

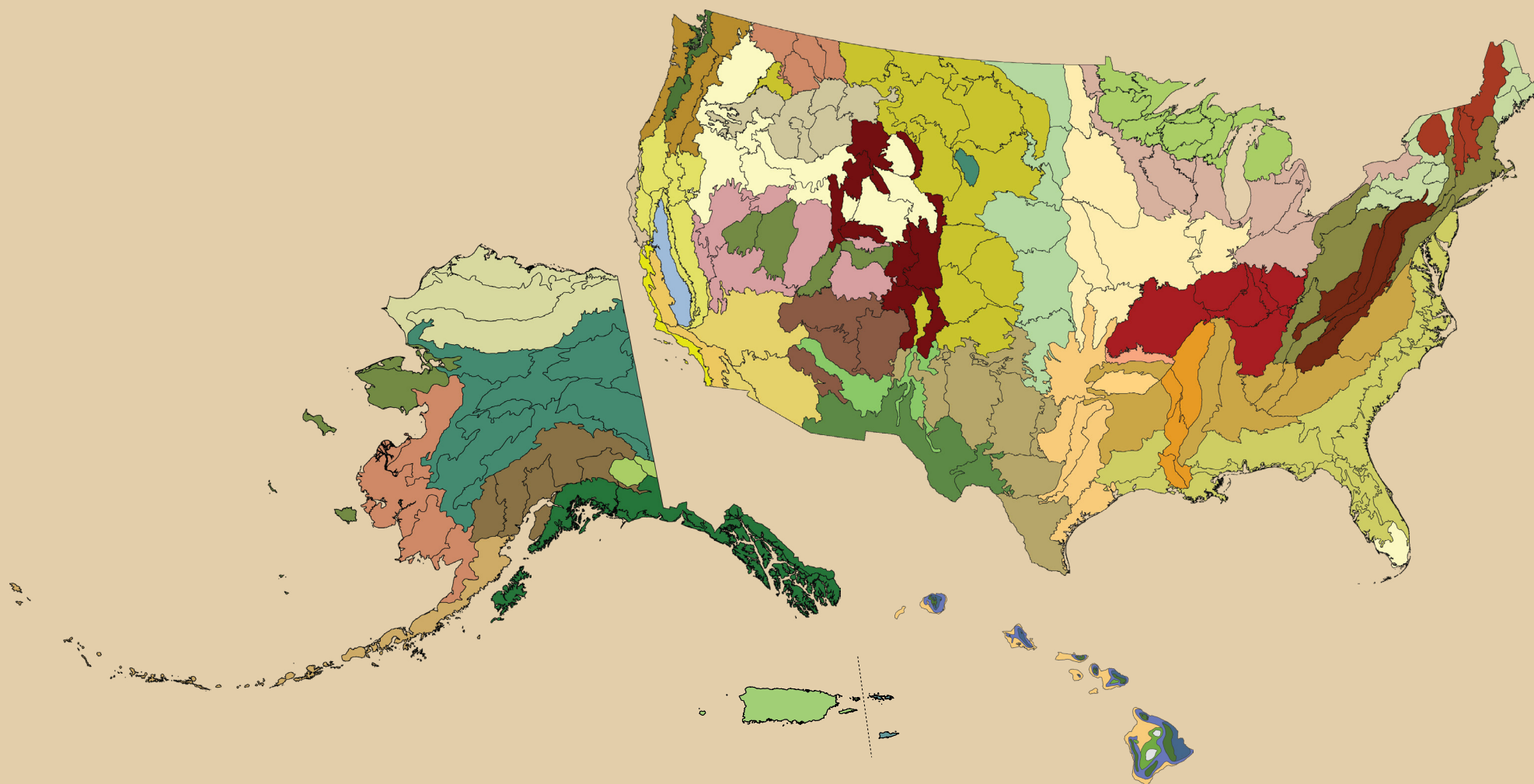


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Forest Health Monitoring: Evaluation Monitoring Project Summaries 2023

EDITORS: Barbara L. Conkling and Karun Pandit



FRONT COVER MAP: Ecoregion provinces and ecoregion sections for the conterminous United States (Cleland and others 2007) and for Alaska (Spencer and others 2002), and ecoregions within the islands of Hawaii (Potter 2023), along with Puerto Rico and the U.S. Virgin Islands, for which no corresponding ecoregion treatments exist.

BACK COVER MAPS: Tree canopy cover (green) for the conterminous United States, Hawaii, Puerto Rico, and the U.S. Virgin Islands based on data from a cooperative project between the Multi-Resolution Land Characteristics Consortium and the U.S. Department of Agriculture, Forest Service, Geospatial Technology and Applications Center using the 2011 National Land Cover Database (NLCD). Forest and shrubland cover for Alaska derived from the 2011 NLCD.

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Forest Health Monitoring: Evaluation Monitoring Project Summaries 2023

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The Forest Health Monitoring (FHM) Program of the U.S. Department of Agriculture, Forest Service annually funds projects under the Evaluation Monitoring (EM) component of FHM. Projects generally are intended to determine the extent, severity, and causes of undesirable changes in forest health. This component of FHM provides opportunities to delve deeper into observed forest health changes than is often possible as part of standard monitoring activities. When an EM project is completed, researchers are asked to provide a concise summary of the project. These summaries were previously (2008–2022) published as part of the FHM annual national status and trends report. Beginning in 2023, the summaries are being published in a separate document to highlight the EM projects and communicate concise results but will continue to be searchable on the FHM website reporting page (<https://www.fs.usda.gov/science-technology/forest-health-protection/publications-reports/forest-health-monitoring-publications>). This report includes summaries of nine recently completed EM projects that: evaluated the extent and severity of black cherry (*Prunus serotina*) regeneration declines in eastern forests and explored potential causes; investigated whether oak (*Quercus* spp.) regeneration and recruitment in eastern upland forests are being affected by forest health disturbances; studied an outbreak of *Diplodia sapinea* and *D. scrobiculata* that caused extensive dieback and mortality of slash pines (*Pinus elliotii*) in Florida; evaluated Carolina hemlock (*Tsuga caroliniana*) distribution and health in the Southern Appalachian Mountains;

applied Forest Inventory and Analysis data and site-specific factors to understand flowering dogwood (*Cornus florida*) decline in the Southeastern United States; evaluated distribution and impacts of invasive species on Southern U.S. forests; evaluated the health and abundance of five-needle pines (*Pinus* spp.) in southern Colorado and northern New Mexico; linked physiology to landscape-scale dieback of Utah juniper (*Juniperus osteosperma*) during an ongoing megadrought; and investigated whether spruce aphid (*Elatobium abietinum*) defoliation alters Engelmann spruce (*Picea engelmannii*) resistance to spruce beetle (*Dendroctonus rufipennis*).

Keywords—Black cherry, Carolina hemlock, diplodia tip blight, Engelmann spruce, five-needle pines, flowering dogwood, forest insects and disease, invasive species, oak, regeneration, spruce aphid, spruce beetle, Utah juniper.

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The national Forest Health Monitoring (FHM) Program of the U.S. Department of Agriculture, Forest Service 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. 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 (Potter and Conkling 2023).

One component of the FHM Program, Evaluation Monitoring (EM), is designed to fund projects to determine the extent, severity, and causes of undesirable changes in forest health identified through Detection Monitoring. Issues, therefore, are identified using data from aerial and ground surveys, as well as observations on the ground by Federal or State forestry professionals.

In this report, nine recently completed EM projects funded by the FHM Program address a wide variety of forest health concerns at a scale smaller than national-scale or multistate regional-scale analyses. These projects:

- Evaluated the extent and severity of declines in black cherry (*Prunus serotina*) seeding populations in eastern forests and explored potential causes using long-term datasets representing multiple spatiotemporal scales. Analyses of trends using those data found changes in black cherry tree and seedling populations, but the changes varied throughout

the range of the species in the Eastern United States. While correlations were found between black cherry and areas with high nitrate and sulfate deposition rates before 1990, researchers could not determine the changes in black cherry tree and seedling populations were caused by the atmospheric depositions documented in the datasets used. Further investigation using controlled experiments could contribute to more causal information useful to managers (ch. 2).

- Investigated if oak (*Quercus* spp.) regeneration and recruitment in eastern upland forests are being affected by forest health disturbances using Forest Inventory and Analysis (FIA) data. Data analysis resulted in the observation that, in broad terms, the number of new upland oak saplings occurring on sites several years after disturbances was not different than sites with harvesting or no disturbance. However, this study revealed a much lower number of new upland oak saplings per acre regardless of disturbance category than the estimates of sapling numbers required to likely avoid composition shifts in upland oak forests. Future investigations could include using data from longer timeframes and clarification of disturbance causes including specific pests (ch. 3).
- Investigated an outbreak of *Diplodia sapinea* and *D. scrobiculata* that caused extensive dieback and mortality of slash pines (*Pinus elliottii*) in Florida. The objectives were to evaluate the effects of nutrients in the outbreak, and to examine the role of nitrogen in disease severity using inoculated saplings in

EXECUTIVE SUMMARY

greenhouse studies. Samples from tree stands with different disease incidence underwent foliar analysis for 16 nutrients. Only nitrogen and molybdenum were significantly different. Nitrogen was found to be higher in stands with greater disease incidence. In the greenhouse studies, higher nitrogen levels were observed to increase disease effects in slash pine (*P. elliotii* var. *elliotii*) and south Florida slash pine (*P. elliotii* var. *densa*) that were inoculated with *D. sapinea* when compared to saplings with lower nitrogen. Saplings inoculated with *D. scrobiculata* did not show any significant differences among different nitrogen levels. Nitrogen management was suggested as an important management consideration for southern pines (ch. 4).

- Evaluated known and reported Carolina hemlock (*Tsuga caroliniana* Engelm.) populations throughout the species range to delineate and map the populations and assess their general health status as part of hemlock woolly adelgid (*Adelges tsugae* Annand) monitoring and genetic conservation efforts. Seventy-nine of 109 sites originally identified and visited supported living Carolina hemlock trees. Of those 79 populations, 74 were identified as having natural origin and the remaining 5 were identified as planted. While details of the health assessments are presented in the summary, the researchers concluded that most of the trees across the sites evaluated had poor to moderate health, indicating the importance of continued monitoring and management (ch. 5).

- Evaluated flowering dogwood (*Cornus florida* L.) mortality data from FIA along with data from a field study to understand the increasing dogwood mortality and population decline in the Southeastern United States, identify the factors that could be contributing to the decline, and investigate interactions among the factors. FIA data showed that flowering dogwood mortality increased significantly over several timespans (1979–1981, 1986–1989, 1999–2000, and 2008–2009). Causes identified in the FIA data were reported over the years of data analyzed and included silviculture/land clearing activities, vegetation impacts, diseases, and other individual agents such as weather, insects, fire, etc. The field data revealed that >70 percent of tagged flowering dogwood trees that were evaluated showed some level of dieback when initially assessed. At final assessment, most of the trees showed some stage of dieback with most showing >1/3 dead crown. Across all locations, about 7 percent were dead. The diverse species of fungi found at the field sites are documented in the summary. The researchers concluded that monitoring the interactions of the factors studied and their roles in tree decline is important in mitigating flowering dogwood decline and mortality in Southeastern U.S. forests (ch. 6).
- Evaluated the distribution and impacts of invasive plant species on forests in the Southern United States using geospatial and statistical modeling approaches and data from FIA. The analyses were divided into five sections: effects of environmental predictors on invasive species distribution; potential

distribution of invasive plants; landscape characteristics of predicted hotspots for plant invasions; relative importance of the biotic community, abiotic factors, and landscape characteristics on nonnative species richness and abundance; and impacts of invasive plants on tree regeneration. The results gave researchers a general idea of environmental effects on invasive species distribution, landscape susceptibility to invasion, and impacts of invasive plants on tree regeneration. Based on this research, the authors indicated the importance of incorporating information such as that presented in the summary into conservation strategies. Data that include a variety of environmental variables collected at different spatial scales are vital in considering invasive species distributions and susceptibility of the landscape to invasion (ch. 7).

- Evaluated the health and abundance of five-needle pines (*Pinus* spp.) in southern Colorado and northern New Mexico. Researchers collected baseline information on the distribution, abundance, and health of the pines; characterized stand structure and site conditions; and determined the extent and severity of white pine blister rust (WPBR) in the study area. The focus of the study was on previously unsurveyed areas where WPBR had not been detected. Results regarding the general health of surveyed trees revealed most surveyed five-needle pine trees were healthy; 90 percent of limber pines (*P. flexilis*) and 80 percent of Rocky Mountain bristlecone pines (*P. aristata*) had no or very minor damages. Researchers identified, however, a lack of

regeneration that was of concern. Results of 25 walk-through transects (not on formal plots) during the study revealed two new locations with WPBR. Periodic monitoring of likely disease spread was suggested (ch. 8).

- Evaluated Utah juniper (*Juniperus osteosperma*) using a multiscale approach spanning ecophysiology, dendrochronology, pathology/ entomology, and remote sensing to quantify the spatial patterns of juniper dieback, diagnose the drivers of the dieback, and understand the ecological functioning of pinyon-juniper woodlands during the megadrought in the Southwestern United States. This study was divided into three sections: detecting juniper dieback, underlying mechanisms of juniper dieback, and carbon-water cycling in pinyon-juniper woodlands. The data revealed widespread juniper dieback in southeastern Utah that followed severe drought in 2018 and numerous hotspots of severe dieback. The role of the megadrought in predisposing junipers to dieback, however, remains unknown. The researchers are conducting ongoing studies to elucidate these relationships and implications for the future (ch. 9).
- Investigated whether spruce aphid (*Elatobium abietinum*) defoliation alters Engelmann spruce (*Picea engelmannii*) resistance to spruce beetle (*Dendroctonus rufipennis*). The study was conducted in 2000–2021 during the multidecadal megadrought in the Southwestern United States with the endemic spruce beetle population. Researchers found that over 18 months after spruce beetle baiting, tree mortality was greater for trees with high

spruce aphid defoliation (40 percent) than for trees with medium or low defoliation (10 and 0 percent, respectively). When considering the data and the stated study caveats, results suggest that trees heavily defoliated by spruce aphid act as a sink for spruce beetle (ch. 10).

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 Evaluation Monitoring projects funded by FHM representing environmental investigations at scales smaller than the national scale 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 website at <https://www.fs.usda.gov/science-technology/forest-health-protection/monitoring>.

LITERATURE CITED

Potter, K.M.; Conkling, B.L., eds. 2023. Forest Health Monitoring: national status, trends, and analysis 2022. Gen. Tech. Rep. SRS-273. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 180 p. <https://doi.org/10.2737/SRS-GTR-273>.

THE NATIONAL FOREST HEALTH MONITORING PROGRAM

The national Forest Health Monitoring (FHM) Program of the U.S. Department of Agriculture, Forest Service 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 (Potter and Conkling 2023). 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 three major approaches (fig. 1.1), which are interlinked and provide feedback to each other:

- **Survey and Detection**—identifying forest health challenges at local and landscape scales through nationally standardized aerial and ground surveys
- **Monitoring and Analysis**—tracking the status and trends of forest health challenges and assessing impacts on forest resources and their benefits through synthesis and evaluation of information from various data sources within and external to the Forest Service
- **Communications**—supporting decision making and communicating regional, long-term forest health trends and their implications to forest resources and their benefits

EVALUATION MONITORING PROJECT SUMMARIES

Beginning in 2008, the annual “Forest Health Monitoring: National Status, Trends, and Analysis” report included a section comprising summaries of completed Evaluation Monitoring (EM) projects (e.g., Potter and Conkling 2023). Each summary included an overview of the project and results, citations for products and other relevant information, and a contact for questions or further information. The summaries introduced the kinds of monitoring projects supported by FHM and included enough information for readers to pursue specific interests.

The EM projects are funded by the FHM Program annually and are intended to determine the extent, severity, and causes of changes in forest health identified through Detection Monitoring (see the FHM website at <https://www.fs.usda.gov/science-technology/forest-health-protection/monitoring/>). The EM project proposals are submitted, reviewed, and selected through a process established by the FHM Management Team. Each year, the FHM Management Team reviews the EM proposal guidelines, application instructions, and priorities, culminating in updated proposal instructions and guidelines, new project proposal form, and progress and final reporting responsibilities being posted on the Forest Health Protection Funding Opportunities (<https://www.fs.usda.gov/science-technology/forest-health-protection/funding-opportunities/>).

CHAPTER 1

Introduction

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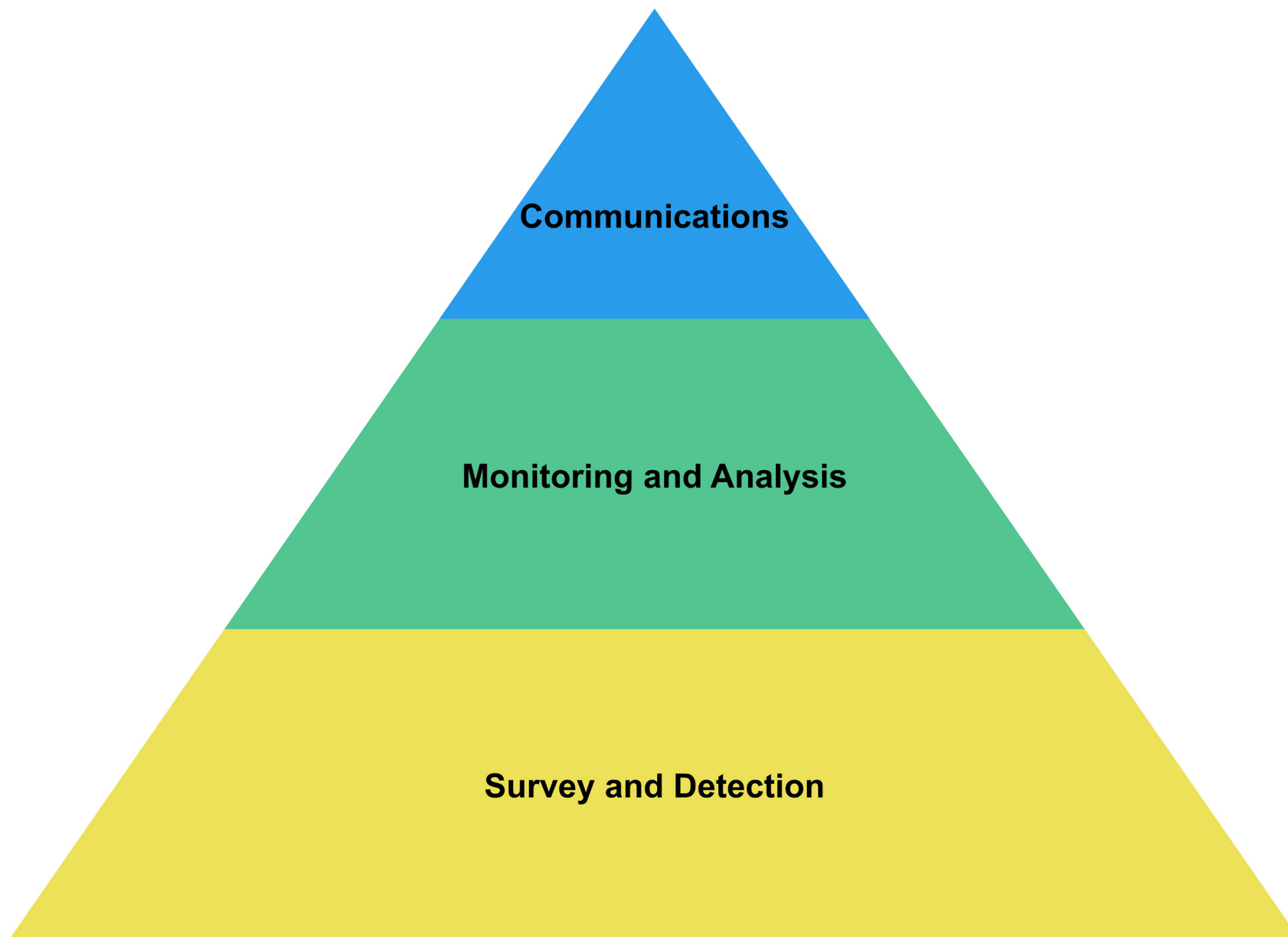


Figure 1.1—The design of the Forest Health Monitoring Program.

In 2023, the FHM Management Team decided to begin compiling the EM summaries each year into a report separate from the annual “Forest Health Monitoring: National Status, Trends, and Analysis” report. This allows FHM to continue documenting and highlighting results from the EM projects and provides an easier way to locate information about the EM component of the program. The EM project summaries will continue to be searchable on the FHM website reporting page (<https://www.fs.usda.gov/science-technology/forest-health-protection/publications-reports/forest-health-monitoring-publications/forest-health-highlights>).

This EM project summaries report is produced by FHM researchers at the Eastern Forest Environmental Threat Assessment Center (EFETAC) and EM project principal investigators across the United States in collaboration with North Carolina State University cooperators in the Forest Health Monitoring Research Group (<https://research.cnr.ncsu.edu/foresthealthmonitoring/>). 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 the EFETAC website (<https://forestthreats.org/about>).

LITERATURE CITED

Potter, K.M.; Conkling, B.L., eds. 2023. Forest Health Monitoring: national status, trends, and analysis, 2022. Gen. Tech. Rep. SRS-273. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 180 p. <https://doi.org/10.2737/SRS-GTR-273>.

INTRODUCTION

This project resulted from conversations between researchers and managers in Pennsylvania, both noting persistent deterioration in tree canopy vigor and seedling numbers for black cherry (*Prunus serotina*) in the Allegheny region, the epicenter of the species' commercial range. The Forest Health Collaborative subsequently identified black cherry decline as a top threat to Allegheny hardwood ecosystems (USDA Forest Service 2018a). Preliminary analyses of Forest Inventory and Analysis (FIA) data showed a ≈50-percent decline in seedling abundance from 2000 to 2017 in Pennsylvania (USDA Forest Service 2018b).

Royo and others (2021) examined the relative strength of available evidence of five factors known or assumed to influence forest health broadly and black cherry dynamics in the Allegheny region: climate variability, deer browsing, senescence, negative plant-soil microbial feedback loops (i.e., pathogens), and reduced nitrogen availability. The authors concluded that declines in black cherry health and regeneration were not temporary anomalies. Their review suggested that altered nitrogen deposition dynamics since the Clean Air Act Amendments of 1990 and strengthened negative soil-microbial feedback loops, particularly given a warmer and wetter climate, represented promising avenues for future research. In contrast, succession, senescence, and deer browsing may have an influence in localized instances but were not suspected to underlie the regional decline. Prior research has shown black cherry to be a nitrogen-demanding species during germination and

growth (Auchmoody 1979, 1982) and that it may accrue critical densities of pathogenic fungi that eventually limit regeneration (Johnson and others 2012, Reinhart and others 2005, Stanosz 1992) in dense stands. This study therefore posited that three mutually nonexclusive mechanisms may limit regeneration of black cherry: (1) advancing forest successional status, (2) changes in soil nutrition, and (3) intensified negative density dependence.

As an Evaluation Monitoring project funded under the U.S. Department of Agriculture, Forest Service, Forest Health Monitoring Program, this study had an overarching goal to provide insight into which locales may be at greatest risk of losing their future black cherry component. It proposed an examination of seedling dynamics, including the extent and severity of declining abundance, over 5–20 years at multiple spatial scales. The hypothesis was that seedling abundance declines would be pronounced in the Allegheny region, where black cherry has historically been most dominant, but less severe across the range of black cherry. Specifically, it was an investigation into whether the occurrence or magnitude of declining black cherry seedling abundance was localized or varied with latitude, varied with historic mature black cherry stocking, or varied geospatially with changing atmospheric deposition.

METHODS

Data From Forest Inventory and Analysis

This project relied on data collected as part of the FIA Program (<https://www.fia.fs.usda.gov/>). This national inventory has a long history of

CHAPTER 2

Evaluating the Extent and Severity of Black Cherry Regeneration Declines in Eastern Forests and Exploring Potential Causes

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HILLE

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data collection and is administered at the State level; however, the methodology has undergone numerous changes and refinements throughout that history, generally trending towards more uniformity across States with time. Disparities in methodologies across States limited the inferential potential of early data (ca. pre-1980) at broad scales and those data are not readily accessible by the public. Therefore, this project only used publicly accessible data, which generally spans ca. 1980 to present, covering a period when inventory methodologies became similar enough to allow broad spatiotemporal-scale investigations (though note that inventories prior to ca. 2000 still suffer from temporal incongruency, i.e., an entire State was inventoried in a single effort, but not all States were inventoried the same year). Around 2000, FIA adopted a nationalized and annualized inventory protocol in which the inventory was continuous in every State, i.e., a fixed percentage of inventory plots is measured every year in every State such that plots are remeasured every 5–7 years and, after the initial cycle, statewide inventories are updated yearly as the oldest data are replaced by the remeasured data.

The FIA Program collects data on permanent sample plots at known locations that are systematically distributed across all forested lands around the country such that each sample plot generally represents about 6,000 acres, though certain areas such as national forests are sampled with higher intensity. At each FIA sample location, one or more conditions, which include information on land/water, forest/nonforest, reserved/not reserved, stand type, ownership, and accessibility, are recorded. The tree data collected

include species, diameter at breast height (d.b.h.), status as live or dead, whether a dead stem is to be counted as a harvest removal or other mortality, and several other attributes, for all trees with d.b.h. ≥ 5.0 inches, on a cluster of nested subplots at each sample location (Bechtold and Patterson 2005). For saplings (d.b.h. = 1.0–4.9 inches), species, d.b.h., and location are consistently collected for live stems only (the fates of dead or harvested stems of this size are not tracked), and seedlings (d.b.h. < 1.0 inch and height ≥ 6.0 inches for softwoods or ≥ 12.0 inches for hardwoods) are simply tallied by species.

