthealth/fhm](http://www.fs.fed.us/foresthealth/fhm). Other examples include Steinman (2004) and Harris and others (2011).

The FHM Program and its partners also produce reports and journal articles on monitoring techniques and analytical methods, including forest health data (Smith and Conkling 2004, Potter and others 2016); soils as an indicator of forest health (O'Neill and others 2005); urban forest health monitoring (Bigsby and others 2014; Cumming and others 2006, 2007; Lake and others 2006); remote sensing of forest disturbances (Chastain and others 2015, Rebbeck and others 2015); health conditions in national forests (Morin and others 2006); crown conditions (Morin and others 2015; Randolph 2010a, 2010b, 2013; Randolph and Moser 2009; Schomaker and others 2007); indicators of regeneration (McWilliams and others 2015); vegetation diversity and structure (Schulz and Gray 2013, Schulz and others 2009, Simkin and others 2016); forest lichen communities (Jovan and others 2012, Root and others 2014); downed woody materials in forests (Woodall and others 2012, 2013); drought (Vose and others 2016); ozone monitoring (Rose and Coulston 2009); patterns of nonnative invasive plant occurrence (Guo and others 2015; Oswalt and others 2015; Iannone and others 2015, 2016); assessments of alien-invasive forest insect and

disease risk (Koch and others 2011, 2014; Krist and others 2014; Yemshanov and others 2014; Vogt and Koch 2016); spatial patterns of land cover (Riitters 2011; Riitters and others 2012, 2016; Riitters and Wickham 2012); broad-scale assessments of forest biodiversity (Potter and Koch 2014; Potter and Woodall 2012, 2014); predictions and indicators of climate change effects on forests and forest tree species (Potter and Hargrove 2013, Heath and others 2015); and the overall forest health indicator program (Woodall and others 2010).

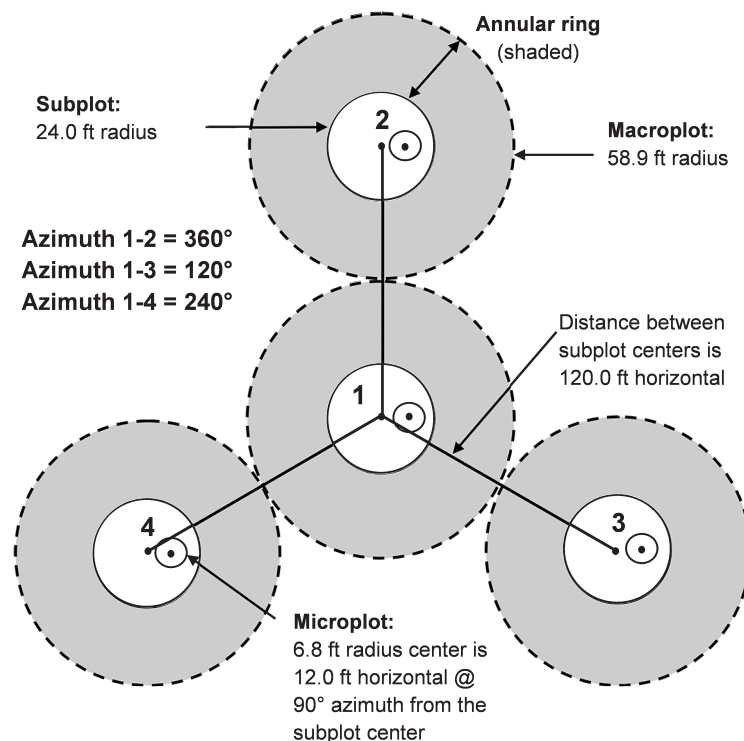
For more information about the FHM Program, visit the FHM Web site at [www.fs.fed.us/foresthealth/fhm](http://www.fs.fed.us/foresthealth/fhm). Among other things, this Web site includes links to all past national forest health reports ([www.fs.fed.us/foresthealth/fhm/pubs](http://www.fs.fed.us/foresthealth/fhm/pubs)),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

## DATA SOURCES

Forest Service data sources in this edition of the FHM national report include FIA annualized phase 2 and phase 3 survey data (Bechtold and Patterson 2005, Woodall and others 2010, Woudenberg and others 2010); FHP National Insect and Disease Survey forest mortality and defoliation data for 2015 (FHP 2014); Moderate Resolution Imaging Spectroradiometer (MODIS) Active Fire Detections for the United States database for 2015 (USDA Forest Service 2016);

and forest cover data developed from MODIS satellite imagery by the U.S. Forest Service Remote Sensing Applications Center. Other sources of data include Parameter-elevation Regression on Independent Slopes Model (PRISM) climate mapping system data (PRISM Climate Group 2016), FIA's publicly available Environmental Monitoring and Assessment Program (EMAP) hexagons (Brand and others 2000), and the 2013–2027 National Insect and Disease Risk Map (Krist and others 2014).

As a major source of data for several FHM analyses, the FIA Program merits detailed description. The FIA Program collects forest inventory information across all forest land ownerships in the United States, and maintains a network of more than 130,000 permanent forested ground plots across the conterminous United States and southeastern Alaska, with a sampling intensity of approximately one plot per 2428 ha. FIA phase 2 encompasses the annualized inventory measured on plots at regular intervals, with each plot surveyed every 5 to 7 years in most Eastern States, but with plots in the Rocky Mountain and Pacific Northwest regions surveyed once every 10 years (Reams and others 2005). The standard 0.067-ha plot (fig. 1.3) consists of four 7.315-m radius subplots (approximately 168.6 m<sup>2</sup> or 1/24th acre), on which field crews measure trees at least 12.7 cm in diameter. Within each of these subplots is nested a 2.073-m radius microplot (approximately 13.48 m<sup>2</sup> or 1/300th acre), on which crews measure trees smaller than 12.7 cm in diameter. A core-optional variant of the standard design includes four “macroplots,” each



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

with a radius of 17.953 m (or approximately 0.1012 ha) that originates at the center of each subplot (Woudenberg and others 2010).

FIA phase 3 plots represent a subset of these phase 2 plots, with one phase 3 plot for every 16 standard FIA phase 2 plots. In addition to traditional forest inventory measurements, data for a variety of important ecological indicators



can be collected from phase 3 plots, including tree crown condition, lichen communities, down woody material, soil condition, and vegetation structure and diversity, whereas data on ozone bioindicator plants are collected on a separate grid of plots (Woodall and others 2010, 2011). Most of these additional forest health indicators were measured as part of the FHM Detection Monitoring ground plot system prior to 2000<sup>1</sup> (Palmer and others 1991).

## FHM REPORT PRODUCTION

This FHM national report, the 16th in a series of such annual documents, is produced by forest health monitoring researchers at the Eastern Forest Environmental Threat Assessment Center (EFETAC) in collaboration with North Carolina State University cooperators. A unit of the Southern Research Station of the Forest Service, EFETAC was established under the Healthy Forests Restoration Act of 2003 to generate the knowledge and tools needed to anticipate and respond to environmental threats. For more information about the research team and about threats to U.S. forests, please visit [www.forestthreats.org/about](http://www.forestthreats.org/about).

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

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# SECTION 1.

## Analyses of Short-Term Forest Health Data



## INTRODUCTION

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

Analyzing patterns of forest insect infestations, disease occurrences, forest declines, and related biotic stress factors is necessary to monitor the health of forested ecosystems and their potential impacts on forest structure, composition, biodiversity, and species distributions (Castello and others 1995). Introduced nonnative insects and diseases, in particular, can extensively damage the

biodiversity, ecology, and economy of affected areas (Brockerhoff and others 2006, Mack and others 2000). Few forests remain unaffected by invasive species, and their devastating impacts in forests are undeniable, including, in some cases, wholesale changes to the structure and function of an ecosystem (Parry and Teale 2011).

Examining insect pest occurrences and related stress factors from a landscape-scale perspective is useful, given the regional extent of many infestations and the large-scale complexity of interactions between host distribution, stress factors, and the development of insect pest outbreaks (Holdenrieder and others 2004, Liebhold and others 2013). One such landscape-scale approach is detecting geographic patterns of disturbance, which allows for the identification of areas at greater risk of significant ecological and economic impacts and for the selection of locations for more intensive monitoring and analysis.

## METHODS

### Data

Forest Health Protection (FHP) national Insect and Disease Survey (IDS) data (FHP 2014) consist of information from low-altitude aerial survey and ground survey efforts by FHP and partners in State agencies. These data can be used to identify forest landscape-scale patterns associated with geographic hot spots of forest insect and disease activity in the conterminous 48 States and to summarize insect and disease activity by ecoregion in Alaska (Potter and

# CHAPTER 2.

## Large-Scale Patterns of Insect and Disease Activity in the Conterminous United States, Alaska, and Hawai'i from the National Insect and Disease Survey, 2015

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Koch 2012; Potter 2012; Potter 2013; Potter and Paschke 2013, 2014, 2015a, 2015b, 2016) and by island in Hawai'i (Potter and Paschke 2015b). In 2015, IDS surveys of the conterminous United States covered about 251.20 million ha, of which approximately 151.10 million ha were forested (about 59.3 percent of the total forested area of the conterminous States); 236.60 million ha of this information was collected using the Digital Aerial Sketch Mapping (DASM) approach (fig. 2.1). An additional 14.96 million ha were surveyed using the new Digital Mobile Sketch Mapping (DMSM) approach in 2015. In Alaska, roughly 13.33 million ha were surveyed in 2015, using only the DASM approach; of this, 8.10 million ha were forested, or about 15.8 percent of the total forested area of the State. Finally, approximately 365 600 ha were surveyed on the Big Island of Hawai'i using DMSM, of which 244 300 ha were forested (59.9 percent of the forested area on the island). DMSM includes tablet hardware, software, and data support processes that allow trained aerial surveyors in light aircraft, as well as ground observers, to record forest disturbances and their causal agents. DMSM replaces the legacy DASM approach, and will greatly enhance the quality and quantity of forest health data while improving safety by integrating with programs such as operational remote sensing (ORS), which uses satellite imagery to monitor disturbances in areas of higher aviation risk (FHP 2016). Geospatial data collected with DMSM are stored in the national IDS.

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

The 2015 mortality and defoliation polygons were used to identify the select mortality and defoliation agents and complexes causing damage on more than 5000 ha of forest in the conterminous United States in that year, and



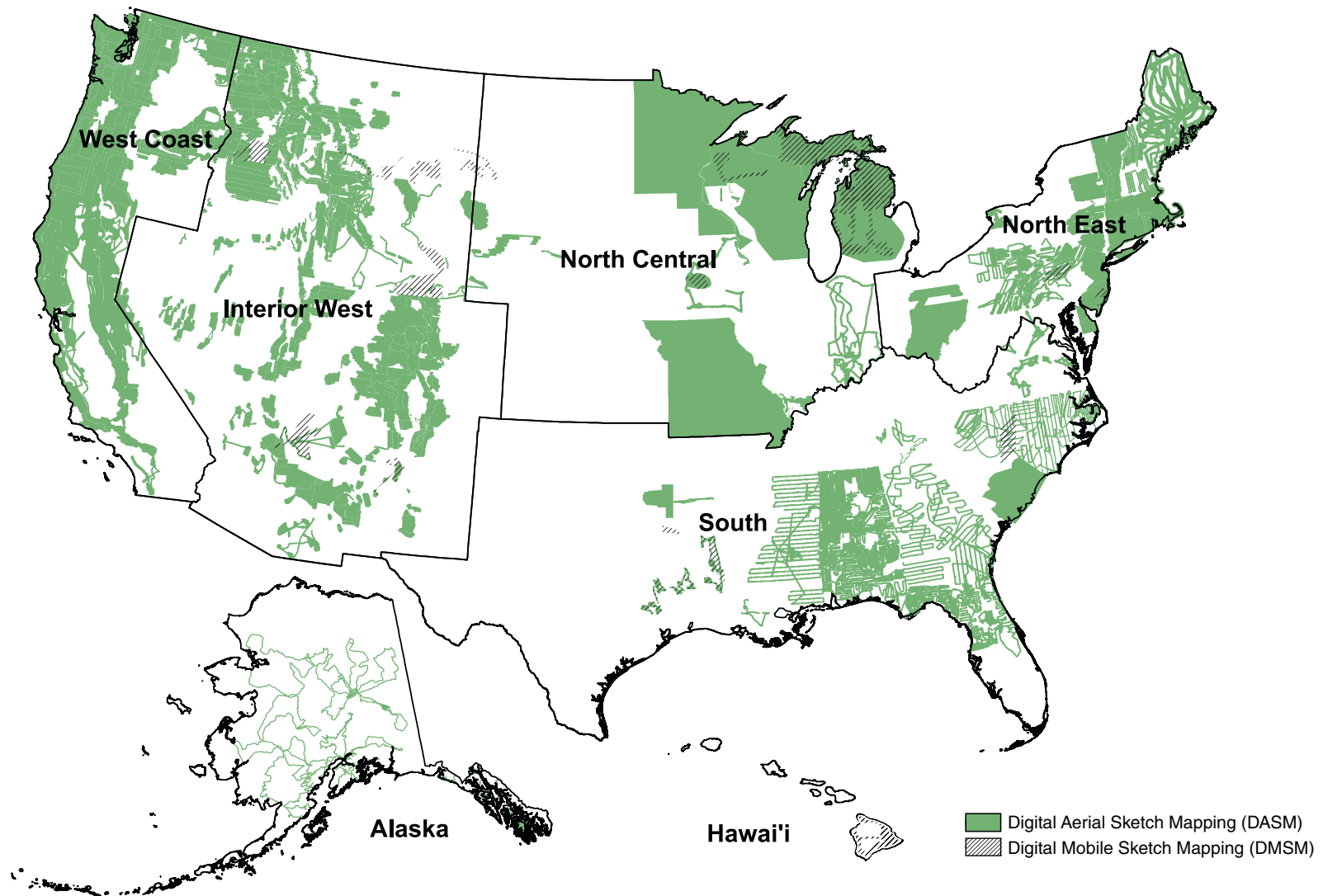


Figure 2.1— The extent of surveys for insect and disease activity conducted in the conterminous United States, Alaska, and Hawai'i in 2015. Cross-hatched areas were surveyed using the new Digital Mobile Sketch Mapping (DMSM) platform, rather than the older Digital Aerial Sketch Mapping (DASM) approach, which is portrayed in green. The black lines delineate Forest Health Monitoring regions. Note: Alaska and Hawai'i are not shown to scale with map of the conterminous United States. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection)

to identify and list the most widely detected mortality and defoliation agents for Alaska and Hawai'i. Because of the insect and disease aerial sketch-mapping process (i.e., digitization of polygons by a human interpreter aboard the aircraft), all quantities are approximate "footprint" areas for each agent or complex, delineating areas of visible damage within which the agent or complex is present. Unaffected trees may exist within the footprint, and the amount of damage within the footprint is not reflected in the estimates of forest area affected. The sum of agents and complexes is not equal to the total affected area as a result of reporting multiple agents per polygon in some situations.

### Analyses

We used the Spatial Association of Scalable Hexagons (SASH) analytical approach to identify surveyed forest areas with the greatest exposure to the detected mortality-causing and defoliation-causing agents and complexes (using data collected using DASM only). This method identifies locations where ecological phenomena occur at greater or lower occurrences than expected by random chance and is based on a sampling frame optimized for spatial neighborhood analysis, adjustable to the appropriate spatial resolution, and applicable to multiple data types (Potter and others 2016). Specifically, it consists of dividing an analysis area into scalable equal-area hexagonal cells within which data are aggregated, followed by identifying statistically significant geographic clusters of hexagonal cells within which mean values are greater or less than those expected by

chance. To identify these clusters, we employ a Getis-Ord ( $G_i^*$ ) hot spot analysis (Getis and Ord 1992) in ArcMap® 10.1 (ESRI 2012).

The units of analysis were 9,810 hexagonal cells, each approximately 834 km<sup>2</sup> in area, generated in a lattice across the conterminous United States using intensification of the Environmental Monitoring and Assessment Program (EMAP) North American hexagon coordinates (White and others 1992). These coordinates are the foundation of a sampling frame in which a hexagonal lattice was projected onto the conterminous United States by centering a large base hexagon over the region (Reams and others 2005, White and others 1992). This base hexagon can be subdivided into many smaller hexagons, depending on sampling needs, and serves as the basis of the plot sampling frame for the Forest Inventory and Analysis (FIA) Program (Reams and others 2005). Importantly, the hexagons maintain equal areas across the study region regardless of the degree of intensification of the EMAP hexagon coordinates. In addition, the hexagons are compact and uniform in their distance to the centroids of neighboring hexagons, meaning that a hexagonal lattice has a higher degree of isotropy (uniformity in all directions) than does a square grid (Shima and others 2010). These are convenient and highly useful attributes for spatial neighborhood analyses. These scalable hexagons also are independent of geopolitical and ecological boundaries, avoiding the possibility of different sample units (such as counties, States, or watersheds) encompassing

vastly different areas (Potter and others 2016). We selected hexagons 834 km<sup>2</sup> in area because this is a manageable size for making monitoring and management decisions in analyses that are national in extent (Potter and others 2016).

The variable used in the hot spot analysis was the percentage of surveyed forest area in each hexagon exposed to either mortality-causing or defoliation-causing agents. This required first separately dissolving the mortality and defoliation polygon boundaries to generate an overall footprint of each general type of disturbance, then masking the dissolved polygons using a forest cover map (1-km<sup>2</sup> resolution) derived from Moderate Resolution Imaging Spectroradiometer (MODIS) satellite imagery by the U.S. Forest Service Remote Sensing Applications Center (USDA Forest Service 2008). The same process was undertaken with the polygons of the surveyed area. Finally, the percentage of surveyed forest within each hexagon exposed to mortality or defoliation agents was calculated by dividing the total forest-masked damage area by the forest-masked surveyed area.

The Getis-Ord  $G_i^*$  statistic was used to identify clusters of hexagonal cells in which the percentage of surveyed forest exposed to mortality or defoliation agents was higher than expected by chance. This statistic allows for the decomposition of a global measure of spatial association into its contributing factors, by location, and is therefore particularly suitable for

detecting nonstationarities in a dataset, such as when spatial clustering is concentrated in one subregion of the data (Anselin 1992).

The Getis-Ord  $G_i^*$  statistic for each hexagon summed the differences between the mean values in a local sample, determined by a moving window consisting of the hexagon and its 18 first- and second-order neighbors (the 6 adjacent hexagons and the 12 additional hexagons contiguous to those 6) and the global mean of all the forested hexagonal cells in the conterminous 48 States. It is then standardized as a  $z$  score with a mean of 0 and a standard deviation of 1, with values  $> 1.96$  representing significant ( $p < 0.025$ ) local clustering of high values and values  $< -1.96$  representing significant clustering of low values ( $p < 0.025$ ), since 95 percent of the observations under a normal distribution should be within approximately 2 (exactly 1.96) standard deviations of the mean (Laffan 2006). In other words, a  $G_i^*$  value of 1.96 indicates that the local mean of the percentage of forest exposed to mortality-causing or defoliation-causing agents for a hexagon and its 18 neighbors is approximately 2 standard deviations greater than the mean expected in the absence of spatial clustering, while a  $G_i^*$  value of -1.96 indicates that the local mortality or defoliation mean for a hexagon and its 18 neighbors is approximately 2 standard deviations less than the mean expected in the absence of spatial clustering. Values between -1.96 and 1.96 have no statistically significant concentration of high or low values.

In other words, when a hexagon has a  $G_i^*$  value between -1.96 and 1.96, mortality or defoliation damage within it and its 18 neighbors is not statistically different from a normal expectation.

It is worth noting that the -1.96 and 1.96 threshold values are not exact because the correlation of spatial data violates the assumption of independence required for statistical significance (Laffan 2006). The Getis-Ord approach does not require that the input data be normally distributed because the local  $G_i^*$  values are computed under a randomization assumption, with  $G_i^*$  equating to a standardized  $z$  score that asymptotically tends to a normal distribution (Anselin 1992). The  $z$  scores are reliable, even with skewed data, as long as the distance band used to define the local sample around the target observation is large enough to include several neighbors for each feature (ESRI 2012).

The low density of survey data from Alaska and Hawai'i in 2015 (fig. 2.1) precluded the use of Getis-Ord  $G_i^*$  hot spot analyses for these States. Instead, mortality and defoliation data were summarized by ecoregion section in Alaska (Nowacki and Brock 1995) and by county council district on the Big Island of Hawai'i (State of Hawai'i 2012), calculated as the percent of the forest within the surveyed areas affected by agents of mortality or defoliation. (As with the mortality and defoliation data, the flown area polygons were first dissolved to create an overall footprint.) For reference purposes,

ecoregion sections (Cleland and others 2007) were also displayed on the geographic hot spot maps of the conterminous 48 United States.

## RESULTS AND DISCUSSION

### Conterminous United States Mortality

The national IDS survey data identified 70 different mortality-causing agents and complexes on approximately 2.12 million ha across the conterminous United States in 2015, slightly less than the land area of New Hampshire. By way of comparison, forests are estimated to cover approximately 252 million ha of the conterminous 48 States (Smith and others 2009).

Fir engraver (*Scolytus ventralis*) was the most widespread mortality agent in 2015, detected on 581 524 ha (table 2.1). Four other mortality agents and complexes were detected on more than 100 000 ha in 2015: mountain pine beetle (*Dendroctonus ponderosae*), Jeffrey pine beetle (*Dendroctonus jeffreyi*), western pine beetle (*Dendroctonus brevicomis*), and spruce beetle (*Dendroctonus rufipennis*). Mortality from the western bark beetle group, which encompasses 18 different agents in the IDS data (table 2.2), was detected on approximately 1.81 million ha in 2015, representing a large majority of the total area on which mortality was recorded across the conterminous States. This was the first year since 2003 that mountain pine beetle was not the most widespread mortality agent; at its

**Table 2.1—Mortality agents and complexes affecting more than 5000 ha in the conterminous United States during 2015**

| Agents/complexes causing mortality, 2015 | Area<br><i>ha</i> |
|------------------------------------------|-------------------|
| Fir engraver                             | 581 524           |
| Mountain pine beetle                     | 534 176           |
| Jeffrey pine beetle                      | 308 567           |
| Western pine beetle                      | 290 729           |
| Spruce beetle                            | 254 749           |
| Pinyon ips                               | 97 001            |
| Emerald ash borer                        | 73 786            |
| Western balsam bark beetle               | 57 482            |
| Douglas-fir beetle                       | 56 181            |
| Unknown bark beetle                      | 38 096            |
| Flatheaded fir borer                     | 35 210            |
| Sudden oak death                         | 24 328            |
| Unknown                                  | 23 616            |
| Root disease and beetle complex          | 23 047            |
| Beech bark disease complex               | 19 846            |
| Spruce budworm                           | 19 104            |
| Western cedar bark beetle                | 18 442            |
| Eastern larch beetle                     | 16 775            |
| Ips engraver beetles                     | 16 662            |
| California flatheaded borer              | 16 059            |
| Gray brown sap rot                       | 12 723            |
| Balsam woolly adelgid                    | 12 368            |
| Hemlock woolly adelgid                   | 6 332             |
| Oak decline                              | 6 007             |
| Pine engraver                            | 5 254             |
| Other (45)                               | 28 073            |
| <b>Total, all mortality agents</b>       | <b>2 212 566</b>  |

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

**Table 2.2—Beetle taxa included in the “western bark beetle” group**

| Western bark beetle mortality agents | Beetle taxa                       |
|--------------------------------------|-----------------------------------|
| Cedar and cypress bark beetles       | <i>Phloeosinus</i> spp.           |
| Douglas-fir beetle                   | <i>Dendroctonus pseudotsugae</i>  |
| Douglas-fir engraver                 | <i>Scolytus unispinosus</i>       |
| Fir engraver                         | <i>Scolytus ventralis</i>         |
| Fir root bark beetle                 | <i>Pseudohylesinus granulatus</i> |
| Flatheaded borer                     | Family Buprestidae                |
| Ips engraver beetles                 | <i>Ips</i> spp.                   |
| Jeffrey pine beetle                  | <i>Dendroctonus jeffreyi</i>      |
| Mountain pine beetle                 | <i>Dendroctonus ponderosae</i>    |
| Pine engraver                        | <i>Ips pini</i>                   |
| Pinyon ips                           | <i>Ips confusus</i>               |
| Root disease and beetle complex      | —                                 |
| Silver fir beetle                    | <i>Pseudohylesinus sericeus</i>   |
| Spruce beetle                        | <i>Dendroctonus rufipennis</i>    |
| Unknown bark beetle                  | —                                 |
| Western balsam bark beetle           | <i>Dryocoetes confusus</i>        |
| Western cedar bark beetle            | <i>Phloeosinus punctatus</i>      |
| Western pine beetle                  | <i>Dendroctonus brevicomis</i>    |

— = not applicable.

annual maximum, 3.47 million ha was exposed to mountain pine beetle mortality in 2009 (Potter 2013).

The FHM West Coast region had the largest area on which mortality agents and complexes were detected, about 1.29 million ha (table 2.3). Of the 42 agents and complexes detected, fir engraver was the leading cause of mortality and was identified on about 541 600 ha, approximately 42 percent of the entire affected area. Other bark beetles, including mountain

**Table 2.3—The top five mortality agents or complexes for each Forest Health Monitoring region, and for Alaska and Hawai‘i, in 2015**

| 2015 mortality agents and complexes              | Area           | 2015 mortality agents and complexes              | Area             |
|--------------------------------------------------|----------------|--------------------------------------------------|------------------|
|                                                  | <i>ha</i>      |                                                  | <i>ha</i>        |
| <b>Interior West</b>                             |                | <b>West Coast</b>                                |                  |
| Spruce beetle                                    | 248 289        | Fir engraver                                     | 541 592          |
| Mountain pine beetle                             | 153 756        | Mountain pine beetle                             | 374 324          |
| Western balsam bark beetle                       | 56 111         | Jeffrey pine beetle                              | 308 286          |
| Fir engraver                                     | 39 932         | Western pine beetle                              | 288 770          |
| Douglas-fir beetle                               | 37 570         | Pinyon ips                                       | 95 905           |
| Other mortality agents (15)                      | 70 507         | Other mortality agents (37)                      | 162 853          |
| <b>Total, all mortality agents and complexes</b> | <b>595 328</b> | <b>Total, all mortality agents and complexes</b> | <b>1,288,423</b> |
| <b>North Central</b>                             |                | <b>Alaska</b>                                    |                  |
| Emerald ash borer                                | 30 736         | Yellow-cedar decline                             | 15 940           |
| Spruce budworm                                   | 19 104         | Spruce beetle                                    | 12 587           |
| Eastern larch beetle                             | 16 757         | Northern spruce engraver                         | 3 289            |
| Gray brown sap rot                               | 12 723         | Unknown                                          | 100              |
| Unknown                                          | 8 611          | Western balsam bark beetle                       | 10               |
| Other mortality agents (19)                      | 17 975         | <b>Total, all mortality agents and complexes</b> | <b>31 814</b>    |
| <b>Total, all mortality agents and complexes</b> | <b>105 804</b> | <b>Hawai‘i</b>                                   |                  |
| <b>North East</b>                                |                | Rapid ‘ōhi‘a death                               | 17 980           |
| Emerald ash borer                                | 43 047         | Unknown                                          | 2 520            |
| Beech bark disease complex                       | 15 398         | <b>Total, all mortality agents and complexes</b> | <b>20 501</b>    |
| Hemlock woolly adelgid                           | 6 332          |                                                  |                  |
| Oak decline                                      | 3 450          |                                                  |                  |
| Unknown                                          | 2 635          |                                                  |                  |
| Other mortality agents (24)                      | 14 472         |                                                  |                  |
| <b>Total, all mortality agents and complexes</b> | <b>76 715</b>  |                                                  |                  |
| <b>South</b>                                     |                |                                                  |                  |
| Unknown                                          | 3 302          |                                                  |                  |
| Ips engraver beetles                             | 2 273          |                                                  |                  |
| Southern pine beetle                             | 763            |                                                  |                  |
| Unknown bark beetle                              | 632            |                                                  |                  |
| Other wood borer, known                          | 7              |                                                  |                  |
| Other mortality agents (3)                       | 5              |                                                  |                  |
| <b>Total, all mortality agents and complexes</b> | <b>6 982</b>   |                                                  |                  |

Note: The total area affected by other agents is listed at the end of each section. All values are “footprint” areas for each agent or complex. The sum of the individual agents is not equal to the total for all agents due to the reporting of multiple agents per polygon.



pine beetle, Jeffrey pine beetle, western pine beetle, and pinyon ips (*Ips confusus*), were also widespread causes of mortality in the region.

A large hot spot of extremely high and very high mortality extended throughout most of the M261E–Sierra Nevada ecoregion in east-central California, and was caused by extensive detections of mortality in singleleaf pinyon (*Pinus monophylla*) caused by pinyon ips; in Jeffrey pine (*Pinus jeffreyi*) caused by Jeffrey pine beetle; in white fir (*Abies concolor*) and California red fir (*Abies magnifica*) caused by fir engraver; and in sugar pine (*Pinus lambertiana*), lodgepole pine (*Pinus contorta*), and western white pine (*Pinus monticola*) caused by mountain pine beetle (fig. 2.2). Mortality associated with many of these same agents extended south into a hot spot of high and moderate mortality in M262B–Southern California Mountain and Valley.

A hot spot of very high exposure to mortality was detected in northern California and south-central Oregon, centered on M261D–Southern Cascades, M261A–Klamath Mountains, and M261G–Modoc Plateau, and was associated primarily with fir engraver in white fir and California red fir, western pine beetle in ponderosa pine (*Pinus ponderosa*), Jeffrey pine beetle in Jeffrey pine, and mountain pine beetle in lodgepole pine (fig. 2.2). Meanwhile, a hot spot of high mountain pine beetle mortality in lodgepole pine and ponderosa pine was detected in M332G–Blue Mountains in eastern Oregon.

Sudden oak death mortality in tanoak (*Lithocarpus densiflorus*) and California live oak (*Quercus agrifolia*) forests, flatheaded fir borer (*Phaenops drummondi*) related mortality in Douglas-fir (*Pseudotsuga menziesii*) stands, western pine beetle-related mortality of Coulter pine (*Pinus coulteri*) stands, and ips engraver beetles in grey pine (*Pinus sabaliana*) resulted in hot spots of moderate mortality near the central California coast, in 263A–Northern California Coast and 261A–Central California Coast.

The FHM Interior West region had approximately 595 300 ha on which mortality-causing agents and complexes were detected in 2015, second only to the West Coast region (table 2.3). About 42 percent of this was associated with spruce beetle; also constituting a considerable area were mountain pine beetle (26 percent), western balsam bark beetle (*Dryocoetes confusus*) (9 percent), fir engraver (7 percent), and Douglas-fir beetle (*Dendroctonus pseudotsugae*) (6 percent). A total of 20 mortality agents and complexes were detected in the region.

The Getis-Ord analysis detected several hot spots of intense mortality exposure in the Interior West region in 2015, but none with more than high exposure to mortality-causing agents (fig. 2.2). A widespread spruce beetle infestation in Engelmann spruce (*Picea engelmannii*), hot spot in south-central Colorado, centered on M331G–South-Central Highlands, M331I–Northern Parks and Ranges, M331H–North-Central Highlands

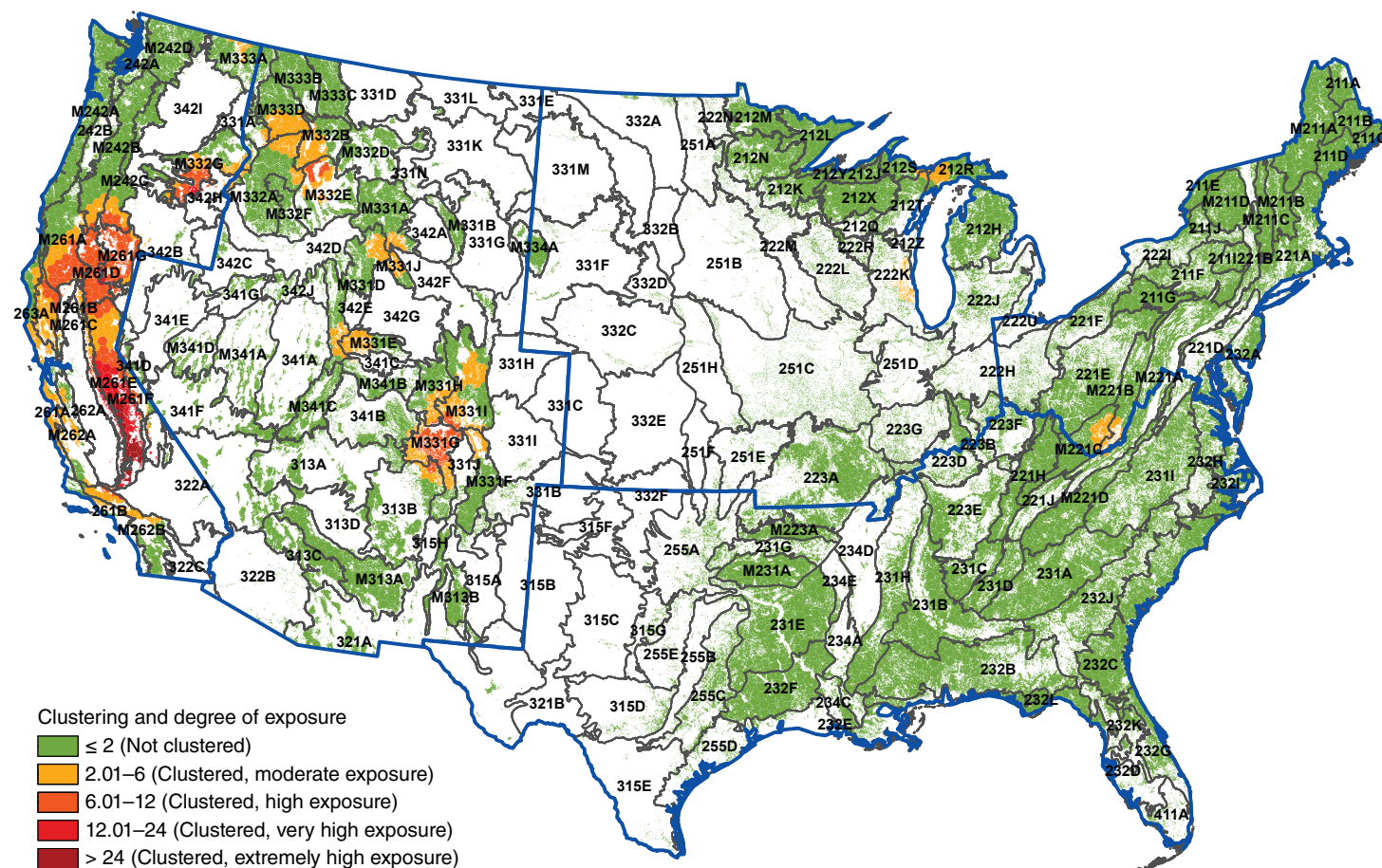


Figure 2.2— Hot spots of exposure to mortality-causing insects and diseases in 2015. Values are Getis-Ord  $G_i^*$  scores, with values  $> 2$  representing significant clustering of high percentages of forest area exposed to mortality agents. (No areas of significant clustering of low percentages of exposure,  $< -2$ , were detected.) The gray lines delineate ecoregion sections (Cleland and others 2007), and blue lines delineate Forest Health Monitoring regions. Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection)



and Rocky Mountains, and M331F–Southern Parks and Rocky Mountain Range. Western bark beetle mortality in subalpine fir (*Abies lasiocarpa*) forests and mountain pine beetle mortality in limber pine (*Pinus flexilis*) also occurred in this area.

As during the last 3 years, a hot spot of mortality exposure was centered on the border between eastern Idaho and western Montana, especially in ecoregions M332B–Northern Rockies and Bitterroot Valley, M332E–Beaverhead Mountains, M332D–Belt Mountains, and M332A–Idaho Batholith (fig. 2.2). The intensity of the mortality—caused mostly by mountain pine beetle in lodgepole pine forests—was lower than in the previous years, however. An additional hot spot of moderate mortality, detected to the south in M331J–Wind River Mountains, M331A–Yellowstone Highlands, and M331D–Overthrust Mountains, was associated primarily with spruce beetle infestation in Englemann spruce stands, along with root disease and beetle complex in subalpine fir and mountain pine beetle in limber pine and whitebark pine (*Pinus albicaulis*). Similarly, a moderate-intensity hot spot in M331E–Uinta Mountains of northeastern Utah was mainly associated with spruce beetle-caused mortality in Engelmann spruce.

In the North East FHM region, the FHP survey recorded 29 mortality-causing agents and complexes on approximately 76 700 ha (table 2.3). Emerald ash borer (*Agrilus planipennis*) was by far the most commonly

detected, on 43 000 ha, approximately 56 percent of the area experiencing mortality. Other widespread agents and complexes were beech bark disease complex (20 percent) and hemlock woolly adelgid (8 percent). Emerald ash borer caused a hot spot of moderate mortality exposure in east-central West Virginia, in M221B–Allegheny Mountains and M221C–Northern Cumberland Mountains (fig. 2.2).

In the North Central FHM region, mortality was recorded on nearly 105 800 ha, with emerald ash borer the most widely identified causal agent, found on almost 31 000 ha (table 2.3). Of the 24 agents and complexes detected in the region, three others affected areas exceeding 10 000 ha: spruce budworm (*Choristoneura fumiferana*), eastern larch beetle (*Dendroctonus simplex*) and gray brown sap rot (*Cryptoporus volvatus*). Mortality with unclassified causes also was detected on about 8600 ha. Emerald ash borer was the cause of a mortality hot spot in southeastern Wisconsin (222K–Southwestern Great Lakes Morainel), and gray brown sap rot mortality in American beech resulted in a hot spot on the upper peninsula of Michigan (212S–Northern Upper Peninsula and 212R–Eastern Upper Peninsula) (fig. 2.2).

In the South, mortality was detected on about 7000 ha, mostly with an unknown cause. Ips engraver beetles were the most commonly identified agent, on 2300 ha (table 2.3). No geographic hot spots of mortality were detected in the South FHM region.

## Conterminous United States Defoliation

In 2015, the national IDS survey identified 58 defoliation agents and complexes affecting approximately 4.85 million ha across the conterminous United States (table 2.4), an area approximately the same as the combined land area of Maryland, Massachusetts, and Rhode Island. The two most widespread defoliation agents were yellow poplar weevil (*Odontopus calceatus*), affecting approximately 2.23 million ha, and western spruce budworm (*Choristoneura occidentalis*), detected on approximately 1.27 million ha. Three other insects—gypsy moth (*Lymantria dispar*), spruce budworm, and loblolly pine sawfly (*Neodiprion taedae linearis*)—each also affected more than 100 000 ha (table 2.4).

The South FHM region had the largest area on which defoliating agents and complexes were detected in 2015, approximately 2.18 million ha (table 2.5). A little more than 91 percent of this, or 2 million ha, was associated with yellow poplar weevil. A total of 12 agents and complexes were found, including loblolly pine sawfly on about 133 800 ha. A large hot spot of extremely high exposure to defoliating agents was detected in western North Carolina (M221D—Blue Ridge Mountains and 231I—Central Appalachian Piedmont), associated with an extensive area of patchy light to heavy defoliation in yellow-poplar (*Liriodendron tulipifera*) by yellow poplar weevil (fig. 2.3). A second hot spot of high defoliation exposure was caused by loblolly pine sawfly in loblolly pine

**Table 2.4—Defoliation agents and complexes affecting more than 5000 ha in the conterminous United States in 2015**

| Agents/complexes causing defoliation, 2015 | Area             |
|--------------------------------------------|------------------|
|                                            | ha               |
| Yellow poplar weevil                       | 2 231 938        |
| Western spruce budworm                     | 1 274 198        |
| Gypsy moth                                 | 612 333          |
| Spruce budworm                             | 271 778          |
| Loblolly pine sawfly                       | 133 834          |
| Forest tent caterpillar                    | 57 921           |
| Unknown defoliator                         | 56 515           |
| Jack pine budworm                          | 49 979           |
| Winter moth                                | 49 797           |
| Fall cankerworm                            | 47 945           |
| Oak leafroller                             | 47 379           |
| Cherry scallop shell moth                  | 23 598           |
| Unknown                                    | 16 774           |
| Large aspen tortrix                        | 13 875           |
| Douglas-fir tussock moth                   | 11 228           |
| Larch casebearer                           | 11 062           |
| Loopers                                    | 9 883            |
| Birch leafminer                            | 8 090            |
| Skeletonizer                               | 8 090            |
| Birch leaf fungus                          | 7 884            |
| Locust leafminer                           | 7 072            |
| Lophodermium needle cast of pines          | 6 820            |
| Tamarisk leaf beetles                      | 6 369            |
| Baldcypress leafroller                     | 6 117            |
| Other (34)                                 | 41 225           |
| <b>Total, all defoliation agents</b>       | <b>4 848 645</b> |

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

**Table 2.5—The top five defoliation agents or complexes for each Forest Health Monitoring region and for Alaska in 2015**

| 2015 defoliation agents and complexes              | Area             | 2015 defoliation agents and complexes              | Area             |
|----------------------------------------------------|------------------|----------------------------------------------------|------------------|
|                                                    | ha               |                                                    | ha               |
| <b>Interior West</b>                               |                  | <b>South</b>                                       |                  |
| Western spruce budworm                             | 1 240 820        | Yellow poplar weevil                               | 1 997 336        |
| Unknown defoliator                                 | 51 449           | Loblolly pine sawfly                               | 133 834          |
| Douglas-fir tussock moth                           | 11 228           | Forest tent caterpillar                            | 31 307           |
| Tamarisk leaf beetles                              | 6 369            | Unknown                                            | 13 140           |
| Spruce aphid                                       | 3 580            | Baldcypress leafroller                             | 6 117            |
| Other defoliation agents (13)                      | 8 070            | Other defoliation agents (7)                       | 6 667            |
| <b>Total, all defoliation agents and complexes</b> | <b>1 320 653</b> | <b>Total, all defoliation agents and complexes</b> | <b>2 183 362</b> |
| <b>North Central</b>                               |                  | <b>West Coast</b>                                  |                  |
| Spruce budworm                                     | 271 778          | Western spruce budworm                             | 33 378           |
| Jack pine budworm                                  | 49 979           | Lophodermium needle cast of pines                  | 6 820            |
| Forest tent caterpillar                            | 26 615           | Needlecast                                         | 4 200            |
| Large aspen tortrix                                | 13 875           | Unknown defoliator                                 | 1 252            |
| Larch casebearer                                   | 10 927           | Larch needle cast                                  | 1 224            |
| Other defoliation agents (12)                      | 7 301            | Other defoliation agents (9)                       | 1 837            |
| <b>Total, all defoliation agents and complexes</b> | <b>378 965</b>   | <b>Total, all defoliation agents and complexes</b> | <b>48 673</b>    |
| <b>North East</b>                                  |                  | <b>Alaska</b>                                      |                  |
| Gypsy moth                                         | 608 510          | Unknown defoliator                                 | 113 123          |
| Yellow poplar weevil                               | 234 602          | Aspen leafminer                                    | 32 112           |
| Winter moth                                        | 49 797           | Willow leaf blotchminer                            | 13 387           |
| Fall cankerworm                                    | 47 492           | Large aspen tortrix                                | 8 287            |
| Oak leafroller                                     | 47 035           | Spruce aphid                                       | 2 600            |
| Other defoliation agents (22)                      | 85 166           | Other defoliation agents (6)                       | 2 105            |
| <b>Total, all defoliation agents and complexes</b> | <b>916 993</b>   | <b>Total, all defoliation agents and complexes</b> | <b>170 397</b>   |

Note: The total area affected by other agents is listed at the end of each section. All values are “footprint” areas for each agent or complex. The sum of the individual agents is not equal to the total for all agents due to the reporting of multiple agents per polygon.

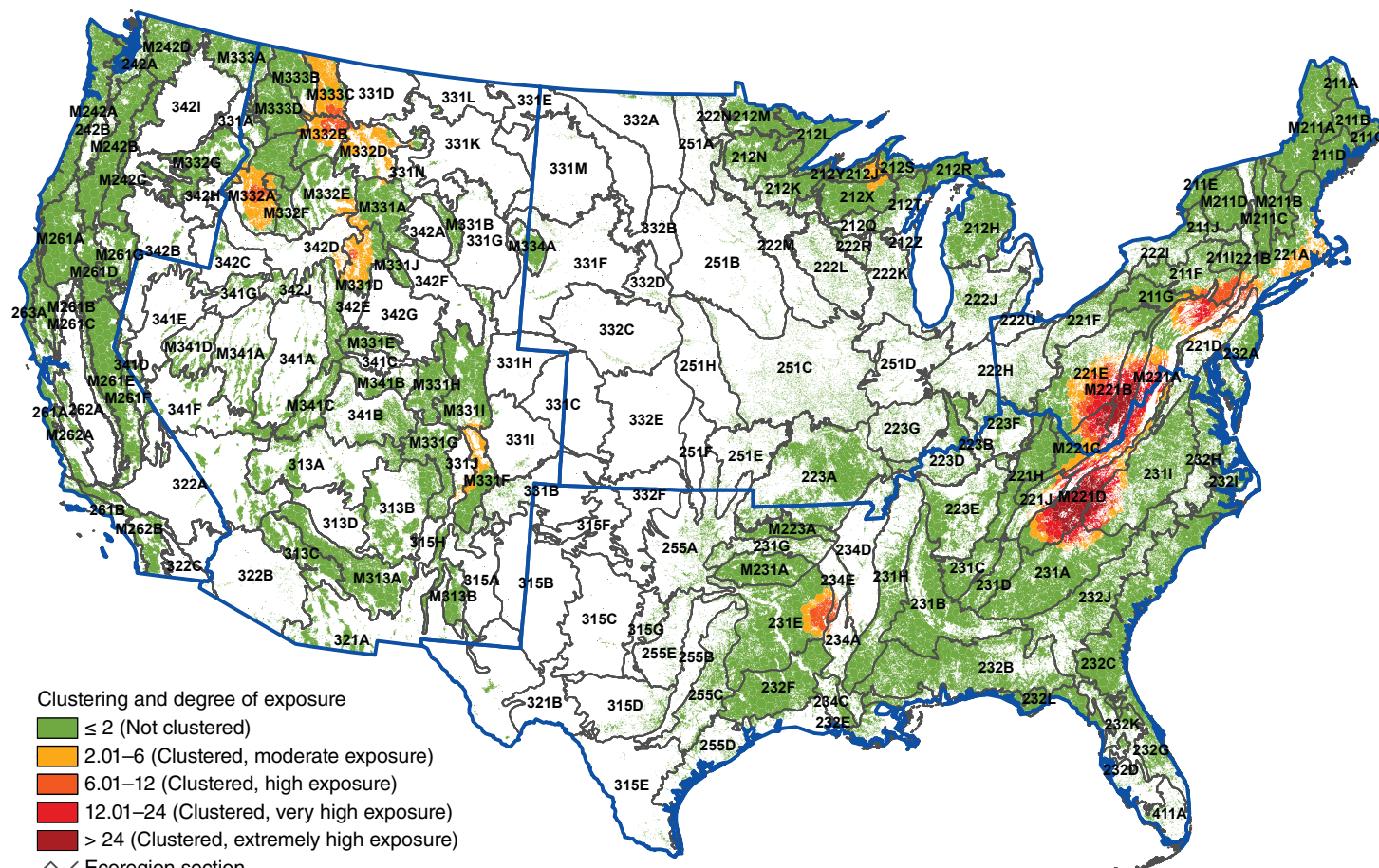


Figure 2.3—Hot spots of exposure to defoliation-causing insects and diseases in 2015. Values are Getis-Ord  $G_i^*$  scores, with values > 2 representing significant clustering of high percentages of forest area exposed to defoliation agents. (No areas of significant clustering of low percentages of exposure, < -2, were detected.) The gray lines delineate ecoregion sections (Cleland and others 2007), and blue lines delineate Forest Health Monitoring regions. Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection)

(*Pinus taeda*) forests of southeastern Arkansas (234E–Arkansas Alluvial Plains and 231E–Mid Coastal Plains-Western).

In 2015, approximately 1.32 million ha of defoliation was documented in the Interior West FHM region, of which the vast majority (94 percent, or 1.24 million ha) was associated with western spruce budworm (table 2.5). Unknown defoliators and Douglas-fir tussock moth (*Orgyia pseudotsugata*) were the next most widely detected defoliation agents of the 18 that were identified.

The Getis-Ord analysis detected four 2015 defoliation hot spots in the region (fig. 2.3), each associated with western spruce budworm (along with other agents) and each overlapping or near a similar hot spot in 2014 (Potter and Paschke 2016). As in 2014, the largest of these, caused by western spruce budworm in fir forests, was a hot spot of high defoliation exposure that centered on M333B–Flathead Valley, M333C–Northern Rockies and M332D–Belt Mountains. Meanwhile, western spruce budworm activity in Douglas-fir forests and subalpine fir resulted in a hot spot of high defoliation exposure in central Idaho (M332A–Idaho Batholith) and another in western Wyoming and southeastern Idaho (M331D–Overthrust Mountains, M331A–Yellowstone Highlands, and M332E–Beaverhead Mountains).

Twenty-seven defoliation agents and complexes were identified on about 917 000 ha in the North East FHM region, with gypsy moth the most widely detected on nearly 66 percent of

this area (more than 608 500 ha). Yellow poplar weevil was recorded on more than 234 600 ha, winter moth (*Operophtera brumata*) on 49 800 ha, fall cankerworm on 47 500 ha, and oak leafroller (*Archips semiferana*) on 47 000 ha (table 2.5). As to the south along the Appalachian Mountains, an outbreak of yellow poplar weevil, along with a smaller gypsy moth component, resulted in a hot spot of extremely high defoliation exposure in eastern West Virginia (M221B–Allegheny Mountains, M221C–Northern Cumberland Mountains, and M221A–Northern Ridge and Valley) (fig. 2.3). Meanwhile, a gypsy moth outbreak in eastern Pennsylvania, northern New Jersey, and southern New York caused an extensive hot spot of high and very high defoliation exposure in M221A–Northern Ridge and Valley, 221A–Lower New England, 211F–Northern Glaciated Allegheny Plateau, and 221B–Hudson Valley (fig. 2.3). Finally, a hot spot of moderate defoliation exposure was caused by gypsy moth and winter moth in 221A–Lower New England (Massachusetts, Rhode Island, and Connecticut).

Meanwhile, 17 agents and complexes were associated with about 379 000 ha of defoliation in the North Central FHM region (table 2.5). Spruce budworm was the most commonly detected defoliation agent in the region, found on a little less than 272 000 ha, or 72 percent of the defoliated area. Other widespread defoliators were jack pine budworm (*Choristoneura pinus*), forest tent caterpillar (*Malacosoma disstria*), large aspen tortrix (*Choristoneura conflictana*), and larch casebearer (*Coleophora laricella*), affecting



approximately 50 000 ha, 26 600 ha, 13 900 ha, and 10 900 ha, respectively (table 2.5). Our geographic hot spot analysis detected a single cluster of moderate defoliation exposure in North Central FHM region (fig. 2.3), caused by spruce budworm in the Upper Peninsula of Michigan (212J–Southern Superior Highlands, 212X–Northern Highlands, and 212S–Northern Upper Peninsula).

Finally, western spruce budworm accounted for about 69 percent of the approximately 48 700 ha of defoliation recorded in the FHM West Coast region (table 2.5). Of the 14 defoliation agents and complexes detected in the region, the next most widely found was *Lophodermium* needle cast of pines (*Lophodermium* spp.), on 6800 ha. No geographic hot spots of defoliation were identified in the region.

