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Cultivar Identification and Genetic Relatedness Among 25 Black Walnut (Juglans Nigra) Clones Based on Microsatellite Markers

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Abstract

Proceedings from the 2016 Central Hardwood Forest Conference in Columbia, MO. The published proceedings include 31 papers pertaining to research conducted on artificial and natural regeneration, biomass and carbon, forest dynamics, forest health, modeling and utilization, prescribed fire, soils and nutrients, and wind disturbance.

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Flowering dogwood. Photo by David R. Larsen, University of Missouri, used with permission.

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FOREWORD

The Central Hardwood Forest Conference is a series of biennial meetings dedicated to the sustainability and improvement of Central Hardwood forest ecosystems. The objective of the conference is to bring together forest managers and scientists to discuss research and issues concerning the ecology and management of forests in the Central Hardwood region. The 20th Central Hardwood Forest Conference was hosted by the University of Missouri, School of Natural Resources and the U.S. Forest Service, Northern Research Station. The conference included presentations pertaining to artificial and natural regeneration, biomass and carbon, forest dynamics, forest health, modeling and utilization, prescribed fire, soils and nutrients, and disturbance. There also was a concurrent session featuring the findings from the Missouri Ozark Forest Ecosystem Project, the Indiana Hardwood Ecosystem Experiment, and other long-term research studies conducted in the Central Hardwood Forest region.

The 21st Central Hardwood Forest Conference (2018) will be hosted by Purdue University and the U.S. Forest Service, Northern Research Station.

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REVIEW PROCEDURES

A single-blind review process was used in reviewing manuscripts submitted for publication. Each manuscript was peer-reviewed by at least two professionals. Reviews were returned to authors to revise their manuscripts. Upon acceptance by the assigned editor, revised manuscripts were forwarded to the U.S. Forest Service, Northern Research Station for final editing and publishing. The conference Editorial Committee returned some of the manuscripts to the authors as being more appropriate for other outlets.

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The Program Committee wishes to thank Lisa Allen, State Forester, Missouri Department of Conservation, for providing a warm welcome to begin the conference. We thank James Guldin, U.S. Forest Service, Southern Research Station, and David Bengston, U.S. Forest Service, Northern Research Station, for their insightful plenary session messages. We would like to express our gratitude to Erica Lovercamp, Debbi Haskamp, and to other staff of the University of Missouri Conference Office for assistance with planning and hosting the conference. We also thank the following individuals who volunteered as session moderators: Stacy Clark, Lauren Cox, Kamal Islam, Brice Hanberry, Tara Keyser, Carter Kinkead, David Larsen, Elizabeth Olson, Terry Robison, Jamie Schuler, Callie Schweitzer, Steve Shifley, Kyle Steele, Alicia Struckhoff, Melissa Thomas-Van Gundy, and Lance Vickers. The Steering Committee is grateful to Susan Wright and other members of the Northern Research Station Communications and Science Delivery group for technical editing and publishing of these proceedings.

HISTORY OF THE CENTRAL HARDWOOD FOREST CONFERENCE

This Conference is the 20th in a series of biennial meetings that have been hosted by numerous universities and U.S. Forest Service Research Stations in the Central Hardwood Forest region including:

1976 Southern Illinois University

1978 Purdue University

1980 University of Missouri

1982 University of Kentucky

1985 University of Illinois

1987 University of Tennessee

1989 Southern Illinois University and the North Central Forest Experiment Station

1991 Pennsylvania State University and the Northeastern Forest Experiment Station

1993 Purdue University and the North Central Forest Experiment Station

1995 Northeastern Forest Experiment Station and West Virginia University

1997 University of Missouri and the North Central Forest Experiment Station

1999 University of Kentucky and the Southern Research Station

2002 University of Illinois, Urbana-Champaign and the North Central Research Station

2004 Ohio State University, Wooster, and the Northeastern Research Station

2006 University of Tennessee, Knoxville, and the Southern Research Station

2008 Purdue University and the Northern Research Station

2010 University of Kentucky and the Northern Research Station

2012 West Virginia University and the Northern Research Station

2014 Southern Illinois University and the Northern Research Station

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ANTICIPATING CASCADING CHANGE IN FORESTS: SEEKING A DEEPER UNDERSTANDING OF THE FUTURE CONSEQUENCES OF FOREST MANAGEMENT ACTION OR INACTION

David N. Bengtson, Michael J. Dockry, and Stephen R. Shifley[1]

Abstract.—This study used a participatory group brainstorming process called the Futures Wheel to identify and evaluate the direct and higher-order implications of this trend: Central Hardwood forests lack age-class diversity and will uniformly grow old. Five 1st-order consequences of this trend were identified: continued significant decrease in early-successional forest, continued significant increase in late-successional forest, decreased resilience to many types of forest disturbances, decrease in carbon sequestration rates, and increase in the popular perception that this is the way all forests are and should be. Twenty-five forestry professionals participated in a Futures Wheel exercise to identify 2nd- and 3rd-order implications of the five 1st orders. Participants identified 25 2nd-order and 121 3rd-order implications and scored them for likelihood and desirability. Analysis of the 2nd and 3rd orders revealed many types of implications that are relevant for policy and management interventions, including high-likelihood but strongly negative implications, and low-likelihood but strongly positive implications.

INTRODUCTION

As an artifact of past disturbance (MacCleery 2011), nearly 60 percent of Central Hardwood forestland is clustered in age classes that span 40–80 years. This is one of the five anthropogenic trends identified by Shifley et al. (2014) as having profound consequences for forest conditions and management needs in the future. Young forests (age 20 years or newer) comprise 7 percent of all forests in the region; forests older than 100 years comprise 5 percent (Fig. 1). Within the region, this unimodal (bell-shaped) pattern of clustered age classes is repeated at smaller spatial scales for individual states and for individual forest-type groups (Miles 2015, Shifley et al. 2012). The unimodal age-class distributions common throughout the Central Hardwood region differ from those observed for other regions of the United States (Pan et al. 2011). Because of built-in inertia and low rates of natural or anthropogenic forest-regenerating disturbance, the uniform aging of these forests will continue for decades.

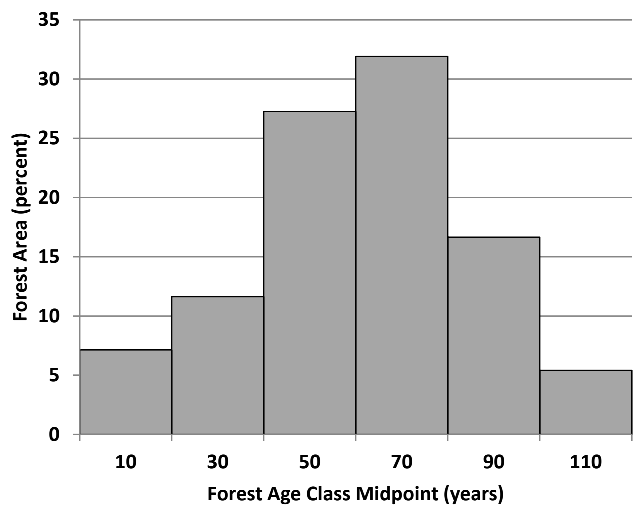

Figure 1.—Central Hardwood forest area summarized by 20-year age classes. Based on the 126 million acres of forest land in the Eastern Broadleaf Forest Province (221), Midwest Broadleaf Forest Province (222), Central Interior Broadleaf Forest Province (223), Prairie Parkland (Temperate) Province (251), and Central Appalachian Broadleaf Forest-Coniferous Forest-Meadow Province (M221) (based on Miles [2015] and McNab et al. [2007]).

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Major but slow-moving trends such as uniform aging of forests are often ignored by planners and policy makers. As business futurist Peter Schwartz (2003) observed, "The dominant intellectual strategy that people bring to bear on the future is denial." It is much easier to focus on the countless immediate problems of today and ignore the far-reaching, slowly unfolding challenges of tomorrow. But major changes—even slow-moving ones—often produce a cascade of unanticipated consequences. These consequences may be direct or indirect, obvious or hidden, immediate or long-range, positive or negative, and insignificant or game changing.

Forestry professionals need to recognize, explore, and prepare for the possible consequences of cascading change. Forest policies and management actions can be designed to avoid negative consequences and to promote positive changes. The direct 1st-order impacts of cascading change may be relatively easy to identify: 1st-order consequences are overwhelmingly what groups identify in unstructured brainstorming about the impacts of change (Schreier 2011). The higher-order consequences, however, are rarely detected without deeper analyses. The higher-order (2nd- and 3rd-order) implications contain the most surprises, both positive and negative, and may have the most relevance for forward-looking policy and management. The Futures Wheel was designed to facilitate uncovering and analyzing the higher-order implications of change.

THE FUTURES WHEEL

The Futures Wheel, also called the Implications Wheel® (Barker 2011), is a participatory research method that uses a structured brainstorming process to discover multiple levels of consequences that result from all types of change: an emerging trend, a new or revised policy, a technological innovation, a major event, new strategic objectives, new laws or regulations, and other types of change. It is a group process that, if conducted properly, incorporates the four conditions required to form a "wise crowd" (Surowiecki 2004): (1) diversity (participants represent diverse perspectives), (2) independence (a participant's opinions are not determined by the opinions of others in the group), (3) decentralization (participants are able to specialize and draw on local knowledge), and (4) aggregation (a mechanism exists for aggregating individual perspectives). The diversity of participants is a key to the effectiveness of the Futures Wheel method. Page (2007) and others have shown that complex problems may be solved more effectively with a diverse team than by the best individual experts.

Originally proposed by futurist Jerome Glenn (1972), the Futures Wheel has been widely used to identify and analyze unanticipated consequences of change in many organizational settings, including corporations, the military, public sector agencies, and nongovernmental organizations. Most applications of the Futures Wheel are not published because they are proprietary or confidential, but examples of published studies include examining the implications of trends that affect tourism (Benckendorff 2008, Benckendorff et al. 2009), European integration (Potůček 2005), research and technology management (Farrington et al. 2013), operational changes in religious organizations (Gebhard and Meyer 2006), and student understanding of the potential consequences of science-related developments (BouJaoude 2000).

The Futures Wheel takes a broad look at the future landscape to quickly gather information on potential opportunities, dangers, and consequences of a particular change. The method provides a means to quickly develop qualitative information that is useful to find the best pathway forward, avoid obstacles, and enhance decision-making information.

Central to the Implications Wheel approach to the Futures Wheel is the "cascade thinking" that emerges through the process and traces how one change leads to multiple possibilities, and each of these possibilities in turn leads to further possibilities (Barker and Kenny 2011). Cascading consequences, and how a seemingly insignificant event can have catastrophic results, are illustrated in the classic proverb "For want of a nail":

> For want of a nail the shoe was lost.
>
> For want of a shoe the horse was lost.
>
> For want of a horse the rider was lost.
>
> For want of a rider the message was lost.
>
> For want of a message the battle was lost.
>
> For want of a battle the kingdom was lost.
>
> And all for the want of a horseshoe nail.

Conversely, insignificant or undesirable changes can also have strongly positive consequences and opportunities. The Futures Wheel uncovers positive and negative cascading changes.

OVERVIEW OF THE FUTURES WHEEL METHOD

The first step in a Futures Wheel exercise is to define and describe the change of interest. In our case, this was the lack of age-class diversity and uniform aging of Central Hardwood forests. Five major direct or 1st-order implications of this trend were identified by the research team before the group process:

1. Continued significant decrease in early-successional forest.
2. Continued significant increase in late-successional forest.
3. Decreased resilience to many types of forest disturbances such as extreme weather events and climate change.
4. Decrease in carbon sequestration rates (older forests sequester carbon at a slower rate).
5. Increase in the popular perception that this is the way all forests are and should be (i.e., older forests are what people are accustomed to and therefore what they prefer).

These outcomes seemed likely given the current state of Central Hardwood forests.

A diverse group of participants—including participants with specialized knowledge about hardwood forests and nonspecialists—was recruited to explore the 1st-order implications in depth. In general, the group sizes range from as few as a single small group of about five for an informal exploration to hundreds of participants divided into small groups. Our study included 25 participants.

Figure 2 illustrates the structure of a Futures Wheel used to guide the group process. The exploration process begins by briefing participants on details of the issue at the center, dividing participants into groups of about five, and assigning a 1st-order implication to each group. The facilitator then asks, "If this occurs, what might happen next?" Participants take turns contributing possible 2nd-order implications, which are added to the diagram, branching out from the 1st orders. The facilitator ensures that contributed implications are clear and specific, and that they follow directly from the preceding implication.

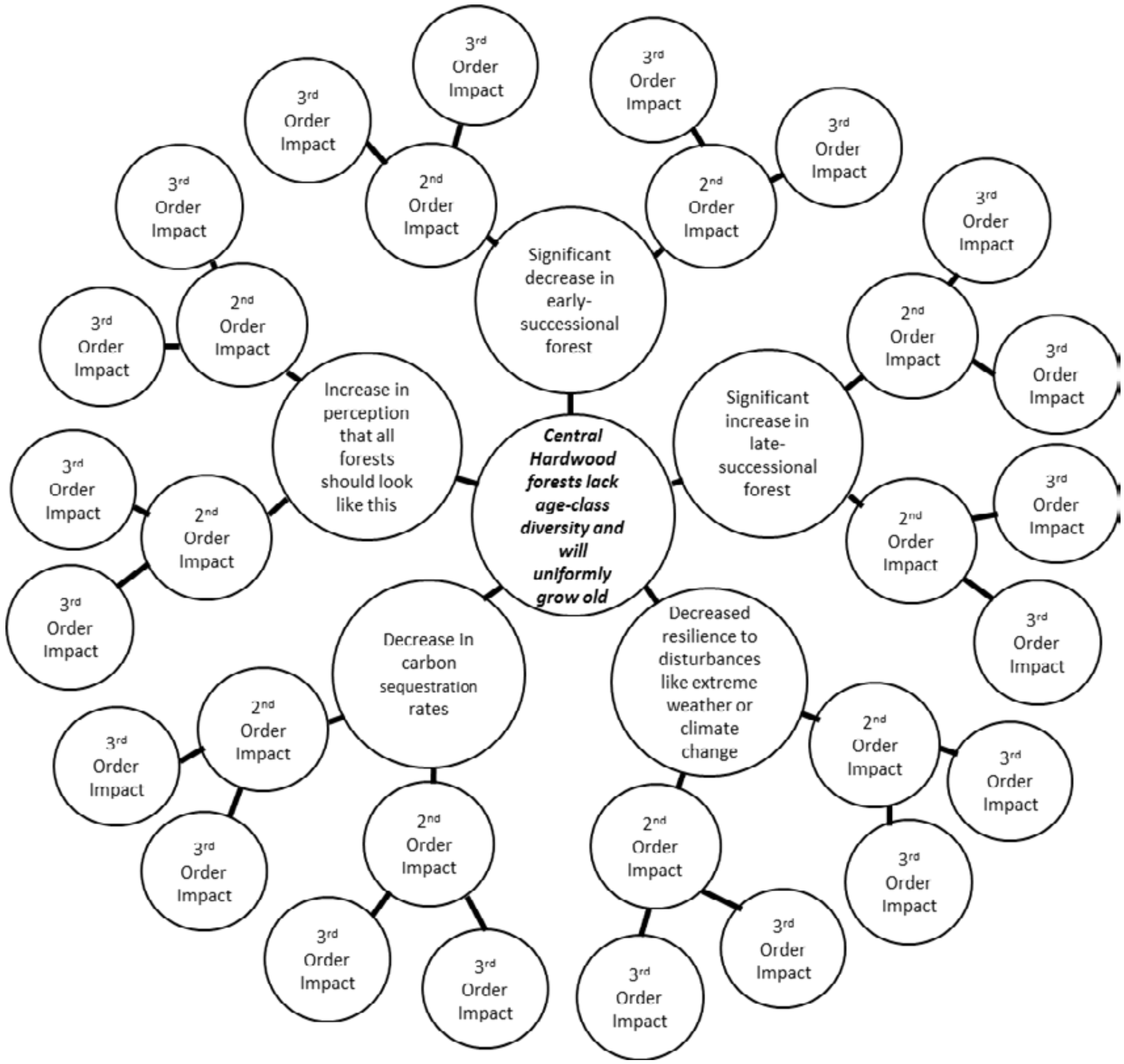

Figure 2.—Futures Wheel diagram with the central trend and 1st-order implications. In our Futures Wheel exercise, participants generated about five 2nd-order impacts for each 1st-order impact, and five 3rd-order impacts for each 2nd-order impact.

Once the brainstorming groups have identified about five 2nd orders for each 1st order, the process is repeated to identify a set of possible 3rd-order implications for each 2nd order.

After the implications are identified, the Implications Wheel version of the Futures Wheel includes a group scoring process. Participants rate each of the 1st-, 2nd-, and 3rd-order implications for desirability and likelihood, arriving at a consensus score in their small groups (Schreier 2005). Scoring provides significant additional information and highlights likely negative and unlikely positive implications that may be most important to decisionmakers. Benckendorff (2008), Bengston (2015), and Glenn (2009) provide additional details on the Futures Wheel method.

SELECTED FINDINGS

This section highlights selected findings from a November 2014 Futures Wheel exercise we conducted to explore the implications of the trend that Central Hardwood forests lack age-class diversity and will uniformly grow old. The 25 participants in our study included diverse forestry and natural resource professionals, administrators, support personnel, and communications specialists.

From the five 1st-order implications that were developed in advance by the research team, our participants generated 25 2nd-order implications and 121 3rd-order implications. The large number of higher-order implications is due to the structure of the brainstorming process, which shifts the focus from the short-term and direct implications to the longer-term and higher-order implications of change.

Similar Implications

Similar implications are ones that emerge repeatedly in different contexts (i.e., from different 1st and 2nd orders) during the Futures Wheel process. The appearance of the same or similar implications suggests that they are robust and more likely to occur. The most common similar implications from our Futures Wheel exercise were the following:

- Increasing conflict over forest management, including legal action
- Decreased biodiversity
- Decreased resilience to disturbance
- Fewer forest management options
- Fewer hunting opportunities
- Worsening climate change

The fact that these most commonly generated similar implications are all undesirable—or strongly negative—is significant. Advanced knowledge that these negative consequences are likely to emerge from the central trend of uniformly aging forests could be quite useful for planners, managers, and policy makers in setting priorities and designing management strategies.

Unique Implications

Unique implications are ones that were mentioned only once throughout the process of generating implications in small groups. They tend to be suggested by participants who are "outside-the-box" thinkers or who have diverse backgrounds and perspectives. Unique implications may represent uncommon but significant issues that most forest planners and managers are not considering. Examples of unique 3rd-order implications from our Futures Wheel exercise include the following (the preceding 1st- and 2nd-order implications are given to provide the context for the unique 3rd orders):

- (1st order) Decreased resilience to many types of future forest disturbances → (2nd order) Increased large-scale land cover conversion caused by low resistance to large-scale natural disturbances → (3rd order) Increased opportunities to create climate-resilient forest.
- (1st order) Continued significant increase in late-successional forest → (2nd order) Decrease in the diversity of understory species (wildlife and vegetation) → (3rd order) Decreased opportunities to gather edible plants.

- (1st order) Continued significant increase in late-successional forest → (2nd order) More big trees → (3rd order) Enhanced tribal connections to the spiritual values of big trees.
- (1st order) Growing popular perception that this is the way all forests are and should be (i.e., older forests are what people are accustomed to and prefer) → (2nd order) Delayed development and application of best forest management practices → (3rd order) Increasing difficulties in changing public perceptions of forests.
- (1st order) Growing popular perception that this is the way all forests are and should be (i.e., older forests are what people are accustomed to and prefer) → (2nd order) Laws change to protect old forests → (3rd order) Limits on motorized recreation.

Many of the unique implications that our participants suggested were of low relevance to forest decisionmakers. But some useful insights emerged—elements of the future that are not immediately evident.

Highly Significant Implications

Two types of highly significant implications have special relevance for forest decisionmakers: likely strong negatives and unlikely strong positives. Likely strong negatives are implications that the participants scored as both highly likely and strongly negative; unlikely strong positive implications were scored as both highly unlikely and strongly positive.

Likely strong negative implications require policies or management actions that are designed to decrease their likelihood or reduce their negative effects. Examples from our Futures Wheel exercise include the following:

- (1st order) Continued significant decrease in early-successional forest → (2nd order) Decrease in abundance of early-successional wildlife → (3rd order) Further declines in bird species already of conservation concern.
- (1st order) Decreased resilience to many types of future forest disturbance → (2nd order) More uncertainty for industry re: timber supply (declining forest-based industry) → (3rd order). Declining quality of local school districts where mills are located because of reduced tax revenues
- (1st order) Increase in popular perception that this is the way all forests are and should be → (2nd order) Laws change to protect old forests → (3rd order) Wildlife and forest diversity declines.

Unlikely strong positive implications represent potential opportunities for forest decisionmakers to provide desirable outcomes through policies or management actions that increase the chances of these events occurring. Our participants generated far more likely strong negatives than unlikely strong positives.

CONCLUDING COMMENT

Forest decisionmakers need to anticipate unforeseen consequences of major trends and other types of change and address them proactively. Most analyses of planning and policy issues do not go beyond identifying the most evident direct consequences of change. But the higher-order consequences may be the most significant. The smart group process and nonlinear thinking that occur when conducting a Futures Wheel make it a powerful tool for identifying and evaluating possible implications of all types of changes that affect forests. This process gives deeper insights into the future consequences of forest change and highlights possibilities for avoiding potential problems and embracing opportunities.

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REGENERATION I

CONTROL OF HAY-SCENTED AND NEW YORK FERNS WITH OUST XP® HERBICIDE: REVISITING RATE AND TIMING REQUIRED IN MIXED OAK AND NORTHERN HARDWOOD STANDS

Todd E. Ristau[1]

Abstract.—Dense rhizomatous fern layers compete with desirable tree seedlings for light, which suppresses development and even kills seedlings. Sulfometuron methyl (Oust XP®) herbicide can be safely and effectively used to control ferns. Previous research showed that depending on application timing, as little as 2 ounces of Oust XP per acre controlled ferns while hardwood tree seedlings growing beneath them survived. It was recently suggested that some sites may require higher rates because of soil differences. This study uses a wider variety of site conditions than previous studies to reevaluate the rates and timing of Oust application. This study examined the efficacy of 2, 3, and 4 ounces of Oust XP herbicide per acre applied in July, August, and September without a surfactant for controlling *Dennstaedtia punctilobula* and *Thelypteris nove-boracensis* (hay-scented and New York ferns) in mixed oak and northern hardwood forests. Ten-foot by ten-foot plots were treated and monitored for fern control and impacts on nontarget species within 10 stands across north-central and northwestern Pennsylvania. Ferns were controlled with all rates and times of application. More surviving ferns in plots were treated in September. Hardwood seedlings survived even at the highest rate. *Prunus serotina* and *Amelanchier arborea* had some nonsignificant reductions in numbers at all rates and times. Some growth malformations were noticed on individual seedlings with July treatments. Oust XP killed *Rubus* spp. (blackberry) at all rates and times, but grasses and other herbs were unaffected. Oust XP continues to be an effective tool for controlling ferns at a variety of sites and allows seedlings to survive in place.

INTRODUCTION

Overbrowsing by white-tailed deer *(Odocoileus virginianus)* has altered forest understories across the eastern United States (Alverson and Waller 1997, Horsley et al. 2003, Porter et al. 2004). Forest understory plants that interfere with the reestablishment of overstory species following disturbance often become established as a result of long-term herbivory. This can apply selective pressure to an ecosystem, driving species composition toward those that are herbivore resistant (Royo and Carson 2006). Many stands in Pennsylvania have dense herbaceous layers that are dominated by the rhizomatous hay-scented (*Dennstaedtia punctilobula* (Michx) T. Moore) and New York (*Thelypteris noveboracensis* (L.) Nieuwl.) ferns, striped maple (*Acer pensylvanicum* L.), and American beech (*Fagus grandifolia* Ehrh.) (Horsley 1991). These understories are the result of long-term deer browsing followed by increased light, which results from beech bark disease mortality, partial cutting practices, and the shade tolerance of these plants (Horsley et al. 2003, Marquis and Brenneman 1981, Marquis et al. 1992, Tilghman 1989). Since 1991, an herbicide prescription for removing interfering ferns has been used in Pennsylvania (Horsley et al.1992, McCormick et al. 1991). McCormick et al. (1991) showed that hardwood and white pine (*Pinus strobus* L.) seedlings were not affected by Oust XP® application when a surfactant was not included with the product application. Herbicides commonly used to control interfering

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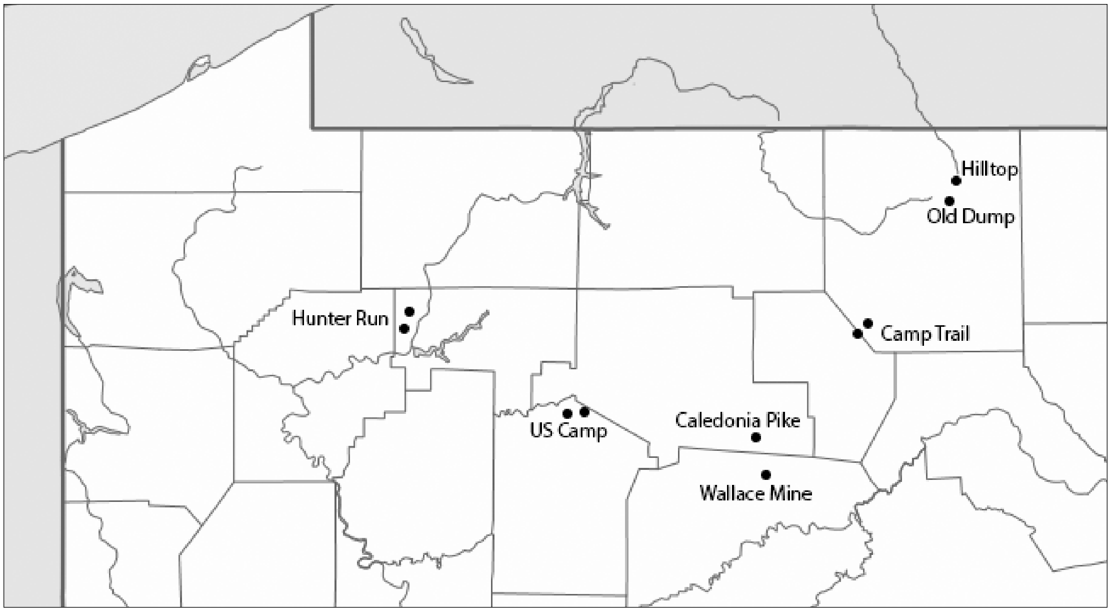

Figure 1.—Study locations throughout north-central Pennsylvania.

plants in Pennsylvania have only short-term impacts on nontarget organisms (Ristau et al. 2011, Stoleson et al. 2011, Trager et al. 2013). Full recovery occurred within 3-5 years. Herbicides are safe and effective for reducing competition and do not harm nontarget organisms.

Public agencies, private companies, and private nonindustrial forest owners currently use herbicides on about 12,000 acres of forest land in Pennsylvania each year. Treating with Oust XP alone does not kill desirable regeneration. Many foresters have expressed concern that fern control is incomplete and unsatisfactory at the recommended 2 ounce per acre rate. They report that higher rates are needed. The Oust XP prescription was developed in Allegheny hardwood stands in northwestern Pennsylvania and in oak stands near State College, PA (Horsley 1988). Differences in soils, genetics, and climate may in turn affect the rate required for adequate fern control. This study expanded the range of sites to include additional oak and northern hardwood stands throughout north-central Pennsylvania to determine if 2 ounces of Oust XP per acre are still effective when applied during July, August, and September.

METHODS

Suitable stands for this study were sought from the Pennsylvania Bureau of Forestry in spring 2012. All stands had 80- to 100-percent summer fern cover and desirable tree seedlings beneath the fern. Overstory density was variable; shelterwood, thinned stands, or stands with mortality provided suitable light conditions for seedlings and interference to develop. Site limitations such as excessively wet or rocky soils were not permitted. Ten sites (see Fig. 1) located in the Clear Creek, Cornplanter, Elk, Moshannon, and Susquehannock districts of the Pennsylvania Bureau of Forestry were selected.