Data From the National Atmospheric Deposition Program

This study also relied on publicly available data from the National Trends Network of the National Atmospheric Deposition Program (NADP; <https://nadp.slh.wisc.edu/>). The National Trends Network is the largest long-term network for precipitation chemistry across the United States and originated in 1977. Basic data elements include measures of free acidity (H^+ as pH), conductance, calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^+), potassium (K^+), sulfate (SO_4^{2-}), nitrate (NO_3^-), chloride (Cl^-), bromide (Br^-), and ammonium (NH_4^+) and are made available on the NADP website. These basic data undergo further modeling and interpolation analyses to develop publicly available annualized continuous cover raster (grid) layers, or maps, for the various elements. Analyses used the raster layers for total rainfall, free acidity (H^+ as pH), calcium (Ca^{2+}), magnesium (Mg^{2+}), sulfate (SO_4^{2-}), nitrate (NO_3^-), and ammonium (NH_4^+), which spanned the years 1985–2018.

Analyses

Data from both FIA and NADP were summarized at various spatial scales including ecological subsections and sections (Cleland and others 2007), and across the entire Eastern U.S. range of black cherry (east of 100° W longitude). The range of black cherry was defined as any ecological subsection in the Eastern United States where the summarized black cherry density for stems with d.b.h. ≥ 1.0 inch was ≥ 1 stem per acre in the most recent FIA inventories from each

State (since the earliest forest inventories were less consistent across States). This study did not uncover any conspicuous, reliable differences in range extent across the time periods used (ca. 1990–2020). The ecological sections included in this range are depicted in figure 2.1.

Because of some temporal inconsistencies in methodology, this study could only examine population metrics such as density, frequency, basal area, size distribution, species composition, and importance value as far back as about 1990 and only for stems with a d.b.h. ≥ 1.0 inch (hereafter trees). Importance values for trees were calculated as the sum of relative stem density, relative basal area, and proportion of occupied plots in a given area (e.g., ecological section) for a given species divided by 300. Since stems with a d.b.h. < 1.0 inch (hereafter seedlings) were not consistently inventoried prior to the adoption of the nationalized and annualized inventory, only seedling data at the species range scale from around 2005 onward (varying by State) were analyzed.

RESULTS

As defined herein, the Eastern U.S. range of black cherry spanned 34 States and 80 ecological sections (fig. 2.1). At this scale, only modest change was observed in black cherry tree populations from ca. 1990 to 2020, and changes that were noted at this scale tended slightly positive for density, dominance, frequency, and importance metrics. However, at the same spatial scale, seedling populations decreased in all metrics from 2005 to 2020, including a 43-percent

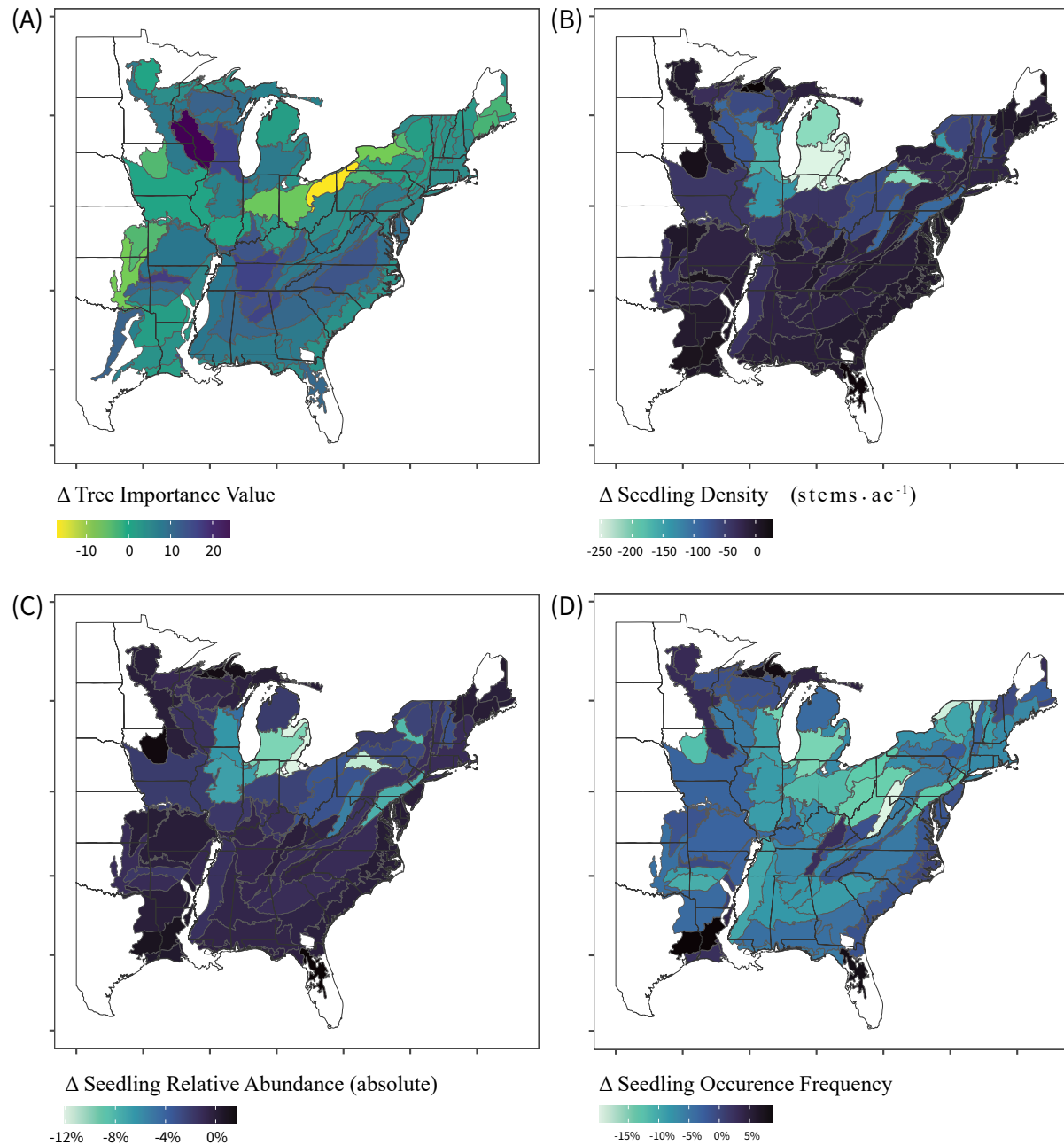


Figure 2.1 —(A) Net change in range-wide importance value from ca. 1990 to 2020 for black cherry trees (defined as stems with diameter at breast height (d.b.h.) ≥ 1 inch) summarized at the ecological section scale. (B) Net change in seedling density from ca. 2005 to 2020. (Seedlings are defined as stems with a total height ≥ 12 inches but d.b.h. < 1 inch.) (C) Net absolute change in seedling relative abundance from ca. 2005 to 2020. (D) Net absolute change in seedling occurrence frequency from ca. 2005 to 2020. Range includes ecological subsections where the species was present beyond a trace amount (tree density per acre ≥ 1) ca. 2020. Approximate year summarizes State-level Forest Inventory and Analysis data that were the closest temporal match to nominal year. Δ = total change from the latest time period to the earliest for each metric.

decline in overall abundance, 38-percent decline in relative abundance, and 28-percent decline in occurrence frequency (table 2.1). In the same study period, seedling abundances for all other species combined also decreased, but only about 8 percent, which further highlights the magnitude of the 43-percent change that was observed for black cherry.

In 1990, black cherry importance was highest in ecological sections within the Allegheny region bordering Lake Erie but not limited to a particular State or ecological section. Instead, the importance of black cherry to forests was evident across a broader region sometimes referred to as the “Rust Belt,” “Industrial Midwest,” or the “Great Lakes megalopolis,” which can include parts of Illinois, Indiana, Michigan, Minnesota, New York, Ohio, Pennsylvania, West Virginia, Wisconsin, Ontario, and Quebec in various combination. By 2020, the relative magnitude of importance between the Allegheny region and the rest of the Rust Belt region was more subdued, indicating that the Allegheny region had experienced a more severe decline in black

cherry importance from 1990 to 2020 (fig. 2.1A). The seedling population declined consistently in total density and relative abundance from ca. 2005 to 2020. A consistent decline trend of change was not detected along a latitudinal gradient, but the greatest declines again occurred in the Rust Belt region and, most prominently, in southern Michigan and northwestern Pennsylvania (fig. 2.1B). Reductions in seedling relative abundance largely mirrored those ecological sections where the density declined the most (fig. 2.1C). Decreasing occurrence of black cherry seedlings overall was widespread (fig. 2.1D).

Declines in black cherry tree and seedling populations were highly correlated with the prior importance value (ca. 1990) or seedling abundance (ca. 2005) of black cherry at the scale of ecological sections, i.e., losses were more extreme in the regions where it was once most prominent. The observed decreases in the absolute density of black cherry seedlings ($r = -0.64$), their relative abundance ($r = -0.71$), and their occurrence frequency ($r = -0.63$) were all magnified in regions with higher black cherry tree importance values.

Table 2.1—Rangewide^a population across 15 years^b for black cherry seedlings (height ≥12 inches)

Year	Abundance (stems/acre)	Relative abundance (% stems/acre)	Frequency (% plots)
2005	94.8	4.5	25.9
2020	53.9	2.8	18.6
Δ	-43%	-38%	-28%

Δ = percent change in metrics from 2005 to 2020.

^a Range includes ecological subsections where the species was present beyond a trace amount (tree density per acre ≥1) ca. 2020.

^b Approximate year summarizes State-level Forest Inventory and Analysis data that were the closest temporal match to nominal year.

The magnitude of recent declines in seedling relative abundance at the scale of the ecological section tended to increase in severity with higher levels of seedling relative abundance at the beginning of the monitoring period ($r = -0.87$).

Black cherry tree importance in 1990 was positively correlated with total nitrogen deposition ($r = 0.78$) and sulfate deposition ($r = 0.65$) as measured in the late 1980s (which approximates the era of peak atmospheric deposition) at the scale of an ecological section. At that time, evidence suggested that both nitrate and ammonium were positively correlated with black cherry importance, though nitrate- had a stronger correlation ($r = 0.72$ vs. $r = 0.56$) and broader range of inputs. By 2020, nitrate deposition rates decreased substantially and the correlation between black cherry importance and total nitrogen deposition was nearly halved. Similarly, the rates of sulfate deposition by 2020 were about one-third of 1990 levels and sulfate deposition was no longer correlated with black cherry importance in 2020. For black cherry seedlings, the ecological sections with the greatest reductions in relative abundance and frequency from 2005 to 2020 also tended to be the regions with the greatest flux in deposition rates since 1990 (fig. 2.2). Changing deposition rates for nitrogen and sulfate were correlated with reductions in relative abundance and frequency and, since no relationship with ammonium was found, nitrate is the more likely driver of any nitrogen-seedling effects.

CONCLUSIONS

The aim of this project was to evaluate the extent and severity of declines in black cherry seedling populations and to explore potential causes of this decline that threaten the sustainable management of this species. This study used statistical analyses of long-term datasets to examine trends and potential effects on seedling populations across multiple ecologically and statistically meaningful spatiotemporal scales. The results of this analysis showed evidence of changes in black cherry tree and seedling populations, but the occurrence and magnitude of those changes varied across the Eastern U.S. range of the species. Regions that experienced the highest magnitude of change tended to be those with historically high black cherry importance which, in turn, also tended to be regions that had high nitrate and sulfate deposition rates prior to 1990. In these regions, deposition rates of nitrate and sulfate have since declined as has the strength of their correlations with black cherry. The observational nature of this study could not determine that the changes observed in black cherry were caused by changes in atmospheric deposition quantity or composition, but the strength of correlations reported here suggest that controlled experiments to test that hypothesis are warranted. In the meantime, managers interested in promoting black cherry may face the most challenges in the very areas where black cherry has historically thrived.

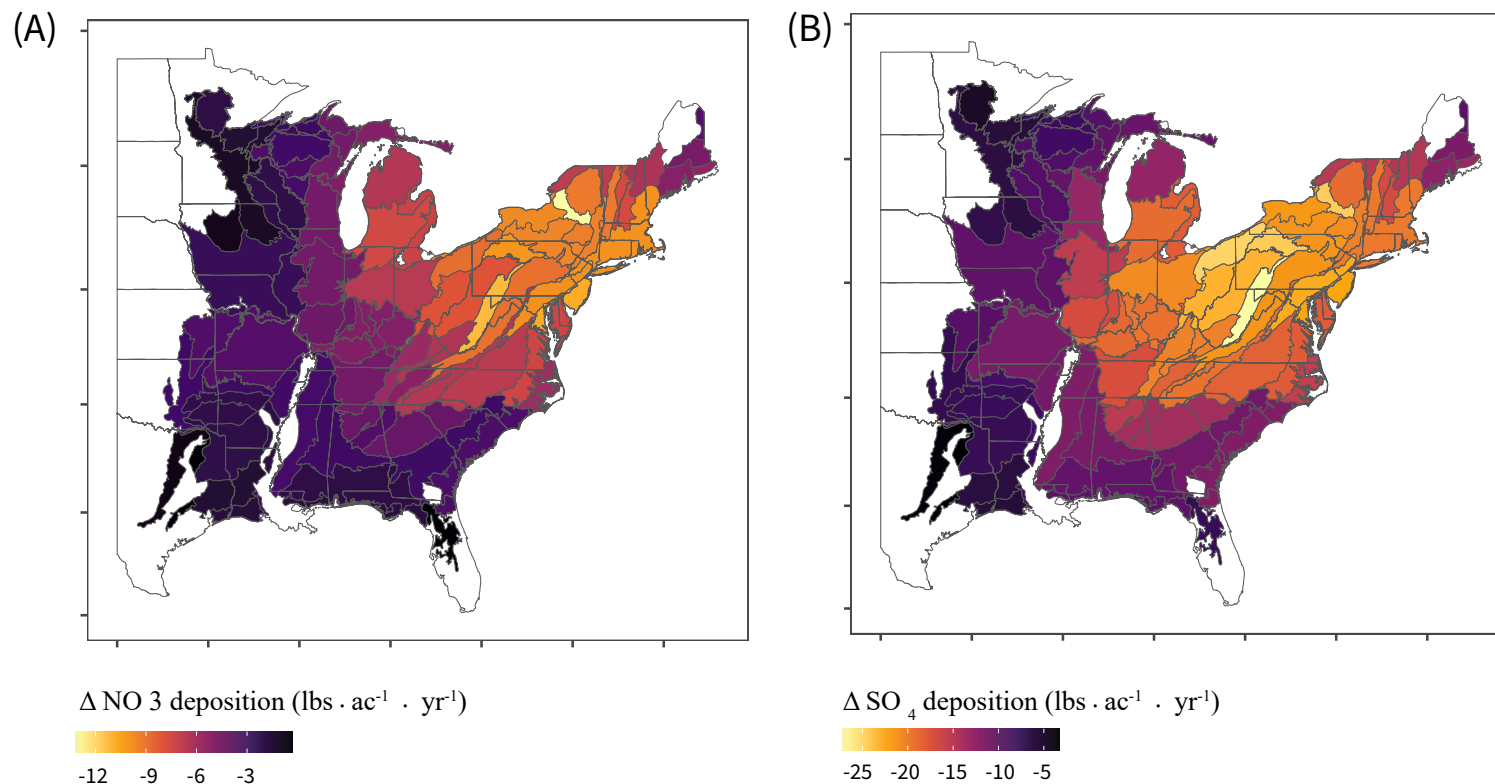


Figure 2.2 —Net change in rangewide atmospheric deposition of nitrate (A) and sulfate (B) from ca. 1990 to 2020 for black cherry summarized at the ecological section scale. Range includes ecological subsections where the species was present beyond a trace amount (tree density per acre ≥ 1) ca. 2020. Approximate year summarizes National Atmospheric Deposition Program data that were the closest temporal match to nominal year. Δ = total change from the latest time period to the earliest for each metric. Data retrieved from National Atmospheric Deposition Program (<https://nadp.slh.wisc.edu/>).

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INTRODUCTION

Oaks (*Quercus* spp.) are an important genus in the Eastern United States, comprising 11 percent of all living trees and 16 percent of all standing dead trees, and occurring on over 187 million acres (Miles 2017). Prominent upland oak species in these communities may include white oak (*Q. alba*), northern red oak (*Q. rubra*), chestnut oak (*Q. prinus*), black oak (*Q. velutina*), post oak (*Q. stellata*), scarlet oak (*Q. coccinea*), bur oak (*Q. macrocarpa*), southern red oak (*Q. falcata*), chinquapin oak (*Q. muehlenbergii*), northern pin oak (*Q. ellipsoidalis*), and blackjack oak (*Q. marilandica*), among others. Maintaining species composition when regenerating upland oak forests is often difficult, potentially requiring decades of active forest management planning (Dey 2014, Johnson and others 2019, Loftis and McGee 1993). This is because upland oaks, in particular, are strongly reliant on competitive advance reproduction to regenerate, which is often scarce, and their recruitment into dominant canopy positions following canopy disturbance is usually commensurate with the abundance of competitive advance reproduction (Brose and others 2008, Sander 1971). Recent broad-scale studies have documented a lack of “regeneration-ready” mature oak stands in the Eastern United States, evidenced by a lack of sufficient advance reproduction (McEwan and others 2011, Miller and McGill 2019, Vickers and others 2019a). The transition of oak forests to non-oak species is a widespread concern that appears to be driven, at least in part, by disturbance regimes that do not favor oak regeneration and recruitment (Fei and

others 2011, Larsen and Johnson 1998, McEwan and others 2011).

Recent studies have documented that mortality associated with forest pest invasions can lead to considerable loss of mature trees and reduced stand occupancy (Fei and others 2019). Throughout much of the Eastern United States, oak forest is relatively old, with at least 70 percent of the acreage approaching stages when regeneration considerations arise (Miles 2017, Shifley and others 2014). Untimely canopy mortality brought on by forest health issues may exacerbate regeneration difficulties and probably accelerates composition shifts (Appel 1995, Gottschalk 1993, Heitzman and others 2007). Recognizing that understory dynamics influence future forest composition, health, and sustainability, this project aimed to evaluate the influence of disturbance-induced oak canopy reductions on understory demographics in upland oak forests across the Eastern United States.

This study was an Evaluation Monitoring project funded under the U.S. Department of Agriculture (USDA), Forest Service, Forest Health Monitoring Program with an overarching goal to yield a better understanding of regeneration dynamics during disturbance-induced canopy reductions and help triage oak forests for regeneration risks. It proposed to examine seedling and sapling dynamics through time at broad spatial scales to examine whether forest health issues in eastern upland oak forests affect oak regeneration and recruitment by addressing two specific questions:

CHAPTER 3

Do Forest Health Issues in Eastern Upland Oak Forests Affect Oak Regeneration and Recruitment?

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- (1) Do forest health disturbances affect upland oak regeneration density (seedlings and saplings)?
- (2) Do forest health disturbances affect upland oak sapling recruitment (sapling ingrowth)?

METHODS

Data Source

This project relied on data collected as part of the Forest Service's Forest Inventory and Analysis Program (FIA; <https://www.fia.fs.usda.gov/>). FIA collects data on permanent sample plots at known locations that are systematically distributed across all forested lands around the country such that each sample plot generally represents about 6,000 acres, though certain areas such as national forests are sampled with higher intensity. At each FIA sample location, one or more conditions—which include information on land/water, forest/nonforest, reserved/not reserved, stand type, ownership, and accessibility—are recorded. The tree data collected for all trees with diameter at breast height (d.b.h.) ≥ 5.0 inches include species, mapped location on the plot, d.b.h., status as live or dead, whether a dead stem is to be counted as a harvest removal or other mortality, the likely cause of death, and several other attributes on a cluster of nested subplots at each sample location (Bechtold and Patterson 2005). For saplings (d.b.h. = 1.0–4.9 inches), species, d.b.h., and location are consistently collected for live stems only (the fates of dead or harvested stems of this size are not tracked). Seedlings (d.b.h. < 1.0 inch and height ≥ 6.0 inches for softwoods or ≥ 12.0 inches for hardwoods) are simply tallied

by species. In addition to various tree-centric measures, inventories record several site attributes such as ownership group, forest type, stand size/maturity, visible signs of disturbances including forest health and harvesting operations, and more ancillary site attributes to help describe the condition (Burrill and others 2018).

Data Filtering

FIA has a long history of data collection and is administered at the State level. Around 2000, FIA adopted a nationalized and annualized inventory protocol in every State in which the inventory was continuous, i.e., a fixed percentage of inventory plots is measured every year in every State such that plots are remeasured every 5–7 years and, after the initial cycle, statewide inventories are updated yearly as the oldest data are replaced by the remeasured data. At the time of data acquisition, complete inventories were available from ca. 2000 through 2018 across the Eastern U.S. study region (forest land portions of 37 States east of 100° W longitude). Analyses were limited to inventories that included a complete seedling count, which was not consistent across the Eastern United States before ca. 2005; thus, the actual timeframe of these analyses within the approximate 2000–2018 temporal window varies slightly by State within the study region. Within this timeframe, this study collated the first three inventories (t1–3) with full seedling counts for each plot location. Any plot locations without at least three consecutive inventories recorded were discarded.

Locations with mature, upland oak characteristics that did not show signs of recent disturbance during the first of the three consecutive inventories (hereafter, t1) were identified. Mature characteristics included the following plot-level characteristics: a preponderance of large or medium tree diameters (STDSZCD = 1 or 2), an average (quadratic) tree diameter ≥ 7 inches, and a recorded stand age ≥ 50 years old (STDAGE). Upland oak characteristics included plots that had at least 10 percent of living plot-level stocking (sensu Gingrich 1967) comprising upland oak species (as defined in Introduction) but were not located on hydric sites or floodplains/bottomlands (PHYSCLCD = 11–23, 29). Evidence of recent disturbance included: disturbances or treatments recorded (DSTRBCD1 > 0 or TRTCD1 > 0), artificial regeneration recorded (STDORGCD = 1), canopy stocking < 60 percent, and a standing-dead basal area more than one standard deviation greater than the average across all mature upland oak plots in the study region.

The second of these consecutive inventories (hereafter, t2) categorized the previously undisturbed, mature upland oak plot locations based on which tree-level cause of death (AGENTCD), if any, affected the most basal area since t1 (the previous measurement, ≈ 5 –7 years earlier). Analyses were limited to plot locations that fell into one of the following four disturbance categories: insect, disease, harvesting, or undisturbed. Plot locations that experienced multiple types of disturbance were discarded. A total of 3,302 plot locations within the data available for the period of interest met all the t1

criteria above, of which 97 were categorized as insect disturbances at t2, 335 were categorized as diseased, 326 were categorized as harvested, and 549 were categorized as undisturbed. This same dataset was used to address the two research questions stated above. Data were retrieved via the rFIA package for R statistical software (Stanke and others 2020), and all analyses were conducted with R statistical software (R Core Team 2020).

Data Analyses

The data selection process described above enabled a “natural experiment” approach in analyzing this observational data where all plot locations began the series as undisturbed mature upland oak forests (t1), encountered one of the disturbance types by the second inventory (t2), and were examined for their responses to those disturbances at the third inventory (t3). Because of variation in State-level sampling schedules, the time since disturbance can range from 5–14 years or be ongoing in the case of a single disturbance type that continued from t2 to t3.

To address the first question (Do forest health disturbances affect upland oak regeneration density?), a negative binomial generalized linear model compared the combined density of upland oak seedlings and saplings at t3 between all disturbance categories with the harvesting and undisturbed categories serving as disturbed and undisturbed “controls,” respectively. To address the second question (Do forest health disturbances affect upland oak sapling recruitment?), a negative binomial generalized linear model compared the combined

density of post-disturbance sapling ingrowth ($t_2 + t_3$) between all disturbance categories with the harvesting and undisturbed categories serving as disturbed and undisturbed “controls,” respectively.