### Alaska and Hawai'i

In Alaska, mortality was recorded on approximately 31 800 ha in 2015, associated with five agents and complexes (table 2.3). This is a very small proportion (< 0.25 percent) of the forested area surveyed. Yellow-cedar (*Chamaecyparis nootkatensis*) decline was the most widely detected mortality agent, found on about 15 900 ha in the Alaska panhandle (M245B–Alexander Archipelago), whereas spruce beetle was identified on almost 12 600 ha, mostly in the south-central part of the State (especially 213B–Cook Inlet Lowlands), and northern spruce engraver (*Ips perturbatus*) was detected on about 3300 ha, mostly in the central forested areas of Alaska. The percentage of surveyed forest

exposed to mortality agents in 2015 exceeded 1 percent only in 213B–Cook Inlet Lowlands, where it was 1.3 percent (fig. 2.4).

Meanwhile, defoliators were detected on a much larger area of Alaska during 2015, with 11 defoliating agents recorded on approximately 170 400 ha (table 2.5). Of this area, about 66 percent (113 000 ha) consisted of unknown defoliators. Aspen leafminer (*Phyllocnistis populiella*) was detected on about 32 000 ha, while willow leaf blotchminer (*Micrurapteryx salicifoliella*) was recorded on 13 400 ha, and large aspen tortrix was found on almost 8300 ha.

The Alaska ecoregion with the highest proportion of surveyed forest area affected by defoliators in 2015 was 135A–Copper River Basin in the southeastern part of the State (5.1 percent of surveyed forest), where extensive areas of aspen leafminer defoliation were detected in quaking aspen (*Populus tremuloides*) forests (fig. 2.5). Several ecoregions in the central and southwestern parts of Alaska had moderate levels of detected defoliation, including M213A–Northern Aleutian Range (3.8 percent, unknown defoliators in hardwood stands), 129B–Yukon-Kuskokwim Delta (3.1 percent, unknown defoliators in hardwood stands), and 139A–Yukon Flats (2.8 percent, willow leaf blotchminer in willow [*Salix* spp.] stands).

In 2015, 244 300 ha of forest on the Big Island of Hawai'i was surveyed for mortality associated with rapid 'ōhi'a death, a disease caused by the fungal pathogen *Ceratocystis fimbriata* that affects 'ōhi'a (*Metrosideros polymorpha*), a highly

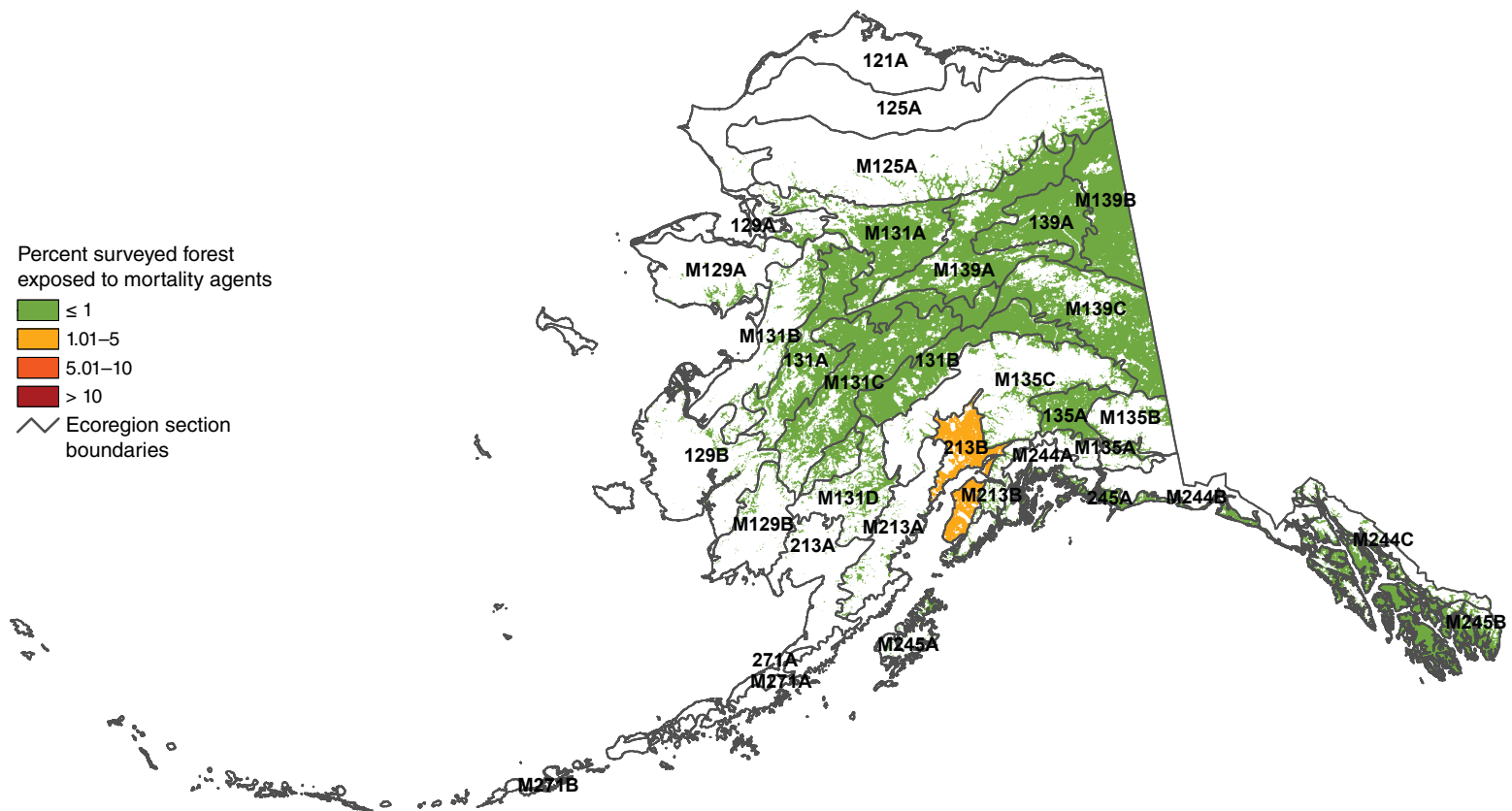


Figure 2.4—Percentage of surveyed forest in Alaska ecoregion sections exposed to mortality-causing insects and diseases in 2015. The gray lines delineate ecoregion sections (Nowacki and Brock 1995). Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: U.S. Department of Agriculture, Forest Service, Forest Health Protection)



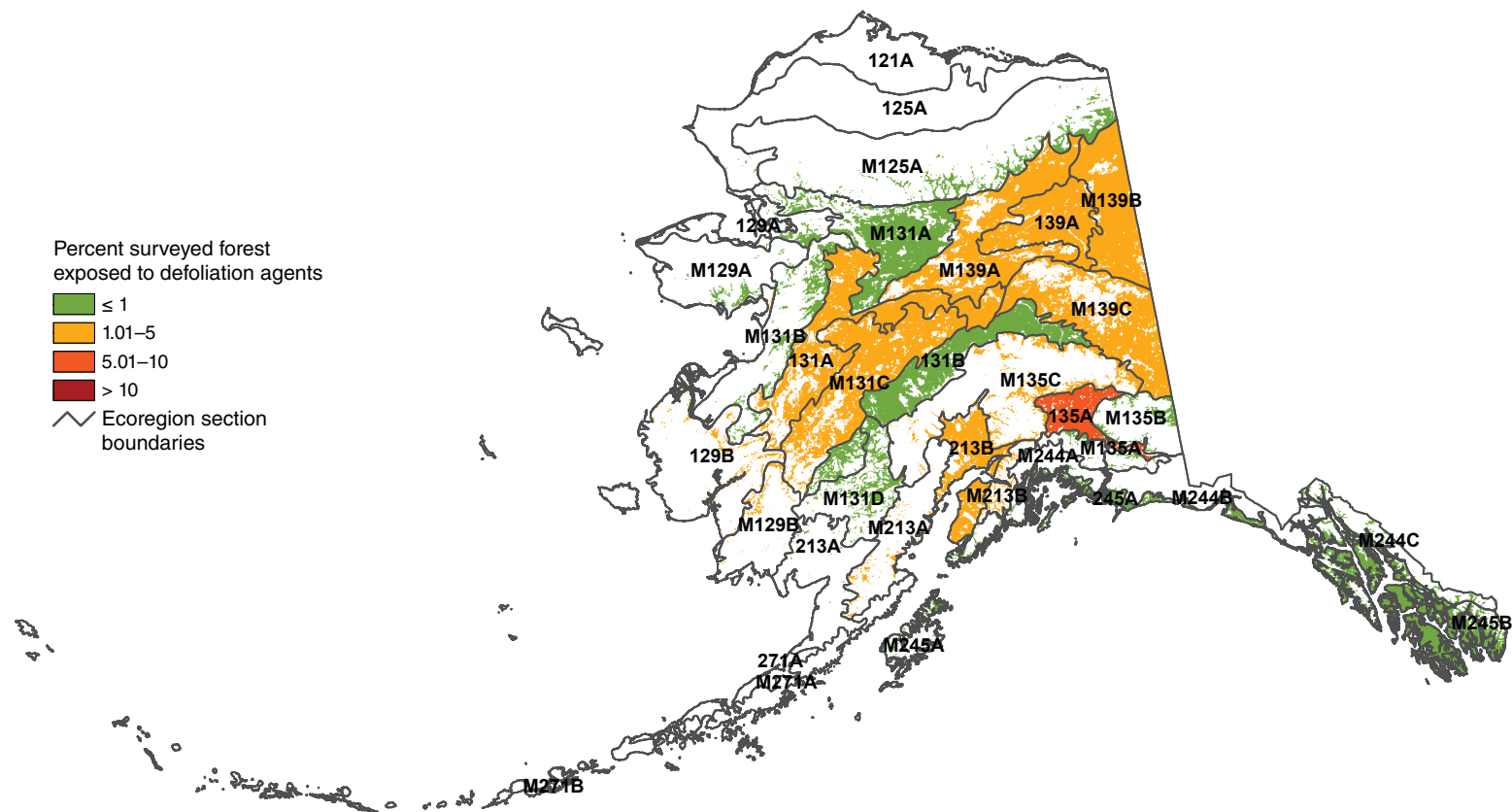


Figure 2.5—Percentage of surveyed forest in Alaska ecoregion sections exposed to defoliation-causing insects and diseases in 2015. The gray lines delineate ecoregion sections (Nowacki and Brock 1995). Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: U.S. Department of Agriculture, Forest Service, Forest Health Protection)

ecologically and culturally important tree species in Hawaiʻian native forests (University of Hawaiʻi 2016). The survey detected approximately 18 000 ha of mortality caused by rapid ʻōhiʻa death (table 2.3), with at least some mortality present in all districts of the island. A higher percentage of surveyed forest was affected by the pathogen in the southeastern part of the island (fig. 2.6), where the pathogen was first identified. Specifically, 17.8 and 15.9 percent of the surveyed forest in council districts

4 and 5, respectively (roughly approximating the Puna region), were affected by rapid ʻōhiʻa death, followed by the neighboring district 3 (South Hilo, 8.1 percent) and district 6 (Kau and South Kona, 6.2 percent). The lowest percentage of mortality was detected on the west and northwest parts of the island farthest from Puna, in districts 8 and 7 (North Kona, 0.6 and 3.4 percent, respectively) and district 9 (North and South Kohala, 1.5 percent).

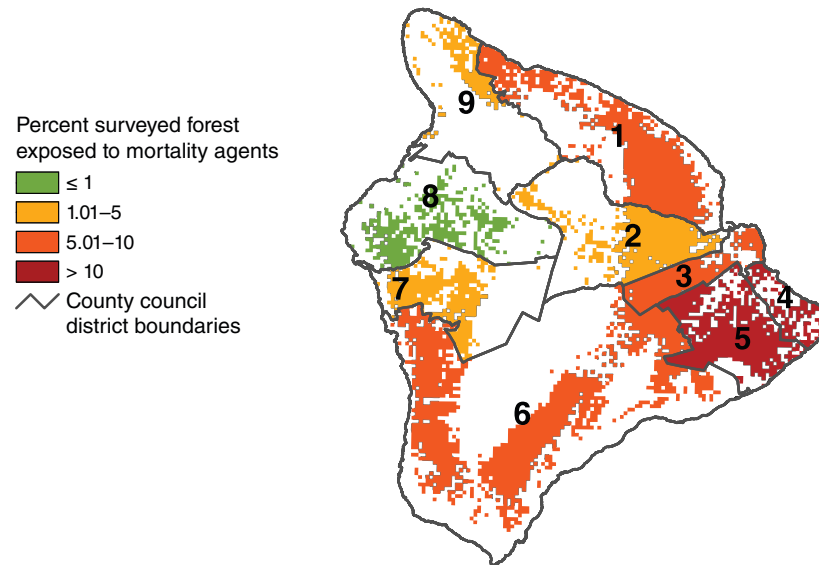


Figure 2.6—Percentage of surveyed forest on the Big Island of Hawaiʻi exposed to mortality-causing rapid ʻōhiʻa death in 2015, by county council district (State of Hawaiʻi 2012). Background forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Data source: U.S. Department of Agriculture Forest Service, Forest Health Protection)

## CONCLUSION

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

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## INTRODUCTION

As a pervasive disturbance agent operating at many spatial and temporal scales, wildland fire is a key abiotic factor affecting forest health both positively and negatively. In some ecosystems, for example, wildland fires have been essential for regulating processes that maintain forest health (Lundquist and others 2011). Wildland fire is an important ecological mechanism that shapes the distributions of species, maintains the structure and function of fire-prone communities, and acts as a significant evolutionary force (Bond and Keeley 2005). At the same time, wildland fires have created forest health problems in some ecosystems (Edmonds and others 2011). Specifically, fire outside the historic range of frequency and intensity can impose extensive ecological and socioeconomic impacts. Current fire regimes on more than half of the forested area in the conterminous United States have been moderately or significantly altered from historical regimes, potentially altering key ecosystem components such as species composition, structural stage, stand age, canopy closure, and fuel loadings (Schmidt and others 2002). As a result of intense suppression efforts during most of the 20th century, the forest area burned annually decreased from approximately 16–20 million ha (40–50 million acres) in the early 1930s to about 2 million ha (5 million acres) in the 1970s (Vinton 2004). Understanding existing fire regimes is essential to properly assessing the impact of fire on forest health because changes to historical fire

regimes can alter forest developmental patterns, including the establishment, growth, and mortality of trees (Lundquist and others 2011).

Fire regimes have been dramatically altered by fire suppression (Barbour and others 1999) and by the introduction of nonnative invasive plants, which can change fuel properties and in turn both affect fire behavior and alter fire regime characteristics such as frequency, intensity, type, and seasonality (Brooks and others 2004). Fires in some regions and ecosystems have become larger, more intense, and more damaging because of the accumulation of fuels as a result of prolonged fire suppression (Pyne 2010). Such large wildland fires also can have long-lasting social and economic consequences, which include the loss of human life and property, smoke-related human health impacts, and the economic cost and dangers of fighting the fires themselves (Gill and others 2013, Richardson and others 2012). In some regions, plant communities have experienced or are undergoing rapid compositional and structural changes as a result of fire suppression (Nowacki and Abrams 2008). Additionally, changes in fire intensity and recurrence could result in decreased forest resilience and persistence (Lundquist and others 2011), and fire regimes altered by global climate change could cause large-scale shifts in vegetation spatial patterns (McKenzie and others 1996).

This chapter presents analyses of fire occurrence data, collected nationally each day by satellite, that map and quantify where fire

## CHAPTER 3.

### Large-Scale Patterns of Forest Fire Occurrence in the Conterminous United States, Alaska, and Hawai'i, 2015

KEVIN M. POTTER

occurrences were concentrated spatially across the conterminous United States, Alaska, and Hawai'i in 2015. It also compares 2015 fire occurrences, within a geographic context, to all the recent years for which such data are available. Quantifying and monitoring such medium-scale patterns of fire occurrence across the United States can help improve our understanding of the ecological and economic impacts of fire as well as the appropriate management and prescribed use of fire. Specifically, large-scale assessments of fire occurrence can help identify areas where specific management activities may be needed, or where research into the ecological and socioeconomic impacts of fires may be required.

## METHODS

### Data

Annual monitoring and reporting of active wildland fire events using the Moderate Resolution Imaging Spectroradiometer (MODIS) Active Fire Detections for the United States database (USDA Forest Service 2016) allows analysts to spatially display and summarize fire occurrences across broad geographic regions (Coulston and others 2005; Potter 2012a, 2012b, 2013a, 2013b, 2014, 2015a, 2015b, 2016). A fire occurrence is defined as one daily satellite detection of wildland fire in a 1-km<sup>2</sup> pixel, with multiple fire occurrences possible on a pixel across multiple days resulting from a single wildland fire that lasts more than a single day. The data are derived using the MODIS Rapid Response System (Justice and others 2002, 2011) to extract fire location and intensity

information from the thermal infrared bands of imagery collected daily by two satellites at a resolution of 1 km<sup>2</sup>, with the center of a pixel recorded as a fire occurrence (USDA Forest Service 2016). The Terra and Aqua satellites' MODIS sensors identify the presence of a fire at the time of image collection, with Terra observations collected in the morning and Aqua observations collected in the afternoon. The resulting fire occurrence data represent only whether a fire was active because the MODIS data bands do not differentiate between a hot fire in a relatively small area (0.01 km<sup>2</sup>, for example) and a cooler fire over a larger area (1 km<sup>2</sup>, for example). The MODIS Active Fire database does well at capturing large fires during cloud-free conditions but may underrepresent rapidly burning, small, and low-intensity fires, as well as fires in areas with frequent cloud cover (Hawbaker and others 2008). For large-scale assessments, the dataset represents a good alternative to the use of information on ignition points, which may be preferable but can be difficult to obtain or may not exist (Tonini and others 2009). For more information about the performance of this product, see Justice and others (2011).

It is important to underscore that estimates of burned area and calculations of MODIS-detected fire occurrences are two different metrics for quantifying fire activity within a given year. Most importantly, the MODIS data contain both spatial and temporal components because persistent fire will be detected repeatedly over several days on a given 1-km<sup>2</sup> pixel. In other words, a location can be counted as having a



fire occurrence multiple times, once for each day a fire is detected at the location. Analyses of the MODIS-detected fire occurrences, therefore, measure the total number of daily 1-km<sup>2</sup> pixels with fire during a year, as opposed to quantifying only the area on which fire occurred at some point during the course of the year. A fire detected on a single pixel on every day of the year would be equivalent to 365 fire occurrences.

### Analyses

These MODIS products for 2015 were processed in ArcMap<sup>®</sup> (ESRI 2012) to determine number of fire occurrences per 100 km<sup>2</sup> (10 000 ha) of forested area for each ecoregion section in the conterminous 48 States (Cleland and others 2007) and Alaska (Nowacki and Brock 1995), and for each of the major islands of Hawai'i. This forest fire occurrence density measure was calculated after screening out wildland fires on nonforested pixels using a forest cover layer derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center (RSAC) (USDA Forest Service 2008). The total numbers of forest fire occurrences were also determined separately for the conterminous States, Alaska, and Hawai'i.

The fire occurrence density value for each ecoregion and island in 2015 was then compared with the mean fire density values for the first 14 full years of MODIS Active Fire data collection (2001–14). Specifically, the difference of the 2015 value and the previous 14-year mean for an ecoregion was divided by the standard

deviation across the previous 14-year period, assuming normal distribution of fire density over time in the ecoregion. The result for each ecoregion was a standardized z-score, which is a dimensionless quantity describing the degree to which the fire occurrence density in the ecoregion in 2015 was higher, lower, or the same relative to all the previous years for which data have been collected, accounting for the variability in the previous years. The z-score is the number of standard deviations between the observation and the mean of the previous observations. Approximately 68 percent of observations would be expected within one standard deviation of the mean, and 95 percent within two standard deviations. Near-normal conditions are classified as those within a single standard deviation of the mean, although such a threshold is somewhat arbitrary. Conditions between about one and two standard deviations of the mean are moderately different from mean conditions, but are not significantly different statistically. Those outside about two standard deviations would be considered statistically greater than or less than the long-term mean (at  $p < 0.025$  at each tail of the distribution).

Additionally, we used the Spatial Association of Scalable Hexagons (SASH) analytical approach to identify forested areas in the conterminous 48 States with higher-than-expected fire occurrence density in 2015. This method identifies locations where ecological phenomena occur at greater or lower occurrences than expected by random chance and is based on a sampling frame optimized for spatial neighborhood analysis,

adjustable to the appropriate spatial resolution, and applicable to multiple data types (Potter and others 2016). Specifically, it consists of dividing an analysis area into scalable equal-area hexagonal cells within which data are aggregated, followed by identifying statistically significant geographic clusters of hexagonal cells within which mean values are greater or less than those expected by chance. To identify these clusters, we employed a Getis-Ord ( $G_i^*$ ) hot spot analysis (Getis and Ord 1992) in ArcMap® 10.1 (ESRI 2012).

The spatial units of analysis were 9,810 hexagonal cells, each approximately 834 km<sup>2</sup> in area, generated in a lattice across the conterminous United States using intensification of the Environmental Monitoring and Assessment Program (EMAP) North American hexagon coordinates (White and others 1992). These coordinates are the foundation of a sampling frame in which a hexagonal lattice was projected onto the conterminous United States by centering a large base hexagon over the region (Reams and others 2005, White and others 1992). This base hexagon can be subdivided into many smaller hexagons, depending on sampling needs, and serves as the basis of the plot sampling frame for the FIA program (Reams and others 2005). Importantly, the hexagons maintain equal areas across the study region regardless of the degree of intensification of the EMAP hexagon coordinates. In addition, the hexagons are compact and uniform in their distance to the centroids of neighboring hexagons, meaning that a hexagonal lattice has a higher degree of

isotropy (uniformity in all directions) than does a square grid (Shima and others 2010). These are convenient and highly useful attributes for spatial neighborhood analyses. These scalable hexagons also are independent of geopolitical and ecological boundaries, avoiding the possibility of different sample units (such as counties, States, or watersheds) encompassing vastly different areas (Potter and others 2016). We selected hexagons 834 km<sup>2</sup> in area because this is a manageable size for making monitoring and management decisions in nationwide analyses (Potter and others 2016).

Fire occurrence density values for each hexagon were quantified as the number of forest fire occurrences per 100 km<sup>2</sup> of forested area within the hexagon. The Getis-Ord  $G_i^*$  statistic was used to identify clusters of hexagonal cells with fire occurrence density values higher than expected by chance. This statistic allows for the decomposition of a global measure of spatial association into its contributing factors, by location, and is therefore particularly suitable for detecting outlier assemblages of similar conditions in a dataset, such as when spatial clustering is concentrated in one subregion of the data (Anselin 1992).

Briefly,  $G_i^*$  sums the differences between the mean values in a local sample, determined in this case by a moving window of each hexagon and its 18 first- and second-order neighbors (the 6 adjacent hexagons and the 12 additional hexagons contiguous to those 6) and the global mean of all the forested hexagonal cells in the conterminous 48

States.  $G_i^*$  is standardized as a z-score with a mean of 0 and a standard deviation of 1, with values  $> 1.96$  representing significant local clustering of higher fire occurrence densities ( $p < 0.025$ ) and values  $< -1.96$  representing significant clustering of lower fire occurrence densities ( $p < 0.025$ ), because 95 percent of the observations under a normal distribution should be within approximately 2 standard deviations of the mean (Laffan 2006). Values between  $-1.96$  and  $1.96$  have no statistically significant concentration of high or low values; a hexagon and its 18 neighbors, in other words, have a normal range of both high and low numbers of fire occurrences per 100 km<sup>2</sup> of forested area. It is worth noting that the threshold values are not exact because the correlation of spatial data violates the assumption of independence required for statistical significance (Laffan 2006). In addition, the Getis-Ord approach does not require that the input data be normally distributed, because the local  $G_i^*$  values are computed under a randomization assumption, with  $G_i^*$  equating to a standardized z score that asymptotically tends to a normal distribution (Anselin 1992). The z scores are reliable, even with skewed data, as long as the distance band is large enough to include several neighbors for each feature (ESRI 2012).

## RESULTS AND DISCUSSION

The MODIS Active Fire database recorded 81,435 wildland forest fire occurrences across the conterminous United States in 2015, the sixth largest annual number of fire occurrences

since the first full year of data collection in 2001 and the smallest number since 2011 (fig. 3.1). This was approximately 23 percent fewer than in 2014 (106,242 total forest fire occurrences), and about 25 percent more than the annual mean of 64,960 forest fire occurrences across the previous 14 full years of data collection. In contrast, the MODIS database captured 21,466 forest fire occurrences in Alaska in 2015, about 2275-percent more than the preceding year (904) and about 91-percent higher than the previous 14-year annual mean of 11,243. It was also the fourth highest annual number of fire occurrences recorded for Alaska. Meanwhile, Hawai'i had only 904 fire occurrences in 2015, down nearly 50 percent from the previous year (1,797), but still 170-percent higher than the average 335 fire occurrences over the previous 14 years.

The decrease in the total number of fire occurrences across the United States is somewhat inconsistent with the official wildland fire statistics (National Interagency Coordination Center 2016). In 2015, 68,151 wildfires were reported nationally, compared to 63,612 the previous year. The area burned nationally in 2015 (4 097 502 ha) was 145 percent of the 10-year average, with 52 fires exceeding 16 187 ha (43 more than in 2014) (National Interagency Coordination Center 2016). The total area burned nationally represented a 182-percent increase from 2014 (1 455 092 ha) (National Interagency Coordination Center 2015). As noted in the Methods section, such

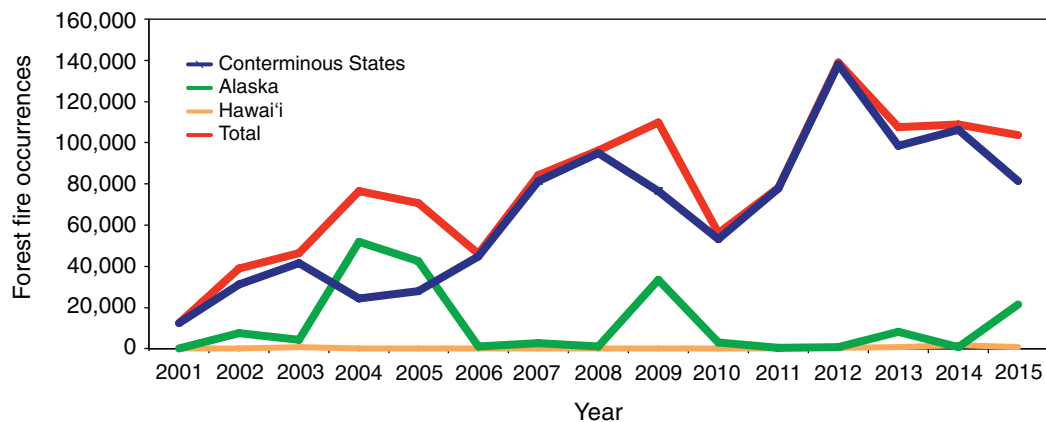


Figure 3.1—Forest fire occurrences detected by MODIS from 2001 through 2015 for the conterminous United States, Alaska, and Hawai'i, and for the entire Nation combined. (Data source: U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center, in conjunction with the NASA MODIS Rapid Response group)

estimates of burned area are different metrics for quantifying fire activity than calculations of MODIS-detected fire occurrences, though the two may be correlated.

In 2015, the highest forest fire occurrence densities occurred in the interior of the Pacific Northwest States, and near the coast of northwestern California and southeastern Oregon (fig. 3.2). This area experienced a warm and dry winter and spring followed by a very hot and dry summer, which led to exceptional drought conditions and extremely dry fuels; this situation was moderated in other parts of the West by a series of tropical systems that brought much-needed rain (National Interagency Coordination Center 2016).

The five ecoregions with the highest wildland forest fire occurrence density in 2015 are all located within inland areas of the Pacific Northwest (fig. 3.2), with M333A—Okanogan Highland first on the list with extremely high fire occurrences, 59.7 per 100 km<sup>2</sup> of forest. This was the area of at least three very large fires: (1) the human-ignited North Star fire that scorched 88 277 ha between August 13 and November 30 and cost approximately \$45 million to contain; (2) the lightning-caused Kettle Complex of fires that burned 30 963 ha between August 11 and September 27; and (3) the human-caused Carpenter Road fire, which burned 25 889 ha from August 14 to December 16, with a containment cost of \$22.7 million (National Interagency

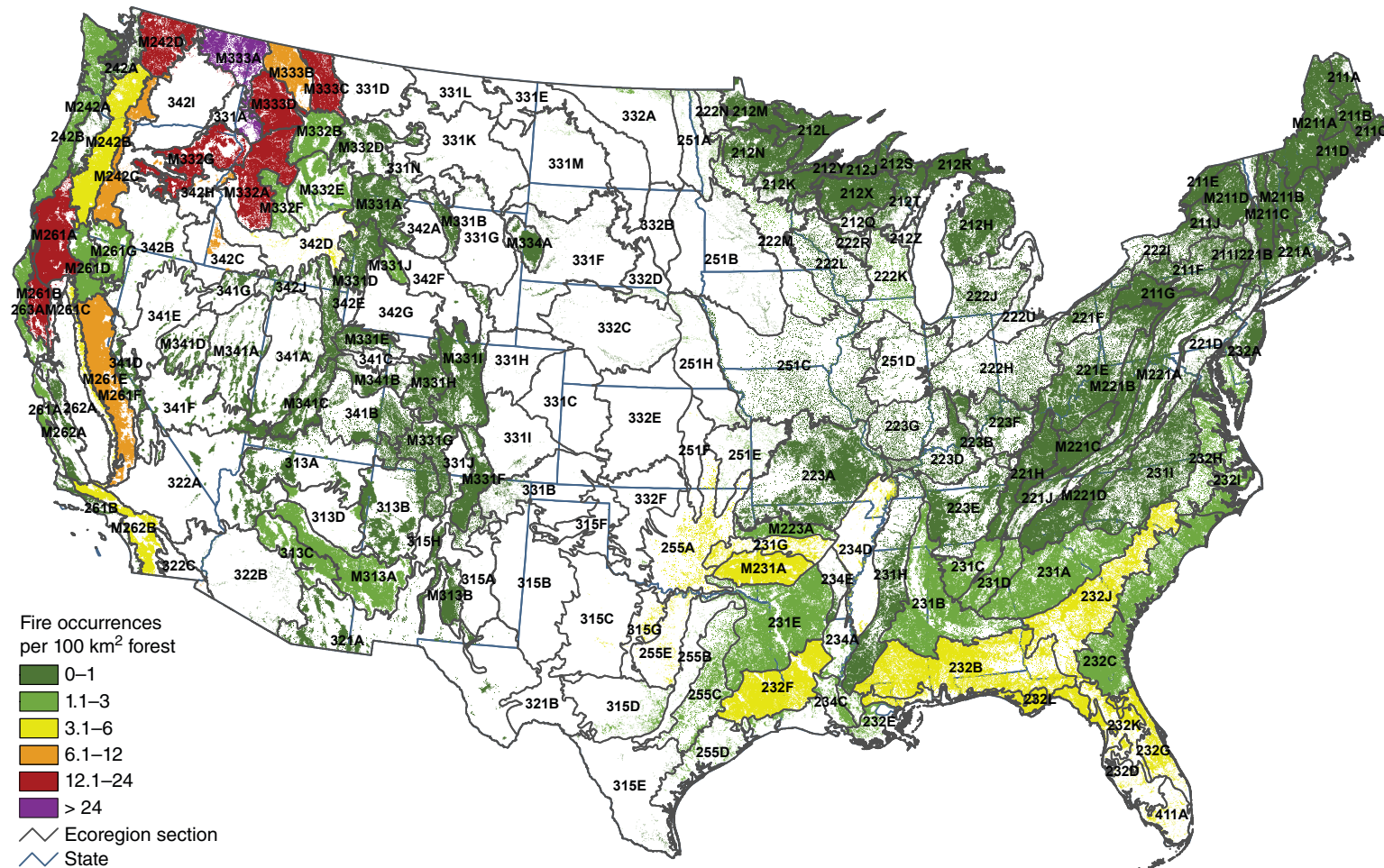


Figure 3.2—The number of forest fire occurrences, per 100 km<sup>2</sup> (10 000 hectares) of forested area, by ecoregion section within the conterminous 48 States, for 2015. The gray lines delineate ecoregion sections (Cleland and others 2007). Forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center, in conjunction with the NASA MODIS Rapid Response group)



Coordination Center 2016). The neighboring M242D–Northern Cascades (19.4 fires per 100 km<sup>2</sup> of forest) encompassed the lightning-ignited Okanogan Complex of fires that scorched 58 794 ha between August 14 and September 29, the Chelan Complex of fires that covered 36 010 ha between August 14 and September 20 and cost \$10 million to contain, and the Wolverine fire that burned 29 187 ha from June 29 until October 3 and cost \$35 million to contain (National Interagency Coordination Center 2016). Other ecoregions with high fire occurrence densities in the region were 331A–Palouse Prairie (36.4 fires), M332G–Blue Mountains (20.6 fires), M333C–Northern Rockies (16.5 fires), M333D–Bitterroot Mountains (14.1 fires), and M332A–Idaho Batholith (13.1 fires).

Three ecoregions in northern California and southwestern Oregon also experienced very high fire occurrence densities: M261A–Klamath Mountains (17.6 fire occurrences), M261B–Northern California Coast Ranges (15.9 fires), and M261C–Northern California Interior Coast Ranges (15.2 fires). The first of these was the location of the River Complex of fires, which cost approximately \$33 million to contain as it scorched 31 394 ha between July 30 and September 25 (National Interagency Coordination Center 2016).

Additionally, moderately high fire occurrence densities were recorded in several other Western ecoregions, both in the Northwest and in California:

- M261E–Sierra Nevada (east-central California), 11.2 fire occurrences;
- M242C–Eastern Cascades (central Oregon and south-central Washington), 9.8 fire occurrences;
- 342H–Blue Mountain Foothills (east-central Oregon), 9.3 fire occurrences;
- 342C–Owyhee Uplands (southwest Idaho and southeast Oregon), 8.0 fire occurrences; and
- M333B–Flathead Valley (northwest Montana), 6.6 fire occurrences.

Several ecoregions with moderate fire occurrence densities stretched across the Southeast, from north-central Texas and central Oklahoma east along the Gulf Coast to much of peninsular Florida as well as central Georgia and South Carolina (fig. 3.2). At the same time, fire densities were low in the Southwestern, Central Rocky Mountain, Midwestern, Mid-Atlantic, and Northeastern States.

Meanwhile, Alaska experienced drought and above-average temperatures in spring 2015, with the resulting limited snowpack leading to fire fuels being exposed earlier than usual; the fire season subsequently extended later than usual into the year (National Interagency Coordination Center 2016). Not surprisingly, much of the interior of the State had moderately high fire occurrence densities during 2015 (fig. 3.3). The ecoregions with the highest values were M131A–Upper Kobuk-Koyukuk (11.8 fire occurrences per 100 km<sup>2</sup> of forest), M139A–Ray Mountains (10.0 fire occurrences), 131A–Yukon Bottomlands (8.1 fire occurrences),

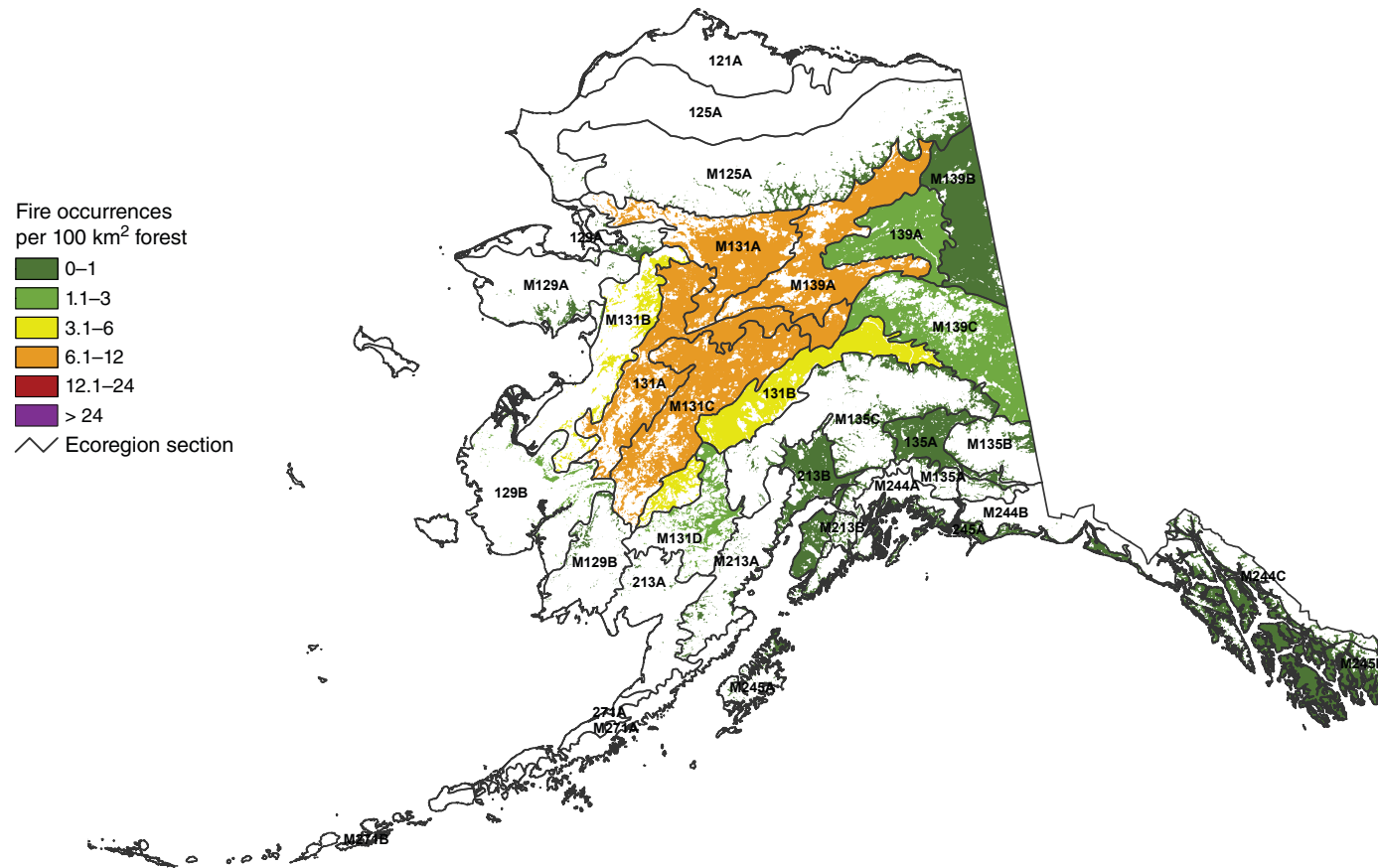


Figure 3.3—The number of forest fire occurrences, per 100 km<sup>2</sup> (10 000 hectares) of forested area, by ecoregion section within Alaska, for 2015. The gray lines delineate ecoregion sections (Nowacki and Brock 1995). Forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center, in conjunction with the NASA MODIS Rapid Response group)



and M131C–Kuskokwim Mountains (7.1 fire occurrences). The two largest U.S. fires in 2015 occurred in central Alaska, the 201 551-ha Tanana Area fire complex and the 170 621-ha Ruby Area fire complex (National Interagency Coordination Center 2016).

In Hawai‘i, lava flows from the 33-year-long eruption of Pu‘u ‘Ō‘ō, a vent on the flank of the Kilauea volcano on the Big Island, were the cause of most forest fire occurrences; specifically, lava flowed into dense forest in the Puna district from August 2014 through March 2015 (West Hawai‘i Today 2016). As a result, fire occurrence density on the Big Island was 60.5 per 100 km<sup>2</sup> of forest in 2015 (fig. 3.4). All the other islands experienced less than one fire per 100 km<sup>2</sup> of forest.

### Comparison to Longer Term Trends

The nature of the MODIS Active Fire data makes it possible to contrast, for each ecoregion section and Hawai‘ian island, short-term (1-year) wildland forest fire occurrence densities with longer term trends encompassing the first 14 full years of data collection (2001–2014). In general, the ecoregions with the highest annual fire occurrence means are located in the Northern Rocky Mountains, the Southwest, California, and the Southeast, while most ecoregions within the Northeastern, Midwestern, Middle Atlantic, and Appalachian regions experienced less than one fire per 100 km<sup>2</sup> of forest annually during the multiyear period (fig. 3.5A). The forested ecoregion that experienced the most fires on average was

M332A–Idaho Batholith in central Idaho (mean annual fire occurrence density of 13.3). Other ecoregions with high mean fire occurrence densities (6.1–12.0 per 100 km<sup>2</sup> of forest) were located along the Gulf Coast and in peninsular Florida in the Southeast, in coastal and central areas of California, in central Arizona and New Mexico, in the Northern Rocky Mountains, and in north-central Texas. Ecoregions with the greatest variation in fire occurrence densities from 2001 to 2014 were also located in central Idaho and near the California coast, with more moderate variation in north-central Washington, southwestern Oregon, northern and central California, western Montana and Wyoming, central Arizona and west-central New Mexico, and eastern North Carolina (fig. 3.5B). Less variation occurred throughout the central and northern Rocky Mountain States, northern Minnesota, the Southeast, and coastal and eastern Oregon and Washington. The lowest levels of variation occurred throughout most of the Midwest and Northeast.

As determined by the calculation of standardized fire occurrence z-scores, ecoregions in the Northern Rocky Mountains, in the Pacific Northwest and northern California, in New England, and in the Great Lakes States experienced greater fire occurrence densities than normal in 2015, compared to the previous 14-year mean and accounting for variability over time (fig. 3.5C). Some of these areas (in New England and the Great Lakes States) had high z-scores despite a relatively low density of fire occurrences in 2015 (fig. 3.2) because

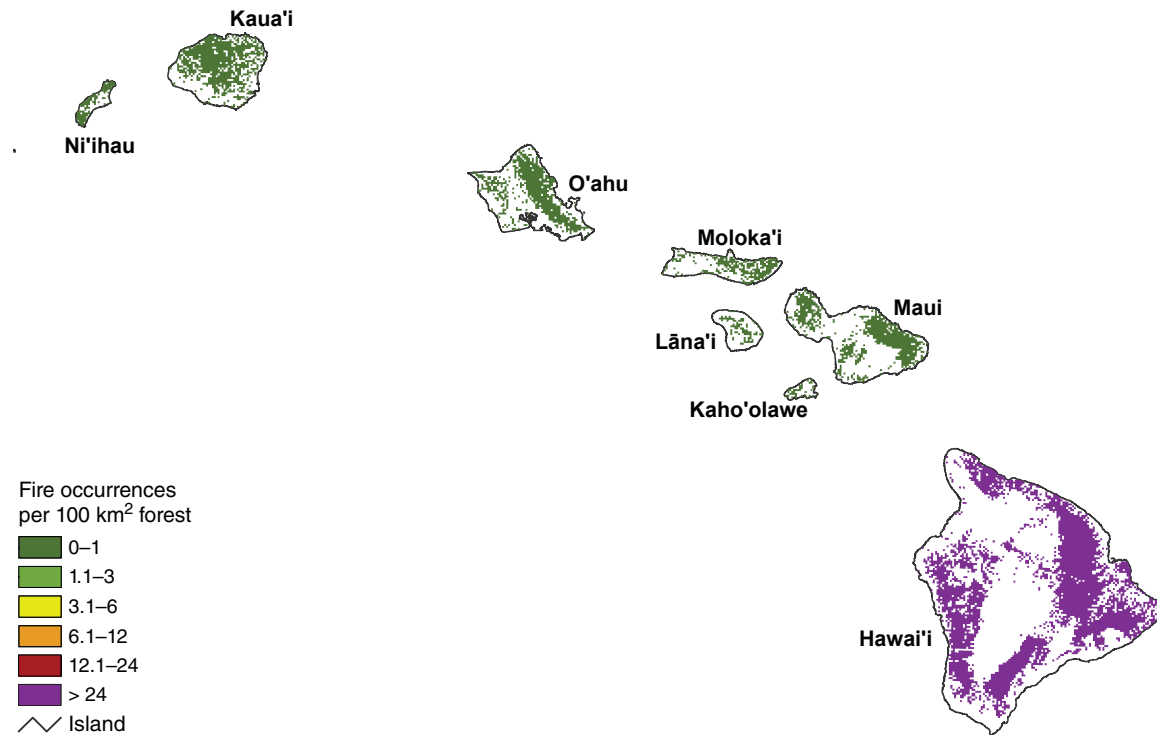


Figure 3.4—The number of forest fire occurrences, per 100 km<sup>2</sup> (10 000 ha) of forested area, by island in Hawai'i, for 2015. Forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center, in conjunction with the NASA MODIS Rapid Response group)

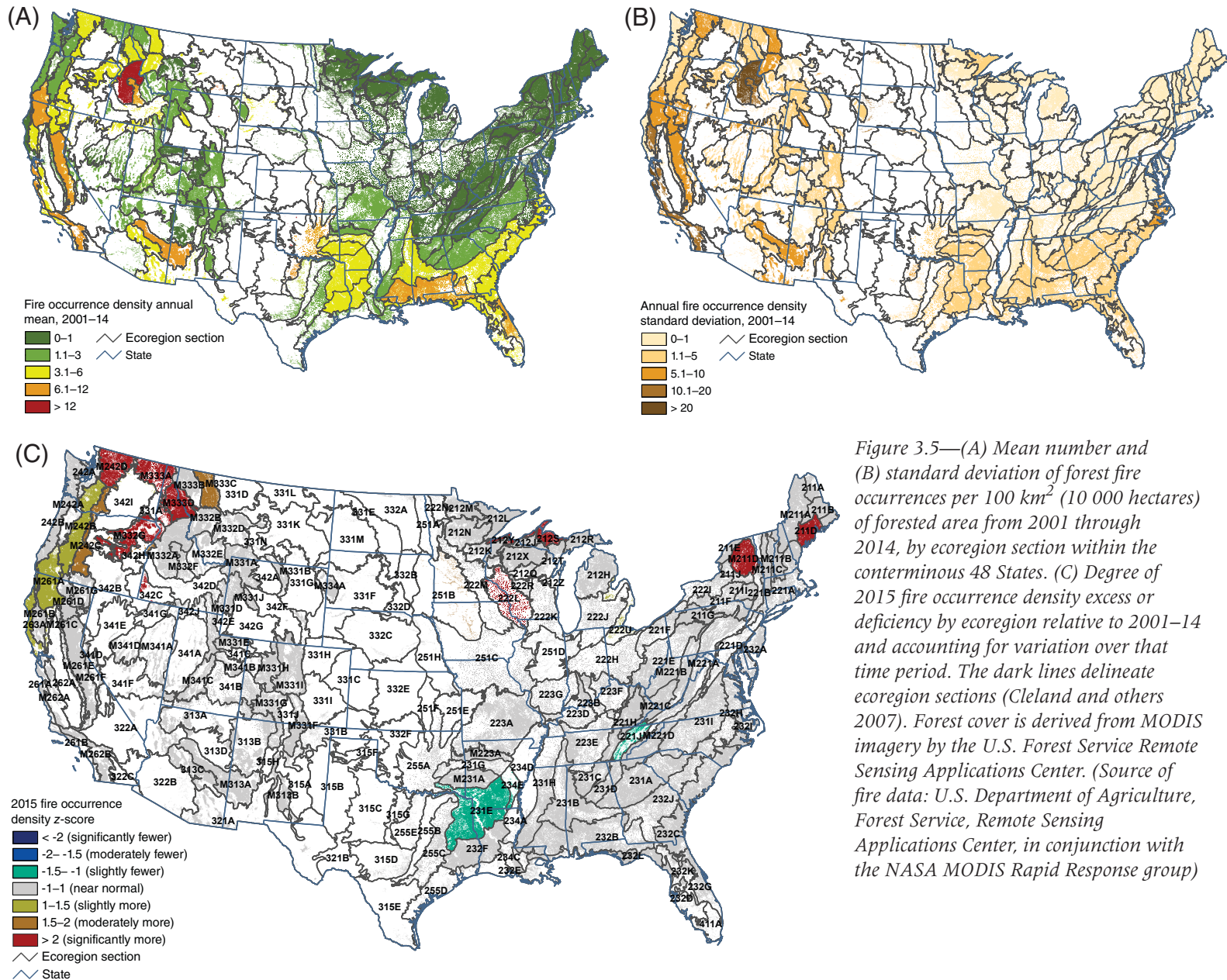


Figure 3.5—(A) Mean number and (B) standard deviation of forest fire occurrences per 100 km<sup>2</sup> (10 000 hectares) of forested area from 2001 through 2014, by ecoregion section within the conterminous 48 States. (C) Degree of 2015 fire occurrence density excess or deficiency by ecoregion relative to 2001–14 and accounting for variation over that time period. The dark lines delineate ecoregion sections (Cleland and others 2007). Forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center, in conjunction with the NASA MODIS Rapid Response group)

these were ecoregions that typically have very little variation over time in fire occurrence density. On the other hand, several of the northwestern ecoregions with high z-scores also had very high fire occurrence densities in 2015 (fig. 3.2), including M333A–Okanagan Highland and M242D–Northern Cascades in northern Washington, M332G–Blue Mountains in northeastern Oregon, M242C–Eastern Cascades in central Oregon and Washington, M261A–Klamath Mountains in northwestern California and southwestern Oregon, M333C–Northern Rockies in northwestern Montana, and M333D–Bitterroot Mountains in northern Idaho (fig. 3.2). Three ecoregions in the Southeast had lower fire occurrence densities in 2015 compared to the longer term: 231E–Mid Coastal Plains–Western in east Texas, southwestern Arkansas, northwestern Louisiana, and southeastern Oklahoma; 234E–Arkansas Alluvial Plains in southeastern Arkansas; and 221J–Central Ridge and Valley in eastern Tennessee. All three had low or very low fire occurrence densities in 2015; the first had a moderate annual mean fire occurrence density and variation from 2001–14, while the second two had low means and variation.

In Alaska, meanwhile, the highest mean annual fire occurrence density between 2001 and 2014 occurred in the east-central and central parts of the State (fig. 3.6A) in the 139A–Yukon Flats ecoregion, with moderate mean fire occurrence density in neighboring areas. As expected, many of those same areas experienced the greatest degree of variability over the 14-year period (fig. 3.6B). In 2015,

several west-central ecoregions, along with one south-central ecoregion, were outside the range of near-normal fire occurrence density, compared to the mean of the previous 14 years and accounting for variability (fig. 3.6C). The ecoregions farthest outside the range of near-normal fire occurrence density were M129B–Ahklun Mountains, M245A–Gulf of Alaska Fjordlands, M131A–Upper Kobuk–Koyukuk, M131B–Nulato Hills, and 129B–Yukon–Kuskokwim Delta. All of these ecoregions had a low or very low mean annual fire occurrence density (fig. 3.6A) and variability (fig. 3.6B) over the previous 14 years.

In Hawai‘i, both the mean annual fire occurrence density (fig. 3.7A) and variability (fig. 3.7B) were highest on the Big Island during the 2001–2014 period. The annual mean was less than 1 fire per 100 km<sup>2</sup> of forest for all islands except the Big Island (11.6) and Kaho‘olawe (1.8). The annual fire occurrence standard deviation exceeded 1 for only the Big Island (17.7), Kaho‘olawe (5.2), and Lāna‘i (1.3). In 2015, the Big Island was well outside the range of near-normal fire occurrence density, controlling for variability over the previous 14 years (fig. 3.7C), with many more fires than expected.