Twelve 100-square-foot (10 foot × 10 foot) treatment plots with a 1/1000th acre (milacre) plot in the center were established in each stand. Application rates used included 0, 2, 3, or 4 ounces of Oust XP per acre (dissolved in 25 gallons of water) applied on or about July 15, August 15, or September 15, 2014 (Fig. 2). Amounts applied to 100-square-foot treatment plots were the proportional equivalents to per-acre amounts (0, 0.13, 0.19, or 0.26 g in 500 mL of water). Surfactant was not included in the applications. Before treatment and full fern expansion (May 15-30, 2014), plant inventories were conducted on milacre plots. These inventories counted

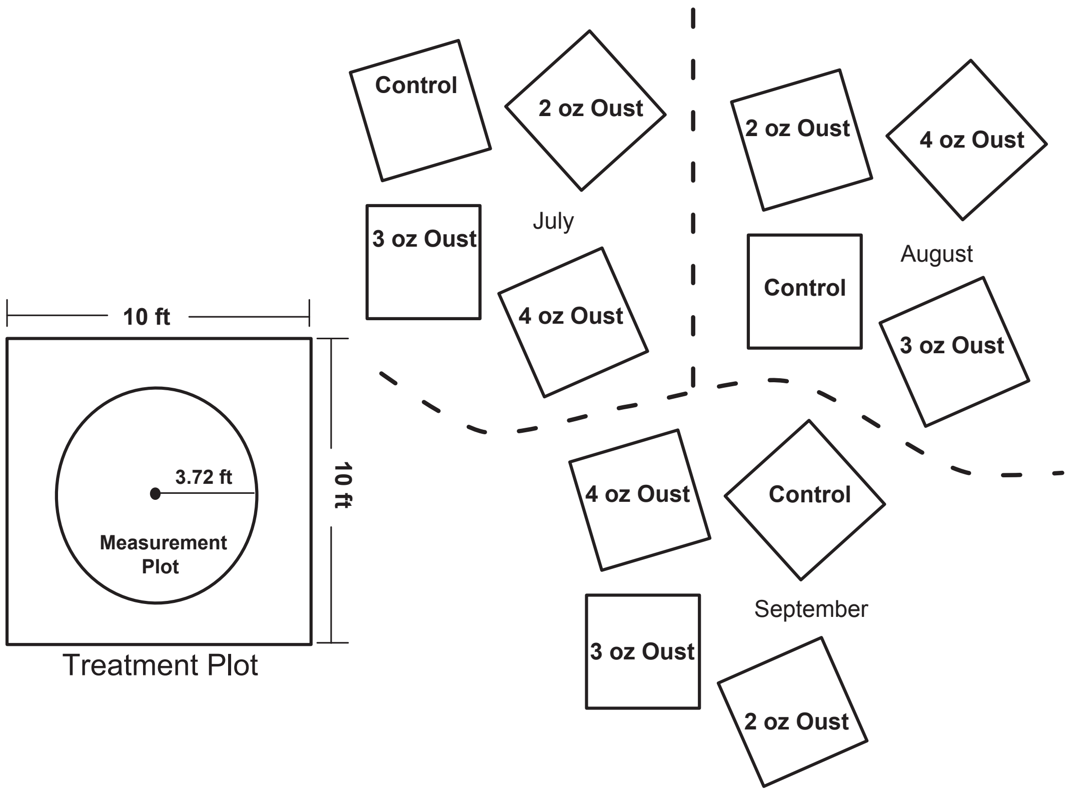

Figure 2.—Stylized plot layout of Oust treatments.

all tree seedlings by species and estimated the percentage of plot covered by broad classes of herbaceous plants, including the targeted fern species. Most fern on the plots was hay-scented fern. Any New York fern encountered was combined with the hay-scented fern cover for analysis because they behave and respond similarly. Inventories were repeated in May 2015 to assess fern control and impacts on nontargeted species. May sampling means that ferns were not fully expressed at the time of the inventory; however, all sample plots had 80- to 100-percent fern cover and were established the year before treatment (2013). Sampling plots with dense fern cover when ferns are fully expressed damages fronds. To avoid that damage, sampling was conducted in May.

The study design was a randomized complete block. Statistical analyses were conducted with SAS PROC MIXED (SAS Institute, Cary, NC 2014). A restricted maximum likelihood technique and the Kenward-Roger correction method for the denominator degrees of freedom were used (Littell et al. 2006). The 10 sites provided replication, and each site contained all rate-by-time combinations. Site was considered as a random effect; year, month of application, and rate applied were treated as fixed effects. Year was considered a repeated measure with a compound symmetry covariance structure (Littell et al. 2006). Treatment effects were evaluated for cover of fern, grass, lycopodium, other herbaceous species combined, and *Rubus* as well as seedling counts of red maple (*A. rubrum* L.), sugar maple (*A. saccharum* Marsh.), serviceberry (*Amelanchier arborea* (Michx.)Fern.), sweet birch (*Betula lenta* L.), American beech (*Fagus grandifolia* Ehrh.), yellow-poplar (*Liriodendron tulipifera* L.), cucumber tree (*Magnolia acuminata* L.), pin cherry (*Prunus pensylvanica* L.), black cherry (*P. serotina* Ehrh.), and northern red oak (*Quercus rubra* L.). Residuals were evaluated graphically using a scatterplot of the residuals, a histogram with normal density, a Q-Q plot, and a box plot of the residuals produced by PROC MIXED. Post-hoc tests to identify differences among experimental treatments were conducted using Tukey-Kramer tests in the LSMEANS option of the MIXED procedure with an alpha probability level of 0.05.

RESULTS AND DISCUSSION

Early summer pretreatment fern cover was 49–60 percent and did not vary between plots of different rates and times of application (Table 1). Following treatment in July or August the percent fern cover was reduced to less than 1 percent for all rates used. All September treatments slightly exceeded 1-percent fern cover for all rates. These results are similar to those reported by Horsley (1988) and Horsley et al. (1992), who indicated no differences in fern control rate across the months tested. Ferns treated in September were often yellowing and the control was expected to be less. Even though the treatments did not differ significantly, there was twice as much fern cover following September treatments as in July and August (Table 1). Other than ferns, the only nontree species that was controlled by Oust XP was *Rubus* spp., which in all treatments except one (2 ounces in August) resulted in substantially lower cover following application. The variability of this species in the 2-ounce treatment was high and there was lower *Rubus* coverage in the plots receiving that rate and timing.

Serviceberry and black cherry were the only hardwood tree seedlings that were affected by Oust XP applications, though these differences were not statistically significant because the sites varied greatly (Table 2). At all rates and times of application, numbers of black cherry seedlings were reduced by 30-50 percent, and serviceberry was reduced by 70–90 percent. Like *Rubus,* which was affected by Oust XP, these species are all in the Rosaceae family, suggesting that members of Rosaceae may be more sensitive to Oust XP herbicide than others in the study. Overall, these results suggest that treatment with Oust XP at any of the rates and times tested does not negatively affect tree seedling numbers. Horsley et al. (1992) cautioned that tree seedling species were affected when a surfactant was added to the herbicide mix.

It is unclear why regional forest managers have observed lack of fern control with the recommended 2 ounce per acre rate (Brose et al. 2008, Marquis et al. 1992). Results of this study show effective control across all study sites, rates, and times. Operational treatments can differ from that of experimental plots, and perhaps the adequacy of coverage is part of the problem. On experimental plots coverage was complete. Mist blower applications may not cover adequately. Greater numbers of fronds were left uncontrolled during the September treatments, suggesting that August treatment with Oust XP may be more effective. July treatment can also work, but these earlier treatments, especially at the higher rates, caused increased malformations on seedlings (data not shown). For an operational recommendation, using 2 ounces Oust XP per acre applied in July or August seems most reasonable, because it reduces the herbicide applied and minimizes damage to desirable hardwood regeneration (Brose et al. 2008, Marquis et al. 1992).

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Table 1.—Percent coverage of herbaceous plants by time and rate of Oust application. Values in parentheses are standard error of the mean. Numbers are from 10 replicates throughout north-central Pennsylvania. Significant ($p < 0.05$) pairwise comparisons between pretreatment and post-treatment are indicated in bold. There were no significant differences among rates and times.

		July (ounce Oust/acre)				August (ounce Oust/acre)				September (ounce Oust/acre)			
	Year	0	2	3	4	0	2	3	4	0	2	3	4
Rhizomatous ferns[a]	2014	56.4 (8.0)	53.6 (5.7)	48.6 (6.5)	55.5 (8.4)	52.7 (6.1)	52.8 (4.7)	54.7 (5.8)	53.0 (6.5)	60.2 (5.0)	56.3 (6.1)	49.7 (6.5)	55.0 (6.4)
	2015	77.0 (4.3)	**0.1 (0.1)**	**0.3 (0.2)**	**0.0 (0.0)**	77.8 (5.6)	**0.6 (0.2)**	**0.4 (0.2)**	**0.5 (0.3)**	76.3 (3.7)	**1.3 (0.5)**	**1.1 (0.2)**	**1.2 (0.4)**
Lycopodium spp.	2014	0.2 (0.2)	0.1 (0.1)	1.5 (1.5)	0.0 (0.0)	2.0 (2.0)	8.5 (8.5)	3.5 (2.5)	0.0 (0.0)	8.4 (8.0)	2.5 (1.7)	1.1 (1.0)	4.5 (4.5)
	2015	2.6 (2.5)	2.0 (2.0)	2.5 (2.5)	0.2 (0.2)	0.1 (0.1)	8.0 (8.0)	9.5 (6.6)	0.1 (0.1)	8.5 (7.0)	7.5 (6.0)	2.5 (2.5)	6.1 (6.0)
Graminoids	2014	0.9 (0.5)	1.8 (1.5)	1.7 (1.0)	0.8 (0.5)	0.2 (0.1)	2.4 (1.0)	1.9 (1.0)	6.2 (4.9)	1.8 (1.0)	1.3 (0.7)	2.1 (1.5)	1.7 (0.6)
	2015	2.0 (1.0)	2.5 (1.5)	1.0 (0.5)	0.7 (0.2)	1.4 (1.0)	4.8 (2.2)	1.9 (1.0)	3.3 (1.6)	1.3 (0.5)	2.0 (1.5)	2.5 (2.0)	1.5 (1.0)
Rubus spp.	2014	2.4 (2.0)	2.4 (1.5)	0.7 (0.2)	1.0 (0.5)	3.8 (1.9)	1.3 (0.7)	5.2 (2.3)	4.0 (2.0)	1.5 (0.6)	7.8 (5.9)	4.3 (2.6)	0.5 (0.2)
	2015	5.0 (3.0)	0.2 (0.1)	0.1 (0.1)	0.1 (0.1)	6.0 (2.4)	0.6 (0.5)	**0.8 (0.3)**	0.9 (0.3)	3.8 (2.0)	**0.9 (0.5)**	0.1 (0.1)	0.2 (0.1)
Other herbaceous	2014	5.7 (1.6)	3.8 (1.9)	4.8 (1.9)	4.1 (1.7)	3.4 (1.2)	2.3 (1.0)	3.9 (1.1)	4.7 (2.0)	4.9 (1.6)	3.0 (1.3)	5.8 (2.3)	5.8 (2.1)
	2015	8.1 (3.8)	3.1 (1.0)	6.6 (2.4)	2.0 (1.5)	4.0 (1.9)	6.4 (2.0)	5.0 (2.0)	5.2 (2.6)	13.2 (4.2)	2.3 (1.1)	8.1 (4.0)	2.4 (0.7)

[a] Combined cover of *Dennstaedtia punctilobula* and *Thelypteris nove-boracensis* (mostly *Dennstaedtia punctilobula*).

Table 2.—Thousands of stems per acre by time and rate of Oust application. Values in parentheses are standard error of the mean. Numbers are from 10 replicates throughout north-central Pennsylvania. There were no significant differences among treatments.

		July (ounce Oust/acre)				August (ounce Oust/acre)				September (ounce Oust/acre)			
Species	Year	0	2	3	4	2	2	3	4	0	2	3	4
		Thousands per acre											
Acer rubrum	2014	14.7	20.0	27.0	18.6	12.7	12.4	11.5	27.9	25.4	21.1	15.0	14.1
		(4.1)	(6.5)	(6.0)	(4.8)	(4.0)	(4.9)	(5.3)	(10.7)	(11.2)	(8.3)	(4.4)	(3.9)
	2015	16.9	27.4	31.5	23.6	17.5	14.0	20.0	24.7	25.5	25.8	21.0	18.0
		(3.5)	(8.6)	(8.4)	(8.5)	(6.6)	(4.3)	(12.8)	(6.5)	(12.6)	(10.4)	(6.1)	(4.1)
Acer saccharum	2014	0.0	0.0	0.0	0.3	0.6	0.1	0.1	0.0	0.3	0.4	0.1	0.0
		(0.0)	(0.0)	(0.0)	(0.3)	(0.6)	(0.1)	(0.1)	(0.0)	(0.2)	(0.3)	(0.1)	(0.0)
	2015	0.0	0.0	0.0	0.3	0.5	0.1	0.1	0.2	0.2	0.0	0.4	0.0
		(0.0)	(0.0)	(0.0)	(0.3)	(0.5)	(0.1)	(0.1)	(0.2)	(0.1)	(0.0)	(0.3)	(0.0)
Amelanchier arborea	2014	4.9	1.4	3.0	0.9	10.1	3.8	2.1	3.0	4.4	6.6	2.6	4.5
		(3.1)	(0.9)	(1.5)	(0.5)	(5.6)	(1.8)	(0.9)	(1.2)	(3.0)	(4.1)	(1.6)	(2.7)
	2015	2.9	0.8	0.4	0.2	4.9	0.6	0.1	0.9	5.1	0.2	0.0	1.1
		(1.5)	(0.7)	(0.2)	(0.2)	(2.2)	(0.5)	(0.1)	(0.4)	(2.9)	(0.2)	(0.0)	(1.1)
Betula lenta	2014	0.6	0.9	0.2	0.7	0.0	0.0	0.1	0.9	0.0	0.1	0.0	0.0
		(0.3)	(0.7)	(0.2)	(0.4)	(0.0)	(0.0)	(0.1)	(0.6)	(0.0)	(0.1)	(0.0)	(0.0)
	2015	0.3	0.5	0.1	0.2	0.0	0.1	0.1	0.1	0.1	0.1	0.0	0.0
		(0.2)	(0.4)	(0.1)	(0.1)	(0.0)	(0.1)	(0.1)	(0.1)	(0.1)	(0.1)	(0.0)	(0.0)
Fagus grandifolia	2014	0.3	0.5	0.1	0.0	0.1	0.0	0.0	0.2	0.0	0.3	0.2	0.1
		(0.3)	(0.5)	(0.1)	(0.0)	(0.1)	(0.0)	(0.0)	(0.2)	(0.0)	(0.2)	(0.2)	(0.1)
	2015	0.4	0.2	0.1	0.2	0.1	0.0	0.0	0.3	0.0	0.4	0.2	0.0
		(0.4)	(0.2)	(0.1)	(0.2)	(0.1)	(0.0)	(0.0)	(0.3)	(0.0)	(0.3)	(0.2)	(0.0)
Liriodendron tulipifera	2014	0.5	1.4	0.7	0.7	0.3	0.7	1.2	1.4	1.8	0.5	0.1	1.7
		(0.4)	(1.4)	(0.5)	(0.5)	(0.3)	(0.6)	(0.9)	(0.9)	(1.8)	(0.5)	(0.1)	(1.7)
	2015	0.9	0.7	1.0	0.8	0.3	0.6	1.4	1.2	1.2	0.9	0.1	1.3
		(0.6)	(0.7)	(0.7)	(0.7)	(0.3)	(0.6)	(1.0)	(0.9)	(1.2)	(0.6)	(0.1)	(1.3)
Magnolia acuminata	2014	0.4	0.7	0.3	0.9	0.3	0.4	0.4	0.0	0.7	0.9	0.5	0.4
		(0.3)	(0.5)	(0.2)	(0.5)	(0.2)	(0.3)	(0.3)	(0.0)	(0.3)	(0.5)	(0.3)	(0.3)
	2015	0.4	0.6	0.3	0.3	0.4	0.7	0.3	0.0	0.7	0.8	0.3	0.2
		(0.2)	(0.5)	(0.2)	(0.2)	(0.2)	(0.6)	(0.2)	(0.0)	(0.3)	(0.4)	(0.2)	(0.2)
Prunus pensylvanica	2014	0.3	0.5	0.2	0.5	0.1	0.0	0.1	0.3	0.0	0.0	0.1	0.0
		(0.3)	(0.3)	(0.2)	(0.4)	(0.1)	(0.0)	(0.1)	(0.3)	(0.0)	(0.0)	(0.1)	(0.0)
	2015	0.1	0.0	0.0	0.1	0.1	0.0	0.0	0.3	0.1	0.0	0.0	0.0
		(0.1)	(0.0)	(0.0)	(0.1)	(0.1)	(0.0)	(0.0)	(0.2)	(0.1)	(0.0)	(0.0)	(0.0)
Prunus serotina	2014	4.0	3.1	7.8	2.7	6.7	4.5	9.6	10.3	4.0	15.4	8.8	2.7
		(1.9)	(1.5)	(5.4)	(1.4)	(4.9)	(2.0)	(6.6)	(8.3)	(1.4)	(9.2)	(5.7)	(1.1)
	2015	3.5	1.4	3.7	1.7	4.7	3.4	3.1	4.7	3.8	8.8	8.9	0.7
		(1.4)	(0.9)	(2.1)	(1.0)	(3.4)	(1.8)	(1.6)	(3.5)	(1.5)	(4.9)	(5.5)	(0.3)
Quercus rubra	2014	5.0	5.0	6.5	8.8	7.2	5.4	3.3	6.9	5.7	3.7	7.5	11.8
		(2.0)	(2.7)	(3.6)	(3.5)	(3.4)	(1.7)	(1.8)	(3.2)	(2.8)	(1.7)	(3.9)	(9.7)
	2015	5.5	6.6	5.2	5.1	3.9	3.3	7.3	3.5	7.3	6.2	5.6	8.0
		(1.7)	(3.3)	(2.7)	(2.5)	(1.4)	(1.4)	(3.7)	(1.8)	(3.1)	(2.6)	(2.3)	(6.2)

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THE IMPACT OF STRIP CLEARCUTTING ON RED OAK SEEDLING DEVELOPMENT

Jamie L. Schuler, Michael Boyce, and Gary W. Miller[1]

Abstract.—A mature upland yellow-poplar/red oak stand was harvested using an alternating strip clearcut method. Red oak seedlings were planted across a light gradient between the cut and residual strips to assess the potential ability of the residual strips to foster the development of competitive oak seedlings over time. After one growing season, no differences in seedling diameter and height growth or survival were detected across the planting positions. Planting shock and drought were assumed to have affected year 1 results.

INTRODUCTION

The challenge of regenerating oak species (*Quercus* spp.) is a widespread and well-known problem throughout eastern hardwood forests. Numerous studies have reported on harvesting techniques to regenerate oak species (e.g., Schuler and Robison 2009). When harvests occur in oak-dominated stands, the subsequent reproduction, although usually present in adequate numbers, often has limited representation of oak species (Dey 2014). Oaks are considered disturbance-dependent species whose successful regeneration is often the combined result of several fortuitous and well-planned events. Oak species regenerate over time, rather than from a single establishment event (e.g., harvest). The process involves the development of competitive sources of reproduction and their timely release into the overstory (Dey 2014, Loftis 2004). The unavailability of competitive seedlings often limits the success of oak regeneration following a harvest.

The general paucity of competitive oak seedlings under mature forests is attributed to low light levels in the understory of the multilayered canopies of undisturbed hardwood forests. With light levels near 5 percent or less of full sun (Gottschalk 1994, Miller et al. 2004, Schweitzer and Dey 2015), oak seedlings rarely persist long enough to achieve competitive status (>3 feet tall) (Brose et al. 2008) and are replaced by more competitive seedlings after a regeneration treatment (Dey et al. 2007). Following harvesting, shade-tolerant advance reproduction outcompetes newer oak germinants; on higher-quality sites, new germinants of fast-growing intolerant species such as yellow-poplar (*Liriodendron tulipifera*) can quickly overtop smaller oak seedlings (Loftis 1990). To make oak seedlings more competitive, treatments that moderate understory light conditions are often recommended well in advance of harvesting (i.e., 10 years), to provide the time necessary to create established and competitive advance reproduction.

A midstory removal treatment that removes or deadens suppressed and intermediate trees to allow increased light to penetrate to the understory layer is a common prescription ahead of a regeneration harvest. This increases light levels to 12-25 percent, a range that improves the survival and growth of oak seedlings compared to seedlings growing in deep shade (Craig et al. 2014, Gottschalk 1985, Miller et al. 2004, Ostrom and Lowenstein 2006). The challenge associated with midstory treatments is that they stimulate the growth of competing species such as red maple (*Acer rubrum*) and sugar maple (*A. saccharum*) (Craig et al. 2014, Schweitzer and Dey 2015); therefore, controlling pre-existing competition is often warranted.

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Despite its demonstrated effectiveness, widespread adoption of midstory removal treatments is hindered by high costs (Bailey et al. 2011, Rathfon 2011), limited markets for small-diameter stems, and the need for immediate landowner income. Therefore, low-cost alternatives that provide meaningful harvest volumes are needed. We hypothesized that a strip clearcut harvest, where 50 percent of the area is cut and the remaining 50 percent is harvested once regeneration is well established (e.g., 5-10 years) is a viable alternative for many landowners. Strip clearcuts are usually prescribed where windthrow is a concern, where seed dispersal distances are limited, and where the impacts of clearcutting need to be lessened (Nyland 2002). Strip clearcutting has been employed in oak stands (Allison et al. 2003, Shostak et al. 2002, Williams 1995). It may provide a means to improve light conditions that are favorable to oak seedlings and provide income from harvesting, without the need for expensive midstory removal treatments.

When designed and implemented appropriately, strip clearcutting creates a gradient of light conditions (Marquis 1965), where the centers of the cut strips have the most light and the centers of the residual strips have conditions that are more similar to the original uncut stand. Orientation of the strips in a north–south direction facilitates moderated light conditions into both sides of the residual strips as the sun progresses from east to west over the course of the day. During the morning hours, light penetrates the eastern edges of the residual strips; by afternoon, light penetrates the western side of each residual strip. This creates elevated light conditions within each strip that affect oak regeneration density and height (Lhotka and Stringer 2013). The increased light conditions are expected to be similar to stands with midstories removed (e.g., 10–20 percent total photosynthetically active radiation). Oak reproduction within the cut strips will depend almost exclusively on advance reproduction, because large harvested oak trees have low sprouting probabilities (Gould et al. 2007, Sands and Abrams 2009). The cut areas are likely to regenerate to a faster-growing, intolerant species such as yellow-poplar. Instead of spending time and money trying to redirect regeneration here, we elected to focus on the residual strips as the major sources of oak for the next stand.

Given the highly variable nature of natural regeneration, both spatially and temporally, we elected to plant northern red oak seedlings (*Q. rubra*) across the light gradient to control for some of this variability. The objective of this study was to assess oak seedling survival and growth along a light gradient from residual strips and extending through the cut strips as a proxy to determine the effectiveness of strip clearcutting to promote oak seedling recruitment.

METHODS

The study was conducted in the central Appalachians on the West Virginia University Research Forest in Preston County, WV. The site lies on a north-facing aspect with 20 percent average slopes. Soils were mapped as Dekalb channery sandy loam. The site index averages 70-75 feet (base age 50) for upland oak using site index curves from Schnur (1937). Overstory species composition was largely yellow-poplar, northern red oak, black cherry (*Prunus serotina*), and red maple, which represented 66 percent, 21 percent, 5 percent, and 3 percent of the harvested volume and 55 percent, 21 percent, 6 percent, and 13 percent of the basal area, respectively. The study site encompasses about 35 acres, with about one-half of the acreage regenerated in fall 2014 using the alternating strip clearcut method. Harvest and residual strips were each 150-feet wide and oriented in a north–south direction. All stems >1 inch diameter at breast height were felled using conventional equipment for the region (e.g., chainsaws and rubber-tired cable skidders). No additional treatments were applied to the site. The residual strips are expected to be harvested in 5-10 years.

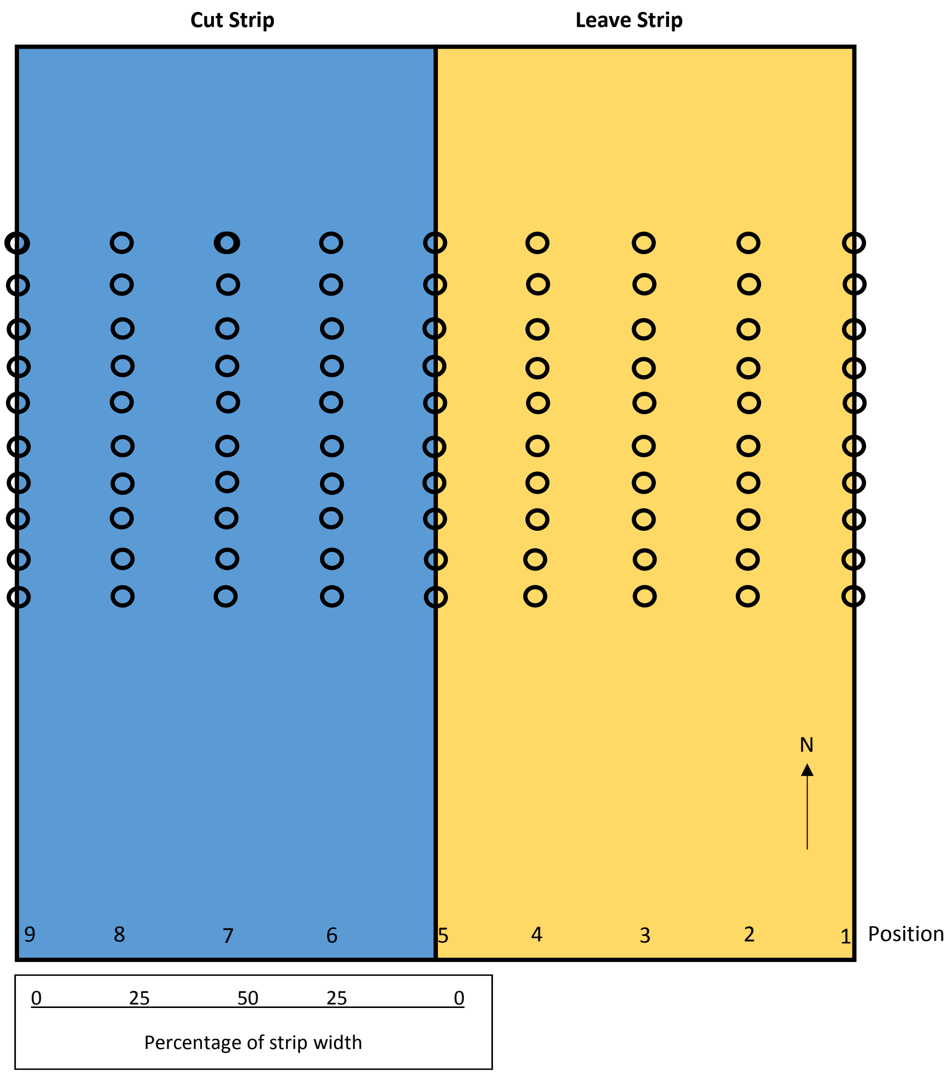

Figure 1.—Illustration showing block layout and planting position. Positions 1, 5, and 9 represent locations along the edges, positions 2, 4, 6, and 8 are 37.5 feet from the nearest edge, and positions 3 and 7 are in the center of each strip.

In April 2015, 360 2-0 northern red oak seedlings were planted across the site. Ten seedlings were planted 5 feet apart in each of nine locations within each residual/cut strip pairing that represented a gradient of light conditions (Fig. 1). Positions 1, 5, and 9 represent locations along the edges of the cut/leave strips. Positions 3 and 7 are in the center of each strip, and the remaining locations (positions 2, 4, 6, and 8) are halfway between the center and the nearest edge.

Four separate strip pairings were used as replication blocks. Each block was consistent relative to slope position, stoniness, and distance from skid trail. The best 360 of 500 seedlings based on morphological characteristics were selected. The seedlings selected for planting had the following average characteristics: 0.32 inch root collar diameter (RCD), 17.6 inches in height, and 7.4 first-order lateral roots (FOLRs) (Table 1). Fern competition developed during the first part of the growing season and was controlled midsummer using a directed application of Oust® herbicide.

Table 1.—Morphological characteristics of the planted seedlings

Characteristic	Average	Range	Recommended[a]
RCD (inches)	0.32	0.23-0.55	0.39
Height (inches)	17.6	11-28	20
No. FOLR	7.4	4-21	5

[a] Based on recommendations in Dey et al. 2012.

Initial measurements recorded for each seedling were RCD, total height, and the number of FOLRs. Basal diameter and total height were measured after the first growing season. In addition, a spherical densitometer was used to estimate overhead canopy cover. Readings were taken in early September 2015 from the center of each 10-seedling row.