RESULTS

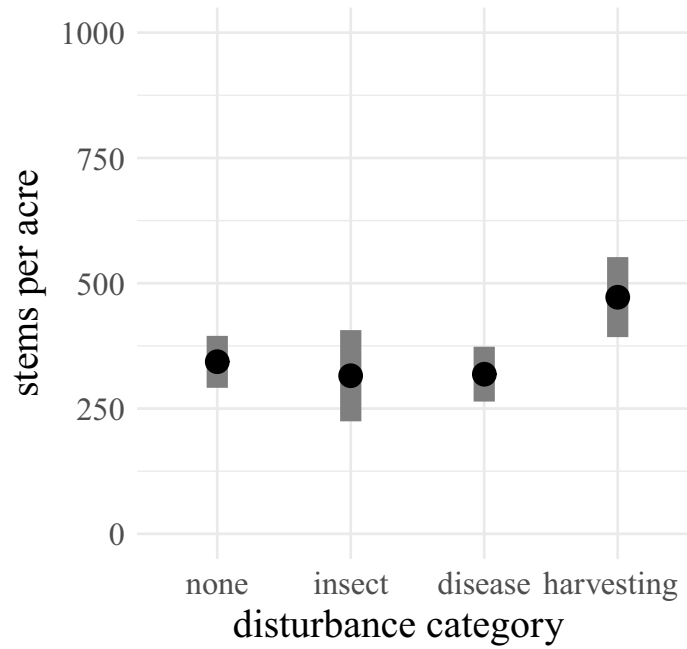
This study found little evidence of direct effects from forest health disturbances on upland oak regeneration and recruitment. On average, more upland oak seedlings and saplings regenerated following harvesting than other types of disturbance ($p = 0.0043$), while the densities of upland oak regeneration following insect, disease, and undisturbed categories were statistically indistinguishable (fig. 3.1A). No statistical differences ($p = 0.6917$) were found in new sapling ingrowth following any of the disturbance types examined (fig. 3.1B). Upland oaks (fig. 3.1A) comprised 10–15 percent of all combined seedlings and saplings across species regardless of disturbance category, while upland oak sapling ingrowth (fig. 3.1B) comprised 5–10 percent of sapling ingrowth.

DISCUSSION

Forest health issues often create a canopy disturbance that releases understory trees. In oak stands where oaks are absent or sparse in the understory, untimely disturbance or canopy disruption may initiate composition shifts by releasing non-oaks. Many times, focused silviculture based on timely release of existing advance reproduction is advocated to improve this condition (Brose and others 2008, Johnson and

others 2019, Loftis 2004). Broad-scale studies have highlighted potential oak regeneration problems given current understory conditions across vast acreages (Miller and McGill 2019, Vickers and others 2019a), but this landscape-scale concern is not regarded as a conventional forest health issue. Collectively, the results of these analyses suggest that forest health-related disturbances, in broad terms, resulted in about as many new upland oak saplings several years following disturbance as did harvesting or no disturbance. However, the 15–25 new upland oak saplings per acre that developed in this study, regardless of disturbance category (fig. 3.1B), fall short of even the lowest estimates of sapling recruitment required to plausibly avoid composition shifts in upland oak forests (86–130 per acre; Vickers and others 2019b, 2019b) as well as the 60–70 saplings per acre estimated as necessary to simply maintain the economic viability of the oak component in such stands (Brose and others 2008, Miller and others 2007, Ward 2009). Moreover, the 5–10-percent average upland oak component of sapling ingrowth by t_3 regardless of disturbance category represents a considerable regeneration debt (sensu Miller and McGill 2019) compared to the nearly 50-percent average upland oak component in the upper canopy of these plots at maturity prior to disturbance (t_1). Thus, forest health disturbances appear to be forcing an untimely initiation of the regeneration process in stands that are not poised for successful oak regeneration. This, in turn, accelerates the widespread phenomena of composition shifts in oak stands, not by negatively impacting oak seedling and sapling

(A)



(B)

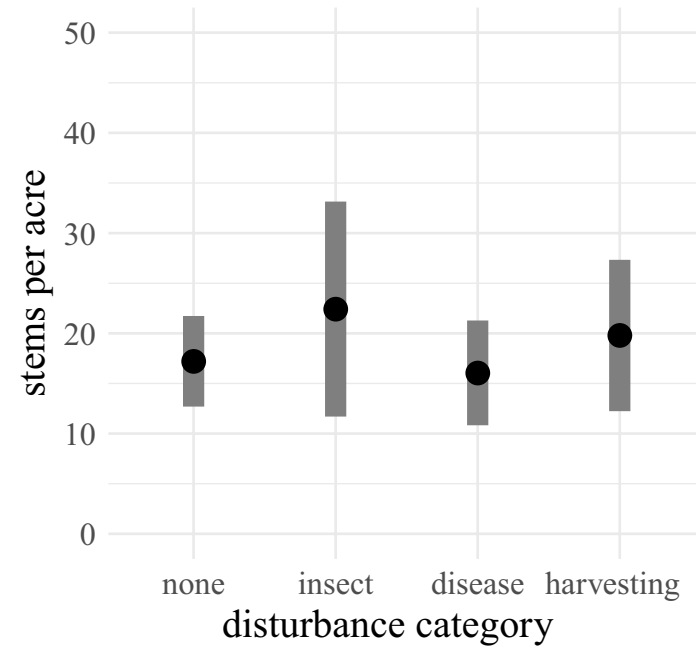


Figure 3.1 —Density of post-disturbance upland oak seedling and sapling regeneration (A) and upland oak sapling ingrowth (B) by disturbance category across 37 Eastern U.S. States. Points depict average values and error bars depict 95-percent confidence intervals.

populations more so than any other kind of disturbance examined in this study, but instead by releasing competing species in the understory in stands without adequate upland oak advance reproduction.

One weakness of this study is that it could only examine forest health issues as detected by new tree deaths recorded in the inventory record. There are likely instances where canopy trees do not exhibit widespread mortality but nonetheless are experiencing forest health-related disturbances that alter canopy health, which subsequently influences understory populations either directly or indirectly through altered light levels and other ways. While FIA does record certain aspects of tree and canopy health on a subset of plots (Schomaker and others 2007), the sample sizes are relatively sparse (Randolph and others 2010) and the attributes collected and protocols used have changed through time (Burrill and others 2017, Woudenberg and others 2010), which may complicate potential inferences that may be attainable from such data. Future research directions would ideally include longer data timeframes and better attribution of disturbance including specific types of pests. This will become increasingly possible as the latest FIA protocols are implemented and plots are remeasured (Burrill and others 2017). The potential incorporation of data from USDA Animal and Plant Health Inspection Service (APHIS) and possibly other datasets that document the range, prevalence, and occurrence of various pest outbreaks, often at the county level, is another potential research avenue to explore.

ACKNOWLEDGMENTS

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INTRODUCTION

Abiotic stressors such as drought, tree wounding, nutritional imbalances, and/or improper management practices generally predispose trees to disease outbreaks caused by endemic pathogens. These factors can weaken trees, increasing their susceptibility to infection by pathogens (Ayres and Lombardero 2000). Nutrient imbalances and excessive fertilizer applications have been implicated as major predisposing factors in certain tree disease outbreaks (Blakeslee and others 1999, Panique and others 1997, Swart and others 1987). Of all plant nutrient imbalances, excessive nitrogen is most associated with elevated disease incidence and severity (Lopez-Zamora and others 2007). Numerous studies have demonstrated the predisposition of trees to outbreaks in relation to nitrogen fertilization rates (Vandijk and others 1992). An excess of nitrogen induces succulent shoot and foliar growth, which decreases lignin concentration and facilitates wounding from insects, hail, or wind (Hesterberg and Jurgensen 1972), as well as decreases the concentration and toxicity of phenols used by plants against pathogens (Torres-Olivar and others 2014). For example, high levels of foliar and soil nitrogen are known to enhance disease severity of pitch canker (*Fusarium circinatum*) in pines (*Pinus* spp.; Lopez-Zamora and others 2007), as well as fire blight (*Erwinia amylovora*) in pear (*Pyrus* spp.) and rust (*Puccinia graminis* f. sp. *Tritici*) on wheat (*Triticum* spp.; Agrios 2005). Likewise, pine trees in the Netherlands severely affected by *Diplodia sapinea* showed approximately 60 percent more nitrogen than healthy trees (Vandijk and others 1992).

This study focused on an outbreak of *D. sapinea* and *D. scrobiculata*, which caused extensive dieback and mortality of slash pines (*P. elliotii*) along a highway in central Florida starting in 2012 (Paez and Smith 2018). The presence of an unusually large and uniform outbreak led to concern about what factors were responsible in the development of the outbreak. The objectives of this study were to (1) examine the effect of nutrients in this outbreak and (2) investigate the role of nitrogen in disease severity using inoculated saplings.

METHODS

Field Foliar Nutrient Analysis

In the summer of 2014, live foliar samples were taken from three slash pine stands in the outbreak area in central Florida. Each stand was categorized into healthy (<10 percent of trees with symptoms), medium (25–50 percent), or high (>50 percent) disease incidence of diplodia tip blight. Each sample was dried to a constant weight in a conventional oven at 55 °C for 2 days. The dry samples were sent to Micro Macro International Laboratory (MMI Labs, Athens, GA) to be ashed (AOAC 900.02B) and analyzed for nutrient analysis using inductively coupled plasma (ICP-EPA 6010a; MMI Labs, Athens, GA). Foliar nutrient analysis was compared among the three levels of disease incidence. These comparisons examined potential relationships between nutrient imbalances and diplodia disease incidence in the field.

CHAPTER 4

Etiology of Diplodia Tip Blight Outbreak on Slash Pine in Florida

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SMITH

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Effect of Nitrogen on Disease Severity

Under controlled conditions, one greenhouse study of loblolly pine (*P. taeda*), slash pine (*P. elliotii* var. *elliotii*), and south Florida slash pine (*P. elliotii* var. *densa*) was performed twice to evaluate the effect of different levels of nitrogen fertilization on disease severity of diplopedia tip blight caused by the two fungal pathogens (*D. sapinea* and *D. scrobiculata*). Each experiment was designed as a randomized complete block design, with three blocks. The treatments were arranged in a factorial design with fungal species (*D. sapinea*, *D. scrobiculata*, and mock-inoculated control) and three nitrogen fertilization rates, for a total of nine treatments.

The fertilization treatments were preceded by a base application of Nutricote® (18:6:8; Florikan®) at the recommended dose, based on operational silvicultural commercial standards (Jokela 2004) downsized for the surface area and volume of the tree pot. The fertilization treatments were then imposed after a month and were applied as (1) no added nitrogen (0X N), (2) twice the recommended nitrogen dose (112.1 kg/ha; 2X N), and (3) three times the recommended nitrogen dose (168.1 kg/ha; 3X N). The nitrogen source was urea (46:0:0) and the dose was estimated for the pot size (11.4 L) used in the study. Stems of 3-year-old loblolly pine, slash pine, and south Florida slash pine saplings were wounded and inoculated using a 6-mm sterile cork borer at approximately 25 cm below the terminal shoot tip. A 6-mm colonized agar plug (from the marginal growth of each in vitro fungal culture) was inserted in the wounded area facing the cambial tissue of the stem and wrapped with Parafilm® M (Pechiney Plastic).

For mock-inoculated controls, the inoculation was made with a sterile (noncolonized) potato dextrose agar plug (full strength) wrapped with Parafilm® M (Pechiney Plastic). Parafilm was not removed until the end of the study. External symptoms and internal lesion lengths were evaluated 11 weeks after inoculation. Lesion length was evaluated by removing the bark with a sterile scalpel blade to expose discolored sapwood and all bark surrounding the healthy tissue on margins of the discolored tissue. The lesions were then measured in centimeters. To complete Koch's Postulates, the inoculated *Diplopedia* strains were reisolated from the discolored tissues and identification was confirmed by internal transcribed spacer (ITS) sequencing.

Statistical Analysis

To determine if differences existed among the levels of macronutrients and micronutrients in the slash pine stands, a Kruskal-Wallis nonparametric one-way analysis of variance (ANOVA) test was used with the disease incidence level (healthy, medium, or high) of the sampled trees as the independent variable. A Steele-Dwass nonparametric multiple comparison procedure was used to test which incidence level differed among the stands.

To analyze lesion lengths in pine saplings treated with different nitrogen levels, a two-factor linear mixed model was used to test for differences, with fungi (*D. sapinea* and *D. scrobiculata*) and nitrogen level (0X N, 2X N, and 3X N) set as the independent variables. A mixed model was used to account for the blocks in the experimental design and for variability between the fungi treatments. After performing the analysis, Tukey's multiple

means comparison honestly significant difference (HSD) was conducted when the lesion length was significant. The significance was evaluated at $p \leq 0.05$. All analyses were performed using the statistical software SAS version 14.

RESULTS AND DISCUSSION

The foliar analysis of samples from tree stands with different disease incidence indicates that among 16 nutrients, only nitrogen and molybdenum were significantly different (fig. 4.1). Mean nitrogen levels were significantly higher in plots exhibiting high disease incidence (~2.3 percent) versus plots with healthy trees (1.2 percent) and those showing a medium disease incidence (1.5 percent) (fig. 4.1A). The level of foliar nitrogen in the high disease incidence plots (~2.3 percent) is considered high (nearly double) when compared to published values for healthy slash pine trees (Jokela 2004). This was expected, as the nitrogen applied in these plots by highway maintenance crews—ranging between 55 and 142 kg/ha (based on 1,568–3,913 trees/ha) every year for 4 years—was much higher than what is recommended in commercial forestry plantations using the same species (45–56 kg/ha of nitrogen at time of planting only (Jokela 2004). Numerous factors are known to cause stress to pine trees, including drought, competition, and overfertilization (Bachi and Peterson 1985, Blodgett and others 1997, Johnson and others 1997, Nicholls and Ostry 1990, Paoletti and others 2001).

In the greenhouse experiment, higher nitrogen levels (3X N) significantly increased disease effects

in slash pine and south Florida slash pine when inoculated with *D. sapinea* compared to those with low nitrogen levels (0X N). However, the nitrogen level did not cause a severity difference with any pathogens in inoculated loblolly pine (fig. 4.2). When inoculated with *D. scrobiculata*, no significant differences in disease severity were found among the different nitrogen levels for any tree species ($p > 0.05$). Previous studies show that if environmental conditions become unfavorable for pines (causing host stress), *D. sapinea* turns pathogenic and moves from the bark to living internal tissues (Flowers and others 2001, Smith and others 2002). Also, previous research has shown that nitrogen increases the susceptibility of pines to *D. sapinea* infection (Blodgett and others 2005).

CONCLUSIONS

As landscapers use native trees in diverse, often very intensively maintained landscapes, more education about the effects of specific stresses created in these situations, such as excessive nitrogen, is needed. Some forestry and horticulture practitioners tend to adopt a philosophy that if the goal is fast plant growth, then the method to achieve this goal is to apply more fertilizers, especially nitrogen. This study highlights a case analysis providing detailed evidence of the potential risk to a specific pathogen resulting from overfertilization. In future management of the southern pines, this risk could be considered by examining nutrition management in relation to disease outbreaks.

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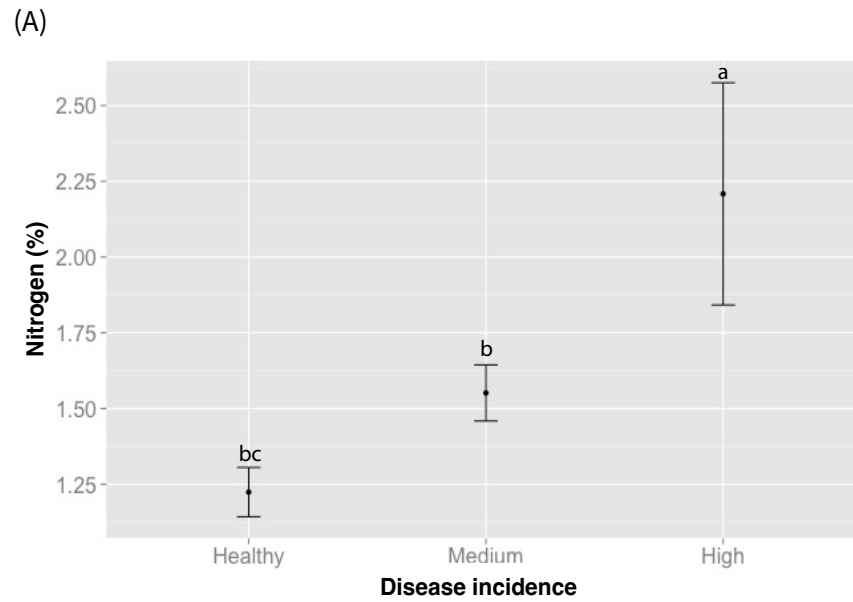
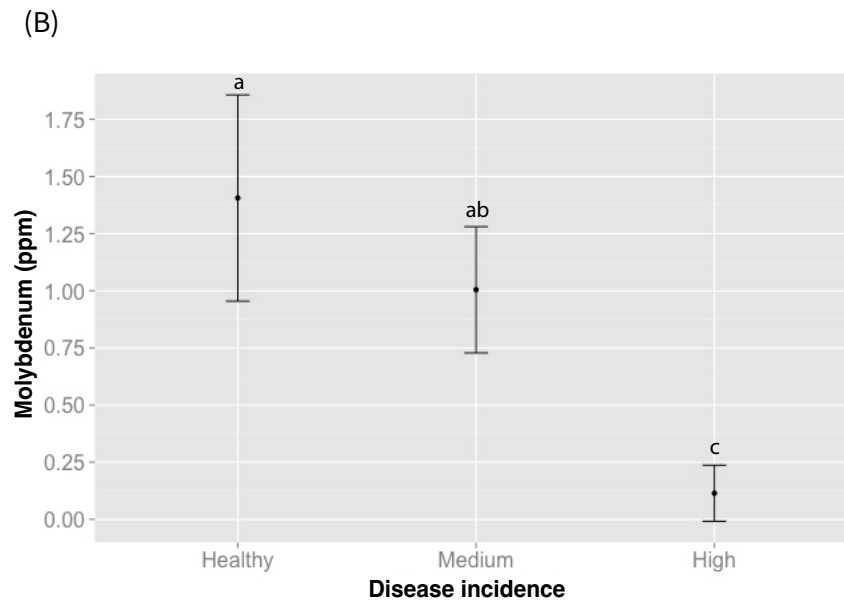


Figure 4.1— Foliar nitrogen concentration (percent) (A) and foliar molybdenum concentration (parts per million) (B) of slash pine trees from stands with different diplodia disease incidence (Paez 2017). Different letters indicate statistical significance ($p \leq 0.05$).



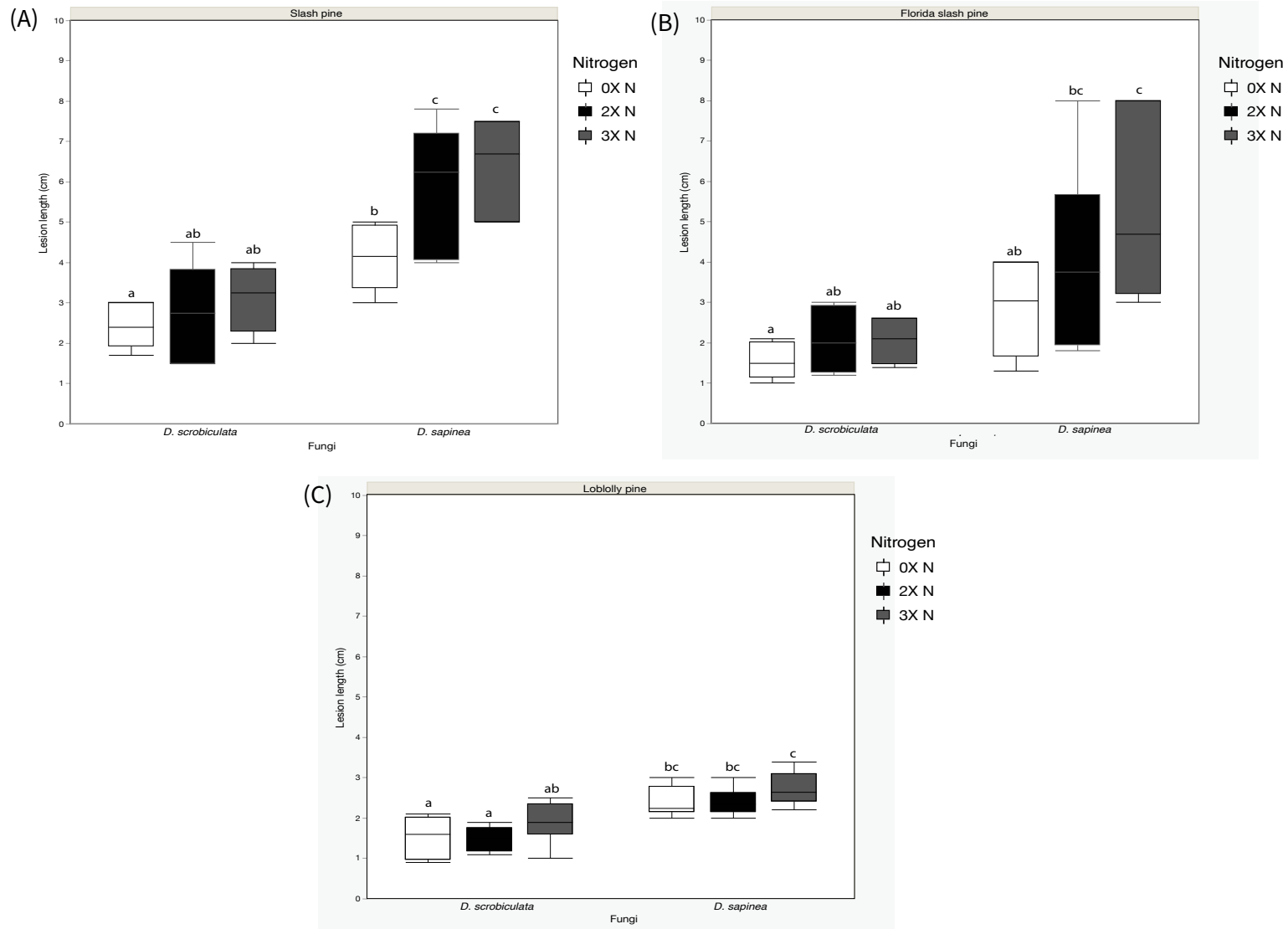


Figure 4.2 —Lesion length (cm) caused by inoculation with two fungal pathogens (*Diplodia sapinea* and *D. scrobiculata*) in slash pine (A), south Florida slash pine (B), and loblolly pine (C) saplings treated with three levels of nitrogen fertilizer (Paez 2017). Different letters indicate statistical significance ($p \leq 0.05$) within tree species among nitrogen treatments.

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INTRODUCTION

Carolina hemlock (*Tsuga caroliniana* Engelm.) is a rare conifer species endemic to the Southern Appalachian Mountain regions of North Carolina, South Carolina, Tennessee, Virginia, and Georgia (Jetton and others 2008). The known distribution covers an area of approximately 465 x 165 km with a latitudinal range from 37°40' N in Rockbridge County, VA, south to 34°73' N in Rabun County, GA (fig. 5.1). Carolina hemlock stands typically grow along the margins of mixed hardwood-conifer forests on exposed rocky outcroppings at elevations between 600 and 1500 m. Individual populations tend to be small (in terms of both area and number of trees) and isolated, with high levels of genetic differentiation from one stand to the next (Potter and others 2017). Because of its narrow geographic distribution and these population characteristics, Carolina hemlock is highly susceptible to extirpation caused by large-scale forest disturbances.

The hemlock woolly adelgid (HWA; *Adelges tsugae* Annand) is an invasive piercing-sucking insect that has caused widespread decline and mortality of Carolina hemlock across its Southern Appalachian distribution. The insect is native to Asia and western North America and was introduced to eastern North America from Japan prior to 1951 when it was first detected in Virginia (Havill and others 2006). Since that time, HWA has spread throughout the hemlock forests of the Eastern United States and Canada where it also impacts the more widely distributed eastern hemlock (*T. canadensis* (L.) Carr.). HWA feeds on

xylem ray parenchyma, depleting stored nutrients leading to premature needle loss, branch and crown dieback, and eventually death of the tree (Havill and others 2014). Hemlocks experiencing predisposing stress can be killed in as little as 4 years, though many trees persist for 10 or more years before succumbing. HWA now infests the entire range of Carolina hemlock and poses a significant threat to the long-term health and sustainability of this already rare conifer species.

The Camcore program at North Carolina State University (NCSU/Camcore) and the U.S. Department of Agriculture, Forest Service have been conducting HWA monitoring and genetic resource conservation efforts in Carolina hemlock stands since 2003. Initially, this work focused on nine well-known occurrences of the species in North Carolina (Tighe and others 2005). Over the next 10 years, the list of sites with active HWA monitoring grew to 15 (Jetton and others 2008) and then 32 (Potter and others 2017) as additional occurrences were reported. National interest in the conservation of Carolina hemlock led to the need to document additional occurrences of the species and to assess tree health, HWA infestation, and treatment status. To accomplish this goal, an Evaluation Monitoring project (EM-SO-15-01) funded by the Forest Service's Forest Health Monitoring Program was initiated in 2015 to delineate and map known and reported Carolina hemlock populations across the species range and assess their general health status. This article presents a project summary on the expanded map of species occurrences, stand characteristics, and overall health.