### **Geographical Hot Spots of Fire Occurrence Density**

Although summarizing fire occurrence data at the ecoregion scale allows for the quantification of fire occurrence density across the country, a geographical hot spot analysis can offer insights into where, statistically, fire occurrences are



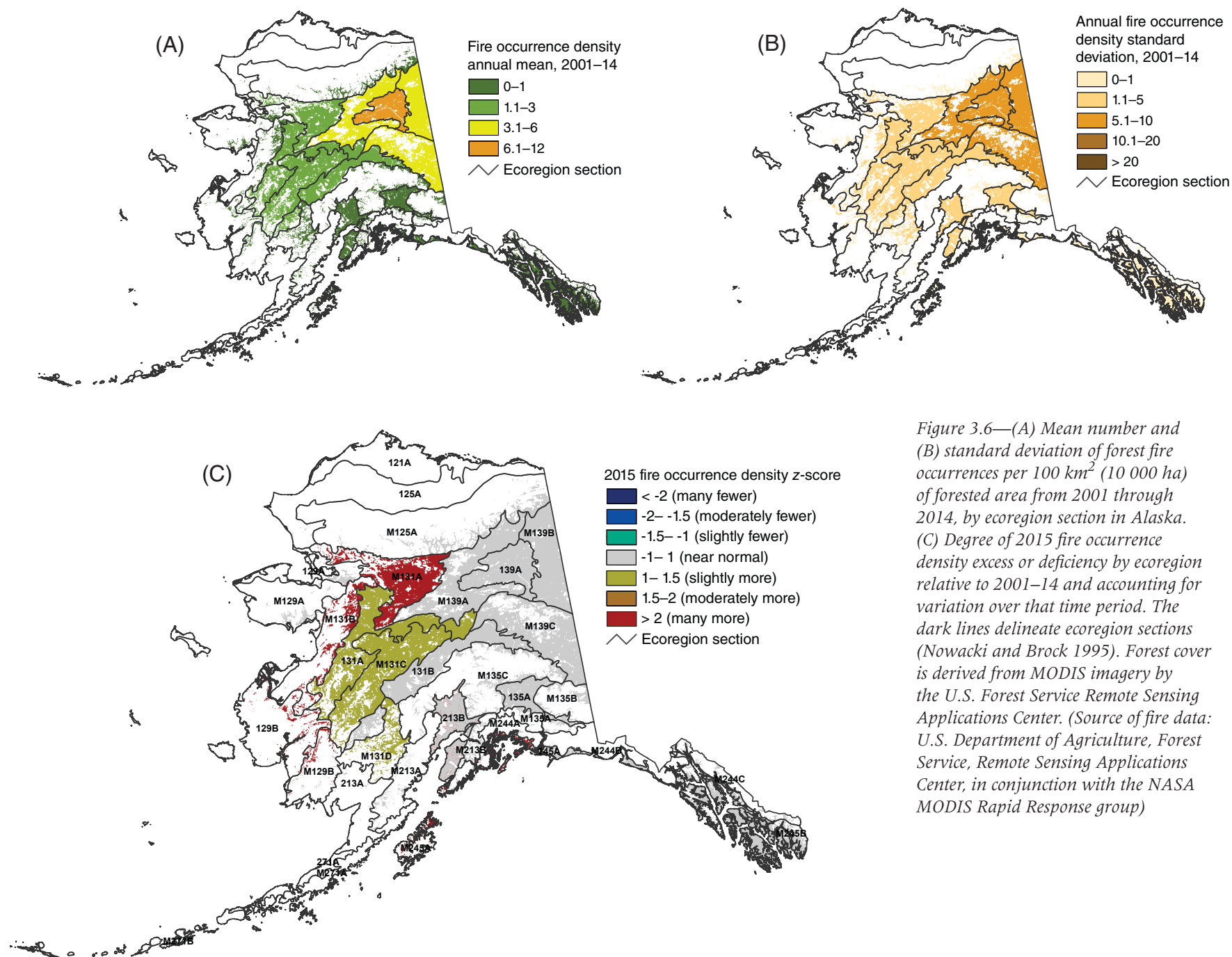


Figure 3.6—(A) Mean number and (B) standard deviation of forest fire occurrences per 100 km<sup>2</sup> (10 000 ha) of forested area from 2001 through 2014, by ecoregion section in Alaska. (C) Degree of 2015 fire occurrence density excess or deficiency by ecoregion relative to 2001–14 and accounting for variation over that time period. The dark lines delineate ecoregion sections (Nowacki and Brock 1995). Forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center, in conjunction with the NASA MODIS Rapid Response group)

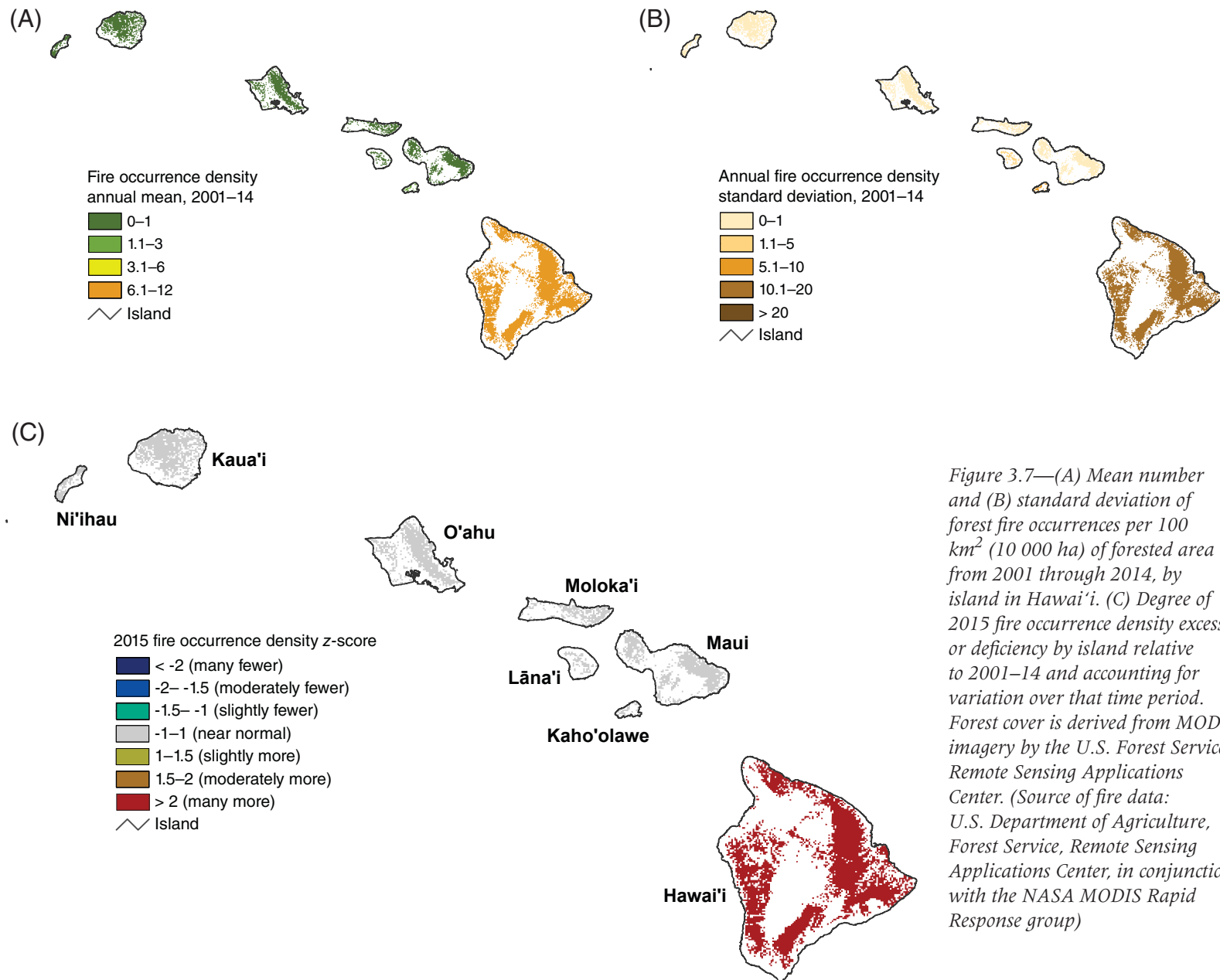


Figure 3.7—(A) Mean number and (B) standard deviation of forest fire occurrences per 100 km<sup>2</sup> (10 000 ha) of forested area from 2001 through 2014, by island in Hawai'i. (C) Degree of 2015 fire occurrence density excess or deficiency by island relative to 2001–14 and accounting for variation over that time period. Forest cover is derived from MODIS imagery by the U.S. Forest Service Remote Sensing Applications Center. (Source of fire data: U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center, in conjunction with the NASA MODIS Rapid Response group)

more concentrated than expected by chance. In 2015, a geographical hot spot of extremely high fire occurrence density was located in north-central Washington, in ecoregions M333A–Okanogan Highland and M242D–Northern Cascades (fig. 3.8). As noted above, this region encompassed numerous very large wildland fires. The analysis detected a hot spot of very high fire occurrence density in east-central Oregon (M332G–Blue Mountains and 342H–Blue Mountain Foothills). This also was associated with a handful of large fires, including the Canyon Creek Complex (44 621 ha) and the Cornet-Windy Ridge fire (41 314 ha).

Hot spots of high fire density were detected in northwestern California (M261A–Klamath Mountains, M261B–Northern California Coast Ranges, and 263A–Northern California Coast), in south-central California (M261E–Sierra Nevada), and in northern Idaho (M332A–Idaho Batholith, 331A–Palouse Prairie, and M333D–Bitterroot Mountains).

Other hot spots of moderate fire density were scattered elsewhere across the Western United States (fig. 3.8), including in the following regions:

- South-central Washington (M242C–Eastern Cascades and M242B–Western Cascades),
- Northwestern Montana (M333C–Northern Rockies),
- Northern Idaho and northwestern Montana (M333B–Flathead Valley and M333D–Bitterroot Mountains and M332G–Blue Mountains), and

- Southwestern Idaho (342C–Owyhee Uplands).

The geographic clustering analysis detected a single moderate-density hot spot in the Southeast, in ecoregion 232B–Gulf Coast Plains and Flatwoods in southwestern Georgia and north-central Florida (fig. 3.8).

## CONCLUSION

The results of these geographic analyses are intended to offer insights into where fire occurrences have been concentrated spatially in a given year and compared to previous years, but are not intended to quantify the severity of a given fire season. Given the limits of MODIS active fire detection using 1-km<sup>2</sup> resolution data, these products also may underrepresent the number of fire occurrences in some ecosystems where small and low-intensity fires are common. These products can also have commission errors. However, these high temporal fidelity products currently offer the best means for daily monitoring wildfire impacts. Ecological and forest health impacts relating to fire and other abiotic disturbances are scale-dependent properties, which in turn are affected by management objectives (Lundquist and others 2011). Information about the concentration of fire occurrences may help pinpoint areas of concern for aiding management activities and for investigations into the ecological and socioeconomic impacts of wildland forest fire potentially outside the range of historic frequency.





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## INTRODUCTION

Droughts affect most forested ecosystems of the United States, although they vary widely in frequency and intensity (Hanson and Weltzin 2000). Most western U.S. forests experience annual seasonal droughts. In contrast, eastern U.S. forests typically exhibit one of two predominant drought patterns: random (i.e., occurring at any time of year) occasional droughts, as observed in the Appalachian Mountains and the Northeast, or frequent late-summer droughts, as commonly seen in the southeastern Coastal Plain and the eastern portion of the Great Plains (Hanson and Weltzin 2000).

In forests, reduced moisture availability during droughts can cause considerable tree stress, especially when this moisture scarcity is accompanied by high temperatures (L.D.L. Anderegg and others 2013, Peters and others 2015, Williams and others 2013). Initially, trees and other plants respond to this stress by decreasing fundamental growth processes such as cell division and enlargement. Photosynthesis is less sensitive than these fundamental processes, and thus decreases slowly at low levels of drought stress, but decreases more sharply as drought stress becomes moderate to severe (Kareiva and others 1993, Mattson and Haack 1987). In addition to such direct effects, drought stress often makes trees susceptible to attack by damaging insects and diseases (Clinton and others 1993, Mattson and Haack 1987, Raffa and others 2008). Droughts also increase wildland fire risk by inhibiting organic matter

decomposition and lowering the moisture content of downed woody debris and other potential fire fuels (Clark 1989, Keetch and Byram 1968, Schoennagel and others 2004, Trouet and others 2010).

In general, forests are resistant to short-term droughts (Archaux and Wolters 2006), although individual tree species differ in their levels of drought resistance. Because of this resistance, the duration of drought events may be more important for forests than their intensity (Archaux and Wolters 2006). For instance, forested areas subjected to multiple consecutive years of drought (2–5 years) are much more likely to experience high tree mortality than areas subjected to a single year of extremely dry conditions (Guarín and Taylor 2005, Millar and others 2007). Therefore, a comprehensive evaluation of drought impact in forests should include analysis of moisture conditions over multiyear time windows.

In the 2010 FHM national report, we described a method for mapping drought conditions across the conterminous United States (Koch and others 2013b). Our objective was to generate fine-scale, drought-related spatial data sets that improve upon similar products available from sources such as the National Climatic Data Center (2015a) or the U.S. Drought Monitor Program (Svoboda and others 2002). The principal inputs are gridded climate data (i.e., monthly raster maps of precipitation and temperature over a 100-year period) created with the Parameter-elevation Regression on Independent Slopes (PRISM)

## CHAPTER 4. Moisture Deficit and Surplus in the Conterminous United States for Three Time Windows: 2015, 2013–2015, and 2011–2015

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climate mapping system (Daly and others 2002). The method utilizes a standardized indexing approach that facilitates comparison of a given location's moisture status during different time windows, regardless of their length. The index is easier to calculate than the commonly used Palmer Drought Severity Index, or PDSI (Palmer 1965), and avoids some criticisms of the PDSI (summarized by Alley 1984) regarding its underlying assumptions and limited comparability across space and time. Here, we applied the method outlined in the 2010 FHM report to the most currently available climate data (i.e., the monthly PRISM data through 2015), thereby providing the seventh installment in an ongoing series of annual drought assessments for the conterminous United States from 2009 forward (Koch and Coulston 2015, 2016; Koch and others 2013a, b, 2014, 2015).

This is the second year in which we also mapped the degree of moisture surplus across the conterminous United States during multiple time windows. Much recent refereed literature (e.g., Adams and others 2009, Allen and others 2010, Martínez-Vilalta and others 2012, Peng and others 2011, Williams and others 2013) has tended to focus on reports of widespread, regional-scale forest decline and mortality due to persistent drought conditions, especially in conjunction with periods of extremely high temperatures (i.e., heat waves). However, surplus moisture availability can also be detrimental to forests. Abnormally high moisture can be a short-term stressor (e.g., an extreme

rainfall event with subsequent flooding) or a long-term stressor (e.g., persistent wetness driven by a macroscale climatic pattern such as the El Niño-Southern Oscillation), either of which may contribute to tree dieback and mortality (Rozas and García-González 2012, Rozas and Sampedro 2013). Such impacts have been observed in both tropical and temperate forests (Laurance and others 2009, Rozas and García-González 2012). While surplus-induced impacts in forests are probably not as common as drought-induced impacts, a single index that depicts both moisture surplus and deficit conditions provides a fuller accounting of potential forest health issues.

## METHODS

We acquired grids for monthly precipitation and monthly mean temperature for the conterminous United States from the PRISM Climate Group Web site (PRISM Climate Group 2016). At the time of these analyses, gridded data sets were available for all years from 1895 to 2015. However, the grids for December 2015 were only provisional versions (i.e., finalized grids had not yet been released). For analytical purposes, we treated these provisional grids as if they were the final versions. The spatial resolution of the grids was approximately 4 km (cell area = 16 km<sup>2</sup>). For future applications and to ensure better compatibility with other spatial data sets, all output grids were resampled to a spatial resolution of approximately 2 km (cell area = 4 km<sup>2</sup>) using a nearest neighbor approach. The nearest neighbor approach is a



computationally simple resampling method that avoids the smoothing of data values observed with methods such as bilinear interpolation or cubic convolution.

### Potential Evapotranspiration Maps

As in our previous drought mapping efforts (Koch and Coulston 2015, 2016; Koch and others 2012a, b; Koch and others 2013a, b, 2014, 2015), we adopted an approach in which a moisture index value is calculated for each location of interest (i.e., each grid cell in a map of the conterminous United States) during a given time period. Moisture indices are intended to reflect the amount of available water in a location (e.g., to support plant growth). In our case, the index is computed using an approach that considers both the amount of precipitation that falls on a location during the period of interest as well as the level of potential evapotranspiration during this period. Potential evapotranspiration measures the loss of soil moisture through plant uptake and transpiration (Akin 1991). It does not measure actual moisture loss, but rather the loss that would occur if there was no possible shortage of moisture for plants to transpire (Akin 1991, Thornthwaite 1948). In simple terms, potential evapotranspiration serves as a measure of moisture demand. By including potential evapotranspiration along with precipitation, our index thus documents the long-term balance between moisture demand and supply for each location of interest.

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

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

where

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

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

$T_m$  = the mean temperature for month  $m$  in degrees C

$I$  = an annual heat index, calculated as

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

where

$T_m$  is the mean temperature for each month  $m$  of the year

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

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

### Moisture Index Maps

To estimate baseline conditions, we used the precipitation (*P*) and *PET* grids to generate moisture index grids for the past 100 years (i.e., 1916–2015) for the conterminous United States. We used a moisture index described by Willmott and Feddema (1992), which has been applied in a variety of contexts, including global vegetation modeling (Potter and Klooster 1999) and climate change analysis (Grundstein 2009). Willmott and Feddema (1992) devised the index as a refinement of one described earlier by Thornthwaite (1948) and Thornthwaite and Mather (1955). Their revised index, *MI'*, has the following form:

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

where

*P* = precipitation

*PET* = potential evapotranspiration, as calculated using equation 1

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

This set of equations yields a symmetric, dimensionless index scaled between -1 and 1. A primary advantage of this symmetry is that it enables valid comparisons between any set of locations in terms of their moisture balance (i.e., the balance between moisture demand and supply). *MI'* can be calculated for any time period, but is commonly calculated on an annual basis using *P* and *PET* values summed across the entire year (Willmott and Feddema 1992). An alternative to this summation approach is to calculate *MI'* on a monthly basis (i.e., from total measured precipitation and estimated potential evapotranspiration in each month), and then, for a given time window of interest, calculate its moisture index as the mean of the *MI'* values for all months in the time window. This “mean-of-months” approach limits the ability of short-term peaks in either precipitation or potential evapotranspiration to negate corresponding short-term deficits, as would happen under a summation approach.

For each year in our study period (i.e., 1916–2015), we used the mean-of-months approach to calculate moisture index grids for three different time windows: 1 year ( $MI_1'$ ), 3 years ( $MI_3'$ ), and 5 years ( $MI_5'$ ). Briefly, the  $MI_1'$  grids are the mean (i.e., the mean value for each grid cell) of the 12 monthly  $MI'$  grids for each year in the study period, the  $MI_3'$  grids are the mean of the 36 monthly grids from January two years prior through December of the target year, and the  $MI_5'$  grids are the mean of the 60 consecutive monthly  $MI'$  grids from January four years prior to December of the target year. Thus, the  $MI_1'$  grid for the year 2015 is the mean of the monthly  $MI'$  grids from January to December 2015, while the  $MI_3'$  grid is the mean of the grids from January 2013 to December 2015 and the  $MI_5'$  grid is the mean of the grids from January 2011 to December 2015.

### Annual and Multiyear Drought Maps

To determine degree of departure from typical moisture conditions, we first created a normal grid,  $MI_{i\text{norm}}'$ , for each of our three time windows, representing the mean (i.e., the mean value for each grid cell) of the 100 corresponding moisture index grids (i.e., the  $MI_1'$ ,  $MI_3'$ , or  $MI_5'$  grids, depending on the window; see fig. 4.1). We also created a standard deviation grid,  $MI_{iSD}'$ , for each time window, calculated from the window's 100 individual moisture index grids as well as its  $MI_{i\text{norm}}'$  grid.

We subsequently calculated moisture difference z-scores,  $MDZ_{ij}$ , for each time window using these derived data sets:

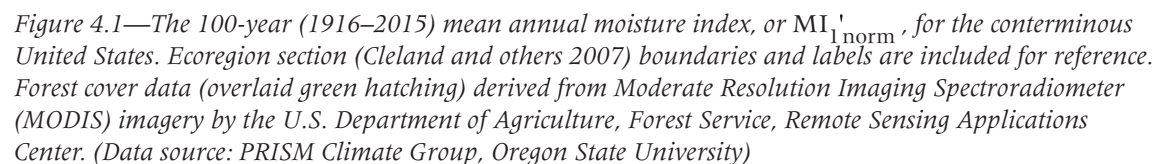
$$MDZ_{ij} = \frac{MI_i' - MI_{i\text{norm}}'}{MI_{iSD}'} \quad (3)$$

where

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

$j$  = a particular target year in our 100-year study period (i.e., 1916–2015).

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



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

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

analysis, we generated and classified *MDZ* maps of the conterminous United States, based on all three time windows, for the target year 2015.

## RESULTS AND DISCUSSION

The 100-year (1916–2015) mean annual moisture index, or  $MI'_{1\text{norm}}$  grid (fig. 4.1) provides an overview of climatic regimes in the conterminous United States. (The 100-year  $MI'_{3\text{norm}}$  and  $MI'_{5\text{norm}}$  grids were very similar to the mean  $MI'_{1\text{norm}}$  grid, and so are not shown here.) Wet climates ( $MI' > 0$ ) are common in the Eastern United States, particularly the Northeast. A noteworthy anomaly is southern Florida, especially ecoregion sections 232D–Florida Coastal Lowlands-Gulf, 232G–Florida Coastal Lowlands-Atlantic, and 411A–Everglades, which appears to be dry relative to other parts of the

East. This is an effect of the region's tropical climate, which has distinct wet (primarily summer months) and dry (late fall to early spring) seasons. Although southern Florida usually receives a high level of precipitation during the wet season, it can be insufficient to offset the region's lengthy dry season (Duever and others 1994) or its high level of evapotranspiration, especially during the late spring and summer months, resulting in negative  $MI'$  values.

The comparatively dry climatic regime of southern Florida is markedly different from the pattern observed in the driest parts of the Western United States, especially the Southwest (e.g., sections 322A–Mojave Desert, 322B–Sonoran Desert, and 322C–Colorado Desert), where potential evapotranspiration is very high, but precipitation levels are very low. In fact, dry climates ( $MI' < 0$ ) are typical across much of the Western United States because of generally lower precipitation than the East. Nevertheless, mountainous areas in the central and northern Rocky Mountains as well as the Pacific Northwest are relatively wet, such as ecoregion sections (Cleland and others 2007) M242A–Oregon and Washington Coast Ranges, M242B–Western Cascades, M331G–South Central Highlands, and M333C–Northern Rockies. This is driven in part by large amounts of winter snowfall in these regions (Hanson and Weltzin 2000).

Figure 4.2 shows the annual (i.e., one-year) *MDZ* map for 2015 for the conterminous United States. Much of the Eastern and Central



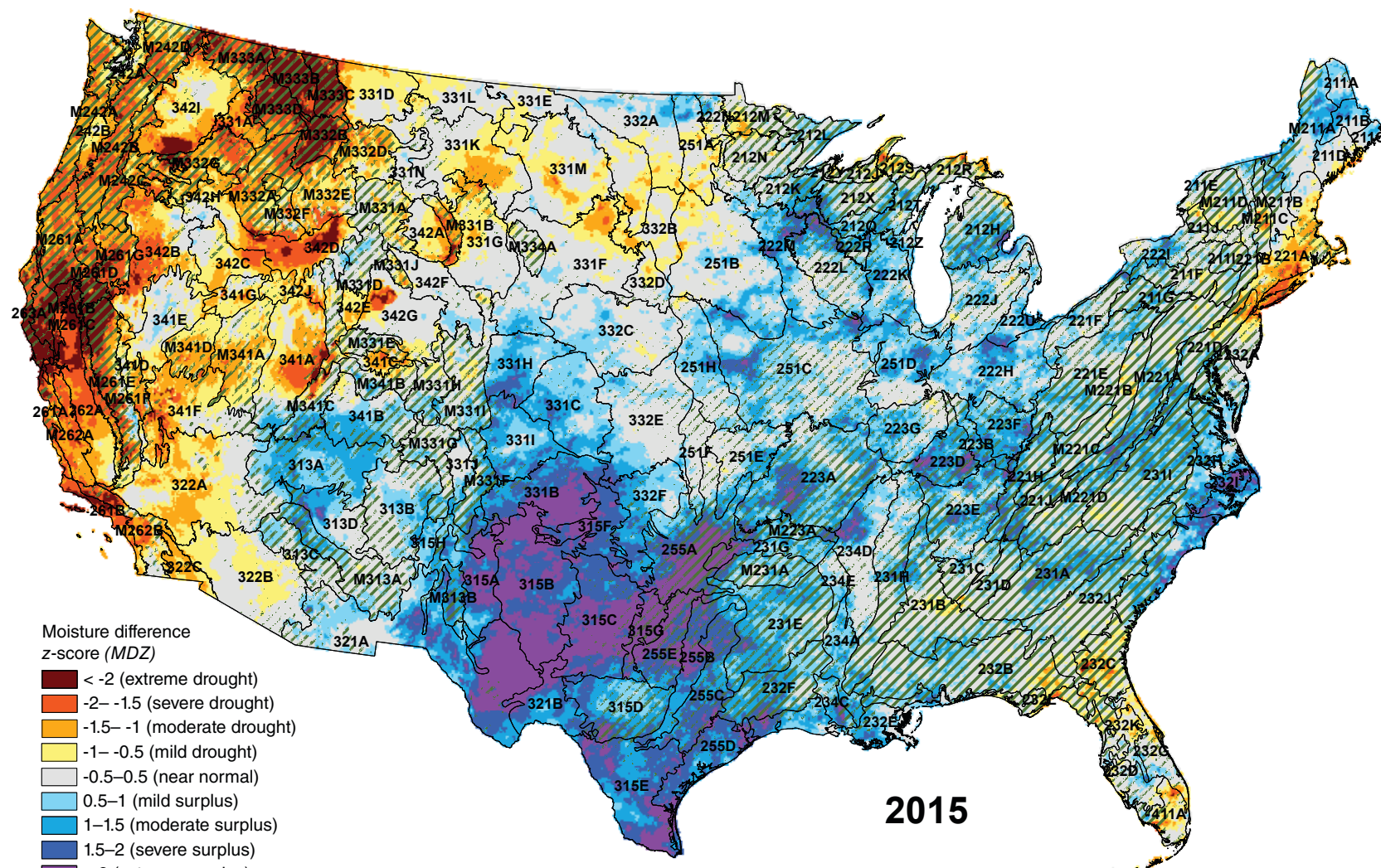


Figure 4.2—The 2015 annual (i.e., 1-year) moisture difference z-score, or MDZ, for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries and labels are included for reference. Forest cover data (overlaid green hatching) derived from MODIS imagery by the U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center. (Data source: PRISM Climate Group, Oregon State University)



United States saw near-normal to moderate surplus moisture conditions during the year. There were a few exceptions to this pattern in the East: an area of mild to severe drought ( $MDZ < -0.5$ ) in Long Island and southern New England (ecoregion section 221A–Lower New England, as well as northern parts of 221D–Northern Appalachian Piedmont and 232A–Northern Atlantic Coastal Plain); a set of small, disconnected hot spots of mild to extreme drought distributed across Florida and southern Georgia (primarily in sections 232B–Gulf Coastal Plains and Flatwoods, 232C–Atlantic Coastal Flatwoods, 232G–Florida Coastal Lowlands–Atlantic, 232K–Florida Coastal Plains Central Highlands, 232L–Gulf Coastal Lowlands, and 411A–Everglades); and hot spots of mild to extreme drought along the southern coast of Lake Superior and St. Mary’s River, which feeds into it (sections 212J–Southern Superior Uplands, 212R–Eastern Upper Peninsula, and 212S–Northern Upper Peninsula).

With respect to the Central United States, areas of mild to severe drought were scattered across mostly nonforested portions of the Northern Great Plains. These are less visually striking than the extensive areas of severe to extreme moisture surplus ( $MDZ > 1.5$ ) in the Southern Great Plains region, covering most of Texas as well as some of Oklahoma and New Mexico. Most of the ecoregion sections in these areas are sparsely forested, except for section 255A–Cross Timbers and Prairie, and affected portions of sections 231G–Arkansas Valley, 255B–Blackland Prairie, 255E–Texas

Cross Timbers and Prairie, and 315G–Eastern Rolling Plains. By contrast, many of these areas experienced mild to extreme drought conditions in 2014 (fig. 4.3). The disparity between the 2014 and 2015 *MDZ* maps in these areas is explained by heavy rains in spring and fall 2015, which partially alleviated the prior drought conditions; indeed, both Oklahoma and Texas experienced their wettest May and October–December periods on record (National Climatic Data Center 2016b). Looking across the entire year, 2015 was the wettest year for both Oklahoma and Texas since 1895, and the fifth wettest for New Mexico (National Climatic Data Center 2016a).

Virtually the entire west coast of the United States, and much of the northern Rocky Mountain region, including parts of Utah and Wyoming, experienced at least moderate drought conditions ( $MDZ < -1$ ) during 2015 (fig. 4.2). Most pronounced were two large, contiguous areas of severe to extreme drought ( $MDZ < -1.5$ ), the first of which included forested portions of sections 261A–Central California Coast, 263A–Northern California Coast, M261A–Klamath Mountains, M261B–Northern California Coast Ranges, M261C–Northern California Interior Coast Ranges, M261D–Southern Cascades, M261E–Sierra Nevada, M261F–Sierra Nevada Foothills, and M261G–Modoc Plateau, while the second was mostly in sections M332B–Northern Rockies and Bitterroot Valley, M333A–Okanogan Highland, M333B–Flathead Valley, M333C–Northern Rockies, and M333D–Bitterroot Mountains.

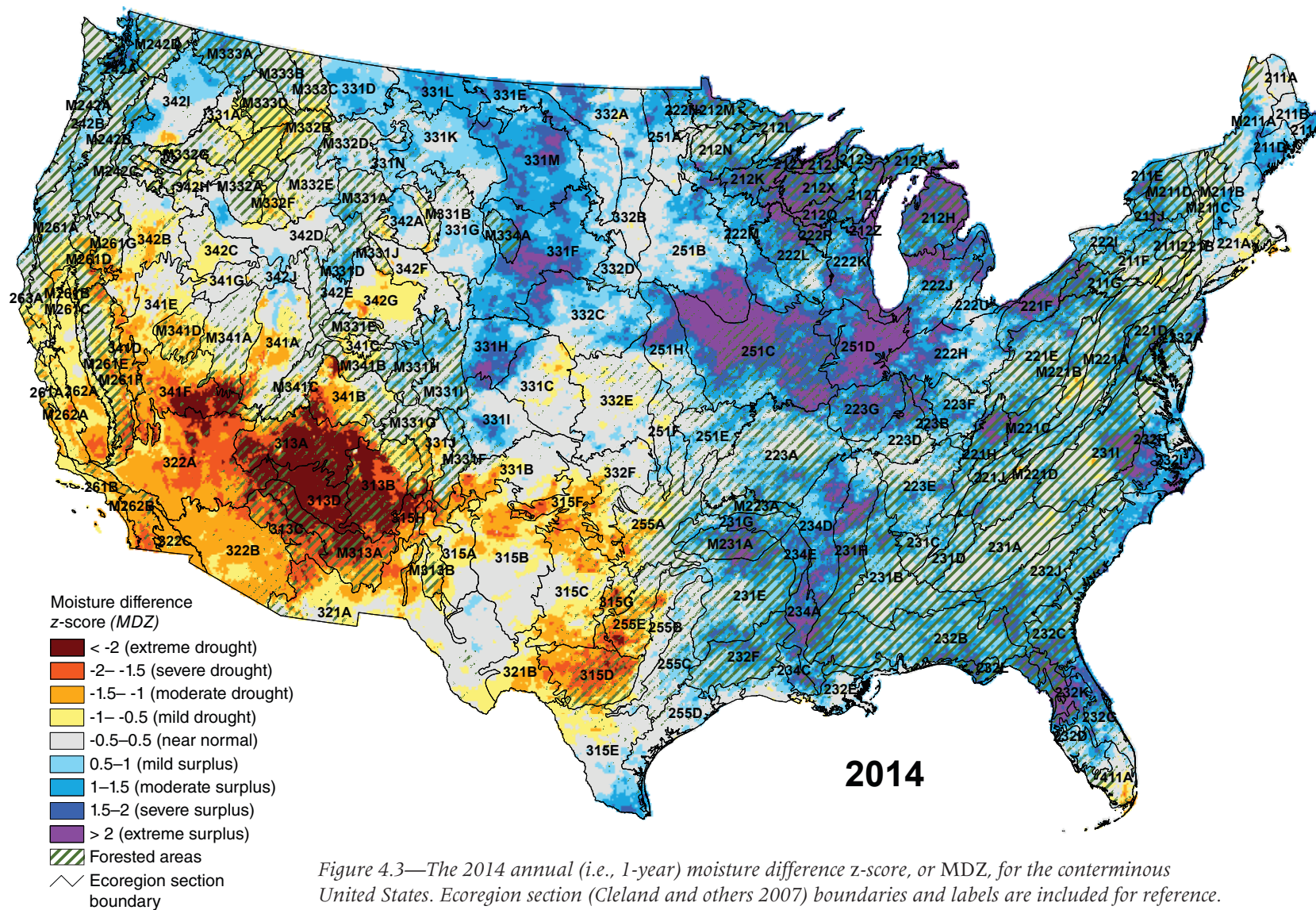


Figure 4.3—The 2014 annual (i.e., 1-year) moisture difference z-score, or MDZ, for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries and labels are included for reference. Forest cover data (overlaid green hatching) derived from MODIS imagery by the U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center. (Data source: PRISM Climate Group, Oregon State University)

This represents a geographic shift for the Western United States as compared to 2014 (see fig. 4.3), when the worst drought conditions in the country occurred in the Southwest region, especially in Arizona, while much of the Pacific Northwest region was unaffected by drought or only experienced mild drought conditions. This shift occurred because the wet conditions noted previously for Texas and New Mexico in 2015 extended similarly across the Southwest, giving the region its wettest year since 1983 and eighth wettest overall (National Climatic Data Center 2016a). At the same time, it was the warmest year on record for the Pacific Northwest, and the high temperatures increased evapotranspiration more than enough to offset the region's relatively normal precipitation levels during 2015 (National Climatic Data Center 2016a).

The widespread drought conditions in California and Nevada during 2015 (fig. 4.2) continued a multiyear trend of severe to extreme drought in these States, particularly in California. As was also the case in 2014, high temperatures played a major role, driving evapotranspiration well above normal levels. Although 2014 was the warmest year on record for both California and Nevada, 2015 was not appreciably cooler: it was the second warmest year on record for California and the third warmest year for Nevada (National Climatic Data Center 2015b, 2016a). Furthermore, storms

that brought heavy precipitation to Northern California during the latter part of 2014, thereby reducing drought severity relative to Southern California and Nevada (fig. 4.3), did not recur in 2015 (National Climatic Data Center 2015b, 2016a, 2016b).

The 3-year (2013–2015; fig. 4.4) and 5-year (2011–2015; fig. 4.5) *MDZ* maps depict the recent history of moisture conditions in the conterminous United States. The 3-year *MDZ* map (fig. 4.4) shows a strong dichotomy between areas west of the Rocky Mountains and the remainder of the country. In the West, nearly all forested areas experienced at least mild drought conditions during this period, most notably California, where extreme drought conditions occurred almost uniformly throughout the State's forested areas. Conversely, most of the Eastern and Central United States experienced at least mild moisture surplus conditions during this period. In fact, areas of severe moisture surplus were widely distributed throughout both the Eastern and Central United States, although many of these areas contain little forest. Deviations from this general pattern of surplus included an area of mild to severe drought in Long Island and along the northern coast of Long Island Sound (primarily in section 221A–Lower New England), as well as an area of mild to extreme drought in southern Florida (primarily in section



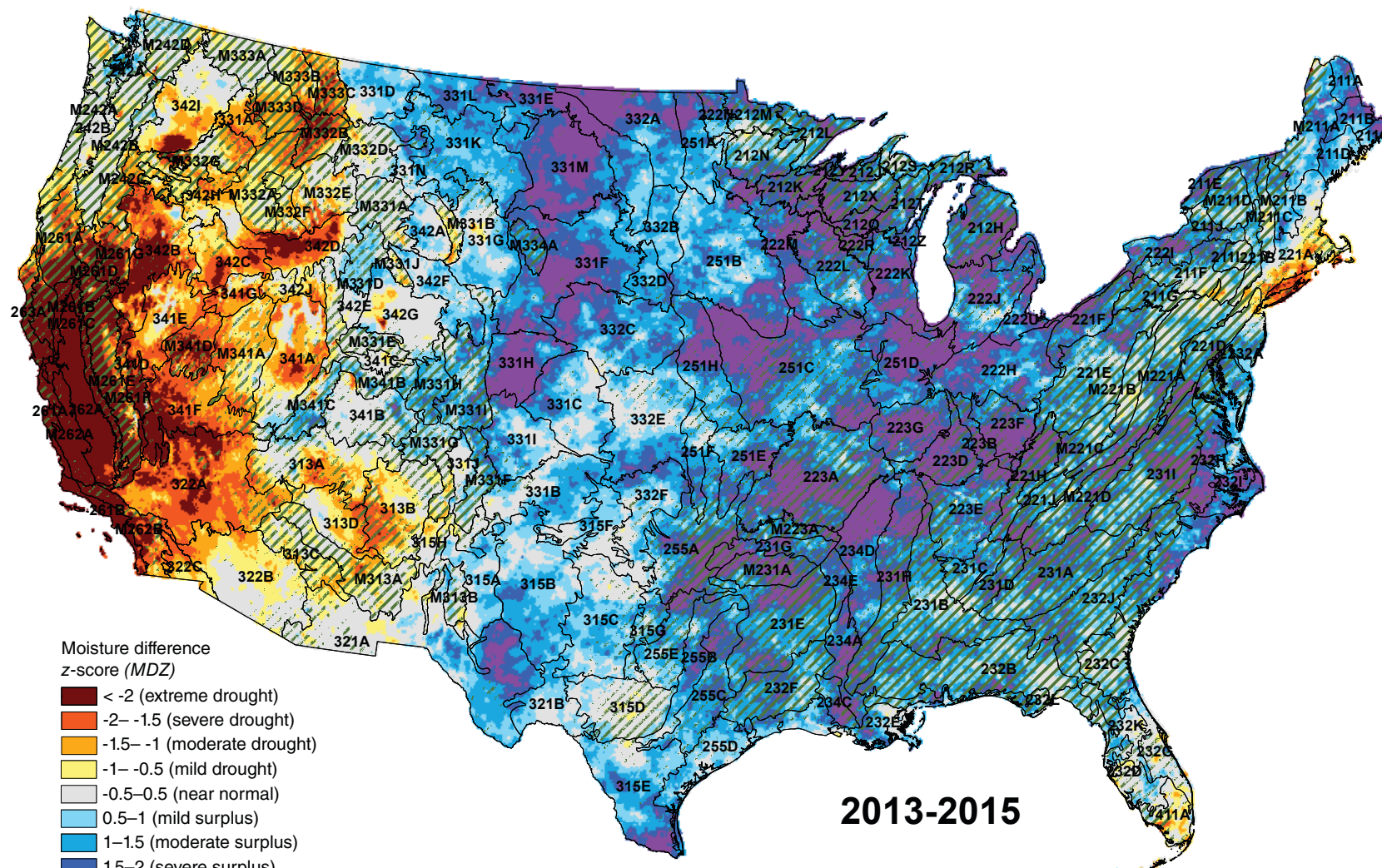


Figure 4.4—The 2013–2015 (i.e., 3-year) moisture difference z-score (MDZ) for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries are included for reference. Forest cover data (overlaid green hatching) derived from MODIS imagery by the U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center. (Data source: PRISM Climate Group, Oregon State University)

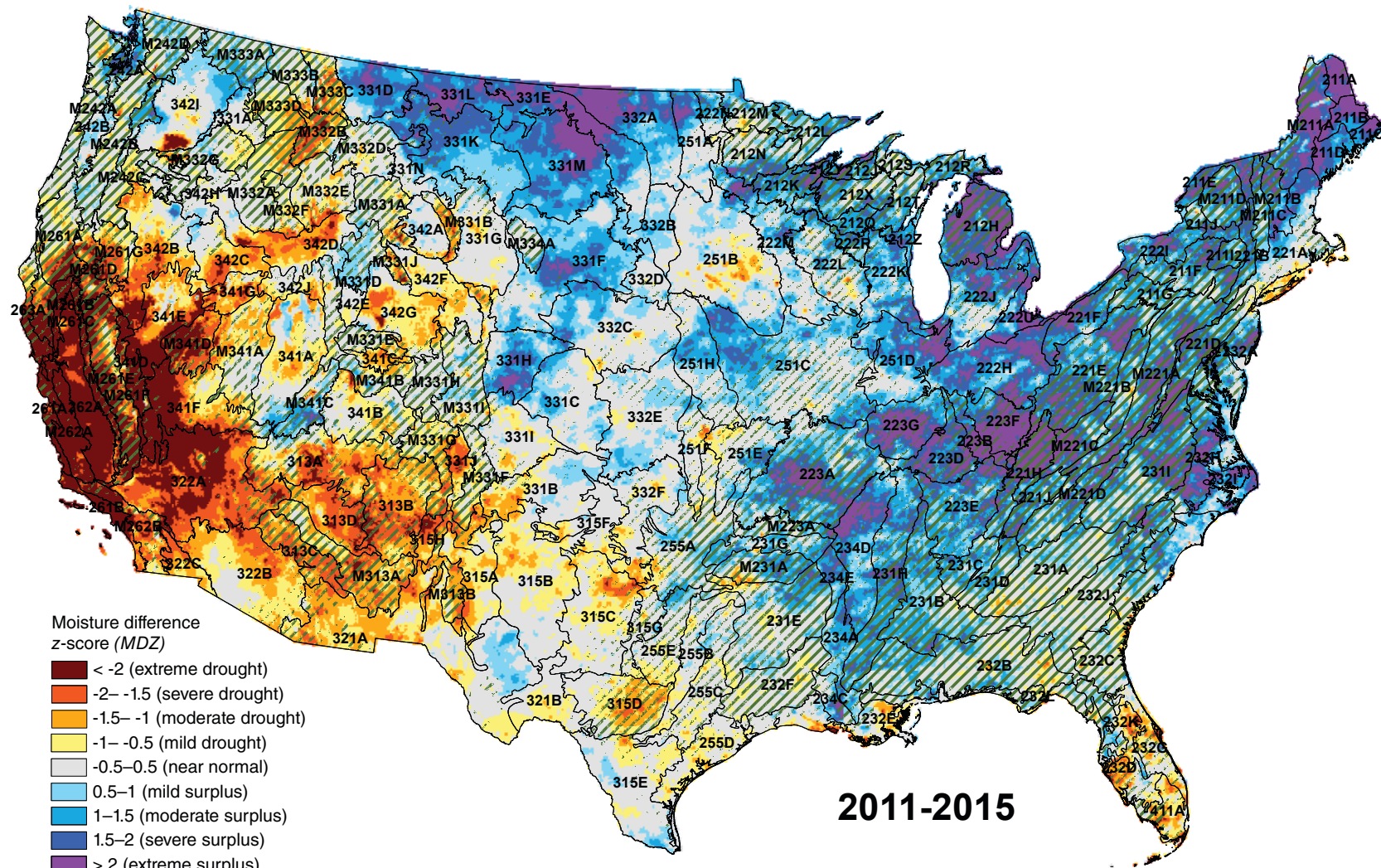


Figure 4.5—The 2011–2015 (i.e., 5-year) moisture difference z-score (MDZ) for the conterminous United States. Ecoregion section (Cleland and others 2007) boundaries are included for reference. Forest cover data (overlaid green hatching) derived from MODIS imagery by the U.S. Department of Agriculture, Forest Service, Remote Sensing Applications Center. (Data source: PRISM Climate Group, Oregon State University)



411A–Everglades). As noted earlier, the Long Island area was one of few hot spots of drought (see fig. 4.2) in the Eastern United States during 2015 (National Climatic Data Center 2016b). While moisture conditions were close to normal for most of this area during 2014 (fig. 4.3), those near-normal conditions came after the area experienced moderate to severe drought conditions in 2013 (Koch and Coulston 2015, 2016). Southern Florida was dry in both 2014 and 2015 (figs. 4.2–4.3), but close to normal in 2013 (Koch and Coulston 2015, 2016; National Climatic Data Center 2016a). This is consistent with the region’s long-term climatic regime (see fig. 4.1), in which seasonal droughts are a fairly common occurrence (Duever and others 1994).

These two areas (i.e., Long Island, southern Florida) are also evident in the 5-year *MDZ* map (fig. 4.5), suggesting longer-term persistence of drought conditions in both locations. Elsewhere in the Eastern and Central United States, including much of the Southwest, moisture conditions appear better—or at least not worse—in the 3-year *MDZ* map (fig. 4.4) than in the 5-year *MDZ* map. West of the Rocky Mountains, and in California in particular, moisture conditions have shown no improvement or have worsened in recent years. Indeed, after multiple consecutive years of extreme drought in California, the Forest Service, U.S. Department of Agriculture, conducted aerial surveys of

drought-induced tree mortality across the State. The agency found that 66 million trees died in California between 2010 and 2016, prompting Governor Jerry Brown to declare a state of emergency and initiate a Tree Mortality task force (USDA Office of Communications 2016).

As with drought, the most relevant moisture surpluses with respect to forest health are probably those that persist for several years. Areas exhibiting severe to extreme surplus conditions in both the 3-year and 5-year *MDZ* maps (figs. 4.4–4.5) may merit further attention, since the former would indicate limited recent movement toward more normal moisture conditions. Forested areas exhibiting severe to extreme surpluses in both maps were distributed widely across the Eastern United States: in North Carolina and the Mid-Atlantic States (sections 221D–Northern Appalachian Piedmont, M221A–Northern Ridge and Valley, 231I–Central Appalachian Piedmont, 232A–Northern Atlantic Coastal Plain, 232H–Middle Atlantic Coastal Plains and Flatwoods, and 232I–Northern Atlantic Coastal Flatwoods); in Kentucky (sections 221H–Northern Cumberland Plateau, M221C–Northern Cumberland Mountains, 223B–Interior Low Plateau-Transition Hills, 223D–Interior Low Plateau-Shawnee Hills, and 223F–Interior Low Plateau-Bluegrass); in Arkansas and Louisiana (section 223A–Ozark Highlands and the northern portion of 234D–



White and Black River Alluvial Plains); in Maine (primarily sections 211A–Aroostook Hills and Lowlands, 211B–Maine-New Brunswick Foothills and Lowlands, and 211C–Fundy Coastal and Interior); and in Michigan (section 212H–Northern Lower Peninsula).

### Future Efforts

If the appropriate spatial data (i.e., high-resolution maps of precipitation and temperature) remain available for public use, we will continue to produce our 1-year, 3-year, and 5-year *MDZ* maps of the conterminous United States as a regular yearly component of national-scale forest health reporting. However, users should interpret and compare the *MDZ* maps presented here cautiously. Foremost, the *MDZ* approach does not incorporate certain factors that may influence a location’s moisture supply at a finer spatial scale, such as winter snowpack, surface runoff, or groundwater storage. Furthermore, although the maps use a standardized index scale that applies regardless of the size of the time window, the window size may still deserve some consideration. For instance, an extreme drought that persists over a 5-year period has substantially different forest health implications than an extreme drought over a 1-year period. While the 1-year, 3-year, and 5-year *MDZ* maps may together provide a comprehensive short-term overview, it may

also be important to consider a particular region’s longer-term moisture history when assessing the current health of its forests. For example, in geographic regions where droughts have historically occurred on a frequent (e.g., annual or nearly annual) basis, certain tree species may be better adapted to a regular lack of available moisture (McDowell and others 2008). Because of this variability in species’ drought tolerance, a long period of persistent and severe drought conditions could ultimately lead to changes in regional forest composition (Mueller and others 2005); compositional changes may similarly arise from a long period of persistent moisture surplus (McEwan and others 2011). In turn, such changes are likely to affect regional responses to future drought or surplus conditions, fire regimes, and the status of ecosystem services such as nutrient cycling and wildlife habitat (W.R.L. Anderegg and others 2013, DeSantis and others 2011). In future work, we hope to provide forest managers and other decisionmakers with better quantitative evidence regarding critical relationships between moisture extremes and significant forest health impacts such as regional-scale tree mortality (e.g., Mitchell and others 2014). We also intend to examine the capacity of moisture extremes to serve as inciting factors for other forest threats such as wildfire or pest outbreaks.

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## INTRODUCTION

Tree mortality is a natural process in all forest ecosystems. High mortality can be an indicator of forest health problems. On a regional scale, high mortality levels may indicate widespread insect or disease impacts. High mortality may also occur if a large proportion of the forest in a particular region is made up of older, senescent stands. The approach presented here seeks to detect mortality patterns that might reflect changes to ecosystem processes due to factors such as air pollution, global climate change, or fire-regime change. However, in many cases, the proximate cause of mortality may be discernable. Understanding proximate causes of mortality *may* provide insight into whether the mortality is within the range of natural variation or reflects more fundamental changes to ecological processes.

## DATA

Forest Inventory and Analysis (FIA) phase 2 (P2) data were the basis of the mortality analysis. FIA P2 data are collected across forested land throughout the United States, with approximately 1 plot per 6,000 acres of forest, using a rotating panel sample design (Bechtold and Patterson 2005). Field plots are divided into spatially balanced panels, with one panel being measured each year. A single cycle of measurements consists of measuring all panels. This “annualized” method of inventory was adopted, State by State, beginning in 1999.

The cycle length (i.e., number of years required to measure all plot panels) ranges from 5 to 10 years.

An analysis of mortality requires data collected at a minimum of two points in time. Therefore, mortality analysis was possible for areas where data from repeated plot measurements using consistent sampling protocols were available (i.e., where one cycle of measurements had been completed and at least one panel of the next cycle had been measured, and where there had been no changes to the protocols affecting tree measurements of trees or saplings). For this report, the repeated P2 data were available for all of the Central and Eastern States. The States included in this analysis, as well as the forest cover within those States, are shown in figure 5.1.