Data were analyzed as a randomized complete block design (n = 4) with subsampling (10 seedlings per location per block) using analysis of variance to test the hypothesis that seedling growth varies for the nine positions within the cut/leave strip pairing. Initial seedling height and diameter were used as covariates. Statistical tests were conducted using an alpha = 0.05 level of significance.

RESULTS

Densitometer measures showed distinct differences in canopy coverage by position within the strip clearcuts (Table 2). The interior portions of the residual strips (positions 2, 3, and 4) had almost complete canopy coverage (~95 percent), the edges (positions 1, 5, and 9) had moderate coverage (49-76 percent), and the interior positions (positions 6, 7, and 8) of the cut strips had 11–31 percent coverage. After one growing season, the position across the residual and cut strips had no significant effects on height growth, diameter growth, or survival (Table 3). Although not significant, the height growth trend was that the east edge of the residual strips, which received mostly morning sun, had more growth than the other positions. For diameter growth, seedlings planted entirely within the cut strips and those planted on the west edge of the residual strips tended to have more growth. Survival varied from 87.5 percent to 100 percent across all positions. Two of the three exposed positions (within the cut areas) had the lowest survival; however, these were not significantly different.

Table 2.—The average canopy cover along a light gradient estimated by spherical densiometer in an Appalachian strip clearcut

Position[a]	Canopy coverage (%)
1	76
2	97
3	97
4	95
5	49
6	12
7	11
8	31
9	68

[a] Position descriptions are given in Figure 1.

Table 3.—Height and diameter growth one growing season after being planted in a strip clearcut

Position[a]	Height (cm)	Diameter (mm)	Survival (%)
1	7	-0.0	95
2	8	-0.3	95
3	5	-0.3	98
4	4	0.0	95
5	3	0.4	100
6	6	0.2	88
7	5	1.0	88
8	4	0.3	95
9	5	0.0	95

[a] Position description provided in Figure 1.

DISCUSSION

The strip thinning approach used here was a regeneration method that provided a compromise between a landowner's need for immediate income and the desire to maintain a strong oak component in the new stand. The continued monitoring of this enrichment planting will enable us to determine whether the gradient of illumination within the residual strips and edges of the cut strips will promote the development of oak seedlings and facilitate their growth into competitive sources.

Oak seedlings, whether natural or planted, require sufficient light resources to survive and grow into competitive sizes. The understories of mature oak stands often have light levels that are at or lower than the compensation point for oak seedlings (3-5 percent of full light) (Johnson et al. 2009). At those levels, oak survival is low. For example, 2-year survival of oak seedlings planted in unmanipulated, intact stands in Tennessee was 58 percent (Oswalt et al. 2006). Midstory removal treatments generally increase light levels to 10-20 percent of full light (Miller et al. 2004), although this response may be short-lived (Schweitzer and Dey 2015). Although no midstory was treated in the residual strips of the strip clearcut, the interior portion of the residual strips is expected to retain higher light levels for a prolonged period of time compared to typical midstory release treatments because of the reduced side shading associated with the strip cuts. With midstory treatments, light levels need to penetrate the overstory canopy before reaching the understory. In strip cuts, the sun will also penetrate laterally from the sides because most of the intercepting canopy is much higher than 50 feet. We have no photosynthetically active radiation data because midsummer to late summer 2015 had no overcast days (Parent and Messier 1996).

Treatments using harvesting equipment, herbicides, and prescribed fires have been used to increase light levels in oak stands. The degree to which canopies are opened up, however, depends on the type and nature of the competition. Care is needed on sites with aggressive intolerant species, where light conditions need to be tempered to not provide too much light that will favor their establishment. On our strip clearcut site and many other higher-quality sites in the Appalachian region, yellow-poplar, a fast-growing, shade-intolerant species, can quickly overtop oak seedlings on harvested sites (Beck and Hooper 1986, Loftis 1990) and lead to failed plantings (Schuler and Robison 2010). The center of the cut strips receives the most light (Marquis 1965), which will promote the development of species such as yellow-poplar and black cherry. Both are important and valuable species in the central Appalachian region. We are focusing on the shaded areas within and proximate to the residual strips for developing the oak component.

Artificial regeneration techniques have been used with mixed results in clearcuts and under shelterwoods. Poor growth and survival of plantings are often due to poor planting stock and competition (Dey et al. 2012). The planting stock used for this study was typical of seedlings grown at commercial nurseries in the area. In all, we culled roughly 30 percent of the seedlings purchased before planting the best available seedlings. Still, the seedlings were generally lower than current grading recommendations (Table 1), especially for height and RCD. Only 15 percent and 25 percent of the planted seedlings met the current guidelines for RCD and height, respectively; 96 percent met the guidelines for FOLRs.

After 1 year, none of the seedlings were considered competitive (i.e., >3 feet tall), nor were they expected to reach competitive status. Transplant shock probably best explains why no differences were found between positions across residual and cut strips. Transplant shock, a common occurrence after planting bareroot seedlings, is generally considered to be the result of

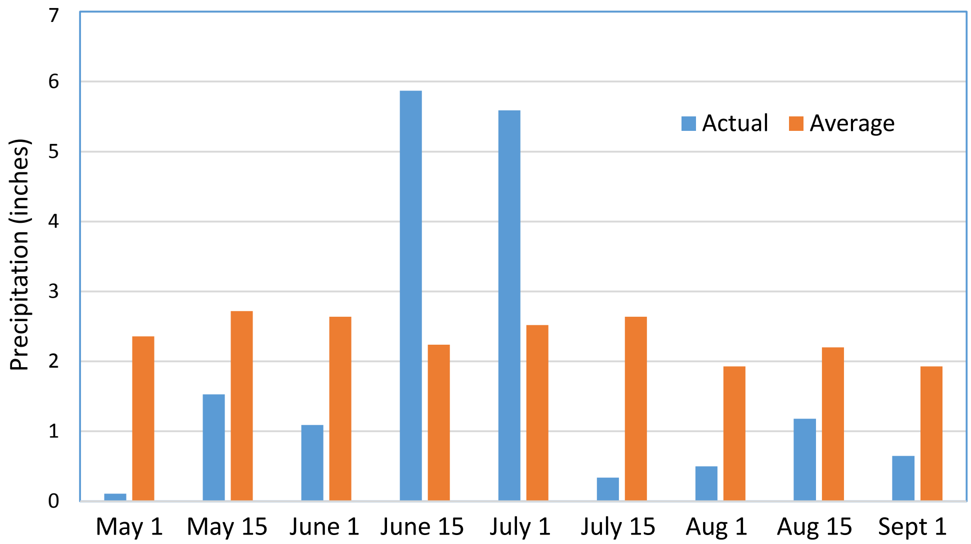

Figure 2.—Semi-monthly precipitation data for the West Virginia University Research Forest, 2015.

a disruption of water uptake caused by damaged root systems (Burdett 1990). Recently planted northern red oak bareroot seedlings under drought stress conditions have decreased biomass accumulation and root growth (Jacobs et al. 2009). Seedlings were planted for this study during early and late summer droughts (Fig. 2). More time will be required to determine if the modified light regime in the strip clearcut is successfully promoting the seedlings and simultaneously retarding the development of faster-growing species.

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EFFECT OF RESERVE TREES ON REGENERATION IN CENTRAL WISCONSIN OAK SITES

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Abstract.—Anecdotally, higher retention levels after harvest are believed to have a negative impact on oak (*Quercus*) regeneration accumulation. For this study, harvests with four retention levels of 0, 15, 30, and 45 percent were established to determine the impact of overstory retention on the success of oak regeneration after harvest. Previous research has shown that an increase in reserve trees can increase the accumulation of the intermediate shade-tolerant oak and of other species with similar shade tolerance and reduce the abundance of more intolerant species such as aspen. Seven to eight years after harvest, the regeneration levels in all plots met the requirements for adequate stocking based on standards set by the Wisconsin Department of Natural Resources. There were no significant differences in oak regeneration between retention levels. At this stage of regeneration, oak can apparently persist under higher retention levels, but further study will be necessary to determine recruitment into the canopy.

INTRODUCTION

Oaks growing on dry, nutrient-poor sites are an important cover type in central Wisconsin. The stands are generally composed of northern pin (*Quercus ellipsoidalis* E.J. Hill) and black oak (*Q. velutina* Lam.) with variable representation of white, jack, and red pine (*Pinus strobus* L., *P. banksiana* Lamb., and *P. resinosa* Aiton), red maple (*Acer rubrum* L.), aspen (*Populus tremuloides* Michx.), and bur and white oak (*Q. macrocarpa* Michx. and *Q. alba* L.). Coppice and overstory removal has been a generally accepted practice for scrub oak regeneration in Wisconsin and can be successful on these poor quality sites where oak has a strong competitive advantage (Arend and Scholz 1969).

Because the prescription, suggested in the past by the Wisconsin Department of Natural Resources, for the lowest quality of these stands has been a pulpwood rotation at 45-70 years old, the overstory trees are small and the number of stems per acre is high. Oak sprouting has been important on similar sites (Lynch and Bassett 1987). On these dry sites, sprout origin regeneration would have an important competitive advantage over the seedling regeneration of more mesic site species (e.g., red maple). In addition to the stump sprouts, oak seedlings with their tap root development (Chadwell and Buckley 2003) and adaptations to drought (Abrams 1990) can compete successfully on dry, nutrient-poor oak sites.

Even though coppice or overstory removal can be a viable harvest method for oak on dry, nutrient-poor sites, the aesthetics of partial harvests often better meets other landowner goals. Johnson (1992) demonstrated that the height of the tallest oak stem per 5.5-m^2 plot was greater as the overstory density declined; however, differences between overstory species and habitat type increased the density of oak seedling regeneration. Because oak is generally favored by disturbance in the areas of North America where fire played a significant role (Dey and Fan 2009), and because oak regeneration tends to accumulate on dry, nutrient-poor sites (Johnson et al. 2002), these scrub oak sites present a competitively favorable environment for oak

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regeneration. Higher levels of overstory retention can, however, decrease the growth rate of the oak seedlings and make them more susceptible to herbivory (Russell et al. 2001) and mortality from other environmental factors (Buckley et al. 1998). Because higher levels of overstory retention might reduce the competitive advantage of oak compared to other more shade-tolerant species (Abrams 1992), determining the level of overstory retention at which regeneration, especially oak regeneration, is compromised would be useful to managers. The objective of this study is to determine the impact of the level of overstory retention on density, aggregate height, and stocking of oak seedlings.

METHODS

Study Sites

Five dry, nutrient-poor oak sites were selected in central Wisconsin for this study. Three sites were selected in Greenwood Wildlife Area, henceforth referred to as Greenwood West (GWW), East (GWE), and South (GWS); one in the Emmons Creek Fisheries Area (EC); and one in the Henry C. Kurtz Memorial Forest (KMF). All sites but EC were harvested during the winter of 2007-2008, EC was harvested in the winter of 2008-2009. The overstory was predominantly northern pin/black oak and white/bur oak with occasional white pine, red pine, black cherry (*Prunus serotina* Ehrh), and red maple. The average ages of the stands and site index (SI) (50 years SI; data based on 10 cores taken from the trees remaining after harvest) were: EC 82 years (SI 51; size 27 acres), KMF 72 years (SI 47; size 18 acres), GWW 89 years (SI 53; size 17 acres), GWE 56 years (SI 65; size 33 acres) and GWS 68 years (SI 48; size 19 acres). Occasional individual trees (especially white and bur oaks) were significantly older. Initial basal area (mean ± se) was 143 ± 3, 101 ± 4, 150 ± 5, 153 ± 3, and 150 ± 3 square feet per acre for EC, KMF, GWW, GWE, and GWS, respectively. The soils were all quite sandy and of glacial origin. KMF was in an outwash sand plain; GWW, GWE, and GWS were in morainal sands formed by glacial deposits that are a matrix of moraine and pitted outwash; and EC was in morainal sands that were partially resculpted by meltwater and postglacial erosion with some exposed till. They consisted of Plainfield and Plainfield/Kranski (mixed, mesic Typic Udipsamments), Richford (mixed, superactive, mesic Arenic Hapludalf), Brems and Meehan (mixed, mesic Aquic Udipsamments), and Coloma (mixed, mesic Lamellic Udisamments).

Experimental Design

The experimental design of this study was a balanced randomized complete block analysis of variance with five blocks (harvest units), each with three replicates of each of four randomly assigned treatments (0, 15, 30, and 45 percent retention levels [RLs]). An alpha level of 0.10 was used to reduce the risk of type II error.

Field Methods

On each harvest unit, three replicates of the four treatments were randomly assigned to one of the twelve 0.5-acre square plots. The initial corner of the first plot was located using a random number of feet from the stand boundary, and additional plots were located based on that plot. Because these stand boundaries were irregular, plot locations were designed to fit within the boundaries of the stand. Distance between plots was roughly 60 feet, although the size and shape of the harvest unit varied. The matrix between the plots was completely harvested; all trees were removed down to a 2-inch diameter at breast height (d.b.h.). Within the plot, an approximately 33-foot buffer inside the plot was not sampled (to avoid the effect of the edge). During 2015, subplots were located at 20-foot spacings along four paced transects that were approximately 27 feet apart. Although corners of plots were initially monumented, logging damage and vandalism (many of the corner posts were stolen from two harvest units) required that these corners be relocated using GPS with its associated inaccuracies. These subplots should not be construed to be identical to the preharvest subplots, because they represent only the plot as a whole. Although this is not ideal, the data for each subplot were aggregated and the level of replication was at the level of plot (the plot was the level to which the treatment was applied). This provided 240 subplots per harvest unit, for a total of 1200 subplots. For each subplot, a round 1/735-acre regeneration subplot was established. A tally of all advanced tree regeneration by species was recorded in 2-foot height classes (0-2, 2-4, 4-6, 6-8, 8-10, and 10+ feet). Because site visit observations indicated that both wind damage and mortality had occurred since harvest, spherical densitometer measurements were taken on an additional site visit on subplots that approximated every other regeneration subplot (retaining the approximately 33-foot buffer from the plot edge). These were not taken on the original site visit because the extent of damage was not expected prior to sampling.

Data Analysis

Data were analyzed using Minitab 15.1.1.0 (Minitab, Inc., State College, PA). Analyses were performed with $\alpha = 0.10$, as the consequences of a type II error are greater than the consequences of type I error. The model for this design is:

$$\gamma = \mu + \beta_i + \tau_j + (\beta\tau)_{ij} + e_{ijk}$$

Where

γ = Aggregate height for plot l, in treatment j found in block i,

μ = Mean aggregate height across all treatments,

β_i = Block (site) effect for treatment i,

i = GWW, GWE, GWS, EC, KMF,

τ_j = Treatment effect for treatment j,

j = 0, 15, 30, or 45 percent RL,

$(\beta\tau)_{ij}$ = Error term associated with the interaction between block i and treatment j,

e_{ijk} = Error term, and

k = 1, 2, … 12 (replicate subplot).

Analysis was completed individually for aggregate height per acre (based on Fei et al. 2006) and number of seedlings per acre for oak seedlings and total tree seedlings.

Because some crown cover was lost between logging and measurement in 2015 (from wind damage and mortality), a general linear model with Block as a random factor and canopy closure (spherical densitometer) as main factors were used to determine if current canopy closure better predicted regeneration conditions.

Modified procedures from Jacobs and Wray (1992) were used to determine an oak stocking value. In our modification of this procedure, a 1-foot size class seedling was assigned a stocking value of 3, a 3-foot seedling was a 15, and a 5-foot or taller seedling was assigned a 30. A plot with a total stocking value of 30 or higher was determined to be stocked. An analysis of variance was completed by treatment level to determine if there were any significant differences.

RESULTS

There were losses of crown cover for each RL except the 0 percent RL. The measured current crown cover for RLs 0, 15, 30, and 45 percent were 0, 11 ± 1, 16 ± 2, and 31 ± 3 percent, respectively. This loss of trees was due to a combination of wind damage and mortality (likely caused by oak wilt, which is an expanding problem in this region, although the cause of mortality was not definitely determined for this project).

There were no significant differences between RL for any of the parameters tested: number of oak seedlings per acre (OS), total number of seedlings per acre (TS), aggregate oak seedling height per acre (OH), or aggregate total seedlings height per acre (TH) (Table 1). There were differences, however, in OH and TS between sites ($p = 0.06$ and 0.02, respectively). There was also no difference between RL for the proportion of plots that were stocked (based on calculated stocking value) (Table 2). Across all sites, regeneration was composed of 6 percent red oak, 37 percent black/northern pin oak, 37 percent white/bur oak, 6 percent red maple, 12 percent black cherry, and 2 percent other species.

Although there were no significant differences between the quantity and size of regeneration in the plots with the various overstory retention levels, there were significant changes in the number of seedlings between 2007 and 2015 (Table 3). Both OS and TS decreased (total number reduced); however, OH and TH essentially did not change, meaning that average

Table 1.—Mean values per acre in 2015 for number of oak seedlings (OS), total tree seedlings (TS), aggregate oak seedling height (OH), and aggregate total seedling height (TH) per acre. Data presented ± standard error. Sample size was 15 plots per retention level.

Retention (%)	OS (number/acre)	TS (number/acre)	OH (feet/acre)	TH (feet/acre)
0	1970 ± 244	2418 ± 244	6759 ± 1028	9622 ± 1445
15	2203 ± 295	2710 ± 296	6251 ± 778	9533 ± 1488
30	2279 ± 287	2849 ± 327	6302 ± 770	9231 ± 957
45	2624 ± 358	3477 ± 355	5575 ± 632	10217 ± 1772

Table 2.—Proportion of plots stocked with oak at a stocking value (SV) of 30 or more in 2015. Data presented ± standard error.

Retention level (%)	Proportion of plots ≥30 SV
0	0.23 ± 0.04
15	0.29 ± 0.05
30	0.26 ± 0.04
45	0.29 ± 0.04

Table 3.—Mean number of oak seedlings (OS), total tree seedlings (TS), aggregate oak seedling height (OH), and aggregate total seedling height (TH) per acre. Data presented ± standard error.

Year	OS (number/acre)	TS (number/acre)	OH (feet/acre)	TH (feet/acre)
2007	5064 ± 336	6279 ± 330	7150 ± 499	10805 ± 635
2015	2269 ± 149	2863 ± 158	6222 ± 401	9651 ± 706

Table 4.—Oak seedlings per acre by height class in each crown cover in 2015

	Height class (feet)					
Crown cover (%)	0-2	2-4	4-6	6-8	8-10	10+
0	588	620	500	191	59	12
15	943	715	350	108	56	29
30	1022	774	277	98	61	47
45	1529	799	196	59	22	20

height of seedlings was increasing. Even though we did not record deer browse levels in this project, a noteworthy number of the stems appeared to have a history of browse damage. The effective height of deer browsing is somewhat less than 6 feet during most years, but 101-262 oak seedlings per acre were taller than normal deer browsing height (Table 4). No significant difference in the number of oak seedlings per acre taller than 6 feet was found between RLs ($p = 0.11$), although there were differences between sites ($p = 0.062$).

DISCUSSION

Some loss of reserve trees is expected as the stands respond to harvest. Indeed, the addition of snags and course woody debris is one advantage of maintaining reserve trees in a harvest. Calculating annual mortality rates from the published values of Starkey and Guldin (2004) in Arkansas and Miller et al. (2006) in Appalachia, approximately 0.5 to 1.6 percent loss per year could be expected. On similar sites in Wisconsin, Mujuri and Demchik (2009) found a 5.5 percent rate of annual mortality for northern pin oak reserve trees and no measured loss for white oak. Considering these published values, the losses are not surprising. Expectations of some mortality and wind damage should be included in planning when reserve trees are retained on similar sites.

Regeneration of oak on dry, nutrient-poor sites is generally simpler than on richer sites. Schlesinger et al. (1993) found that through shelterwood harvesting (i.e., low-density management) adequate regeneration will develop on lower quality sites; however, higher quality sites may require further understory treatment to remove competing species. On the lower quality sites they found no detrimental effects of using the shelterwood method. Schwartz and Demchik (2013) found both natural regeneration and stump sprouts to succeed after overstory removal (clearcut/coppice) on dry, nutrient-poor oak sites (12 years after harvest) where 41 percent of the plots had at least one oak seedling taller than 10 feet even in the presence of significant deer populations (average 39 deer per square mile during the first 5 years after harvest). Oak is much better adapted to dry, nutrient-poor sites than many of its competitors. Although fire has undoubtedly played a role in the establishment of these stands in the past, it does not appear to be necessary on these dry, nutrient-poor sites for oak to dominate the regeneration. This is likely not true as soil becomes either more mesic or more nutrient rich.

The regeneration levels of all species in all treatments were generally high and met "rule of thumb" guidelines of 1000 seedlings of desirable species per acre. Following modified protocols of Jacobs and Wray (1992), the sites were 23-29 percent stocked with oak seedlings. When examining stocking based on aggregate height according to the standards described by Steiner et al. (2008), expected oak stocking would be 41-47 percent. These results support previous studies (Weigel and Johnson 1998, Weigel and Peng 2002) that showed oak can regenerate successfully on poor quality sites. Without additional site treatments that favor oak, these stands will likely be more diverse in species composition than before harvest. Oak maintenance is a management goal on these sites specifically and across state lands in Wisconsin as a whole, but increased species diversity on these dry, nutrient-poor sites is probably desirable. Oak wilt is an expanding problem in central Wisconsin, and diversifying these stands may reduce this disease's impact and speed of transmission. Because these data are from only the 7th and 8th seasons after harvest, however, we cannot predict success beyond that timeframe.

Many authors have studied the impact of modified light levels on oak regeneration success. Shelterwood has been shown to promote the growth of regeneration, but Steiner et al. (2008) suggested that it may not increase the regeneration that is established after harvest. Preharvest regeneration or site treatments to promote regeneration are often needed if adequate advanced regeneration is not present. Without sufficient regeneration before harvest, recruitment often fails. On lower quality sites, particularly those used in the current study, oak advance regeneration is typically abundant and rotation is often at a much younger age (main product of pulpwood, not saw logs as is common on richer sites). We found an average of more than 5000 oak seedlings per acre before harvest, although most were in the smallest size class (0-2 feet tall). In addition to the often copious regeneration, the contribution of stump sprouts can be significant and their growth rates very high. Lynch and Bassett (1987) found northern pin oak to stump sprout frequently even to advanced ages. Schwartz and Demchik (2013) found average stump sprout height after 12 years to be 24 feet even under significant browse pressure.

Although there is currently no statistical difference between regeneration under different overstory RLs, this may change over time as the sites continue to develop. Because crown cover decreased over time in these stands from wind damage and mortality, this may well have increased the success of the regeneration. Because these sites were measured 7-8 years since harvest, however, that may be too soon in the regeneration phase to determine any differences. Smith et al. (1989) measured regeneration in deferment harvests 5 years post-treatment and found no significant differences from what would be expected in a clearcut. This same pattern was noted in regeneration surveys 2-5 years after a shelterwood harvest (Johnson et al. 1989). Limited differences in light levels between treatments may explain the lack of differences between treatments. A study by Atwood et al. (2009) that involved three harvest levels (clearcut, leave-tree [BA = 21 square feet per acre], and shelterwood [BA = 59-51 square feet per acre]) found no differences in light levels between the leave-tree and shelterwood levels of residual trees. Although there were differences in residual densities, these differences were not significant enough to alter the light levels that reached the forest floor. In our study, oak seedlings were still the largest part of the regeneration. The lower light levels created by higher RLs could still give oak a higher competitive advantage over more shade-intolerant species such as red maple or aspen. This would provide oak sufficient time to reach a height that is more likely to achieve dominance in the canopy, as seen in studies by Sander (1971, 1972). Overall, a longer timeframe may be necessary to determine which treatment will have a greater rate of recruitment of oak, because oak success after harvest changes through time.

SUMMARY

Retention of reserve trees can help attain some ecological and aesthetic goals. For oak on dry, nutrient-poor sites, retention of 0-45 percent crown cover results in comparable number, aggregate height, and stocking value for oak seedlings during the first 7–8 years after harvest. Whether this will continue is unknown. Some mortality and wind damage of reserve trees should be expected.

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STUMP SPROUT DOMINANCE PROBABILITIES OF FIVE OAK SPECIES IN SOUTHERN INDIANA 25 YEARS AFTER CLEARCUT HARVESTING

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Abstract.—When regenerating oak or mixed-hardwood forests in southern Indiana, oak (*Quercus* spp.) stump sprouts are vital to sustaining their presence and long-term dominance. In 1987, a study began in the Hoosier National Forest in southern Indiana. The study goal was to predict the sprouting potential and dominance probability of oaks. Before clearcut harvesting, we sampled 2,188 trees of five oak species—white oak, chestnut oak, black oak, scarlet oak, and northern red oak. Measurements were taken before and 1, 5, and 25 years after clearcut harvesting. We used logistic regression to develop two preharvest predictive models and four postharvest models for dominance probabilities of the five species 25 years after harvest.

INTRODUCTION

Oaks (*Quercus* spp.) form a major component of the upland forests throughout the Central Hardwood region, and maintaining them on the regional landscape is important for wildlife, timber, and biodiversity. Regenerating oaks has been a problem in southern Indiana and surrounding regions (Fischer et al. 1987; Lorimer 1989, 1993) because of changes in forest management strategies and disturbance regimes, shifts in species composition, and unpredictable climatic influences.

Adequate oak reproduction in advance of the final harvest of even-aged stands is generally considered to be the key to ensuring oak in future stands (Dey 2014, Dey et al. 1996, Johnson et al. 2009, Sander et al. 1984). Stump sprouts that originate from harvested trees are, however, another potential source of oak reproduction. Because these sprouts are often the fastest-growing sources of oak reproduction, their contribution to future stocking is often important even when their numbers are relatively low. In fact, in many stands that originate from clearcutting, stump sprouts may comprise 50-75 percent of the basal area in oak that is free to grow or in dominant positions (Beck and Hooper 1986, Gould et al. 2002). In former oak forests in southern Indiana, Morrissey et al. (2008) found that 45 percent of dominant oak trees in 21- to 35-year-old clearcuts were from oak stump sprouts, and Swaim et al. (2016) reported that the few dominant oak trees in 23-year-old clearcuts were primarily of stump sprout origin.

Our objectives were to determine significant predictors of oak stump sprouting success in southern Indiana and to develop dominance probability models for oaks at year 25, which update previously developed models (Weigel and Peng 2002; Weigel et al. 2006, 2011). Dominance probability models permit the forest manager to make an informed prediction of the approximate number of dominant or codominant oak stump sprouts in future stands. Two model types were developed to enable the forest manager to predict dominance probability when either preharvest or postharvest data are available.

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METHODS

The study was conducted on the Hoosier National Forest in south-central Indiana. Nine stands scheduled to be clearcut were selected for measurement. There were three stands in each of three age classes: 71-90, 91-110, and 110+ years. Harvesting was done between October 1987 and May 1989. We could not determine the season (growing or dormant) during which individual stems were harvested because harvesting occurred over two seasons. For a complete discussion of the study sites, measurements, model building, and data analysis, see Weigel and Peng (2002).

Before harvest, 0.04-ha plots were established along transects in the nine stands. We inventoried and tagged 1,371 white oak (*Quercus alba* L.), 180 chestnut oak (*Q. prinus* L.), 399 black oak (*Q. velutina* Lam.), 130 scarlet oak *(Q. coccinea* Muenchh.), and 108 northern red oak (*Q. rubra* L.) >4 cm diameter at breast height (d.b.h.) on the plots. Measurements included d.b.h. on all trees and heights and ages of selected trees that were used to determine the site index. First-year measurements included determining parent-tree age by counting rings on the stump surface and noting whether any sprouts were present. Fifth-year measurements included recording the number of stump sprouts and measuring the height of the tallest sprout. At year 25, we remeasured surviving oak stump sprouts and recorded the number of sprouts, the height of the tallest sprout, and the crown class of the tallest sprout.

At age 25, oak sprout success was determined by its crown class position. By year 25 the crowns had closed; therefore, crown class provided a meaningful metric of sprout success. This measure of sprout potential, success, or competitiveness is embodied in the concept of dominance probability (Spetich et al. 2002). A successful sprout was characterized as a sprout in the dominant or codominant crown class at year 25.

We used the five-step model building approach suggested by Hosmer and Lemeshow (2000) to develop logistic regression models. We used the maximum likelihood method implemented in PROC LOGISTIC of SAS version 9.1 (SAS Institute Inc. 2004) to perform the logistic modeling.

The species were grouped into the white oak group and the red oak group for both model types. The white oak group consisted of white and chestnut oaks; the red oak group consisted of northern red, black, and scarlet oaks.