CHAPTER 5

Carolina Hemlock Distribution and Health in the Southern Appalachian Mountains

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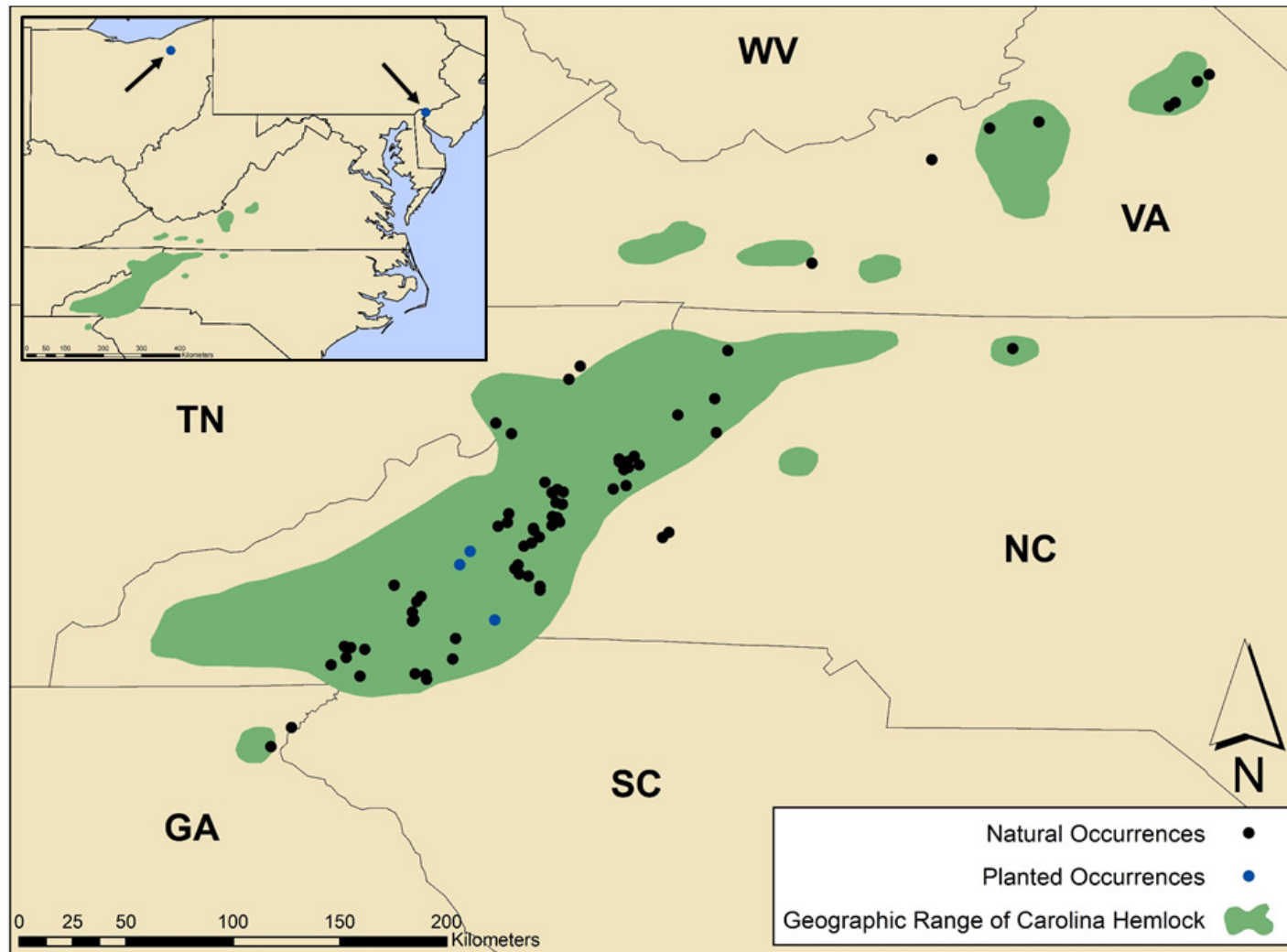


Figure 5.1—Geographic range of Carolina hemlock with locations of the 79 natural and planted occurrences assessed during the project. Inset shows the locations of planted occurrences outside the natural range of the species in Delaware and Ohio.

METHODS

A list of 109 candidate Carolina hemlock occurrences was developed through reviews of the scientific literature and outreach to Federal and State resource managers, natural heritage programs, and private landowners across the natural distribution of the species. Between 2016 and 2020, field visits were made to each of the candidate occurrences to verify the presence of Carolina hemlock, map the trees, and assess health status. At sites where the species was confirmed, a custom form for the ArcGIS Collector smartphone app was used to gather stand- and tree-level attribute data as follows. Stand-level data included a total tally of Carolina hemlock trees, the presence/absence of Carolina hemlock seedling/sapling regeneration, and a list of dominant co-occurring species. Tree-level data included latitude (D.d), longitude (D.d), elevation (m), height (m), diameter at breast height (d.b.h.; cm), aspect, tree health rating, presence/absence of active HWA infestation, and presence/absence of previous HWA treatments. Tree health ratings were based on a 0–5 scale as follows: 0 = dead, 1 = 1–25 percent live crown remaining, 2 = 26–50 percent live crown remaining, 3 = 51–75 percent live crown remaining, 4 = 76–99 percent live crown remaining, and 5 = full, healthy crown with no signs of crown dieback. HWA infestation status was determined by examining the underside of four lower crown branches for presence of adelgid ovisacs. Presence of HWA treatments was based on treatment records provided by site managers and/or the presence of tree tags, tree paint, or other markers commonly used for indicating adelgid treatments. Tree-level data

were collected for 14 percent of all trees tallied during the project. Analysis of variance was used to determine the effect of HWA infestation status, HWA treatment status, and their interaction on Carolina hemlock crown health ratings. Where significant differences were found, mean separation analysis was performed using Tukey-Kramer honestly significant difference (HSD).

RESULTS AND DISCUSSION

Of the 109 sites visited during this project, 79 were confirmed to support living Carolina hemlock trees (fig. 5.1). The remaining 30 sites had no Carolina hemlock and consisted mostly of misidentified occurrences of eastern hemlock or red spruce (*Picea rubens* Sarg.). Others were likely occurrence reports for which inaccurate coordinate information was obtained, or possibly areas where HWA had already eliminated Carolina hemlock, though standing dead stems were not observed at these locations.

Among the 79 sites where Carolina hemlock was confirmed, 74 were identified as being of natural origin, while five were determined to have been planted (table 5.1, fig. 5.1). Two of these sites, Rockwood Park in Delaware and Cuyahoga Valley National Park in Ohio, are far outside the natural distribution of the species, and available records confirm the planted status of the trees at these locations (e.g., MacKay and Trimbath 2022). The three remaining planted sites are all found within the natural range of Carolina hemlock in North Carolina and include Bent Creek Experimental Forest, Biltmore Estate, and the Carl Sandburg Home National Historic Site.

Table 5.1—Stand- and tree-level data for Carolina hemlock occurrences assessed during the project

Variable	Mean	SE	Range	Total
Stand-level attributes				
Total occurrences	—	—	—	79
Natural	—	—	—	74
Planted	—	—	—	5
Regeneration present	—	—	—	27
HWA infestation present	—	—	—	62
HWA treatments present	—	—	—	27
Tree-level attributes				
Total trees	—	—	—	13,766
Elevation (m)	980.43	26.61	330–1580	—
Height (m)	12.32	0.50	3.70–22.38	—
d.b.h. (cm)	22.94	0.95	6.44–44.11	—
Health (0–5) ^a	2.70	0.07	0.80–4.00	—

d.b.h. = diameter at breast height; HWA = hemlock woolly adelgid; SE = standard error.

^a Tree health ratings were based on a 0–5 scale as follows: 0 = dead, 1 = 1–25 percent live crown remaining, 2 = 26–50 percent live crown remaining, 3 = 51–75 percent live crown remaining, 4 = 76–99 percent live crown remaining, and 5 = full, healthy crown with no signs of crown dieback.

The planted nature of trees at these locations was confirmed by onsite resource managers. Among the 74 populations of natural origin, 48 were new to the NCSU/Camcore-Forest Service monitoring and conservation program for Carolina hemlock and 26 had been included in previous efforts (Potter and others 2017).

A total of 13,766 Carolina hemlocks were tallied across the 74 natural occurrences, ranging in elevation from 330 to 1580 m (table 5.1). Trees in five locations occurred below the lower elevation limit of 600 m previously reported for the species (Jetton and others 2008, Rentch and others 2000). Individual Carolina hemlocks were found growing on all aspects, but most (65 percent) occurred on northerly and westerly exposures (table 5.2). This is consistent with species descriptions indicating that Carolina hemlock typically grows on less productive, dry, exposed, nutrient-poor sites where its stress-tolerant life history strategy (drought and shade tolerance, slow growth, evergreen needles, long lifespan) allows the species to be successful

(Humphrey 1989). Similar to other studies of Carolina hemlock communities (Keyser and others 2023, Rentch and others 2000), common associates included shade-tolerant blackgum (*Nyssa sylvatica* Marsh.), red maple (*Acer rubrum* L.), and sourwood (*Oxydendrum arboreum* L.); mid-tolerant chestnut oak (*Quercus montana* Willd.) and northern red oak (*Q. rubra* L.); and understory ericaceous shrubs such as Catawba rosebay (*Rhododendron catawbiense* Michx.), mountain laurel (*Kalmia latifolia* L.), and a variety of *Vaccinium* spp.

The number of trees per occurrence varied greatly across the 74 natural Carolina hemlock sites, ranging from a single tree along the east bank of the Chattooga River on the South Carolina-Georgia border to more than 1,700 trees along the Appalachian Trail on Sinking Creek Mountain near Blacksburg, VA. The majority of occurrences (47) consisted of fewer than 100 scattered trees making up a minor component of the overall forest canopy (table 5.2). The remaining 27 were larger with 100 or more trees,

Table 5.2—Percentage of trees per aspect and number of occurrences per occurrence size for Carolina hemlock occurrences assessed during the project

Variable								
Aspect	North	Northeast	East	Southeast	South	Southwest	West	Northwest
Trees (percent)	20	9	7	6	7	6	19	26
Occurrence size (trees per occurrence)	1-25	26-50	51-99	100-499	500-999	1,000+		
Number of occurrences	24	13	10	20	4	3		

but only 7 of these were pure stands with 500 or more trees dominating the canopy. The relative scarcity of mature Carolina hemlock in the forest canopy across sites was reflected in its lack of representation in the understory, with seedling/sapling regeneration present at only 27 of the sites. Keyser and others (2023) found similar trends for the species, with Carolina hemlock being of relatively low importance value in the overstory layer and representing a very small component (~3 percent) of the regeneration layer. Because Carolina hemlock requires the presence of mature trees in the canopy to maintain itself on site (Humphrey 1989) and due to the absence of significant advanced regeneration across most occurrences and likely loss of mature trees from the overstory due to HWA infestation, persistence of the species in these ecosystems without management intervention seems unlikely.

HWA infestations were present in 62 of the 74 natural occurrences and evidence of previous adelgid treatments was noted in 27 (table 5.1). Based on the tree-level assessments, the number of individual Carolina hemlocks with active HWA infestations was nearly double that of uninfested trees across occurrences, and nearly half had received previous adelgid management treatments (fig. 5.2A). Fifty-two percent of trees were confirmed to be untreated or were of unknown treatment status. Treatments consisted primarily of applications of the systemic neonicotinoid insecticide imidacloprid, while one site was reported to have been treated with dinotefuran and three had documented releases of HWA biological control agents in addition to insecticide applications. The average crown health

rating across all trees assessed was moderate at 2.70 (table 5.1), ranging from 0.80 (near total crown mortality) to 4.00 (>75 percent live crown remaining). Tree health ratings were significantly affected by both HWA infestation status ($df = 1$, $F = 186.81$, $p < 0.0001$) and treatment status ($df = 2$, $F = 9.23$, $p < 0.0001$) but not their interaction (fig. 5.2B).

Although significant, the differences in Carolina hemlock health ratings based on treatment status were quite small, and trees across all three treatment types were of relatively poor crown health. Most trees had received treatment 5 years or more prior to assessment, and because imidacloprid effectiveness begins to wane 4 to 6 years following application (Benton and others 2016), most trees were likely no longer fully protected from HWA infestation. Similarly and encouragingly, uninfested Carolina hemlocks were identified and had significantly better crown health than infested trees. However, the 3.1 average health rating among uninfested trees indicates that this cohort had experienced an average of 51–75 percent crown decline and dieback, meaning these trees had likely been impacted by HWA in the past. This trend can be explained by expected variations in adelgid population density over time (Havill and others 2014) as populations can cycle from very high to very low or undetectable levels in response to variations in hemlock health or extreme winter and summer temperatures. In response, hemlocks decline gradually, cycling between periods of improved and reduced health, lingering in the forest for 10 or more years before finally succumbing to the infestation. This points to the

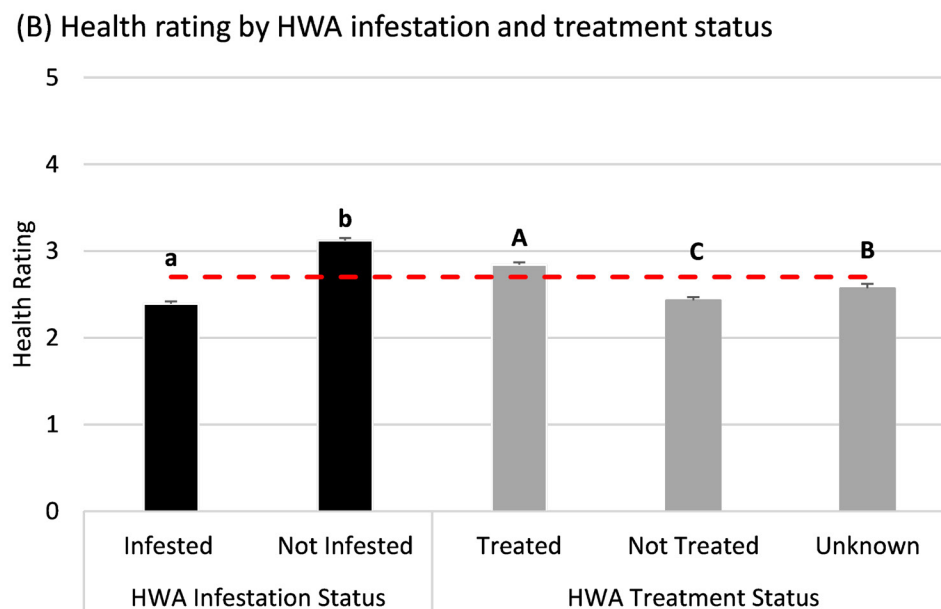
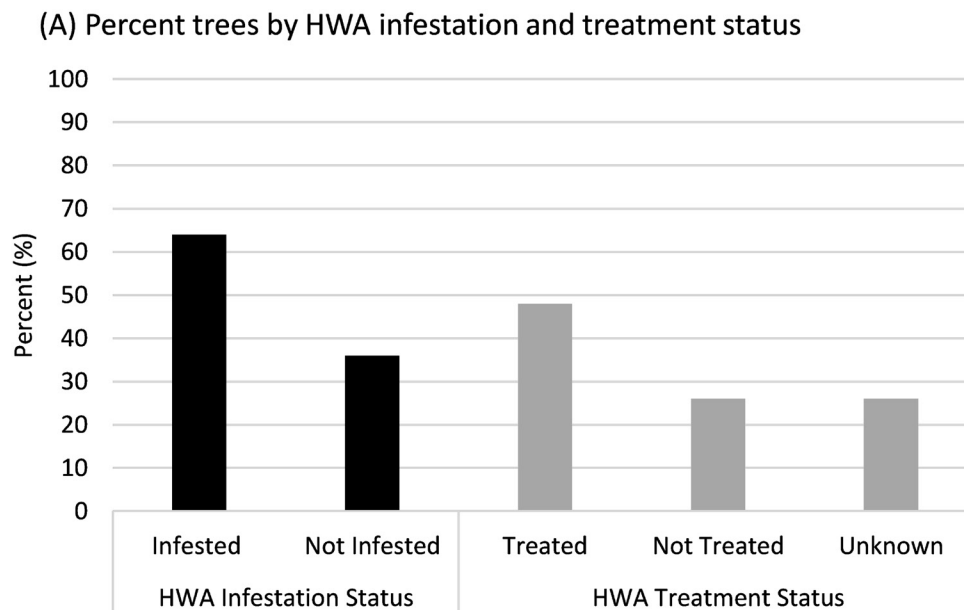


Figure 5.2—Percentage of trees by hemlock woolly adelgid (HWA) infestation and treatment status (A) and response of crown health rating to HWA infestation and treatment status (B) across Carolina hemlock occurrences assessed during the project. Red dashed line represents the average health rating of all trees in the assessment (2.70). Bars with different lowercase or uppercase letters are significantly different $\alpha = 0.05$.

limitations of single “snapshot in time” monitoring efforts like the one reported here and the need for the establishment of long-term monitoring plots to better track Carolina hemlock health and identify areas most in need of management intervention.

CONCLUSIONS

HWA has been established across the range of Carolina hemlock for approximately two decades. During this time, HWA monitoring and management efforts have focused on a small number of well-known occurrences of the tree species. This project added 48 occurrences to the list for which monitoring data were gathered to provide a better picture of Carolina hemlock health and HWA infestation and treatment status across the species range. While Carolina hemlock persists at all sites, the health of most trees assessed is poor to moderate, indicating that additional management intervention is needed to maintain the species as part of the Southern Appalachian ecosystem. Continued use of systemic insecticides integrated with expanded releases of HWA biological control agents will benefit the health of the trees (Mayfield and others 2020). However, opportunities also exist to explore the use of silvicultural tools that will promote seed production and germination, increase seedling/sapling recruitment into older age classes, and improve Carolina hemlock health and resilience to HWA infestation as has been documented for eastern hemlock (Mayfield and others 2023). Management success will depend on continued monitoring in stands where treatments are applied. The development of monitoring

tools designed for Carolina hemlock ecosystems and the establishment of a plot network across the species range are the subject of a currently ongoing Evaluation Monitoring project (EM-SO-21-02).

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INTRODUCTION

Flowering dogwood (*Cornus florida* L.) is a common forest understory tree that is widely distributed in the Eastern United States, where it ranges from eastern Oklahoma and eastern Texas through the southern Gulf Coast to northern Florida, extending north as far as southern Michigan and New England (Little 1971). Flowering dogwood is an important understory component of old-growth forests (Goebel and Hix 1996, McCune and others 1988) that has seen varying degrees of localized and regional health problems across its natural habitat in recent years (Oswalt and others 2012). Although disease in the northeastern population of dogwoods in the United States has been mainly attributed to dogwood anthracnose caused by a fungal pathogen (*Discula destructiva* Redlin; Britton 1994), in the Western Gulf region where the pathogen has yet to be detected, the apparent prevalence of unhealthy dogwoods may indicate a decline complex according to the general description, i.e., predisposing, inciting, and contributing factors that result in disease (Manion 1981). The objectives of this study were (1) to understand the increasing flowering dogwood mortality and population decline in the Southeastern United States over the past few decades; (2) to identify the predisposing, inciting, and contributing factors that could be leading to a decline complex; and (3) to investigate interactions between the identified factors based on Forest Inventory and Analysis (FIA) data and site-specific field survey observations. Findings from the study could facilitate future development of flowering dogwood decline risk assessment tools to assist land managers with management decisions.

MATERIALS AND METHODS

Analysis of Flowering Dogwood Mortality From Forest Inventory and Analysis Data

The flowering dogwood mortality in the FIA plots in the Southeastern United States was obtained from the FIA online data repository for 1958 to 2018, using multiple specific FIA subset databases and the FIA Database User Guide P2_8-0 (<https://usfs-public.box.com/v/FIA-FIADB-UserGuides>). Data from 12 States including Alabama, Arkansas, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, South Carolina, Tennessee, Texas, and Virginia were analyzed to identify biotic and abiotic variables and determine the relationships between the flowering dogwood year of mortality (MORTYR) and the causal agents (AGENTCD) (i.e., animal, diseases, fire, insects, silviculture, competition by vegetation, weather related factors, and others) in the FIA plots surveyed. A total of 186,871 flowering dogwood trees from the FIA plots were evaluated. Data analyses were performed using JMP-SAS statistical software.

CHAPTER 6

Application of Forest Inventory and Analysis Data and Site-Specific Factors in Understanding Flowering Dogwood Decline in the Southeastern United States

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Evaluations of Site-Specific Field Plots for Evidence of Flowering Dogwood Decline

A limited-scope, multiyear field survey was conducted from 2020 to 2022 at the Catahoula, Evangeline, Kisatchie, and Winn Ranger Districts of the Kisatchie National Forest, LA, as well as a privately owned site in Pollock, Grant Parish, LA (adjacent to the Catahoula Ranger District). The survey primarily focused on the health of flowering dogwood in Louisiana, where dogwood anthracnose is currently absent, and was aimed at investigating the role site-specific factors may have on a population decline in Louisiana and the Western Gulf region. A total of 103 flowering dogwood trees were identified, tagged, and repeatedly evaluated during the study period. Healthy and unhealthy flowering dogwood trees within diverse forest canopy structures across survey sites were evaluated using selected variables associated with the potentially declining flowering dogwood trees (table 6.1). Visual evaluation of crown conditions and the assessment of crown dieback incidence and severity on individual trees were recorded. The overall crown health condition based on severity of dieback was scored on a scale ranging from 0 to 4, where 0 = no dead branches, 1 = <1/3 dead crown, 2 = 1/3–2/3 dead crown, 3 = >2/3 dead crown, and 4 = entirely dead. Symptomatic tissue samples for laboratory analysis and site-specific data were collected at survey sites. Data analyses were performed using JMP-SAS statistical software.

Tissue samples were collected from symptomatic (i.e., discolored foliage) flowering dogwood trees to determine the presence of *D. destructiva*, the causal pathogen of dogwood anthracnose. Other fungi isolated from tissue samples were cultured on potato dextrose agar at ambient temperature in the laboratory and subsequently identified using morphological characteristics and DNA analysis of the nuclear ribosomal internal transcribed spacer (ITS) region. Insect specimens associated with declining trees were collected and identified based on morphological characteristics.

Field assessments and other planned study activities, including movements within and across States, were limited to sites solely in Louisiana due to the COVID-19 (coronavirus) pandemic, which substantially hindered travel/research activities and negatively impacted the installation and evaluation of field plots and survey components of the study.

Table 6.1—Field survey data variables evaluated for flowering dogwood decline

Variable	Definitions/categories
Site type	Residential/park = RES; privately owned forest = PVTF; publicly owned forest = PUBF
Forest type	Pine dominated = PINE; mixed pine/hardwood = MPH; hardwood = HDWD
Burn history	Not known = NK; last 5 years (based on knowledge or bole scorch) = 5; 5 or more years = 5+
Canopy exposure	Open canopy condition = O; partially open canopy = P; closed canopy condition = C
Dieback	Score (0–4): 0 = no dead branches; 1 = <1/3 dead branches, 2 = 1/3–2/3 dead crown; 3 = >2/3 dead crown; 4 = entirely dead
Position of dieback mortality	Upper = U; midcrown = M; lower = L
Epicormics	Yes/no. If yes, 1 = 1–10; 2 = 11–20; 3 = >20.
Woody stem galls	None = N; light = L; moderate = M; heavy = H. If limbs are too high to accurately assess, note "TH"; if only a few limbs are low enough to observe, note such.
Club galls on fine twigs	None = N; light = L; moderate = M; heavy = H. If limbs are too high to accurately assess, note "TH"; if only a few limbs are low enough to observe, note such.
Leaf spots/lesions	None = N; light = L; moderate = M; heavy = H. If limbs are too high to accurately assess, note "TH"; if only a few limbs are low enough to observe, note such.
Flower spots/lesions	None = N; light = L; moderate = M; heavy = H. If limbs are too high to accurately assess, note "TH"; take representative picture if possible.
Trunk scarring	Presence of burn damage (meaning cambium, callus tissue formed around exposed xylem) = BD; mechanical damage = MD; wildlife damage = WD
Fungal/bacterial canker	If damage is present, classify intensity by 1 = <25% of bole circumference affected; 2 = 25–50%; 3 = >50%
Woodborer exit holes	Presence = yes/no. If yes, denote number observed and any other notes on such.
Fungal fruiting bodies	Presence = yes/no. If yes, note at tree base/soil line "S" or on tree as "T." Take representative picture if possible.
Seedlings	Presence/absence of seedlings (defined here as trees <3 feet tall) within 25 feet of tree = yes/no. If yes, note approximate number.
Other notes	Note any other leaf/branch/trunk abnormalities and attempt to quantify.

RESULTS

Analysis of Flowering Dogwood Mortality From Forest Inventory and Analysis Data

Results showed that 17.9 percent (33,382) of flowering dogwood trees were lost between 1958 and 2018 due to various factors, including silviculture/land clearing activities, vegetation impacts (such as suppression of growth, competition with other plant species, and vines/kudzu (*Pueraria montana*)), diseases, weather, fire, and insect damage, among others. Flowering dogwood mortality increased significantly during periods from 1979 to 1981, 1986 to 1989, 1999 to 2000, and 2008 to 2009, while the highest tree mortality across the region occurred between 1986 and 1989 (fig. 6.1). Approximately 36 percent of tree mortality or losses was related to silviculture/land clearing activities, 31 percent was from vegetation impacts, and 17 percent was associated with diseases, while the remaining individual causal agents—i.e., weather, insects, animal, fire, and “unknown”—accounted for tree mortality ranging from 0.1 to 7 percent.

Silviculture and land clearing activities increased flowering dogwood mortality steadily from 1958 for more than a decade, then accelerated in mid-1970s, and peaked in 1986/1987 before decreasing gradually. A similar trend was observed for the vegetation impacts (i.e., suppression, competition, vines/kudzu etc.) on flowering dogwood mortality during the same period. Further analyses showed that unknown or unidentified factor(s) may have contributed to mortality as well; however, the overall impacts of the “unknown,” along with other agents (including animal, fire, insects, and weather), were minimal. Variability in flowering dogwood mortality was noticeable across FIA plots in the 12 States. In Kentucky, flowering dogwood mortality was significantly higher from 1980 to 1982 and from 1986 to 1988 than at any other periods. The highest mortality in Kentucky was recorded in 1987, when 722 trees died across the FIA plots, attributed to “no agents recorded.” Although no causal agent was recorded as responsible for tree death during these two periods, the periods corresponded with years of extended and extreme drought events according to the statewide Standardized Precipitation Index (SPI), which measures the deviation from the climatological average for a given period.