## METHODS

FIA calculates the growth, mortality, and removal volume on each plot over the interval between repeated measurements. These values are stored in the FIA database (version 6.1) (USDA Forest Service, Forest Inventory and Analysis Program 2014). FIA's EVALIDator (ver. 1.6.0.03) is an online tool for querying the FIA database and generating area-based reports on forest characteristics (Miles 2015). EVALIDator was used to obtain net growth rates and mortality rates over the most recent measurement cycle for each of 100 ecoregion sections (Cleland and others 2007, McNab and

# CHAPTER 5.

## Tree Mortality

MARK J. AMBROSE



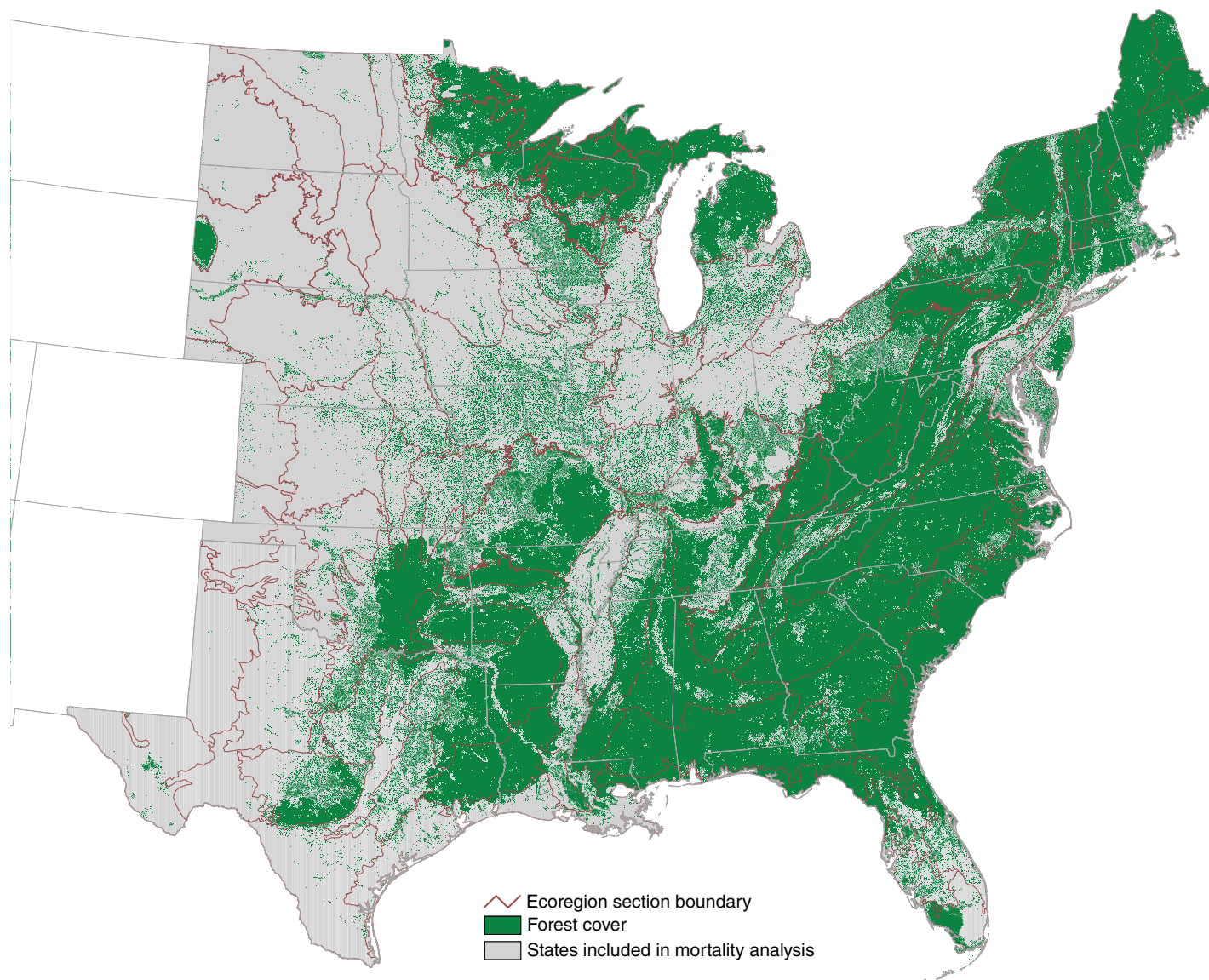


Figure 5.1—Forest cover in the States where mortality was analyzed by ecoregion section (Cleland and others 2007). Forest cover was derived from MODIS satellite imagery (USDA Forest Service 2008).



others 2007) covering the Eastern and Central United States. For most States, the most recent cycle of available data ran through 2015 (e.g., data were collected 2011 through 2015).

To compare mortality across forest types and climate zones, the ratio of annual mortality to gross growth (MRATIO) was used as a standardized mortality indicator (Coulston and others 2005). The MRATIO was calculated from the EVALIDator output for each ecoregion section, using the formula:

$$\text{MRATIO} = m / (m + r + g_n)$$

where

$m$  = annual mortality (cubic feet/year)

$r$  = annual removals, e.g. harvests, thinnings, (cubic feet/year)

$g_n$  = net annual growth (cubic feet/year).

In addition, mortality rates were derived for each forest type group (USDA Forest Service 2008) for each ecoregion section. Identifying the forest types experiencing high mortality in an ecoregion is a first step in identifying what forest health issue may be affecting the forests. Although determining particular causal agents associated with all observed mortality is beyond the scope of this report, often there are well-known insects and pathogens that are “likely suspects” once the affected forest types are identified.

To identify possible causal agents for the observed mortality, EVALIDator was also used to report disturbances that were recorded on plots where mortality occurred. Care must be used in interpreting these disturbances because disturbance is a location level variable (e.g., recorded for each stand included on a plot) rather than a tree level variable, so a given disturbance may not be directly related to the mortality of a particular tree. Nevertheless, such disturbances may indicate stressors that played a role in the observed mortality. Further information about the cause of mortality is provided by the aerial survey of insects and disease (see chapter 2 in this report). It is difficult to directly match aerial survey data to mortality observed on FIA plots. However, I incorporate the results of this survey into the discussion by consulting State Forest Health Highlights, which reflect in large part the results of aerial surveys.

## RESULTS AND DISCUSSION

The MRATIO values are shown in figure 5.2. The MRATIO can be large if an over-mature forest is senescing and losing a cohort of older trees. If forests are not naturally senescing, a high MRATIO ( $> 0.6$ ) may indicate high mortality due to some acute cause (insects or pathogens) or due to generally deteriorating forest health conditions. The 10 ecoregion sections with the highest MRATIOS are labeled

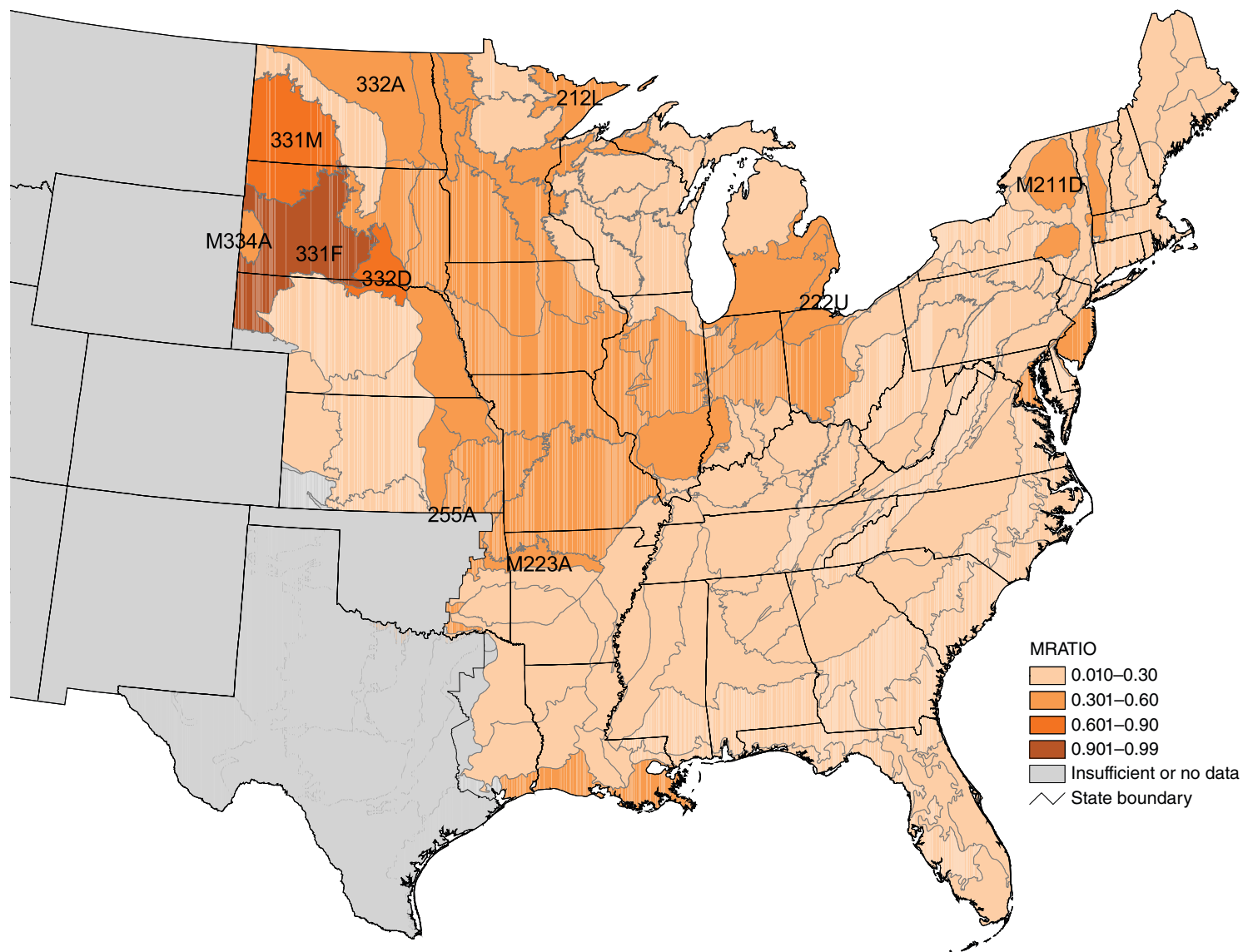


Figure 5.2—Tree mortality expressed as the ratio of annual mortality volume to gross annual volume growth (MRATIO) by ecoregion section. Ecoregions with high MRATIOS are identified by section number (Cleland and others 2007). (Data source: USDA, Forest Service Forest Inventory and Analysis Program)

on the map. In the discussion that follows, I focus on the ecoregions having MRATIOS  $> 0.5$  (i.e., where mortality was greater than half of gross growth).

The highest MRATIOS occurred in ecoregion sections 331F–Western Great Plains (MRATIO = 0.93) and 332D–North-Central Great Plains (MRATIO = 0.87), both in South Dakota and Nebraska. Another area of high mortality relative to growth on the Great Plains was ecoregion 331M–Missouri Plateau (MRATIO = 0.72) in North and South Dakota. In these Great Plains ecoregions where mortality is high relative to growth, the predominant vegetation is grassland. In most of them, though the ecoregions were quite large, there was relatively little forest land to measure (114 plots in region 331F, 86 plots in region 331M, and 60 plots in region 332D). In the Plains, tree growth is generally slow in these ecoregion sections because of naturally dry conditions. Where the number of sample plots is small and tree growth is slow, care must be taken in interpreting mortality relative to growth.

Both ecoregions 331F and 332D have had high mortality relative to growth in recent years (Ambrose 2013, 2014, 2015a, 2015b, 2016), so the observed mortality is **not a new phenomenon**. Tree growth rates in these regions (especially in 331F) are quite low, so the high MRATIOS are due to a combination of low growth and high mortality. Much of the forest in these sections is riparian, and most of the species experiencing the greatest mortality are commonly found in riparian areas. The major

exception was high ponderosa pine mortality in ecoregion section 331F–Western Great Plains. Ponderosa pine is not a riparian tree species, but like the riparian species, it only occurs in a relatively small area of the ecoregion, on discontinuous mountains, plateaus, canyons, and breaks in the plains (Burns and Honkala 1990). In both of these ecoregions, fire and damage from domestic animals were associated with large proportions of the mortality (table 5.1).

In ecoregion section 332D–North-Central Great Plains, over half of the mortality (by volume) occurred in the elm-ash-cottonwood forest-type group. This type group includes most of the riparian forests found in the region. A number of different factors may be responsible for the high mortality in the ecoregion. Domestic animal damage was associated with 74 percent of the mortality in this ecoregion. However, it is unclear how much of this damage was severe enough to induce mortality. The drought in 2012 and 2013, as well as associated winter desiccation, has been reported as severely stressing trees in much of South Dakota and Nebraska. Dutch elm disease has been responsible for American elm mortality in both States (Ball and others 2015; Nebraska Forest Service 2012, 2013, 2015; South Dakota Department of Agriculture 2012). Cedar bark beetle (*Phloeosinus* spp.) combined with drought stress has been reported as causing mortality in juniper (redcedar) in South Dakota (South Dakota Department of Agriculture 2012, 2013). Green ash have been affected by ash/lilac borer

**Table 5.1—Ecoregion sections having the highest mortality relative to growth (MRATIO); annual growth, mortality, and removal rates; and disturbances associated with areas of mortality**

| Ecoregion section                           | Average annual mortality | Average annual removals | Average annual net growth | MRATIO | Major disturbances associated with areas with mortality                 |
|---------------------------------------------|--------------------------|-------------------------|---------------------------|--------|-------------------------------------------------------------------------|
| ----- cubic feet/year -----                 |                          |                         |                           |        |                                                                         |
| 331F—Western Great Plains                   | 14,308,113               | 3,731,435               | -2,694,641                | 0.93   | Fire (56%), domestic animals (34%)                                      |
| 332D—North-Central Great Plains             | 12,126,364               | 304,034                 | 1,527,629                 | 0.87   | Domestic animals (74%), fire (18%)                                      |
| 331M—Missouri Plateau                       | 6,694,677                | 123,283                 | 2,480,803                 | 0.72   | Flooding (58%)                                                          |
| 255A—Cross Timbers and Prairie              | 16,062,132               | 2,957,525               | 8,384,393                 | 0.59   | Fire (16%), beaver (13%), domestic animals (12%)                        |
| 332A—Northeastern Glaciated Plains          | 7,131,800                | 566,035                 | 4,596,716                 | 0.58   | Domestic animals (21%)                                                  |
| M334A—Black Hills                           | 37,002,272               | 34,125,600              | 649,279                   | 0.52   | Insects (52%), fire (18%)                                               |
| 222U—Lake Whittlesey Glaciolacustrine Plain | 47,565,620               | 22,143,394              | 23,341,691                | 0.51   | Insects (61%)                                                           |
| 212L—Northern Superior Uplands              | 121,066,267              | 49,363,356              | 75,932,039                | 0.49   | Fire (15%)                                                              |
| M223A—Boston Mountains                      | 80,938,965               | 38,834,443              | 50,970,788                | 0.47   | Ice (30%), fire (7%)                                                    |
| M211D—Adirondack Highlands                  | 175,362,862              | 78,066,326              | 137,881,360               | 0.45   | Human caused (4%), insects (3%), domestic animals (3%), vegetation (3%) |

Percentages are the percent of mortality volume occurring on forested conditions that were affected by the given disturbance type.

(*Podosesia syringae*) in South Dakota (Ball and others 2015, South Dakota Department of Agriculture 2012).

In ecoregion 331M—Missouri Plateau, over 90 percent of the mortality (by volume) occurred in the elm-ash-cottonwood forest-type group. Prior analyses identified three species, eastern cottonwood, bur oak, and green ash, as suffering high mortality in this region (Ambrose 2015b, 2016). Green ash have been affected by ash/lilac borer, as well as other native ash borers, in both North and South Dakota (Ball and others 2015, North Dakota Forest Service 2012, South Dakota

Department of Agriculture 2012). Cottonwood canker fungi have been identified as a problem throughout North Dakota (North Dakota Forest Service 2014); these fungi may be contributing to the observed cottonwood mortality. Adverse weather conditions, including both drought and excessively wet conditions, both of which occurred during the remeasurement cycle (North Dakota Forest Service 2012, 2013; South Dakota Department of Agriculture 2012), may have contributed to mortality by stressing trees. Flooding was associated with 58 percent of the observed mortality (table 5.1).

The majority of the mortality in ecoregion 332A–Northeastern Glaciated Plains of North Dakota (MRATIO = 0.58) was split about evenly between the elm-ash-cottonwood and aspen-birch forest-type groups (about 40 percent in each). About 21 percent of the mortality was associated with domestic animal damage (table 5.1). This ecoregion includes the Turtle Mountains, where thousands of acres of forest tent caterpillar and large aspen tortrix defoliation have occurred in recent years. Also, 4,000 acres of aspen decline related to over-mature stands have been identified in this region (North Dakota Forest Service 2014). The defoliation together with the aspen decline may be the cause of most of the mortality in the aspen-birch forest type. Cottonwood canker fungi are a problem throughout North Dakota (North Dakota Forest Service 2014) and may be a cause of the mortality in the elm-ash-cottonwood forest type.

In ecoregion section M334A–Black Hills (MRATIO = 0.52), almost all of the mortality occurred in the ponderosa pine forest type group; however, previous analyses showed that this represented a relatively small proportion of the ponderosa pine in the ecoregion (about 2 percent of ponderosa pine forest stems) (Ambrose 2015b). In the adjacent ecoregion section 331F–Western Great Plains, where the MRATIO was highest, the vast majority (about 90 percent) of the mortality also occurred in the ponderosa pine forest-type group. Here, too, this mortality represented a relatively small proportion of the growing stock in the

ponderosa pine forest type (1.4 percent) in the region. About 52 percent of the mortality in the Black Hills was associated with insects (table 5.1). The pine mortality in both ecoregions is very likely related to mountain pine beetle (*Dendroctonus ponderosae*). There has been an ongoing pine beetle outbreak in the Black Hills (South Dakota Department of Agriculture 2011, 2012, 2013, 2014), and mountain pine beetle killed about 21 million trees in the Black Hills from 2000 to 2015 (Ball and others 2015). Mountain pine beetle-related mortality also has been reported in western Nebraska (Nebraska Forest Service 2011, 2012), with an outbreak that began in 2009, though pine beetle-related mortality there has fallen significantly recently (Nebraska Forest Service 2014, 2015). More recently, several other agents have been reported as affecting ponderosa pine in western Nebraska, including *Ips* beetles and *Diplodia* blight (Nebraska Forest Service 2015). Drought in 2012 and 2013, affecting much of South Dakota and Nebraska (Nebraska Forest Service 2012, 2013; South Dakota Department of Agriculture 2012), may also have contributed to pine mortality, as well as that of other species, in these ecoregions.

Mortality was split almost evenly between the oak-hickory and elm-ash-cottonwood forest-type groups in ecoregion 222U–Lake Whittlesey Glaciolacustrine Plain (MRATIO = 0.51). About 61 percent of the mortality in this ecoregion was associated with insects (table 5.1). The mortality in the elm-ash-cottonwood group is likely due to emerald ash borer (*Agrilus planipennis*), which has produced extremely high ash mortality

throughout Ohio and Michigan (Michigan Department of Natural Resources 2014, 2015; Ohio Department of Natural Resources, Forestry Division 2014, 2015). The cause of mortality in the oak-hickory forest-type group is less clear. Several oak pests were reported in Ohio as well as “leaf-curl syndrome” of unknown origin (Ohio Department of Natural Resources, Division of Forestry 2015), while in Michigan oak wilt (caused by the pathogen *Ceratocystis fagacearum*) has been confirmed in at least part of the ecoregion (Michigan Department of Natural Resources 2015).

Ecoregion 255A–Cross Timbers and Prairie experienced relatively high mortality (MRATIO = 0.59). However, the majority of the ecoregion is located in western Oklahoma, where mortality data are not yet available. Therefore, the results shown are based on data collected in the relatively small portion of the ecoregion located in eastern Oklahoma and southeastern Kansas. About 60 percent of the mortality (in terms of tree volume) occurred in the oak-hickory forest-type group; another 26 percent of the mortality occurred in the elm-ash-cottonwood forest-type group. Disturbances associated with mortality included fire, beavers, and domestic animal damage. A record-setting drought in 2011 affected Oklahoma and Texas. It was reported as weakening both pines and hardwoods in Texas, making them susceptible to a variety of pests and pathogens (Smith 2013, 2014). Oklahoma has been working with Texas to monitor the impacts of drought on forest health in both States (Oklahoma Forestry Services 2014).

## SUMMARY

This analysis shows that in most of the Eastern and Central United States, mortality is low relative to tree growth. Mortality is rather low in most of the areas for which data are available. The areas of highest mortality occur in the mostly riparian forests of Great Plains ecoregions. A common characteristic of most of these ecoregions with high mortality is that they are on the margins of land suitable for forest growth, being very dry. Thus, they tend to be extremely vulnerable to changes in weather patterns that might produce prolonged and/or extreme drought. Drought, combined with a variety of other biotic and/or abiotic stressors are likely responsible for much of the mortality observed.

It is important to realize that this analysis alone cannot tell the complete story regarding tree mortality. Mortality that is concentrated in highly fragmented forest or nonforest areas adjacent to human development may not be detected because the available FIA data processed by EvalIDator do not cover urban areas or other places not defined as forest by FIA. Also, should a particular species be dying due to a pest or pathogen in mixed-species forests where other species are growing vigorously, this analysis is unlikely to detect it. This is especially true of species (e.g., ash) that make up a relatively small proportion of many eastern forests. To gain a more complete understanding of mortality, one should consider the results of this analysis together with other indicators of forest health, including insect



and disease activity (chapter 2) and Evaluation Monitoring projects that focus on specific mortality-causing agents (chapters 9–14).

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## SECTION 2.

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



## INTRODUCTION

The Forest Health Monitoring (FHM) indicators were designed to provide data to describe current conditions of forest ecosystems and detect changes and trends over time (Woodall and others 2011). Collectively, the FHM indicators provide a nationwide systematic sample of a wide array of measurements on forested ecosystems, which are used by a diverse set of customers for a variety of purposes. The individual indicators developed and matured at various rates during the 10-plus years of their implementation by the U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis (FIA) Program. The Vegetation Diversity and Structure Indicator (VEG) entails a complete survey of vascular plant species on each forested subplot, including species identification and percentage of total canopy cover as a measure of abundance. VEG was the last to be implemented and least widely collected of the FHM indicators. However, despite discontinuous data collection over 10 years, there are more than 500 plots that were visited twice, providing an opportunity to explore what has changed over the time between plot visits.

VEG data have been used for assessing species diversity, the distribution and abundance of individual species or groups of species (e.g., introduced species, species by growth habits), and species composition by forest types across States and ecological regions (Morin and others 2011, Moser and others 2013, Schulz 2011a, Schulz and Gray 2013). Species diversity can be measured in many ways. Species richness,

the number of species observed in a defined area, is the most basic measure of diversity. Species diversity also can be assessed in terms of compositional differences or dominance of individual or groups of species. Because VEG data were collected on standard-sized sample units, it is possible to use them to detect trends over large regions and across many different forest ecosystem types.

Species distribution can be assessed as frequency: the proportion of sample units where a species is present (McCune and Grace 2002; Schulz 2011a, 2011b; Schulz and others 2013). When plots are visited more than once, frequency is a robust measure to detect change in distribution of individual or groups of species. Nonnative plant species represent a group of species of high interest to land managers and are prevalent throughout the Northeastern United States (Schulz and Gray 2013). Many surveys of introduced or invasive species do not include information about the native species on the site. VEG data provide the ability to look at occupancy, reported as a proportion of introduced species number and abundance (measured as cover) compared to all species present, as an indication of the impact of nonnative species.

The data collected as part of FIA's standard forest inventory provide a considerable amount of information about the physical setting of each plot, including observations of chronic or discrete disturbances (O'Connell and others 2015). These variables can serve as predictors for changes observed in forest understory

## CHAPTER 6.

### Changes on Vegetation Indicator Plots: Small Sample Reveals Significant Trends

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species composition. For example, consider plots harvested or experiencing an ice storm between inventory visits. One might expect tree cover to be reduced and some non-tree species to become more abundant while others became less abundant, depending on sensitivity responses to increased light. When analyzing change in species composition over time, understanding the types of disturbances occurring between plot visits is important in the interpretation of results.

In this report, I first describe the changes in physical setting or forest stand conditions recorded between visits on plots that could influence vegetation composition or growth. I then highlight examples of change analysis using the vegetation indicator data, specifically: 1) the differences in the total number of species observed and average subplot species richness between visit periods; 2) changes to common individual species plot frequency between visits, comparing different subsets of plots; and 3) changes over time in introduced species frequency and relative abundance. My objectives are to show the dynamic nature of the forests examined, to demonstrate appropriate types of change analyses using forest health vegetation indicator data, and to highlight features of the database that are easily overlooked but essential to understand when compiling data prior to analyses.

## METHODS

### Area and Data Collection

The majority of VEG plots visited twice are located across the North Central and Northeastern United States (fig. 6.1), with additional plots in coastal Alaska. FIA maintains a network of permanent plots across the Nation, and the forest health indicators were collected on a 1/16<sup>th</sup> subset of these plots. Plots are visited on a 5- or 10-year rotation (depending on the region of the country) to collect standard forest inventory tree variables, along with physical descriptors of the plot site (O'Connell and others 2015, USDA Forest Service 2007a). Each plot location is also assigned to an ecological subsection in accordance to hierarchical ecological regions described by Bailey (1995) and refined by Cleland and others (2007).

On plots where VEG data were collected, all vascular plants rooted in or hanging over the four 7.32-m radius subplots were identified. Species were recorded using the Natural Resources Conservation Service PLANTS database taxonomy (USDA NRCS 2010). VEG crews were allowed to record plants to subspecies when they were confident of the identification. The PLANTS database also designates species origin as native or introduced. For each species on the subplot, total percent canopy cover was estimated and recorded. Unknown species were collected near the plot and identified later by an FIA vegetation specialist or submitted to a qualified herbarium



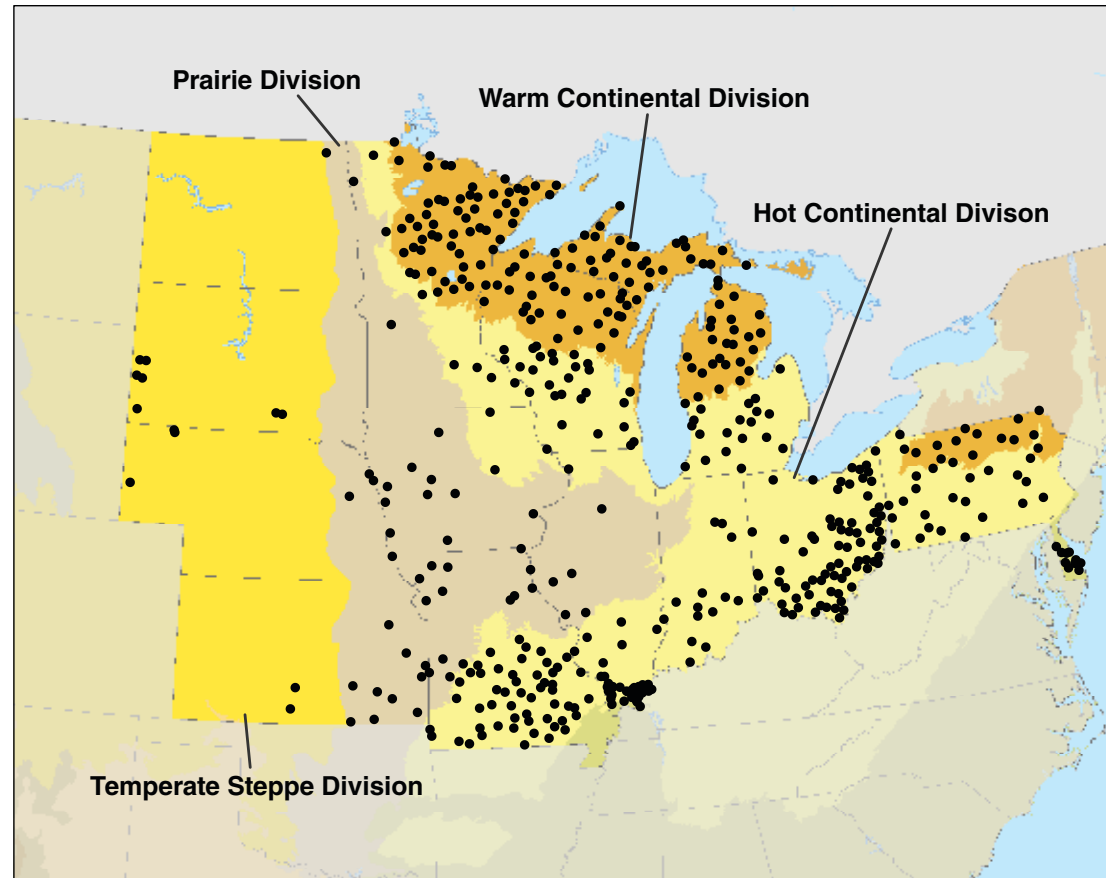


Figure 6.1—Approximate locations where the suite of Vegetation Indicator measurements were collected twice in the conterminous United States.

(USDA Forest Service 2007b). FIA VEG field guide protocols recommended that when plots are remeasured at the 5- or 10-year rotation, they should be visited within a 2-week calendar window of the previous visit so that seasonally-limited, short-lived species may be observed during both visits (USDA Forest Service 2007b).

### Dataset

VEG data were collected between 2001 and 2011 and downloaded from the Forest Inventory and Analysis Database (FIADB) Data Mart ([http://apps.fs.fed.us/fiadb-downloads/CSV/datamart\\_csv.html](http://apps.fs.fed.us/fiadb-downloads/CSV/datamart_csv.html)) in 2012. Species records in

this dataset were then subjected to post-field quality control, in which tree species recorded by the VEG crew were compared with tree data collected as part of FIA's standard forest inventory. Sometimes large trees were missed by the VEG crew, or species identifications conflicted. After thorough examination of plot records, some subplot species records were edited or augmented to include tree species missed by the VEG crew. The augmented dataset differs from the original FIADB dataset in that some subplots have one or two more species records (in most cases), with a modeled or estimated percentage of total canopy cover. The adjusted or augmented records result in an increase in the sum of individual species total cover. The entire VEG dataset was subjected to this review, so both first and second visits were scrutinized. These data are available upon request from the author.

I focused on the 514 plots where visits were at least 5 years apart. In some cases, plots were revisited on a shorter rotation, while in other cases the revisit cycle was extended due to regional logistic or access challenges. The vast majority of plots were revisited after 5 years, some after 6 years, and four plots were visited after 10 years.

### Analysis

**Plot conditions**—To assess changes in plot conditions, I examined the standard inventory records at both the initial and second visit, noting changes in condition and forest types,

distance from roads, and water on plot code. In addition, records of observed treatments and disturbances that occurred between visits were examined. I then noted the calendar date of each plot visit and determined if revisits occurred within the recommended 2-week calendar window. For this analysis, plot ecological subsections were aggregated to the higher division level within the ecoregion hierarchy, a level representing broad climatic zones.

At each visit, each plot was also assigned to one of three fragmentation classes based on the number and types of condition classes assigned. If the plot was 100 percent forest and was determined to be a single condition, it was classified as an “intact” stand. Plots that were 100 percent forested but had more than one condition were classified as “multiple condition.” Plots that were less than 100 percent forest were classified as “forest edge.” This method served as a coarse filter for fragmentation, although some “intact” forest plots could be located close to forest edges or within small parcels of remnant forests, and some “edge” plots might be adjacent to naturally occurring nonforest plant communities.

**Changes in species richness**—To first explore the total number of species observed on the 514 plots during the initial and second visit periods, I compiled a list of unique plants species identified for each visit period by ecological division. Special attention was taken to correct for unknowns that were resolved to a species

already recorded on each sample unit, while maintaining individual records for unresolved unknown codes. I also summed the number of species that were recorded on one plot only, and the number of unresolved (genus or unknown) plant codes used for each visit period. I used all plots for these summaries. This provides some insight into the nature of the dataset, and possibly the experience and botanical skills of the crews collecting data.

To compare changes in subplot species richness, I used only those plots that were intact (four fully forested subplots) at both visits (n=267). I totaled the number of unique species for each visit at both the plot and subplot levels, and determined the average number of subplot species per plot. I used the paired Student's t-test (2-tailed) to determine if the mean species richness at visit one was equal to mean species richness at the time of the second visit.

**Changes in species distribution**—To examine changes in the species distribution of 47 of the more commonly recorded individual species in the best-represented ecological divisions (Hot Continental and Warm Continental), the McNemar's test (Conover 1980, Elzinga and others 1998, Schulz and others 2009) was used. The analysis was conducted on three subsets of plots:

1) Plots with no recorded changes in water codes, disturbances, or treatments between visits, that were not within 100 feet of a road at the second visit, and that were revisited within

the recommended 2-week calendar window (n=71), representing the least disturbed subset of plots, with forb species included in the analysis;

2) Plots with no changes in percentage of accessible forest, with some disturbances and changes, measured within the 2-week window (n=97), representing some disturbances, with forb species included; and

3) Plots with no changes in percentage of forest, with some disturbances, with no regard to revisit calendar date (n=286), with only woody and two invasive grass species easily recognized year-round included in the analysis.

Plants recorded at the subspecies level were consolidated at the species level. Plot species records were examined to determine if a species (or one of its subspecies) was present or absent at either visit. The McNemar's test statistic ( $T$ ) can be used for assessing differences in proportions (such as frequency) when permanent plots are remeasured. This test is powerful because the observations are highly correlated.

$$T = \frac{(b - c)^2}{(b + c)}$$

Where:

$b$  = the number of subplots where a species was found in time 1 but not time 2

$c$  = the number of subplots where the species was not found in time 1 but found in time 2

$n = b + c$

The test statistic  $T$  is then compared to the chi-square distribution with 1 degree of freedom. The value ( $n$ ) is used to assess the chi-squared distribution. The null hypothesis of no change is rejected if the test statistic exceeds the chi-square table value for 1 degree of freedom. Results were summarized by ecological division for the three datasets analyzed.

**Changes in the constancy and occupancy in introduced species**—Constancy, the proportion of plots with at least one introduced species, was computed for each visit period by fragmentation class. Measures of occupancy—relative richness and relative cover of introduced species—were computed at the plot level for each plot visit. The number of plants identified to species and the number of introduced species were determined for each plot. Total cover for all species and for introduced species were summed across the plot as well. The relative richness of introduced species is simply the number of introduced species divided by the total number of all species identified to species per plot. The relative cover of introduced species is the sum of all introduced species divided by the sum of cover of all species for each plot.

In contrast to how occupancy was compiled by Schulz and Gray (2013), both plots with and without introduced species were used to compile the relative richness and relative cover of introduced species. This approach is used to highlight change over the entire population of plots, as opposed to elucidating the distribution of introduced species, which is more meaningful for a single time series (Schulz and others 2009).

The paired Student's  $t$ -test (2-tailed) was used to determine if the occupancy values at time 1 and time 2 are from populations with equal means.

These analyses were limited to plots where the level of fragmentation or percentage of accessible forest had not changed ( $n=382$ ) and to those plants identified to species. Only subplots that were fully forested were included in the plot-level summaries. Some plants observed on VEG plots were never identified to species due to their stage of development at the time of plot visits. It is assumed that the proportion of introduced species among the unidentified plants is similar to their proportion of all plants identified to species.

## RESULTS AND DISCUSSION

### Plot Conditions

A summary is shown in table 6.1 of the number of plot pairs and forest types represented by State, and of changes that occurred between first and second visit in level of fragmentation, forest type, disturbances or treatments, water on plot, and distance from roads. Note that several changes in conditions may occur on a single plot. The number of revisits within the recommended time window is also included. Only one-third of the plot revisits were within a 2-week calendar window. This summary reveals the diversity of forested conditions this dataset represents. The spatial extent of plot locations (fig. 6.1) is great enough to capture a range of climatic and edaphic conditions, as represented by the ecological divisions.

**Table 6.1—Plot pair numbers by State, types of changes recorded, and numbers of revisits within the recommended calendar window for 514 revisited VEG plots**

| State                     | Total number of plots | Number of forest types | No change | Type of change      |             |                            |                          |               |                    |                          | Visits within 14 calendar days |
|---------------------------|-----------------------|------------------------|-----------|---------------------|-------------|----------------------------|--------------------------|---------------|--------------------|--------------------------|--------------------------------|
|                           |                       |                        |           | Fragmentation level | Forest type | Disturbance between visits | Treatment between visits | Water on plot | Distance from road | Road now within 100 feet |                                |
|                           |                       |                        |           |                     |             |                            |                          |               |                    |                          |                                |
| -----number of plots----- |                       |                        |           |                     |             |                            |                          |               |                    |                          |                                |
| Alaska                    | 21                    | 8                      | 2         | 3                   | 0           | 6                          | 0                        | 11            | 11                 | 1                        | 7                              |
| Delaware                  | 10                    | 4                      | 4         | 2                   | 0           | 0                          | 0                        | 2             | 5                  | 0                        | 0                              |
| Illinois                  | 48                    | 11                     | 12        | 3                   | 4           | 13                         | 0                        | 9             | 19                 | 3                        | 1                              |
| Indiana                   | 13                    | 5                      | 0         | 0                   | 2           | 5                          | 1                        | 6             | 9                  | 0                        | 3                              |
| Iowa                      | 13                    | 5                      | 5         | 1                   | 2           | 3                          | 0                        | 6             | 5                  | 0                        | 1                              |
| Kansas                    | 12                    | 5                      | 1         | 2                   | 0           | 5                          | 0                        | 6             | 3                  | 0                        | 6                              |
| Michigan                  | 78                    | 19                     | 12        | 8                   | 7           | 5                          | 4                        | 29            | 32                 | 2                        | 28                             |
| Minnesota                 | 69                    | 21                     | 13        | 8                   | 4           | 2                          | 3                        | 20            | 42                 | 0                        | 29                             |
| Missouri                  | 67                    | 11                     | 13        | 4                   | 3           | 20                         | 4                        | 22            | 23                 | 0                        | 28                             |
| Nebraska                  | 6                     | 1                      | 1         | 1                   | 0           | 5                          | 1                        | 1             | 1                  | 0                        | 4                              |
| North Dakota              | 1                     | 1                      | 1         | 0                   | 0           | 0                          | 0                        | 0             | 0                  | 0                        | 0                              |
| Ohio                      | 75                    | 16                     | 11        | 7                   | 14          | 20                         | 12                       | 18            | 33                 | 1                        | 24                             |
| Pennsylvania              | 38                    | 12                     | 13        | 3                   | 1           | 3                          | 2                        | 2             | 15                 | 0                        | 11                             |
| South Dakota              | 7                     | 2                      | 2         | 2                   | 0           | 1                          | 1                        | 0             | 3                  | 0                        | 2                              |
| Wisconsin                 | 56                    | 21                     | 17        | 7                   | 9           | 3                          | 8                        | 6             | 21                 | 0                        | 17                             |
| TOTAL                     | 514                   | 60                     | 107       | 51                  | 46          | 91                         | 36                       | 138           | 222                | 7                        | 161                            |

Overall, few changes in level of fragmentation were explained by recorded disturbance or treatment. The same holds true when there was a change of forest type on an intact plot. Although there are a few disturbances or treatments noted between visits on plots where proportion of accessible forested land or forest type changed, most often the cause for the change is not explained by recorded variables.

The type of disturbances occurring between visits varies by region. Disturbances included insect and disease observations in Alaska, but were most often caused by weather in Illinois. In Ohio, competing vegetation was recorded as a disturbance on 11 plots out of 20, with grazing animals, wind, and human-caused disturbance also recorded. In Missouri, Indiana, Iowa, Kansas, and South Dakota, domestic livestock grazing was the most common disturbance, followed by ice and flooding damage.

The “water on plot” coding system is intended to be hierarchical or directional from large permanent water to temporary water, but it is not quantitative, making these changes a bit more difficult to assess. For example, a “permanent stream or pond too small to be included in noncensus water” may be present at one visit but not at another. Plant composition is affected by fluxes in moisture; these plots may be areas that experience changes in moisture levels seasonally/annually. Because the changes in codes are difficult to interpret, but changes in the presence of water will influence changes

in forb and grass-like species composition, plots with changes in water on plot code should be recognized and handled appropriately if assessing the influence other factors may be playing in a change analysis.

Changes in Species Richness

Overall, more species were recorded during the second visits than first, with fewer genus-only and unknown codes used during the second visit (table 6.2).

Table 6.2—Number of plot pairs with two visits by ecological division, with total number of species identified, number of species recorded only once, and number of genus and unknown codes used for each visit period

| Ecological division  | Number of plots | Total plant species identified |              | Species identified only once |              | Genus and unknown codes used |              |
|----------------------|-----------------|--------------------------------|--------------|------------------------------|--------------|------------------------------|--------------|
|                      |                 | First visit                    | Second visit | First visit                  | Second visit | First visit                  | Second visit |
|                      |                 | -----number of species-----    |              |                              |              | --number of codes--          |              |
| 130 Subarctic        | 2               | 55                             | 63           | 40                           | 51           | 8                            | 7            |
| 210 Warm Continental | 160             | 687                            | 839          | 221                          | 271          | 156                          | 108          |
| 220 Hot Continental  | 260             | 1026                           | 1240         | 331                          | 372          | 216                          | 189          |
| 230 Subtropical      | 14              | 161                            | 185          | 85                           | 113          | 23                           | 15           |
| 240 Marine           | 19              | 156                            | 195          | 44                           | 70           | 23                           | 23           |
| 250 Prairie          | 46              | 540                            | 564          | 222                          | 269          | 69                           | 38           |
| 330 Temperate Steppe | 13              | 248                            | 303          | 179                          | 198          | 30                           | 11           |
| Total for population | 514             | 1654                           | 1886         | 477                          | 567          | 286                          | 222          |

Note: the only column where “total for the population” is the sum of the numbers above is “number of plots.” Totals for the population in the “numbers of species” and “number of codes” do not equal the sum of numbers in the above columns because many species are found in many divisions, so these are the total (cumulative) number of species identified if one had ignored the divisions and looks at all plots together. The point being, botanists identified more species and used fewer unknown codes at the time of the second visit.



The differences by ecoregion division in plot and subplot species richness for those plots that were intact for both visits (n=267) are shown in table 6.3, with the exception of two plots from Subarctic ecoregion division 130 that were excluded because of insufficient sample size. There were significant increases in plot and average subplot richness at the second visit for the best represented ecological divisions, as well as the Marine division. Although these results could indicate a true increase in the number of species present, one must consider if the observed increase is an artifact of field crews gaining experience and better observational skills by the time of the second visit. During the first few years of implementation of the VEG protocol, the novelty of hiring crew members with botanical skills and the development of data collection and management systems proved challenging, and some units did better than

others at achieving data quality targets. These factors could have influenced the apparent increase in species richness.

### Individual Species Distribution

Twenty-three of the 47 species tested showed some significant change in plot frequency using the McNemar test statistic in one or both ecological divisions included (table 6.4). Most species showed an increase in frequency; only one was decreasing (*Quercus velutina* plot frequency in the Hot Continental division). The 24 species with no change in frequency are tabulated in table 6.5.

Results show common serviceberry and black cherry increasing in each dataset tested (table 6.4). Note that as the sample size increases, the probability of equal frequencies at both visits decreases. Common serviceberry

**Table 6.3—Number of intact plots with two visits by ecological division and average plot and subplot species richness for intact plots for each visit, with the Student’s t-test probability of average subplot species richness being equal at both visits**

| Ecological division  | Number of plots | Average plot species richness |              | Probability of plot richness being equal at both visits | Average subplot species richness |              | Probability of subplot richness being equal at both visits |
|----------------------|-----------------|-------------------------------|--------------|---------------------------------------------------------|----------------------------------|--------------|------------------------------------------------------------|
|                      |                 | First Visit                   | Second Visit |                                                         | First visit                      | Second visit |                                                            |
| 210 Warm Continental | 103             | 41.47                         | 53.63        | <0.0001                                                 | 24.27                            | 31.63        | <0.0001                                                    |
| 220 Hot Continental  | 122             | 57.54                         | 61.72        | <0.0001                                                 | 28.54                            | 36.84        | <0.0001                                                    |
| 230 Subtropical      | 7               | 36.00                         | 36.43        | 0.9204                                                  | 19.71                            | 22.50        | 0.2921                                                     |
| 240 Marine           | 15              | 35.67                         | 41.60        | 0.0265                                                  | 22.73                            | 26.63        | 0.0127                                                     |
| 250 Prairie          | 13              | 60.15                         | 52.62        | 0.2622                                                  | 30.98                            | 31.90        | 0.7731                                                     |
| 330 Temperate Steppe | 5               | 33.20                         | 36.40        | 0.6509                                                  | 17.55                            | 21.95        | 0.2863                                                     |

**Table 6.4—Scientific and common names of 23 species with probability of plot frequency being equal at both visits, as determined with McNemar’s test in at least one of the compared datasets, in two ecological divisions**

| Scientific name                  | Common name                   | Warm Continental Division |                            |                                  | Hot Continental Division |                            |                                  |
|----------------------------------|-------------------------------|---------------------------|----------------------------|----------------------------------|--------------------------|----------------------------|----------------------------------|
|                                  |                               | No change, 2-week window  | Some change, 2-week window | Some change, no time restriction | No change, 2-week window | Some change, 2-week window | Some change, no time restriction |
| -----level of significance-----  |                               |                           |                            |                                  |                          |                            |                                  |
| <i>Acer rubrum</i>               | Red maple                     | –                         | –                          | –                                | 0.1796                   | 0.2668                     | –                                |
| <i>Amelanchier arborea</i>       | Common serviceberry           | 0.0574                    | 0.0308                     | 0.0100                           | 0.1460                   | 0.1796                     | 0.0750                           |
| <i>Anemone quinquefolia</i>      | Wood anemone                  | 0.1250                    | 0.1094                     | fb                               | –                        | –                          | fb                               |
| <i>Athyrium filix-femina</i>     | Common lady fern              | 0.2266                    | 0.0574                     | fb                               | –                        | –                          | fb                               |
| <i>Berberis thunbergii</i> (i)   | Japanese barberry             | –                         | –                          | –                                | –                        | –                          | 0.0704                           |
| <i>Botrychium virginianum</i>    | Rattlesnake fern              | –                         | –                          | fb                               | –                        | 0.1184                     | fb                               |
| <i>Celtis occidentalis</i>       | Common hackberry              | –                         | –                          | –                                | –                        | 0.3438                     | 0.1670                           |
| <i>Fraxinus pennsylvanica</i>    | Green ash                     | –                         | –                          | 0.0030                           | –                        | –                          | –                                |
| <i>Microstegium vimineum</i> (i) | Nepalese browntop             | –                         | –                          | –                                | 0.0624                   | 0.0156                     | <0.0001                          |
| <i>Phalaris arundinacea</i> (i)  | Reed canarygrass              | 0.2188                    | 0.0390                     | 0.0072                           | –                        | –                          | –                                |
| <i>Prunus serotina</i>           | Black cherry                  | 0.2188                    | 0.0654                     | 0.0250                           | 0.0214                   | 0.0018                     | 0.0001                           |
| <i>Pteridium aquilinum</i>       | Western brackenfern           | 0.0078                    | 0.0010                     | fb                               | –                        | –                          | fb                               |
| <i>Quercus rubra</i>             | Red oak                       | 0.0040                    | 0.0020                     | 0.0050                           | –                        | –                          | 0.2000                           |
| <i>Quercus velutina</i>          | Black oak ( <i>decrease</i> ) | –                         | –                          | –                                | –                        | 0.0624                     | 0.0490                           |
| <i>Rhamnus cathartica</i> (i)    | Common buckthorn              | –                         | –                          | –                                | –                        | –                          | 0.0574                           |
| <i>Rosa multiflora</i> (i)       | Multiflora rose               | –                         | –                          | –                                | 0.289                    | 0.0386                     | <0.0001                          |
| <i>Rubus idaeus</i>              | American red raspberry        | 0.0078                    | 0.0020                     | <0.0001                          | –                        | –                          | 0.0070                           |
| <i>Sassafras albidum</i>         | Sassafras                     | –                         | –                          | –                                | –                        | –                          | 0.0130                           |
| <i>Toxicodendron radicans</i>    | Eastern poison ivy            | –                         | –                          | –                                | 0.125                    | 0.0214                     | 0.0700                           |
| <i>Trientalis borealis</i>       | starflower                    | 0.1250                    | 0.1250                     | fb                               | –                        | –                          | fb                               |
| <i>Ulmus americana</i>           | American elm                  | –                         | –                          | 0.0128                           | –                        | –                          | 0.1000                           |
| <i>Viburnum acerifolium</i>      | Mapleleaf viburnum            | –                         | –                          | 0.1094                           | –                        | –                          | –                                |
| <i>Vitis aestivalis</i>          | Summer grape                  | –                         | –                          |                                  | 0.289                    | 0.0212                     | <0.0001                          |

– = tested, but  $n < 5$  and/or  $T < 2$ , indicating no detectable change.

fb = forb species not included in dataset with no time restriction.

(i) = introduced species.

**Table 6.5—The 24 species with no change in frequency**

| Scientific name                    | Common name          |
|------------------------------------|----------------------|
| <i>Abies balsamea</i>              | Balsam fir           |
| <i>Aralia nudicaulis</i>           | Wild sarsaparilla    |
| <i>Asplenium platyneuron</i>       | Ebony spleenwort     |
| <i>Cornus canadensis</i>           | Bunchberry dogwood   |
| <i>Cornus florida</i>              | Flowering dogwood    |
| <i>Eurybia macrophylla</i>         | Big leaf aster       |
| <i>Galium circaeans</i>            | Licorice bedstraw    |
| <i>Glechoma hederacea</i> (i)      | Ground ivy           |
| <i>Hieracium aurantiacum</i> (i)   | Orange hawkweed      |
| <i>Impatiens capensis</i>          | Jewelweed            |
| <i>Lonicera japonica</i> (i)       | Japanese honeysuckle |
| <i>Maianthemum canadense</i>       | Canada mayflower     |
| <i>Oxalis montana</i>              | Mountain woodsorrel  |
| <i>Parthenocissus quinquefolia</i> | Virginia creeper     |
| <i>Plantago major</i> (i)          | Common plantain      |
| <i>Polygonum convolvulus</i> (i)   | Black bindweed       |
| <i>Polystichum acrostichoides</i>  | Christmas fern       |
| <i>Quercus alba</i>                | White oak            |
| <i>Rubus allegheniensis</i>        | Allegheny blackberry |
| <i>Rumex crispus</i> (i)           | Curly dock           |
| <i>Symphoricarpos orbiculatus</i>  | Coralberry           |
| <i>Trientalis borealis</i>         | Starflower           |
| <i>Uvularia sessilifolia</i>       | Sessileleaf bellwort |
| <i>Veronica officinalis</i> (i)    | Common gypsy weed    |

Species names followed by (i) indicates an introduced species.

is considered a late successional species, often occurring in the understory as a small tree. It is reported to be a preferred food for gypsy moth larvae, and may increase in abundance following defoliation (Gottschalk 1988, Hix and others 1991). Black cherry is shade intolerant (Marquis 1990), but the increase is detected on both disturbed and undisturbed plots, especially in the Hot Continental division. Marquis (1990) also notes that black cherry seedlings are common in understories of uncut stands, but do not live long without an opening in the overstory. Black cherry trees (*Prunus serotina*), while toxic to livestock, are important food sources to a variety of wildlife species, including birds who carry and deposit seeds away from the parent plant. Most other species showing change are restricted to one ecological division or the other, but there are several that show change in both ecological divisions, including red oak (*Quercus rubra*), American red raspberry (*Rubus idaeus*), and American elm (*Ulmus americana*). Evidence of increases in vine growth with elevated CO<sub>2</sub> has been demonstrated (Granados and Korner 2002 Smart 2004), which could influence several species listed in table 6.4 (including *Toxicodendron radicans* and *Vitis aestivalis*). Five species are well-established invasive species that continue to spread.

Using the augmented dataset for this analysis allowed for the correction of the missed tree species not included in the original dataset. Plots visited twice were reviewed as part of the quality control process to decide when to augment

species records, in addition to comparison of standard inventory tree records. As a result, plots where records were augmented had species lists for each visit that were more similar after the editing process than before. It was also important to consider species that had been recorded to the subspecies level to avoid false-positive results for some common species. One example is red maple, which was often recorded by some VEG crew as *Acer rubrum* var. *rubrum* rather than *Acer rubrum*. If this subspecies had not been consolidated to the species level, a false positive for changing frequency would have resulted.