For the preharvest models the dependent variable was a dominant or codominant stump sprout 25 years after the parent stem was harvested. The independent variables were species, parent tree age, d.b.h., natural log of d.b.h., site index, natural log of site index, aspect, and interactions between two or more of these independent variables. According to Johnson et al. (2009), these are commonly the driving variables that affect sprouting and competitive relationships in regenerating oak forests.

The postharvest models used the same dependent variable as the preharvest models, but the number of independent variables was reduced so that only stump diameter, species, aspect, and site index were required.

Table 1.—Preharvest models: logistic regression models for estimating the probability that an oak stump sprout will be in either the dominant or codominant crown class at year 25

		Parameter estimates[a,b]			Model evaluation statistics	
Model No.	Species	b_0	b_1	b_2	X^2	H-L[c]
1	White oak	0.4168	2.4015	-0.1272	472.4107	12.8360
	Chestnut oak	1.8604	2.4015	-0.1272	(p<0.001)	(p=0.1176)
2	Red & black oak	0.0992	2.0586	-0.0637	200.7304	13.5027
	Scarlet oak	1.7435	2.0586	-0.0637	(p<0.0001)	(p=0.0957)

[a]Regression models are of the form $P = [1+e^{-(b_0+b_1X1+b_2X2)}]^{-1}$, where P is the estimated probability that a cut tree will produce a successful (dominant or codominant) stump sprout at age 25: *X*1 is aspect (north 315°-135° = 0, south 136°-314° = 1; Hannah 1968); *X*2 is d.b.h. in centimeters.

[b]All parameter estimates differ significantly from zero at p < 0.01.

[c]Hosmer-Lemeshow goodness-of-fit test (Hosmer and Lemeshow 2000).

RESULTS

Preharvest White Oak Group Model

At year 25, the best model developed from the preharvest variables included three of those variables: species, d.b.h., and aspect (model 1 in Table 1).

Preharvest Red Oak Group Model

Northern red oak and black oak were combined into a single group because in all previous models (Weigel and Peng 2002; Weigel et al. 2006, 2011) they did not differ significantly. This combined group did, however, differ significantly from scarlet oak.

The best model that predicted the dominance of the red oak group at year 25 (model 2 in Table 1) included species, d.b.h., and aspect.

Postharvest White Oak Group Models

Year 1: At year 1, white and chestnut oak did not differ significantly (p >0.05) from each other, so they were combined. The significant predictors of stump sprout dominance at year 25 using data collected at year 1 were diameter at stump height, aspect, and site index (model 3 in Table 2).

Year 5: Using data at year 5 to predict dominance, we found that species, diameter at stump height, and site index were all significant predictors (model 4 in Table 2).

Postharvest Red Oak Group Models

As in the red oak preharvest models, northern red and black oaks were combined into a single group that differed significantly from scarlet oak.

Year 1: At year 1 the red oak group showed three significant predictors of dominance: species, diameter at stump height, and aspect (model 5 in Table 2).

Year 5: The only significant predictors from year 5 data for the red oak group were species and site index (model 6 in Table 2).

Table 2.—Postharvest models: logistic regression models for estimating the probability that an oak stump sprout will be in either the dominant or codominant crown class at year 25 when sprouts were present at year 1 or 5 after clearcutting

		Parameter estimates[a,b]				Model evaluation statistics	
Model	Species and year	b_0	b_1	b_2	b_3	X^2	H-L[c]
3	White and chestnut oaks, 1	4.4361	1.1263	-0.0594	-0.1770	110.8413 (p<0.0001)	13.0023 (p=0.1118)
4	White oak, 5	6.6776		-0.0314	-0.2821	61.5603	10.0860
	Chestnut oak, 5	6.0473		-0.0314	-0.2821	(p<0.0001)	(p=0.2590)
5	Black and red oaks, 1	0.2922	1.1033	-0.0294		69.8513	12.9706
	Scarlet oak, 1	2.2087	1.1033	-0.0294		(p<0.0001)	(p=0.1129)
6	Black and red oaks, 5	5.1520			-0.2450	52.0984	5.0943
	Scarlet oak, 5	6.5769			-0.2450	(p<0.0001)	(p=0.6485)

[a]Regression models are of the form $P = [1+e^{-(b_0+b_1X1+b_2X2+b_3X3)}]^{-1}$, where P is the estimated probability that a cut tree will produce a successful (dominant or codominant) stump sprout at age 25: *X*1 is aspect (north 315°-135° azimuth = 0, south 136°-314° azimuth = 1) (Hannah 1968); *X*2 is cut tree stump diameter in centimeters 15 centimeters above ground level; *X*3 = site index (where site index is derived from Carmean et al. 1989).

[b]All parameter estimates differ significantly from zero at p < 0.01 except for species in the white oak group year 1 model that differs significantly from zero at p < 0.10.

[c]Hosmer-Lemeshow goodness-of-fit test (Hosmer and Lemeshow 2000).

DISCUSSION

For all models, whether the prediction input variables were from preharvest data or postharvest data, and regardless of broad species grouping (red or white oak), dominance probabilities decreased with increasing diameters of cut trees.

Preharvest

As in previous year models (Weigel and Peng 2002; Weigel et al. 2006, 2011), chestnut oak had higher dominance probabilities at year 25 than white oak (Fig. 1). Parent tree age and site index were, however, no longer significant variables in predicting dominance. A more general site quality variable, aspect, was significant (Hannah 1968). Aspect is one of several topographic factors that influences site index (Johnson et al. 2009), therefore, aspect and site index are correlated. The inclusion of one or both variables in the best model is likely influenced by sample size, data structure nuances, and artifacts of statistical analysis. The interpretation remains the same: oak dominance probabilities increase as site quality decreases, which is indicated by lower site index or hotter, drier, more exposed aspects. In general, oaks are more drought tolerant than many of their major competitors and are better able to persist on more xeric sites of lower productivity (Johnson et al. 2009). The diversity, abundance, and growth potential of oak competitors are also significantly less on the lower-quality sites (Kabrick et al. 2008, 2011, 2014). Parent trees for both species located on south aspects (136°-314°) had a higher probability of producing dominant or codominant sprouts than those found on a north aspect (315°-135°). Because southern aspects tend to have lower site indices (Johnson et al. 2009), this is similar to previous models where lower quality sites had higher dominance probabilities. Competition from faster growing species that are typically found on higher-quality sites limited the oaks' dominance probabilities on the better quality sites; however, on the lower-quality sites the oaks were able to successfully compete with the other species. Several studies in oak forests of the Central Hardwood region have shown increases in diversity, abundance, and competitiveness of oak competitors with increasing site quality, productivity, and index (Dey et al. 2009; Kabrick et al. 2008, 2011, 2014).

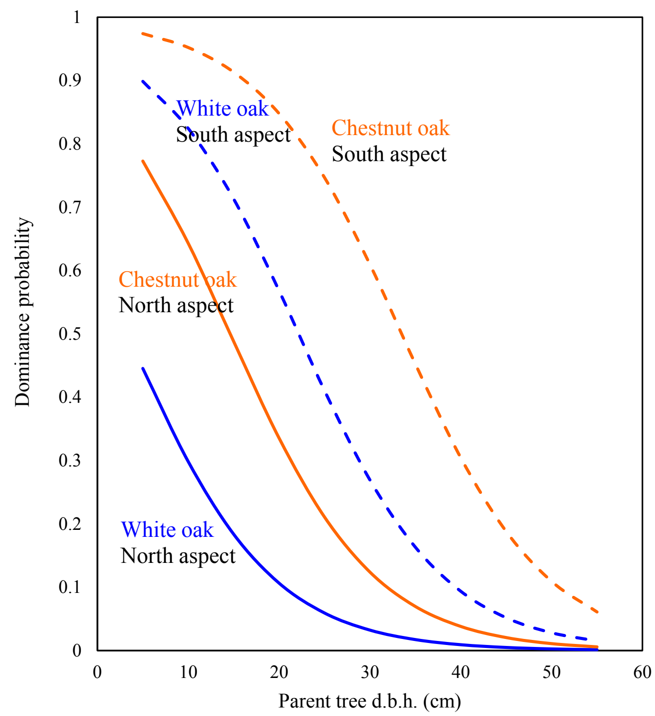

Figure 1.—Estimated probability (Table 1) that a white oak or chestnut oak stump will produce a sprout that is either dominant or codominant 25 years after the parent tree is cut in a clearcut regeneration harvest based on preharvest d.b.h. and aspect.

Chestnut oaks that are 10 cm d.b.h. and grow on a south aspect have a 95 percent probability of producing a dominant sprout 25 years after harvest; white oaks of a similar size have an 82 percent probability. The forest manager would know after inventorying the stand that 95 percent of all 10-cm chestnut oak on the south slopes would produce a dominant or codominant sprout 25 years after the harvest.

For the red oak group, data related to preharvest scarlet oak again resulted in higher dominance probabilities at age 25 than those of northern red or black oaks (Fig. 2), a result similar to previously published models (Weigel and Peng 2002; Weigel et al. 2006, 2011). For estimating dominance at age 25, parent tree age was no longer a significant predictor. The more general site quality variable, aspect, was significant; southern aspects have higher probabilities than northern aspects. As with the white oak group, the red oak group species were better able to compete on the lower-quality sites that tend to have less competition.

Postharvest

Year 1: Species was not a significant predictor of dominance at age 25 for the white oak group (Table 2). This is similar to previous models (Weigel and Peng 2002; Weigel et al. 2006, 2011). Site index remained a significant predictor as in these models; the more general variable, aspect, became significant. As in previous models, trees found on poorer-quality sites had higher dominance probabilities (Fig. 3).

The combined northern red and black oak grouping differed significantly from scarlet oak at year 1 (Table 2). By year 25, site index was no longer a significant predictive variable, but aspect was, a change from models developed for previous years. Scarlet oak had higher dominance probabilities than northern red or black oaks regardless of aspect, which was similar to previous models that indicated scarlet oak had higher dominance probabilities regardless of site index (Fig. 4).

Year 5: Aspect was no longer a significant predictor for the white oak group. Compared to year 1 results, white oak had higher dominance probabilities than chestnut oak (Fig. 5). Although aspect was no longer a significant predictor, site index did remain significant. As in previous models, trees on lower site index sites were predicted to have higher dominance probabilities.

Species for the white oak group became a significant predictor at year 25, which is different than in previous models (Weigel and Peng 2002; Weigel et al. 2006, 2011). A possible explanation was an increased mortality of chestnut oak sprouts from previous years. Field observations indicated that some large chestnut oak sprouts broke loose from the parent stump.

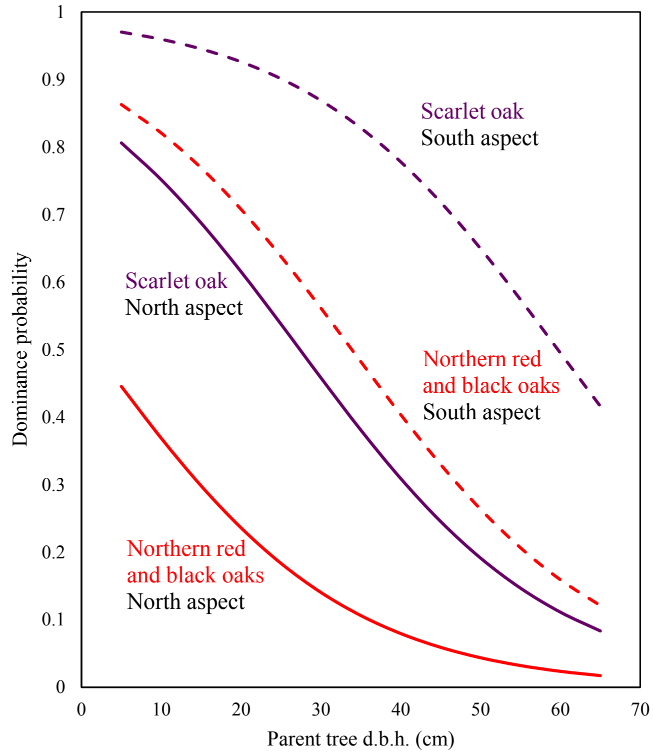

Figure 2.—Estimated probability (Table 1) that a black and northern red oak or scarlet oak stump will produce a sprout that is either dominant or codominant 25 years after the parent tree is cut in a clearcut regeneration harvest based on preharvest d.b.h. and aspect.

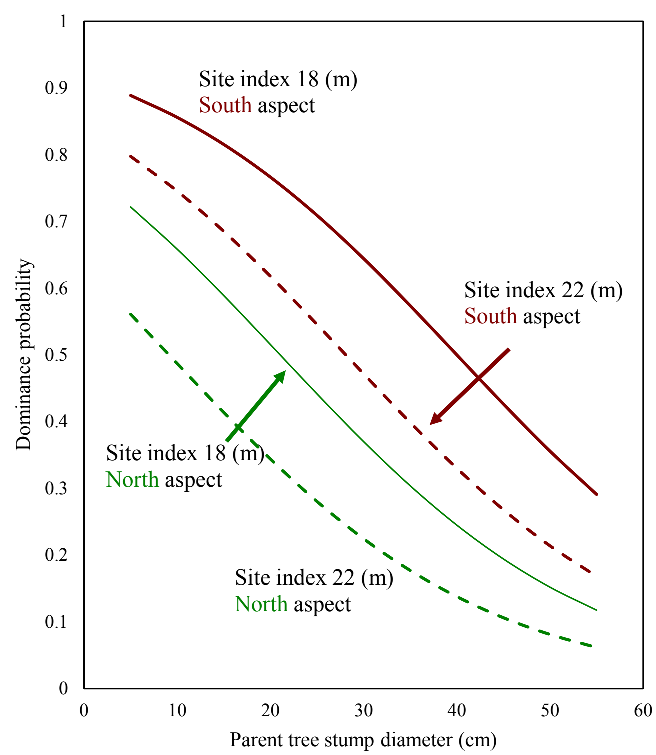

Figure 3.—Estimated probability (Table 2) that a white oak or chestnut oak stump sprout will be either dominant or codominant at year 25 when sprouts were present at year 1 after clearcutting based on aspect, stump diameter, and site index.

Species and site index were significant predictors for the red oak group; however, the dominance probabilities exceed 99 percent for both the combined northern red and black oak grouping and scarlet oak. This suggests very low mortality after year 5. If a sprout is present at year 5, it will be dominant or codominant at year 25. This is a higher dominance probability than reported by Weigel et al. (2011). Previous models included the interaction of site index with the natural log of site index. The present model does not contain this interaction, which may be related to the increased dominance probability.

CONCLUSION

The six models presented are valuable for predicting the contribution of stump sprouts to forest regeneration. The models allow forest managers to predict the percentage of oak stump sprouts that will be competitive 25 years after an even-age timber harvest. Models 1 and 2 can be used to predict the probability of dominant and codominant stump sprouts 25 years after a clearcut harvest based on preharvest information. These models also permit forest managers to assess the contribution of stump sprouts to the desired stocking of oak advanced reproduction. In addition, it allows them to adjust stand prescriptions to promote oak advance reproduction by reducing the vigor and abundance of major woody competitors.

Models 3 through 6 predict the probability of dominant and codominant stump sprouts at year 25 based on stumps sprouts being present at year 1 or year 5. Forest managers can then assess the need for crop tree release or another type of precommercial thinning to maintain the desired stocking of oak. Forest modelers can use these models to predict and describe the influence of oak stump sprouts on future stands and stand stocking.

Our analysis differs from many other stump sprout studies by predicting the contribution of stump sprouts to the future stand and hence the sustainability of oak in that stand. Our model incorporates data regarding whether a stump produces sprouts, whether those sprouts survive and grow, and how competitive these sprouts are relative to competing vegetation. Another unique quality of this study is that it provides a long-term evaluation of stump sprouts. We examined the fate of oak stump sprouts at age 25, when crowns close and differentiate; this gives forest managers a better understanding of stump sprout potential in the future stand.

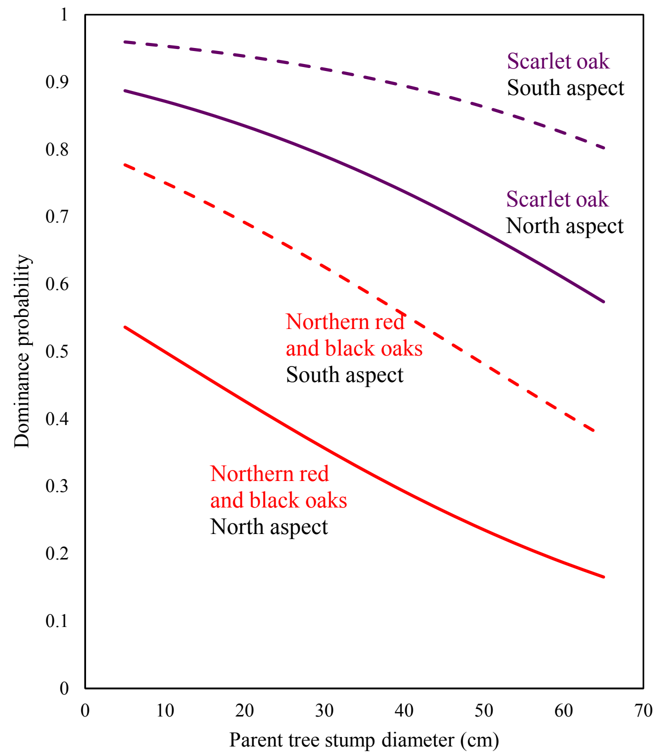

Figure 4.—Estimated probability (Table 2) that a black and northern red oak or scarlet oak stump sprout will be either dominant or codominant at year 25 when sprouts were present 1 year after clearcutting, based on stump diameter and aspect.

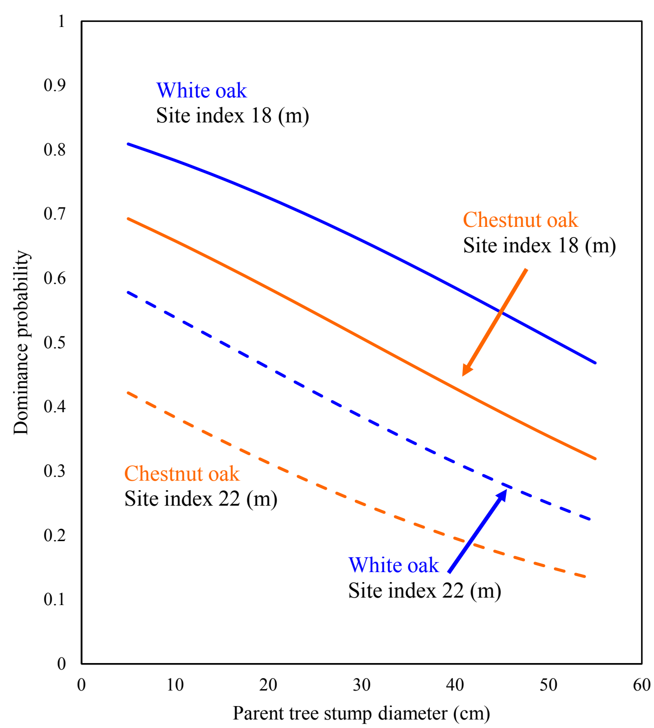

Figure 5.—Estimated probability (Table 2) that a white oak or chestnut oak stump sprout will be either dominant or codominant at year 25 when sprouts were present at year 5 after clearcutting, based on stump diameter and site index.

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SOILS AND NUTRIENTS

MASS LOSS AND NUTRIENT CONCENTRATIONS OF BURIED WOOD AS A FUNCTION OF ORGANIC MATTER REMOVAL, SOIL COMPACTION, AND VEGETATION CONTROL IN A REGENERATING OAK-PINE FOREST

Felix Ponder, Jr., John M. Kabrick, Mary Beth Adams, Deborah S. Page-Dumroese, and Marty F. Jurgensen[1]

Abstract.—Mass loss and nutrient concentrations of northern red oak (*Quercus rubra*) and white oak (*Q. alba*) wood stakes were measured 30 months after their burial in the upper 10 cm of soil in a regenerating forest after harvesting and soil disturbance. Disturbance treatments were two levels of organic matter (OM) removal (only merchantable logs removed or removal of all woody material plus forest floor) with two levels of soil compaction (C), not compacted and severely compacted. Treatments were arranged in a factorial design with each treatment plot split with and without vegetation control (VC). The VC treatment increased the northern red oak stake decomposition by 32 percent and increased nitrogen and sulfur concentrations in northern red oak and white oak stakes. The VC treatment also increased phosphorus and calcium concentrations in white oak stakes, even though decomposition of these stakes was nominal and not statistically significant. Findings suggest that postharvesting efforts to reduce competition during forest regeneration will have a greater impact on belowground woody debris decomposition and nutrient availability than do OM removal and C in xeric, low fertility oak-hickory and oak-pine ecosystems in the Central Hardwood region.

INTRODUCTION

Nutrient availability in forest ecosystems can be influenced by organic matter (OM) decomposition that is in turn influenced by factors such as climate, initial site nutrient level, and litter quality (Melillo et al. 1982). Decomposition is a critical process in nutrient cycling within temperate forest ecosystems (Binkley and Fisher 2013). Decomposition of detritus can provide 69–87 percent of the nutrients needed annually for forest growth (Waring and Schlesinger 1985). This material provides habitat for numerous vertebrates and invertebrates and serves as substrate for a variety of fungi and other microorganisms. Nutrient deficiencies, especially nitrogen (N) and phosphorus (P), are often found in highly leached soils, such as Alfisols and Ultisols, which are common soils throughout the Central Hardwood Forest region. Therefore, on these soils, the removal of forest biomass could represent the loss of many potential site nutrients that would otherwise be available after decomposition. This would affect site productivity.

Forest harvesting affects the decomposition rates of OM and consequently nutrient availability through a number of mechanisms (Ponder et al. 2012, Yanai et al. 2003). Forest harvesting changes soil moisture and temperature and removes nutrients stored in the aboveground biomass (Page-Dumroese et al. 2010). Soil compaction caused by harvesting equipment can increase or

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decrease water-holding capacity, reduce aeration, and restrict root growth and nutrient uptake (Page-Dumroese et al. 2006). The subsequent regrowth of forest vegetation reduces nutrient availability through plant uptake where excessive biomass has been removed (Mendham et al. 2003). The forest floor is particularly important for moderating water storage, for protecting the soil surface from erosion, and for providing nutrients through mineralization. Harvesting has the most impact on the forest floor (Page-Dumroese et al. 2010, Tiedemann et al. 2000).

Considerable research has been conducted to identify factors that influence litter decomposition rates. Factors such as litter quality (Giardina et al. 2001, Melillo et al. 1982), macroclimate and microclimate variables (Zhang et al. 2008), soil fertility (Prescott 2010), and microbial and fungal biotic activity (Kamplichler and Bruckner 2009, Seastedt 1984) have been identified. During decomposition, nutrient changes generally follow three phases: (1) initial nutrient release through leaching, (2) net immobilization when the microorganisms retain or import nutrients during decomposition, and (3) nutrient release from the OM at a rate close to mass loss (Prescott et al. 1993). Variability in factors associated with OM decomposition such as soil fauna (Yan et al. 2004), in addition to differences in OM type and quality, species, and ecosystem can, however, alter this pattern (Sanchez 2001).

After conventional harvesting in mature oak-hickory (*Quercus-Caraya*) and oak-shortleaf pine (*Quercus-Pinus*) forest types, large amounts of forest biomass from the harvested trees remain on site as branch wood and roots. In these ecosystems less than half the standing biomass in a forest clearcut is removed (Ponder and Mikkelson 1995). The relative contribution of woody as opposed to leaf biomass, however, has received less attention as a component of decomposition and nutrient cycling. Depending on the size of the material, wood may take longer than leaf litter to decompose. Several studies have shown how land management alters forest floor decomposition (Berg 2000, Berg et al. 1996, Breymeyer 2003) or woody material on the soil surface (Brown et al. 1996, Palviainen et al. 2004), but few studies describe wood decomposition within the mineral soil (Jurgensen et al. 2006).

The importance of woody debris decomposition for retaining carbon, replenishing nutrients, and maintaining soil quality in managed forests remains controversial (Holub et al. 2001, Laiho and Prescott 2004, Smith et al. 2007). Although some studies suggest that woody debris comprises only a small proportion of carbon and nutrient pools in forest ecosystems (Laiho and Prescott 2004), others indicate that woody debris serves as an important sink for nutrients during decomposition (Johnson et al. 2014, Shortle et al. 2012, Smith et al. 2007) and that the slow release of stored nutrients during woody debris decomposition may be a significant source of nutrients, particularly in infertile soils.

The primary aim of this study was to measure mass loss and nutrient (N, P, potassium [K], calcium [Ca], magnesium [Mg], and sulfur [S]) dynamics of decomposing wood in mineral soil. Wood stakes from two tree species that commonly inhabit forests in the Central Hardwood region—northern red oak (*Q. rubra* L.) and white oak (*Q. alba* L.)—were used as surrogates for woody debris of these two species. The specific objective was to compare the effects of OM removal and soil compaction, with and without vegetation control, on mass loss and nutrient concentrations in the decomposing wood stakes after burial in a regenerating forest for 30 months.

MATERIALS AND METHODS

Site Description

The study is in the U.S. Department of Agriculture, U.S. Forest Service's Long-Term Soil Productivity (LTSP) site located in the Current River Conservation Area in Shannon County, MO. The soil is primarily derived from Ordovician and Cambrian dolomite with some areas of pre-Cambrian igneous rock (Missouri Geological Survey 1979). Weathered material forms a deep mantle of cherty hillslope sediments, which overlie cherty residuum (Gott 1975). Soils derived from this residuum are primarily of the Clarksville series (loamy-skeletal, mixed, mesic Typic Paleudults). The soils are excessively drained and have an available water-holding capacity of <10 cm. Initial soil chemical properties of the 0- to 20-cm depth were pH, 5.7; total carbon, 33 g kg^{-1}; total N, 1.1 g kg^{-1}; P, 16.9 mg kg^{-1}; Ca, 789 mg kg^{-1}; and Mg, 61 mg kg^{-1} (Ponder et al. 2000). Before harvest, the site was occupied by a fully-stocked, mature, second-growth oak-hickory forest. The site index indicated that the height of 50-year-old black oak (*Q. velutina*) ranges from 23 to 24 m on this site (Hahn 1991). Data from a weather station on site indicated that the mean annual precipitation is 1120 mm and the mean annual temperature is 13.3 °C.

EXPERIMENTAL DESIGN

The experiment was a three-factor randomized split-plot design with three replications of two forest floor removal treatments (whole plot), two compaction treatments (whole plot), and two vegetation control (VC) treatments (split plot). The two levels of OM removal were (1) merchantable boles only (OM_0), in which only merchantable saw logs were removed, and (2) removal of all woody material during harvesting plus forest floor (OM_2), in which all aboveground biomass (merchantable and unmerchantable trees plus forest floor) was removed. Litter that accumulated during the study was not removed. The two levels of soil compaction (C) were none (C_0) and severe compaction (C_2). During harvesting, heavy equipment was kept off the C_0 plots. Soil compaction was accomplished by using heavy road-construction equipment to make multiple passages over the plot. Mean bulk density in the C_2 plots increased to 1.8 g cm^{-3} after site preparation compared to 1.3 g cm^{-3} for the C_0 treatment. The two levels of VC were no vegetation control (VC_0) and vegetation control (VC_1). VC included the annual application of glyphosate to all vegetation that competed with northern red oak, white oak, and shortleaf pine seedlings that were planted at this location as part of the LTSP study. The only plants remaining in the VC_1 treatment were the planted northern red oak and white oak and shortleaf pine seedlings at a spacing of 3.66 m. There was a ratio of three oaks of each of the two species to each shortleaf pine seedling planted. This treatment was applied to half of each 0.4-ha square plot. Ponder and Mikkelson (1995) give a complete description of the site and the LTSP installation.

Treatment units were established in 1994 and wood stakes were installed 5 years later. Three control (unharvested) plots were located adjacent to the harvested stand, with stocking and site index that were similar to harvested treatment plots.

Preparation of Wood Stakes

Two mature trees—a northern red oak and a white oak from the site—were processed into stakes that were 15.24 cm long × 2.54 cm wide × 0.31 cm thick. The fresh logs were first cut into 2.54 cm thick boards (96.52 cm × 15.24 cm wide × 2.54 cm thick). After drying for 3 months, boards were ripped lengthwise to 0.31 cm thick and cut into 15.24 cm lengths. Holes that were 0.31 cm were drilled into stakes for bundling replicates. Machining burs, sawdust, and other wood fragments were removed from the stakes. The stakes were numbered and weighed before oven-drying at 70 °C for 72 hours. Once oven-dried, 10 stakes of the same species were weighed, strung together (grouped) in a circle with aluminum wire, labeled according to plot, and stored in plastic bags in a cool dry storage room until they were placed in plots. Each group contained a numbered aluminum identity tag.