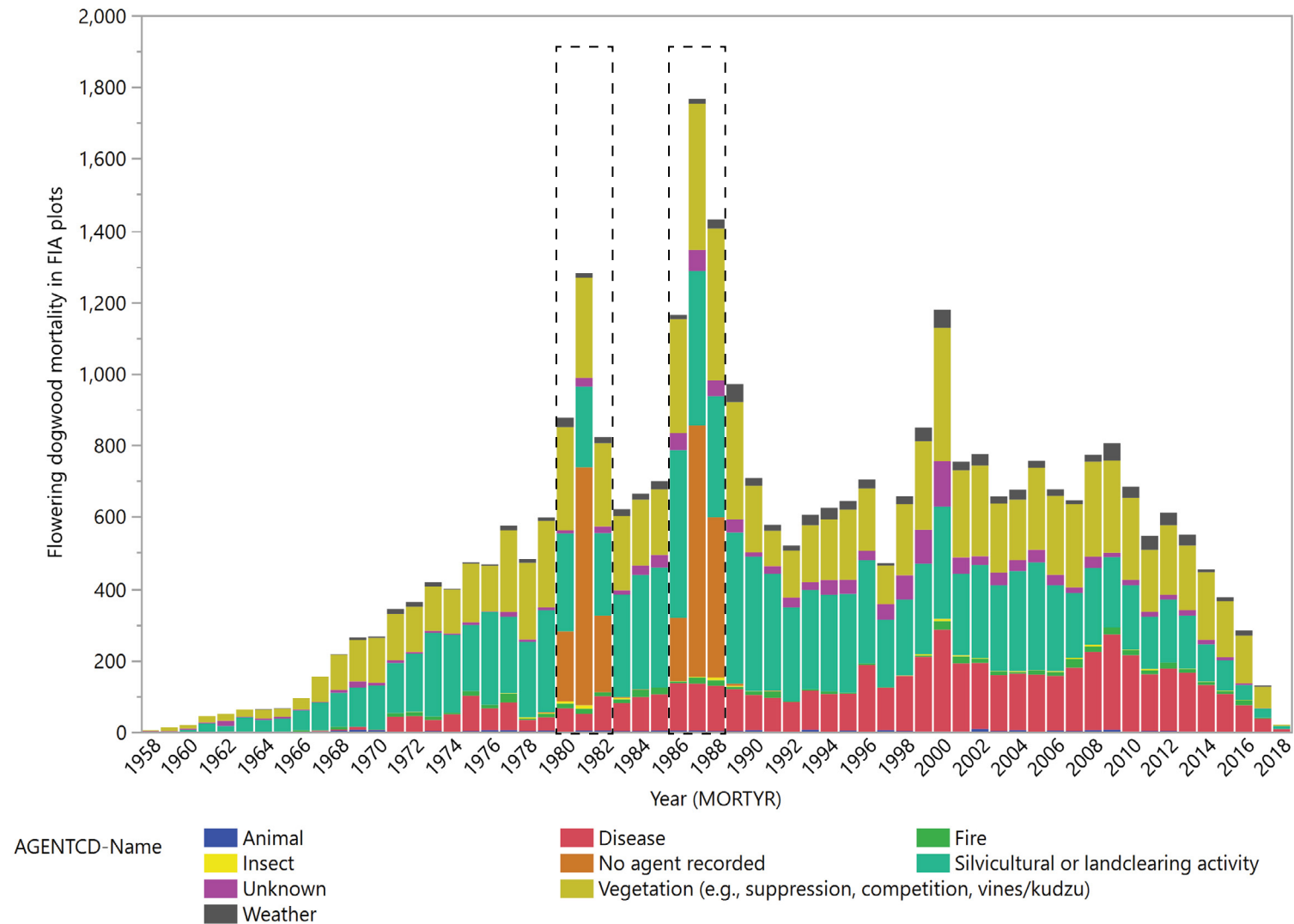


Figure 6.1 —Abiotic and biotic agents associated with flowering dogwood mortality across Forest Inventory and Analysis (FIA) plots in the Southeastern United States for 1958–2018, based on observations and the causal agents from the FIA Database (FLADB) according to the FLADB user guide. Although FIA data do not indicate what could have caused flowering dogwood mortality during periods shown inside dashed rectangles, climate data suggest that periods of extreme drought events corresponded with the two periods of significant tree mortality in Kentucky.

Evaluations of Site-Specific Field Plots for Evidence of Flowering Dogwood Decline

Results from data collected from field plots showed that over 70 percent of tagged flowering dogwood trees evaluated had some level of observed dieback during initial assessments, with 75 percent at the Catahoula Ranger District and 59.2 percent at the Pollock site in 2020, and 84 percent at the Evangeline Ranger District, 50 percent at the Kisatchie Ranger District, and 100 percent at the Winn Ranger District in 2021. The affected trees were at various stages of dieback with most showing >1/3 dead crown, while approximately 7 percent were dead across locations by the time the final assessments were conducted (fig. 6.2). Further analysis identified an interaction between canopy exposure and the incidence of dieback on flowering dogwood at the Evangeline Ranger District in 2021 and 2022. Results indicate a higher incidence of dieback in partially open canopies compared to closed or open canopy stands (fig. 6.3).

Visual assessments of flowering dogwood conducted during spring 2020 showed most trees had healthy foliage and minor blemishes on flowers across sites. Swollen twig terminals

caused by the dogwood club gall midge (Diptera: Cecidomyiidae: *Resseliella clavula* (Beutermüller)) were commonly observed. Other symptoms including callused cankers were found in the secondary and/or tertiary branches of flowering dogwoods, while insect exit holes were noticed on symptomatic twigs examined at the Pollock site. Also, at the Kisatchie National Forest in late summer of 2020, the black twig borer (*Xylosandrus compactus* (Eichhoff)) was found in twigs on a few declining trees (fig. 6.4).

Diverse species of pathogenic and opportunistic fungi isolated from declining flowering dogwood tissue samples collected from the Kisatchie Ranger District include *Botrytis cinerea* (botrytis blight), *Cytospora brevispora* (canker), *C. pruinos*a (canker), *Clonostachys rosea*, *Epicoccum nigrum*, *Fusarium lateritium* (stem canker), *Lasiodiplodia theobromae* (canker), *Nigrospora sphaerica*, and *Pestalotiopsis microspora*. Other fungal species isolated from samples collected at the Winn Ranger District in 2021 include *Colletotrichum acutatum*, *C. fioriniae*, *C. fructicola*, *C. siamense*, *Diaporthe phaseolorum*, *Fusarium verticillioides*, and *Myrothecium* spp. However, *D. destructiva*, the causal pathogen of dogwood anthracnose, was not detected, nor were disease-related symptoms observed on the evaluated dogwood trees.

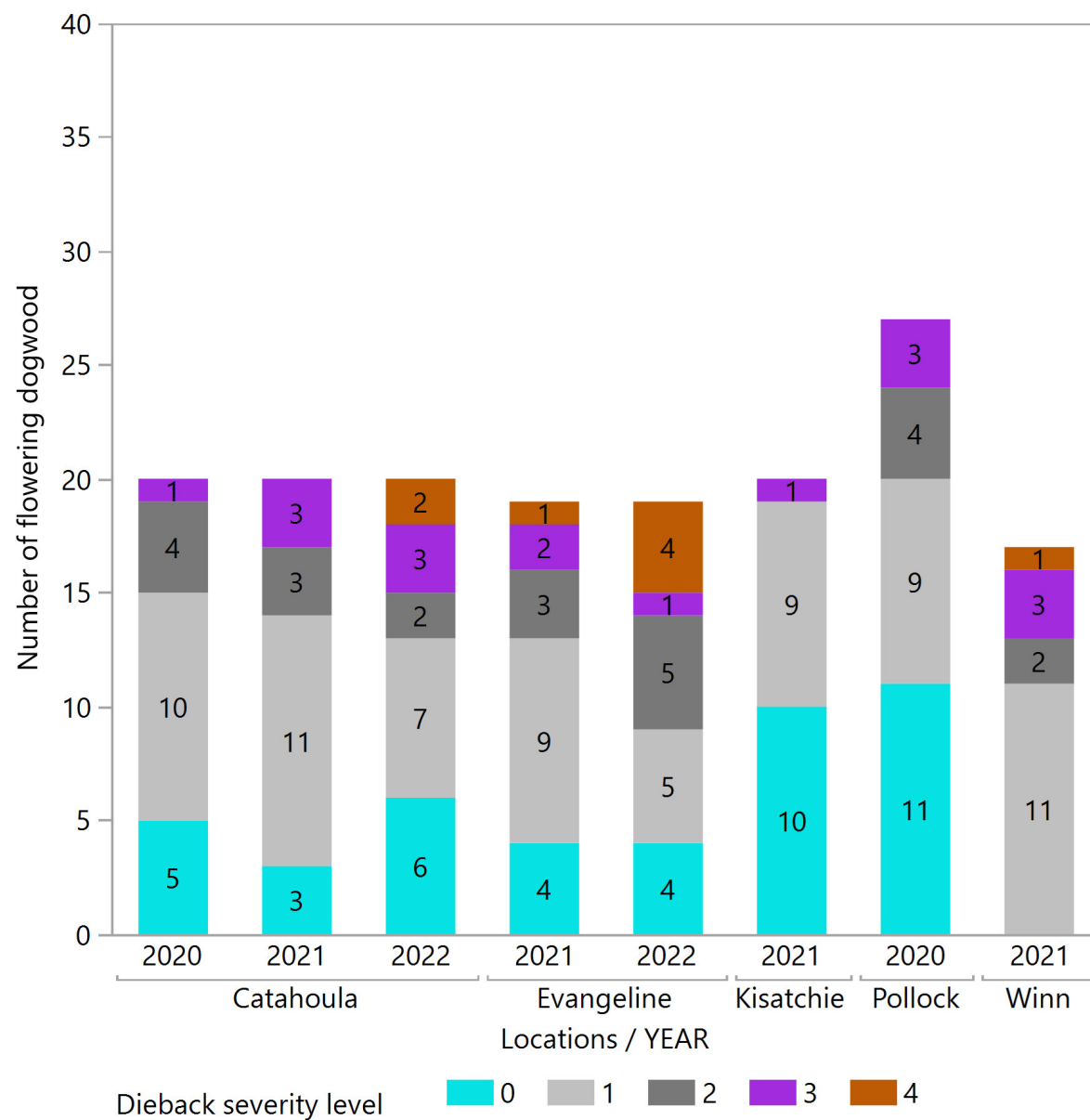


Figure 6.2—
Incidence and
severity of dieback on
flowering dogwood at
multiple field survey
locations from 2020
to 2022 in Louisiana.
The severity was
scored on a scale of
0 to 4, where 0 = no
incidence or dead
branches, 1 = <1/3
dead crown, 2 =
1/3–2/3 dead crown,
3 = >2/3 dead crown,
and 4 = entirely dead.

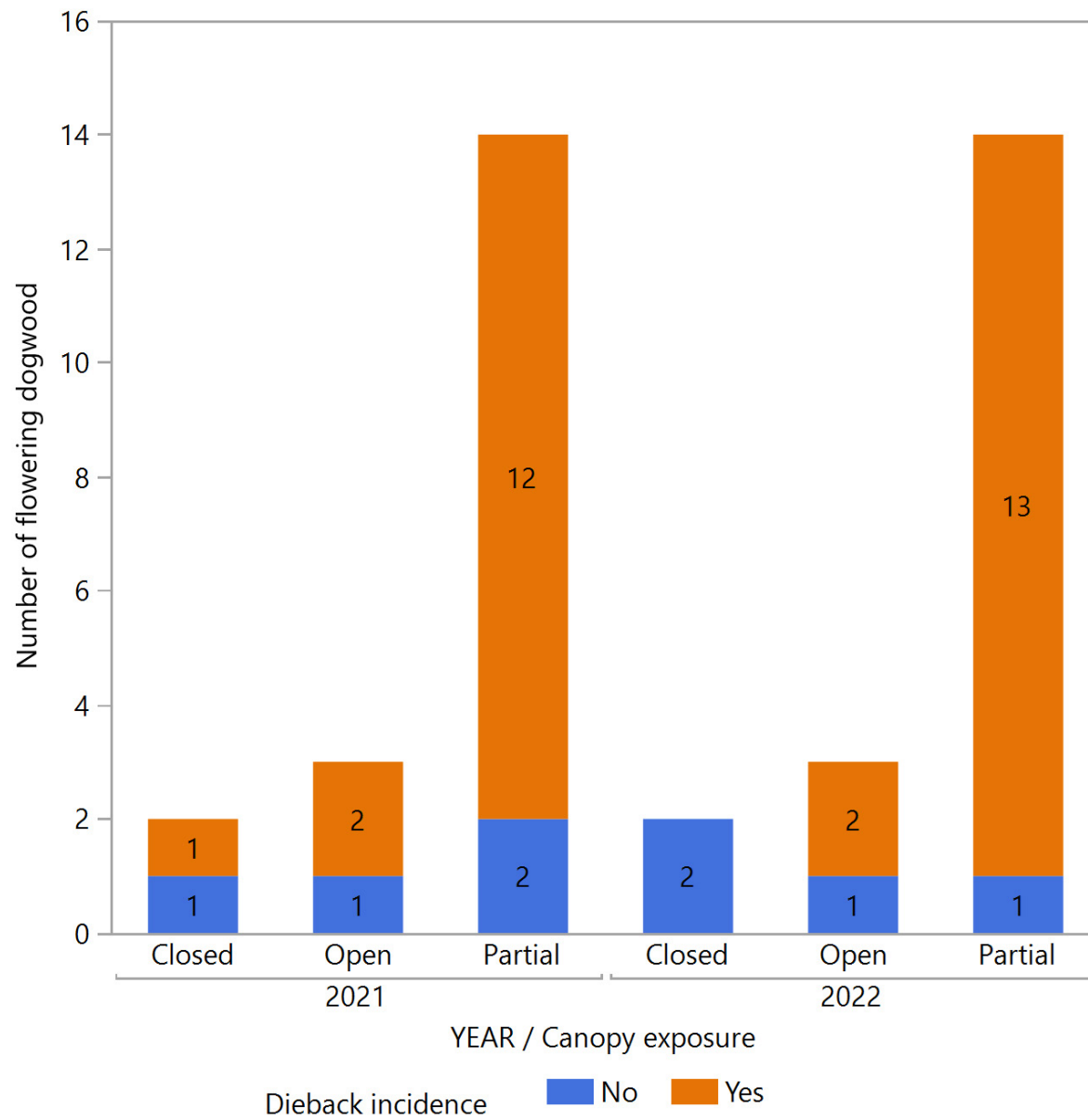


Figure 6.3—Canopy exposure and the incidence of dieback on flowering dogwood trees evaluated in 2021 and 2022 at the Evangeline Ranger District, Kisatchie National Forest, LA.



Figure 6.4 —(A) Healthy flowering dogwood trees observed within a pine stand; (B) leaves; (C) ripe berries, at the Kisatchie National Forest, LA, in late summer in 2020; (D) blighted flower caused by *Botrytis cinerea*; (E and F) callused-over cankers in twigs observed in the secondary/tertiary branches; (G) black twig borers on symptomatic twigs; (H) flowering dogwood tree decline observed in an open forest canopy structure; and (I and J) abiotic factors including old wound and fire injury observed on declining trees at the Kisatchie National Forest, LA. A–C and G–J: USDA Forest Service photos by Rabiw Olatinwo; D–F: USDA Forest Service photos by Wood Johnson.

DISCUSSION

Flowering dogwood is a very shallow-rooted forest understory tree that prefers indirect afternoon sunlight or direct morning light, hence requiring overstory protection particularly in late summer when critical water needs must be met to prevent stress or death from the hot afternoon sun (Smith and others 2023). Flowering dogwood is not drought tolerant and therefore is vulnerable to water stress that may also predispose it to diseases and insect pest attacks (e.g., borers). These characteristics as well as site-specific predisposing factors relating to forest canopy structure could be important in determining the sensitivity of flowering dogwood to decline causal agents, the decline progression, and subsequent tree mortality.

Whatever the causal agents or interactions among multiple agents that might be involved, documented geographic distribution (Little 1971) and the FIA flowering dogwood mortality data collected since the 1980s across the Southeastern United States clearly indicate that a substantial decline has occurred in flowering dogwood populations. In the absence of any long-term management and restoration plans, the damaging effects of the dogwood anthracnose pathogen (in some regions), combined with various abiotic and biotic factors including competition by other species and defoliators, diminish the hope of decelerating the decline of flowering dogwood populations on the landscape (Oswalt and others 2012). Although dogwood anthracnose was not detected nor any related symptoms observed on the evaluated flowering dogwood trees in Louisiana from this study, other opportunistic and

pathogenic fungi such as *B. cinerea* and various types of cankers were recovered from vulnerable declining dogwoods in the Kisatchie National Forest and nearby forests in Louisiana.

Results from field surveys in this study suggest a higher incidence of dogwood dieback in partially open canopies than in closed or open canopies. Ward and Stephens (1993) noted that shade-intolerant woody species declined slowly in some plots as the forest matured and the canopy closed. In a study on the influence of site factors on dogwood anthracnose in the Nantahala Mountain range of western North Carolina, Chellemi and others (1992) observed higher disease incidence and greater severity in interior forest areas where the lowest evaporative potential was found. These observations highlight the complexity of the forest canopy structure that protects vulnerable flowering dogwood from exposure to direct sun on one hand, and the conducive environment within the interior forest area that may support the proliferation of pathogens and subsequent disease development on the other hand.

Silviculture and land clearing activities, vegetation impacts due to suppression of growth, competition from other plant species, diseases caused by pathogens, climate/weather, fire, and insect pests were each implicated to varying degrees as causal agents in flowering dogwood mortality according to the FIA survey data examined in this study. The extreme and extended period of drought events associated with a changing climate in part of the Southeastern United States may continue to affect flowering dogwood populations negatively, perhaps shifting habitat suitability in parts of the region that could

expose the flowering dogwood population to further competition from climate change-induced species migration (Oswalt and others 2012, Woodall and others 2009).

CONCLUSIONS

Monitoring the complex interactions among predisposing, inciting, and contributing factors and their roles in tree decline would be helpful in assessing the impacts of a changing climate on flowering dogwood populations as well as mitigating tree decline or mortality in forest stands across the region. Continuous exposure of flowering dogwood to stressors such as drought, pathogens/pests, and direct sunlight because of overstory tree harvesting/canopy opening may increase and expand the distribution of flowering dogwood decline in southeastern forests over time.

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INTRODUCTION

Invasive plants cause significant ecological and economic harm in natural and managed forested areas throughout the world (Levine and others 2003). Previous research has documented multiple ways in which invasive plants alter functions of recipient forested ecosystems, including disturbance regimes, primary productivity, nutrient cycling, carbon sequestration, and tree regeneration (Levine and others 2003, Peltzer and others 2010). Unfortunately, invasion events are likely to be more frequent in the future, as the rate of introduction of novel species continues to increase due to human-mediated dispersal through multiple trade networks (e.g., Lucardi and others 2020).

Factors influencing the successful establishment of invasive species are complex and interact with one another at multiple spatial and temporal scales, complicating adaptive management efforts. The major aim in this project was to determine biotic and abiotic factors that are most closely associated with successful forest plant invasions across the Southern United States, which is an area of primary concern with regard to the spread of invasive plant species (Oswalt and Oswalt 2011, Oswalt and others 2015).

Although researchers in this study had an interest in evaluating spatial patterns of potential future invasions within the region, many invasive plant species are already well established in the region and are, presumably, having significant impacts on forests. Many studies have shown that invasive plants pose great challenges to forest regeneration at local spatial scales (e.g., Boyce 2009, Levine and others 2003, Mack and

D'Antonio 1998, Merriam and Feil 2002, Oswalt and Oswalt 2011); however, little is known about how those local-scale impacts of invasive plants may collectively influence forest regeneration at a regional scale. A second aim of the work described here was to assess the potential for negative impacts of established invasive plant species on forest regeneration across the region.

Using geospatial and statistical modeling approaches, this study evaluated how biotic interactions, abiotic conditions, and landscape factors affect the distribution of invasive plant taxa within the Southern United States. It also examined, using data from the U.S. Department of Agriculture (USDA), Forest Service, Forest Inventory and Analysis (FIA) Program, the potential impact of invasive plant taxa on desirable tree species recruitment within the region.

METHODS

Effects of Environmental Predictors on Invasive Species Distribution in the Southern United States

Environmental variables and presence-absence data were compiled for 16 invasive plant species known to have high rates of spread across 52,690 Southern U.S. (Forest Service Region 8) FIA plots that were surveyed between 2015 and 2019 (Lázaro-Lobo and others 2021b). Working directly with FIA personnel, species data from the FIA Database were matched with georeferenced environmental variables from publicly available data sources, including a U.S. Department of the Interior, U.S. Geological Survey (USGS) digital elevation model (DEM); WorldClim climate data;

CHAPTER 7

Evaluating Distribution and Impacts of Invasive Plant Species on Southern U.S. Forests

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Multi-Resolution Land Cover (MRLC) land cover and tree canopy data; MRLC normalized difference vegetation index (NDVI); and USDA Soil Survey Geographic Database (SSURGO). Generalized linear models (GLMs) with binomial distribution of residuals were used to evaluate the effect size of each environmental variable on invasive species occurrence.

Potential Distribution of Invasive Plants in the Southern United States

This study used an ensemble species distribution modeling (SDM) approach to model the potential distribution of invasive plants throughout the Southern United States, based on the species and environmental data collected above (Lázaro-Lobo and others 2021b). A model ensemble for each species was generated by averaging the output of individual modeling techniques (generalized additive model (GAM), maximum entropy (MaxEnt), and generalized boosted regression trees (GBM)), weighted by their model performance scores (area under the curve (AUC) values).

This ensemble modeling approach resulted in SDM outputs with 30-m-resolution pixels, where pixel values ranged from 0 (low susceptibility to invasion) to 1 (high susceptibility to invasion). Lastly, the mean susceptibility to invasion across the evaluated species was calculated for each cell in the resulting risk maps. This allowed delineation of areas at high risk of invasion (hotspots) in Southern U.S. forests.

Landscape Characteristics of Predicted Hotspots for Plant Invasions

The web-based Early Detection and Distribution Mapping System (EDDMapS) collects occurrence data of invasive species across different land cover types. This analysis used occurrence points throughout the Southern United States, collected between 2007 and 2018, for 45 major invasive plant species causing negative effects (Lázaro-Lobo and others 2020). As environmental predictors, georeferenced environmental variables were obtained from publicly available data sources (USGS DEM topographic data, WorldClim climate data, and MRLC land cover and tree canopy data from 2011).

To examine the relationship between landscape characteristics and susceptibility to invasion at different spatial scales, this study first modeled the susceptibility of areas within the Southern United States (30-m-resolution grid) to be invaded by each invasive species using MaxEnt. Then, random points were placed in the resulting invasion risk maps and two buffers were created around those points at different distances: medium (0–150 m) and long (0–500 m). Landscape composition and configuration were examined using MRLC land cover data and within the above-mentioned buffers using the ClassStat function of the SDMTTools package in R.

Relative Importance of the Biotic Community, Abiotic Factors, and Landscape Characteristics on Nonnative Species Richness and Abundance

A dataset was collected by visiting 460 sites across the State of Mississippi during 2006–2009 (Lázaro-Lobo and Ervin 2021). These sites were selected randomly from all land in the State that could be accessed, including State and national parks, national wildlife refuges, State wildlife management areas, and national forests. At each site, researchers collected data on the plant species present from the ground layer to the forest canopy (when present), along with soil samples and data on local land cover and use. Using ArcGIS, data on land cover and proximity to transportation corridors were added to the data for each random point.

This study used these data to assess the effect of the biotic community, abiotic factors, and landscape characteristics on nonnative species richness and abundance. It used GLMs, regression tree, and variation partitioning analyses to evaluate how selected predictors interacted to favor or limit nonnative and/or native species richness and percent area cover.

Impacts of Invasive Plants on Tree Regeneration

Presence/absence data were compiled for 16 major invasive plant species from the above-mentioned 52,690 FIA plots and calculated the number, richness, and diversity of seedlings and saplings for all major tree species present in those same plots (Lázaro-Lobo and others 2021a). GLMs were used to examine the relationship of the selected invasive plant species to tree seedlings and saplings. Principal component analysis then evaluated whether correlations between invasive species and tree seedlings or saplings appeared to be related to overlap in niche associations for each possible pair of species. Niche associations were represented by the same set of environmental variables as for the FIA-based invasive species distribution modeling described above (USGS DEM, WorldClim climate data, MRLC land cover and tree canopy data, MRLC NDVI, and USDA SSURGO data).