### Constancy and Occupancy of Introduced Species

There is a trend of increasing constancy and occupancy across all levels of fragmentation (fig. 6.2 and table 6.6) (n=382). Significant increases are in relative richness and cover for intact plots, and relative richness on edge plots. The trends for increasing constancy and occupancy of introduced species are not surprising. Introduced species are well-established in the North Central States (Schulz and Gray 2013, Schulz and others 2013), with about 67 percent of plots where VEG was collected having at least one introduced species. Although intact plots were least infested, the constancy of introduced species by the second visit was approaching that of multiple condition and forest edge plots (fig. 6.2). Also, although intact plots remain less occupied by introduced

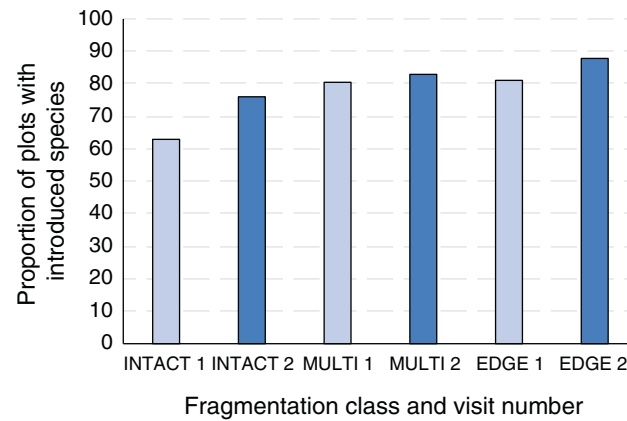


Figure 6.2— Constancy of introduced species by condition.

**Table 6.6—Relative richness and cover of introduced species by fragmentation class for each visit period, and the 2-tailed probability of equal means, as determined by the paired Student's t-test**

| Fragmentation class  | Metric            | Visit 1 | Visit 2 | Probability of equal means |
|----------------------|-------------------|---------|---------|----------------------------|
| -----percentage----- |                   |         |         |                            |
| Intact (n = 267)     | Relative richness | 5.04    | 5.72    | 0.0079                     |
|                      | Relative cover    | 3.48    | 4.18    | 0.0316                     |
| Multi (n = 41)       | Relative richness | 8.65    | 8.31    | 0.6238                     |
|                      | Relative cover    | 7.26    | 7.38    | 0.8964                     |
| Edge (n = 74)        | Relative richness | 10.22   | 12.28   | 0.0024                     |
|                      | Relative cover    | 11.40   | 10.50   | 0.4747                     |

species, they are significantly more occupied over time. Relative richness has increased significantly on forest edge plots, but relative cover is a bit lower, though not significantly so (table 6.6). Sample sizes of multiple condition and forest edge plots are smaller than for intact plots. Introduced species occupancy metrics are less affected by varying levels of botanical expertise by field crews because they are based on relative richness and cover of a group of species compared to all species identified. Using the augmented dataset resulted in higher sums of individual species cover on some plots. Both first and second visit data were reviewed and corrected during the quality assessment and augmentation process. Because most tally tree species are native, the overall effect on results would be to lessen the relative cover of introduced species for the plot visit where a tree was missed. Therefore, the proportion of cover of introduced species reported here for either visit period is less than what would have resulted if the original data were used in the analysis.

## CONCLUSIONS

For many ecological studies, having 514 plots to analyze would seem like a treasure trove of possibilities. However, a number of factors limit the types of analyses appropriate for this particular dataset. The spatial extent and number of forest types encountered depicts the diversity of the forests in the North Central and Northeastern United States and beyond, but limit the number of samples of any one particular forest or community type. The

number and types of changes that have occurred on those plots demonstrate the dynamic nature of the forests of the North Central and Northeastern United States, but limit the ability to relate the change in vascular plant composition to any individual factors. Potential differences in botanical skills of field crews could be a factor in the increases in observed species richness. Finally, fewer than 35 percent of plots were revisited within a 2-week calendar window, limiting the analysis of herbaceous species that may be present or observable in a short window of time due to phenology.

However, some conclusions can be drawn, and cautions for future analyses are noted:

1. Increasing species richness overall may be an artifact of early data collection challenges, but the trend is so widespread that it is also hard to dismiss. Another cycle of measurement with experienced crews would be required to determine if species richness was increasing with more certainty, but this is unlikely to occur in the foreseeable future. Others who may be initiating vegetation monitoring are cautioned to anticipate and scan for potential problems (crew experience, data management) in the early stages of a program so that they may be quickly resolved.
2. About one-half of the species tested showed some level of increasing frequency, with only one species showing a decrease (black oak). Common serviceberry and black cherry showed the most consistent increase across all datasets tested. Increasing frequency of other

species recorded are supported by the work of others. Using augmented data corrects for missed tally tree species and including species recorded as subspecies both reduce probability of false positives for false changes in species distribution.

3. Introduced species continue to increase in constancy and occupancy, especially on intact plots. These results are less influenced by differences in the botanical skills of monitors. Using the augmented dataset added to the cover of native species on those plots where trees were missed, resulting in a lower proportion of cover by introduced species than if the original plot data were used. The more conservative estimates of relative cover using the augmented dataset nonetheless indicate an increase in the occupancy of introduced species.

Although the number of plots available for these analyses was limited compared to most FIA-data based studies, the value of the VEG data is nonetheless demonstrated for particular analyses. Changes in species richness must be approached with care and are perhaps not the best fit for this particular dataset; analysts using VEG data to investigate trends in species richness should use the most recent data collected. Changes in species distribution over all, regardless of the recommended 2-week window, can be explored and reported for woody or non-ephemeral species over the entire dataset, and the changes on plots further explored for

possible hypothesis development. Additional analyses to explore include changes in overall structure (total cover by layer), shifts in ground cover variables, and rates of change for other metrics along ancillary environmental gradients or trends on plots with disturbance. These topics provide opportunities for analysts to gain experience working with a complex but unique vegetation database covering large regions of the United States.

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## INTRODUCTION

Young forests are pivotal in the forest life cycle because they set future compositional and structural trajectories following stand initiation disturbance, and hence, the potential for sustaining future forest values (Dey 2014). Heavily disturbed forests either regenerate to healthy young forest, lack stocking of desirable reproduction, or convert to shrub or other plant communities. Young forest is defined here as beginning with stand initiation and concluding early in the stem exclusions phase. Although specific ages, stem sizes, and stocking levels vary by forest type and other factors, all young forests represent an ephemeral stage of succession. Young forests of the Midwest and Northeast United States have emerged as a policy and management issue because they are rare and difficult to regenerate. This foretells a shortage of early succession forest habitat crucial for obligate and facultative flora and fauna, as well as landscape diversity.

Difficulty regenerating Midwest and Northeast forests has been expressed in the literature for decades (Loftis and McGee 1993, McWilliams and others 1995). These forests were largely established following large-scale clearing and timbering over a century ago when fire was prevalent; deer were rare; and the invasion of nonnative pests, pathogens, and plants had yet to occur. Regeneration management of today's forests is made difficult by contemporary drivers and stressors that were non-existent at the time of their establishment. Forest managers

must now consider several inter-related factors that challenge prescriptive decisions for re-establishing forest ecosystems:

- Herbivory,
- Lack of fire,
- Competing native vegetation such as mountain laurel (*Kalmia latifolia*) and eastern hayscented fern (*Dennstaedtia punctilobula*),
- Competing nonnative vegetation such as bush honeysuckle (*Lonicera* spp.),
- Nonnative pests and pathogens such as gypsy moth (*Lymantria dispar*),
- Lack of planning and poor cutting practices,
- High expense of regeneration management prescriptions (shelterwood, herbicide, fencing),
- Fragmentation and urban influence,
- Conflicting owner objectives,
- Pollutant deposition, and
- Changing climate.

These factors often act together to make it difficult and costly to successfully regenerate heavily disturbed forests.

Ongoing concern over an aging forest, dwindling young forest habitat, and regeneration problems resulted in the implementation of a Regeneration Indicator (RI) by the Northern Research Station (NRS) Forest Inventory and Analysis (FIA) Program that is responsible for monitoring forests of the Midwest and Northeast United States (McWilliams and others 2012). The RI was derived from more than 20 years

## CHAPTER 7. Forest Regeneration Indicator Results for the Midwest and Northeast United States

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of experience measuring regeneration and herbivory in Pennsylvania. The Pennsylvania Regeneration Study (PRS) is an ongoing investigation of regeneration adequacy for forest ecosystems under stress from herbivory and other factors (McWilliams and others 2017). The RI protocols include a suite of tree-seedling and browse impact measurements. The FIA program's three-phase inventory and monitoring system is described by Bechtold and Patterson (2005). The NRS-FIA modified the phase 3 protocols in 2012 to roughly double the sample size. Measurements are collected during the leaf-on season on a 12.5-percent subsample of the phase 2 samples, referred to as phase 2-plus. The baseline phase 2-plus inventory will take 7 years to complete. Each RI sample is coincident with phase 2 (USDA Forest Service 2013a) and other phase 2-plus indicator samples, including down woody material, vegetation structure, tree damage, invasive plants, and soils (USDA Forest Service 2013b).

The overall goal of this chapter is describe and demonstrate RI analyses conducted following the first 3 years of data collection (2012–2014). The examples utilize State-level analyses that have been used in NRS-FIA comprehensive reports to summarize findings and issues every 5 years. With 43 percent of the samples collected to date, it is recommended that substate-scale analyses consider statistical variation of estimates (sampling error) and the appropriate scale. Future research needs are discussed and conclusions are drawn concerning how to best

fill knowledge gaps on the status and condition of regeneration in the Midwest and Northeast United States.

## METHODS

### Study Area

The study area consists of 24 States that make up the NRS-FIA reporting region that spans the Plains, Lake, Central, Mid-Atlantic, and New England States (fig. 7.1). The area covers wide latitudinal and longitudinal range and diverse physiographic and climatic biomes. The study area's forests contain a mix of deciduous and coniferous forest-type groups but are dominated by broadleaf deciduous forests. Oak/hickory (*Quercus/Carya*) is the most prevalent forest-type group (36 percent of forest land) and is common across the southern tier of the study area. The maple/beech/birch group (25 percent) is found mostly across the northern tier. Only 5 percent and 4 percent of these two groups, respectively, are early succession forest from 0 to 20 years of age. Regeneration of oak/hickory and maple/beech/birch (*Acer/Fagus/Betula*) forest requires that seedling and sapling reproduction be established prior to harvest to ensure adequate stocking of desirable species following harvest disturbance, or "advance regeneration." Other common groups that are typified by younger forest are aspen/birch (*Populus/Betula*, 9 percent of forest land) and spruce/fir (*Picea/Abies*, 8 percent). These two groups have 23 percent and 7 percent of forest land in the 0 to 20 year age class, respectively.

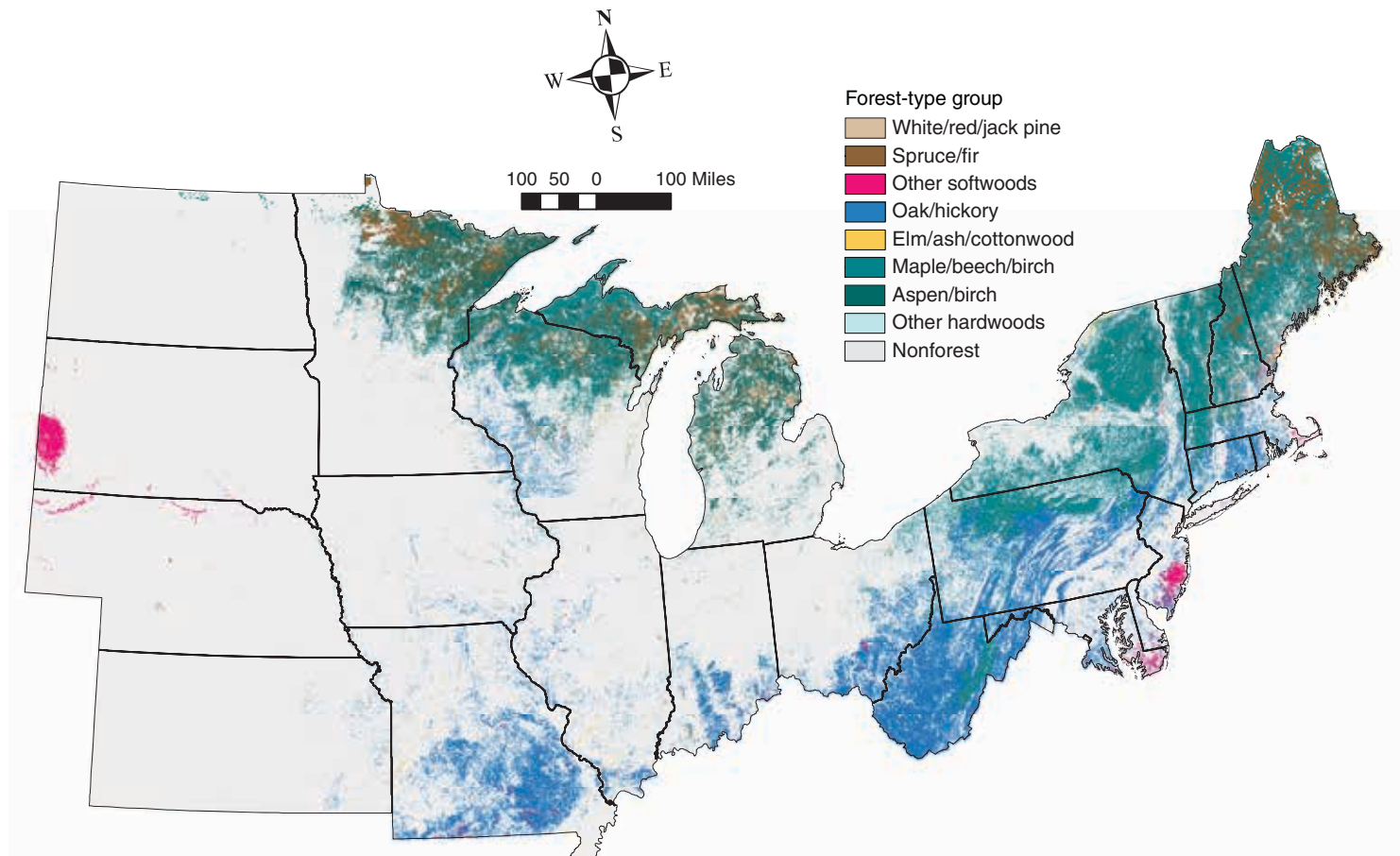


Figure 7.1—Distribution of forest land and dominant forest-type groups for the Midwest and Northeast United States, 2008 (Homer and others 2012, Ruefenacht and others 2008).

### Regeneration Indicator

Seedling height is an important factor for estimating likelihood of survival for naturally and artificially established seedlings, especially in areas with competing vegetation (Sander 1972). Most factors that influence tree seedling establishment and development are in play in the study area, including competing

vegetation, invasive plants, and herbivory. Tree seedling counts are needed to provide estimates of the number of seedlings per unit area for populations of interest and to gauge the adequacy of the tree regeneration process. Regeneration adequacy assessment requires thresholds for the number of seedlings needed to satisfy regeneration stocking requirements for FIA forest types.

The standard FIA phase 2 protocols for seedlings requires seedlings to be at least 1-foot tall for inclusion in the sample. The phase 2-plus RI protocols expands the tally to seedlings at least 2-inches tall that bear normally sized leaves and do not bear cotyledons, or established seedlings that have survived for at least one growing season. Tree seedlings are counted on forested microplot conditions by source and height class. Seedling source is coded as seedling, stump sprout, or a separate code expressing competitiveness of large-seeded species such as oak, hickory, and black walnut (*Juglans nigra*). The stump sprout code is useful because very often, sprouts outcompete seedlings. In the case of large-seeded species only, the level of competitiveness is determined primarily by evaluating root-collar diameter (RCD). An RCD of at least 0.25 inch defines a stem that is considered “established.” Large-seeded seedlings at least 3.0 feet in height are also included as established. Seedlings with an RCD of at least 0.75 inch and a d.b.h. (diameter at breast height) less than 1.0 inch are coded as “competitive.” The competitive code is used along with thresholds for regeneration stocking requirements in the analysis of regeneration adequacy. Seedlings are assigned to one of several height classes: 2.0 to 5.9 inches, 6.0 to 11.9 inches, 1.0 to 2.9 feet, 3.0 to 4.9 feet, 5.0 to 9.9 feet, and greater than or equal to 10.0 feet. The methodology for tree-seedling counts, seedling height measurement, and

RCD evaluation were adapted from long-term ecological and silvicultural research for mixed oak (Brose and others 2008) and northern hardwood (Marquis 1994) forests of the Mid-Atlantic region.

The browse impact code indicates the amount of pressure that herbivores are exerting on tree seedlings and other understory flora for the area surrounding the sample plot. Browse is defined as “the consumption of tender shoots, twigs, and leaves of trees and shrubs used by animals for food.” Browse impact is a function of browser population and the amount of available food sources within 1-square mile surrounding the plot. Five codes expressing browse impact are used:

- Very Low: plot is inside a well-maintained enclosure.
- Low: minimal browsing observed or vigorous seedlings present and of varied height (no well-maintained enclosure present). Herbaceous plants are present and they are able to flower and fruit.
- Moderate: browsing evidence observed but not common. Seedlings are common but with limited variability in height. Stump sprouts are heavily browsed or not evident. Herbaceous plants show a lack of or inhibited flowering and fruiting. There is little or no evidence of browsing on non-preferred plants.



- High: browsing evidence common on preferred vegetation. Preferred seedlings and herbaceous plants are rare or absent. Non-preferred plants show some evidence of herbivory and browse-resistant vegetation is limited in height growth. A browse line is beginning to be visible.
- Very High: browsing evidence is omnipresent. Non-preferred and browse-resistant plants show signs of heavy repeated browsing. A browse line is obvious.

The level of browse impact directly influences forest management practices and prescriptions (Latham and others 2005). In areas where ungulates exert moderate or high herbivory impact, it is recommended that forest managers consider silvicultural prescriptions to forestall impacts during the stand-initiation process (Brose and others 2008). The browse impact code may also be used to adjust regeneration stocking guidelines for comparison with existing stocking of seedlings and saplings.

### Spatial Analysis

In order to create continuous maps of herbivory impacts, indicator kriging (Isaaks and Srivastava 1989) within ESRI's ArcMap 10.2.2 Geostatistical Analyst package was used to create an interpolated surface from the browse impact

sample data. Indicator kriging is a geostatistical technique that models autocorrelation between observations as a function of distance. The spatial database consisted of the more than 1,700 phase 2-plus samples (fig. 7.2). Application of indicator kriging provided a visualization of the probability of occurrence by browse impact level. The resulting probabilistic visualization represents sub-continental risk of herbivory impacts on the composition, structure, function, and health of forest understories.

## RESULTS AND DISCUSSION

A series of examples of RI analyses are provided to demonstrate results and findings that address common regeneration issues. The examples provide guidance on using the results for reporting and research, as well as background for discussing research needs to expand the use of the RI results.

### Tree Seedling Regeneration

Extending the tally of tree seedlings from 1-foot tall down to 2 inches and including height-class measurements creates opportunities for more detailed investigation of young forest dynamics than previously possible from FIA data. The results are used for 5-year NRS-FIA reports, annual issue reports, and other FIA outlets.

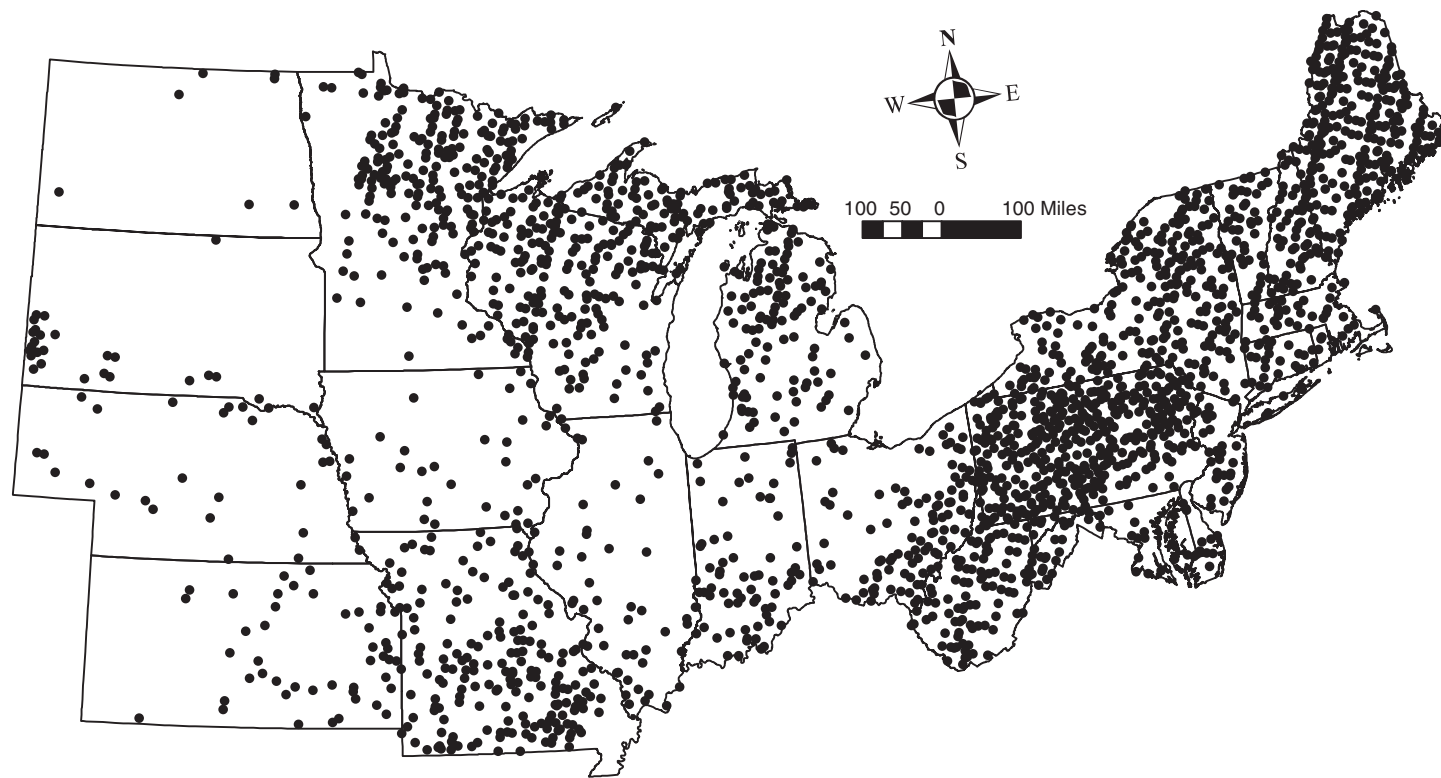


Figure 7.2—Distribution of Forest Inventory and Analysis (FIA) sample locations, Midwest and Northeast United States from 2012 to 2014, and Pennsylvania from 2009 to 2014. (All locations are approximate).

The basic regeneration storyline for standard FIA reports is a description of the abundance, composition, and structure of seedlings and what this means for resource sustainability. The example for West Virginia utilizes population estimates for seedlings derived by expanding the phase 2-plus sample to the State level. The results illustrate an overwhelming dominance of red maple, seedlings less than 1-foot tall, and a lack of seedlings representing the State's dominant overstory species, e.g., northern red oak (*Q. rubra*), white pine (*Pinus strobus*), aspen

(*Populus grandidentata* and *P. tremuloides*), and red pine (*P. resinosa*) (fig. 7.3). These findings provide evidence that oak and other important species may not be sustainable for the long-term future. The seedling estimates shown do not include a breakdown of the seedling population by species and height class because it is based on only two of seven inventory panels (2012 and 2013), which results in very high sampling errors. This is apparent by examining the error bars for individual species. Chestnut oak is the third most important seedling species, but is associated

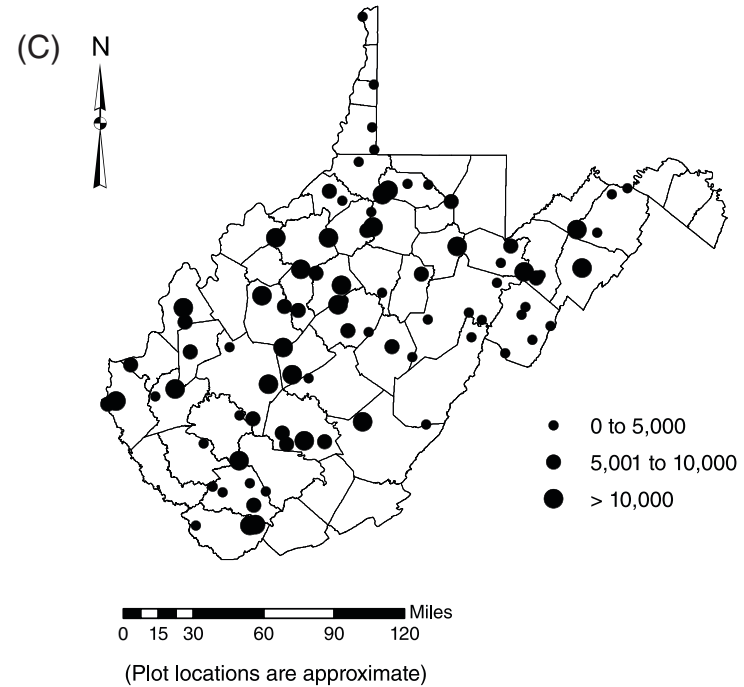
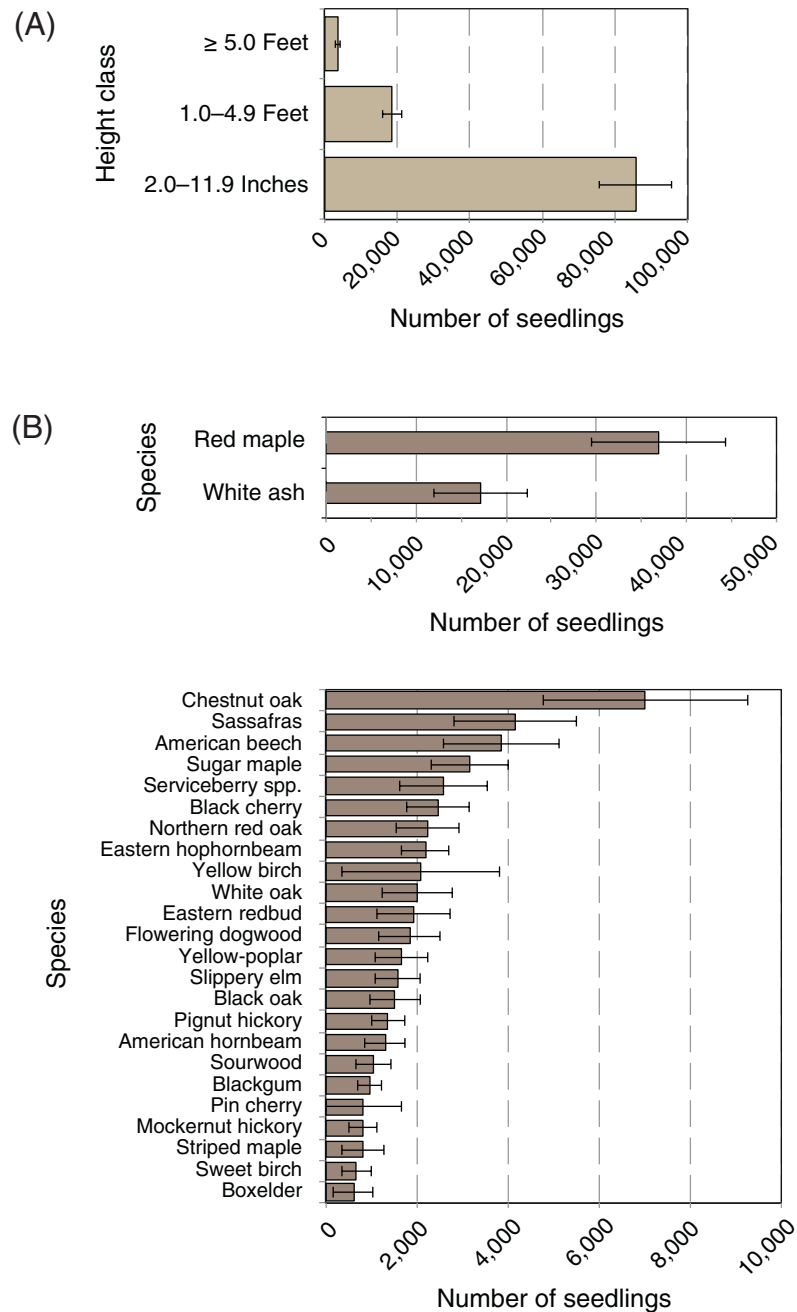


Figure 7.3—(A) Number of seedlings by height class, (B) number of seedlings for species with at least 1 percent of the total number of seedlings, ranked by species, and (C) distribution of forested phase 2-plus samples by the average number of seedlings per acre in West Virginia, 2012–2013. Error bars represent 1 standard error or a 68-percent confidence interval.

with a 33-percent sampling error. Typically, FIA analyses do not consider estimates with sampling errors greater than 25 percent, but rather consider the relative distribution of the number of seedlings by species.

Although the limits of sample size cannot be completely overcome, combining species and focusing on the more prevalent taxa offers insight into impacts of seedlings on future forest condition. The results for Missouri highlight how the seedling information can be compared with saplings and adult trees (fig. 7.4). It is apparent that oak seedlings are under-represented when

compared to saplings and adults. Maple seedlings are over-represented. These results provide an early indication of prospective maple incursion into larger size classes. Other species that are dominant in the overstory but under-represented in the sample are black oak (*Q. velutina*), shortleaf pine (*P. echinata*), and black walnut.

Considerable value is added to the analyses when the tree reproduction sample is compared with accepted regeneration stocking guidelines to assess regeneration adequacy. So far, applicable research for estimates of regeneration adequacy is only available for the Mid-Atlantic region

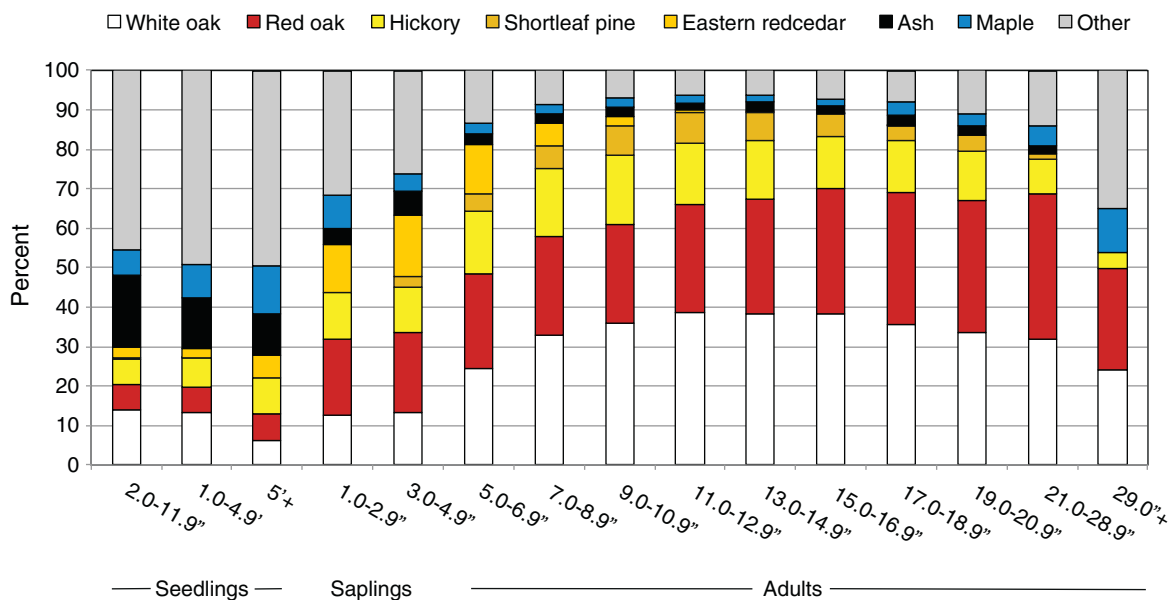


Figure 7.4—Percent of total number of seedlings and dominant/co-dominant live saplings and growing-stock trees on forest land for select species by species group and size in Missouri. Seedling estimates are for 2012–2013. Sapling and tree estimates are for 2009–2013.

based on research from the PRS. The approach evaluates samples in a range of stocking with enough light for seedling establishment. The regeneration stocking guidelines are adjusted using the browse impact code, based on the assumption that more seedlings are required for higher impacts (McWilliams and others 2014). Findings are reported as the percent of samples with adequate regeneration of seedlings and saplings.

The FIA definition of a tree includes all perennial woody species that typically produce a single stem and grow to at least 15 feet in height. Because this definition includes such a wide array of phenotypes, simply assuming that a tree seedling of any species is acceptable for adequate regeneration can produce uninformed results. The FIA list of species provides the opportunity to characterize regeneration for a wide range of research objectives. Some common analytical questions of interest include evaluating FIA species that are: (1) native, (2) endemic, (3) nonnative, (4) invasive, (5) preferred for timber production, (6) able to achieve high canopy position, (7) match overstory species, and (8) combinations of these and other characteristics.

Two broad species groups from the PRS were used as indications of regeneration success to provide a means to evaluate groupings of taxa based on regeneration composition objectives. The most basic approach assesses whether a forest has the ability to regenerate with high canopy species. The High Canopy Dominant group includes species with at least 2 percent

of the total aboveground tree biomass in Pennsylvania and that have the potential to achieve a high canopy position, or a replacement index (fig. 7.5). The All High Canopy group includes all species with the potential to reach high canopy regardless of contribution to total

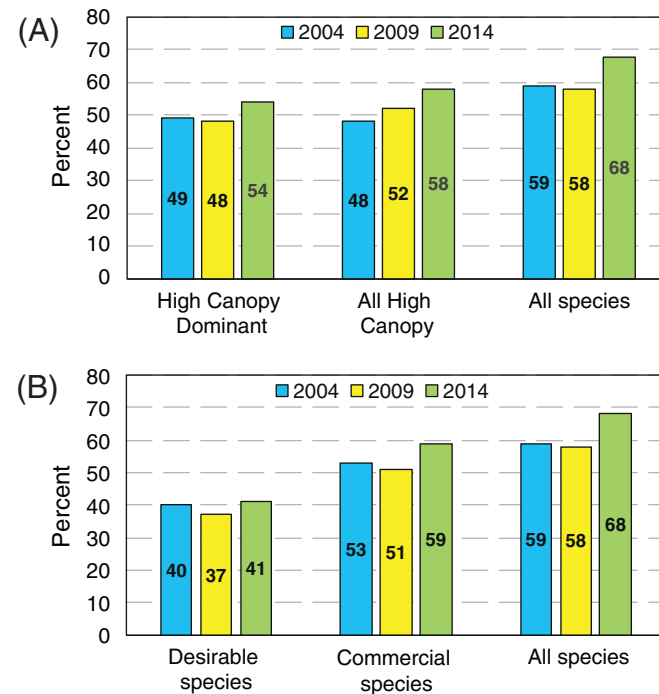


Figure 7.5—(A) Percent of regeneration samples adequately stocked with advance regeneration for stands from 40- to 75-percent stocked with live trees by canopy species group in Pennsylvania for 2004, 2009, and 2014; and (B) percentage of regeneration samples adequately stocked with advance regeneration for stands from 40- to 75-percent stocked with live trees by commercial species group, also in Pennsylvania for 2004, 2009, and 2014.

biomass. An alternative approach is to group species based on preferences for producing crops of timber. The Desirable group contains species most desired for timber production in Pennsylvania. The Commercial group includes all species capable of producing a commercial crop of timber.

After decades of little change, the first signs of improvement in regeneration success for canopy species appeared in the 2014 inventory results. The results show improvement for the Commercial species group, but no improvement for the Desirable species group.

### Herbivory Impact

Assessment of herbivory usually includes discussion of the percentage of forest land by impact level and the distribution of impacts across the forested landscape. Pennsylvania was chosen as the example because it has a full set of inventory panels, making for more statistically reliable results than States with partial inventory completion. Results indicate that about 8 out of 10 samples had moderate (60 percent) or high (21 percent) levels of herbivory impact on understory plants. (Note: the very low and low classes and the high and very high classes were combined to minimize sampling error of the estimates.) The impact of herbivory will need to be taken into account for most of the forest land in Pennsylvania following stand-initiation disturbance. Examination of impact across the State reveals areas at higher risk of impact relative to areas at lower risk. The visualization

of the probability of exceeding low (moderate and high) impact shows high probabilities of impact in all but the south central part of the State (fig. 7.6A). The visualization of the probability of exceeding medium impact shows that these samples were most common in the central and northeast areas (fig. 7.6B).

### Small Area Reporting

Small sample sizes often limit the ability to make conclusions about herbivory impacts and other variables. This was the case when the NRS-FIA 5-year reports for Delaware, Maryland, and New Jersey were due for the year 2013. With only two out of seven inventory panels complete, it was necessary to use regional results to interpret the visualization of herbivory impact (fig. 7.7). The Pennsylvania samples provide context for bordering States. The implication is that although sampling errors for individual States are too high to make meaningful conclusions, combining the results with the Pennsylvania data assists in making broad interpretation of the findings. The relatively high impacts of the Outer Coastal Mixed province align with findings for the adjacent Eastern Broadleaf province of Pennsylvania.

### Research Needs

As with any new ecological health indicator, there are numerous research extensions and applications to enhance the utility of science products. Future research should focus on an analysis of composition and structure of the



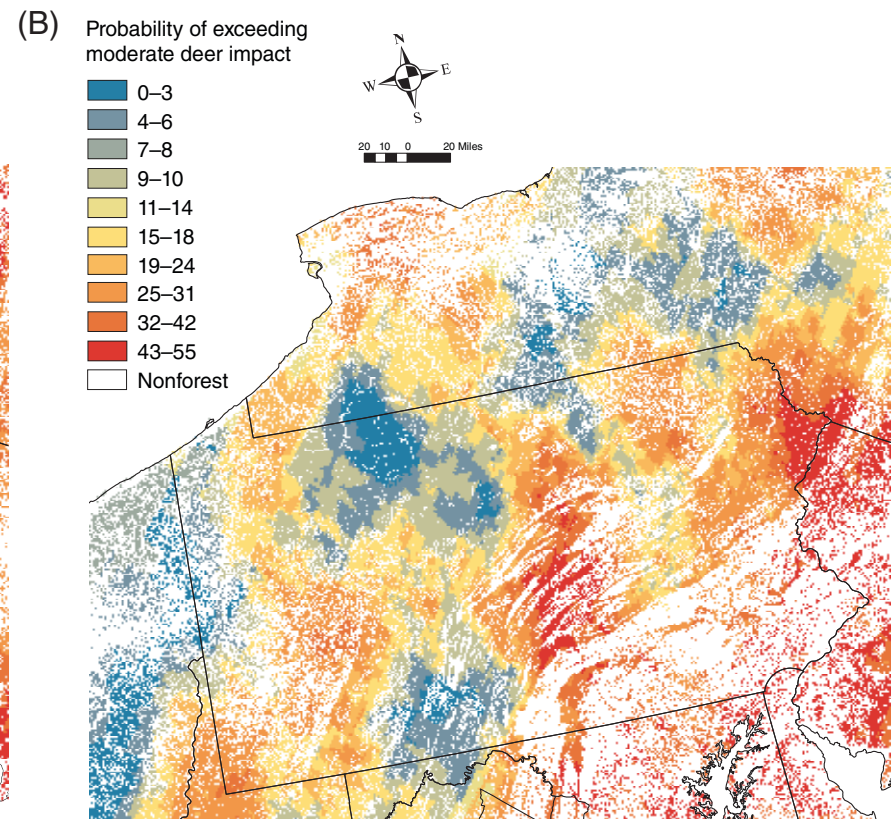
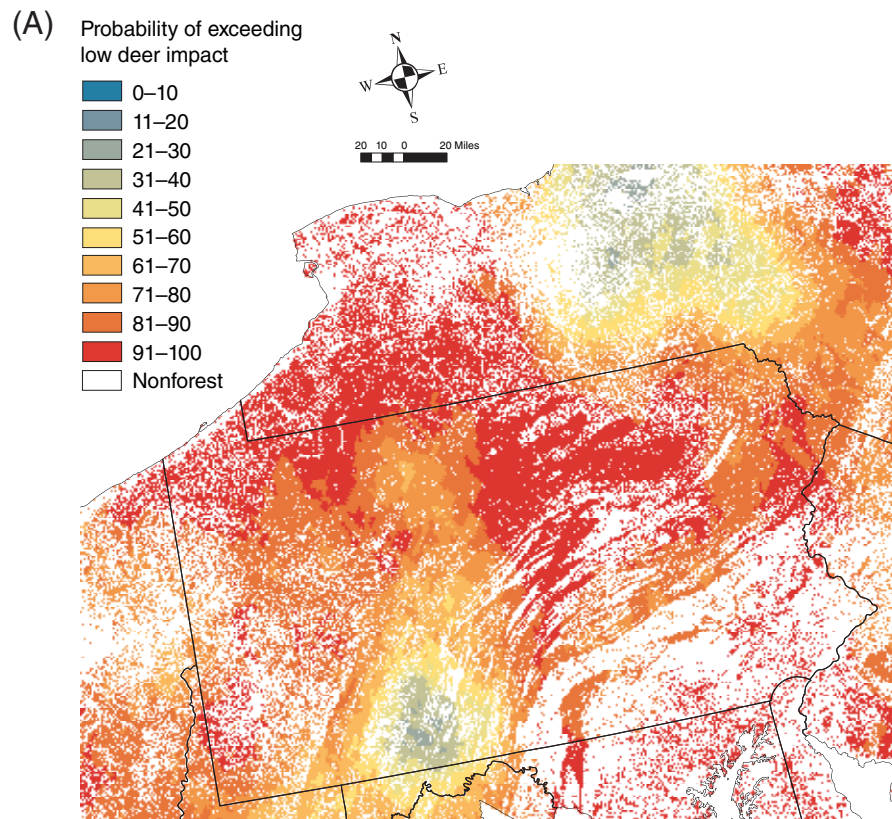


Figure 7.6—(A) Probability of exceeding low browse impact and (B) probability of exceeding moderate browse impact for Pennsylvania in 2014.

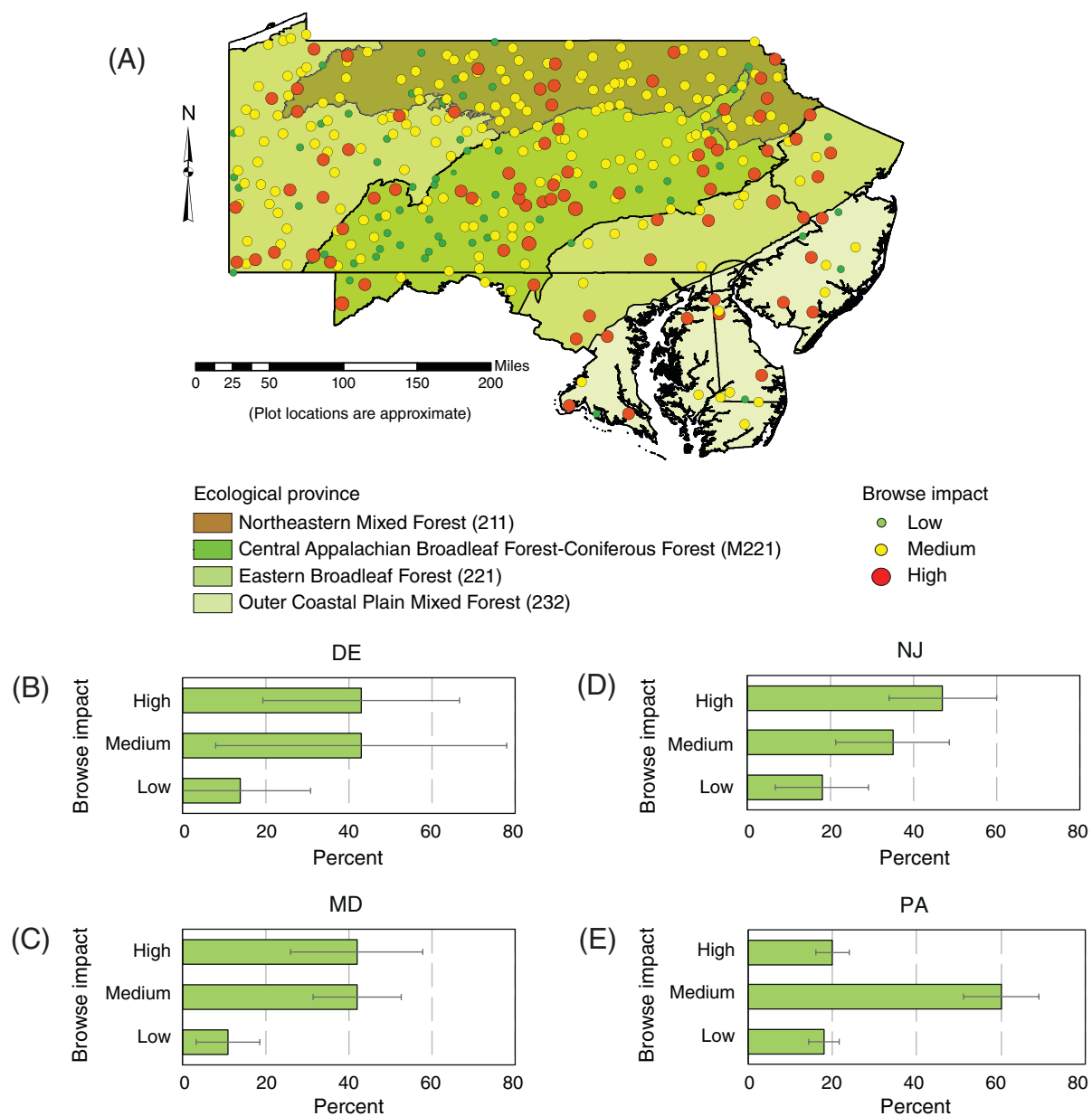


Figure 7.7—Distribution of forested phase 2-plus samples (A) by browse impact and ecological province (Cleland and others 2007), in Delaware (B), Maryland (C), New Jersey (D), from 2012 through 2014, and in Pennsylvania (E), from 2010 through 2014. Error bars represent 1 standard error or a 68-percent confidence interval.

seedling component as the dataset expands and on geospatial analyses to identify high-risk areas, such as forest types with poor regeneration.

Research is also needed to address the viability of the regeneration process. While analytics that adjust existing regeneration guidelines to reflect herbivory impact have been applied for the Mid-Atlantic States (McWilliams and others 2014), similar analytics for the Central, Lake, and New England States are needed. Once complete, such metrics will facilitate a seamless and transparent assessment of regeneration adequacy for the major forest types of the Northeast and Midwest. This is a critical need due to aging of mixed oak and northern hardwood forests of the region and regeneration stress factors that interact to make regeneration difficult.

Modern resource questions often require multivariate studies that combine geographic data to better understand complex relationships. There are many research opportunities to integrate the RI data with other publicly available geographic datasets. For example, tree-species migration studies would benefit from including soils (USDA Natural Resources Conservation Service 2015), climate (NOAA National Weather Service 2015), and disturbance (USDI Geological Survey 2015).

The RI was designed to supplement NRS-FIA's vegetation profile and invasive plant survey information. Combining results from these three components of the vegetation structure measurements provides a fuller appraisal of the forest understory that will better address

emerging issues, e.g., the status and condition of new forest communities (Royo and Carson 2006). In turn, this should improve our ability to assess sustainability of future forest values in the Midwest and Northeast.

## SUMMARY

The forest reproductive process requires established seedlings and saplings to ensure future forest cover. Regeneration management challenges often make it difficult to establish a desirable species mix with the abundance needed to restore conditions favorable for an even flow of goods, services, and values that the public has become accustomed to. A quarter century of analyzing the results from the PRS has re-affirmed some basic management practices for broadleaf forests under herbivory stress:

- Herbivory needs to be considered before harvest for stand initiation in moderate and high pressure conditions;
- Establishing, recruiting, and retaining competitive seedlings from the mature stand prior to harvest through the stem exclusion phase will need to consider contemporary drivers and stressors, e.g., invasive plants;
- A successful technique is the shelterwood system and related management prescriptions described by Brose and others (2008);
- Fire is a useful and well understood practice in Pennsylvania restoring a natural process; and
- Fencing is often needed to reduce herbivory and facilitate application of fire, herbicides, and other controls.

Jackson and Finley (2011) summarize these practices as control of competing vegetation, deer, and light.

Extending the PRS protocols to all of the NRS-FIA States means that similar kinds of conclusions can be made across the region. The results presented here reflect only a subset of the seven panels of measurements that will eventually comprise the first sub-continental full baseline dataset for the RI. The dataset will also facilitate research to evaluate plot-level regeneration adequacy for the major forest-type groups and a more complete understanding of future trends in composition, structure, and health of the region's forest ecosystems.

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## INTRODUCTION

The 2012 National Insect and Disease Risk Map (NIDRM) estimates potential tree mortality from insects and diseases (pests) over a 15-year future time period (2013–2027) across most treed areas of the United States (Krist and others 2014). It is derived from science-based, transparent geospatial methods, as well as input from forest health specialists in the field. The NIDRM comprises 186 forest insect and disease models parameterized by entomologists and pathologists, each representing specific pest-host combinations within Bailey’s ecoregions (Bailey 1995). The results of these models support prevention and suppression activities related to integrated forest pest management, forest restoration efforts, ecosystem resiliency management, short- and long-term monitoring efforts, and fuels and fire management (Potyondy and Geier 2011). Furthermore, NIDRM supported the 2014 Farm Bill (H.R.2642–Agricultural Act of 2014) by aiding the designation of areas for treatments that reduce hazard from forest insects and diseases; more than 200 thousand square kilometers (approximately 50 million acres) have been designated to date (USDA Forest Service 2016).

Since its introduction in 2000 (Lewis 2002), NIDRM has moved beyond a static cartographic product to an interactive set of consistent and repeatable models and geographic information system (GIS) layers that provide comparable results across all land ownerships and political boundaries (fig. 8.1). The 2006 (Krist and

others 2007) and 2012 (Krist and others 2014) assessments focused heavily on improving GIS data consistency and quality, model parameters, and forest health specialist participation. More than 150 Forest Service staff and Federal partners contributed directly to the development of the most recent NIDRM. This participation included informal comparisons of interim NIDRM results with existing forest conditions and local field knowledge. Despite this and other enhancements since 2000, products from NIDRM have not been formally validated over large areas. Model validation is critical for informing discussions about improving NIDRM as well as appropriate uses of NIDRM at various scales.

The goal of this paper is to ascertain if—for insects and diseases that cause significant measurable tree mortality at the regional level on an annual basis (Krist and others 2007)—the general magnitude of potential mortality predicted by NIDRM across all host tree species is confirmed by actual observations of mortality. NIDRM models were re-run using forest parameter input datasets characterizing the forest conditions of 2002, thereby providing an estimate of mortality for the period 2002–2016. These results were then compared to Forest Inventory and Analysis (FIA) estimates of mortality for a similar period. FIA data were chosen as a comparative tool because FIA is the only source of rigorously collected and statistically valid ground data that reports on the status and trends in health and mortality of trees across all forested land ownerships in the

## CHAPTER 8. Validating the National Insect and Disease Risk Map using Mortality Estimates from Forest Inventory and Analysis

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# NIDRM Progression .....

## INCREASING:

- Complexity
- Repeatability
- Transparency
- Accuracy
- Precision
- Usability

Improved resolution (more management relevant)  
Better accuracy and precision  
Technology transfer of GIS to entomologists  
and pathologists

2001

Simple cartographic exercise  
Motivated new thinking

2006

Engaged regions  
Advanced standard GIS data and models

2012



Figure 8.1—Progression of the National Insect and Disease Risk Map (NIDRM) in recent history.

United States. We restrict our analysis to the Eastern United States for a variety of reasons, including: (1) the availability of complete FIA panels is more robust in the East, and (2) many western NIDRM models use drought drivers that would need to be retrospectively adjusted to adequately perform our evaluation. This analysis retrospectively adjusts only the forest parameter inputs discussed below. Future validation studies will consider retrospectively modifying a broader range of model drivers, such as drought frequency.