Field Installation of Wood Stakes

Approximately 5 years after the OM was removed and C treatments were applied, two groups of 10 stakes of each species were buried within the buffer area on each of 12 treated plots and 3 uncut control plots. The buffer area, which received the same treatment as the core or measurement plot, was the outer 6- to 9-m-wide area of the plot that was designated for auxiliary studies that would not interfere with the integrity of core plot area. An area measuring 182.9 cm × 121.9 cm was excavated to a depth of 10.2 cm. Within the excavated area of each plot, the two groups of each species of stakes were placed flat (wide side down) in the soil, side by side with wire still attached. Numbered pins were inserted through the aluminum wire strung through each group of stakes and in the soil to mark the location. Stakes were covered with soil to the original approximate soil depth and bulk density. Reference stakes (those not buried) of each species were kept in plastic bags in a frost-free refrigerator for the duration of the study and analyzed for macronutrient element concentrations when buried stakes were analyzed.

Stake Removal, Mass Loss Calculation, and Chemical Analysis

Approximately 29-30 months after burial, a masonry tool (Fig. 1A) and broom (Fig. 1B) were used to carefully excavate the stakes. The groups of stakes (Fig. 1C and Fig. 1D) were placed in individual plastic bags; labeled according to plot, treatment, date, and species; and transported to the laboratory. At the laboratory, they were removed from the bags and inspected for decay. All attached soil was also gently removed before the stakes were oven-dried at 70 °C for 72 h and

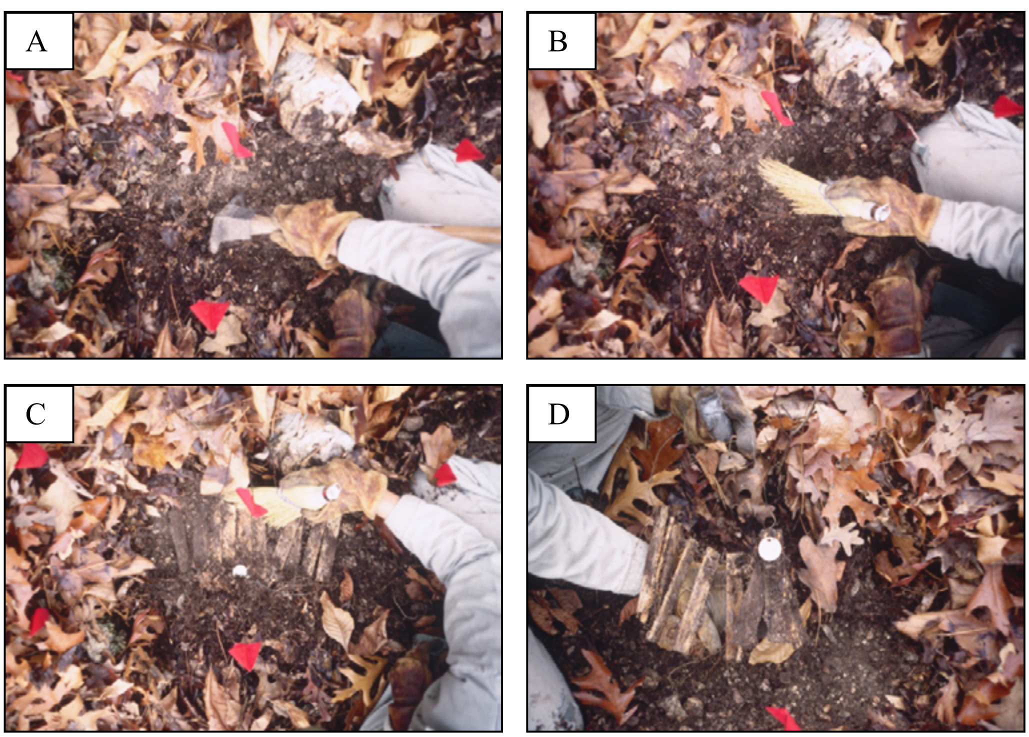

Figure 1.—Procedure for lifting stakes after 30 months for mass loss and nutrient determinations. Photo sequence includes excavating with a masonry tool (A) and broom (B and C). Final photo shows a group of 10 stakes fastened together with a wire loop (D).

weighed. Percent mass loss was calculated as the difference between the initial oven-dry-mass minus the 30-month (final) oven-dry mass expressed as a percentage of the initial oven-dry mass. We followed the approach used by Risch et al. (2013) to calculate the decay constant k and the half-life of the stakes using equations (1) and (2):

$$k = \ln (M_0/M_t)/t \qquad (1)$$

$$\text{half-life} = 0.693/k \qquad (2)$$

Where

M_0 = the initial mass of the wood stake before burial, and

M_t = the mass t months after removal (which was 29 or 30 months depending on the exact removal date).

Depending on the amount of decay, stakes were either broken by hand or sawed with a band saw into pieces small enough to be ground in a Wiley mill to pass through a 2 mm mesh sieve. Samples of unburied (reference) stakes were also prepared and analyzed for nutrients. Chemical analyses were conducted at the University of Arkansas Agricultural Diagnostic Laboratory at Fayetteville. Total N was determined by dry combustion (Pella 1990). Other nutrients (P, K, Ca, Mg, and S) were determined by inductively coupled plasma emission spectroscopy (PerkinElmer Corporation 1983) from samples extracted with a HNO_3-H_2O_2 digestion procedure (Plank 1992).

Statistical Analysis

Analysis of variance was used to test for treatment effects on mass loss and N, P, K, Ca, Mg, and S concentrations for wood stakes removed after 30 months. The data were analyzed with the GLIMMIX procedure (Version 9.3, SAS Institute, Inc., Cary, NC) where fixed effects included the OM, C, and VC treatments, their first-order interactions, and the three-way interaction among OM removal, C, and VC treatments. Because of the split-plot design, a random statement was included in the model to specify that the VC × plot interaction was the error term. Control plots were not included in the analysis because they did not include the OM removal, C, or VC treatments, but data are presented for comparative purposes.

Table 1.—Percent mass loss ± standard error for northern red oak and white oak wood samples 30 months after burial in the soil surface in regenerating forest plots with organic matter removal, compaction, and vegetation control treatments

Treatment[a]	Northern red oak	White oak
	------------------------------ Percent loss	----------------------------
OM_0	65.3 ± 5.4	52.3 ± 4.2
OM_2	81.0 ± 6.6	65.8 ± 5.5
C_0	74.4 ± 6.1	61.9 ± 4.9
C_2	71.4 ± 5.8	55.6 ± 4.7
VC_0	**62.8 ± 4.7**	56.5 ± 4.5
VC_1	**83.2 ± 6.2**	60.9 ± 5.2
Control	81.6 ± 6.1	60.1 ± 12.8

[a] Treatment codes are OM_0 (bole removal), OM_2 (whole tree plus forest floor removal), C_0 (no compaction), C_2 (severe compaction), VC_1 (with vegetation control), VC_0 (without vegetation control), and control. For each OM, C, or VC treatment, values that are significantly (P < 0.05) different from each other are shown in bold. Control was not included in the analysis.

RESULTS

Mass Loss

Thirty months after burial, the mass loss (percent weight basis) of northern red oak stakes was 1.3 times greater (P = 0.02) in the VC_1 treatment than in the VC_0 treatment (Table 1). There were no other significant treatment effects on northern red oak stake decomposition, although there was a nominal increase in decomposition in the OM_2 relative to the OM_1 treatment. There were no significant treatment effects for the mass loss of buried white oak stakes, even though there were nominal differences that paralleled those of the northern red oak stakes. In addition, the mass loss for northern red oak stakes was about 1.5 times greater than for the white oak stakes, independent of treatment.

Because VC was the only significant treatment associated with mass loss, we calculated the mean (± standard error) half-life for northern red oak and white oak stakes for this treatment. For northern red oak stakes, the half-lives for stakes were estimated to be 11.2 ± 1.2 and 17.2 ± 1.2 months for the VC_1 and VC_0 treatments, respectively. For the white oak stakes, the half-lives were 20.1 ± 1.2 and 23.2 ± 1.2 months for the VC_1 and VC_0 treatments, respectively.

Nutrient Concentrations

The VC treatment, with one exception, was the only treatment that exhibited significant effects on nutrient concentrations in the buried stakes (Table 2). In stakes of both species, the N (P < 0.01) and S (P < 0.03) concentrations increased by at least 1.5 times in the VC_1 treatment compared to the VC_0 treatment after burial for 30 months. In the white oak stakes, the VC_1 treatment also increased the P concentrations by 2 times (P = 0.01) and the Ca concentrations by 1.5 times (P = 0.04). The only other significant treatment effect occurred with the OM treatment; the Mg concentration of the buried white oak stakes that received OM_2 was 1.5 times greater (P = 0.04) than that observed after the OM_0 treatment. For most nutrients, concentrations in the buried stakes were similar to those buried in the control treatment and generally 2-10 times greater than in the unburied reference stakes (Table 2).

Table 2.—Mean nutrient concentrations ± standard error for northern red oak and white oak stakes buried in the surface soil of a regenerating forest plots with organic matter removal, compaction, and vegetation control treatments

Treatment[a]	N	P	K	Ca	Mg	S
	g kg^{-1}					
Northern red oak						
OM_0	5.63 ± 0.65	0.18 ± 0.04	0.76 ± 0.08	5.46 ± 1.17	0.33 ± 0.07	0.41 ± 0.06
OM_2	6.26 ± 0.67	0.25 ± 0.05	0.73 ± 0.07	4.27 ± 1.16	0.54 ± 0.11	0.39 ± 0.05
C_0	6.16 ± 0.70	0.22 ± 0.05	0.78 ± 0.08	5.32 ± 1.17	0.40 ± 0.08	0.41 ± 0.06
C_2	5.72 ± 0.63	0.20 ± 0.04	0.71 ± 0.07	4.45 ± 1.16	0.44 ± 0.09	0.39 ± 0.05
VC_0	**4.43 ± 0.42**	0.18 ± 0.04	0.73 ± 0.34	4.11 ± 1.15	0.42 ± 0.09	**0.32 ± 0.04**
VC_1	**7.95 ± 0.77**	0.25 ± 0.05	0.75 ± 0.07	5.67 ± 1.15	0.42 ± 0.09	**0.49 ± 0.06**
Control	3.86 ± 0.74	0.18 ± 0.06	0.74 ± 0.25	3.97 ±1.04	0.31 ± 0.11	0.31 ± 0.05
Not buried (reference)	1.42 ± 0.46	0.02 ± 0.01	0.55 ± 0.05	1.08 ± 0.81	0.10 ± 0.05	0.10 ± 0.03
White oak						
OM_0	4.61 ± 0.41	0.16 ± 0.02	0.72 ± 0.12	5.85 ± 1.15	**0.29 ± 0.04**	0.38 ± 0.04
OM_2	4.84 ± 0.42	0.18 ± 0.02	0.58 ± 0.09	5.22 ± 1.15	**0.43 ± 0.05**	0.35 ± 0.04
C_0	5.35 ± 0.48	0.19 ± 0.02	0.78 ± 0.13	5.85 ± 1.15	0.41 ± 0.05	0.41 ± 0.05
C_2	4.18 ± 0.37	0.15 ± 0.02	0.54 ± 0.09	5.22 ± 1.15	0.30 ± 0.04	0.33 ± 0.04
VC_0	**3.89 ± 0.31**	**0.12 ± 0.01**	0.59 ± 0.09	**4.56 ± 1.14**	0.35 ± 0.04	**0.30 ± 0.03**
VC_1	**5.74 ± 0.43**	**0.25 ± 0.03**	0.71 ± 0.10	**6.70 ± 1.13**	0.35 ± 0.04	**0.45 ± 0.04**
Control	3.95 ± 0.82	0.19 ± 0.07	0.61 ± 0.22	3.76 ± 0.96	0.40 ± 0.09	0.32 ± 0.08
Not buried (reference)	1.49 ± 0.05	0.02 ± 0.01	0.58 ± 0.03	0.69 ± 0.51	0.06 ± 0.04	0.08 ± 0.02

[a] Treatment codes are OM_0 (bole removal), OM_2 (whole tree plus forest floor removal), C_0 (no compaction), C_2 (severe compaction), VC_1 (with vegetation control), VC_0 (without vegetation control), and control. For each OM, C, or VC treatment, values that are significantly ($P < 0.05$) different from each other are shown in bold.

DISCUSSION

We observed that the OM and C treatments had little effect on the mass loss or nutrient concentrations in oak stakes that were buried for approximately 30 months. The OM_2 treatment may have marginally enhanced decomposition compared to the OM_0 treatment, but not sufficiently to cause statistically significant increases in decomposition or in nutrient accumulation in buried stakes. The only exception was an increased Mg concentration observed in buried white oak stakes. The reason why only the Mg concentration increased in this treatment is unclear.

The lack of a compaction effect may partly be a result of the wood stake installation process, which may have alleviated some of the compaction as experienced by the wood stakes. The method used caused some soil disturbance, but probably no more than that caused by a coring tool used to create standardized holes for wood stakes described by Jurgensen et al. (2006). In addition, Shestak and Busse (2005) reported that despite increases in forest soil bulk density, variables such as pore-size distribution, water-holding capacity, gas diffusion, and microbial biomass indicators—all factors that could affect the mass loss of buried wood—were either unaffected by compaction or showed inconsistent responses. In a study of soil compaction in an agricultural field, Entry et al. (2006) reported a similar lack of response. The researchers found that significant increases in bulk density had no consistent effect on active bacterial or active fungal biomass either in the top 7.5 cm of soil or in the 15- to 20-cm depth of soil.

Table 3.—Soil water and temperature by treatment measured in 2000 (adapted from Page-Dumroese et al. 2006)

	OM_0	OM_2	C_0	C_2	VC_0	VC_1
Soil water (%)						
June	13.4	20.4	20.6	18.4	10.5	30.1
July	19.4	26.3	23.9	29.7	13.9	36.2
August	24.9	36.2	29.5	34.9	20.6	40.6
September	19.9	24.6	22.2	26.0	15.6	34.1
Mean	**19.4**	**26.9**	**24.1**	**27.3**	**15.2**	**35.3**
Soil temperature (°C)						
June	13.3	16.4	15.2	15.1	10.6	18.7
July	18.0	18.8	18.6	19.8	15.1	21.7
August	15.7	20.3	17.8	21.3	15.8	22.3
September	12.4	15.9	12.7	18.3	14.1	17.9
Mean	**14.9**	**17.9**	**16.1**	**18.6**	**13.9**	**20.2**

Although environmental factors were not measured during this wood stake decomposition study, another report from this study site (Page-Dumroese et al. 2006) showed that when this study began, the surface soil temperature in the VC_1 treatment was 6 °C warmer and the soil water content was 2.3 times greater (15 vs. 35 percent) than in the VC_0 treatment from June through September (Table 3). The greater temperature and soil water content of the VC_1 treatment was due to the lack of vegetation other than the planted seedlings to shade the ground and transpire soil water. Soil moisture and temperature are positively correlated to woody debris decomposition rates across a number of ecosystems (Risch et al. 2013). In our study region, soil moisture becomes limiting to tree growth during dry periods in midsummer because of high evaporative demand and the low water-holding capacity of the soils (Jenkins and Pallardy 1995). It is also reasonable to assume that a lack of soil moisture is limiting to microbial activity and consequently to wood decay during those dry periods regardless of soil temperature. We could not determine, however, whether temperature or moisture was the most important factor causing the decomposition differences between VC_0 and VC_1 treatments.

We expected the faster decomposition and the shorter half-life of northern red oak stakes compared to white oak stakes. The heartwood of white oak group species has long been known to resist decay much more effectively than that of red oak group species (Carteret al. 1976, Scheffer et al. 1949). As decomposition progressed, nutrient concentrations in the wood increased for both species compared to reference stakes. Decaying wood in soil has been shown to induce nutrient deficiencies in plants by stimulating microbial growth, which depletes the surrounding soil of some available nutrients. In the current study, however, correlations between soil nutrient concentrations and mass loss of wood were not statistically significant (data not presented). There was also no strong correlation between nutrient concentrations of buried wood stakes and nutrient concentrations of soil in the plots. Perhaps this is partially due to the relatively low nutrient status of the soil and the small differences in nutrient concentrations commonly reported for the Clarksville soils in the study area (King 1997).

Others have reported significant increases in nutrient concentrations within decomposing woody debris including N, P, and Ca (Johnson et al. 2014) and Ca and Mg (Smith et al. 2007), which suggests that woody debris plays an important role as a nutrient sink during early stages of decomposition. The duration and magnitude of nutrient immobilization are reportedly dependent on substrate quality (Bosatta and Berendse 1984) before a period of release or mineralization (Lousier and Parkinson 1978). Periods of nutrient accumulation longer than 24 months have been reported for forest litter (Weber 1987). Results from Blair (1988) and Kaczmarek et al. (1998) show that total P content of deciduous leaf litter tended to increase for at least 2 years, despite significant losses in litter mass. Idol et al. (2001) suggested that a delay can occur in N mineralization for dead and down wood for several decades depending on the age and decomposition class of the wood. Idol et al. (2003) also reported that for a chronosequence of upland hardwood stands in Indiana, N mineralization was greatest in the oldest stand (80-100 years old) and lowest in the next-oldest stand compared to stands ranging from 1 to 14 years since harvesting. They speculated that large inputs of logging debris, the mixing of fresh forest floor litter, and the death of coarse root systems in recently harvested stands led to increases in N immobilization because of the generally high C:N ratio of woody debris (Idol et al. 2003). During the accumulation period, the decomposing material is N poor and limiting to microbial activity, causing microorganisms to import nutrients from the immediate surroundings into the material (Blair 1988).

Despite apparent increased decomposition and enhanced nutrient immobilization, Ponder et al. (2012) demonstrated that the growth of planted tree seedlings in the VC_1 treatment was greater than that of planted seedlings in the VC_0 treatment at our study site and elsewhere where similar experiments were conducted. Foliar nutrient concentrations in planted seedlings were also higher in the VC_1 treatment (Ponder et al. 2012). Collectively, this suggests that if enhanced nutrient immobilization by decaying wood was occurring in this treatment that it was not sufficient to reduce tree growth during the first decade after planting, even in the nutrient-deficient sites such as those examined in this study.

CONCLUSION

The findings indicate that in the regenerating mixed-oak forests on these xeric and nutrient-poor sites, OM removal and compaction resulted in negligible changes in the mass loss and nutrient concentrations of buried wood stakes. VC, however, appears to enhance stake decomposition and increase nutrient concentrations within the stakes, at least initially. This suggests that extensive biomass removals during harvesting operations or incidental compaction associated with skidder activity would have little impact on woody debris decomposition dynamics in mineral soil on this and similar sites. Immobilization of N and possibly P and Ca may occur where VC is practiced as part of a regeneration strategy. The immobilization of nutrients in buried woody debris on these nutrient-poor Missouri Ozark sites may also serve to reduce nutrient loss to leaching. The slow release of immobilized nutrients in wood through decomposition and mineralization appears to be one mechanism for retaining nutrients on xeric, nutrient-deficient sites.

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FOLIAR NUTRIENT RESPONSES OF OAK SAPLINGS TO NITROGEN TREATMENTS ON ALKALINE SOILS WITHIN THE MISSOURI RIVER FLOODPLAIN

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Abstract.—Bottomland afforestation is frequently unsuccessful, partly because of low-quality planting stock and low soil fertility following row cropping. In autumn 1999, two 16.2-ha fields at two conservation areas in central Missouri were seeded to redtop grass or allowed to revegetate from the seedbank. In spring 2004, one of five nitrogen (N) treatments was applied to one row of five-row tree plots planted in 2000 with all combinations of bare-root and root production method (RPM) seedlings of swamp white oak and pin oak. In late July 2006, foliar N averaged 1.83 percent after annual application of 83 g of 20N-10P-10K as slow-release ammonium nitrate, 1.82 percent after annual application of 87 g of 19N-6P-9K as slow-release urea, 1.78 percent after planting two N-fixing false wild indigo seedlings adjacent to each oak sapling, 1.79 percent after planting two buttonbush seedlings, and 1.81 percent when left untreated. Foliar N averaged 1.70 percent for swamp white oak from bare-root planting stock, 1.80 percent for swamp white oak from RPM planting stock, and 1.88 percent for pin oak from RPM planting stock. Foliar N averaged 1.74 percent for oaks growing in cover of weeds and 1.81 percent for oaks planted in redtop at one site; no differences were caused by ground cover at the other site (1.83 percent). Compared with estimated sufficiency ranges, foliar phosphorus, potassium, calcium, magnesium, zinc, boron, and copper were adequate for both species. Foliar N, manganese, and iron were deficient for both species with only foliar sulfur deficient for pin oak. A high soil pH likely limited micronutrient availability, especially manganese, and negated any response to applied N. Soil pH will need to be neutralized and quality planting stock planted for bottomland restoration of hard-mast species on moderately alkaline soils within the lower Missouri River floodplains.

INTRODUCTION

Reforestation efforts by public land managers and private landowners within the lower Missouri River and upper Mississippi River region frequently result in failures (Patterson and Adams 2003, Schweitzer and Stanturf 1997, Stanturf et al. 2001). Reasons for the failures included the use of species that are poorly adapted to frequent flooding or wet site conditions, competition from light-seeded hardwoods, altered soil properties, and depletion of soil nutrients following row cropping (Allen et al. 2001, Schweitzer and Stanturf 1997, Stanturf et al. 2004). Soils in these floodplains are frequently alluvial deposits and soil pHs of 7.5-8.0 are common. High soil pHs have the potential to limit the availability of essential macronutrients and micronutrients, especially inorganic nitrogen (N), iron (Fe), manganese (Mn), boron (B), zinc (Zn), and possibly copper (Cu) (Mills and Jones 1996).

Nitrogen is the nutrient that most often limits tree growth in temperate ecosystems. Recommendations for increasing available soil N to tree crops on old-field sites have included applying synthetic fertilizers and incorporating N-fixing plants into the planting (Ponder 1997;

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Van Sambeek et al. 1986, 1989). Johnson (1980) found only two of five oak species had increased growth 8 years after incorporating 112 and 436 kg ha^{-1} of N and phosphorus (P), respectively, at the time of planting. Schlesinger and Williams (1984) reported that planting N-fixing shrubs with black walnut (*Juglans nigra* L.) improves tree growth on all but the best sites. Funk et al. (1979) found that black walnut planted within 1.7-2.4 m of autumn olive (*Eleagnus umbellata* Thunb.) had improved tree growth as early as 4 to 5 years after establishment.

The objectives of our study were to

1. Determine if differences in leaf morphology and nutrient concentrations remain because of site, ground cover, or fertilizer treatments for 7-year-old oak saplings established as bare-root and RPM (root production method, Forrest Keeling Nursery, Elsberry, MO)
2. Evaluate the effectiveness of applying slow-release synthetic N fertilizers or planting N-fixing woody legumes to improve foliar N of planted swamp white oak (*Quercus bicolor* Willd.) and pin oak (*Q. palustris* L.) saplings exhibiting N deficiency symptoms
3. Determine if the uptake of essential macronutrients and micronutrients by established pin or swamp white oak saplings is altered by the N treatments.

Slow-release fertilizer treatments were chosen to provide continuous sources of N that presumably most closely mimic biological sources of N within forest ecosystems.

MATERIALS AND METHODS

Our fertilization study began in spring 2004 on two sites within the Missouri River floodplain planted with pin and swamp white oak. We used RPM planting stock in fall 1999 and bare-root stock in spring 2000. Table 1 summarizes each site's location, flooding history, soil types and characterization, site preparation, and planting histories. Soils at the two study sites were remapped by soil survey staff following soil alterations caused by flooding during the 1990s. Soils at the Plowboy Bend site are currently mapped as Lowmo silt loam (coarse-silty, mixed, superactive, mesic Fluventic Hapludolls); Treloar-Sarpy-Kenmoor soils (sandy over loamy, mixed, superactive, calcareous, mesic Oxyaquic Udifluvents; mixed, mesic Typic Udipsamments; and sandy over clayey, mixed, superactive, calcareous, mesic Oxyaquic Udifluvents, respectively); and Buckney fine sandy loam (sandy, mixed, mesic Typic Hapludolls). Soils at the Smoky Waters site are currently mapped as Blencoe silty clay loam (clayey over loamy, smectitic over mixed, superactive, mesic Aquertic Hapludolls); Haynie silt loam (coarse-silty, mixed, superactive, calcareous, mesic, Mollic Udifluvents); and SansDessein silty clay (fine, smectitic, mesic Fluvaquentic Vertic Endoaquolls). The soils at both sites are characterized by high soil pHs (Plowboy Bend ~7.6; Smoky Waters ~7.9) resulting from alluvial deposits eroded from calcareous soils in the midwestern states. At both sites, two square 16.2-ha areas were tilled in fall 1999 with one field seeded to redtop grass (*Agrostis gigantea* L.) and the other field allowed to revegetate naturally from the seedbank. The latter was initially dominated by dense stands of lambsquarter (*Chenopodium album* L.) and rough pigweed (*Amaranthus retroflexus* L.), followed by Johnson grass (*Sorghum halepense* [L.] Pers.), and, most recently, tall tick trefoil (*Desmodium paniculatum* [L.] DC.).

All four 16.2-ha fields were laid out with forty-four 380-m-long rows spaced 9.1 m apart and oriented parallel with the Missouri River. Excluding four border rows, each field was subdivided into eight replications, each of which contained five rows. Rows within four replications were bedded with a levee plow to create 30- to 40-cm-high soil beds approximately 0.6 m wide at the top and 2.1 m wide at the base. Each replication was divided into 54.8-m-long plots to be

Table 1.—Characterization of the study sites at Plowboy Bend and Smoky Waters Conservation areas

	Plowboy Bend Conservation Area	Smoky Waters Conservation Area	Citation
County and location	Moniteau; Section 24 and 25, T47N, R14W	Cole; Section 1, T44N, R10W	Shaw et al. 2003
Latitude	38°45'5" N	38°35'9"N	Shaw et al. 2003
Longitude	92°24'17"W	91°58'3"W	Shaw et al. 2003
Flooding history (1998 to 2007)	100-year levee protected	Flooded in 2001 and 2002 for up to 3 weeks in June	Dey et al. 2006
Soil type	Lowmo silt loam, Treloar-Sarpy-Kenmoor soils, Buckney fine sandy loam	Blencoe silty clay loam, Haynie silt loam, SansDessein silty clay	Web Soil Survey; Kabrick et al. 2005
Texture classes (0–50 cm)	Sandy loam to silt loam	Silt loam to silty clay	Web Soil Survey; Kabrick et al. 2005
Clay (0–50 cm)	3 to 9%; 6 to 7%	16 to 40%; 21 to 33%	Shaw et al. 2003; Kabrick et al 2005
Soil pH (0–50 cm)	7.2 to 7.7; 7.4 to 7.7	7.7 to 8.0; 7.8 to 8.0	Shaw et al. 2003; Kabrick et al. 2005
Organic carbon	0.3 to 0.4%; 0.5 to 0.8%	0.7 to 1.3%; 1.2 to 1.5%	Shaw et al. 2003; Kabrick et al. 2005
CEC ($cmol_c$ kg^{-1})	5.1 to 10.3; 5.4 to 8.4	13.6 to 29.7; 21.2 to 27.3	Shaw et al. 2003; Kabrick et al. 2003
Bulk density (Mg m^{-3})	1.4 to 1.5	1.2 to 1.4	Kabrick et al. 2003
Soil sulfur (µg g^{-1})	140 to 210	Not determined	Plassmeyer 2008
Soil iron (mg g^{-1})	6.0 to 9.9	Not determined	Plassmeyer 2008
Soil manganese (µg g^{-1})	130 to 275	Not determined	Plassmeyer 2008
Soil zinc (µg g^{-1})	25 to 40	Not determined	Plassmeyer 2008
Site preparation	Cropped, offset disked in August 1999; mounded and seeded in September 1999	Cropped, sprayed August 1999; offset disked, mounded, and seeded in September 1999	Shaw et al. 2003
RPM stock planted	Late November 1999	Late November 1999	Kabrick et al. 2005
Bare-root stock planted	Spring 2000	Spring 2000	Shaw et al. 2003
Install weed barrier and fertilize seedlings	March 2000	March 2000	Kabrick et al. 2005
Average water content (wt wt^{-1}) (0–30 cm)	Redtop = 11%; No redtop = 15%	Redtop = 22%; No redtop = 27%	Shaw et al. 2003
Swamp white oak foliar nitrogen in July 2002	Redtop = 1.70%; No redtop = 1.68%	Redtop = 2.05%; No redtop = 2.11%	Kabrick et al. 2005
Mounded pin oak foliar nitrogen in July 2002	Redtop = 1.62%; No redtop = 1.61%	Redtop = 1.99%; No redtop = 2.03%	Kabrick et al. 2005

planted with 30 trees on a 9.1 m × 9.1 m spacing. Planting stock consisted of 1-0 bare-root seedlings; 11-L container-grown, 8-month-old RPM saplings; and 19-L container-grown, 22-month-old RPM saplings of both swamp white oak and pin oak. RPM was used to produce large containerized seedlings that have a dense, fibrous root system as a result of air-root pruning (Lovelace 1998, 2002; Walter et al. 2013). A single combination of the two species and the three planting stock types was randomly assigned to the first six plots within each replication. The RPM saplings were planted in late November 1999 and the bare-root seedlings in March 2000. For weed control, a 1.2-m^2 water-permeable mat of black landscape fabric (DeWitt, Sikeston, MO) was placed around each tree. All trees received approximately 30 g of a slow-release 33N-3P-6K fertilizer across the weed barrier mat in March 2000 with no additional fertilizer until spring 2004.