RESULTS

Effects of Environmental Predictors on Invasive Species Distribution in the Southern United States

Invasive species presence in the FIA database was favored by proximity to pastures, croplands, and developed land cover, as well as by high light availability (low tree canopy cover) and soil capacity to retain nutrients and water. In contrast, soil organic matter generally was negatively correlated with invasive species presence. The effects of climatic variables and other soil properties, such as pH and soil depth, varied in their direction and were species specific.

Potential Distribution of Invasive Plants in the Southern United States

Most of the examined invasive plant species have the potential to occupy large areas of forest throughout the region (fig. 7.1). The invasive taxa with the greatest potential to establish across Southern U.S. forests were *Ligustrum* spp. (nonnative privets) and *Lonicera japonica* (Japanese honeysuckle) (37–39 percent of the forested area), followed by *Rosa* spp. (nonnative roses), *Imperata cylindrica* (cogongrass), and *Lygodium japonicum* (Japanese climbing fern) (18–23 percent). Each evaluated species is likely to occupy different areas of the Southern United States, with some being restricted to coastal areas, whereas others were more likely to occupy interior areas.

Landscape Characteristics of Predicted Hotspots for Plant Invasions

Most of the species examined were more prone to invade areas with anthropogenic land cover, such as developed areas and barren lands, than areas that experienced minimal human interference. Landscape heterogeneity and the presence of corridors for propagule dispersal significantly increased susceptibility to invasion by most species. However, susceptibility to invasion by some species increased in landscapes with large core areas and/or less-aggregated patches. Forest understory species and vines tended to respond most favorably to unfragmented forest cover, as might be expected. Thus, even though this study found the expected general landscape patterns for susceptibility to invasion by most species, the influence of landscape composition and configuration on invasion risk can be species specific, and species biology must be considered in predicting responses to land use.

Relative Importance of the Biotic Community, Abiotic Factors, and Landscape Characteristics on Nonnative Species Richness and Abundance

Landscape attributes were especially important for the arrival of nonnative species, whereas biotic factors seemed to regulate their abundance. The combined effects of different types of predictors (biotic, abiotic, and landscape variables) explained more variation than did any individual category of predictor in all analyses. Nonnative species richness and abundance were increased by low

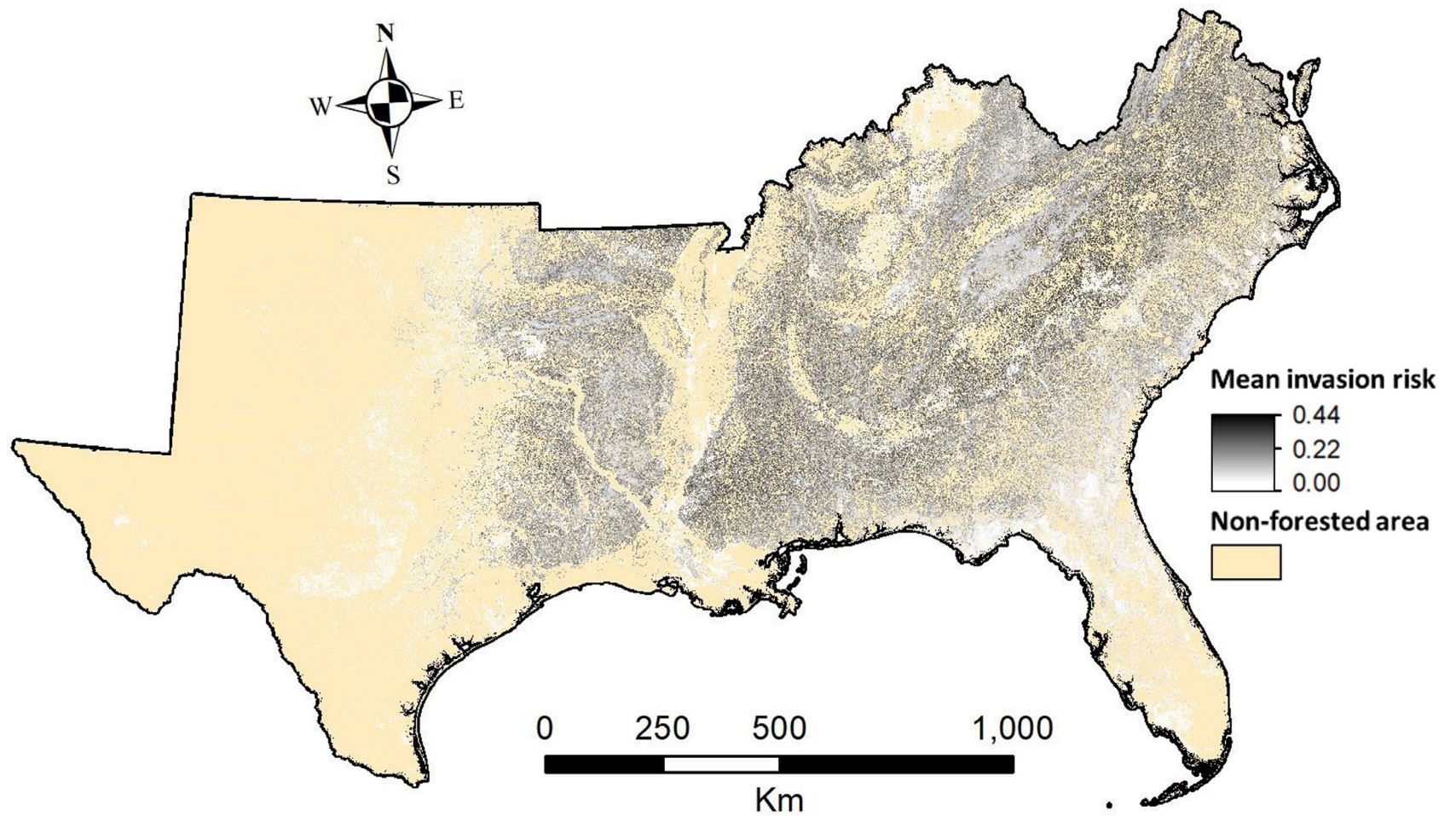


Figure 7.1—Average susceptibility to invasion by all invasive plants species examined for each pixel occupied by forested areas. Darker colors indicate higher risk of invasion, while lighter colors illustrate the opposite (modified from Lázaro-Lobo and others 2021b).

values of native species cover and diversity, high degree of landscape fragmentation, human disturbances, anthropogenic land use, and proximity to transportation systems, especially highways. Nonnative species were less common in areas with low anthropogenic disturbance.

Impacts of Invasive Plants on Tree Regeneration

For Region 8 FIA plots surveyed during 2015–2019, the presence of invasive plants generally was correlated with reduced abundance of seedlings and saplings of desirable tree species (fig. 7.2). Those negative interactions appeared to be independent of environmental niche associations between the invasive species and the desirable trees. Fast-growing native hardwood tree species (e.g., ash (*Fraxinus* spp.), cottonwood (*Populus* spp.), hackberry (*Celtis* spp.), sweetgum (*Liquidambar styraciflua*)), however, had the fewest negative species-wise associations with invasive plants, suggesting that effects of plant invasion on forest regeneration may be mediated, in part, by ecological traits of the tree species present. The invasive plants evaluated in this study also influenced seedling abundance and diversity more strongly than sapling abundance and diversity, which is not unexpected but does reinforce the importance of invasive species management for forest regeneration.

DISCUSSION

These results provide a general idea of environmental effects on invasive species distribution, landscape susceptibility to invasion, and impacts of invasive plants on tree regeneration. Climate, land use, and soil variables were important factors to explain invasive species distributions at the regional scale, but the direction and magnitude of their influence were species-specific. Results also indicate that most of the focal invasive plants are likely to occupy large, forested areas throughout the study region (fig. 7.1). Thus, incorporating estimates of invasion risk into conservation strategies could prevent further establishment of invasive plants in the Southern United States. These studies also demonstrate that landscape composition and configuration highly influence susceptibility to invasion across the Southern United States. Although highly fragmented landscapes with spatially aggregated dispersal corridors were important for invasive species in general, the biology and ecology of some invaders (e.g., forest understory specialists) still provide important, more refined insight for invasion management. Therefore, this knowledge can inform management plans that consider how invasive species distribution is affected by the proportion of the landscape occupied by the different land cover types, the amount of edge in the landscape, and the degree of aggregation and cohesion of patch types, within the context of larger scale environmental drivers. Regarding the invasive species themselves, the greatest threats to tree regeneration were from Japanese honeysuckle, Japanese climbing fern, nonnative privet species, and Chinese tallow (*Triadica sebifera*), each of

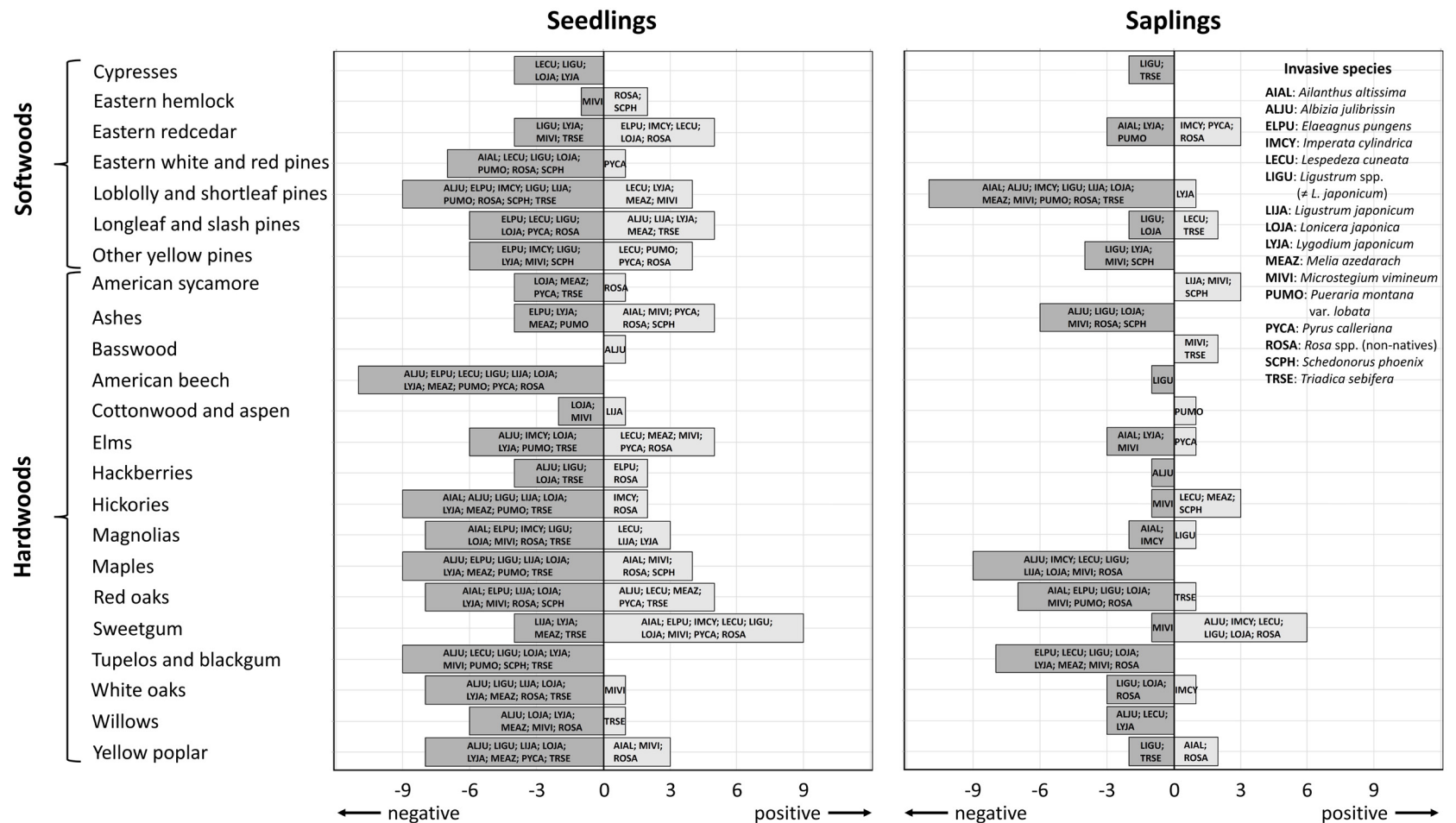


Figure 7.2—Number of significant relationships (with $p < 0.05$) between invasive taxa presence and tree seedling and sapling abundance. Values on the left of the x-axis (negative sign) represent the number of negative relationships, whereas values to the right indicate the number of positive associations. The invasive taxa responsible for the significant relationships are included inside the bars of each tree taxon (modified from Lázaro-Lobo and others 2021a).

which showed negative correlations with the abundance of tree seedlings in at least 12 tree species. Modeling results also showed that 35–40 percent of the region’s forests could be susceptible to invasion by Japanese honeysuckle and nonnative privets; thus, these invasive plants show the greatest potential for long-term, broad-scale negative effects on southern forests, based on this research.

Results also show a hierarchical nature to biotic and abiotic factors in their influence on invasive species distribution and abundance. Land use and land cover can provide significant filters influencing the arrival and establishment of nonnative species. At a finer scale, however, biotic factors may regulate the abundance of nonnative species once they are present within the system. Nonnative species richness and abundance were increased by low values of native species cover and diversity, by anthropogenic land use and disturbance (e.g., developed land cover, agricultural areas, and soil disturbance), by high landscape heterogeneity and edge density, and by close proximity to transportation systems.

In the context of longer time scales, significant potential impacts of invasive plants on tree regeneration exist within the region. This is likely to hamper the sustainable management of Southern U.S. forests (both natural and managed) by the reduction of extant biodiversity, via reduced recruitment success. The seedling stage of desirable tree species in southern forests was negatively correlated with invasive species occurrence. Thus, invasive plant control could be incorporated into management strategies to help ensure the persistence of native forest

communities and the ecological services they provide.

CONCLUSIONS

Integration of multiple environmental variables collected at different spatial scales is pivotal to provide insights into invasive species distributions and landscape susceptibility to invasion. Furthermore, measuring the impact of biological invasions on natural systems illustrates the importance of invasive species within the context of global change.

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Additional Publications Arising From the Project:

- Lázaro-Lobo, A.; Ervin, G.N. 2019. A global examination on the differential impacts of roadsides on native vs. exotic and weedy plant species. *Global Ecology & Conservation*. 17: e00555.
- Lázaro-Lobo, A.; Ervin, G.N. 2021. Wetland invasion: a multi-faceted challenge during a time of rapid global change. *Wetlands*. 41: 64.
- Lázaro-Lobo, A.; Ervin, G.N.; Caño, L.; Panetta, F.D. 2022. Biological invasion by *Baccharis*. In: Fernandes, G.W.; Oki, Y.; Barbosa, M., eds. *Baccharis: from evolutionary and ecological aspects to social uses and medicinal applications*. Cham, Switzerland: Springer Nature: 185–214.
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INTRODUCTION

Limber pine (*Pinus flexilis*) and Rocky Mountain bristlecone pine (*P. aristata*) are ecologically important five-needle pine species in the Southern Rocky Mountains. Limber pine and southwestern white pine (*P. strobiformis*), a species native to southern New Mexico and Arizona, also hybridize across a broad zone spanning the New Mexico-Colorado border (Menon and others 2018). While none of these pines have commercial value, they are keystone species in that they provide habitat and a food source for wildlife, regulate snow retention and runoff, facilitate the establishment of late successional tree species at treeline, and maintain cover on harsh, rugged sites where few other tree species can grow (Schoettle 2004, Steele 1990).

Five-needle pine populations are declining in many parts of the Rocky Mountains, largely due to the combined impacts of the nonnative, invasive disease known as white pine blister rust (WPBR; *Cronartium ribicola*) and mountain pine beetle (MPB; *Dendroctonus ponderosae*) (Burns and others 2023, Schoettle and others 2022, Shepherd and others 2018). While mortality of pines caused by MPB has been significant in northern Colorado, very little mortality has been reported in southern Colorado and northern New Mexico (Bentz and others 2021, Burns and others 2023, Gibson and others 2008, Schoettle and Coop 2017). WPBR has only recently been detected in this area in the Sangre de Cristo Mountains (limber pine and Rocky Mountain bristlecone pine) and Wet Mountains (limber pine) of southern Colorado in 2003 (Blodgett and Sullivan 2004, Burns 2006)

and on the Santa Clara Pueblo and Santa Fe National Forest (limber pine) of northern New Mexico in 2006 (Conklin and others 2009). A rangewide survey of Rocky Mountain bristlecone pine in 2010–2011 found no new outbreak areas and predominantly healthy stands (Schoettle and Coop 2017). Comprehensive surveys of limber pine in southern Colorado and northern New Mexico are lacking. The purpose of this study was to gather baseline information on the distribution, abundance, and health of five-needle pines; characterize stand structure and site conditions; and determine the extent and severity of WPBR in the study area.

METHODS

The study area extended across the range of five-needle pines in southern Colorado and northern New Mexico focusing on previously unsurveyed areas where WPBR has not been detected. Potential survey sites were identified using local knowledge, vegetation layers, and records of herbaria specimens from SEINet (<https://swbiodiversity.org/seinet/>). Plots were established following the methods of Burns and others (2011).

Site data collected included geospatial coordinates, elevation, aspect, slope, slope position, stand structure, stand type (Rocky Mountain bristlecone pine, limber pine, or both species), disturbances, and the presence of alternate WPBR hosts (currants and gooseberries (*Ribes* spp.), Indian paintbrush (*Castilleja* spp.), and louseworts (*Pedicularis* spp.)). Data collected for all trees (≥ 4.5 feet tall) included species, diameter at breast

CHAPTER 8

Health and Abundance of Five-Needle Pines in Southern Colorado and Northern New Mexico

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height (d.b.h.; 4.5 feet), health status (healthy, declining, dying, recent dead, or old dead), crown class, and damages and their severities. Additional data collected for five-needle pines included live crown ratio (percent), recent canopy kill (percent), presence of mature green cones, and signs and symptoms of WPBR. This study did not attempt to distinguish between limber pine and limber-southwestern white pine hybrids and refer to both as limber pine for the purpose of this study. Approximately five dominant or codominant trees, including at least two five-needle pines, were cored at each site to determine stand age structure. Within each plot, three circular, fixed-area subplots (1/100 acre, 11.8-foot radius) were installed at equal increments along the transect line. Within each subplot, ground cover was estimated as percentage of bare soil, vegetation, lichen and moss, rock, litter, and trees and logs. Additionally, regeneration (<4.5 feet tall) was tallied by species and WPBR presence/absence for five-needle pines. Five-needle pine regeneration was also tallied across the entire plot. To cover as much ground as possible, researchers also conducted walk-through transects along roads adjacent to formally surveyed sites, which consisted of examining and tallying all five-needle pines by species and WPBR status within 25 feet perpendicular to the measured transect line in both directions. Data analyses were conducted using SAS software.

RESULTS

Site and Stand Characteristics

Fifty-eight plots were established in 2018–2019 on the Rio Grande (34), Carson (8), San Juan (3), and Gunnison National Forests (8), and along the eastern edge of the San Luis Valley in Great Sand Dunes National Park and Preserve and private land (5) (fig. 8.1). Limber pine occurred in 47 plots and Rocky Mountain bristlecone pine occurred in 26. Researchers examined 4,016 trees (3,372 live) within plots including 621 (602 live) limber pines and 722 (688 live) Rocky Mountain bristlecone pines. Plots ranged in elevation from 8,087 to 11,580 feet and were 9,671 feet on average. Most plots were on middle (47 percent) and upper (42 percent) slope positions and the majority were on south-facing slopes (45 percent). Average slope was 17 degrees (range 1–43). *Ribes* spp. occurred on 53 percent, *Castilleja* spp. on 17 percent, and *Pedicularis* spp. on 7 percent of plots. Wax currant (*R. cereum*) was the most common *Ribes* spp. followed by whitestem gooseberry (*R. inerme*), gooseberry currant (*R. montigenum*), and Mescalero currant (*R. mescalerium*). The most common site disturbance observed was fire with 41 percent of sites exhibiting evidence of previous fire.

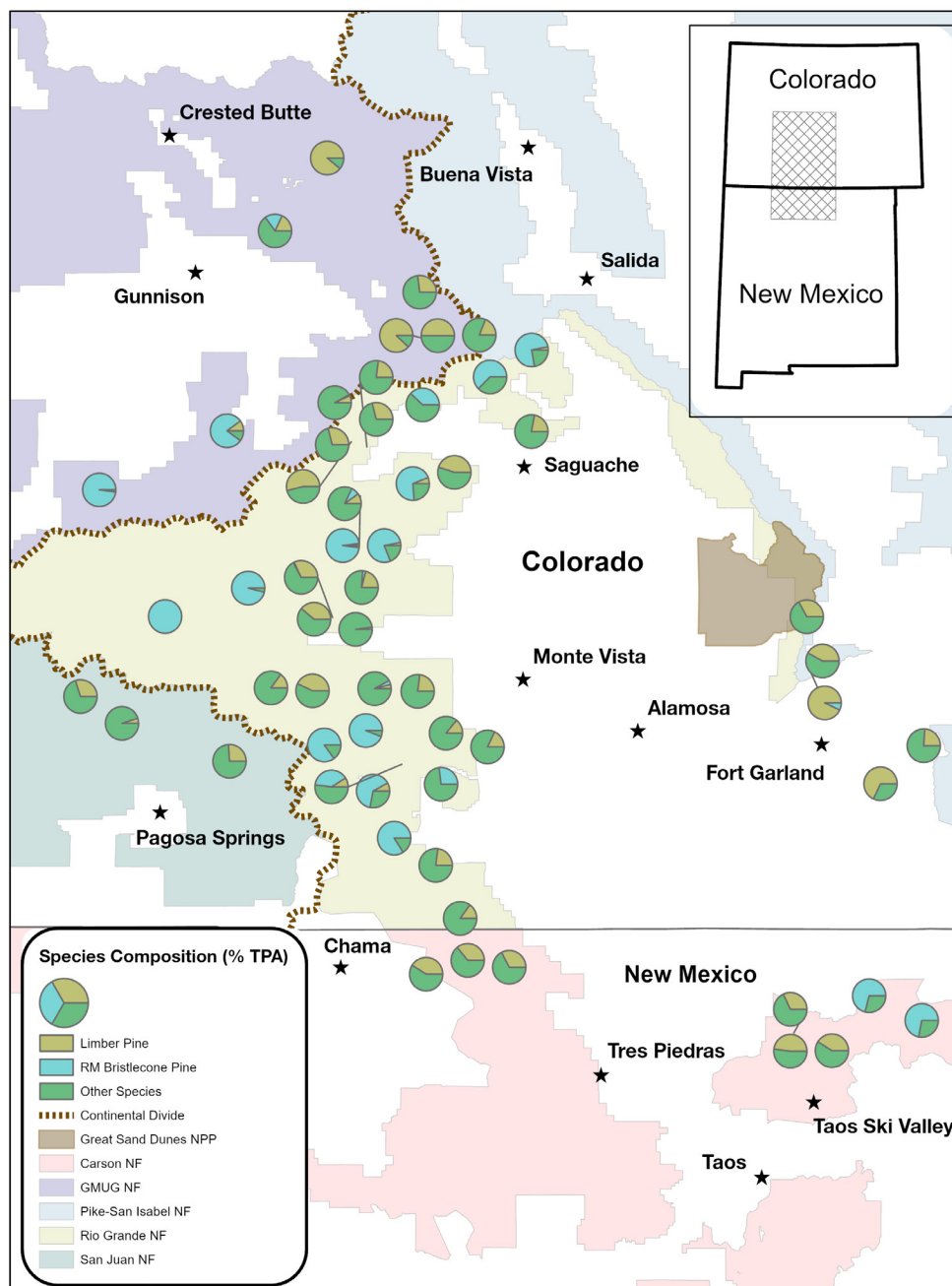


Figure 8.1—Locations of 58 limber pine and Rocky Mountain bristlecone pine health monitoring plots in southern Colorado and northern New Mexico. Pie charts represent species composition based on percentage of trees per acre made up of limber pine (dark green), Rocky Mountain bristlecone pine (light green), and other species (magenta). NF = National Forest; NPP = National Park and Preserve; RM = Rocky Mountain; TPA = trees per acre.

Live tree density of all species was 253 (range 52–1,279) trees per acre on average. Limber pine occurred as a smaller component of mixed species stands than Rocky Mountain bristlecone pine (table 8.1). In stands with limber pine, quaking aspen (*Populus tremuloides*) was the most abundant species followed by limber pine, Douglas-fir (*Pseudotsuga menziesii*), Rocky Mountain bristlecone pine, Engelmann spruce (*Picea engelmannii*), white fir (*Abies concolor*), and ponderosa pine (*Pinus ponderosa*). Rocky Mountain bristlecone pine was the most abundant species in stands where it occurred, followed by quaking aspen, Douglas-fir, limber pine, Engelmann spruce, white fir, and ponderosa pine. Stand basal area for trees was 79 (range 15–210) square feet per acre on average. Both five-needle pine species contributed the most to basal area where they occurred with Rocky Mountain bristlecone pine making up 53 percent and limber pine 38 percent of basal area on average (table 8.2). Average d.b.h. was 8.2 and 6.1 inches for limber pine and Rocky Mountain bristlecone pine, respectively. While quaking aspen was abundant, it only contributed 8 percent to total basal area and d.b.h. was only 3.6 inches on average. Other than

five-needle pines, Douglas-fir contributed the most to basal area in all stands (table 8.2). Most stands were intermediate in age with limber pines being 168 (range 46–388) and Rocky Mountain bristlecone pines 207 (range 68–383) years old on average. The oldest trees were Douglas-firs, which were approximately 550 years old.