NIDRM summarizes trends in tree mortality in support of broad-scale (national and landscape) planning. Analysis and interpretation of NIDRM results below the county or national forest level is not recommended (Krist and others 2014), but questions remain as to what minimum spatial unit is most accurate and informative. Therefore, we focus on comparisons between FIA and NIDRM mortality at various spatial scales. This comparison seeks to not only determine how reliable NIDRM is for broad-scale planning, but also how well the results perform at smaller summary units.

## METHODS

Despite the need for validation, the results arising from a comparison of NIDRM and FIA are conditional on a number of important assumptions and caveats that merit consideration upfront. First, key drivers of mortality can change over time (e.g., stand density may have been a stronger controller of mortality in the past, whereas drought may be

a stronger driver now), meaning that a NIDRM model developed in one time period may not be equally well parameterized for another time period. Second, NIDRM simulations assume no management is occurring on the ground. Many areas across the United States undergo forest management activities that can reduce or eliminate hazards associated with insects and diseases. Third, NIDRM has known weaknesses in certain areas of model development, especially in areas of low forest density, as well as areas where certain hosts could not be modeled. In these cases, we expect NIDRM to under-predict mortality relative to observations. Fourth, areas of high hazard according to NIDRM may not realize insect or disease mortality at the predicted levels within the 15-year time horizon, principally because of varying modeled and other non-modeled factors. Examples include variations in weather patterns, such as the onset of drier or wetter conditions, which may be needed to actually trigger mortality. Under these scenarios, we expect NIDRM to over-predict mortality. Fifth, due to the periodic nature of FIA plot re-measurements, FIA inventories may not detect mortality until many years after it has occurred. Sixth, FIA inventory plots typically represent many thousands of hectares when they are used to derive population-level estimates. It is likely that the relatively coarse geographic scale of the FIA inventories often fails to detect insect and disease activity occurring at finer spatial scales. Seventh, this analysis assumes that FIA's attribution of tree mortality from insects and diseases is accurate. Thus, interpretation

of NIDRM and FIA mortality estimates is complicated, especially at fine spatial scales. Accordingly, we examine and report patterns over broad spatial extents.

### Areas of Interest

We evaluated five different ecological and administrative areas of interest (AOI) that varied in size by approximately two orders of magnitude. At the largest scale, we utilized the publicly available FIA Environmental Monitoring and Assessment Program (EMAP) hexagons (approximately 650 km<sup>2</sup>) (Brand and others 2000), while at the smallest scale we used States (2 838–218 546 km<sup>2</sup>; average approximately 111 000 km<sup>2</sup>) and ecological sections, hereafter termed “ecosections” (average approximately 42 500 km<sup>2</sup>) (McNab and others 2007). Mid-scale, we considered ecological subsections (“ecosubsections”; average approximately 6 300 km<sup>2</sup>) (McNab and others 2007) and counties (average approximately 2 500 km<sup>2</sup>). Note that the ecosections and ecosubsections by which we summarize data represent the land classification system used by FIA (McNab and others 2007), which differs from Bailey’s (1995) classification system used by NIDRM model developers when they built (and associated with ecoregions) individual risk models. Although different in their exact designations, these two classification systems are comparable. Combined, these AOIs allowed us to examine similarities versus differences in tree mortality across a variety of spatial units commonly used in reporting forest conditions. In the past (Krist and others 2014), NIDRM has

used U.S. Geological Survey 12-digit hydrologic unit code subwatersheds (U.S. Geological Survey and others 2013) in reporting. Although we did not consider subwatersheds in the present study (FIA estimates of mortality were not readily available for watersheds), for comparison purposes in the results below it is worth noting that their average size of approximately 90 km<sup>2</sup> is nearly an order of magnitude smaller than the EMAP hexagons.

### FIA Estimates of Tree Mortality

We obtained population-level estimates, by AOI, of live tree basal area (BA) and mortality BA derived from FIA’s most recent phase 2 inventories (Bechtold and Patterson 2005). These estimates were obtained across all areas designated “forest land” by FIA and were from inventory cycles ending between 2013 and 2015. “Mortality,” as designated by FIA, includes those trees that have died between the previous and current measurement cycles. The population-level estimates are derived from the point-based FIA inventory estimates using previously published, plot-specific expansion factors reflecting the areal extent represented by each inventory plot, as described in Scott and others (2005). Mortality estimates were originally partitioned among seven classes designating the *cause* of mortality (insect, disease, fire, animal, weather, vegetation, and unknown/other). Our primary focus was on the causes attributable to insects or diseases, but we also considered “unknown/other” for two major reasons. First, it is reasonable to assume that mortality trees assigned to the unknown/other class in many

cases have a biotic agent involved in their demise. While it is unlikely that *all* mortality trees assigned to the unknown/other class died as a result of insects or diseases, we believe that consideration of this mortality class is reasonable for this analysis. Second, we found the use of the unknown/other category to be concentrated in certain areas (data not shown). For example, across Eastern counties, the total mortality attributable to insects and diseases (combined) and to unknown/other were 30 percent and 33 percent, respectively. In counties where attribution to insects and diseases (combined) was <null> (approximately 12 percent of all counties), the attribution to unknown/other was 72 percent of total mortality. In counties where the unknown/other category was <null> (about 25 percent of all counties), the percentage of total mortality attributable to insects and diseases is 50 percent. Together, these observations suggest that it is likely that the unknown/other category is at least partially confounded with what other observers would attribute to insects or diseases, at least in some locations.

Therefore, we developed three FIA mortality scenarios for comparison with NIDRM's estimates of insect- and disease-caused tree mortality. Under the first scenario, we summed within each AOI the FIA annual mortality estimates attributable to insects and diseases. In the second scenario, we also derived an FIA mortality estimate based upon FIA's assignment to the causes of insects, diseases, and 100 percent of unknown/other. These two scenarios likely bracket the range of mortality

that is due to insects and diseases, but for reasons described above they likely either underestimate (first scenario) or overestimate (second scenario) mortality caused specifically by insects and diseases. Consequently we developed a third intermediate scenario that estimated FIA mortality based upon insects, diseases, and 50 percent of unknown/other.

FIA estimates of annual mortality under the three scenarios were obtained only for areas east of Montana, Wyoming, Colorado, New Mexico, and west Texas. East Texas is inventoried as a distinct unit in FIA and is treated as a separate "State" in this analysis. The population-level estimates from FIA represent average annual stand densities and average annual mortality rates for the period of time between the current measurement cycle and the previous measurement cycle. Assuming a 5-year re-measurement cycle, common in the Eastern United States, this would correspond to an estimate spanning the range from approximately 2007–2012 (i.e., the midpoints of the two recent measurement cycles). We inflated this periodic annual rate to a 15-year mortality rate for comparison with NIDRM.

### **NIDRM Estimates of Tree Mortality**

The pest and host combinations modeled in the NIDRM were selected because they account for the vast majority of the annual insect and disease caused tree mortality nationwide. Of the 186 risk models composing NIDRM nationally, 41 models exist within our Eastern U.S. study area and were re-run for this analysis. Each predicts potential 15-year mortality for one



or more tree species that serve as host to a single pest in a specific geographic area. A total of 43 pests and 60 host tree species or species groups were modeled in the 2013–2027 NIDRM (Krist and others 2014). The subset of 41 models in this analysis includes 25 different insect and disease agents or agent groups acting on 33 different host tree species or species groups (table 8.1). Each NIDRM model uses a combination of various *model criteria* (or *model drivers*) as weighted GIS overlays in the “Risk Modelling Application” (RMAP), a custom software application designed to facilitate the development of spatial risk models by entomologists and pathologists who may have little to no GIS expertise (Krist and others 2014). Model criteria include forest parameters as well as physiographic, climatic, edaphic, and other factors.

Forest parameter layers used as input to NIDRM include metrics such as BA per acre, stand density index, and quadratic mean diameter for nearly all tree species inventoried by FIA in the continental United States. Their development included modeling of conditions circa 2002 as well as estimates of conditions circa 2012 derived by accounting for growth and mortality since 2002. For details on the creation of these datasets, refer to Krist and others (2014), and Ellenwood and others (2015). Although the time periods represented by the forest parameter layers are approximate, we refer to these datasets by their dates. The 2012 parameters were used to model potential effects of insects and diseases for the

**Table 8.1—Forest pest and host tree species combinations modeled for this analysis in the Eastern United States**

| Forest pest                 | Host tree species  |
|-----------------------------|--------------------|
| Asian longhorned beetle     | Red maple          |
| Asian longhorned beetle     | Sugar maple        |
| Balsam woolly adelgid       | Balsam fir         |
| Beech bark disease          | American beech     |
| Blight                      | Bur oak            |
| Decline                     | Cottonwood spp.    |
| Decline                     | Quaking aspen      |
| Dutch elm disease           | American elm       |
| Eastern larch beetle        | Tamarack           |
| Emerald ash borer           | Ash spp.           |
| Forest tent caterpillar     | Aspen spp.         |
| Fusiform rust               | Southern pines     |
| Hemlock woolly adelgid      | Hemlock spp.       |
| Heterobasidion root disease | Southern pines     |
| Heterobasidion root disease | Eastern white pine |
| Heterobasidion root disease | Jack pine          |
| Heterobasidion root disease | Longleaf pine      |
| Heterobasidion root disease | Red pine           |
| Ips engraver beetles        | Southern pines     |
| Jack pine budworm           | Jack pine          |
| Laurel wilt                 | Redbay             |

| Forest pest                 | Host tree species  |
|-----------------------------|--------------------|
| Laurel wilt                 | Sassafras          |
| Maple decline               | Sugar maple        |
| Mountain pine beetle        | Limber pine        |
| Mountain pine beetle        | Ponderosa pine     |
| Oak decline                 | Red oaks           |
| Oak decline                 | White oaks         |
| Oak wilt                    | Red oaks           |
| Oak wilt                    | Live oak           |
| Other root or butt diseases | Paper birch        |
| Other root or butt diseases | Ponderosa pine     |
| Other root or butt diseases | White spruce       |
| Sirex woodwasp              | Southern pines     |
| Southern pine beetle        | Southern pines     |
| Southern pine beetle        | Eastern white pine |
| Southern pine beetle        | Longleaf pine      |
| Southern pine beetle        | Pitch pine         |
| Spruce budworm              | Balsam fir         |
| Spruce budworm              | Spruce spp.        |
| White pine blister rust     | Eastern white pine |
| Winter moth                 | Oaks               |



2013–2027 NIDRM; the 2002 datasets were used in this analysis to evaluate the NIDRM’s 15-year mortality prediction over the 2002–2016 time frame, hereafter referred to as the *retrospective NIDRM*.

For this study, in order to estimate mortality over a time period corresponding to observations of mortality from FIA, the 2002 forest parameters datasets were loaded into the RMAP and the NIDRM models were re-run. None of the other NIDRM model criteria were adjusted. Although many if not most of the model criteria used in the NIDRM are relatively static (e.g., topography, soils), a few, such as drought, used as a driver primarily in the Western United States, are dynamic within a 15-year time frame. In the models analyzed in the present study, drought was used in only 5 of the 41 eastern NIDRM models.

The retrospective NIDRM outputs representing potential 15-year mortality across all treed lands were summed and masked to include only those locations recognized as “forest land” by FIA. Retrospective NIDRM results were summarized for each AOI, using Spatial Analyst Zonal Statistics tools in ESRI’s ArcGIS software.

### Comparative Analysis

We performed linear regressions on estimates of both mortality and live BA from the retrospective NIDRM against corresponding estimates from FIA; live BA estimates were compared in order to evaluate the bases for the mortality comparisons. The purpose of

the regressions was to assess goodness of fit (R-squared), slope, and y-intercept. Perfect agreement between NIDRM and FIA would be indicated by an R-squared of 1, a slope of 1, and a y-intercept of 0. For AOIs where regression parameters are reasonably close to these values, we mapped the residuals (the difference between the observed value and the regression’s fitted value) for each AOI, expressed as a percentage of the regression’s fitted value. Areas of high agreement in mortality BA are indicated by residual percentages near 0; large positive percentages suggest NIDRM over-predicts mortality and large negative percentages suggest NIDRM under-predicts mortality relative to FIA estimates. We considered NIDRM mortality estimates to (1) match FIA regression model expectations if residuals were  $\pm 50$  percent, (2) under-predict if residuals were  $< -50$  percent, and (3) over-predict if residuals were  $> 50$  percent. The  $\pm 50$  percent threshold represents one standard deviation of the residual percentages at the State level.

## RESULTS AND DISCUSSION

In this section, we begin by discussing statistical relationships between NIDRM and FIA for live and mortality BA, in order to evaluate whether overall patterns predicted by the retrospective NIDRM are consistent with those observed by FIA. We then drill down and examine geographically where NIDRM and FIA mortalities are different across AOIs, focusing especially on areas where the retrospective NIDRM tended to under- or over-predict mortality relative to FIA expectations. We

emphasize these areas so that future hazard models for forest pests may be improved. When interpreting these geographic patterns in our examples below, we often examine specific insects and diseases that are major contributors to overall tree mortality. Finally, we conclude our chapter with discussion of ideas for extending and improving these comparative analyses, and how they may inform development of a future NIDRM.

### Overall Relationships between NIDRM and FIA

We first assess relationships between estimates of live BA originating from NIDRM and FIA. Across all five AOIs, slopes from the regressions are all close to 1 (0.93-0.94);  $y$ -intercept values are slightly negative; and R-squared values range from 0.86 (EMAP hexagons) to 0.97 (States and ecoregions) (table 8.2). Thus, although the relationship indicates that NIDRM's estimates of live BA are generally slightly lower than FIA's, the relationship is near to ideal and consistent across scales. This strong agreement reveals that the starting conditions in NIDRM are commensurate with those from FIA, and thus do not constitute a significant source of variation when evaluating differences in mortality estimates.

An example of comparing live and mortality BA estimates (across the three FIA mortality scenarios) is presented for States (fig. 8.2, with regression results reported in table 8.2). With this AOI, live BA estimates were found to be highly correlated between NIDRM and FIA (fig. 8.2A). State-level NIDRM mortality was in

closest agreement with FIA mortality scenario 3 (mortality from insects, diseases, and 50 percent of unknown/other mortality, fig. 8.2D), based on all three regression parameters: slope,  $y$ -intercept, and R-squared (table 8.2). Although neither scenario 3 nor scenario 1 (FIA mortality from insects and diseases only, fig. 8.2B) had slopes that were significantly different than 1 (table 8.2), scenario 3 exhibited a smaller (closer to zero)  $y$ -intercept and larger (closer to 1) R-squared value, compared to scenario 1. Scenario 2 (FIA mortality from insects, diseases, and 100 percent of unknown/other, fig. 8.2C) had the smallest slope (furthest from 1) and, moderate  $y$ -intercept and R-squared values.

Regressions results for the four other AOIs are also shown in table 8.2 and figure 8.3. In most cases, scenario 3 exhibited the strongest associations (i.e., slopes and R-squared values closer to 1,  $y$ -intercepts closer to 0). Furthermore, under scenario 3, slopes were not significantly different from 1 for States, counties, and ecoregions. Of these three AOIs evaluated under scenario 3, ecoregions exhibited the highest R-squared value (0.69), which may be due to the fact that NIDRM models were developed by ecoregion (Krist and others 2007). The stronger model fits observed under scenario 3, compared to models for the other two mortality scenarios, are also consistent with our hypothesis that some of the FIA-observed mortality attributed to unknown/other may be actually due to insect and disease mortality. Therefore, we chose to focus our results and discussion below on scenario 3.

**Table 8.2—Linear regression parameters for each area of interest (AOI), metric, and mortality scenario**

| Area of interest              | Metric    | Scenario              | Slope              | <i>y</i> -intercept | R-squared |
|-------------------------------|-----------|-----------------------|--------------------|---------------------|-----------|
| <i>millions m<sup>2</sup></i> |           |                       |                    |                     |           |
| State                         | Live BA   | —                     | 0.94               | -56.01              | 0.97      |
|                               | Mortality | (1) I&D only          | <b><i>0.83</i></b> | 3.06                | 0.47      |
|                               | Mortality | (2) I&D & other       | 0.56               | 1.63                | 0.51      |
|                               | Mortality | (3) I&D and 1/2 other | <b><i>0.81</i></b> | 1.32                | 0.60      |
| County                        | Live BA   | —                     | 0.93               | -1.34               | 0.96      |
|                               | Mortality | (1) I&D only          | 0.78               | 0.06                | 0.19      |
|                               | Mortality | (2) I&D & other       | 0.71               | 0.01                | 0.57      |
|                               | Mortality | (3) I&D and 1/2 other | <b><i>0.96</i></b> | 0.00                | 0.51      |
| Ecoregion section             | Live BA   | —                     | 0.94               | -23.54              | 0.97      |
|                               | Mortality | (1) I&D only          | 1.12               | 0.68                | 0.56      |
|                               | Mortality | (2) I&D & other       | 0.82               | -0.24               | 0.66      |
|                               | Mortality | (3) I&D and 1/2 other | <b><i>1.06</i></b> | -0.11               | 0.69      |
| Ecoregion subsection          | Live BA   | —                     | 0.94               | -3.75               | 0.96      |
|                               | Mortality | (1) I&D only          | 1.15               | 0.13                | 0.50      |
|                               | Mortality | (2) I&D & other       | 0.84               | -0.06               | 0.58      |
|                               | Mortality | (3) I&D and 1/2 other | 1.10               | -0.05               | 0.62      |
| EMAP Hexagon                  | Live BA   | —                     | 0.94               | -0.49               | 0.86      |
|                               | Mortality | (1) I&D only          | 0.37               | 0.04                | 0.09      |
|                               | Mortality | (2) I&D & other       | 0.40               | 0.02                | 0.19      |
|                               | Mortality | (3) I&D and 1/2 other | 0.48               | 0.02                | 0.17      |

— = Not applicable.

Slopes not significantly different than 1 (at the 95-percent confidence level) are in bold italics.

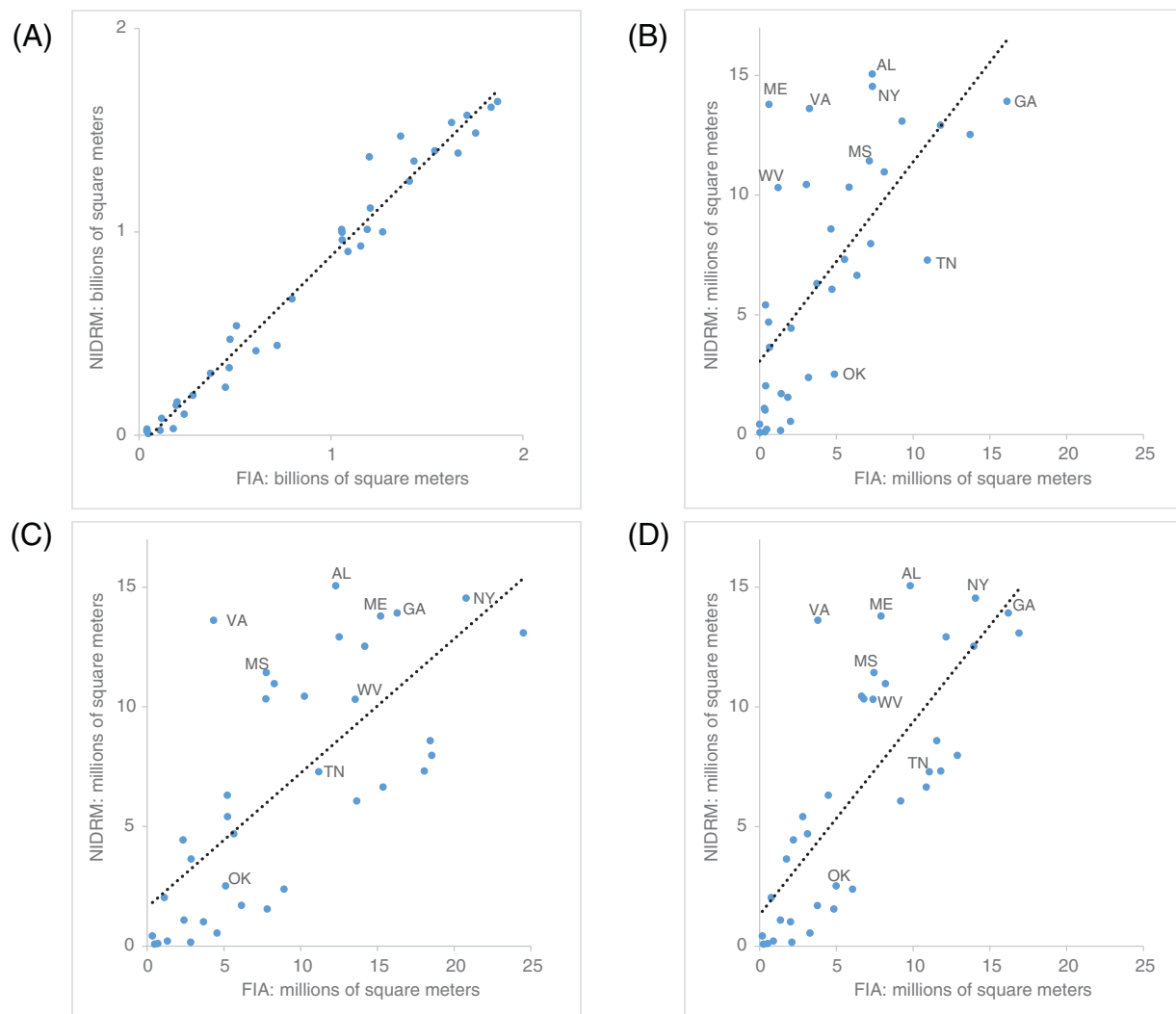


Figure 8.2—State-level National Insect and Disease Risk Map (NIDRM) (y-axis) versus Forest Inventory and Analysis (FIA) (x-axis) plots and linear regressions for (A) live basal area (BA) and (B-D) mortality BA scenario. The x-axis values for the three FIA mortality scenarios represent mortality from: (B) insect and disease only (scenario 1); (C) insect and disease and all of unknown/other (scenario 2); and (D) insect and disease and one-half of unknown/other (scenario 3). A few State points are labeled for reference. See table 8.2 for regression parameters.

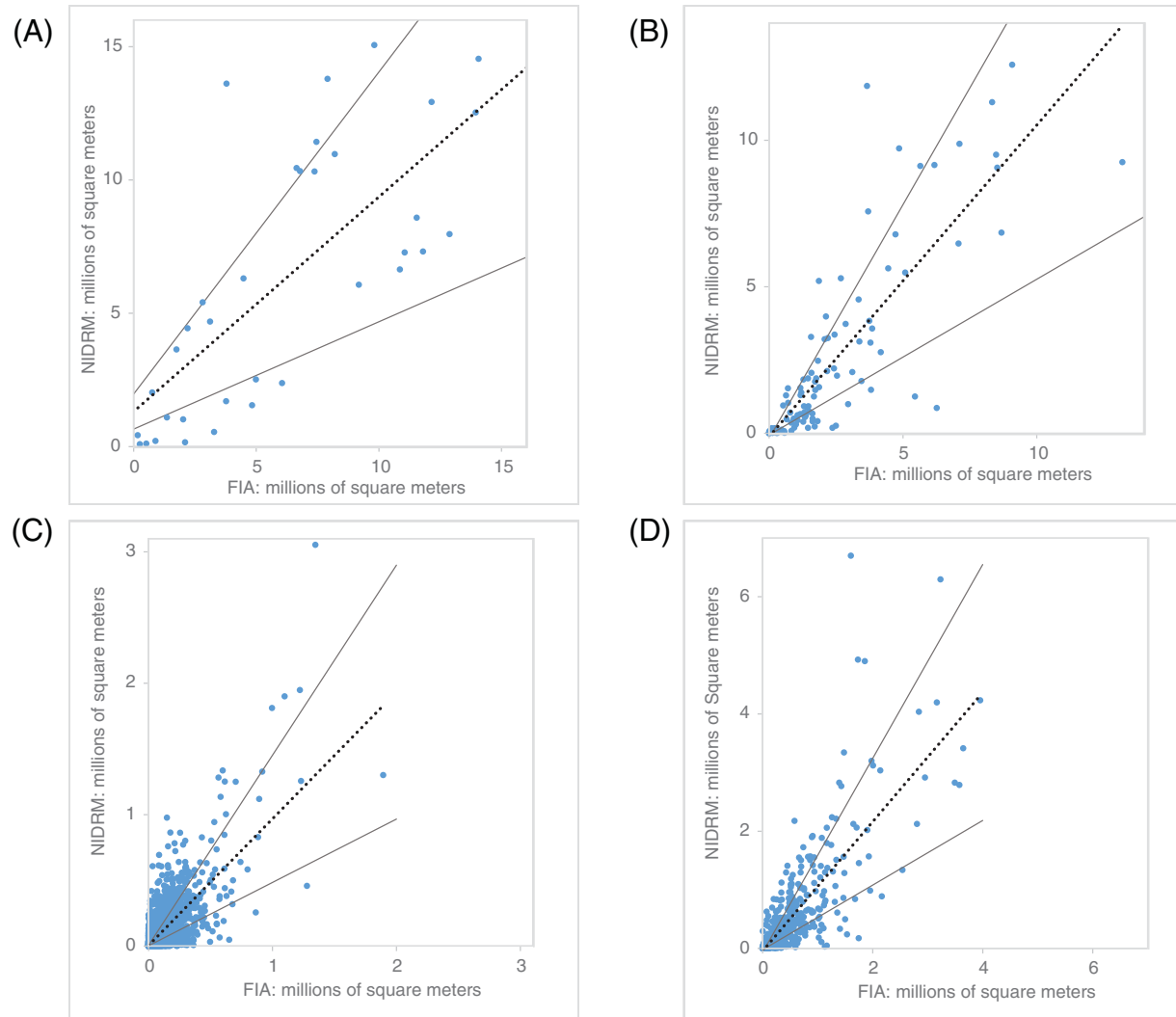


Figure 8.3—National Insect and Disease Risk Map (y-axis) versus Forest Inventory and Analysis scenario 3 (x-axis) plots for mortality basal area by (A) State, (B) section, (C) county, and (D) subsection. Dotted line is the linear regression fit. Solid lines bracket the + 50-percent (upper line) and -50-percent (lower line) residual zone. See table 8.2 for regression parameters.

Estimates of goodness of fit declined dramatically below the county level (table 8.2). At the EMAP hexagon scale, the relationship between NIDRM and FIA mortality degrades very significantly with an R-squared value less than 0.2 and a slope near 0.5, indicating that NIDRM more often under-estimates the amount of mortality relative to FIA. However, FIA does not generally report statistics below the county level because the density of plots provides less reliable estimates at finer scales (Scott and others 2005). Therefore, it is unknown if the degradation in slopes and R-squared values is due to limitations with NIDRM or FIA at these scales. Regardless of the cause, we exclude the EMAP hexagons from the results we report below, emphasizing instead results for the other four AOIs (States, counties, ecosections, and ecosubsections).

### Areas Where NIDRM Mortality Under-predicted FIA Estimates

NIDRM under-prediction (fig. 8.4) often occurred in areas with either low host density or few forest pests modeled in NIDRM. Examples of low host density leading to NIDRM under-prediction include the southern Midwest and the Mississippi Valley (fig. 8.5), areas dominated by agriculture. Areas with isolated or small patches of trees contain lower basal areas of

host tree species and, consequently, are more challenging to model for tree extent and density (Ellenwood and others 2015). Southern Florida and Louisiana are examples of areas with higher forest density but comparably few hosts modeled by NIDRM (fig. 8.5). The under-prediction in those areas may be explained by NIDRM's omission of tree species such as baldcypress (*Taxodium distichum*). In Louisiana, baldcypress is being impacted by repeated defoliation events from baldcypress leafroller (*Archips goyerana*) and forest tent caterpillar (*Malacosoma disstria*) in areas where the natural hydrology has been severely altered, leading to its ongoing degradation and mortality (Effler and Goyer 2006, Effler and others 2006). These were not modeled in NIDRM.

NIDRM under-prediction relative to FIA expectations may also have occurred because of non-modeled factors. In particular, dispersal of insects and pathogens, whether by anthropogenic factors such as the transport of nursery stock or firewood, or by natural factors such as prevailing winds, are challenging to model. For example, the 2012 NIDRM did not predict the 2013 arrival and establishment of emerald ash borer (EAB, *Agrilus planipennis*) in Boulder, CO, because this area was not included in the EAB model's extent.



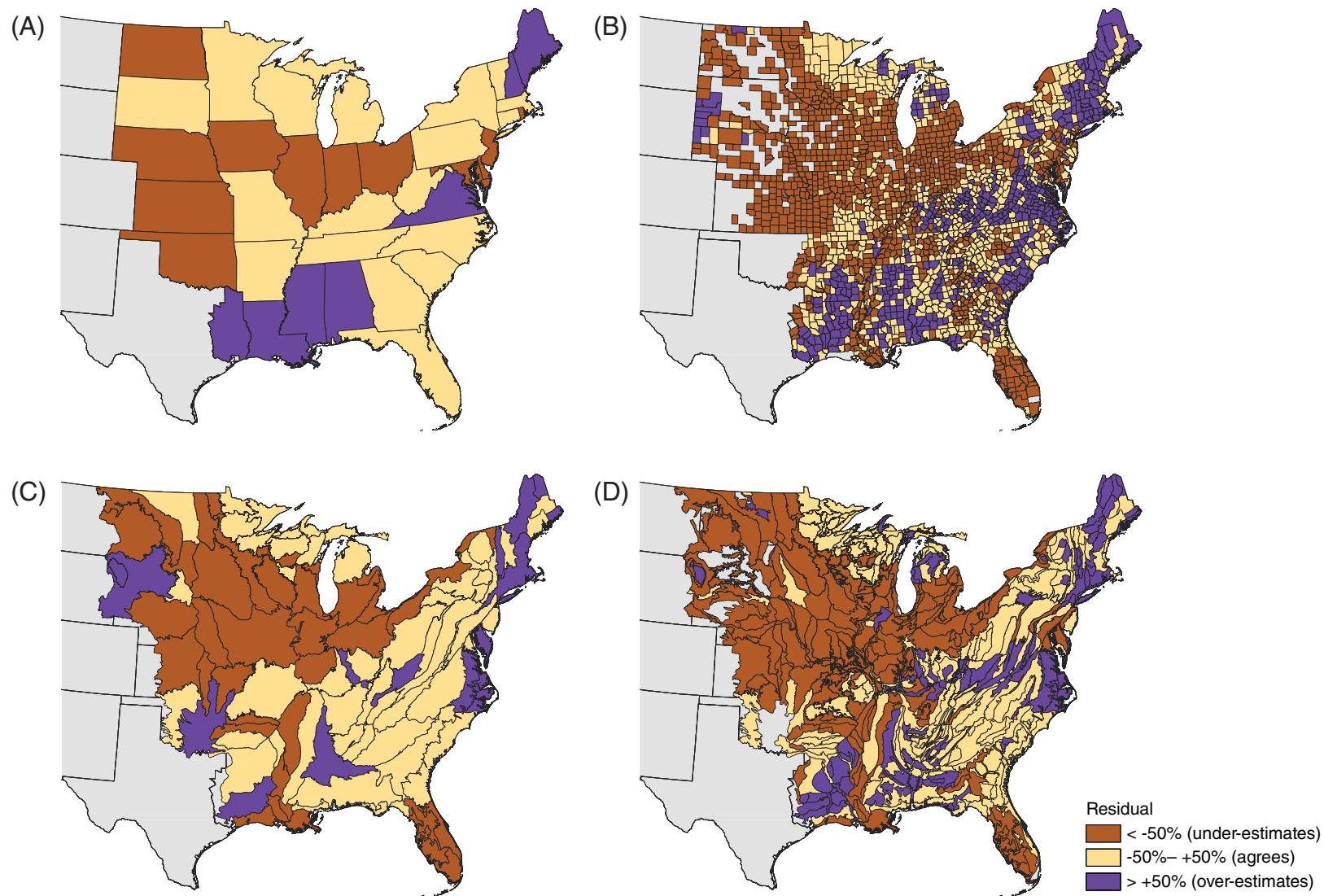


Figure 8.4— Residuals (as a percent) from the National Insect and Disease Risk Map (NIDRM)– Forest Inventory and Analysis (FIA) mortality linear regressions by (A) State, (B) county, (C) ecosection, and (D) subsection. Maps depict areas where mortality predicted by NIDRM approximately agrees, over-estimates, or under-estimates relative to FIA.

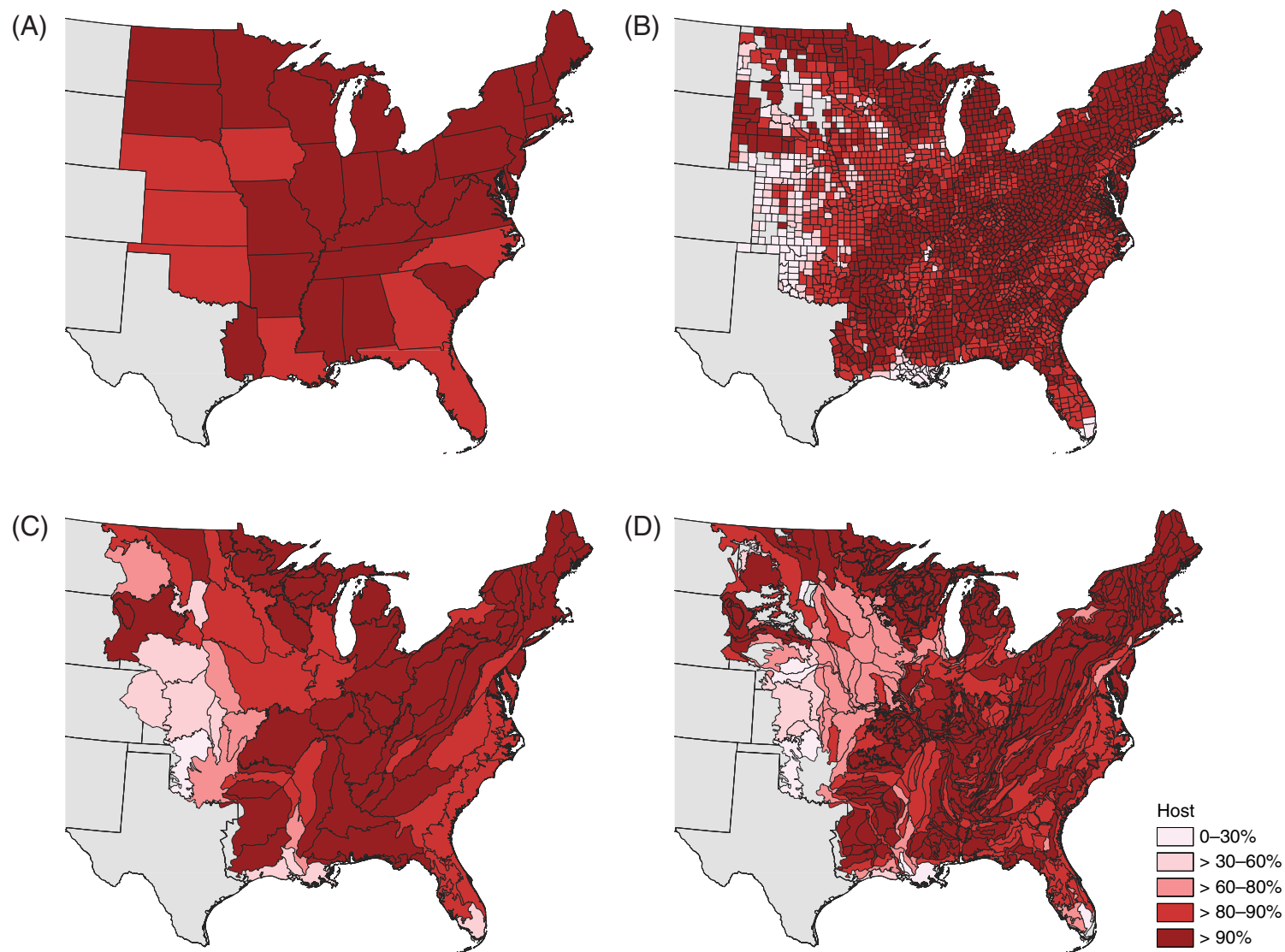


Figure 8.5—Percentage of total basal area that is modeled as potential host for insects or diseases in the National Insect and Disease Risk Map (NIDRM), by (A) State, (B) county, (C) ecosection, and (D) subsection. Analysis restricted to FIA forest land.

### Areas Where NIDRM Mortality Over-predicted FIA Estimates

NIDRM predicted mortality at rates greater than FIA estimates often in areas affected by destructive forest pests with periodic or stochastic outbreaks as part of their natural cycles. In the case of eastern spruce budworm (*Choristoneura fumiferana*), which could impact large areas of Maine, budworm populations can be greatly affected by factors that are difficult to simulate, such as diseases, local weather conditions, and predators (Kucera and Orr 1981). Conditions are still considered favorable in parts of New England for a spruce budworm outbreak. NIDRM over-predictions may also have been due to lags in the FIA inventory cycle, whereby FIA inventories had not yet detected recent or incipient mortality. An example of this is mountain pine beetle (*Dendroctonus ponderosae*) mortality in the Black Hills National Forest of South Dakota (fig. 8.4D), where mortality events are still ongoing. In eastern Texas and across other parts of the South (fig. 8.4B) where higher levels of mortality modeled in NIDRM are driven by southern pine beetle (*Dendroctonus frontalis*), frequent and widespread forest management and pest suppression activities have probably played a large role in reducing hazard. Therefore, mortality seen in FIA is low relative to the prediction of the retrospective NIDRM, which does not consider management.

In some areas of eastern North Carolina where laurel wilt—caused by a fungus (*Raffaelea lauricola*) spread by the exotic invasive redbay ambrosia beetle (*Xyleborus glabratus*)—is not

yet established, we would not expect FIA to detect significant mortality that the retrospective NIDRM estimated for this agent (figs. 8.4B, C, D). Note that the retrospective NIDRM used 2012 extents of the fungus as a model driver. Likewise, the retrospective EAB model used 2012 knowledge of EAB ranges. (It was not backwardly adjusted for the 2002–2016 time frame). Thus, we would expect NIDRM to show higher levels of tree mortality than FIA for EAB.

In areas where more pests are modeled in NIDRM, predicted mortality tends to be higher, and thus the chances of potential mortality being fully realized is lower. In other words, NIDRM estimates of potential hazard from individual agents could be correct, but not all mortality for all modeled pests may have occurred within a 15-year time frame because of pest competitive interactions, differing environmental requirements of agents, or the statistical likelihood of multiple agents acting simultaneously; these factors are omitted when the NIDRM composite result is derived from its component models. Consequently, NIDRM over-predictions often occur in areas where the number of modeled agents is higher. This could help explain the NIDRM over-predictions in areas of New England, the Adirondacks, and central Appalachians, areas that all tend to have high pest diversity and, consequently, a large number of NIDRM models (fig. 8.6).

### Future Directions

A key finding from our study is that NIDRM is reasonably accurate at predicting tree mortality due to insects and diseases at landscape scales.

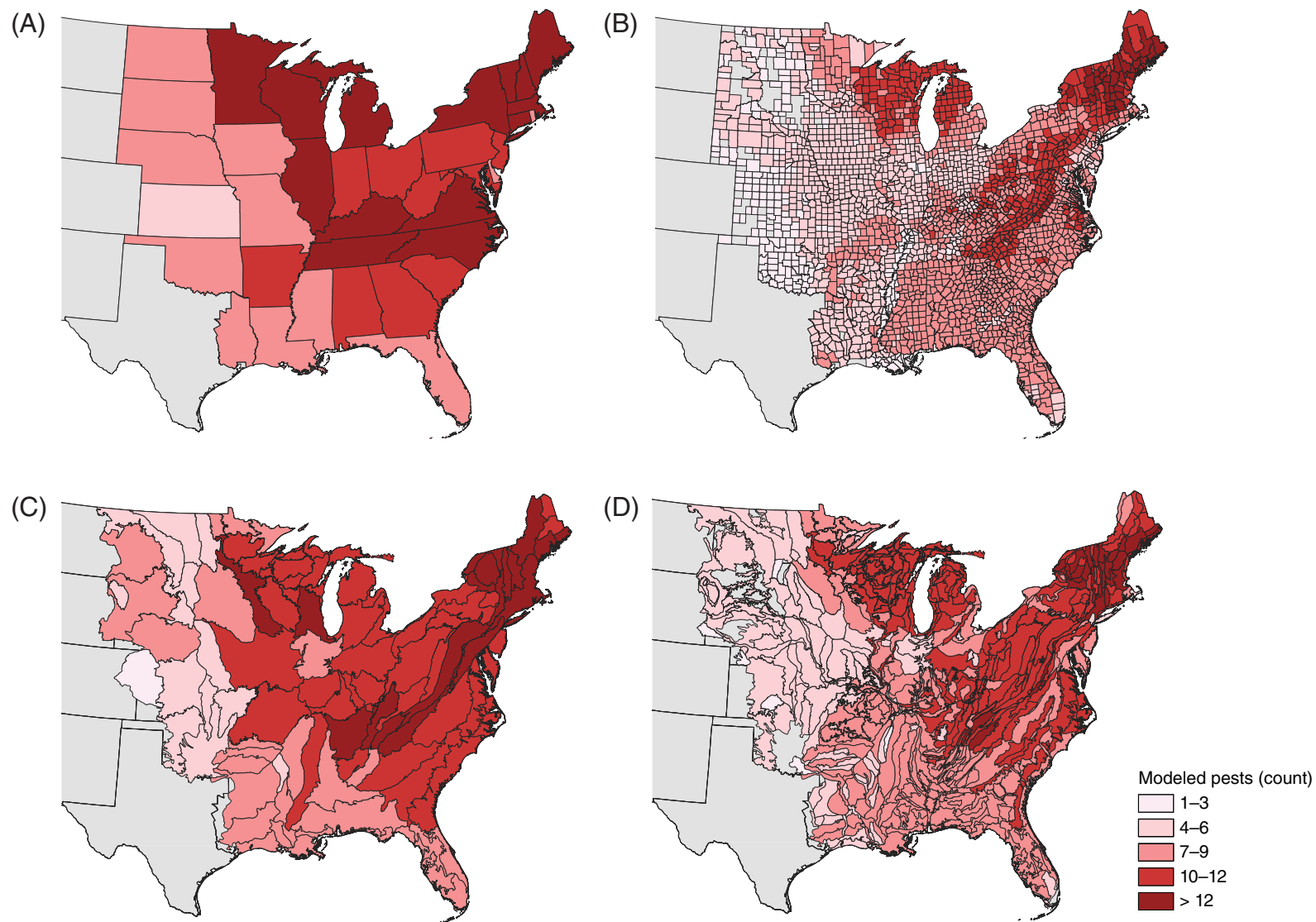


Figure 8.6—Number of forest pests modeled in the National Insect and Disease Risk Map (NIDRM) by (A) State, (B) county, (C) ecosection, and (D) subsection.

This finding validates the use of NIDRM in a variety of management and planning activities. For example, the NIDRM currently plays a key role in forest action plans (occurring at the State level): it has been used in a “forests on the edge” assessment of private lands (Stein and others 2010), in watershed assessments (Potyondy and Geier 2011), and to support the 2014 Farm Bill (H.R.2642–Agricultural Act of 2014). Nevertheless, there are opportunities to augment and improve our comparative analysis. First, it should be extended to include at least the entire conterminous United States, thus including major pest and host combinations that are missing from the present analysis. Second, especially in adding the Western United States to the analysis, the retrospective NIDRM models should be re-run while adjusting all dynamic variables (e.g., drought, which is an important driver of pest population dynamics in western forests, and in influencing the geographic ranges of exotics). Third, because we only considered a fraction of the AOIs commonly used in forest condition reporting, additional AOIs need to be factored into future comparisons (e.g., watersheds and sub-watersheds). Lastly, additional explanatory variables and datasets should be analyzed to better interpret why (and where) NIDRM may be over- or under-predicting actual mortality.

Our comparative analysis also offers insights into how future forest pest hazard models may be improved. One is to improve models in areas with low forest density. This may be achieved by informing the model developers

(i.e., entomologists and pathologists) about areas where NIDRM has historically under-predicted mortality, thus facilitating the “tuning” or re-parameterization of models for these locations. Related, additional pest and host combinations could be modeled, which should further reduce NIDRM’s tendency to under-predict in certain areas. For example, black cherry (*Prunus serotina*) decline in the Northeast has resulted in significant mortality, yet this was not evaluated in the 2012 NIDRM. Another idea for reducing under-prediction is to relax the forest pest dispersal constraints on NIDRM so that models encompass rare, long-distance dispersal events. This would come at the cost of increasing the area where NIDRM over-predicts mortality, but as a model of *potential* hazard it is preferable to have NIDRM err on the side of overestimation rather than underestimation to ensure places with potential for significant insect and disease activity are not overlooked.

Additional ideas for a future NIDRM build on the progression of past NIDRMs (fig. 8.1), as well as feedback from the field. We propose four non-mutually exclusive directions for developing a future NIDRM. The first direction is to integrate NIDRM with other datasets and models, principally national insect and disease surveys (IDS), to iteratively monitor hazard conditions and update NIDRM predictions over time. Relating the magnitude of NIDRM-projected hazard to observed mortality from IDS helps crosswalk our results to management actions that tie to the U.S. Forest Service Strategic Plan ([www.fs.fed.us/strategicplan](http://www.fs.fed.us/strategicplan)). For example,



areas of high NIDRM-projected hazard and low IDS-observed mortality may benefit especially from prevention tactics (e.g., thinning), while areas with high hazard and high mortality may require restoration, either to previous conditions or desired future conditions. Similarly, areas with low NIDRM hazard and low IDS mortality may simply necessitate periodic monitoring to ensure resiliency, while areas having low projected hazard and high current mortality may require suppression and a possible re-evaluation of specific NIDRM models (e.g., the recent California drought, where the 2013–2027 NIDRM has underestimated mortality that has been detected by IDS). In areas with high risk and high mortality, suppression efforts could be overwhelmed by the magnitude of outbreaks and may thus only be suitable for post-hoc restoration efforts. Regions with potential for high risk activity but little current mortality may require frequent surveillance and targeted suppression efforts to prevent activity from rapidly expanding.

The second direction is to simply update the existing NIDRM to factor in new knowledge, including biological knowledge about pests and hosts from the emerging literature (e.g., Morin and Liebhold 2015, 2016). For example, host conditions for mountain pine beetle have changed dramatically since the last NIDRM. Intervening tree mortality could be quantified using IDS and other data sources to revise estimates of current host conditions, which serve as the basis for NIDRM's hazard predictions. Similarly, the geographic ranges of exotics and

certain native pests (e.g., southern pine beetle) have changed and could be updated to reflect the new hazard extent.

Lastly, the third and fourth directions involve modifying the spatial or temporal scales of NIDRM. While NIDRM is presently used to support landscape-level planning, looking out to 2013–2027, local insect and disease risk maps (LIDRMs) may be developed to support management decisions at local scales. New LIDRMs could be extended to include stand-level and other local finer resolution data. Similarly, at the landscape level, new NIDRMs could explicitly integrate future climate input variables, thereby facilitating projections of potential tree mortality and growth further into the future, in support of long-range planning. The forecasts could help to identify areas on the landscape where we might expect loss or extirpation, high stability and resiliency, and new areas for growth or colonization of tree species that serve as major hosts for pests. In turn, the forecasts could suggest a range of passive to active management options pertaining to monitoring (e.g., monitoring on the margins), treatments (e.g., for western bark beetles), and *ex situ* conservation (e.g., genetic banking).

In conclusion, NIDRM is a collaborative assessment, and we offer these ideas to engage partners and collaborators in the development of the next NIDRM. Such collaboration is important not only for ensuring use and application of NIDRM results, but also in helping overcome a number of conceptual and implementation challenges that must be addressed in pursuit



of the next NIDRM. In the past, one of these challenges has been to address outstanding questions as to whether NIDRM is accurately predicting mortality and, if so, at what spatial scales. We demonstrate that NIDRM reasonably predicts observed mortality over broad areas (in the Eastern United States) and further that predictions are informative across a range of scales (e.g., down to individual counties). These findings bolster the value of NIDRM and confirm that it is a useful tool in forest health management.

## ACKNOWLEDGMENTS

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

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

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## SECTION 3.

### Evaluation Monitoring Project Summaries



## INTRODUCTION

Southern pine beetle (SPB; *Dendroctonus frontalis*) is the most destructive forest insect in the Southeastern United States, with greatest damage occurring in high-density pine stands with basal area above approximately  $18 \text{ m}^2 \text{ ha}^{-1}$  (Aoki and others 2016, Ayres and others 2011, Nowak and others 2015). As SPB migrates northward, it is impacting pine-dominated forests throughout much of the Mid-Atlantic region. In the New Jersey Pinelands National Reserve (PNR), Forest Inventory and Analysis (FIA) data indicate that approximately 65 percent of the stands dominated by pitch pine (*Pinus rigida*) and shortleaf pine (*P. echinata*) are at or above the density at which SPB can cause significant pine mortality (Crocker 2014, 2015). Since 2001, SPB damage has been detected in southern New Jersey, with over 500 SPB locations identified and ground-truthed on public lands alone (New Jersey Agricultural Experiment Station 2017, Weed and others 2013). Extensive mortality in pitch and shortleaf pine-dominated stands of the PNR could dramatically alter forest structure, future species composition, and wildfire risk in impacted stands. Although FIA field sampling efforts have been accelerated in New Jersey and enhanced FIA forest census protocols are appropriate for quantifying changes to understory vegetation, fine fuels, and coarse woody debris on the forest

floor, we currently have little information on the regeneration phase following extensive overstory mortality driven by SPB in naturally regenerated pitch and shortleaf pine stands. However, this phase has significant implications for wildfire activity in the PNR, especially if it resembles early successional pine- and scrub oak-dominated forests of the early 1900s, which were characterized by extremely large, destructive wildfires (Clark and others 2014a, 2014b). Any potential impacts on wildfire occurrence and intensity are especially important on the Mid-Atlantic Coastal Plain, where high human population densities, high-value property, and air quality concerns in the adjacent large urban areas are significant issues.

In this project, we addressed how SPB damage and predominant management options alter forest structure, species composition, and canopy fuel loading using FIA forest census protocols in pairs of control and impacted stands. We focused on three major treatments employed by the New Jersey Department of Environmental Protection (NJ DEP) and contractors: (1) “natural” or no treatment; (2) “cut and leave,” where infested pines and a buffer are cut and stems are left in place; and (3) “cut and chip,” where infested pines and buffers are cut, bunched, chipped, and occasionally hauled offsite. We then used forest census data to inform a canopy energy balance and hydrology model to evaluate potential

## CHAPTER 9.

### Impact of Southern Pine Beetle on Forest Structure and Fuel Loading in a Wildfire-prone Landscape (Project NE-EM-F-13-01)

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changes to hazardous fuel loading and wildfire risk in this fire-adapted ecosystem. Specific questions addressed in this project were:

- How are stand structure and composition altered by SPB-driven mortality of overstory pines? Are treated (e.g., cut and leave, cut and chip) plots different from natural plots in terms of retained trees and saplings?
- How will mortality (and removal) of overstory pines potentially affect the long-term forest structure and composition of stands impacted by SPB?
- To what extent are canopy and understory fuels altered by SPB? Using a multi-level energy exchange and hydrologic model informed using comparative forest census data, how do changes in overstory canopy cover and altered fuel loads affect fuel moisture dynamics compared to intact stands?

## METHODS

We used aerial and ground-based surveys conducted by NJ DEP and Dartmouth College researchers to locate impacted stands. We then installed paired field census plots to quantify trees, saplings, seedlings, and understory composition and structure in control and impacted areas using U.S. Department of Agriculture (USDA) Forest Service FIA forest census protocols ([www.fia.gov](http://www.fia.gov)) during 2014 and 2015. All stands were located on public lands (primarily State forests and wildlife management areas) and encompassed the range of treatment strategies: natural ( $n = 12$ ), cut and leave ( $n = 27$ ), and cut and chip ( $n = 12$ ) treatments.