A fertilization study was superimposed on the original study in spring 2004 after most saplings exhibited leaf chlorosis with low foliar N concentrations (1.7 to 1.8 percent) during the second growing season (Kabrick et al. 2005). One of five treatments was randomly assigned to each row within mounded and nonmounded replications. Fertilizer treatments applied to each oak sapling within a row included applying 83 g of 20N-10P-10K as slow-release ammonium nitrate (Osmocote, Scotts-Sierra Horticultural Products Company, Marysville, OH) in the spring each year; applying 87 g of 19N-6P-9K as slow-release urea (T&N, Inc., Foristell, MO) in spring each year; planting two N-fixing false wild indigo (*Amorpha fruticosa* L.) seedlings; planting two buttonbush (*Cephalanthus occidentalis* L.) seedlings; and leaving an unfertilzed control. Fertilizers were broadcast across the weed barrier mat annually in spring 2004, 2005, and 2006. Wild false indigo and buttonbush seedlings were planted 0.6 m on either side of the oak seedlings adjacent to the weed barrier mat within the tree row.

To determine prefertilization foliar N values, an intact leaf was harvested in late July 2003 from both the middle and upper canopies of approximately ten trees within each plot within three bedded and three nonbedded replications from all four plantings. In late July 2006, leaves were again harvested from the middle and upper canopies of up to ten trees within each plot of the four bedded replications in all four plantings. Leaves in both 2003 and 2006 were dried in a forced-air oven at 60 °C for 48 h, weighed, and ground to pass a 1-mm mesh screen. The collected leaves from each plot in 2006 were counted, photographed, and scanned to determine leaf area before oven drying. Leaf samples were analyzed by the Arkansas Diagnostic Laboratory to determine N content in 2003 and 2006 by combustion on a LECO FP428 (AOAC International official method 990.03) and to determine other macronutrients and micronutrients using inductively coupled plasma atomic emission spectrometry of a nitric acid digest. Kabrick et al. (2005) had previously reported that neither the size of RPM planting stock nor the bedding showed treatment differences for oak growth or foliar N in any of the four fields, so leaves from the saplings planted as 11-L and 19-L RPM planting stock within the same replication were combined into one composite sample before chemical analyses.

Foliar leaf area, dry weight, specific leaf mass, and macronutrient and micronutrient concentrations were subjected to analysis of variance for a nested design with four blocks nested within two plantings and two different ground covers with three species-planting stock type combinations and five N treatments as subplots to determine effects of treatment variables and their interactions (Table 2). High mortality of the bare-root pin oak saplings in all four plantings negated current and future analyses of factorial combinations of species and planting stock (RPM or bare-root seedlings).

Table 2.—ANOVA probability of a significant F-value when planting stock types and fertilizer treatments were nested within planting site and ground cover

Variance Source	df	Foliar nitrogen (%)	Leaf area (cm^2)	Specific leaf mass (g cm^{-2})	Nitrogen specific leaf mass (g N cm^{-2})
Planting location (P)	1	0.036	0.109	<0.001	<0.001
Ground cover (C)	1	0.059	0.490	0.205	0.030
Error A = Blocks (P x C)	12	------	-------	-------	-------
Species stock type (S)	2	<0.001	<0.001	<0.001	<0.001
N fertilizer treatment (N)	4	0.296	0.097	0.589	0.499
S x N	8	0.716	0.857	0.833	0.904
P x S	2	0.059	0.354	0.178	0.093
P x T	4	0.996	0.013	0.539	0.589
P x S x T	8	0.780	0.182	0.352	0.735
C x S	2	0.011	0.385	0.016	<0.001
C x N	4	0.312	0.468	0.525	0.440
C x S x N	8	0.294	0.290	0.493	0.738
P x C x S	2	0.477	<0.001	0.326	0.253
P x C x N	4	0.092	0.806	0.006	0.143
P x C x S x N	8	0.583	0.600	0.066	0.096
Error B = Residual error mean squares	169	0.013854	104.576	696.292	518.078

RESULTS AND DISCUSSION

A significant planting × ground cover × species-stock type interaction was observed for nearly all variables analyzed. When this interaction was not found, a significant ground cover × species-stock type was frequently observed. Nearly all variables showed strong main effects between the two planting sites and among the three species-stock type combinations either when included or not found as part of these interactions.

Foliar Nitrogen Concentration

When established as traditional bare-root planting stock, 7-year-old saplings of swamp white oak had a greater average foliar N concentration at Plowboy Bend (1.79 percent) than at Smoky Waters (1.69 percent) (Table 3). After 7 years, redtop continues to dominate the ground cover in the two plantings seeded to redtop except in depressions at Smoky Waters that remained flooded for more than 3 weeks after the 2001 spring flood. No differences in foliar N (1.79 percent) were found for swamp white oak saplings from bare-root planting stock established in a ground cover of redtop or vegetation from the seedbank at Plowboy Bend. In contrast, swamp white oak saplings from bare-root planting stock established in redtop had greater foliar N (1.78 percent) than saplings grown with a succession of forbs from the seedbank (1.61%) at the Smoky Waters site. Dey et al. (2004, 2006) indicated that bare-root planting stock was less likely to be shaded by redtop grass than by a ground cover of mixed seedbank forbs. This may also have contributed to the greater foliar N in the bare-root stock planted into redtop.

Swamp white oak saplings established as RPM planting stock also had greater foliar N at Plowboy Bend (1.84 percent) than at Smoky Waters (1.75 percent). Unlike for saplings from bare-root planting stock, no differences in foliar N were found between RPM saplings growing in redtop or vegetation from the seedbank at either Plowboy Bend or at Smoky Waters.

Table 3.—Mean percent foliar nitrogen (N), leaf area, specific leaf mass (SLM), and N specific leaf mass (NSLM) after applying five N treatments for 3 years to 4-year-old saplings of swamp white oak or pin oak at two conservation areas with a ground cover of either seeded redtop grass or naturally regenerating forbs

Stock type, Species, and Location	Nitrogen treatment	Ground Cover								
		Redtop grass					Mixed forbs from seedbank			
		N	Area	SLM	NSLM		N	Area	SLM	NSLM
		%	cm^2	mg cm^{-2}	µg cm^{-2}		%	cm^2	mg cm^{-2}	µg cm^{-2}
SWAMP WHITE OAK FROM BARE-ROOT PLANTING STOCK										
Plowboy Bend	Check	1.82	69.0	11.5	210		1.65	71.4	12.3	204
	NH_4NO_3	1.83	58.6	12.4	227		1.82	61.0	12.0	219
	Urea	1.75	70.6	13.3	232		1.78	72.1	12.3	220
	False indigo	1.67	54.0	11.8	196		1.72	63.1	12.4	214
	Buttonbush	1.83	71.9	12.9	236		1.65	71.2	13.0	214
Smoky Waters	Check	1.69	58.0	11.1	188		1.65	63.1	10.4	171
	NH_4NO_3	1.76	70.9	10.8	191		1.68	67.4	10.0	168
	Urea	1.82	57.1	10.7	195		1.63	50.7	10.5	170
	False indigo	1.82	56.8	11.5	208		1.51	61.8	10.0	151
	Buttonbush	1.80	60.1	11.4	205		1.59	51.8	9.8	155
SWAMP WHITE OAK FROM RPM CONTAINER PLANTING STOCK										
Plowboy Bend	Check	1.88	83.6	11.9	222		1.80	70.7	12.9	232
	NH_4NO_3	1.86	72.3	12.4	230		1.83	70.2	13.1	239
	Urea	1.88	81.6	13.3	251		1.85	68.0	13.2	244
	False indigo	1.82	74.8	12.3	224		1.88	66.6	12.9	244
	Buttonbush	1.81	83.8	12.2	221		1.88	72.1	12.4	232
Smoky Waters	Check	1.81	77.4	10.7	193		1.74	75.1	11.6	201
	NH_4NO_3	1.74	58.6	11.8	204		1.77	88.1	11.1	196
	Urea	1.80	57.4	11.0	198		1.75	73.9	11.5	202
	False indigo	1.77	58.3	11.1	197		1.77	76.2	11.6	204
	Buttonbush	1.76	55.0	11.1	195		1.70	83.1	11.2	191
PIN OAK FROM RPM CONTAINER PLANTING STOCK										
Plowboy Bend	Check	1.90	17.6	10.2	192		1.97	16.1	10.2	201
	NH_4NO_3	1.89	19.0	10.2	193		1.85	18.2	10.6	196
	Urea	1.91	20.0	10.3	197		1.90	24.3	9.9	190
	False indigo	1.85	17.4	10.7	197		1.97	14.9	9.5	180
	Buttonbush	1.86	18.5	9.9	184		1.84	18.5	10.1	187
Smoky Waters	Check	1.83	15.0	9.7	178		1.93	18.5	7.5	145
	NH_4NO_3	2.00	8.0	10.5	210		1.97	17.0	7.4	145
	Urea	1.89	20.8	7.7	147		1.81	13.5	9.7	175
	False indigo	1.81	16.4	9.0	164		1.81	15.6	9.1	164
	Buttonbush	2.00	17.9	10.8	216		1.74	19.6	8.1	140
	Average 5% LSD =	0.17	14.3	1.4	32		0.17	14.3	1.4	32

Unlike with swamp white oak saplings, no differences were found for pin oak saplings established as RPM planting stock in average foliar N (1.89 percent) at the Smoky Waters and Plowboy Bend sites. Likewise no differences in foliar N were found for pin oaks of either planting stock type when established in redtop or successional vegetation from the seedbank. Van Sambeek and Garrett (2004) found that most grasses reduce hardwood growth and presumably foliar N concentrations more than a cover of broad-leafed forages or weeds. Our results confirm previous reports that redtop may not be as competitive as most other grasses and can be recommended over most grasses as a living mulch in tree plantings.

Color photographs of the leaves collected from representative plots of each treatment show wide ranges in leaf area, color, and amount of chlorosis (Fig. 1). Compared to foliar N concentration for oak saplings in the nontreated control rows (1.83 percent), annual application of 16.6 g of N for 3 years to each tree as slow-release ammonium nitrate or urea N failed to increase foliar

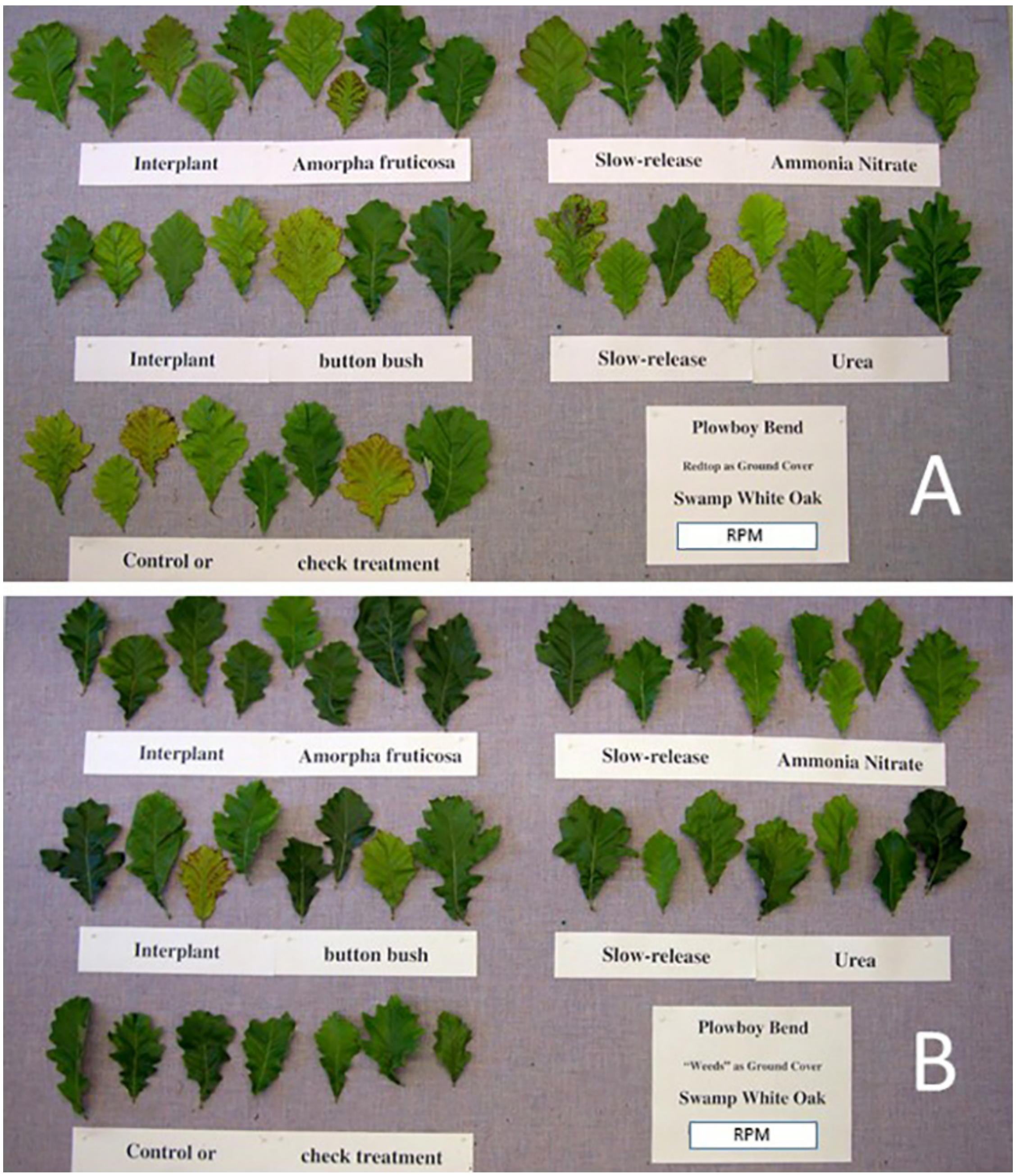

Figure 1.—Variation in leaf area and color of swamp white oak saplings from RPM planting stock growing at Plowboy Bend in (A) redtop grass or (B) seedbank vegetation 3 years after initiating five nitrogen fertilization treatments. Additional photos are available from the authors. Photographs by Jerry Van Sambeek, U.S. Forest Service.

N (1.82 percent). Our fertilization rates of 50 g N per sapling may have been too low. Johnson (1980) found only two of five oak species had increased growth 8 years after incorporating 112 kg ha^{-1} of N at the time of planting. Himelick et al. (1965) reported increased stem diameter growth of 7-year-old pin oak by applying 272 g of N as either ammonium nitrate or urea grown on nearly neutral silt loam soils.

Three years after planting two N-fixing false indigo seedlings adjacent to each oak sapling, foliar N in the oak saplings was no different than when two buttonbushes had been planted adjacent to the oak saplings (1.80 percent and 1.79 percent, respectively). There is good evidence that false indigo should have added N to the ecosystem. Navarrete et al. (2003) found that false wild indigo seedlings are readily nodulated by rhizobial bacteria in greenhouse studies. Plassmeyer (2008) and Van Sambeek et al. (2008) reported that foliar N for false wild indigo leaves averaged more than 3 percent in July. The high summer foliar N is a strong indication that false wild indigo had been nodulated by native rhizobia that can fix atmospheric N. Plassmeyer (2008) also found that foliar N of false wild indigo remained high through abscission (more than 2.5 percent), producing N-rich leaf litter.

Changes in the ground cover also suggest N in the soil is not readily available to any non-N-fixing plants. Mortality of the buttonbush has been high, but many of the false wild indigo have survived and are growing well (personal observation). In addition, *Desmodium* species now dominate the ground cover. Houx et al. (2009) found these native legumes are readily nodulated by rhizobial bacteria. The soils have not been tested to see if there are any changes in total and available soil N. We suspect the high soil pH may directly or indirectly suppress uptake of available soil N. Nannipieri et al. (1982) reported that soil ureases involved in the process for conversion of plant-bound N to available nitrate N can be strongly inhibited by high soil pH.

Leaf Area and Specific Leaf Mass

Increased allocation of N to leaves can result in changes to leaf morphology without changing percent foliar N (Takashima et al. 2004). As with foliar N, neither area nor dry weight per leaf increased in response to any of the treatments (Table 3). Leaf area showed a planting × ground cover × species-stock interaction primarily because swamp white oak from RPM planting stock at Plowboy Bend had a greater leaf area when grown in redtop (79 cm^2) than in seedbank vegetation (70 cm^2). In contrast, swamp white oak from RPM stock at Smoky Waters had a greater leaf area when grown in seedbank vegetation (79 cm^2) than when grown in redtop (62 cm^2). Pin oak averaged 19 cm^2 at Plowboy Bend and 16 cm^2 at Smoky Waters with no differences caused by ground cover.

Increased allocation of N into leaves could also result in changes in leaf thickness or specific leaf mass, with thicker leaves having higher photosynthetic rates. Although a planting × ground cover × species-stock interaction was not found (Table 2), specific leaf mass was greater at Plowboy Bend (11.7 mg cm^{-2}) than at Smoky Waters (10.4 mg cm^{-2}). At Plowboy Bend, specific leaf mass for N averaged 12.4 mg cm^{-2} for swamp white oak from bare-root planting stock and averaged 12.7 and 10.1 mg cm^{-2} for swamp white oak and pin oak from RPM planting stock, respectively; no differences were caused by ground cover. In contrast, at Smoky Waters, specific leaf mass of swamp white oak from bare-root stock averaged 10.1 and 11.1 mg cm^{-2} in seedbank vegetation and redtop, respectively, with no differences found between ground covers for swamp white oak from RPM planting stock. At Smoky Waters, specific leaf mass for pin oak was substantially lower when grown in seedbank vegetation (8.4 mg cm^{-2}) and when grown in redtop (9.5 mg cm^{-2}).

Nitrogen-Specific Leaf Mass

Sage and Pearcy (1987) found that maximum photosynthetic rates for C_3 plants are linearly correlated with leaf N concentrations. The leaf N concentrations average 0.15 to 0.2 μmol CO_2 m^{-2} s^{-1} for each μg N cm^{-2} increase, depending on leaf temperature (maximum photosynthetic rates range from 40 to 50 μmol CO_2 m^{-2} s^{-1} in full sun). Although a planting × ground cover × species-stock interaction was not found, specific leaf mass for N was greater at Plowboy Bend (215 μg cm^{-2}) than at Smoky Waters (184 μg cm^{-2}). At Plowboy Bend, specific leaf mass for N averaged 218 μg cm^{-2} for swamp white oak from bare-root planting stock and averaged 234 and 191 μg cm^{-2} for swamp white oak and pin oak from RPM planting stock, respectively, with no differences caused by ground cover (Table 3). In contrast, at Smoky Waters, specific leaf mass for N averaged 154 and 183 μg cm^{-2} for pin oak in seedbank vegetation and redtop, respectively. Specific leaf mass for N of swamp white oak from RPM planting stock averaged 198 μg cm^{-2} in both ground covers; specific leaf mass for N of swamp white oak from bare-root stock averaged 162 and 197 μg cm^{-2} in seedbank vegetation and redtop, respectively. This suggests that using large swamp white oak planting stock continued to show benefits for more than 7 years after planting.

Macronutrient and Micronutrient Concentrations

Foliar macronutrient and micronutrient concentrations before (summer 2003) and 3 years after the fertilization study began (summer 2006) were for the most part unchanged by the fertilizer treatments for swamp white oak or pin oak (Table 4). Although no differences were found between oak species in foliar potassium (K) and sulfur (S), swamp white oak foliar nutrients were greater in P, calcium (Ca), magnesium (Mg), Fe, Cu, and B than were pin oak foliar nutrients. In contrast, pin oak foliar nutrients were greater in Zn and Mn than for swamp white oak. In general, foliar nutrients for the oaks tended to be greater at Smoky Waters than Plowboy Bend except for N and Fe. Kabrick et al. (2005) reported that the cation exchange capacity and concentrations of K, Ca, and Mg in the soils at Smoky Waters are two to three times greater than at Plowboy Bend. The greater foliar Fe of the oaks at Plowboy Bend than at Smoky Waters is likely a consequence of the higher soil pH at Smoky Waters that are limiting Fe availability in the soil.

Examining the literature for acceptable foliar nutrient concentrations for pin and swamp white oak yielded few publications reporting sufficiency ranges at which healthy trees appear normal and capable of acceptable growth. When ranges for foliar nutrients are found, they can vary considerably both within species and section depending on collection time and growing environment. Sufficiency ranges for broad-leaved plants are also not very useful because oak foliar nutrients tend to deviate from the expected ratios of 1 N:1 K, 4 K:1 Ca, and 2 Ca:1 Mg for plants in general (Mills and Jones 1996).

Table 4 shows estimated sufficiency ranges for the red and black oak group (Section *Lobatae*) and the white oak group (Section *Quercus*) based on average values for reported minimum and maximum values of the different oak species (compiled primarily from Blinn and Bucker 1989, Gerloff et al. 1964, Kennedy 1993, Kennedy et al. 1986, Mills and Jones 1996, Mitchell 1936, Ponder 1993, Scherzer et al. 2003). These sufficiency values suggest the foliar nutrients of swamp white oak are deficient in N, Mn, and Fe, and the foliar nutrients of pin oak are deficient in N, S, Mn, and Fe. Mills and Jones (1996) indicate fertilization to correct for N deficiencies is unlikely to be successful if an element other than N is limiting plant growth. Soil pHs >7.5 limited availability of Mn and Fe in the soil and may in turn limit the ability of the trees to take up the applied ammonium nitrate, applied urea, or any fixed N added to the soil by the false wild indigo.

Table 4.—**Foliar macronutrient and micronutrient concentrations with standard deviations for swamp white oak and pin oak saplings before (July 2003) and 3 years after (July 2006) initiating a fertilizer study along with estimated sufficiency ranges for oak species within the *Quercus* and *Lobatae* Sections**

	Swamp white oak (*Quercus*)			Pin oak (*Lobatae*)		
	July 2003 foliage	July 2006 foliage	Section sufficiency ranges[a]	July 2003 foliage	July 2006 foliage	Section sufficiency ranges[a]
LEAF PROPERTIES:						
Foliage samples (#)	85 - 86	175 - 182	------	70 - 73	63 - 77	-----
Leaf dry mass (mg)	-----	802 ± 214	------	------	168 ± 45	-----
Leaf area (cm^2)	-----	68.6 ± 14.9	------	------	17.6 ± 4.3	-----
MACRONUTRIENTS:						
Nitrogen (mg/kg)	17.15 ± 1.53	17.67 ± 1.32	19.1 - 22.7	18.57 ± 1.51	18.83 ± 1.43	19.6 – 21.8
Phosphorus (mg/kg)	1.53 ± 0.19	1.60 ± 0.16	1.4 - 1.7	1.38 ± 0.16	1.52 ± 0.17	1.4 – 2.0
Potassium (mg/kg)	9.72 ± 1.32	8.64 ± 0.85	8.3 - 9.7	9.81 ± 1.16	8.78 ± 1.16	8.0 – 10.2
Calcium (mg/kg)	15.90 ± 4.64	21.28 ± 4.35	9.0 – 11.2	8.40 ± 1.20	11.25 ± 2.35	8.1 – 10.3
Magnesium (mg/kg)	3.10 ± 0.60	3.28 ± 0.66	1.7 - 2.3	2.13 ± 0.29	2.14 ± 0.31	2.0 - 2.6
Sulfur (mg/kg)	1.17 ± 0.09	1.27 ± 0.09	1.2 - 1.5	1.14 ± 0.12	1.23 ± 0.11	1.4 - 1.6
MICRONUTRIENTS:						
Manganese (µg/kg)	26.8 ± 14.7	35.1 ±12.9	320 - 900	69.2 ± 44.0	62.4 ± 24.0	600 - 1000
Iron (µg/kg)	45.4 ± 21.0	28.1 ± 9.9	75 – 100	45.2 ± 27.8	20.4 ± 6.7	70 – 110
Zinc (µg/kg)	23.2 ± 4.2	25.8 ± 8.9	15 - 25	43.0 ± 8.6	48.2 ± 12.4	30 - 45
Boron (µg/kg)	31.3 ± 7.9	47.8 ± 10.1	40 - 60	27.5 ± 7.1	32.9 ± 8.9	25 - 50
Copper (µg/kg)	10.5 ± 1.2	6.1 ± 1.2	5 - 8	9.3 ± 1.1	5.1 ± 1.2	6 – 10

[a] Sufficiency ranges are average values of the minimum and maximum values of the ranges reported in the literature for summer foliage of nine oak species within the *Quercus* section and 12 species within the *Lobatae* section. Values were compiled primarily from Blinn and Bucker (1989), Gerloff et al. (1964), Kennedy (1993), Kennedy et al. (1986), Mills and Jones (1996), Mitchell (1936), Ponder (1993), and Scherzer et al. (2003).

Ponder et al. (2008) found that the addition of ferrous sulfate with N fertilizers decreased soil pH slightly and increased foliar N and Mn concentrations of planted pin and swamp white oak saplings. The greater nutrient contents for the oaks established as RPM seedlings may in part be due to residual carryover of the available Mn and Fe in the potting medium.

CONCLUSIONS

Three years after initiating a fertilizer study to correct for deficiencies in foliar N of pin and swamp white oak, the trees had not yet responded with an increase in foliar N, leaf area, or leaf thickness. Van Sambeek et al. (1986) found forage legumes increased soil N within 1 to 3 years correlated with increased tree growth after three growing season. Oak saplings were found to also be deficient in foliar Mn and Fe. Soil pHs >7.5 limit the availability of these micronutrients in the soil and may be limiting the ability of the oak saplings to take up the applied ammonium nitrate, applied urea, or N fixed by false wild indigo or the invading *Desmodium* ground cover. The greater nutrient contents for the oaks established as RPM seedlings may in part be due to residual carryover of the available Mn and Fe in the potting medium.

The usual approach to neutralizing alkaline soils is application of elemental S or aluminum sulfate ions; however, this is not economically practical for large-scale use on the floodplains along the lower Missouri River. Application of N as ammonium sulfate or S-coated urea may gradually acidify the soil and correct the problem. A more practical approach may be the foliar application of N fertilizers supplemented with micronutrients (Lovatt 1999).

A more feasible approach to reforesting the Missouri River bottomlands (other than trying to ameliorate the soil condition) would be to choose hardwood species better adapted to alkaline soil such as bur oak (*Q. macrocarpa* Michx.), Shumard oak (*Q. shumardii* Buckl.), or the relatively slow growing chinkapin oak (*Q. muehlenbergii* Engelman) on higher elevations. Pin and swamp white oak were selected for this study because of their importance as a source of acorns for migratory waterfowl. Suggested pH ranges are 5.0 to 6.5 for pin and up to 7.0 for swamp white oak, although the published ranges for swamp white oak may need to be re-evaluated based on the acceptable growth of this species on the alkaline soils at one of the floodplain sites. Future plantings should evaluate other oaks found in the lower Mississippi and Missouri River floodplains such as northern selections of *Q. lyrata* Walt. (overcup), *Q. pogoda* Rufinesque-Schmaltz (cherrybark), and *Q. nuttallii* Palmer (nuttall) for survival, growth, and precocity in reforestation projects, especially when established using high-quality RPM planting stock. Successful reforestation projects will likely require a better mix of flood-tolerant oak species, higher-quality planting stock than bare-root, nursery-grown seedlings, and possibly foliar application of N fertilizers supplemented with micronutrients on these moderately alkaline soils within the lower Missouri River floodplains.