Five-needle pine regeneration was present in only 69 percent of all surveyed stands and 17 percent had no regeneration of any species. Plots had 18.5 five-needle pine seedlings per acre on average. Unlike what others have reported, this study found no relationships between five-needle pine regeneration density and stand or site factors (Cleaver and others 2016, Schoettle and Coop 2017). The most abundant regenerating species were quaking aspen and Douglas-fir. WPBR was not detected on regeneration within plots. Active cone production was found on 15.8 percent and 18.6 percent of all live limber pines and Rocky Mountain bristlecone pines, for an average of 7.1 and 9.6 cone-bearing limber pines and Rocky Mountain bristlecone pines per acre, respectively. Cone production was light during both years, and neither was a mast year.

Table 8.1—Comparison of live tree density (trees per acre ≥ 4.5 feet tall) by species between all sites and sites where limber pine and Rocky Mountain bristlecone pine occurred

Species	All sites				Sites with limber pine ^a				Sites with Rocky Mountain bristlecone pine ^b			
	# ^c	Mean TPA (SD)	Range	% of TPA	# ^c	Mean TPA (SD)	Range	% of TPA	# ^c	Mean TPA (SD)	Range	% of TPA
Limber pine	47	45 (40)	0–135	18%	47	56 (37)	4–132	21%	15	22 (36)	0–126	8%
Rocky Mountain bristlecone pine	26	52 (90)	0–365	21%	15	25 (62)	0–305	10%	26	115 (104)	4–365	40%
Quaking aspen	28	73 (183)	0–1,183	29%	25	85 (200)	0–1,183	33%	12	84 (265)	0–1,183	30%
Douglas-fir	39	41 (58)	0–278	16%	34	45 (59)	0–278	17%	13	37 (75)	0–278	13%
Engelmann spruce	30	16 (25)	0–109	6%	26	17 (26)	0–109	7%	15	17 (26)	0–78	6%
Ponderosa pine	21	10 (17)	0–78	4%	18	10 (17)	0–78	4%	7	7 (14)	0–48	2%
White fir	11	9 (27)	0–131	4%	11	11 (30)	0–131	4%	2	2 (10)	0–52	1%
Other ^d	12	8 (27)	0–139	3%	13	9 (29)	0–139	4%	7	1 (2)	0–9	0%
Overall	58	253 (203)	52–1,279	—	47	260 (219)	57–1,279	—	26	286 (242)	52–1,279	—

SD = standard deviation; TPA = trees per acre.

^a Includes all plots with at least one limber pine.

^b Includes all plots with at least one bristlecone pine.

^c The number of plots where the species occurred.

^d Other species include Rocky Mountain juniper (*Juniperus scopulorum*), twoneedle pinyon (*Pinus edulis*), lodgepole pine (*P. contorta*), and narrowleaf cottonwood (*Populus angustifolia*).

Table 8.2—Comparison of live tree basal area (square feet per acre ≥ 4.5 feet tall) by species between all sites and sites where limber pine and bristlecone pine occurred

Species	All sites				Sites with limber pine ^a				Sites with Rocky Mountain bristlecone pine ^b			
	# ^c	Mean BA (SD)	Range	% of BA	# ^c	Mean BA (SD)	Range	% of BA	# ^c	Mean BA (SD)	Range	% of BA
Limber pine	47	24 (35)	0–178	30%	47	30 (36)	0.1–178	38%	15	14 (34)	0–170	18%
Rocky Mountain bristlecone pine	26	18 (31)	0–123	23%	15	9 (25)	0–123	12%	26	39 (37)	0.1–123	53%
Quaking aspen	28	6 (11)	0–50	8%	25	6 (12)	0–50	8%	12	5 (8)	0–26	6%
Douglas-fir	39	14 (19)	0–64	18%	34	16 (20)	0–64	19%	13	10 (17)	0–64	13%
Engelmann spruce	30	5 (10)	0–37	6%	26	5 (10)	0–37	7%	15	3 (4)	0–18	3%
Ponderosa pine	21	6 (13)	0–59	8%	18	6 (13)	0–59	8%	7	4 (10)	0–40	6%
White fir	11	3 (12)	0–66	4%	11	4 (13)	0–66	5%	2	0.1 (0.5)	0–3	0%
Other ^d	12	2 (10)	0–62	3%	13	3 (11)	0–62	4%	7	0.2 (0.5)	0–2	0%
Overall	58	79 (40)	15–210	—	47	80 (43)	19–210	—	26	74 (38)	15–172	—

BA = basal area; SD = standard deviation.

^a Includes all plots with at least one limber pine.

^b Includes all plots with at least one bristlecone pine.

^c The number of plots where the species occurred.

^d Other species include Rocky Mountain juniper (*Juniperus scopulorum*), twoneedle pinyon (*Pinus edulis*), lodgepole pine (*P. contorta*), and narrowleaf cottonwood (*Populus angustifolia*).

Current Health Status of Surveyed Trees

Most five-needle pine trees surveyed were healthy, with over 90 percent of limber pines and 80 percent of Rocky Mountain bristlecone pines having no or very minor damages. The most common damages on limber pine were animal damage and twig beetles. Twig beetles were the most common damage agent in Rocky Mountain bristlecone pine followed by animal damage and fire. Less-common damages observed in both species included bark beetles (*Ips* spp. and MPB), cytospora canker (*Cytospora* spp.), western gall rust (*Endocronartium barknesii*), needle cast (*Bifusella* spp.), and WPBR. Dwarf mistletoe (*Arceuthobium cyanocarpum*) was identified on only one Rocky Mountain bristlecone pine. Mean percent recent canopy kill was significantly greater ($p < 0.0001$) in Rocky Mountain bristlecone pine (12.5 ± 0.5) than limber pine (5.6 ± 0.4) and was positively correlated with percent ground cover of bare soil, Rocky Mountain bristlecone pine basal area, and d.b.h. and negatively correlated with limber pine basal area and northerly aspect. Much of the dieback was associated with twig beetles, but many trees had no obvious cause. Incidence of WPBR in plots was very low (<1 percent). All infected trees detected within formal plots were limber pines located in Great Sand Dunes National Park and Preserve (fig. 8.2). Limber pine health status was negatively correlated with slope and ground cover dominated by vegetation and rock.

Overall, 644 dead trees (>4.5 feet tall) were surveyed (16 percent of all trees) and only a small portion of these were Rocky Mountain bristlecone pine (5 percent) or limber pine (3 percent). Most mortality was quaking aspen (61 percent), Engelmann spruce (17 percent), or Douglas-fir (9 percent). Of the 15 recently killed five-needle pines, 6 were killed by bark beetles (MPB and *Ips* spp.). No signs or symptoms of WPBR were evident on surveyed snags.

Walk-Through Transects

Twenty-five walk-through transects (fig. 8.2), averaging 1.2 miles long, were completed and included observations of an additional 2,187 limber pines and 1,154 Rocky Mountain bristlecone pines. WPBR was detected at two new locations (fig. 8.2).

The first new location where WPBR was detected was a riparian area on private land about 35 miles east of Alamosa, CO, and 25 miles southeast of the previously described outbreak near Mosca Pass, CO (Blodgett and Sullivan 2004, Burns 2006). WPBR was confirmed on whitestem gooseberry and both limber pine and Rocky Mountain bristlecone pine at this site. Overall, 47 percent of trees were infected with WPBR and incidence was higher in limber pine (18 of 29 trees) than Rocky Mountain bristlecone pine (3 of 16 trees). Infected trees had five cankers on average and no rust-caused mortality was detected. This is only the second location where WPBR has been confirmed on Rocky Mountain bristlecone pine within its natural habitat (Blodgett and Sullivan 2004).

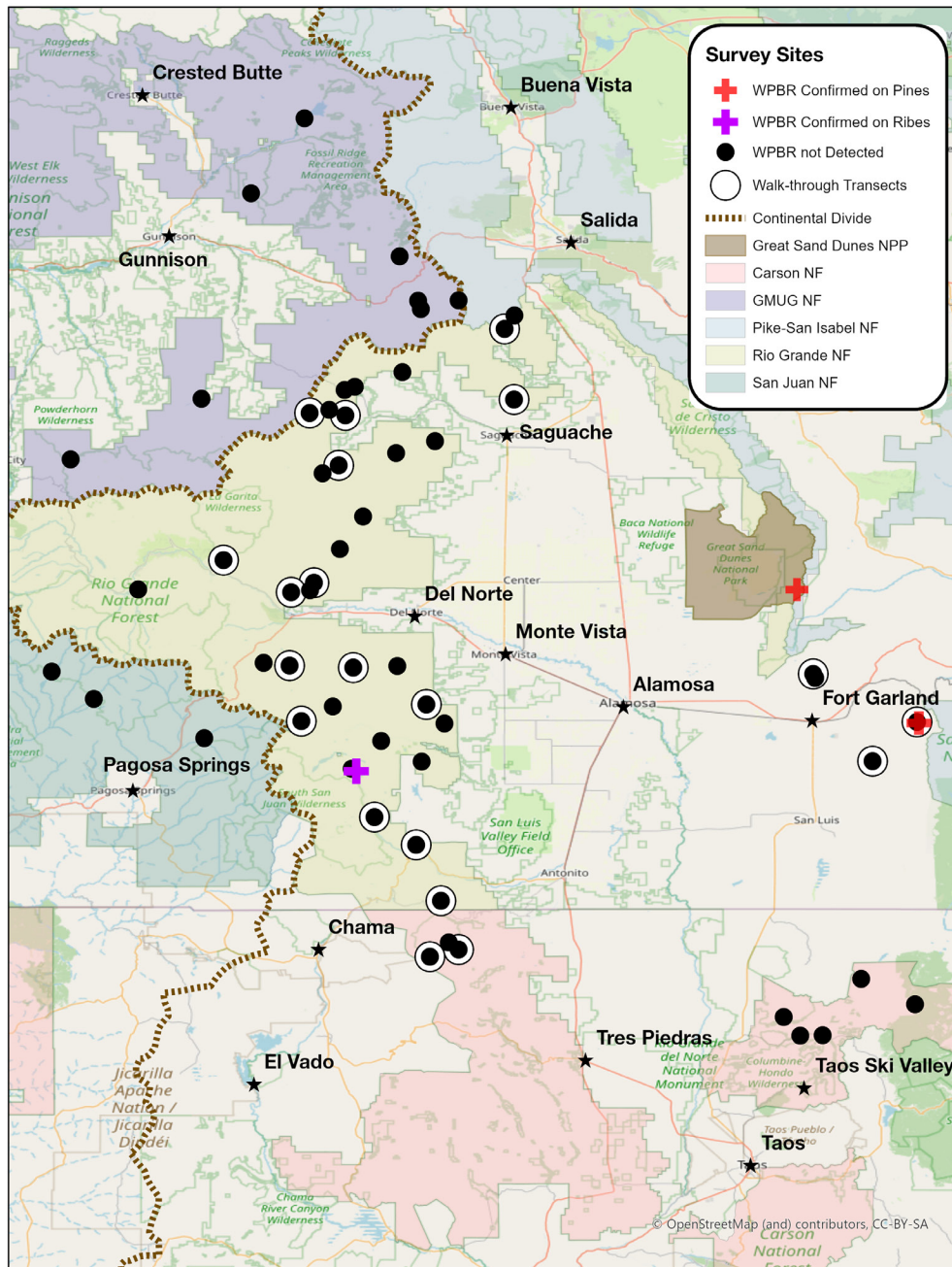


Figure 8.2—Distribution of white pine blister rust in surveyed plots, walk-through transects, and field observations. NF = National Forest; NPP = National Park and Preserve; WPBR = white pine blister rust.

The second new location where WPBR was detected was in the Lake Fork drainage, on the Conejos Peak Ranger District, Rio Grande National Forest, about 25 miles south of Del Norte, CO. An infected whitestem gooseberry bush was identified along the creek, but WPBR was not detected on surrounding pine hosts. This is roughly 60 miles west of the nearest outbreak area in Great Sand Dunes National Park and Preserve (Blodgett and Sullivan 2004, Burns 2006). Infected leaf samples were collected and submitted to a lab at Colorado State University for molecular analysis, which confirmed *Cronartium ribicola*.

DISCUSSION AND CONCLUSIONS

Limber pine and Rocky Mountain bristlecone pine are well distributed in southern Colorado and northern New Mexico but occur at lower frequencies than in other parts of their ranges (Burns and others 2023, Cleaver and others 2015, Schoettle and Coop 2017, Smith and others 2013). Density of live limber pine trees of 56 stems per acre observed in this study was considerably lower than the 126 stems per acre that Cleaver and others (2015) reported from 508 measured plots in northern Colorado, Wyoming, and Montana. However, basal area (square feet per acre) of live limber pine was slightly higher in this study (30 vs. 25 square feet per acre, respectively), indicating that limber pines, while less frequent, were larger on average. This could be explained, in part, by the low incidence of recent MPB activity in the plots. Most stands comprised non-pines, with very few pure stands of limber pine and Rocky Mountain bristlecone pine. MPB outbreaks

are more likely to occur in areas with high basal area of susceptible pine hosts (Fettig and others 2007). Moreover, MPB has historically not been found attacking trees in New Mexico. Similarly, stems per acre of live Rocky Mountain bristlecone pine were considerably lower (115 stems per acre) than the 156 stems per acre Schoettle and Coop (2017) reported from 61 plots throughout the range of Rocky Mountain bristlecone pine. Live Rocky Mountain bristlecone pine basal area (39 square feet per acre) was also well below the 96 square feet per acre that Schoettle and Coop (2017) reported as the rangewide average. Most stands were mixed species with very few pure stands of either species.

Limber pines were healthier in this study than what has been reported in the northern portions of their range (Burns and others 2023, Cleaver and others 2022, Smith and others 2013). Over 90 percent of limber pines were healthy compared to only 50 percent classified as healthy in a 2016–2017 survey by Burns and others (2023). Recent mortality (1 percent), WPBR incidence (1 percent), and mortality attributed to bark beetles were also substantially lower in the southern portion of limber pine's range (compared to Burns and others 2023, Cleaver and others 2022, and Smith and others 2013). Findings were more in line with what Looney and others (2015) reported from their 2010–2011 surveys of southwestern white pine in New Mexico and Arizona. Similarly, Rocky Mountain bristlecone pines were healthier in this study compared to Schoettle and Coop's (2017) rangewide survey (80 percent vs. 71 percent, respectively).

While no new disease outbreak areas were detected in formal plots, two new infection areas were identified in walk-through surveys. *Cronartium ribicola* was confirmed on whitestem gooseberry and both limber pine and Rocky Mountain bristlecone pine approximately 25 miles southeast of the previously described outbreak in Great Sand Dunes National Park and Preserve (fig. 8.2). WPBR was also confirmed on whitestem gooseberry about 60 miles west of the park on the Rio Grande National Forest (fig. 8.2). No infected pines were detected in this area. These new outbreak areas confirm that inoculum is present and disease spread is occurring. Detection of WPBR in pine hosts is especially difficult during the early stages of invasion (Kozhar and others 2023). More frequent surveys coinciding with sporulation on pines are needed to improve estimates of disease distribution at the infection front.

Five-needle pine regeneration was found on fewer sites and at lower densities than other studies in these host types (Burns and others 2023, Cleaver and others 2016, Schoettle and Coop 2017). Burns and others (2023) reported limber pine regeneration on 76 percent of sites with an average density of 73 stems per acre. Similarly, Schoettle and Coop (2017) reported Rocky Mountain bristlecone pine regeneration averaged 60 seedlings per acre. In contrast, five-needle pine regeneration was found on only 69 percent of plots with an average density of 18.5 seedlings per acre. While Rocky Mountain bristlecone pine regeneration is common in the Rocky Mountain bristlecone pine forest type, recent studies suggest otherwise for limber

pine (Goeking and Windmuller-Campione 2021). Goeking and Windmuller-Campione (2021) analyzed a large U.S. Department of Agriculture, Forest Service, Forest Inventory and Analysis dataset and found limber pine regeneration occurred more abundantly in forest types dominated by other (non-five-needle pine) species. A more thorough assessment of the distribution and abundance of regeneration is needed to determine if this is the case in southern Colorado and northern New Mexico.

The combined pressures of WPBR, bark beetles, and climate change threaten five-needle pine populations of western North America (Burns and others 2023, Kearns and others 2014, Schoettle and others 2022). Fortunately, five-needle pines, particularly limber pine, were comparatively healthy in southern Colorado and northern New Mexico. However, the lack of regeneration is concerning and suggests management intervention may be needed to stimulate recruitment on some sites. Forest management plans could include measures to increase resiliency of five-needle pines (Schoettle and others 2019a, 2019b). In areas where WPBR is not yet or only recently established, proactive silvicultural treatments may position the ecosystem to facilitate the evolution of disease resistance in tree populations before they are impacted (Schoettle and Snieszko 2007). Diversifying age class structure, by stimulating regeneration in healthy stands, will provide a larger population for resistance selection upon invasion and simultaneous selection in both the younger and older cohorts (Schoettle and others 2019a, 2019b, 2022). Periodic forest health

monitoring is warranted as disease spread is likely to continue over time in the study area.

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INTRODUCTION

Climate change is threatening dryland ecosystems globally due to aridification coupled with an increase in the frequency and severity of drought (Bradford and others 2020, Dai 2013, Giorgi and others 2011). These ongoing changes could lead many dryland species to tipping points beyond which they cannot recover, resulting in degraded ecosystem conditions and permanently altered landscapes (Bestelmeyer and others 2015, Hoover and others 2020). The American Southwest serves as a “leading edge” of these climate change impacts due to its rapid rate of warming and anomalous increases in projected aridity (Cook and others 2020). Indeed, the region has been in the grip of a “megadrought” since 2000, the driest multidecadal period of the last 1,200 years (Williams and others 2022). Over this period, drought-induced tree mortality events across the region have altered the face of the landscape permanently. In 2018, a severe drought caused widespread dieback of two dominant southwestern tree species, Utah juniper (*Juniperus osteosperma*, hereafter referred to as juniper) and twoneedle pinyon (*Pinus edulis*, hereafter referred to as pinyon). Drought-induced pinyon dieback has occurred several times over the past three decades. However, this dieback of juniper (generally considered one of the most drought-tolerant tree species in the United States) is likely unparalleled in severity and spatial extent (Campbell and others 2020, Kannenberg and others 2021). This raises significant concerns regarding the fate of woodland ecosystems in the American Southwest as the climate continues to change. The recent severe drought in 2020–2021 was also associated with widespread juniper

mortality, largely clustered in northern Arizona and across southern Utah. However, prior to this project, the extent and spatial patterns of juniper dieback had yet to be resolved, along with its underlying mechanisms and impacts on ecosystem function.

This study leveraged a multiscale approach spanning ecophysiology, dendrochronology, pathology/entomology, and remote sensing to ask questions pertaining to three key objectives: (1) quantify the spatial patterns of juniper dieback, (2) diagnose the drivers of juniper dieback, and (3) understand the ecological functioning of pinyon-juniper woodlands during the megadrought.

METHODS

Detecting Pinyon-Juniper Dieback

Initial reports of pinyon-juniper dieback were received in fall 2018 and were clustered in San Juan County, UT. The majority of this dieback was observed in juniper trees located in juniper-dominant stands (i.e., in lower elevation pinyon-juniper woodlands), though significant pinyon dieback occurred across the landscape despite the general scarcity of the species. To quantify the extent and severity of these dieback events, a multiscale remote sensing approach was combined with 35 field plots to generate maps of pinyon-juniper woodland dieback following the 2018 drought (Campbell and others 2020). Briefly, aerial drone imagery was paired with Lidar data to delineate individual tree crowns and quantify their percentage of dieback based on canopy color. These stand-level dieback maps were then paired with Landsat imagery across the county, and a machine learning algorithm was developed to

CHAPTER 9

Linking Physiology to Landscape-Scale Dieback of Utah Juniper During a Megadrought

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predict dieback based on a number of climatic, vegetation, and topo-edaphic variables. These maps were then validated with field surveys from 35 sites.

Underlying Mechanisms of Juniper Dieback

To understand the mechanisms behind juniper dieback, 12 field plots were centered around Blanding, UT, that encompassed a gradient of low (<5 percent) to severe (>50 percent) canopy dieback (Kannenberg and others 2021). At each plot, stands were surveyed in spring and fall 2019 to quantify canopy dieback, leaf water potential and branch hydraulic conductivity were measured to understand the physiological ramifications of drought for plant water relations, and sampling was conducted to detect the presence of any insects or diseases that could be contributing to the observed dieback.

Carbon-Water Cycling in Pinyon-Juniper Woodlands

Finally, this study sought to understand the interactions between meteorological conditions (e.g., vapor pressure deficit, light availability, soil moisture), juniper physiology, and ecosystem-scale carbon-water fluxes. To do so, an eddy covariance flux tower was installed at one of the juniper dieback field sites. This tower (operational from

2019 through 2023) measured net ecosystem exchange, along with a suite of environmental variables, most notably a depth profile of soil moisture from surface to bedrock. Stand surveys also conducted at this location amassed a record of continuous measurements of juniper water potential in 2021 (Kannenberg and others 2023).

RESULTS AND DISCUSSION

Detecting Juniper Dieback

Our multiscale remote sensing approach found that pinyon-juniper dieback was widespread across San Juan County. Almost 40 percent of pinyon-juniper woodlands had experienced some detectable level of dieback in 2019, with some patches exceeding 50 percent dieback (see Campbell and others 2020, figs. 2 and 8). These areas of high-severity dieback were clustered into small hotspots across the landscape. While the drivers of these hotspots remain largely unresolved, high canopy cover and low winter precipitation in 2018 were the most important drivers of dieback in 2019, which points to the importance of deeper moisture stores (that are recharged by winter precipitation) and canopy microclimate for regulating drought-induced dieback.

Underlying Mechanisms of Juniper Dieback

While the remote-sensing approach did identify some covariates with dieback severity, their explanatory power was relatively low. Thus, this study next took a physiological approach for understanding the mechanisms of juniper dieback at the network of field sites. Three hotspots of juniper dieback (>50 percent dieback) were identified and plots were set up to radiate from the center of those hotspots. This approach revealed a very strong elevational and climatic signal, whereby dieback of both pinyon and juniper was much higher at hotter, drier, lower elevation sites (fig. 9.1). Juniper dieback at these sites was associated with losses of branch hydraulic conductivity, a widespread and severe consequence of drought stress that is known to precede mortality (Anderegg and others 2013). Decreases in hydraulic conductivity were, surprisingly, the only physiological consequence of drought observed in these trees; sampling for insects and diseases revealed no possible vectors of dieback, and any less harmful insects or diseases had a very low prevalence. Thus, the etiology of this juniper dieback seems straightforward. A multidecadal drought, in conjunction with acute drought stress in 2018, caused significant hydraulic damage in juniper trees that led to their dieback.

Carbon-Water Cycling in Pinyon-Juniper Woodlands

This study next attempted to develop a predictive understanding of how the megadrought is impacting pinyon-juniper woodland ecosystem function by quantifying the drivers of pinyon-juniper carbon and water fluxes and linking that knowledge to ongoing hydrological trends. By correlating daily fluxes of carbon and water (along with juniper midday water potential) with various environmental drivers, soil moisture was overwhelmingly the most important driver of pinyon-juniper ecosystem function (see Kannenberg and others 2023, fig. 3). Specifically, carbon uptake was highly sensitive to fluctuations in both shallow and deep soil moisture pools, while evapotranspiration was highly sensitive to shallow soil moisture. Atmospheric aridity, air temperature, and light availability did little to control carbon-water fluxes or juniper hydraulic status. The importance of deep soil moisture for pinyon-juniper woodlands is unsurprising yet concerning. In 3 years of monitoring, significant recharge of deep soil moisture pools only occurred in 1 year, due to the accumulation of a moderate snowpack that melted and percolated to deeper layers in the spring. One of the most widespread ramifications of the ongoing megadrought has been reductions in winter precipitation, which have exceeded 50 percent in some regions (Mote and others 2018, Siirila-Woodburn and others 2021). If these reductions in snow accumulation continue, results suggest that pinyon-juniper woodland health will become increasingly reliant on transient, and inconsistent, summer monsoonal rains.