Impacted areas in sampled stands ranged from approximately 0.5 to 35.0 ha. All live and dead trees and saplings were censused for species, diameter at breast height (d.b.h.), height, and crown condition: canopy cover was estimated for each 168-m<sup>2</sup> subplot. Understory height, species composition, and cover by species (including tree seedlings) were recorded for each subplot. Pine seedlings were tallied by height class in each subplot when present. We used recently developed allometric equations for pines based on destructive harvests and LiDAR to estimate total aboveground biomass and biomass of available fuel for crown fires, defined as all live and dead needles and live and dead twigs < 0.635 cm diameter, for pine trees and saplings in each subplot (Clark and others 2013).

We collected understory meteorological data in defoliated ( $n = 3$  to 5 towers) and control stands, in addition to meteorological data collected from seven above-canopy towers throughout the Pinelands in conjunction with New Jersey State Climatologist's Office (Clark and others 2012; <http://climate.rutgers.edu/usfs/monitoring.php>). Information on the immediate and longer term effects of SPB on complex fuel beds was integrated with a multi-level model of fuel moisture dynamics to assess fire danger and wildfire risk.

## RESULTS

SPB damage in infested portions of all stands resulted in significant mortality of pitch, shortleaf, and Virginia pine (*P. virginiana*) trees, averaging 95 percent and 98 percent of pine



tree basal area and aboveground biomass, respectively (fig. 9.1; table 9.1). Pine sapling basal area and aboveground biomass were similar in control and natural stands, but much lower in treated stands (fig. 9.2). In contrast to their response following wildfires, resprouting from epichormic meristems or root crowns of impacted pine trees or saplings was rarely observed following SPB attack. Pine seedlings were abundant only in cut and chip stands, where extensive disturbance of the forest floor had occurred because of vehicle and equipment use, exposing bare sand. Primarily oaks in the upland stands and other hardwood trees and saplings in lowland stands were present to abundant in natural plots, and many were retained in treated plots, with basal areas as high as in non-impacted stands (figs. 9.1 and 9.2).

Total canopy cover was reduced in natural and treated stands compared to control stands, with canopy reduction occurring in the order natural stands < cut and leave stands < cut and chip stands (table 9.1). In contrast, understory cover and height remained nearly unchanged in natural and cut and leave stands, and there was a trend towards increased understory cover in cut and chip stands sampled 2 to 4 years following treatment (table 9.1). Canopy fuels

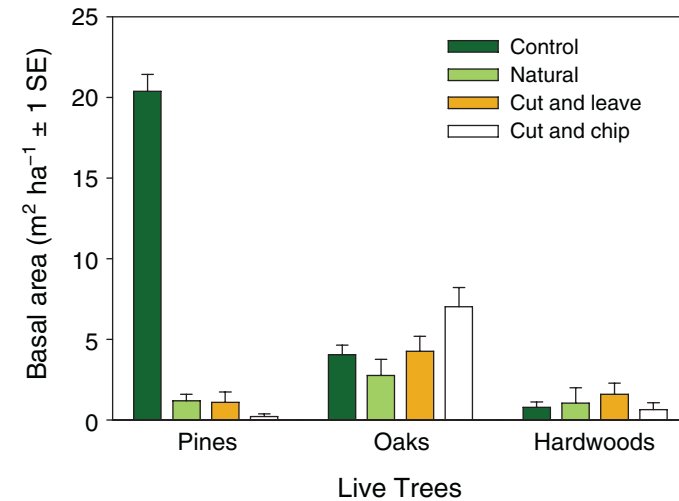


Figure 9.1—Basal area of live trees separated into pines (*Pinus rigida*, *P. echinata*, *P. virginiana*), oaks (*Quercus alba*, *Q. prinus*, *Q. marlandica*, *Q. velutina*, *Q. stellata*, *Q. falcata*, *Q. bicolor*), and other hardwoods (*Acer rubra*, *Nyssa sylvatica*, *Carya glabra*, *Magnolia virginiana*, *Sassafras albidum*) in control, natural, and treated plots.

**Table 9.1—Structural characteristics of canopy and understory in control, natural infested, and treated plots**

| Characteristic                                    | Control                   | Natural                  | Cut and leave            | Cut and chip             | Significance        |
|---------------------------------------------------|---------------------------|--------------------------|--------------------------|--------------------------|---------------------|
| Canopy                                            |                           |                          |                          |                          |                     |
| Height (m)                                        | 15.2 ± 0.3 <sup>a</sup>   | 10.1 ± 1.0 <sup>b</sup>  | 10.4 ± 1.0 <sup>b</sup>  | 12.0 ± 1.5 <sup>b</sup>  | F = 12.1, P < 0.01  |
| Cover (%)                                         | 61.9 ± 2.4 <sup>a</sup>   | 30.8 ± 6.0 <sup>b</sup>  | 20.2 ± 4.8 <sup>b</sup>  | 16.6 ± 6.1 <sup>b</sup>  | F = 29.5, P < 0.01  |
| Understory                                        |                           |                          |                          |                          |                     |
| Height (m)                                        | 0.7 ± 0.1                 | 0.6 ± 0.1                | 0.6 ± 0.1                | 0.5 ± 0.1                | F = 0.9, P = NS     |
| Cover (%)                                         | 71.6 ± 4.6                | 71.3 ± 8.3               | 72.0 ± 5.5               | 88.7 ± 1.0               | F = 1.7, P = NS     |
| Aboveground pine biomass (tons ha <sup>-1</sup> ) |                           |                          |                          |                          |                     |
| Trees                                             | 74.18 ± 4.18 <sup>a</sup> | 2.59 ± 0.88 <sup>b</sup> | 0.33 ± 0.21 <sup>b</sup> | 0.71 ± 0.40 <sup>b</sup> | F = 57.1, P < 0.01  |
| Saplings                                          | 3.96 ± 0.84 <sup>a</sup>  | 3.93 ± 1.27 <sup>a</sup> | 0.08 ± 0.06 <sup>b</sup> | 0.02 ± 0.01 <sup>b</sup> | F = 3.0, P < 0.05   |
| Available fuels (tons ha <sup>-1</sup> )          |                           |                          |                          |                          |                     |
| Canopy                                            | 8.04 ± 0.47 <sup>a</sup>  | 0.28 ± 0.09 <sup>b</sup> | 0.04 ± 0.04 <sup>b</sup> | 0.08 ± 0.04 <sup>b</sup> | F = 53.0, P < 0.001 |
| Sub-canopy                                        | 0.71 ± 0.15 <sup>a</sup>  | 0.80 ± 0.24 <sup>a</sup> | 0.01 ± 0.01 <sup>b</sup> | 0.01 ± 0.01 <sup>b</sup> | F = 3.1, P < 0.05   |

Available fuels were defined as live and dead pine needles, and live and dead stems with a diameter of up to 0.635 cm. Estimates were based on destructive sampling and LiDAR data described in Clark and others (2013).

Data are shown for control stands averaged together ( $n = 51$ ), and natural ( $n = 12$ ), cut and leave ( $n = 27$ ), and cut and chip ( $n = 12$ ) stands. Values are means ± 1 SE. Significance levels were tested using ANOVAs and Tukey's MSD tests. Values indicated with different superscripts among plot types are significantly different.

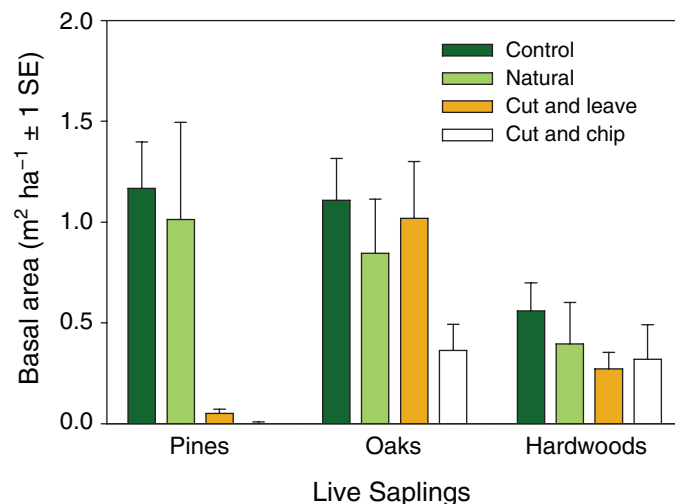


Figure 9.2—Basal area of live saplings separated into pines (*Pinus rigida*, *P. echinata*, *P. virginiana*), oaks (*Quercus alba*, *Q. prinus*, *Q. marlandica*, *Q. velutina*, *Q. stellata*, *Q. falcata*, *Q. bicolor*), and other hardwoods (*Acer rubra*, *Nyssa sylvatica*, *Carya glabra*, *Magnolia virginiana*, *Sassafras albidum*) in control, natural, and treated plots.

were reduced rapidly in impacted stands because the “red needle” phase is short in pitch and shortleaf pines. Available fuels in the canopy of control plots totaled  $8.8 \pm 0.5$  tons  $\text{ha}^{-1}$  (mean  $\pm 1$  standard error), while they were only  $1.1 \pm 0.3$ ,  $0.1 \pm 0.1$ , and  $0.1 \pm 0.1$  tons  $\text{ha}^{-1}$  in plots in natural, cut and leave, and cut and chip treatments, respectively (table 9.1). Impacted stands initially had greater needle mass, and 1-hour + 10-hour woody fuels on the forest floor compared to control stands, estimated to range from  $5.5 \pm 0.8$  to  $8.6 \pm 0.7$  tons  $\text{ha}^{-1}$ . In the years following pine mortality, reduced litterfall inputs

and litter decomposition resulted in reduced forest floor mass compared to undisturbed control plots. Coarse wood mass totaled up to  $105 \pm 12$  tons  $\text{ha}^{-1}$  in cut and leave stands, was highly variable in natural stands due to a large proportion of standing dead trees in some stands, and was minimal in some of the cut and chip treatments.

Meteorological conditions in larger cleared areas driven by insect damage were more similar to above-canopy conditions for wind speed and relative humidity, while mid-day air temperatures near the forest floor were greater than those above canopy or within unimpacted stands with greater canopy cover (table 9.1). Fuel moisture model simulations indicated that larger, treated stands experienced greater amplitudes in moisture dynamics of fine and woody fuels on the forest floor compared to unimpacted stands with intact canopies.

## DISCUSSION

The extensive mortality of pines in infested portions of stands with an average tree and sapling basal area of  $21.5 \text{ m}^{-2} \text{ ha}^{-1}$  reported here is consistent with SPB impacts in southern pine-dominated forests (Nowak and others 2015, 2016). Many of the stands that we sampled in the PNR have regenerated naturally following the cessation of intensive forest management and intense wildfires in the Pinelands, and pines averaged  $77 \pm 24$  years old (Aoki and others 2017). However, our field observations indicate that the impacts of SPB differ significantly from wildfire or wind damage because they cause

mortality and not solely damage to crowns or aboveground portions of stems of mature pitch and shortleaf pines. Contrary to our initial hypothesis, we encountered very few pine seedlings in natural or cut and leave plots. Pine regeneration from seedlings is occurring only in areas where the litter layer and soil has been disturbed significantly, such as in cut and chip stands and in areas associated with the use of vehicles in treated plots, or in areas that had experienced previous disturbance, such as following a summer wildfire in 2010 in Bass River State Forest. Advance regeneration of oak and hardwood saplings in natural stands, and retained trees and saplings in treated stands suggest that many of these stands will become hardwood-dominated in the near future. In the absence of wildfire, oaks in upland stands and other hardwoods in lowland stands will eventually replace pitch and shortleaf pines (La Puma and others 2013). Thus, SPB is accelerating successional sequences in the Pineland, and without forest management intervention, impacted areas will likely not return to pine-dominated stands.

In contrast to the effects of mountain pine beetle (*Dendroctonus ponderosae*) in western forests, needle abscission occurs quickly following mortality in pitch and shortleaf pines, thus the “red needle” phase is relatively short-lived (Jenkins and others 2014, Page and others 2015), reducing the potential for crowning fire behavior in impacted stands. We expected that increased fuel loading on the forest floor would likely significantly alter fuel beds, and that the more open canopies would

alter the meteorological conditions on the forest floor, contributing to higher fire danger in impacted stands. Fuel loads on the forest floor were initially high, with an increase of up to 170 percent from pine needles and twigs, but then reduced litterfall inputs and high rates of decomposition reduced fuel loads on the forest floor. High decomposition rates reflected the relatively high nitrogen and phosphorus content of pine needles when they fell following tree mortality, and may also be related to higher temperatures on the forest floor due to reduced canopy cover. The analogous “grey phase” has been characterized by lower wildfire danger in western forests, but does allow for greater penetration of wind and incident radiation, potentially driving greater fuel moisture dynamics in impacted stands (Jenkins and others 2014, Page and others 2015). Coarse wood on the forest floor was much greater in cut and leave and to a lesser extent natural stands in the Pinelands, but these fuels are usually not consumed during the relatively fast-moving spring wildfires that typically occur in the Pinelands. Our hydrologic simulations indicated an initial increase in fuel moisture dynamics in impacted stands, but when canopy closure occurs these will be reduced. In the long term, decreased available fuels in the canopy and a shift from pine to hardwood litter on the forest floor will very likely reduce wildfire risk in stands impacted by SPB.

To date, management of SPB in the Pinelands has been highly reactive; extensive outbreaks were detected primarily by aerial surveys conducted by NJ DEP, ground-truthed, and

then either not treated, infested trees and buffers were cut and left, or infested trees were cut, bunched, and then chipped. Proactive management should be considered in the future because both thinning of dense stands and higher intensity prescribed fires that reduce basal area and increase tree spacing have been effective in limiting SPB outbreaks in other pine-dominated forests (Nowak and others 2015). To address the long-term impacts of SPB outbreaks and management options, we are now coupling a forest dynamics model (LANDIS II) to predict changes to forest structure and composition in the PNR and surrounding areas.

## CONCLUSIONS

Southern Pine Beetle outbreaks are causing > 90-percent mortality of mature pine trees in infested stands in southern New Jersey. While pine saplings are present in natural stands, regeneration is very low in cut and leave treatments, and pine seedlings were encountered only in the most disturbed treatment (cut and chip), where the forest floor and shrub layer were disturbed. Oaks (upland stands) and other hardwoods (lowland stands) were present to abundant in all natural and treated stands, indicating that impacted stands would become oak- or hardwood-dominated in the future. Canopy fuels were reduced rapidly in impacted stands because needles abscised soon after mortality and the “red needle” phase was short. Fuel loading on the forest floor was initially enhanced in the order cut and leave > natural > cut and chip, but then reduced litterfall

and litter decomposition following overstory mortality resulted in lower fuel loading on the forest floor with time since SPB outbreaks.

## CONTACT INFORMATION

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## INTRODUCTION

Southern pine beetle (SPB; *Dendroctonus frontalis* Zimmermann) is a native pine bark beetle whose range extends from Central America to the Mid-Atlantic States, and from southern Arizona to the southeastern seaboard. SPB populations are very high during their periodic outbreaks, but low to undetectable in intervening years. Outbreaks occurred regularly throughout the South during the 20th century, causing significant economic damage. A single outbreak in the Eastern United States, from 1999–2002 resulted in over 1 billion dollars in lost timber (Clarke and Nowak 2009). At the turn of the 21st century, an outbreak began in southern New Jersey. Although included in the northernmost extent of the beetle's range map, southern New Jersey had not experienced a significant outbreak since the 1930s. The 2000s outbreak spread northward across the State, causing an estimated 14,000 acres of damage in 2010 alone. In the fall of 2014, SPB was detected for the first time on Long Island, NY (Schlossberg 2014), and since then small numbers have been trapped in Connecticut, Rhode Island, and Massachusetts. With this range expansion, SPB moves from its usual hosts—loblolly and shortleaf pine (*Pinus taeda* and *Pinus echinata*, respectively)—to pitch pine (*Pinus rigida*). Although pitch pine has been noted as one of a number of host species, little previous research has studied SPB in pitch pine systems. Studies in the Southeastern United States have demonstrated that forest stand characteristics and tree defenses exert important controls on SPB (Gara and Coster 1968, Reeve

and others 1995, Showalter and Turchin 1993), and knowledge of whether these controls are also important in this naïve ecosystem will help inform predictions of risk and management tactics for suppression and prevention in New Jersey and elsewhere in the Mid-Atlantic and New England States.

Unlike some bark beetle species, SPB attacks living pines, forming discrete infestations, known as “spots,” within an outbreak area. Pine trees have evolved oleoresin defenses against such herbivores, so the initial attack phase in a new spot usually begins with a weakened tree, such as one struck by lightning, or impacted by other factors such as storms, disease, or other insects (Coulson and others 1986, Gara and others 1965). If a local background population of SPB exists in the area, pioneer beetles will be drawn to the tree; these in turn will begin producing frontalin, an aggregation pheromone that draws in many more beetles (Borden 1974). The large number of beetles overcome the defenses of the tree, enabling thousands of beetles to lay their eggs and successfully reproduce within the phloem. If the initial tree draws in enough beetles, these larger numbers are then able to move on to attack adjacent, healthy trees and the infestation grows. Stand characteristics that optimize these behaviors may increase the risk of infestation. In addition, at least some of the variation in resin defenses among stands may be due to how productive trees are at a given site, an effect described in the Growth Differentiation Balance Hypothesis (GDBH) (Herms and Mattson 1992). Since primary and secondary metabolism are both

## CHAPTER 10.

### Assessing Southern Pine Beetle Risk in the New Jersey Pinelands (Project NE-EM-B-12-01)

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carbon-based processes, the hypothesis posits that trees with high resources for growth will have lower resources for defenses, whereas stressed trees with fewer resources will allocate carbon to secondary metabolism, leading to higher resin flows. Stressed trees may thus be better defended, while healthier, more productive trees may be less well-defended. Previous research has shown that greater resources can reduce resin defenses in loblolly pine, the most common host of SPB in the Southern United States, and that SPB attack success declines with increased resin flow (Reeve and others 1995).

We sought to investigate the attributes of pitch pine defenses and stand structure relative to the current SPB outbreak in New Jersey. Our objectives were (1) to determine the scope of SPB infestation relative to forest type; (2) to characterize the defenses of pitch pine against SPB; (3) to determine the range of variation in stand characteristics across the Pinelands; (4) to determine the relationship between pitch pine stand characteristics, resin flow, and mortality; and (5) to use these data to develop an SPB risk profile for the New Jersey Pinelands.

## METHODS

### Study Site

The New Jersey Pinelands, spanning forests in the south and central portions of the State, represent a unique State-Federal partnership designed to preserve and protect over 1.1 million acres of land. Close to half of the Pinelands area is in public ownership (New Jersey Pinelands

Commission 2015). Because SPB management efforts fall almost exclusively on the State, we restricted our data collection and analysis to public lands within the Pinelands and the immediately surrounding area.

### Forest Type

To determine whether some forest types might be more susceptible to SPB than others, we combined GIS forest cover type data with spot occurrence data. The Land Use Land Cover (LULC) dataset produced by the New Jersey Department of Environmental Protection (NJ DEP) uses color infrared photography to classify land cover types across the State (New Jersey Department of Environmental Protection 2010a). Classifications are based on a modified Anderson system developed by NJ DEP (New Jersey Department of Environmental Protection 2010b). We selected all forest types including a conifer component. The overwhelming majority of pine in our study area is pitch pine. To simplify analyses to categories that would be meaningful for management, some cover classifications were merged. The final types were: upland conifer (>75 percent pine), wetland conifer, upland mixed (between 25 and 75 percent conifer), and wetland mixed. We used these data to determine the relative proportion of each forest type across the landscape, and then to determine whether forest types differed in susceptibility to SPB spots. Spot data were produced by NJ DEP and comprise ground-truthed infestations, initially identified in aerial detection surveys. Based on the area covered by each forest type, we conducted a

chi-square analysis to determine whether any one forest type exhibited a higher or lower than expected occurrence of spots. Because the occurrence of spots was largely concentrated south of the Mullica River, and because NJ DEP SPB suppression and monitoring efforts are largely restricted to public land, we restricted the spot analysis to State-owned land in the counties south of the Mullica affected by SPB, including State lands outside the Federal Pinelands boundary.

### Forest Structure

Using the NJ DEP LULC data described above, we used ArcGIS to select plots at random across the four forest types described. Consideration was made for efficiency of accessing plots by foot from established roads. We established 12 control plots each in the northern and southern regions. In each plot, we sampled three 50-m transects with sampling points every 10 m, sampling the nearest tree >15 cm to each sampling point for age, diameter at breast height (d.b.h.), height, basal area of pine, basal area of hardwoods, distance to next nearest pine >15 cm, and percent canopy cover at that point. Hemispherical photography was used, in conjunction with the program ImageJ (Schneider and others 2012), to estimate percent canopy cover. We implemented the same sampling protocol at 24 “spot plots,” or points of SPB infestation, identified by NJ DEP ground-truth surveys. Many of the spots had been previously cut for suppression purposes, so in these cases, we identified immediately

adjacent stands that appeared to have similar forest structure features to the stands that had been cut.

Due to high correlation among variables, principal components analysis (PCA) was used to reduce the dimensionality of the data. The analysis was performed on plot-level means of each variable. To determine the relationship between the principal components and the probability of a stand being infested or not, logistic regression was applied to each of the first two principal components. Analysis of deviance comparing each model to its null was used to obtain a *p*-value. Because the vast majority of the infestations documented by NJ DEP were in the southern region, the northern control plots were not used in the initial PCA or logistic regression analysis. A second PCA was then performed using both northern and southern control plots along with the infested plots, in order to assess the relationship of each region’s plots to the probability of infestation.

### Tree Defenses

To test the GDBH, we selected soil type as a proxy for variation in access to water resources. Using the USDA Natural Resources Conservation Service’s Soil Survey Geographic Database (SSURGO), we selected four study sites within each of the three soil types occurring in highest proportion under coniferous and mixed conifer forests (Soil Survey Staff 2012). We divided the area north and south of the Mullica River into separate regions, for a total of 6 soil types,

and 24 sites all together. The three soil types in the northern region were: Atsion sand; Lakehurst sand; and Lakewood sand. The soil types in the southern region were: Berryland and Mullica soils; Hammonton sandy loam; and Aura sandy loam. Sites were chosen to facilitate accessibility. At each site, seven trees were selected at random. A 2.54-cm punch was used to remove the outer bark and phloem without damaging the xylem, and resin flow from the resulting wound was captured over the next 24 hours and weighed. Each site was sampled four times, in late August 2012, early and mid-July 2013, and September 2013. To specifically test the upland versus the wetland sites, we conducted a separate analysis comprising nine sites each in upland versus wetland locations. Trees were sampled as described above, and sampling occurred in August 2014. Data in both studies were analyzed using a random effects mixed model.

## RESULTS

### Forest Type

Forests north of the Mullica River are overwhelmingly comprised of the upland conifer type, totaling nearly 45 000 ha. Forests south of the Mullica were dominated by the mixed conifer type, both in upland and wetland areas, which together covered over 65 percent of the forested landscape. The pure conifer type, in both upland and wetland areas, covered approximately 33 percent of the landscape. Forty percent of spots were found in the upland conifer type, though this type

covered less than 25 percent of the landscape. A further 27 percent of spots were found in the wetland conifer type, which comprised just over 10 percent of the area. Spots in the upland mixed type represented just over 20 percent of the total, and spots in the wetland mixed type covered approximately 12 percent of the total (fig. 10.1A).

The chi-square analysis results showed that we can reject the hypothesis that the occurrence of spots is proportional to the area of each forest type (chi-square = 121.47,  $df = 3$ ,  $p < 2.2e-16$ ). The log odds for both pure conifer forest types were positive (more spots than expected), and the log odds for both mixed forest types were negative (fewer spots than expected, fig. 10.1B).

### Forest Structure

Seventy-five percent of the variation between all infested and uninfested stands was explained by the first two principal components. Variables clustered on the positive side of component one included higher hardwood basal area, taller height, older age, larger d.b.h., and lower density (farther neighbor distances). High pine basal area and high percent pine were clustered on the negative end of the first component. In the second component, high total basal area and high percent live crown opposed one another across the axis. Logistic regression of status (infested vs. not infested) on the first component showed an increased probability of infestation on the negative end of component one, and on the positive end of component two (PC1  $p = .043$ , PC2  $p = .018$ , fig. 10.2).

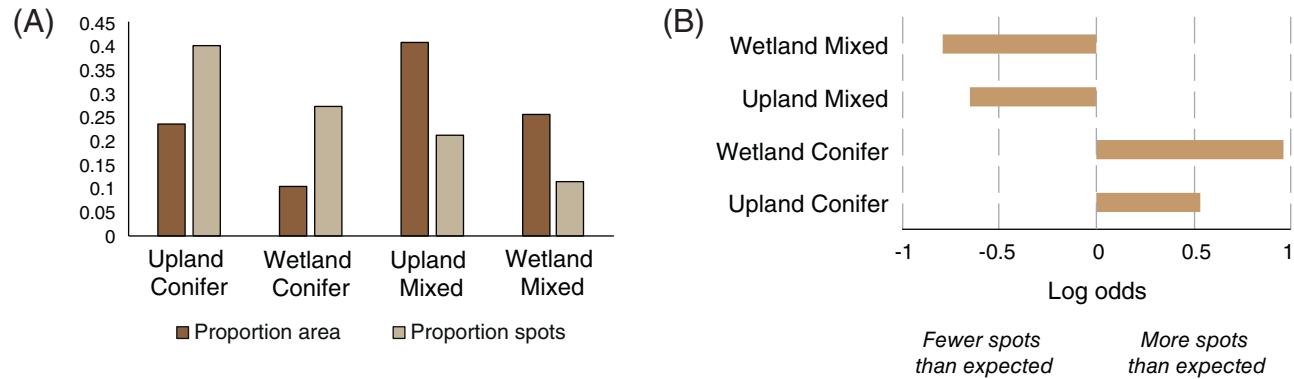


Figure 10.1—(A) Comparison of land area in each forest type with number of spots in each forest type; data from southern Pinelands only. (B) Log odds of number of spots per forest type, based on land area per forest type.

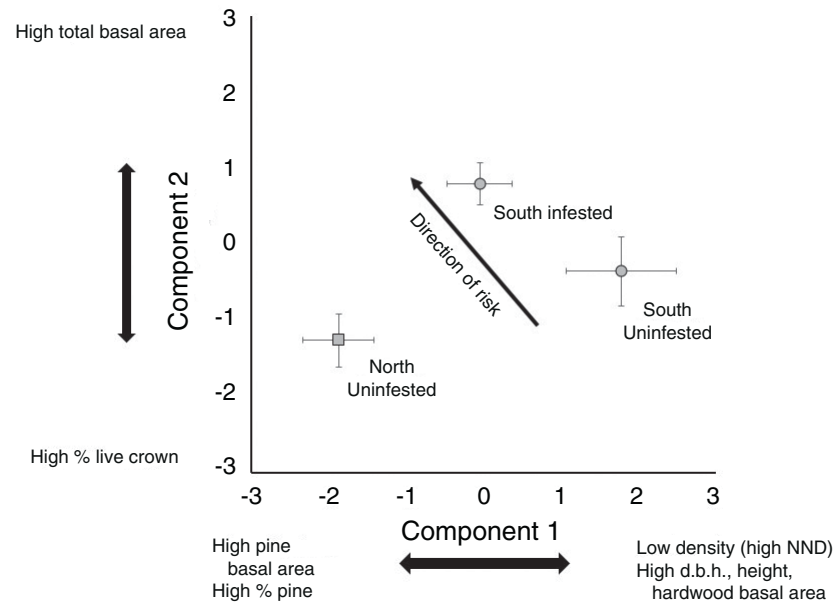


Figure 10.2—Means and standard errors of northern and southern region uninfested forest structure plots and southern region infested forest structure plots, in principal component space. Directions of major variables in relation to each component axis are also indicated.

## Tree Defenses

Variation in resin flow between individual trees within a plot was high, representing close to 60 percent of the variation in the study. There was also a significant fixed effect of soil type ( $F = 10.60_{5,90}$ ,  $p < .0001$ ). However, if the data supported the GDBH, we would expect drainage types (poorly-drained to well-drained, or moist to dry) to align with low to high resin flow, and this was not the case (fig. 10.3A). In the sampling between upland and wetland sites specifically, resin flow in wetland sites was significantly higher than in upland sites ( $F = 9.63_{1,124}$ ,  $p = .0024$ , fig. 10.3B).

## DISCUSSION AND CONCLUSIONS

SPB overwhelmingly preferred pure conifer stands to mixed oak/pine stands, with spots occurring in greater numbers than expected in conifer stands in both wetland and upland areas. Wetland conifer stands were disproportionately

affected, with a very small area of occurrence in the southern Pinelands, but a much larger than expected number of spots. Wetland mixed sites, on the other hand, had fewer spots than expected. The GDBH predicts that SPB spots would occur in greater numbers in wetland sites due to decreased resin flow when water availability is high. Our resin flow data showed that, by contrast, wetland sites in fact have higher resin flow than dry sites. The forest structure data demonstrated mixed results with regard to the GDBH, with wetland forest types having both greater and fewer spots than expected, depending on species composition. Taken together, these results suggest that the percent conifer component at the stand level may overwhelm any effects of resin flow. The extremely high variance among trees also may outweigh the stand level resin flow average.

Stand structure analysis provided further support for the idea that SPB prefer pure conifer stands, with infestation being more

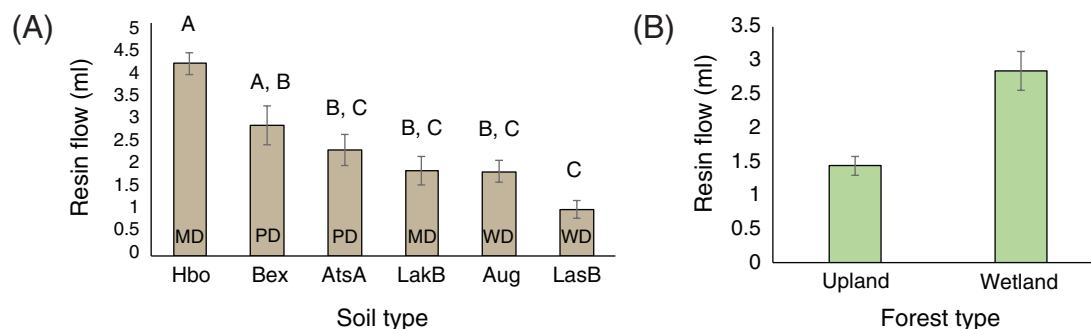


Figure 10.3—(A) Mean resin flow with standard error for each of six soil types. PD = poorly drained, MD = moderately well-drained, WD = well-drained. (B) Mean resin flow with standard error comparing upland vs. wetland sites.



likely in stands with high pine basal area and high percent pine. Stands with lower density (distance between trees) and larger trees (larger d.b.h. and taller heights) were less likely to be infested. NJ DEP has aggressively suppressed SPB north of the Mullica River, resulting in few large infestations in the northern Pinelands. However, as the suppression program draws to a close, the northern Pinelands may be at greater risk of SPB infestation. Stands in the north, on average, are comprised of smaller trees, at a closer distance to one another, and are more likely to be pure rather than mixed conifer—all characteristics that place stands at greater risk of infestation. SPB shows no sign of retreating in the near future, and indeed seems likely to continue its march northward into New England. The northern Pinelands, and stands in the New England States with similar structural characteristics, are particularly at risk of infestation so long as local SPB populations remain at outbreak levels.

## CONTACT INFORMATION

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## INTRODUCTION

Lichens are among the most responsive biological indicators of climate because they depend on atmospheric water and nutrients. Climate change is increasingly associated with shifts in community compositions of tree-dwelling lichens (Aptroot and van Herk 2007, Evju and Bruteig 2013). This suggests that lichen communities will be useful indicators of corresponding changes in forest vegetation as climate change proceeds (Jovan 2008). Currently, the mechanisms by which climate change will affect the growth and demography of lichen species are not well characterized. Will enhanced CO<sub>2</sub> and climatic warming alter lichen growth rates and mortality? Will they result in changes in the relative abundance of boreal forest species? Will they result in directional change of lichen floras?

Our main objective was to link experimental changes in growth and diversity of epiphytic lichens to observed lichen diversity across forests of the Great Lakes and Midwest regions of the United States. This provides a mechanistic basis for interpreting monitoring results from the Lichen Communities Indicator of the Forest Inventory and Analysis (FIA) program, and for developing scientifically defensible climate change indicators. Understanding critical temperature and CO<sub>2</sub> thresholds for lichens helps us evaluate the vulnerability of their roles in forest nutrient cycling, water cycle buffering, and providing wildlife forage in

northern hemisphere forests. This work provides the scientific basis for the use of lichens in ecological forecasting of climate change, a tool to provide policymakers with an early warning of biological effects.

## METHODS

Our approach combined a local climate change experiment with a regional climate gradient analysis. For the climate change experiment, we compared year-to-year variability in epiphytic lichen diversity and growth as part of the SPRUCE experiment (Spruce and Peatland Responses Under Climatic and Environmental change: <https://mnspruce.ornl.gov/>). The SPRUCE site, an ombrotrophic *Picea mariana*-peatland in northern Minnesota, represents a widespread ecosystem of exceptional importance to climate change science because large carbon stocks stored as peat can be mobilized as greenhouse gases CO<sub>2</sub> and CH<sub>4</sub>. Briefly stated, the SPRUCE experiment consists of twelve, 8-m tall enclosures assigned to six different temperature treatments (ambient, control, 2.25, 4, 6.75, and 9 °C above ambient) crossed with two CO<sub>2</sub> levels (ambient vs. enriched). “Control” treatments are stations outside the enclosures, while “ambient” treatments are inside an enclosure maintained at equilibrium with conditions outside (in effect, these are like negative and positive controls). Warming treatments began in August 2015, while CO<sub>2</sub> treatments were set to begin in spring 2016.

# CHAPTER 11.

## Lichen Responses to Experimental Temperature and CO<sub>2</sub> Enhancement: Implications for Climate Change Monitoring

(Project NC-EM-B-14-01)

ROBERT J. SMITH  
SARAH JOVAN  
BRUCE McCUNE

We recorded baseline (pretreatment) responses of lichens by measuring single-species abundance, whole-community diversity, and single-species biomass over the 2 years preceding climate treatments (2013–2015). In 2013, we installed 72 permanent lichen community transects on *Picea mariana* branches in the 12 treatment enclosures (5–7 transects per enclosure). On each 30- to 50-cm length transect, we visually estimated abundance of all lichen species, both in August 2013 and again in August 2015. We also installed 320 *Evernia mesomorpha* lichen transplants on *P. mariana* branches in the SPRUCE enclosures (27 transplants per enclosure) to investigate how warming and CO<sub>2</sub> treatments affect the growth of a common boreal lichen at the southern edge of its geographic distribution. Transplant biomass was re-weighed every August in 2013, 2014, and 2015. We used linear mixed-effects models and permutational multivariate analysis of variance to evaluate differences in diversity and biomass between treatment enclosures and through time. The mixed-effects model was a separate-means linear model permitting different

variances for each year and a compound symmetry correlation structure for years within enclosures. Therefore, responses could differ among years within enclosures, which allows us to distinguish possible treatment effects while accounting for the effect of time.

To interpret local changes at the SPRUCE site within a regional context, we analyzed existing FIA lichen data collected from 336 sites between 1994 and 2005 in the Midwest region (MN, MI, WI, IL, IN, OH). From the 158 lichen species recorded, we created a regional lichen-based climate index (ordination scores from nonmetric multidimensional scaling, NMS). Each site was assigned a score based on lichen community composition (fig. 11.1), which was then regressed or otherwise related to climatic variables. Thirty-year climate data allowed us to establish nine “climate zones” sharing similar climate regimes (*k*-means clustering of climate principal components). We then identified lichen indicator species based on fidelity and exclusivity to a climate zone (or group of zones).

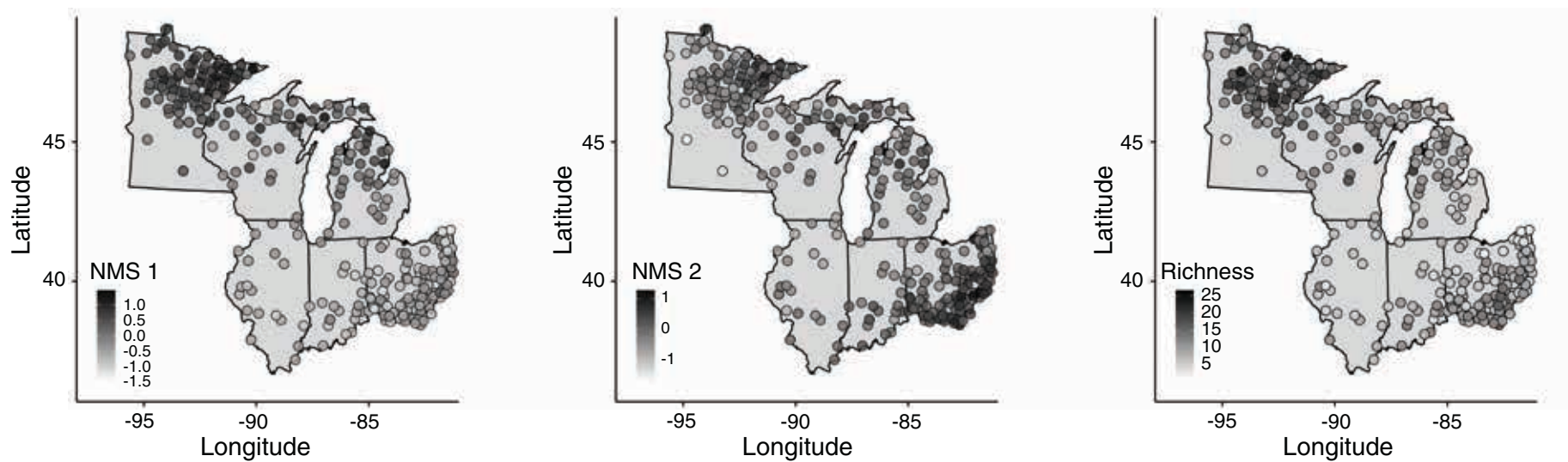


Figure 11.1—Lichen index scores (“NMS 1” and “NMS 2”) and species richness for each Forest Inventory and Analysis (FIA) site in the Great Lakes and Midwest regions. Similar NMS scores among sites indicate similar lichen community compositions. Linking existing regional FIA plots to climate change effects at the Spruce and Peatland Responses Under Climatic and Environmental change (SPRUCE) experiment is a key goal. Plot locations are approximate.

## RESULTS

We recorded a total of 21 lichen species of varying abundance and frequency in the 12 SPRUCE enclosures (table 11.1). Over the 2 years of observation, mean abundance of lichen species remained consistent (no more than 2 percentage points change for any species, most changes were < 0.5 percentage points, table 11.1). As may be expected and desired before treatments were applied, there was no significant difference in either species richness or Shannon diversity index values among unactivated treatment enclosures. Species richness increased over the 2-year pretreatment period, but Shannon diversity did not change. As expected, we also detected no difference in initial, pretreatment species composition among the enclosures destined for particular treatments.

Following a substantial 32-percent decline over the harsh 2013/14 winter, mean lichen biomass of *Evernia* transplants increased 15 percent over the following 2014/15 interval (fig. 11.2), changes which were significant. Critically, these changes were not significantly different among the unactivated “treatment” groups (i.e., no differences prior to treatments).

The NMS lichen-based climate index (fig. 11.1) explained 75.1 percent of the variation in community composition and was correlated with climatic variables (including precipitation, mean annual temperature, continentality, and frost-free period). For

**Table 11.1—Epiphytic lichen species abundance and frequency among 72 transects over a 2-year period prior to treatments at SPRUCE experimental site**

| Species                              | Percentage cover |            | Relative frequency (%) |       |
|--------------------------------------|------------------|------------|------------------------|-------|
|                                      | 2013             | 2015       | 2013                   | 2015  |
| <i>Bryoria furcellata</i>            | 0.1 ± 0.6        | 0.1 ± 0.6  | 1.4                    | 1.4   |
| <i>Buellia</i> sp.                   | 0 ± 0.2          | 0 ± 0.1    | 4.2                    | 16.7  |
| <i>Candelaria fibrosa</i>            | 0                | 0.1 ± 0.1  | 0                      | 1.4   |
| <i>Cetraria halei</i>                | 7.5 ± 9.6        | 5.8 ± 7.7  | 83.3                   | 91.7  |
| <i>Evernia mesomorpha</i>            | 3.8 ± 4.7        | 4 ± 4.4    | 93.1                   | 100.0 |
| <i>Flavoparmelia caperata</i>        | 2.1 ± 4.1        | 2.3 ± 4.3  | 54.2                   | 76.4  |
| <i>Flavopunctelia flaventior</i>     | 0                | 0.1 ± 0.6  | 0                      | 2.8   |
| <i>Flavopunctelia soledica</i>       | 0                | 0.1 ± 0.6  | 0                      | 1.4   |
| <i>Fuscidea arboricola</i>           | 0.2 ± 0.8        | 0.2 ± 0.9  | 6.9                    | 25.0  |
| <i>Hypogymnia physodes</i>           | 2.3 ± 3.3        | 2.1 ± 3.2  | 63.9                   | 75.0  |
| <i>Lecanora circumborealis</i> group | 7.9 ± 10.1       | 8.6 ± 10.5 | 88.9                   | 94.4  |
| <i>Lecanora strobilina</i>           | 0.6 ± 0.9        | 0.4 ± 0.7  | 52.8                   | 87.5  |
| <i>Melanelixia subaurifera</i>       | 0.6 ± 2          | 0.6 ± 1.9  | 13.9                   | 20.8  |
| <i>Ochrolechia arborea</i>           | 4.3 ± 5.4        | 5.2 ± 5.7  | 52.8                   | 81.9  |
| <i>Parmelia sulcata</i>              | 1.4 ± 2.5        | 1.5 ± 3.1  | 54.2                   | 75.0  |
| <i>Punctelia caseana</i>             | 0.1 ± 0.6        | 0.2 ± 0.7  | 1.4                    | 12.5  |
| <i>Ramalina dilacerata</i>           | 0.1 ± 0.3        | 0.1 ± 0.3  | 9.7                    | 16.7  |
| <i>Scoliciosporum chlorococcum</i>   | 7.4 ± 12.3       | 9.4 ± 13.4 | 52.8                   | 81.9  |
| unknown sterile crust                | 0.3 ± 1.0        | 0.6 ± 1.4  | 18.1                   | 77.8  |
| <i>Usnea hirta</i>                   | 0.9 ± 1.3        | 0.6 ± 1.2  | 58.3                   | 69.4  |
| <i>Vulpicida pinastri</i>            | 0.1 ± 0.6        | 0.1 ± 0.6  | 1.4                    | 1.4   |

Means ± one standard deviation for each species.



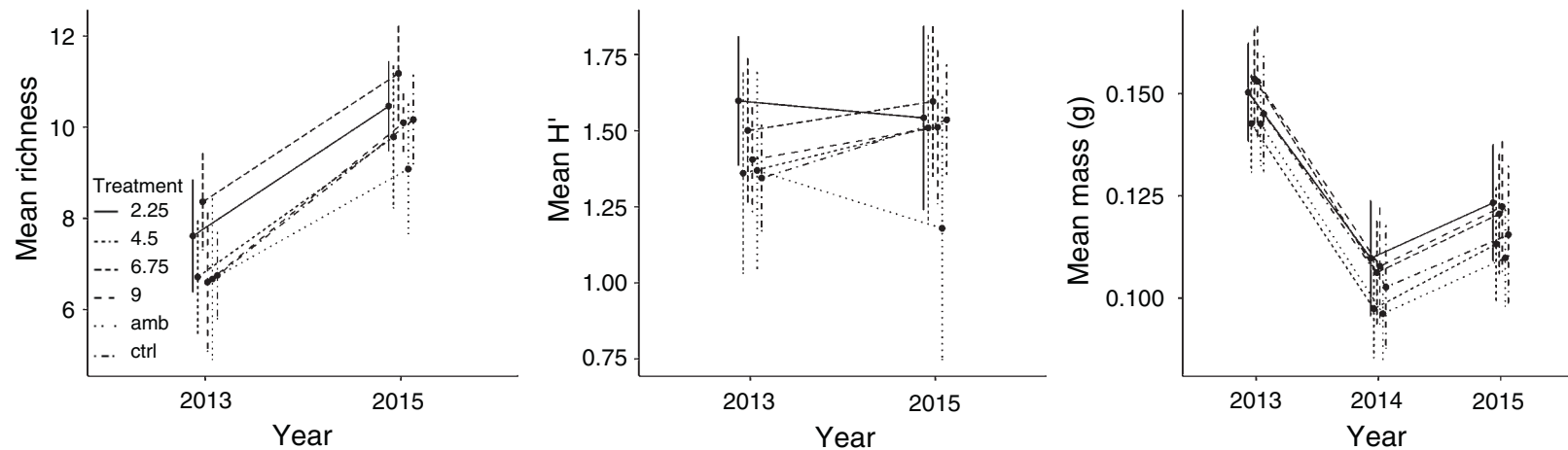


Figure 11.2—Changes in epiphytic lichen species richness, Shannon diversity ( $H'$ ), and growth of *Evernia* transplants over the 2-year period prior to treatments at the Spruce and Peatland Responses Under Climatic and Environmental change (SPRUCE) experimental site. Means  $\pm$  95 percent CI for each treatment (units are  $^{\circ}\text{C}$  above ambient). Shannon diversity is an index incorporating species evenness based on proportional abundance of species.

example, correlations between climate index scores and temperature, continentality, and precipitation all exceeded 0.50. An array of lichen indicator species were faithful and specific to the climate zones.

## DISCUSSION

Our objective was to study growth and diversity responses of epiphytic lichens experiencing experimental climate change. Prior to climate treatments, inter-annual diversity was consistent, but inter-annual growth showed large variation. Future monitoring of *Evernia mesomorpha* on an annual basis will be critical

because of the high variability in biomass that we observed in the years preceding climate treatments. Future aims are to predict range shifts and abundance of this boreal species under different regional climate warming and  $\text{CO}_2$  scenarios. Together, these efforts could lead to lichen-based risk and vulnerability assessments (Ellis 2013) that can guide forest policy.

Regional lichen diversity and the lichen-based climate index varied reliably in relation to climate. Changes in climate index scores at sites remeasured in the future would be useful indicators of climate change. Lichen indicator species reflected well the differences

among climate zones along the transition from boreal forests to northern mixed hardwoods to southern hardwoods forests. Substantial increases or decreases in lichen indicator species would be indicative of corresponding climatic shifts, suggesting their utility as visible and easy-to-measure monitoring targets to indicate forest health and trends.

Linking existing regional lichen data to climate change effects at the SPRUCE experiment remains a key goal as the 10-year experiment unfolds. To make this linkage, we completed 54 supplemental FIA-style plots along a belt transect from southern Illinois to Ontario, passing through the immediate area of the SPRUCE experimental site. New information will permit us to evaluate the suitability of existing FIA lichen community data for establishing regional climatic responses of lichens. This work contributes to a growing body of evidence that early-responding epiphytic lichens are sensitive indicators of climate-induced shifts in forest vegetation.

## ACKNOWLEDGMENTS

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## CONTACT INFORMATION

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## INTRODUCTION

**S**outhern pine (*Pinus* spp.) forests in the Southeastern United States appear to be threatened by a variety of interacting abiotic and biotic factors. However, the exact etiology, nature, impact, and extent of growth loss remains controversial (Coyle and others 2015, Eckhardt and Menard 2009, Eckhardt and others 2010, Meyerpeter 2012). Forest composition and structure in the Southeastern United States have undergone tremendous changes during the last two centuries due, in part, to major alterations to wildfire regimes, loss of original habitat, and forest fragmentation and isolation resulting from farming and urbanization activities (USDA Forest Service 2011). In addition, there has been large-scale conversion of native longleaf pine (*P. palustris*) forests to agricultural lands to loblolly pine (*P. taeda*) plantations. This has resulted in the presence of overstocked stands that are generally more susceptible to insects and diseases (Baker and Langdon 1990, Fettig and others 2007, MacCleery 1992). These pine plantations are now an economic force in the region and nationwide, providing up to 58 percent of total timber in the United States, worth >\$30 billion per year (Hanson and others 2007). Further, the pine plantations are ecologically critical to many wildlife species, assist in soil stabilization and reforestation of degraded lands, and provide other cultural and tourism benefits (Baker and Landon 1990).

Pine health issues have been reported since the 1950s on various public lands, military bases, and private lands in Alabama and

Georgia (Eckhardt and Menard 2009; Eckhardt and others 2007, 2010; Otrosina and others 2008). Studies indicate that pine trees greater than 35 years old in these areas are showing symptoms of dieback such as thinning of crowns, yellowing and loss of needles, and fine and lateral root decay (fig. 12.1). Various



*Figure 12.1—Typical dieback and mortality in a pine stand in Georgia. (Photo by: Kamal Gandhi, University of Georgia)*

## CHAPTER 12.

### Interactions Among Abiotic and Biotic Factors Affecting the Health of Pine Forests in the Southeastern United States

(Project SO-EM-B-12-03)

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root pathogens including heterobasidion root disease (by *Heterobasidion irregulare*), littleleaf disease (by *Phytophthora cinnamomi*), and *Leptographium* spp. transmitted by root-feeding weevils (*Hylastes*, *Hylurgops*, and *Pachylobius* spp.) have been found associated with symptomatic pines, however without consistent relationships between their presence and symptomology. More recently, pine health issues have been defined under the umbrella of “pine decline” or “southern pine decline” with a similar etiology. *Leptographium* spp. have been found to be prevalent within these sites as are symptomatic trees on steeper (7–32 percent) and south-facing slopes (Eckhardt and Menard 2009; Eckhardt and others 2007, 2010). This so-called “pine decline” is most commonly associated with loblolly pines, but longleaf pines have also been found with similar symptoms, especially on military installations (Duke and others 2008, Orosina and others 2008). The major areas for pine decline have been hypothesized to be in the Piedmont, Atlantic and Gulf Coastal Plains, and along the Sandhills Fall-line (Eckhardt and others 2010). Other abiotic factors (e.g., deleterious weather conditions, stand history, soil types) and biotic factors such as bark beetles (*Ips* and *Dendroctonus* spp.) and woodboring beetles (*Monochamus* spp.) have been hypothesized to contribute to stand dieback and mortality under varying conditions (e.g., Zahner and others 1989). However, none of the previous studies have holistically assessed interactions among predisposing factors such as soil conditions, climate, land use, and disturbance history, inciting factors such as drought, damage by insects, and current

management practices, and contributing factors such as bark beetles on southern pine health (Manion 1981).

Our project aims to determine the extent and severity of pine health issues in the Southeast. Additional questions include what are the main factors affecting pine health; what kinds of management practices can prevent and mitigate damage; and could decision models assist forest and land managers to manage pine stands effectively and maintain their resilience over a long-term period?

## METHODS

To delineate the extent of pine health issues and associated topographic factors, we analyzed USDA Forest Inventory and Analysis (FIA) data. We focused on remeasured plots ( $n = 5,396$ ) from all Southeastern States that had unharvested (since the start of inventory) loblolly pine and loblolly pine-hardwood forest types (Coyle and others 2015). For each plot, we calculated the annual rate of pine mortality, net growth, and gross growth based on differences in tree measurements between inventory cycles. We identified the remeasured plots that reported negative net pine growth (i.e., where pine mortality exceeded growth). Two average nearest neighbor distance analyses, one for the set of negative net growth plots and the other for the set of all (i.e., non-harvested) remeasured plots, were conducted as based on the plots’ geographic coordinates. An index was calculated as the ratio between the observed average distance and the distance expected given a hypothetical random distribution with

the same number of observations over the same total area. Two outcomes were identified with an index value:  $<1$  indicated a clustered spatial pattern and  $>1$  indicated a dispersed spatial pattern. These patterns were compared for the set of negative net growth plots and for all plots. We also evaluated whether loblolly pine decline was more prevalent on steep slopes and southern-southwestern aspects as previously hypothesized (Eckhardt and Menard 2008). This hypothesis was manifested as a geographic risk model in Meyerpeter (2012) that assigned risk ratings based on sites' slope and aspect values. For both net growth and mortality, we tested for differences between the risk levels based on regression models that accounted for stand age, site index, and pine percentage of a plot.