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ECOLOGICAL SITES: A USEFUL TOOL FOR LAND MANAGEMENT

Alicia N. Struckhoff, Douglas Wallace, and Fred Young[1]

Abstract.—Developing ecological sites in Missouri is a multiagency, multidiscipline effort led by the Missouri Department of Conservation and the U.S. Department of Agriculture (USDA) Natural Resources Conservation Service. The methodology developed in Missouri has recently served as a model for ecological site development across the country and has aided in an initiative to accelerate the development of provisional ecological site descriptions nationwide in 5 years. Provisional ecological site concepts have been developed for the entire state; eight ecological sites have received additional field verification and review. Provisional ecological sites contain sufficient information for use in conservation planning and land management. Ecological site products include map services and reference documents that are available through federal Websites. Ecological site information is already being used by the Missouri Department of Conservation, the Natural Resources Conservation Service, and additional partners to develop inventories, identify key communities, and recommend management actions.

INTRODUCTION

Ecological classification is the process of dividing the landscape into repeatable, unique, and discrete units. Ecological sites are the basic components of a terrestrial classification system that describes ecological potential and ecosystem dynamics of land areas. An ecological site is defined as "a distinctive kind of land based on recurring soil, landform, geological, and climate characteristics that differs from other kinds of land in its ability to produce distinctive kinds and amounts of vegetation and in its ability to respond similarly to management actions and natural disturbances" (U.S. Natural Resources Conservation Service [NRCS] 2013). Examples of ecological sites in Missouri include loess upland prairies, low-base chert protected backslope woodlands, and loamy floodplain forests.

Ecological sites serve as a framework for linking soils and landscapes to natural communities and ultimately to natural resource management. The sites can help drive resource planning and management at a variety of scales, from ecoregion to landscape to the field and woodlot. Ecological sites are tied to the USDA's biogeographic divisions of major land resource areas (Fig. 1). In Missouri, the NRCS recently required that private landowners who have enrolled in certain federal cost-share programs, such as the Environmental Quality Incentives Program, the Conservation Stewardship Program, and the Conservation Reserve Program, use ecological sites. The process of developing ecological sites generates opportunities to improve our understanding of natural communities in Missouri and refine the statewide soil database by supplementing it with ecological data (in addition to soil surveys) where needed.

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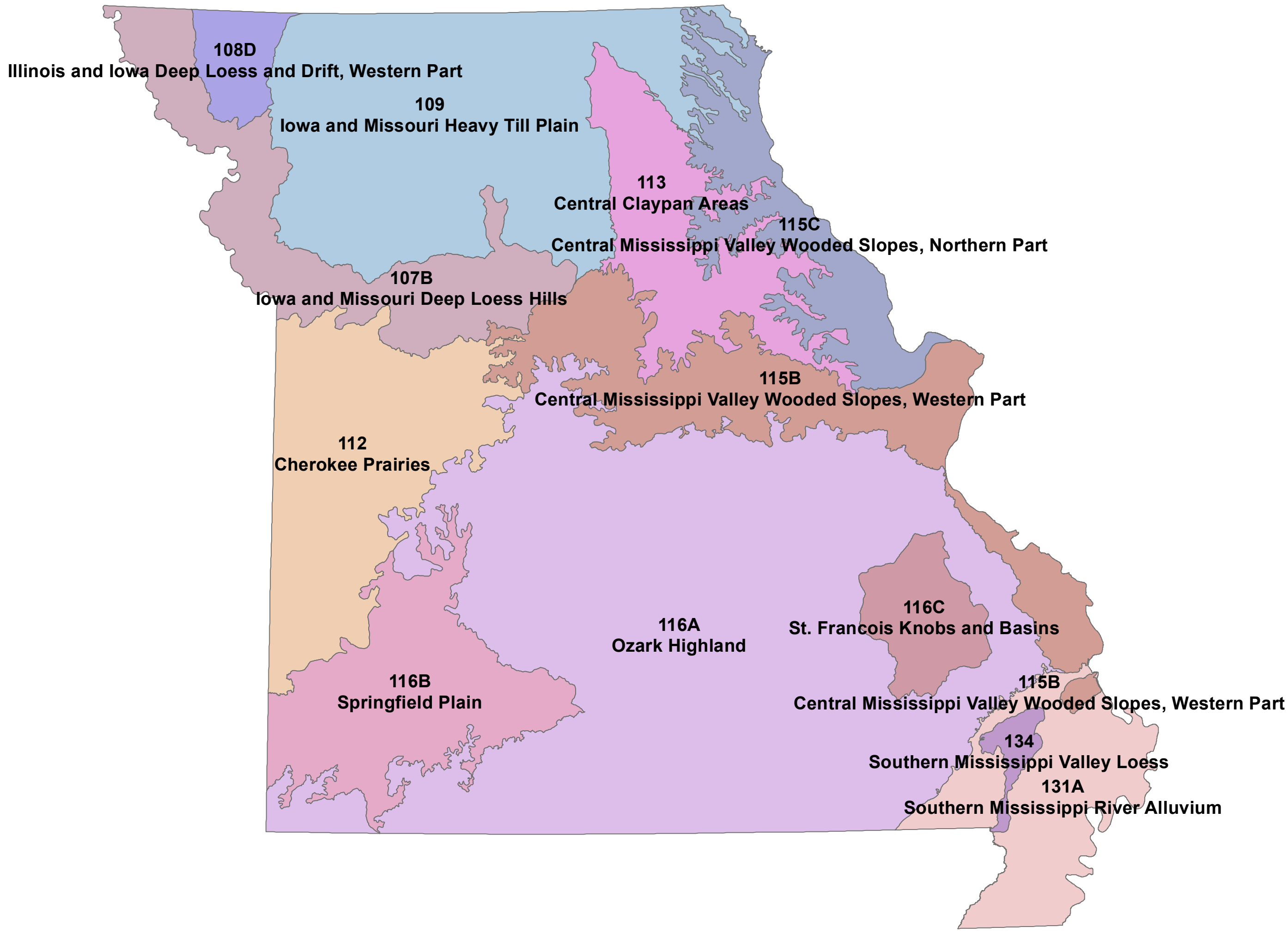


Figure 1.—U.S. Department of Agriculture biogeographic divisions of major land areas in Missouri.

METHODOLOGY

Ecological site development in Missouri is based on a framework of soil properties and select ecological factors. Statewide, eight essential soil and ecological factors have been identified that significantly influence vegetation and site productivity: landform, parent material, root restriction, base saturation, drainage, texture, flooding, and ponding. After delineation by physical properties, historical vegetation patterns or potential vegetation communities are correlated to each land unit, resulting in a provisional ecological site. Each ecological site receives a three-part name that includes soil or substrate plus landform plus vegetation community.

Every ecological site is field verified to strengthen and refine the concepts and to determine reference community locations. Field requirements for ecological site verification include a three-tiered data collection process, beginning with the most minimal requirement of field observations at the tier I level to the intermediate level tier II reconnaissance plot data to the ultimate requirement of highly detailed plot data collection. This comprehensive data collection results in species composition, canopy cover, biomass estimates, and ground cover information for the tier III level (NRCS 2014). The NRCS conducts several reviews that involve soil scientists and vegetation specialists on each ecological site before information is made available to the public. The first level of review results in a provisional ecological site and ultimately, tier III level data collection results in a correlated ecological site.

Provisional ecological site concepts have been developed for the entire state of Missouri; eight ecological sites have received additional field verification and review. With provisional ecological sites completed statewide, Missouri will be able to make more detailed data products available in a relatively short time frame. The methodology for ecological site development created in Missouri through the partnership of the Missouri Department of Conservation (MDC) and the NRCS has been used as an example to inform national ecological site development and promote the development of provisional ecological sites nationwide in 5 years.

PRODUCTS

With provisional ecological sites completed statewide (Fig. 2), a number of products are now available for land management, including the reference documents known as ecological site descriptions (ESDs). ESDs function as the primary repository of ecological knowledge about a given ecological site (Fig. 3). Although still in the provisional development stage, the Missouri ESDs contain sufficient information for use in conservation planning and land management, including the following sections: ecological site extent maps, physiographic features, landscape block diagrams, soil feature descriptions, ecological dynamics with state and transitional models, plant lists, site interpretations for wildlife and forestry, and references and definitions. The MDC, the Missouri office of NRCS, and additional partners are already using the ecological site data. Ecological site information can be used to develop inventories, identify key communities, and recommend management planning. For example, knowledge of an ecological site can guide restoration efforts by supplying species lists and associated canopy cover, basal area estimates, and information about historical disturbance regimes for community management.

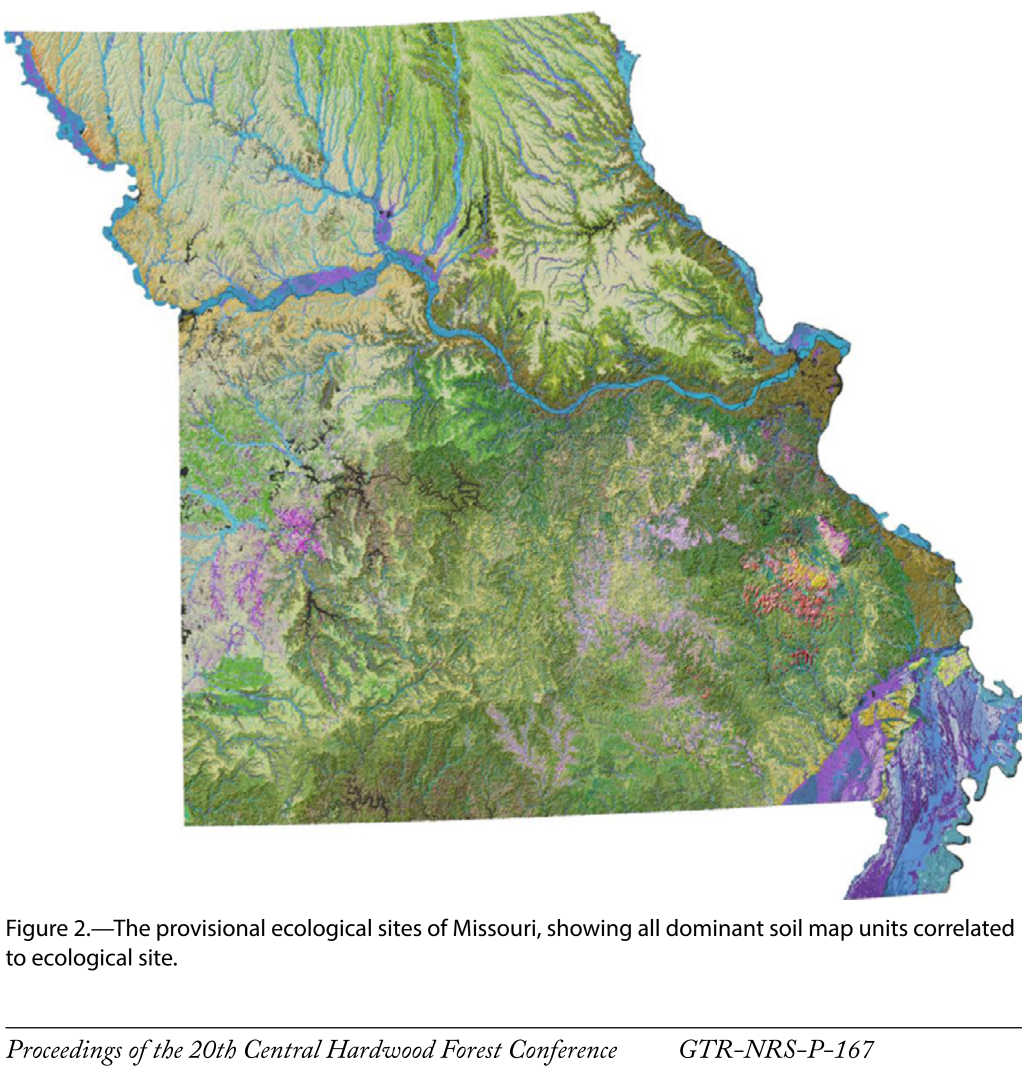
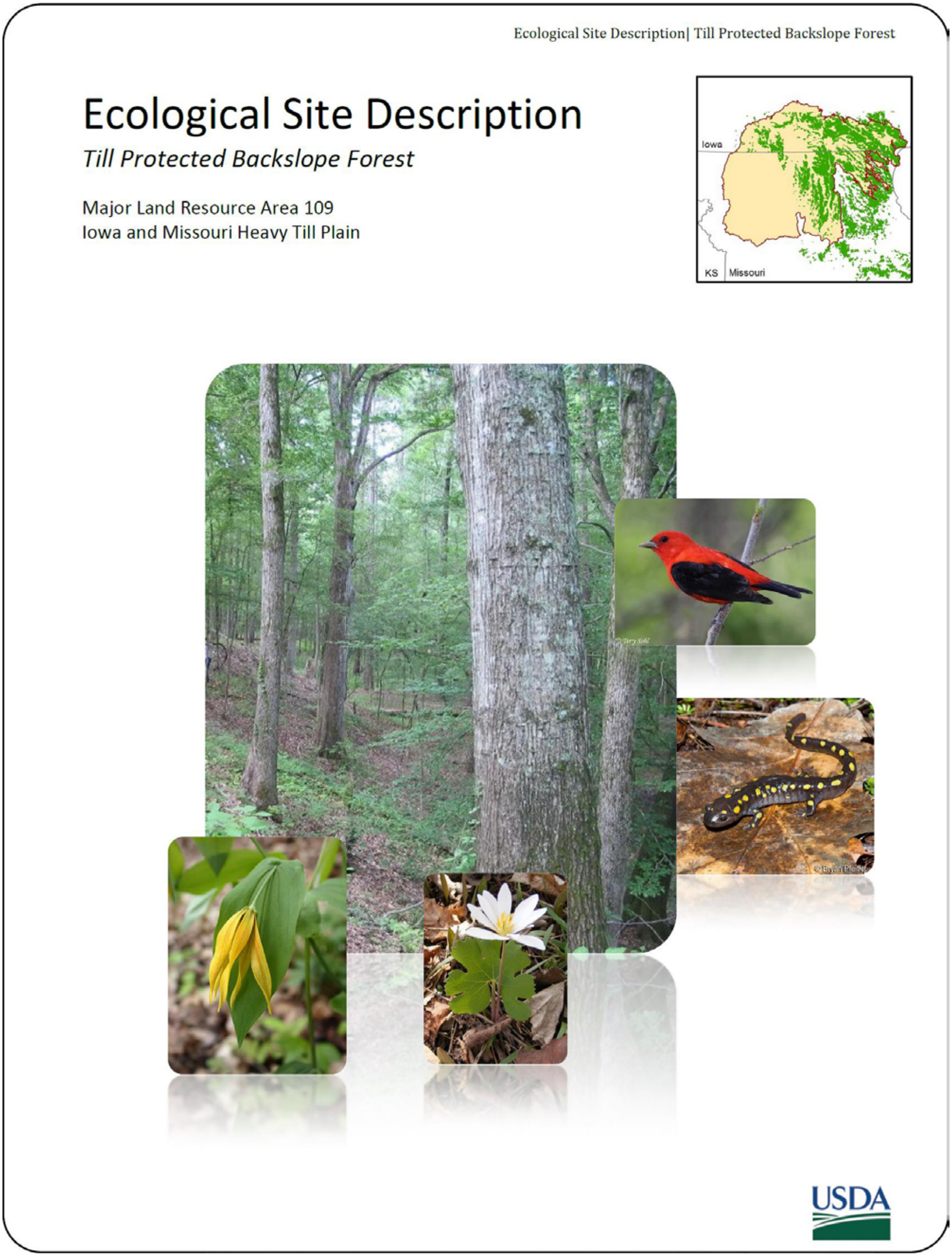


Figure 2.—The provisional ecological sites of Missouri, showing all dominant soil map units correlated to ecological site.



Ecological Site Description| Till Protected Backslope Forest

Ecological Site Description

Till Protected Backslope Forest

Major Land Resource Area 109
Iowa and Missouri Heavy Till Plain

Figure 3.—Example of an approved ecological site description (ESD) cover page. The ESD is the written reference document associated with each ecological site and acts as repository of all knowledge associated with that ESD.

Ecological site descriptions are maintained on the NRCS Ecological Site Information System Website (https://esis.sc.egov.usda.gov/), which is the national repository for information associated with ESDs and the collection of all site data. For Missouri, an additional Website location for provisional ESDs is the Missouri Field Office Technical Guide (NRCS, n.d. a), with ecological site information available in Section II of the Website (heading: Soil and site information). Ecological site information, including maps, is also available via the NRCS web soil survey (NRCS, n.d. b) using tabs marked "Soil Data Explorer" and "Ecological Site Assessment" options which can be accessed by creating an "Area of Interest" (AOI) after selecting "Start WSS".

SUMMARY

Ecological site development in Missouri is a multiagency team effort. The team creates products designed to drive resource planning and management at a variety of scales, from ecoregion to landscape to the field and woodlot. Ecological sites will allow us to understand how ecosystem attributes vary within and among regions and can be influential in developing sound management goals and objectives. Multiple disciplines such as silviculture, wildlife management, grassland planning, natural community management, ecosystem restoration, private lands conservation, and scientific research can all benefit from using ecological site information. The Missouri ecological site project now has ecological sites statewide, and numerous products are available for land managers.

ACKNOWLEDGMENTS

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REGENERATION II

EVALUATION OF SAPLING HEIGHT AND DENSITY AFTER CLEARCUTTING AND GROUP SELECTION IN THE MISSOURI OZARKS

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Abstract.—Silvicultural decisions often affect the development and characteristics of a stand. Silvicultural regeneration events can have immediate and gradual impacts on stand development. The objective of this study was to evaluate the effects of two silvicultural regeneration methods, clearcutting and group selection, on the composition of trees that are likely to recruit to the canopy within stands near the end of the regeneration period. We compared the mean density of saplings among 16-year-old clearcuts and 18-year-old group openings that were above height thresholds based on the 90th percentile of all sampled individuals on exposed or protected aspects within each treatment. The calculated percentiles provide a framework to determine whether the canopy that surrounds group selections causes a significant loss in productivity or desired species. We found significant treatment and aspect interaction ($p = 0.032$) for hickories and a significant treatment effect for red oaks ($p = 0.004$) and white oaks ($p < 0.001$). Our results provide valuable information about the effects of clearcutting and group selection on stand development.

INTRODUCTION

Historical ideologies for upland hardwood forest management have centered on the use of even-age silviculture systems (Roach and Gingrich 1968). In the Missouri Ozarks, these systems were largely successful at regenerating timber species such as oaks (Johnson 1993). More recently interest in multi-aged forests and management objectives has increased beyond commercial optimization to include wildlife habitat and aesthetics. This has led agencies such as the Missouri Department of Conservation (MDC) to enact policy changes that encourage canopy retention during regeneration, such as clearcutting with reserves or shifts to uneven-age practices of single-tree and group selection. Few studies have, however, contrasted the impacts of clearcutting and uneven-age management practices on species composition and density in the Missouri Ozarks.

Although oak regeneration is a complex process that has been a concern of land managers throughout the eastern United States, the evidence suggests that a wide variety of silvicultural practices can successfully regenerate oaks in the Missouri Ozarks because of the unique accumulation of oak seedlings (Dey et al. 1996, Liming and Johnston 1944). Oaks sprout vigorously after top-kill, contributing to the success of oak regeneration with the use of even-age silvicultural systems such as clearcutting. In contrast, Pioneer Forest in southeastern Missouri has used modified single-tree selection for uneven-age management with evidence of successful oak regeneration through 40 years of management (Loewenstein et al. 2000). The capacity of oaks to sprout and grow quickly is reduced, however, with increased overstory retention. An experiment examining the survivability and growth of oak stump sprouts in the Missouri Ozarks

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determined that single-tree harvesting resulted in significantly greater stump sprout mortality than group selection and clearcutting (Dey et al. 2008). Subsequent effects of harvesting practices on regeneration dynamics and stand development are not well understood. Site conditions also affect regeneration outcomes after harvest. Variable solar radiance caused by site aspect can be a contributing factor in regeneration success of oaks (Collins and Carson 2004). Thus, the effects of silvicultural practices on regeneration may vary by aspect.

The objective of this paper is to compare the density (stems per hectare) of the regenerating cohort after clearcutting and group selection in the Missouri Ozark Highlands. Our analyses focused on the tallest 10 percent of trees in both treatments to highlight individuals that are most likely to achieve canopy dominance. The effectiveness of each regeneration method is evaluated across protected and exposed aspects for six prevalent species groups, shortleaf pine, and the combination of all species present.

METHODS

Data were collected from the Missouri Ozark Forest Ecosystem Project (MOFEP), which encompasses more than 3,700 ha in the Current River watershed in Carter, Reynolds, and Shannon counties of southeastern Missouri (Kabrick et al. 2000). MOFEP is a long-term, landscape-scale experiment that was designed as a randomized complete block by MDC in 1989 to evaluate the effects of forest management on ecosystem composition, structure, and function (Shifley and Brookshire 2000).

The MOFEP study design includes three blocks, each with site replicates of even-age management, uneven-age management, and no management treatments. The even-age regeneration method (clearcutting with reserves) and uneven-age regeneration methods of single-tree selection and group selection were applied in 1996. The size of the group openings varied by aspect, with a diameter of one standard tree height (21.3 m) on exposed slopes and two standard tree heights (42.6 m) on protected slopes. The variation in harvest size was designed to allow one-third full sunlight at plot center for maximum photosynthesis of oak saplings (Dey et al. 2010, Law and Lorimer 1989).

The original MOFEP study design included 648 circular, 0.2-ha plots in a nested design for sampling vegetation composition and structure (Jensen 2000). The species, height, and diameter at breast height (d.b.h.) of each tree greater than 11.43 cm d.b.h. were recorded in each plot. In addition, four 0.02-ha plots were nested within each plot and the species, height, and d.b.h. were recorded for all trees, shrubs, and woody vines with d.b.h. measurements greater than 3.81 cm and less than 11.43 cm (Jensen 2000). Centered on each 0.02-ha subplot, one 0.004-ha circular plot was nested to collect species, height, and d.b.h. of trees, shrubs, and woody vines taller than 1 m and with a d.b.h. less than 3.81 cm (Jensen 2000). Clearcut plots were last sampled in 2012 (age 16).

The original MOFEP plots were randomly located (with stratification, see Shifley and Brookshire 2000) before the various harvest treatments were applied, and the proportion of plots that were located within a group opening was low. Moreover, the proportion of a plot that was actually within the opening varied among those plots that did occupy group-selection openings. Therefore, a more explicit sample of group-selection openings within the MOFEP framework was commissioned. We randomly selected 74 group selections. Within each, we established a similar sampling design of 0.004-ha plots nested within 0.02-ha plots. Data for this study were recorded in 2014 (age 18) and included the aspect of each plot and the species, height, and d.b.h. of all individuals taller than 1 m and with a d.b.h. greater than 3.81 cm. We used data from the

74 group selection 0.02-ha subplots along with data from 18 clearcut 0.02-ha subplots that included the same measurements collected in 2012 (age 16).

The 47 species encountered in clearcuts and the 36 species in group selections were condensed into six species groups: hickories (e.g., *Carya tomentosa* and *C. glabra*); maples (*Acer rubrum and A. saccharum*); mixed hardwoods (e.g., *Juglans nigra, Prunus serotina,* and *Ulmus rubra*); red oaks (e.g., *Quercus coccinea, Q. rubra,* and *Q. velutina*); white oaks (e.g., *Q. alba, Q. muehlenbergii,* and *Q. stellata*), and others (e.g., *Rhus copallina* and *Vaccinium arboreum*). We created an additional category for shortleaf pine (*Pinus echinata)*. These species groups are derived from the original MOFEP establishment report (Brookshire and Dey 2000). Because the clearcut and group selection plots were sampled in different years (ages 16 and 18, respectively), we standardized tree height at 16 years after harvesting for each treatment. To accomplish this, we first calculated annual height increment as each individual's height at sampling divided by the number of years since harvest and then multiplied the annual height increment by 16.

We selected the 90th percentile height as a threshold indicator of the most competitive trees in the regenerating cohort to allow comparison of densities among the regeneration treatments that would not be possible if we focused our analyses on a fixed number of trees per hectare. Within each treatment we used all sampled individuals to determine the 90th percentile heights for clearcuts on exposed (90th percentile = 7.406 m) and protected (90th percentile = 8.189 m) sites and for group selections on exposed (90th percentile = 7.707 m) and protected (90th percentile = 8.631 m) sites. Clearcut 90th percentile thresholds were also applied to group selection data to directly ascertain the differences in regeneration density, composition, and heights of individuals of the two treatments.

We calculated the average number of trees per hectare above the 90th percentile height values at the plot level for each species group. To determine treatment effects, we calculated the following response variables:

1. Clearcut^{CC90}: average number of trees per hectare in the clearcuts that met the 90th percentile height threshold calculated from clearcut plots
2. Group^{CC90}: average number of trees per hectare in group selections that met the 90th percentile height threshold calculated from clearcut plots
3. Group^{GP90}: average number of trees per hectare in group selections that met the 90th percentile height threshold calculated from the group selection plots

Each response variable was calculated using data from the exposed and protected sites separately. We calculated species group percent composition for aspect within clearcuts and group openings and for all treatments and aspects above the 90th percentile. We examined the fixed effects of harvest treatment, aspect, and the interaction of treatment and aspect using randomized complete block analysis of variance, using site block as a random effect. We determined statistical significance with $\alpha = 0.05$ and conducted analyses using SAS 9.3 (SAS Institute, Cary, NC).

Table 1.—Mean and maximum heights for all sampled individuals

	Clearcut				Group selection harvest			
	Exposed		Protected		Exposed		Protected	
Species group	Mean	Maximum	Mean	Maximum	Mean	Maximum	Mean	Maximum
	meters							
Hickories	8.4	10.3	10.3	13.2	12.1	17.7	12.4	22.9
Maples	8.2	9.6	9.9	12.6	9.6	10.6	10.9	15.0
Mixed hardwoods	8.4	11.1	9.3	12.2	10.5	15.4	11.5	15.9
Red oaks	9.9	17.1	10.4	14.3	10.3	14.4	10.9	13.6
Shortleaf pine	8.7	9.5	0.0	0.0	9.7	12.4	0.0	0.0
White oaks	8.8	18.3	9.9	13.9	9.4	13.0	10.7	18.1
Others	7.6	7.6	9.4	10.1	0.0	0.0	0.0	0.0

RESULTS

The tallest mean heights for each species group, excluding others, were within group selections. Because of the absence of individuals on protected slopes, shortleaf pine had a greater mean height within exposed group selections than exposed clearcuts (Table 1). Within clearcuts, red oaks had the tallest mean height (10.4 m) followed by hickories (10.3 m) and white oaks (9.9 m) (Table 1). Hickories (12.4 m) and mixed hardwoods (11.5 m) had the tallest mean heights within group selections (Table 1). The tallest oaks were found within exposed clearcuts, and hickories, maples, and mixed hardwoods had higher maximum heights in group selection harvests.

Mixed hardwoods were the most common species group of both treatments and aspects when all individuals were considered (>31 percent) (Fig. 1A). White oaks were the second most abundant (>17 percent) aside from maples within the protected aspect of clearcuts (23 percent) (Fig. 1A). The others species group were comparable to red oaks for all sampled individuals (Fig. 1A), but within the 90th percentile, red oaks contribute a much greater percentage to the overall composition (Fig. 1B). Above the 90th percentile, the red oak group is more common within clearcuts than group selections, and red oaks are more common on the exposed aspect of each treatment than on the protected aspect (Fig. 1B). The white oak group makes up at least 22 percent of all stems across all aspects and treatments above the 90th percentile. Shortleaf pines are found only on the exposed aspect of each treatment, and the greatest abundance of shortleaf pine is within Group^{GP90} (Fig. 1B).

When analyzed across all species, we found significant treatment ($p < 0.001$) and aspect ($p = 0.048$) effects on the number of trees per hectare taller than the 90th percentile. Clearcuts^{CC90} resulted in significantly more trees per hectare than Group^{CC90} and Group^{GP90} (Fig. 2A). There was no significant interaction between aspect and treatment ($p = 0.114$) when all species were included. Hickories were the only species group to exhibit a significant interaction between treatment and aspect ($p = 0.032$) (Fig. 2B). Aspect had a significant effect on density for maples ($p = 0.014$) and mixed hardwoods ($p = 0.001$) (Fig. 2C and Fig. 2D), and there was a significant treatment effect for red oaks ($p < 0.001$) (Fig. 2E) and white oaks ($p < 0.001$) (Fig. 2G). Shortleaf pine and others exhibited no significance in regards to treatment, aspect, or the interaction of treatment and aspect (Fig. 2F and Fig. 2H).