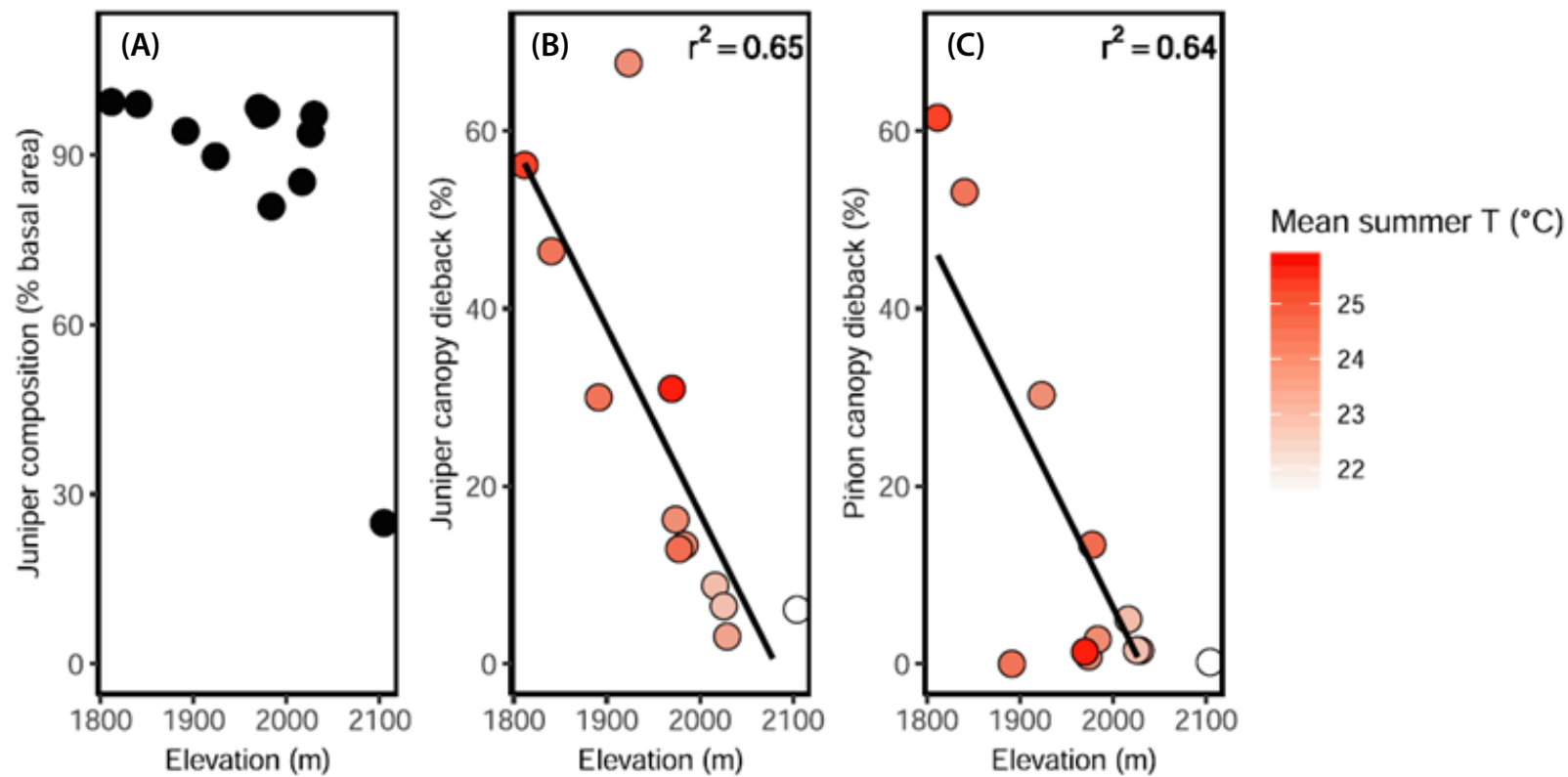


Figure 9.1 —Relationships between site elevation and (A) juniper species composition, (B) juniper canopy dieback, and (C) piñon canopy dieback. Color of points represents the mean summer (June–August) temperature (T) during the 2018 drought at each site. From Kannenberg and others (2021).

CONCLUSIONS

In conclusion, widespread juniper dieback occurred in southeastern Utah following the 2018 drought, with numerous hotspots of severe dieback. While dieback was linked to recent acute drought stress, the explanatory power was still low, and hotspots of juniper dieback may be due to spatially heterogeneous subsurface features (e.g., a lack of deep moisture access). Moreover, the role of the megadrought in predisposing junipers to dieback remains unknown. Ongoing work seeks to expand this remote sensing approach to neighboring States to capture other dieback events that occurred in 2020–2021, and tree-ring chronologies will be used to reconstruct the impacts of the megadrought on tree growth within observed dieback hotspots. The speed and scale of this drought-induced juniper dieback is likely to be historically unprecedented in the region and foreshadows an uncertain future for pinyon-juniper woodlands as the region continues to get warmer and drier.

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INTRODUCTION

Nonnative species are a major threat to the composition and functioning of global forests (Parry and Teale 2011). Impacts of nonnative tree-feeding insects often occur in concert with impacts of native insects, yet interactive effects between nonnatives and natives on tree performance are poorly understood and have rarely been investigated. Nonnative insects could make trees more or less susceptible to native insects, depending on the magnitude of impacts of nonnatives on tree defense, vigor, and host quality.

At the very southern extent of its range in Arizona, Engelmann spruce (*Picea engelmannii*) is being attacked by the nonnative spruce aphid (SA; *Elatobium abietinum*). The SA was first found in the Southwestern United States in 1976 in urban settings, and then in wildland settings in Arizona beginning in 1989 (Lynch 2004). Outbreak frequency increased beginning in the early 2000s, especially in high-elevation spruce-fir-dominated forests of eastern Arizona (Grady 2019, Lynch 2009). Attacks from the native spruce beetle (SB; *Dendroctonus rufipennis*) on Engelmann spruce and the intensity of regional drought also increased during this period, which led to high tree mortality (Grady 2019). The SB has been observed attacking SA-defoliated trees (Lynch 2004, 2009), although whether SB attacks are facilitated by SA defoliation is uncertain. The potential for SA to incite SB outbreaks is of great concern for managers as spruce populations are already declining across the West and SA outbreaks may increase with warmer winters (Straw and others 2019). No previous studies

have investigated impacts of SA defoliation on SB performance or defenses of Engelmann spruce. This project addressed impacts of SA defoliation on SB performance and tree defenses with a novel experiment during an exceptional multidecadal megadrought (2000–2021).

METHODS

This study included an experiment to evaluate whether defoliation by the nonnative SA affects resistance of Engelmann spruce to the native SB in a natural mixed-conifer forest near Greens Peak in the Apache-Sitgreaves National Forests in the White Mountains of east-central Arizona (34.067017, -109.553811) between elevations of 2895 and 2987 m. Selection of this location was based on a well-documented history of SA defoliation of Engelmann spruce by annual aerial surveys and site visits by U.S. Department of Agriculture, Forest Service, Forest Health Protection personnel in the vicinity of Greens Peak (Grady 2019). Spruce in the area incurred defoliation between 2015 and 2018, with a particularly severe defoliation event during winter 2017–2018 (USDA Forest Service 2019). SA defoliation varied substantially among individual trees, ranging from low to high (see next paragraph).

In October 2020, researchers selected 30 Engelmann spruce trees with a range of recent SA defoliation to represent a gradient of SA attack and impact. This approach assumed that the amount of tree defoliation approximated the severity of SA attack. Direct data on the number of aphids attacking each tree were

CHAPTER 10

Does Spruce Aphid Defoliation Alter Engelmann Spruce Resistance to Spruce Beetle?

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lacking. Criteria for selection was diameter at breast height (d.b.h.) between 30 and 43 cm and location within 50 m of a forest road to facilitate access. A team of four experienced observers scored SA defoliation of each tree using a crown rating system conceptually based on the dwarf mistletoe (*Arceuthobium* spp.) rating system (Lynch 2004). Briefly, the live tree crown of each tree was divided into thirds with each section receiving a rating from 0 to 3: 0 = no evidence of defoliation, 1 = 1–33 percent of foliage missing or affected (chlorotic), 2 = 34–66 percent, and 3 = 67 percent or more. The scores were summed to give each tree a rating ranging from 0 to 9. These scores were used to group trees into three defoliation classes, with 10 trees per class: low (score of 3), medium (score between 4 and 6; 2 trees with score of 4, 3 trees with score of 5, and 5 trees with score of 6), and high (score between 7 and 8; 8 trees with score of 7 and 2 trees with score of 8).

In April 2021, samples of phloem and resin were taken from each tree at two breast-height locations at north and south aspects. A 2.54-cm arch punch extracted a disk of phloem that was immediately put in a plastic bag and frozen (-20 °C) until extraction of terpenes. Thickness of each phloem disk was measured with a digital micrometer. Resin that flowed from the phloem wounds over 7 days after wounding was collected using a small aluminum funnel and test tube and then its mass was measured.

In May 2021, one SB pheromone bait (containing frontalin; item # 3300, spruce beetle/Douglas-fir beetle bait; purchased from Synergy, Delta, BC, Canada) was attached to each tree in an attempt to instigate mass SB attack. Baits were replaced in July 2021 and remained on the trees until September 2021. All trees were surveyed in September 2021 to assess SB success by removing a small (3.8 x 7.6 cm) section of bark above four to six pitch tubes per tree to assess whether galleries were established and if beetles were developing. For each tree, the percentage of pitch tubes with successful attacks was determined based on galleries and larval development.

In the second fall after baiting of trees in 2021 with SB pheromones (November 2022), three increment cores (4- or 5-mm diameter) were collected from each tree after cessation of annual growth. Cores were collected at a height of 1.3 m and evenly distributed around the tree circumference. Because this study's focus was on recent growth and resin ducts, core depth in sapwood was 5 cm to obtain rings from the last decade.

After drying and mounting, cores were prepared for measurement of annual ring widths by progressively sanding to a final grit of 600, and then scanned using a flatbed scanner (Epson America, Inc.) at 2,400 dots per square inch. The width of each annual ring was measured to 0.1 mm on scanned images with ImageJ software, aided by inspection under higher magnification (up to $\times 40$) under a scope (Omano Inc). The number of traumatic resin ducts in each annual ring was counted under $\times 40$ magnification but

were not measured due to their small size. Most ducts were formed in 2021 and/or 2022 during the pheromone-mediated bark beetle attacks. They occurred in distinct lines of 8 to 26 ducts in areas darkened by resin. Ducts in years before 2021 were rare constitutive ducts that were excluded from statistical analyses.

Dating of years of ring formation was informed by knowledge of the date of core collection (late fall 2022) and the occurrence of traumatic resin ducts starting in 2021 when pheromone lures induced SB attacks. Assumptions were that one ring was formed in each of the 10 focal years of the study (2013 to 2022) and that no rings were absent. Absent rings in Engelmann spruce can occur but are uncommon in Arizona (O'Connor and others 2015), and Engelmann spruce does not show a strong response of tree ring growth to regional drought compared to lower elevation species on drier sites (Adams and Kolb 2005).

The frequency of traumatic resin ducts formed during the pheromone-mediated SB attacks relative to annual tree ring growth was calculated. First, the area of the combined 2021 and 2022 annual rings was calculated as: (ring growth year 2021 + ring growth 2022) × core width. Duct frequency (#/mm²) was then calculated as the number of ducts over both years divided by the combined area of 2021 and 2022 rings. Values from multiple cores per tree were averaged to produce one value for each tree for statistical analysis. Tests for differences among SA defoliation classes used one-way analysis of variance (ANOVA) and Tukey honest mean comparisons, supplemented by nonparametric Wilcoxon tests.

Annual basal area increment (BAI) in mm² was calculated for each tree. To reconstruct tree radius for each year, calculations used d.b.h. in November 2022 and annual radial increments from cores (assuming no change in bark thickness). These annual radii were used to calculate BAI using the usual geometric approach: $BAI_{year\ x} = 3.14159 \times ((radius_{year\ x}^2) - (radius_{year\ x-1}^2))$. Values from multiple cores per tree were averaged to produce one value for each tree for statistical analysis. BAI was averaged over the 5 years ending with the year of pheromone baiting (2017 to 2021) and differences compared among SA defoliation levels with ANOVA. JMP Pro® 16 was used for all analyses.

Terpenes in phloem were extracted and analyzed using gas chromatography and mass spectrometry. Approximately 30 mg of frozen phloem from each tree was cut into small pieces (ca. 2 mm³) and placed in 1 mL of 95-percent n-hexane in 2-mL glass gas chromatograph (GC) vials and agitated on a shaker plate for 24 hours (20 °C, 250 rpm). After shaking, the solvent was transferred to new GC vials using a pipette. The phloem vial was rinsed twice with 0.25 mL of hexane for a final volume of 1.5 mL and 100 µg of 2-nonanone was added as the internal standard. Phloem was dried in an oven at 30 °C for 1 week and then weighed. Quantifications were made relative to the internal standard by using ChemStation software. Compounds were identified by comparison of retention times and mass spectra with those of commercially available standards or classified using mass spectra as monoterpenes

(MT) or sesquiterpenes (ST). Terpene concentrations were analyzed on a dry phloem weight basis (ug/g phloem dry weight).

RESULTS

Tree mortality over 18 months after SB baiting (May 2021 to November 2022) was greater for trees with high SA defoliation (40 percent) than trees with medium (10 percent) or low defoliation (0 percent) (table 10.1). BAI differed among SA defoliation classes ($p = 0.0402$) and was greatest in lightly defoliated trees, intermediate in moderately defoliated trees, and lowest in highly defoliated trees (table 10.1). Resin duct frequency ($p = 0.17$) and resin flow ($p = 0.363$) did not differ among SA defoliation classes (table 10.1). Phloem thickness differed among defoliation classes ($p = 0.034$) and was lower in trees with high defoliation compared to trees with medium or low defoliation (table 10.1). Phloem concentration of total terpenes ($p = 0.4680$) and each of 34 constituent terpenes ($p > 0.05$) did not differ among defoliation classes (table 10.1).

In September 2021, 5 months after baiting, an average of 67 percent of pitch tubes had SB galleries, but this percentage did not differ ($p = 0.1446$) over defoliation classes (table 10.1). The percentage of pitch tubes with larvae (9 percent overall) was lower than for galleries (67 percent) and differed among defoliation classes ($p = 0.0156$; table 10.1). Larvae were absent in trees with low defoliation, scarce in trees with moderate defoliation (6 percent), and most common in trees with high defoliation (20 percent). In November 2022, no SB exit holes or galleries with live beetles

were observed in any of the trees (table 10.1), indicating that attacks were not successful.

DISCUSSION

An important caveat is that this study occurred during a period of declining SB activity in the study site region. In other words, the SB population was endemic during this study. For example, aerial survey data indicated that SB activity decreased between 2020 and 2021 (when trees were baited) on both the Springerville Ranger District of the Apache-Sitgreaves National Forests (decrease of 25 percent) and adjacent White Mountain Apache Tribal lands (decrease of 96 percent). The finding that SA defoliation of Engelmann spruce had little impact on tree defense against SB applies only to endemic SB populations. The degree to which these findings apply to higher SB populations, or outbreak levels, is not known and further investigation is recommended.

Elevated SB activity and mortality in SA-defoliated trees was documented in Engelmann spruce at Greens Peak (Grady 2019) and broadly across the White Mountains during 2015 to 2018 (Gaylord and others 2020). However, attack success of SB on all trees in this study was low despite lures with aggregation pheromone being on trees for the entire summer of 2021. Analysis of galleries beneath pitch tubes showed that SB attacked and started to reproduce similarly in trees over all SA defoliation levels. Most of these galleries did not produce larvae, yet larvae were more common in trees with high defoliation than lower levels of defoliation. This pattern is

Table 10.1—Results of experiment where Engelmann spruces in each of three spruce aphid defoliation classes (low, medium, high) were experimentally baited with spruce beetle pheromones in May 2021

Tree response; mean (SE)	Low defoliation	Medium defoliation	High defoliation	p-value ANOVA F-test ^a
(1) Mortality May 2021 to November 2022 (percent)	0	10	40	NA
(2) Mean 5-year (2017 to 2021) BAI (mm²/year)	3168.5a (839.8)	2603.0ab (433.5)	1131.6b (144.5)	0.0402
(3) Resin duct frequency combined over 2021 and 2022 (#/mm²)	2.081a (0.791)	5.75a (2.038)	5.76a (1.314)	0.17
(4) Resin flow (g)	0.102a (0.0381)	0.328a (0.2090)	0.106a (0.0502)	0.363
(5) Phloem thickness (mm)	2.29ab (0.109)	2.34a (0.132)	1.93b (0.101)	0.034
(6) Phloem total terpene concentration (ug/g)	1793.1a (361.52)	1338.0a (202.88)	1461.1a (202.57)	0.4680
(7) Phloem concentrations of 34 constituent terpenes^b	Not listed for brevity	Not listed for brevity	Not listed for brevity	>0.05
(8) Percentage of beetle attacks with galleries, September 2021	56.0a (10.24)	82.0a (6.29)	64.5a (10.44)	0.1446
(9) Percentage of beetle attacks with larvae, September 2021	0.0b (0.0)	6.0ab (3.06)	20.0a (7.45)	0.0156
(10) Number of adult beetle exit holes	0	0	0	NA

ANOVA = analysis of variance; BAI = basal area increment; SE = standard error.

Number of trees per defoliation class was 10, except for row (1) where n = 9 for low defoliation and row (4) where n = 9 for medium defoliation and n = 6 for high defoliation.

Means with different letters are significantly different ($p < 0.05$) in Tukey honestly significant difference (HSD) comparisons.

^a ANOVA results were similar to nonparametric Wilcoxon tests in all cases. Bold shows statistically significant differences among defoliation classes.

^b Tricyclene, alpha-pinene, camphene, sabinene, beta-pinene, myrcene, alpha-phellandrene, 3-carene, terpinolene, cymene1, beta-phellandrene, limonene, gamma-terpinene, cymene2, terpinolene, linalool, methyl thymyl ether, terpineol acetate, bornyl acetate, alpha-terpineol acetate, alpha-longipinene, caryophyllene, beta-farnesene, cymene3, two unnamed monoterpenes, and eight unnamed sesquiterpenes.

surprising because highly defoliated trees had thinner phloem than less-defoliated trees, which suggest controls over SB larval development by factors not investigated in this study (e.g., moisture content, tissue sugar concentration). Despite this evidence of partial SB larval development in some highly defoliated trees in the fall of 2021, subsequent SB development over the next year was minimal. In November 2022 (the second year after baiting), no SB exit holes or galleries with beetles at any life stage were observed in any of the trees. These results suggest that tree mortality in this experiment was the result of SA defoliation, chronic drought, and other factors but not the pheromone-mediated SB attacks during the study.

The low 5-year BAI of trees with high SA defoliation shows that these trees were more stressed than trees with less defoliation for several years before the experiment, likely due in part to SA defoliation in 2018. The high mortality during the experiment for trees with high defoliation (40 percent) suggests that even a single year of heavy SA defoliation reduces tree survival under the conditions at the study site, which include the most severe multidecadal drought in the last thousand years (Williams and others 2022). Similar tree mortality rates have been reported during previous SA outbreaks in Arizona (Lynch 2004, 2009). The lower mortality for trees with either light (0 percent) or moderate (10 percent)

SA defoliation compared to high defoliation (40 percent) shows that trees can survive light SA defoliation for at least several years, whereas heavy defoliation results in fast mortality for some trees, and this generally occurs within the first 2 to 4 years following severe defoliation (Lynch 2004).

Root disease is another site factor that likely influenced results. Near the end of the experiment (August 29, 2022), the study site was surveyed for root disease during a period of heavy rainfall. Evidence of tomentosus root rot (*Onnia tomentosa*) was widespread on the site. Fruiting bodies were present within a 3-m radius of every tree in the study; several snapped trees in the study area exhibited “honeycomb” rot at the base, which is typical of tomentosus. Furthermore, evidence of armillaria (fungal mats) was present on several trees. Root disease was suspected to have contributed to tree mortality. Continued study of relationships among root disease, SA, and SB is highly recommended.

Trees with high or medium SA defoliation still have some defense against SB despite their slow growth. These trees were still able to form traumatic resin ducts in response to SB attack. Resin flow and terpene concentrations of moderately or highly defoliated trees were similar to lightly defoliated trees, suggesting similar quantitative and qualitative defenses among SA defoliation levels.

In summary, the results of this study suggest that, under endemic conditions, SB does not gain a reproductive advantage from Engelmann spruce trees defoliated by SA, and SB outbreaks likely do not initiate from SA defoliation. In other words, the results suggest that trees heavily defoliated by SA act as a sink for SB. Further monitoring and investigation of the interaction between SA defoliation and SB performance under outbreak, epidemic conditions are needed.

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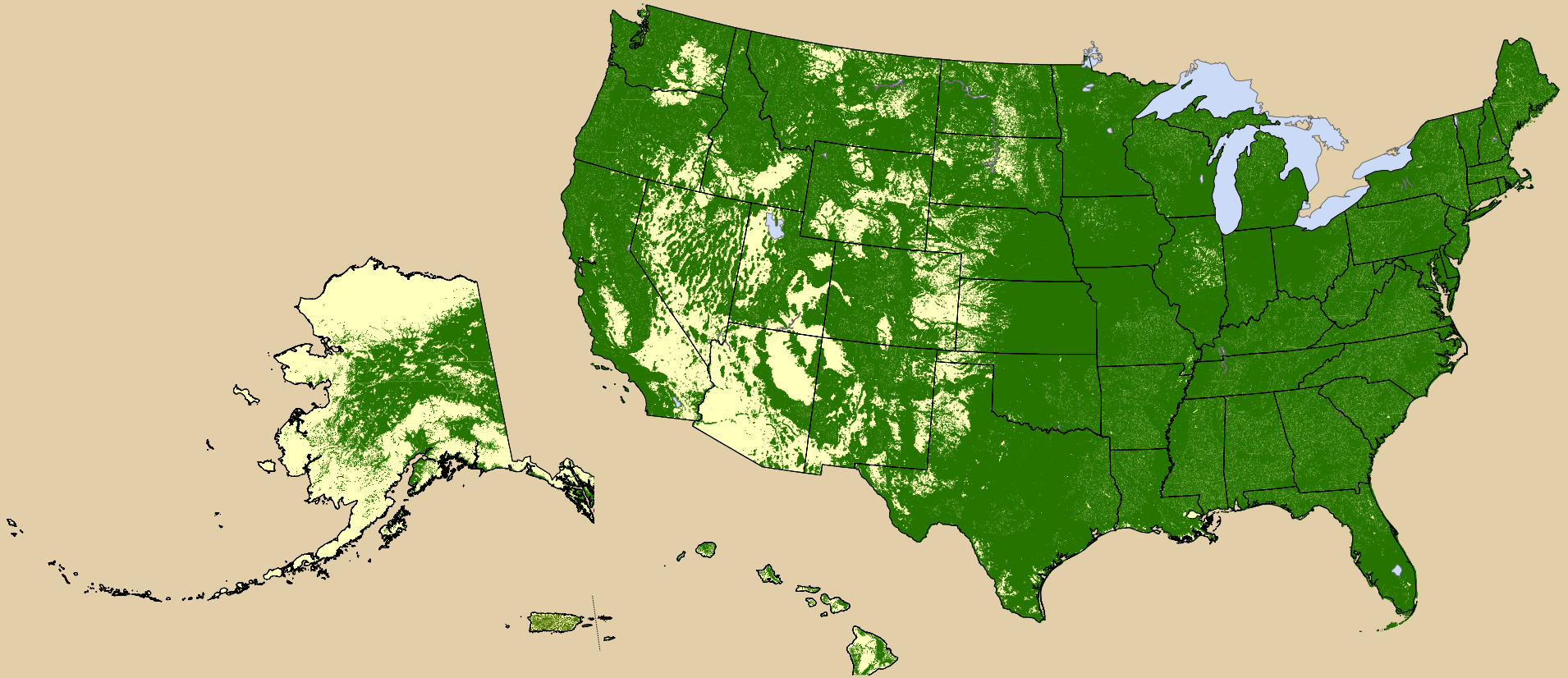
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The Forest Health Monitoring (FHM) Program of the U.S. Department of Agriculture, Forest Service annually funds projects under the Evaluation Monitoring (EM) component of FHM. Projects generally are intended to determine the extent, severity, and causes of undesirable changes in forest health. This component of FHM provides opportunities to delve deeper into observed forest health changes than is often possible as part of standard monitoring activities. When an EM project is completed, researchers are asked to provide a concise summary of the project. These summaries were previously (2008–2022) published as part of the FHM annual national status and trends report. Beginning in 2023, the summaries are being published in a separate document to highlight the EM projects and communicate concise results but will continue to be searchable on the FHM website reporting page (<https://www.fs.usda.gov/science-technology/forest-health-protection/publications-reports/forest-health-monitoring-publications>). This report includes summaries of nine recently completed EM projects that: evaluated the extent and severity of black cherry (*Prunus serotina*) regeneration declines in eastern forests and explored potential causes; investigated whether oak (*Quercus* spp.) regeneration and recruitment in eastern upland forests are being affected by forest health disturbances; studied an outbreak of *Diplodia sapinea* and *D. scrobiculata* that caused extensive dieback and mortality of slash pines (*Pinus elliottii*) in Florida; evaluated Carolina hemlock (*Tsuga caroliniana*) distribution and health in the Southern Appalachian Mountains; applied Forest Inventory and Analysis data and site-specific factors to understand flowering dogwood (*Cornus florida*) decline in the Southeastern United States; evaluated distribution and impacts of invasive species on Southern U.S. forests; evaluated the health and abundance of five-needle pines (*Pinus* spp.) in southern Colorado and northern New Mexico; linked physiology to landscape-scale dieback of Utah juniper (*Juniperus osteosperma*) during an ongoing megadrought; and investigated whether spruce aphid (*Elatobium abietinum*) defoliation alters Engelmann spruce (*Picea engelmannii*) resistance to spruce beetle (*Dendroctonus rufipennis*).

Keywords—Black cherry, Carolina hemlock, diplodia tip blight, Engelmann spruce, five-needle pines, flowering dogwood, forest insects and disease, invasive species, oak, regeneration, spruce aphid, spruce beetle, Utah juniper.



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