To better understand the interactions between abiotic and biotic factors affecting pine health, we conducted field studies in Alabama and Georgia. For the purpose of this study, sites are defined as a continuous forest stand (approximate acre in size), plots nested within sites, and circular subplots nested within plots. We established plots in 13 healthy and 24 unhealthy (symptomatic areas as deemed by landowners/managers) sites in loblolly pine-dominated areas in Alabama and Georgia. Sites in Alabama were within the Talladega National Forest (previously deemed as southern pine decline areas), while those in Georgia were located on Federal, State, and private land. At each site, we sampled three plots, each 250 m apart. At each location we sampled four subplots arranged FIA-style, i.e., center plot of 10-m

radius, and plots at 0, 120, and 240 degrees from the center, 50 m apart on the center. Hence, a total of 111 plots and 444 subplots were established for the study.

The following data were collected at the site level: stand age and area, disturbance history, and local weather conditions. We measured the slope and aspect of each subplot, and collected soil samples from the 0–30 cm depth increment from five locations per subplot. Soils were analyzed for physical (e.g., texture and bulk density) and chemical (e.g., pH, nitrogen, phosphorus, and potassium content) characteristics. Stand characteristics such as pine and hardwood basal area, and pine tree crown size and health were measured. We recorded tree height, tree age, and insect incidence, and damage using conventional means on up to three mature pine trees in each subplot. We collected samples from one small, medium, and large root on up to three mature pine trees in each subplot; these samples were taken to the laboratory to determine the incidence of *Leptographium* spp. and *Heterobasidion*. All putative *Leptographium* isolates were identified to morphospecies, and subset of cultures sent to Forestry Agricultural and Biotechnology Institute (FABI), South Africa, for molecular identification.

## RESULTS AND DISCUSSION

Using FIA data, we found negative net pine growth on 181 plots, representing only about 3 percent of the plots in the southeastern region (Coyle and others 2015) (fig. 12.2). Most of



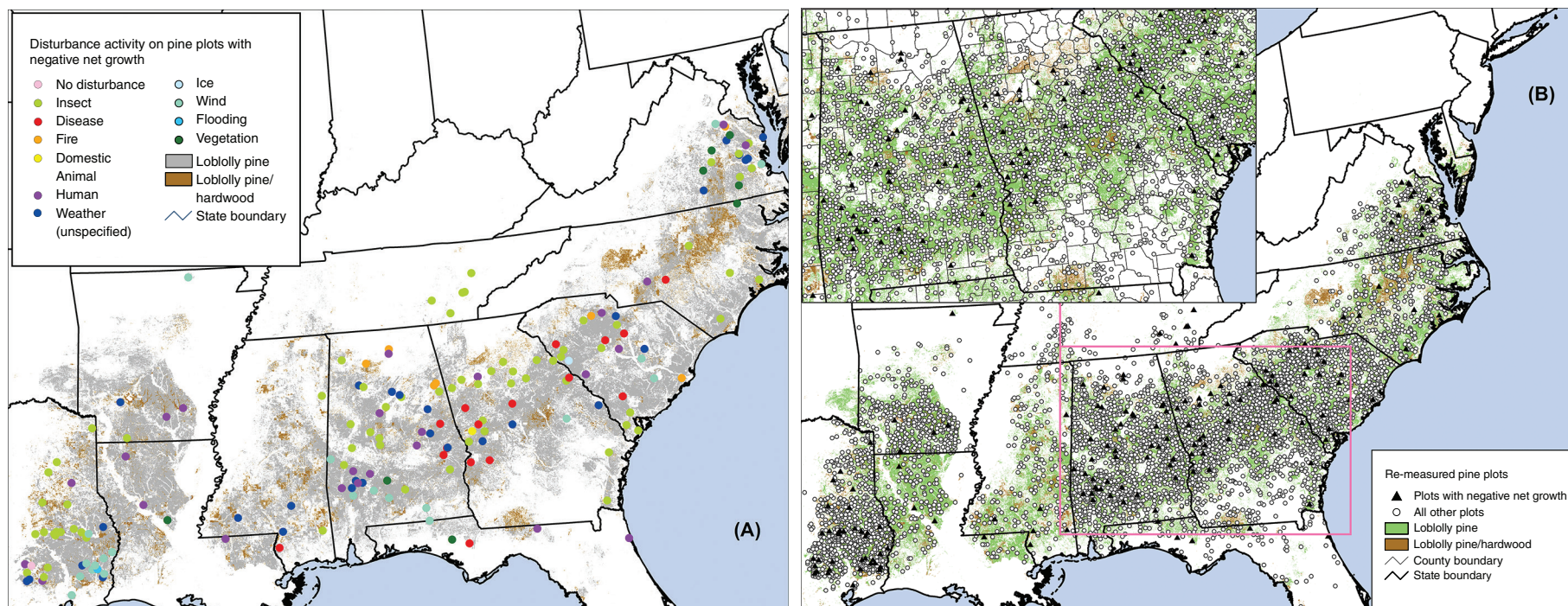


Figure 12.2—Remeasured Forest Inventory and Analysis (FIA) plots in the loblolly pine and loblolly pine/hardwood forest types where (A) plots exhibiting negative net pine growth, labeled according to disturbance agent, and (B) locations of plots with negative net pine growth versus all other pine plots (adapted from Coyle and others 2015). Plot locations are approximate.

the 181 plots were disturbed, most commonly by insect activity or damaging weather events (unspecified). Similar index values for the set of negative net growth plots and set of all plots were observed using the average nearest neighbor distance analyses. After accounting for variation in stand age, site index, and stand composition, slope and aspect were significant factors in only 1 plot each out of 36 comparisons for net pine growth in the loblolly pines and none for the loblolly pine-hardwood forest type. Overall, we found little evidence to support

the existence of a regional pattern of decline in loblolly pine forest health from FIA data. Only a small percentage of non-harvested plots exhibited negative net growth (i.e., abnormally high mortality), and virtually all of these plots were associated with common disturbance agents. A lack of a distinctive spatial pattern in these negative net growth plots (fig. 12.2) suggests that there are no obvious hot spots of high mortality that can be linked to a more local-scale phenomenon impacting loblolly pine growth. After accounting for expected



variation due to stand age, site index, and plot pine proportion, there were minimal differences in pine growth and mortality between slopes and aspects.

Soil texture was predominantly sandy loam or sandy clay loam with greater sand content in unhealthy sites in Alabama. As compared to the healthy pine sites, pH and micronutrients, especially phosphorus and zinc, were lower than the preferred range of loblolly pine in unhealthy sites in Alabama, and high levels of calcium suggest an old-field effect and historical agricultural land-use in these sites (fig. 12.3). Similar patterns for nutrients were not observed in Georgia. Higher basal area particularly for hardwood species was observed in unhealthy sites, especially in Georgia. Total basal area was approximately 16-percent (Alabama) and approximately 41-percent (Georgia) greater in unhealthy stands (fig. 12.4). Both lower nutrient levels and high basal area are likely major stress agents in these symptomatic areas. A similar incidence of *Leptographium* spp. and *Heterobasidion* spp. was found in healthy and unhealthy sites. No spatial pattern of *Leptographium* spp. was observed from the middle of the stand to 250 m and 500 m away from the original plot. Our results indicate that there are no clear patterns of a widespread decline of pines in the Southeast. The dieback we observed was variable and limited to the local scale. Finally, we found that multi-level abiotic (e.g., soil nutrient levels) and biotic factors (e.g., basal area) may be combining to affect pine health and cause decline-like symptoms in this region.

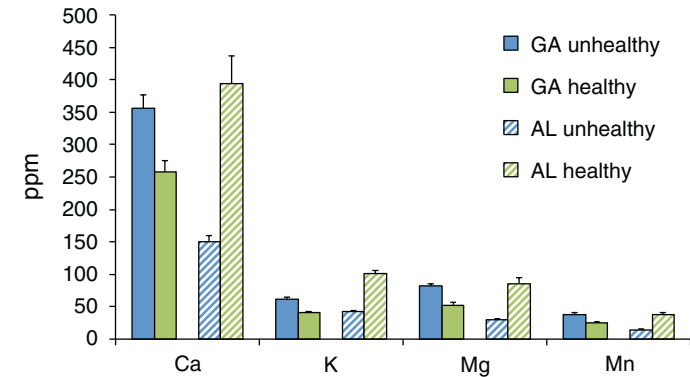


Figure 12.3—Soil levels of calcium (Ca), potassium (K), magnesium (Mg), and manganese (Mn) in healthy and unhealthy (symptomatic) pine stands in Georgia (GA) and Alabama (AL).

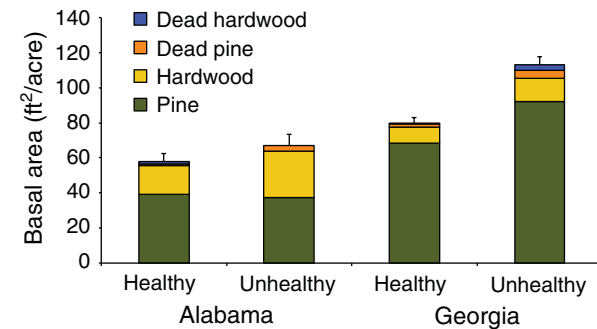


Figure 12.4—Basal area in healthy and unhealthy pine stands in Alabama and Georgia.

## ACKNOWLEDGMENTS

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## INTRODUCTION

Stresses to trees under a changing climate can lead to changes in forest tree survival, mortality, and distribution. For instance, a study examining the effects of human-induced climate change on forest biodiversity by Hansen and others (2001) predicted a 32-percent reduction in loblolly-shortleaf pine habitat across the Eastern United States. However, they also predicted an average increase in area of 34 percent for oak-hickory forests and a 290-percent increase in oak-pine forests. Drought is often a leading cause of stress and mortality in forest trees (Allen and others 2010, Choat and others 2012).

Drought, whether induced by climate change or other mechanisms, is considered a major inciting factor of forest decline (Leininger 1998, Manion 1981). For example a drought-induced oak decline event in Arkansas and Missouri that began in 2000 affected up to 120 000 ha in the Ozark National Forest of Arkansas alone (Starkey and others 2004). A study that examined 1991–2005 data across the Southeastern United States found that drought negatively impacted both growth and mortality of pines and mesophytic species, but not oaks (Klos and others 2009).

In that regard, three phases of research were addressed through this investigation. In the first phase, regional relationships of mortality and drought across the Ozark Highlands of

Arkansas and Missouri were examined. In the second phase, terrestrial vegetation response to climatological influences across 10 States in the Southeastern United States was investigated. In the third phase, probabilities were generated predicting future tree species distribution across the Southeastern United States.

### Phase I: Mortality in the Ozark Highlands

For decades, oak decline has impacted Midwestern upland oak-hickory forests, particularly species in the red oak group (*Quercus* Section Lobatae) across the Ozark Highlands of Missouri, Arkansas, and Oklahoma. Drought is a common inciting factor of oak decline, and advanced tree age is considered a predisposing factor. Opportunistic organisms that take advantage of the already stressed trees such as armillaria root-rotting fungi and opportunistic wood-boring insects are believed to contribute to oak decline. An objective of this phase of research was to examine the relationship of drought and oak mortality.

### Phase II: Mortality in Southeastern States

The objective of this phase of research was to expand our examination of the relationship between drought and red oak mortality across the following 10 Southeastern States: Alabama, Arkansas, Florida, Georgia, Mississippi, North Carolina, Oklahoma, South Carolina, Tennessee, and Texas.

## CHAPTER 13.

### Tree Mortality Estimates and Species Distribution Probabilities in Southeastern United States Forests (Project SO-EM-F-10-01)

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### Phase III: Future Distribution of Species in Southeastern States

This study area included the same 10 Southeastern States examined in Phase II. The objective of this phase of the research was to examine the likely spatial distribution of six major coastal species [loblolly pine (*Pinus taeda*), longleaf pine (*P. palustris*), red maple (*Acer rubrum*), water oak (*Quercus nigra*), southern red oak (*Q. falcata*), and post oak (*Q. stellata*)] through the year 2070.

## METHODS

### Mortality in the Ozark Highlands

To accomplish this phase of work, Forest Inventory and Analysis (FIA) annualized 1999–2010 data from plots in the Ozark Highlands of Arkansas and Missouri were used as well as corresponding cumulative monthly Palmer Drought Severity Indexes (PDSIs) and cumulative yearly PDSIs. The FIA data included 6,997 plots containing oaks. These plots were measured and remeasured from 1999 through 2010. Trees were divided into three species groups: red oak species group, white oak species group, and a non-oak species group. The percentage of standing dead trees was calculated by basal area and number of trees for each species group. This provided an indirect measure of oak decline and mortality. Cross-correlation analyses were used to examine the relationship between growing season PDSI and mortality. Detailed methods can be found in Fan and others (2012).

### Mortality in Southeastern States

Mortality data were obtained from the 1991–2000 Forest Inventory and Analysis database for 10 Southeastern States (Alabama, Arkansas, Florida, Georgia, Mississippi, North Carolina, Oklahoma, South Carolina, Tennessee, and Texas). To calculate percent mortality, basal area of standing dead trees was divided by total basal area (live and dead trees) of each FIA plot. Total annual precipitation, maximum temperature, minimum temperature, and temperature range were extracted from the PRISM (Parameter elevation Regression on Independent Slopes Model) climate database (Oregon State University 2012). Climate data (maximum and minimum temperatures and precipitation) were extracted from PRISM for 24 years prior to each inventory year. Potential spatial trends were examined through kernel smoothing. For detailed methods, see Crosby and others (2012).

### Future Distribution of Six Species across the Southeastern States

The likelihood of presence of six economically important southeastern tree species was projected through the year 2070 by using three climate envelope model simulation platforms. The six species were loblolly pine (*Pinus taeda*), longleaf pine (*Pinus palustris*), red maple (*Acer rubrum*), water oak (*Quercus nigra*), southern red oak (*Q. falcata*), and post oak (*Q. stellata*). The three simulation platforms compared were BIOCLIM, GLM, and MaxEnt, where BIOCLIM is a classic climate envelope model with percentile distributions (Hijmans and Graham 2006), GLM is a parametric method, and MaxEnt

is a maximum entropy model (Hijmans and others 2012). Detailed methods can be found in Sui (2015).

## RESULTS

### Mortality in the Ozark Highlands

Of the three species groups examined in this study, the red oak group consistently dominated dead basal area as a proportion of the total basal area from 1999 through 2010. The proportion of dead basal area and dead trees for the red oak group reached a peak in 2008 to 2009, 2 years after the end of the last drought event. In contrast, the white oak group and the non-oak group had comparable mortality rates that remained relatively stable over time (fig. 13.1).

Regionally, this mortality was significantly correlated with growing season PDSI with a

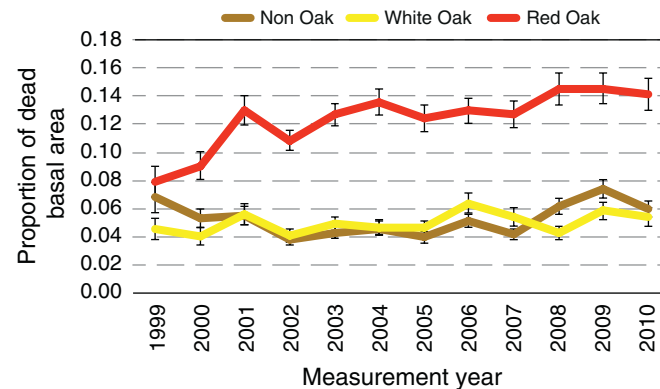


Figure 13.1—The 1999–2010 proportion of dead basal area for the non-oak group, white oak group, and the red oak group. Vertical lines are standard error for dead basal area of each species group and year. Based on 6,997 Forest Inventory and Analysis plots across the Ozark Highlands.

2- to 3-year time lag behind single drought events. Cumulative impacts of drought may last up to 9 years based on the previous 17 years of PDSI data. There were two droughts during the study period: a major drought from 1998 through 2000 and a somewhat milder drought in 2005–2006. These droughts triggered the oak decline events and associated mortality observed during the study period. Most of the red oak hot spots of mortality occurred toward the central part of the Ozarks and gradually increased in area over the Ozark Highlands. Small areas of white oak group mortality were more sporadic. See Fan and others (2012) for further details.

### Mortality in Southeastern States

Levels of mortality varied between the upland red oak group and bottomland red oak group; mortality rate also varied within each of the two groups across the landscape. The highest mortality in upland red oak species extended from the coastal areas of southern Alabama northeastward through the Appalachian Mountains. Mortality rates in the coastal area of South Carolina ranged from 25 to 30 percent. Ozark Highland mortality was lower at 10 to 15 percent. For detailed results see Crosby and others (2012).

### Future Distribution of Six Species across the Southeastern States

The three climate envelope models BIOCLIM, GLM, and MaxEnt were used to project the future distribution of six species during three time periods: 2010 through 2020, 2021 through 2050, and 2051 through 2070. Predicted distribution maps for southern red



oak (fig. 13.2), water oak (fig. 13.3), post oak (fig. 13.4), and red maple (fig. 13.5) show changes through time and the differences among the three models. Further details of this phase of the research as well as distribution maps for loblolly pine and longleaf pine can be found in Sui (2015).

## DISCUSSION

### Mortality in the Ozark Highlands

This analysis of mortality and of PDSI as it correlated with mortality between 1999 and 2010 corroborated the important function of

drought in this oak decline and mortality event in the Ozark Highlands. The regionwide droughts that began in 1998 and again in 2005 were followed by an escalation in relative mortality in the red oak group. Decline sites appear to be in areas where during the preceding century red oak established on dry, rocky, shallow soils. Much of this red oak establishment took place after the extensive harvesting and agricultural clearing that occurred during the early part of the 20th century and was facilitated by a period with relatively cool and moist climate.

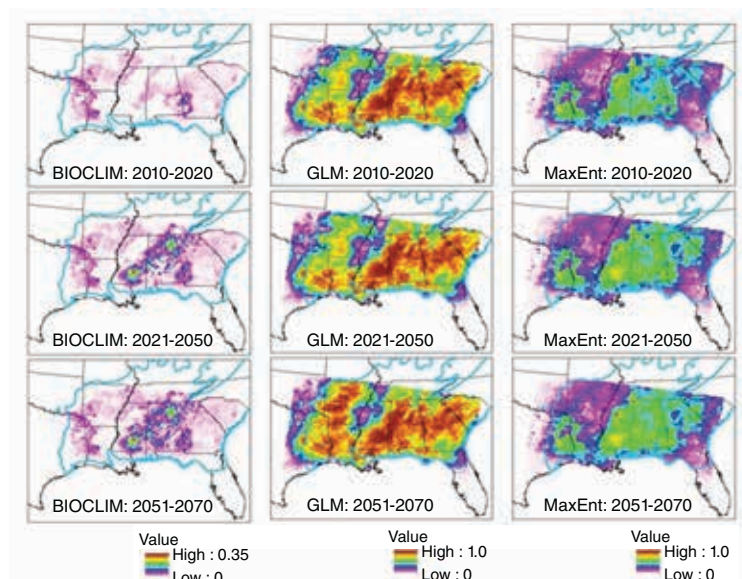


Figure 13.2—Distribution probabilities for southern red oak for the three climate envelope models: BIOCLIM, GLM, and MaxEnt.

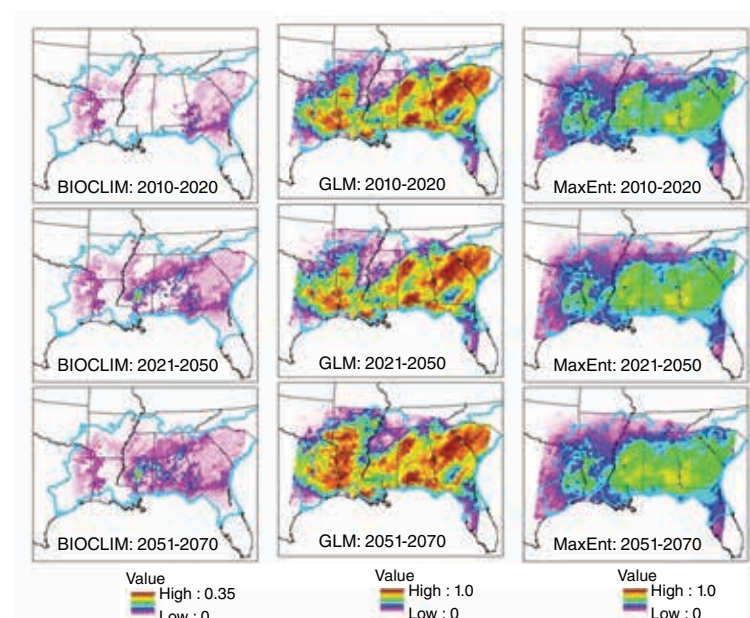


Figure 13.3—Distribution probabilities for water oak for the three climate envelope models: BIOCLIM, GLM, and MaxEnt.



### Mortality in Southeastern States

During the 1991–2000 study period, mortality was higher in the eastern portion of the study area than in the Ozark Highlands in the west. The most important climate variables associated with mortality were average annual temperature for bottomland species and average annual precipitation for upland species. This difference in influential climate variables between bottomland and upland sites could inform future management strategies.

### Future Distribution of Six Species across the Southeastern States

For all species, the BIOCLIM model produced the lowest species distribution probabilities, and the GLM model produced the highest species distribution probabilities. In all three models southern red oak, water oak, and post oak distribution probabilities increased with time, with post oak displaying the highest distribution probabilities. Other studies support a future increase in oak species (Hansen and others 2001,

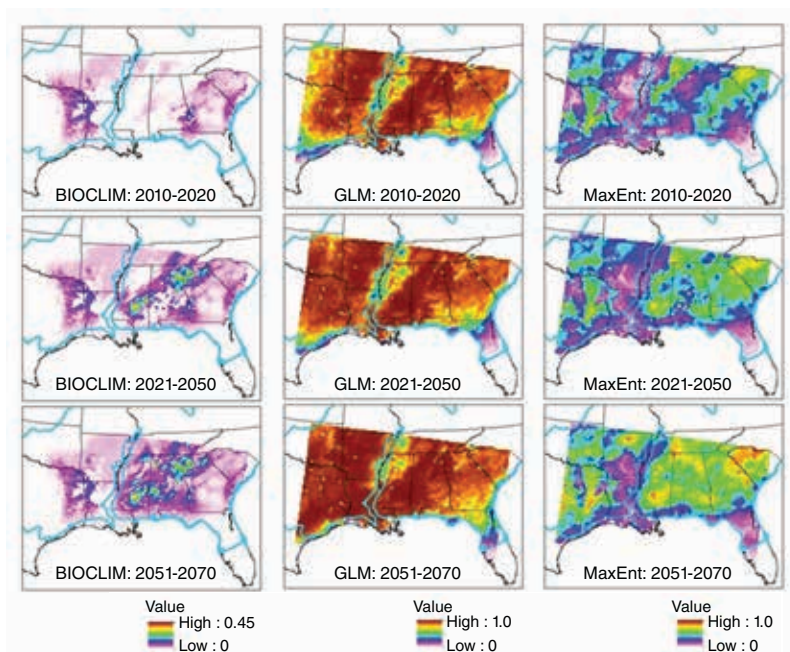


Figure 13.4—Distribution probabilities for post oak for the three climate envelope models: BIOCLIM, GLM, and MaxEnt.

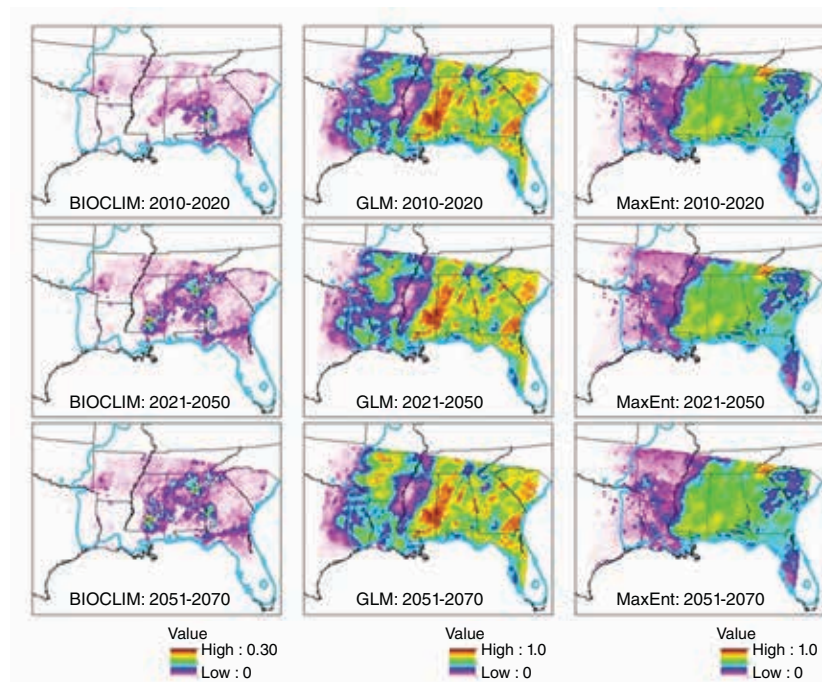


Figure 13.5—Distribution probabilities for red maple for the three climate envelope models; BIOCLIM, GLM and MaxEnt.

Vose and Elliott 2016). For red maple there was a small increase in distribution with time in the BIOCLIM and GLM models at the core of the region; the MaxEnt model indicated relatively little change.

### Overall Summary

These combined results point to a future in which impacts of climate change on drought, temperature, and precipitation may well lead to changes in forest species composition, structure, and spatial distribution across the Southeastern United States. Support continues to grow for the idea of climate-influenced forest change. For instance, a study published in *Nature* that examined global vulnerability of forests to drought predicted extensive forest decline in areas where droughts are predicted to increase in length and severity (Choat and others 2012). An assessment of global tree mortality indicated the prospect for expanded tree mortality as influenced by drought and heat (Allen and others 2010). A recent review article looking at large-scale “mega-disturbances” on temperate forests suggests future large-scale transformations in forest composition and structure (Miller and Stephenson 2015). A more recent paper that examined the influence of changes in fire, climate change, and other disturbances in the Eastern United States suggests that these disturbances will favor oak in the long run (Vose and Elliott 2016). All of this points to a future in which past methods of forest management may require novel

implementation to mitigate unfavorable changes in forest species diversity, composition, structure, and function.

## CONCLUSIONS

In the Ozarks, proactive management will be necessary to reduce the impacts of future oak decline events. However, it may be necessary for management intervention to be taken in young stands before they mature and become more susceptible to decline (Spetich and others 2016).

The complexity of predicting future distributions of tree species is compounded by an unknown degree of influence from future disturbance and climate. In the analyses of future species distributions, the three alternative models predicted different future tree species distributions. There are no currently available models that completely reflect the dynamic ecology of these forests as trees compete with one another and interact with climate, site conditions, and disturbance factors. Continued monitoring of forest health and tree mortality across the region will be important so that management intervention can be applied in an appropriate and timely way. Well-informed managers will be prepared to guide these forests in a way that maintains structure, function, diversity, and health to the benefit of both the environment and society.

## CONTACT INFORMATION

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## INTRODUCTION

**P**olyphagous and Kuroshio shot hole borers (PSHB, *Euwallacea* sp. #1 and KSHB, *Euwallacea* sp. #5 respectively; SHB collectively) are invasive ambrosia beetles to California, which are morphologically indistinguishable, but genetically distinct species (O'Donnell and others 2015). Each beetle carries its own novel pathogenic species of fungi: PSHB is mutually associated with *Fusarium euwallaceae*, *Graphium euwallaceae*, and *Paracremonium pembeum* while KSHB is associated with *Fusarium* sp. and *Graphium* sp. Among all fungal species associated with the beetles, *Fusarium* spp. are found to be the most prevalent fungi that SHB carry in their mycangia (Eskalen, unpublished data;<sup>1</sup> Eskalen and others 2013; Freeman and others 2013; Lynch and others 2016).

SHB inoculate host trees as they bore into the wood while forming brood galleries in the active wood (sapwood). Fungal spores are released from their mycangia as they establish in the sapwood and survive by utilizing their pathogenic fungi as a food source. Fungal colonization in the sapwood blocks the movement of water and nutrients in over 137 susceptible tree species representing 62 families (13 species native to CA), including arroyo willow (*Salix lasiolepis*), red willow (*Salix laevigata*), cottonwood (*Populus fremontii*), coast live oak (*Quercus agrifolia*), California sycamore

(*Platanus racemosa*), Engelmann oak (*Quercus engelmannii*), big leaf maple (*Acer macrophyllum*), and avocado (*Persea americana*).

Hardwoods are an integral component of low-elevation riparian forests, and losses of these species can have potential impacts to water quality, fuel structure, wildlife habitat, recreation, and urban forest. In addition to landscape habitat, the insect-disease complex has also already spread to the \$450 million avocado industry in California. The cost of preventive treatments and hazard tree removal has already begun to impact arboreturns, botanical gardens, and urban forests in southern California. Through this research, we aim to determine the host tree species at risk in urban forests from this invasive insect-disease complex in California, as well as explore possible biological control treatment methods for beetle-infested trees by targeting the beetles and their mutualistic fungi. This knowledge will allow better understanding of the potential impact of *Euwallacea* spp. and *Fusarium* spp. on California native hardwood species, agriculturally important crops, and allow for monitoring of beetle infestations throughout southern California. To this end, we had three main objectives for this study: (1) delimit the distribution of the invasive ambrosia beetles, *Euwallacea* spp., and their mutualistic *Fusarium* species on public and private land with ground surveys; (2) screen potential biocontrol agents and their efficacy at inhibiting the growth of *Fusarium* species on infested and non-infested trees for preventive control; and (3) evaluate the effectiveness of treating SHB-infested trees

## CHAPTER 14.

### Detection and Impact of an Invasive Insect-Disease Complex, the Polyphagous and Kuroshio Shot Hole Borers, *Euwallacea* spp., and Their Mutualistic Fungi in California (Project WC-EM-F-13-02)

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with *Beauveria bassiana*, *Metarhizium brunneum*, and *Isaria* sp. as potential entomopathogenic biological control agents.

## METHODS

### Objective 1: Delimit the Distribution of Ambrosia Beetles and Their Mutualistic Fungal Species

The research team visited areas near confirmed SHB infested sites, including public parks and city landscape areas in 2013–15. Beetle-susceptible tree species within these areas were observed for any visible symptoms of beetle attack (entry holes), and infestation status of each tree was entered into handheld GPS devices along with location, tree species, and estimated tree age. Wood samples were collected from any visible galleries and processed in lab to confirm the presence of their fungal symbionts. This process was repeated, radiating away from confirmed infestation sites. The GPS data of infested locations were compiled and the beetle distribution maps were generated using the ArcGIS application (ESRI 2014).

### Objective 2: Screen Potential Biocontrol Agents on Infested and Non-Infested Trees

A collection of plant endophytic microorganisms obtained from sapwood of sycamore and avocado was tested in the laboratory for antagonistic properties against *Fusarium* spp. Endophytic microorganisms were obtained from trees in beetle-infested locations that harbored relatively fewer numbers of beetle attacks compared to those immediately nearby.

All endophytic microorganisms obtained were screened for antagonistic properties against *Fusarium* spp. by using dual-culturing methods to simultaneously culture the endophytic microbes and *Fusarium* spp. on the same agar plates. Endophytes were placed along edges of potato dextrose agar (PDA) plates and nutrient agar plates containing agar plugs of *Fusarium* isolates in the centers of the plates. The plates were incubated at 25 °C for 5 days. Antagonistic microorganisms were visually identified by observing zones of inhibition in the *Fusarium* spp. colonies. The distances were compared to *Fusarium* colony growth in control plates, and the differences in colony growth were used to calculate the percentage of inhibition for each bacterial isolate against each *Fusarium* isolate.

### Objective 3: Evaluate the Effectiveness of Treating SHB-infested Trees with Foam Containing Entomopathogenic Fungal Spores

Three fungal species, *B. bassiana*, *M. brunneum*, and *Isaria* sp., were provided by Angela Payne (USDA-ARS). Fungi were maintained on PDA plates at 25 °C.

Adult female beetles were collected from infested box elder branches located at the Huntington Botanical Gardens (San Marino, CA). Infested branches were removed, split longitudinally, and sprayed with 95 percent ethanol to entice female beetles out of galleries. Emerging females were collected with a mouth-operated aspirator and stored in vials for *in-vitro* studies. Fungal spores from the entomopathogens were harvested from fresh



culture plates and initial spore concentrations were determined with a haemocytometer. Spore suspensions were diluted to a final concentration of about  $1 \times 10^5$  spores/ml.

To assess fungal entomopathogen efficacy, collected adult female beetles were placed into Petri dishes, and prepared fungal spore suspensions were applied to beetles using an atomizer. Control beetles were misted with sterile 0.01 percent TritonX-100 solution. Five beetles were used per treatment and two technical reps were performed for a total of 10 beetles per treatment. Following treatment application, beetles were transferred to sterile Petri dishes containing a moistened filter paper upon which an autoclaved wood disk of box elder approximately 2 inches in diameter was placed. Petri dishes were placed in an incubator at 25 °C in the dark, and beetle survival/mortality was monitored for 11 days.

#### **Objective 4: Evaluate the Effectiveness of Biocontrol Agents as Protection for Avocado Pruning Wounds**

From our previous surveys, we have observed that the beetle entry holes are most commonly found in branch collars on avocado. To protect the branch collars from further beetle infestation after pruning of infested branches, we tested applications of *Bacillus subtilis*, *B. bassiana*, vegetable oil, water (control), and no application (control). Vegetable oil was included in the study because it was found to reduce *Fusarium* spp. conidial germination in laboratory conditions in a preliminary experiment. The experiment

was set up in an infested grove; we selected and pruned 25 uninfested branches and applied one of the five treatments to each pruning surface randomly for a total of 5 replications with each treatment. The pruned branches were monitored each week for the appearance of new SHB attack and recorded at 0, 1, 2, 3, 7, and 11 weeks post-application.

## **RESULTS AND DISCUSSION**

### **Objective 1: Delimit the Distribution of Ambrosia Beetles and Their Mutualistic Fungal Species**

The current SHB distribution map containing the locations of PSHB and KSHB-infested areas shows two largely distinct areas of SHB infestation, a large area containing Los Angeles, Orange, Riverside, San Bernardino, and Ventura Counties, and KSHB infestation in San Diego County (fig. 14.1). The range of PSHB infestation in the greater Los Angeles area consists of heavy infestations in the Pasadena area extending northward to San Fernando Valley in Sun Valley, and westward to Brentwood. To the east of Los Angeles, the infestation distribution includes areas extending to Ontario in southwest San Bernardino County, Eastvale in Riverside County, and the western edge of Corona in Riverside County. In Orange County, the beetle has been found as far south as Aliso Viejo and San Juan Capistrano. Heavy infestation of KSHB has also been confirmed in El Cajon, the San Diego Zoo, Tijuana River Valley, eastern portions of San Diego, and in several commercial avocado groves in Escondido, Bonsall, and Valley Center.

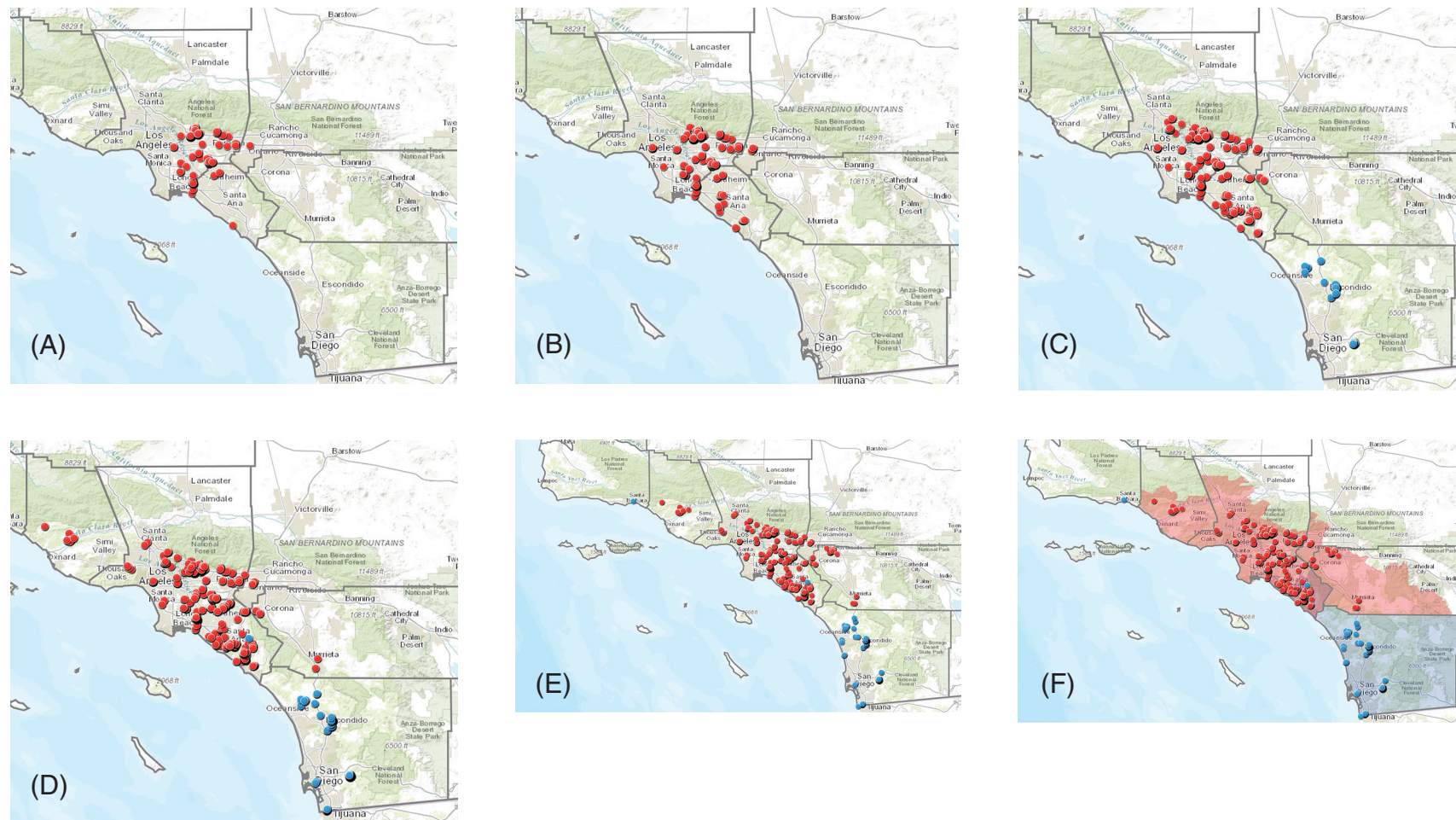


Figure 14.1—Distribution of polyphagous shot hole borers (PSHB) and Kuroshio shot hole borers (KSHB) infestation in California (A) as of 2012, (B) as of 2013, (C) as of 2014, (D) as of 2015, and (E) as of 2016. Red circles indicate infestation locations of PSHB, and blue circles indicate infestation locations of KSHB. Painted areas (F) indicate the possible infestation zones of P/KSHB in California.

## Objective 2: Screen Potential Biocontrol Agents on Infested and Non-Infested Trees

A total of 120 isolates of plant endophytic fungi and bacteria were isolated from xylem samples of sycamore and avocado. Among all endophytic microbes obtained in this way, 13 isolates of bacteria were found to form zones of inhibition in *Fusarium* spp. fungal colonies associated with SHB. None of the fungal isolates showed observable inhibition of *Fusarium* spp. The 13 bacterial isolates were identified as *B. subtilis* by PCR amplification and sequencing of the 16s ribosomal RNA gene.

Mean percent inhibition of *Fusarium* spp. isolates for the 13 bacterial isolates varied from 0.328 (32.8 percent) to 0.5938 (59.38 percent). Isolates E2, E6, E9, and E5 exhibited significantly more percent inhibition than the other bacterial isolates tested against *Fusarium* spp. associated with SHB ( $p < 0.05$ ) (fig. 14.2). There were no significant differences in mean percent inhibition between the four *Fusarium* isolates or between the two trials.

The discovery of these *Fusarium* spp.-inhibiting plant endophytic bacteria presents an opportunity to develop an effective injectable biological control treatment to prevent successful *Fusarium* spp. colonization from SHB inoculation in susceptible tree species. As *Fusarium* spp. serve as the primary source of food for SHB, the reduction or elimination of the fungus in bacteria-treated xylem tissues should lead to a decrease in future beetle colonization in treated trees.

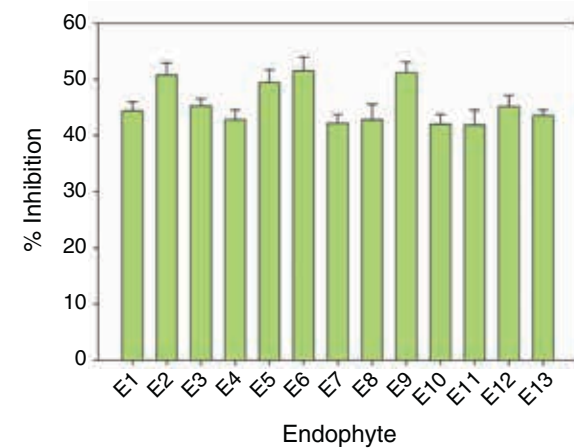


Figure 14.2—Percent inhibition of *Fusarium* spp. by endophytic bacterial isolates obtained from sycamore.

## Objective 3: Evaluate the Effectiveness of Treating SHB-infested Trees with Foam Containing Entomopathogenic Fungal Spores

The *in-vitro* application of *B. bassiana*, *M. brunneum*, and *Isaria* sp. showed that the beetles sprayed with *M. brunneum* exhibited the highest mortality rate. The onset of mortality in the treatment group that was treated with *M. brunneum* was the earliest, with the first death being observed 2 days after treatment. While the mortality rate in *M. brunneum*-treated beetles remained constant throughout the 11-day period, *B. bassiana*-treated beetles exhibited a sharp increase in mortality in one of the replicates toward the end of the experiment period. *M. brunneum*-treated and *B. bassiana*-treated groups each had 5 surviving beetles

11 days after treatment (fig. 14.3). All beetles treated with *Isaria* sp. were found to be dead 1 day after the treatment, but none of the dead beetles showed signs of mycosis. Therefore, the mortality data obtained from the *Isaria* sp.-treated beetles were not considered for this study.

#### Objective 4: Evaluate the Effectiveness of Biocontrol Agents as Protection for Avocado Pruning Wounds

Field application of the entomopathogen *B. bassiana*, antifungal biocontrol agent *B. subtilis*, and vegetable oil on branch collars showed that branch collars sprayed with *B. subtilis* had the lowest level of new colonization compared to all other treatments in preventive treatments (fig. 14.4). Interestingly, vegetable oil treatment seemed to attract more beetles to treated areas compared to control in preventive treatments of the branch collars.

### CONCLUSIONS

Surveys for SHB in California showed that the beetles have spread in urban forests throughout Orange, Los Angeles, Ventura, San Diego, Riverside, San Bernardino, Santa Barbara, and San Luis Obispo Counties, and several commercial avocado groves in San Diego, Los Angeles, and Ventura Counties. These results highlight the need for effective methods to prevent further spread of the beetles and

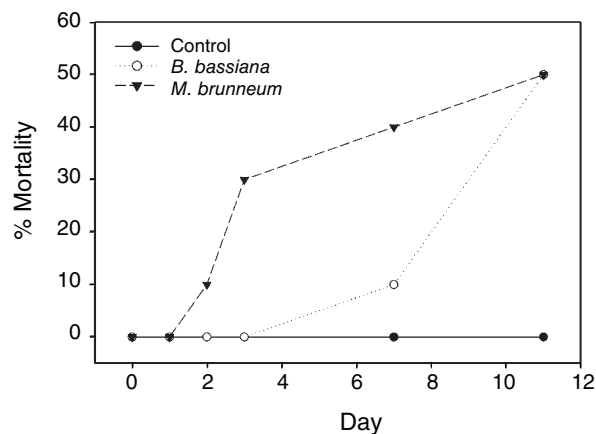


Figure 14.3—Beetle mortality over time after spray application with *Beauveria bassiana*, *Metarhizium brunneum*, or 0.01 percent Triton-X100 solution (control) in laboratory conditions.

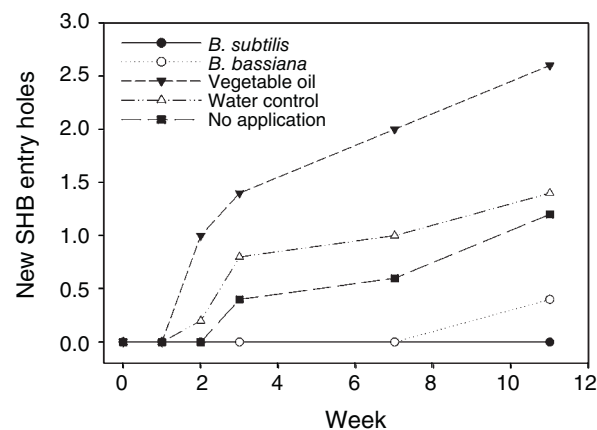


Figure 14.4—Mean new beetle galleries over time in branch collars of spray-treated trees. Pruning wounds were applied with *Bacillus subtilis*, *Beauveria bassiana*, vegetable oil, water, or no treatment.

their pathogenic symbiotic fungi. A potential treatment method using plant endophytic bacteria showed promise in laboratory experiments, and a study is currently underway to evaluate the treatment method's effectiveness in mature plants, as well as to expand the collection of *Fusarium* spp.-inhibiting plant endophytic bacteria. Results from a pruning wound protection experiment showed that *Bacillus subtilis* is a potential candidate biological control agent which can be utilized to protect pruning wounds from new SHB attack after branch removal. This may also have potential application for removal and treatment of infested branches in other affected hosts.

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**Potter, Kevin M.; Conkling, Barbara L., eds.** 2017. Forest health monitoring: national status, trends, and analysis 2016. Gen. Tech. Rep. SRS-222. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 195 p.

The annual national report of the Forest Health Monitoring (FHM) Program of the Forest Service, U.S. Department of Agriculture, presents forest health status and trends from a national or multi-State regional perspective using a variety of sources, introduces new techniques for analyzing forest health data, and summarizes results of recently completed Evaluation Monitoring projects funded through the FHM national program. In this 16th edition in a series of annual reports, survey data are used to identify geographic patterns of insect and disease activity. Satellite data are employed to detect geographic patterns of forest fire occurrence. Recent drought and moisture surplus conditions are compared across the conterminous United States. Data collected by the Forest Inventory and Analysis (FIA) Program are employed to detect regional differences in tree mortality. Change over time in the understory Vegetation Diversity and Structure Indicator is assessed on more than 500 FIA plots in the North Central and Northeastern States. Remeasured vegetation data across several Northern States are used to assess change over time in plant species diversity, occupancy and constancy. A new Regeneration Indicator, which includes a suite of tree-seedling and browse impact measurements, is described. The general magnitude of tree mortality predicted by the National Insect and Disease Risk Map was compared to FIA estimates of mortality. Six recently completed Evaluation Monitoring projects are summarized, addressing forest health concerns at smaller scales.

**Keywords**—Change detection, drought, fire, forest health, forest insects and disease, regeneration, tree mortality.

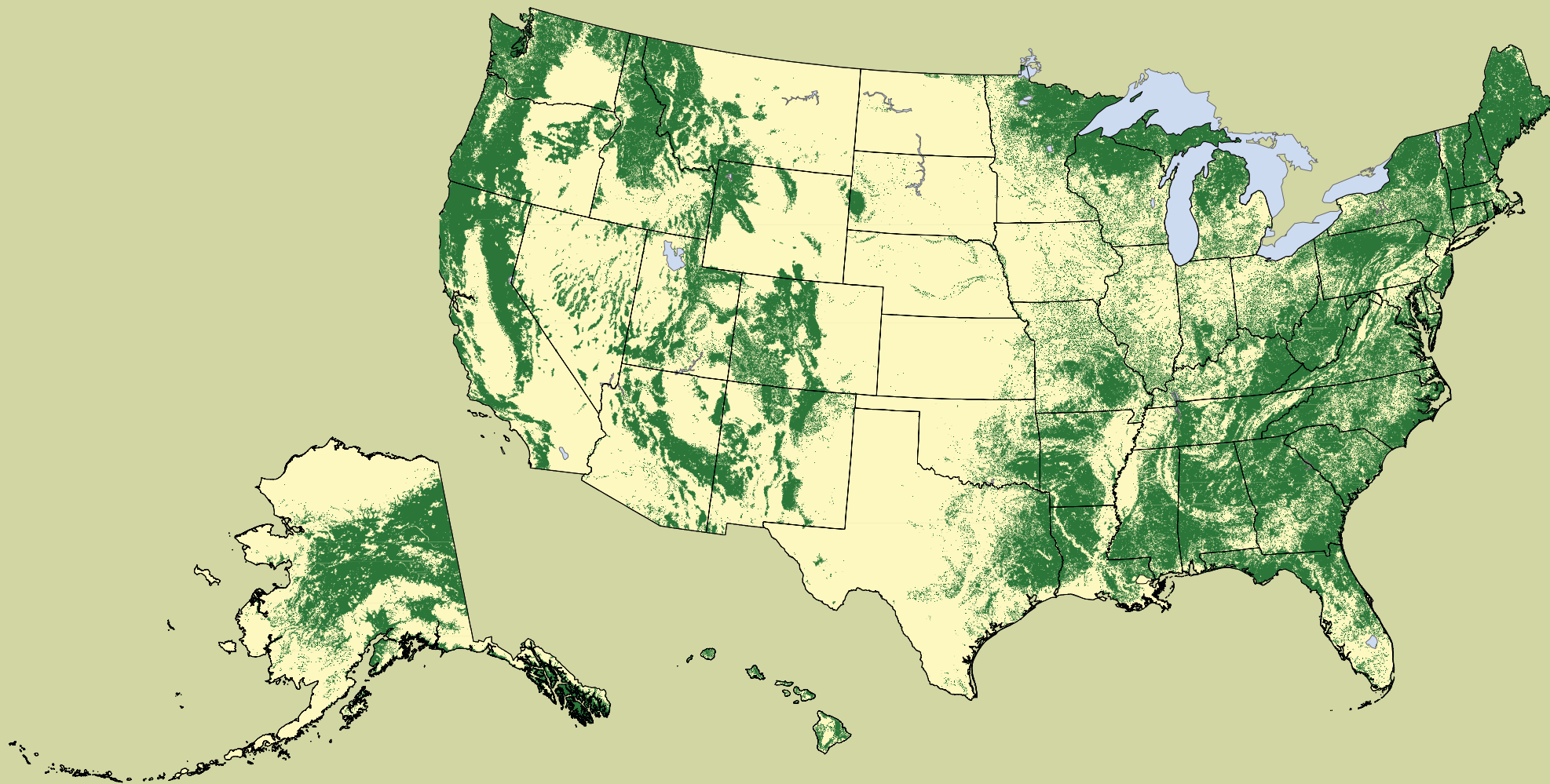


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