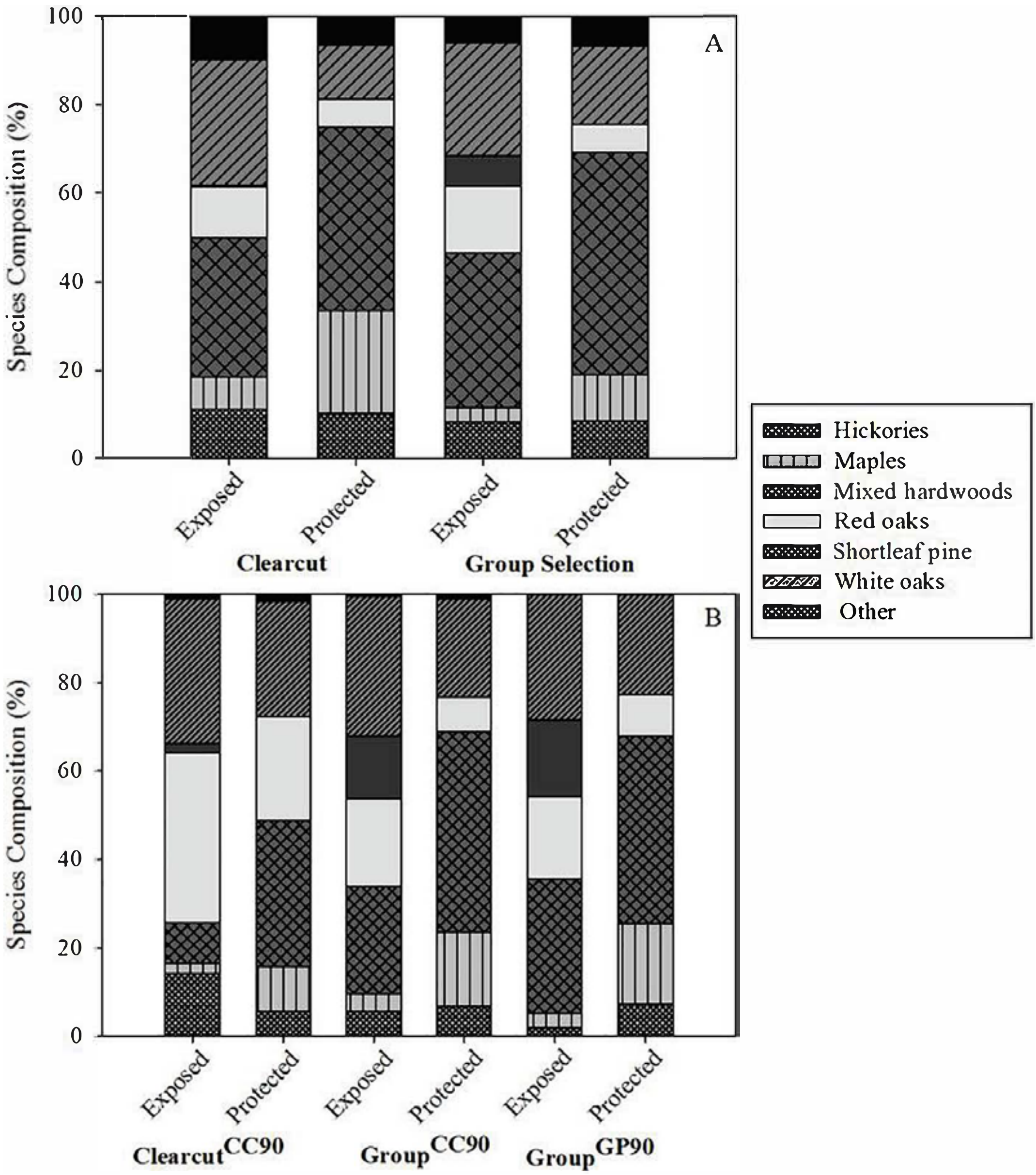

Figure 1.—Species composition (percentage of all stems present) for (A) exposed and protected sties by treatment using all trees present, and (B) exposed and protected sites of all individuals greater than the 90th percentile threshold used in this study.

Clearcutting produced a lower 90th percentile height threshold on both exposed (7.39 m) and protected (8.12 m) sites compared to the group selection harvests on exposed (8.67 m) and protected (9.71 m) sites, and this resulted in greater densities within clearcuts for all species aside from shortleaf pine (Fig. 2B-2H). Group^{CC90} had greater densities for all species compared to Group^{GP90}, and Group^{CC90} had densities greater than clearcut^{CC90} for maples and mixed hardwoods (Fig. 2B-2H).

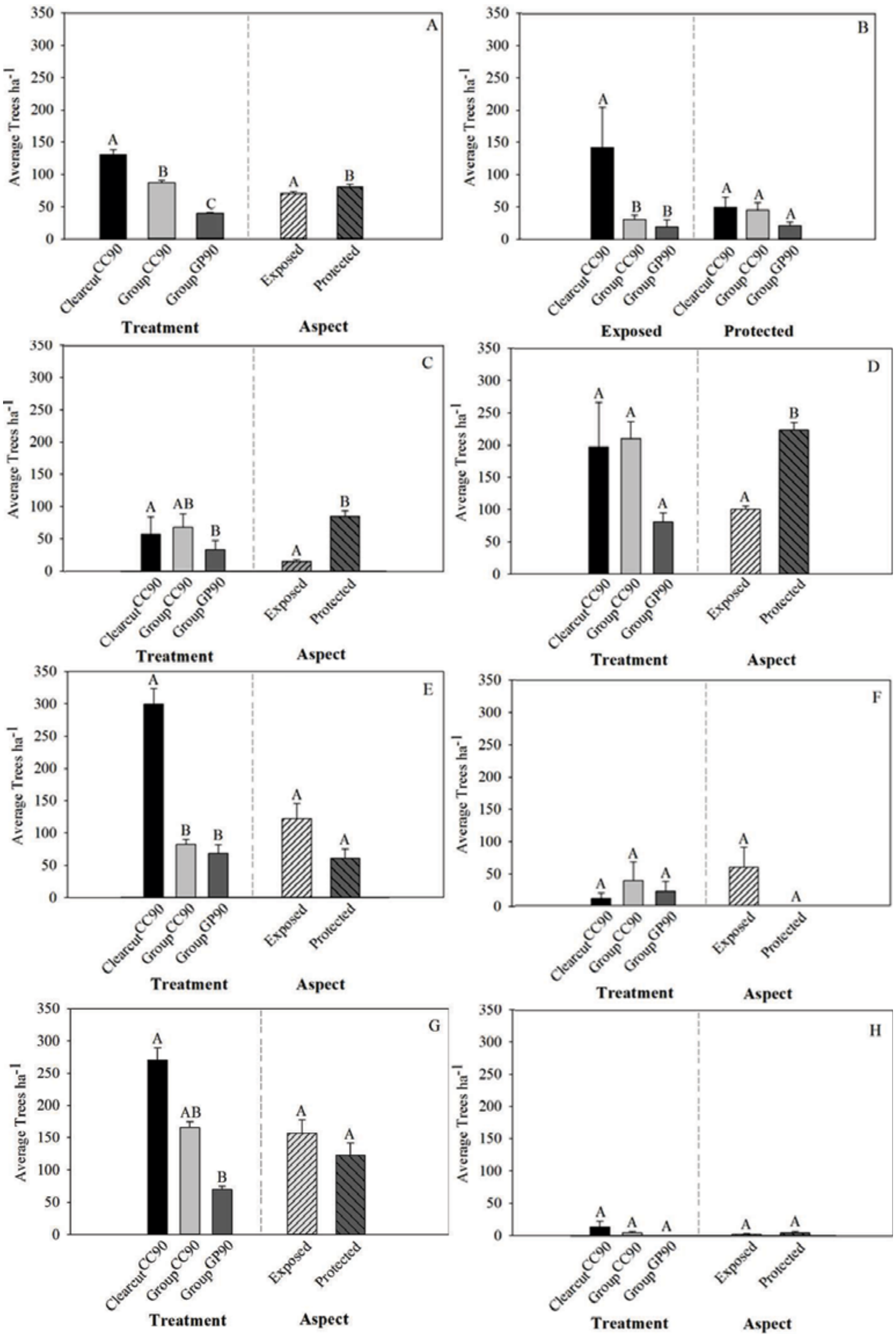

Figure 2.—Average trees per hectare above 90th percentile on protected and exposed aspects for all species and groups: (A) total; (B) hickories; (C) maples; (D) mixed hardwoods; (E) red oaks; (F) shortleaf pine; (G) white oaks; (H) others. Threshold^{CC90} exposed = 7.4066 and protected = 7.707 m. Threshold^{GP90} exposed = 8.189 m and protected = 8.6314 m. The significance (α = 0.05) denoted within graph B indicates significance within each aspect. Within treatment and aspect classes, significantly different (α = 0.05) densities are denoted by differing letters above error bars.

DISCUSSION

Even-age management has been recommended for regenerating oaks on upland sites in the Central Hardwood Forest region (Roach and Gingrich 1968), and the results of our study indicate that clearcutting resulted in the greatest density of oaks that are most likely to achieve canopy dominance. Species diversity was greatest within clearcuts, but the population of the tallest trees was dominated by fewer species than in group openings. The reduction in light levels reaching the regeneration in group openings is a likely factor in the reduction of oak species density. Species composition as a result of group selection did not vary greatly from clearcuts, however. The most notable difference between treatments was the decrease in density for almost every species group in group openings.

The 90^{th} percentile thresholds for heights were similar for each aspect, but group selection thresholds were higher than clearcuts on both aspects. Collins and Carson (2004) found that *Q. alba* showed significant differences in density across aspects, but our white oak species group did not exhibit the same trend. Density of stems in the red oak and white oak species groups did not differ between aspects of the same treatment. The mean heights of all species except shortleaf pine and others were highest in protected group openings. The greater quantity of small trees in clearcuts resulted in higher mean heights in group selections for all species groups except the others group. The elevated 90^{th} percentile thresholds of the group selections were possibly a result of having fewer small trees than in the clearcuts to skew the distribution, which perhaps accelerated the treated area through stand development. If a greater quantity of small trees is the cause, the existence of these smaller trees could be a result of variable budburst phenology, but additional analysis will be required to determine if this is a source of variability (see Hunter and Lechowicz 1992). In addition, competition within the shaded group selections could be the cause of more growth resources attributed to stem elongation.

Our analyses did not take into account sapling locations within the groups and examined only the averages of each plot sampled. Regenerating oaks near plot center may have received the optimum sunlight for growth (Minckler et al. 1973) and were capable of performing similarly to oaks within the clearcuts, but oaks closer to the group edge were subject to greater shading from the mature canopy. Examining the growth and location of trees within the group selections would yield valuable information on the success of group selection in promoting individuals across the selection as well as on the utility of additional treatments such as thinning the forest matrix that surrounds a group selection (Lhotka and Stringer 2013).

Group openings were not placed or shaped selectively to encourage any one species group to dominate. The effect of the treatment may increase with deliberate placement and shape and could be further enhanced by cultural practices such as thinning to accomplish management objectives (Miller et al. 1995). The evaluation of group selection success could have been improved by pretreatment sampling to compare species composition before and after management actions.

CONCLUSION

The Missouri Ozarks are well suited for oak regeneration, and the structure and composition of the regenerating stand depend largely on the composition of the mature canopy before a regeneration event (Dey et al. 1996). Our findings suggest that group selection could be a viable alternative to clearcutting despite significantly different densities after clearcutting and group selection. Although even-age management resulted in the greatest number of stems per hectare of oak species, oaks also dominated the recruiting cohort in group openings. Consideration of management objectives such as aesthetics, economics, and specific wildlife requirements may discourage clearcutting, and group selection harvest appears to be a viable alternative for forest management to succeed in meeting the goals of myriad stakeholders.

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MIDSTORY SHELTERWOOD TO PROMOTE NATURAL *QUERCUS* REPRODUCTION ON THE MID-CUMBERLAND PLATEAU, ALABAMA: STATUS 4 YEARS AFTER FINAL HARVEST

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Abstract.—In 2001, we initiated a study in Jackson County, AL, to examine shelterwood prescriptions in mixed mesophytic upland hardwood forests located on the escarpment of the mid-Cumberland Plateau. We were particularly interested in testing a shelterwood method that was successfully applied in other upland hardwood systems to recruit *Quercus* into competitive size classes. In the midstory herbicide shelterwood (MSW) we used an herbicide to inject more than 380 stems per acre (SPA) in three stands; average diameter of killed trees was 3.5 inches diameter at breast height (d.b.h.). After 8 growing seasons, all merchantable trees were removed in a commercial harvest. Control stands were left untreated for 8 years and then underwent the same final harvest. Before treatment, densities of trees ≥1.5 inches d.b.h. were 320 SPA for all species and included 27 SPA of *Quercus* spp., 67 SPA of *Acer saccharum* Marsh., and 3 SPA of *Liriodendron tulipifera* L. The MSW reduced the total SPA to 117. *Quercus* and *L. tulipifera* SPA were unchanged compared to pretreatment densities, and *A. saccharum* was reduced to 13 SPA. The MSW also increased the amount of full sun reaching the seedling layer for 3 years compared to the control. After the final harvest, there were only 19 SPA in the MSW, with no stems of *Quercus*, *A. saccharum*, or *L. tulipifera*. Control stands after final harvest had 104 SPA, with no stems of *Quercus*. The reproduction cohort in both control and MSW treatments shifted from 5 percent of stems in the smallest size class to 56 percent in the largest size class 4 years after the final harvest. Large *Quercus* seedling stems increased by 33 percent in the MSW and by 9 percent in the control. Survival and growth of tagged *Quercus* seedlings did not differ at any time between treatments. Competitor species of *L. tulipifera*, *A. saccharum*, *Fraxinus* spp., *Viburnum* spp., and *Cercis canadensis* L. may need additional treatment to maintain *Quercus* in these stands.

INTRODUCTION

Contemporary upland hardwood forests in the Tennessee Valley of northern Alabama and adjacent regions contain a mixture of species with wide ranges of shade tolerance and growth rates. Failure of *Quercus* to germinate and recruit into sapling size classes and the concurrent shift in dominance by mesophytic species remains a concern here as in other eastern forests (Nowacki and Abrams 2008). Manipulating light levels by reducing overstory and/or midstory stem densities is commonly recommended to promote oak over its competitors (Brose et al. 2008; Loftis 1990a, 1990b; Parker and Dey 2008; Schweitzer and Dey 2011). The disturbance severity and regime needed to accomplish this remain unknown because prescriptions need to be site-specific. High-severity disturbances such as clearcutting may result in a conversion of stands to *Liriodendron tulipifera* L. (Beck and Hooper 1986, Groninger and Long 2008, Jenkins and Parker 1998, Loftis 1990b). Intermediate-severity density reductions via shelterwood prescriptions have been tested as a means to alter light to favor *Quercus* over non-*Quercus* species

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(Johnson et al. 2009, Loftis 1990a, Sander 1979, Schlesinger et al. 1993, Schweitzer and Dey 2011, Spetich et al. 2002).

For decades those interested in forest management that facilitates the maintenance of *Quercus* in eastern hardwood forests have tried a variety of silviculture treatments. An initial promising prescription reported by Loftis (1990a) was a two-stage shelterwood in which the first stage involved removing the midstory, commonly implemented via herbicide injection of midstory stems. The premise behind this treatment was that it would create light levels on the forest floor that would enhance the growth and competitive position of small *Quercus* advance reproduction, while keeping competitive species that require higher light levels in check. Often the primary competitors are shade-intolerant species such as *L. tulipifera* and mid-tolerant species such as *Acer rubrum* L. On mesic side slopes, or the escarpment of the mid-Cumberland Plateau, *Quercus* reproduction competes with shade-intolerant species and with shade-tolerant *A. saccharum* Marsh. Positive results of this two-stage shelterwood were predicated on an assessment of the competitive position and density of *Quercus* reproduction following the midstory treatment (Craig et al. 2014, Janzen and Hodges 1987, Lockhart et al. 2000, Loftis 1990a, Miller et al. 2004, Parrott et al. 2012). No studies have reported on the results obtained from treatments applied at a stand level, or on results after phase II, which is a complete removal of the overstory.

We initiated a study in mature upland hardwood forests to evaluate the effectiveness of the midstory herbicide shelterwood prescription to promote the development of advanced *Quercus* reproduction. Our primary objective was to evaluate the longer-term efficacy of a two-phase shelterwood (herbicide midstory followed by overstory removal) applied at the stand level by monitoring stand regeneration and development after both phases. We compared the species composition and stand structure responses to control stands that were left untreated when phase I of the shelterwood treatment was implemented but were clearcut concurrent with removing the overstory in the shelterwood treatment. A secondary objective included characterizing the light environment at the seedling stratum under the two regeneration treatments.

METHODS

Study Site Description

This study was conducted in mature upland hardwood forested sites located at the southern end of the mid-Cumberland Plateau in northeastern Jackson County, AL, within the Cumberland Plateau section of the Appalachian Plateaus physiographic province (Fenneman 1938). The area was classed into the Cliff section of the Cumberland Plateau in the Mixed Mesophytic Forest region by Braun (1950) and the Eastern Broadleaf Forest (Oceanic) province and Northern Cumberland Plateau section by Bailey (1995). The area is characterized by steep slopes dissecting the plateau surface and draining to the Tennessee River. Soils are shallow to deep, stony and gravelly loam or clay, well drained, and formed in colluvium from those on the plateau top (Smalley 1982). Climate of the region is temperate with mild winters and moderately hot summers; mean annual temperature is 55 °F and mean annual precipitation is 59 inches (Smalley 1982).

We used a randomized complete block design with three replications of two treatments. Each site (block) comprised one replication of each treatment established along the slope contour. One replication located on Miller Mountain (34°58'11"N, 86°12'2"W), had a southwestern aspect and a mean elevation of 1,600 feet. Two replications located at Jack Gap (34°56'30"N, 86° 04'00"W), had northern aspects. One Jack Gap replication was located at an elevation of 1,496 feet and the other at an elevation of 1,200 feet. Treatments were randomly assigned to 10-acre areas within each replicated block. Dominant canopy tree taxa on both sites included *Quercus* that represented 46 percent of pretreatment basal area (BA; square feet per acre), including *Q. velutina* Lam., *Q. rubra* L., *Q. alba* L., and *Q. montana* L. *Carya* species were 15 percent of pretreatment BA, *A. saccharum* was 13 percent, and *L. tulipifera* was 9 percent. Common understory species included *Cornus florida* L., *Cercis canadensis* L., and *Oxydendrum arboretum* DC.

The midstory-herbicide shelterwood prescription (MSW) was implemented in two phases. Our goal was to retain 75 percent of the BA by removing midstory stems. In 2001, the stands were treated using an herbicide (Arsenal®, active ingredient imazapyr) by means of the tree injection technique to deaden the midstory. Rates of application were within the range recommended by the manufacturer. Watered solutions were made in the laboratory and then trees received application via waist-level hatchet wounds using a small, handheld sprayer. One incision was made per 3 inches of diameter, and each incision received approximately 0.15 fluid ounce of solution. Herbicide treatments were completed in autumn 2001, before leaf fall. The goal was to minimize the creation of overstory canopy gaps while removing 25 percent of the BA in the stand midstory. All injected trees were in lower canopy positions, reducing the creation of canopy gaps. Control stands received no phase I treatment.

Phase II was implemented in 2010 after 8 growing seasons. Phase II was the release or final harvest. Merchantable trees, primarily those ≥5.5 inches diameter at breast height (d.b.h.) in both treatments, were removed through chainsaw felling and grapple skidding.

Prior to treatment, five measurement plots were systematically located in each treatment area. Plot centers were permanently marked with a 24-inch piece of reinforcing steel, and GPS coordinates for plot centers were recorded. At each plot center, a 0.025-acre plot was established and all trees 1.5 inches d.b.h. and greater were monumented (distance and azimuth measured and recorded from plot center; each tree was tagged with a numbered aluminum tag and species and d.b.h. were recorded). Concentrically located in each of these plots was one 0.01-acre plot used to record the reproduction. Two additional reproduction plots were randomly located within each stand for a total of seven reproduction plots. At each reproduction plot, all woody stems were counted by species and by 1-foot height classes up to 4 feet tall, and then from 4 feet tall up to 1.5 inches d.b.h. A subsample of stems (6-10) across all species encountered on each plot was selected and stems were tagged and monumented. Digital calipers were used to measure each seedling's ground line diameter (g.l.d.) to the nearest 0.01 inch, and a calibrated measuring pole was used to measure height corresponding to the tallest shoot to the nearest 0.1 foot. Data were recorded in late summer of 2001, 2002, 2003, 2004, 2005, 2006, 2009, 2011, and 2014.

To determine how much of full sunlight was penetrating the canopy and reaching the seedling layer through the residual stand structure, we measured photosynthetically active radiation (μmol $m^{-2}s^{-1}$) in the understory and compared those values to measurements taken simultaneously in full sunlight. We used two AccuPar Linear Par Ceptometers, Model PAR-80 (Decagon Devices Inc., Pullman, WA). Measurements with one ceptometer were taken at 4.5 feet above the forest floor at each plot center and along transects equally dissecting each plot. Measurements with the second ceptometer were taken in completely open conditions adjacent to each stand. By matching two simultaneous readings, we could take into account some variation caused by changes in cloud cover. Before analysis, the square root of the percentage of full sunlight data was arcsine transformed to meet normality and homogeneity assumptions.

For this analysis, we combined all *Quercus* (*Q. alba*, *Q. veluntina*, *Q. montana*, *Q. rubra*, and *Q. muehlenbergii* Englem.). We used an analysis of variance by implementing PROC MIXED in the SAS 8.01 system (SAS Institute 2000). We specified a random effect (block) and a repeated statement (time) with the type of covariance matrix assigned unstructured by TYPE = UN option specified as stand (treat). Differences in stems per acre and BA and stems were assessed using Tukey's honest significant difference test with significance set at $\alpha \leq 0.05$.

Table 1.—Stems per acre (SPA) (standard deviation) and basal area (BA; square feet per acre) (standard deviation) for all species 1.5 inches diameter at breast height and greater, for a midstory shelterwood treatment (MSW) and a control, over five time periods, for upland hardwood stands located in Jackson County, AL. Times are as follows: 2001, pretreatment; 2002, 1-year post-treatment; 2009, 8 growing seasons post-treatment; 2011, 1 year after final harvest; 2014, 4 years after final harvest.

	2001		2002		2009		2011		2014	
Treatment	SPA	BA	SPA	BA	SPA	BA	SPA	BA	SPA	BA
Control	291 (112)	126.6 (71.8)	280 a (105)	126.6 (71.8)	315 a (100)	141.0 (80.1)	104 (135)	26.5 (37.5)	141 (186)	27.1 (38.9)
MSW	320 (138)	120.6 (77.4)	117 b (78)	94.6 (77.4)	88 b (50)	101.6 (70.3)	19 (40)	14.1 (44.4)	35 (77)	14.4 (44.5)

Different letters within columns indicate significant difference between treatments at α ≤ 0.05.

RESULTS

Overstory Canopy Composition and Structure

Stands initially contained 17 species common to both treatments, and composition was typical for Cumberland Plateau escarpment forests. Pretreatment inventories showed that stands were fully stocked and had BAs of 119.4-147.6 square feet per acre for all trees ≥ 1.5 inches d.b.h., averaging 129.2 square feet per acre (Table 1). Diameters were 1.5-28.3 inches d.b.h. Trees with diameters greater than 20 inches d.b.h. included *Q. alba*, *Q. rubra*, *Q. veluntina*, *Q. montana*, *L. tulipifera*, and *Fagus grandifolia* Ehrh. Stem densities were 291-320 stems per acre (SPA) (Table 1). Across all stands, stem densities were dominated by *A. saccharum* (30.7 percent), *Quercus* (11.1 percent), and *L. tulipifera* (4.6 percent). Distribution of stems by diameter class for all stands resembled a typical inverse J-shaped curve. On average, 60.2 percent of the stems were 1.5-5.5 inches d.b.h.

From 2001 through 2009, the control treatment accrued 24 additional SPA and 14.4 square feet per acre of BA on average. Ingrowth species were *F. grandifolia*, *C. canadensis*, and *Magnolia acuminata* L. Control treatment BA did not differ from that of the MSW treatment across phase I and phase II of the shelterwood treatments (Table 1). In 2010, the control stands were clearcut, resulting in 26.5 square feet per acre of residual BA and 104 SPA. In 2014, 4 growing seasons postcut, the control stands had no stems ≥ 15.5 inches, and 62.4 percent of the stems were 1.5–3.5 inches d.b.h. There were 37 SPA of *L. tulipifera* in the smallest diameter class, 13 SPA of *A. saccharum* in the smallest diameter class, 13 SPA in the 3.6- to 5.5-inch diameter class, and no stems of *Quercus*. In 2014, 17 species were tallied in the control, and the species lost compared to the pretreatment richness included *Nyssa sylvatica* M., *Robinia pseudoacacia* L., *Q. montana*, *C. florida*, *O. arboretum*, and *Q. alba*.

In the MSW treatment, the herbicide treatment targeted midstory and not overstory trees. We applied herbicide to 202 SPA of ≥1.5-10.5 inches d.b.h. BA removed in this treatment was 19.4 square feet per acre, or 16.1 percent of the total. Initially, residual BA did not differ from the control ($P = 0.39$), but stem densities differed from that of the control treatment ($P < 0.001$), and this difference was sustained through 2009 (Table 1). Nine species were targeted in the herbicide treatment. *A. rubrum* L. was the primary species for removal (56 SPA treated), followed by *A. saccharum* (53 SPA treated) and *N. sylvatica* (40 SPA treated). Following the midstory removal, the diameter distribution curve changed. There were 27 SPA of *Quercus* after the herbicide treatment, 13 SPA in the 7.5- to 19.5-inch diameter classes, and 14 SPA ≥19.5 inches d.b.h. There were also 13 SPA of *A. saccharum*, all ≤13.5 inches d.b.h., and 3 SPA of *L. tulipifera* in the

7.6- to 9.5-inch d.b.h. classes. No stems of ingrowth were recorded in the 8 growing seasons postherbicide. Before phase II—the final overstory removal harvest—these stands had 101.6 square feet per acre of BA and 88 SPA. After the harvest the stands had 14.5 square feet per acre of BA and 19 SPA. The species of residual stems after the final harvest were *Carya ovalis* Sarge., *F. grandifolia*, *Fraxinus americana* L., *C. canadensis*, *Prunus serotina* Ehrh., and *R. pseudoacacia.* The *C. ovalis* was the largest diameter tallied at 11.5 inches d.b.h.; the other stems were all ≤4.0 inches d.b.h. Only *C. canadensis* and *F. americana* were tallied more than once across all plots, and were the majority of the residual stems (8 SPA and 16 SPA, respectively). Results 4 years after the final harvest (2014) were similar to results 1 year after the final harvest (2011).

Understory Light Levels

We examined the percentage of full sunlight reaching the seedling layer for the control and MSW using values collected in August of each year. In 2004 (P = 0.0333), 2005 (P = 0.02), and 2006 (P = 0.012), greater light was transmitted in the MSW treatments compared to the controls (Table 2). The percentage of full sunlight penetrating the MSW canopy was twice that or greater than the control from 2002 through 2006. The range of full sunlight peaked immediately after the herbicide treatment, at 16.5 percent, and it remained around 8.5 percent over the next 7 years. For the control, the greatest light was recorded in 2002 at 8 percent and averaged half that amount, 4 percent, in the following years. After the final harvest, two to four times the light was received in both treatments, but this amount diminished with subsequent growing seasons.

Table 2.—Percentage of full sunlight reaching the seedling layer under a midstory shelterwood treatment (MSW) and a control over eight time periods. The MSW treatment was implemented in 2001; both treatments had complete overstory removal in 2010.

Treatment	Year	Percent full sunlight
Control	2002	8.0
MSW		16.5
Control	2003	5.9
MSW		10.3
Control	2004	3.0 a
MSW		6.3 b
Control	2005	3.6 a
MSW		8.5 b
Control	2006	2.5 a
MSW		9.3 b
Control	2009	5.2
MSW		8.3
Control	2011	57.7
MSW		79.8
Control	2014	44.0
MSW		57.3

Different letters within columns indicate significant difference between treatments at $\alpha \leq 0.05$.

Reproduction Composition and Structure

The reproduction cohort had 30 species common to the control and MSW. Control stands had 8,648 SPA and MSW stands had 9,819 SPA (Table 3). Most stems were ≤1 foot tall (61 percent of control stems and 72 percent of MSW stems), and we found significantly more small stems in the MSW compared to the control before treatment (P = 0.0203). In 2002, the first growing season post-treatment, these small stem densities did not differ between treatments. In 2003 (P = 0.0134), 2004 (P = 0 0.0022), and 2005 (P = 0.0193), the ≤1 foot tall stem densities were again greater in the MSW than in the control. No other differences between size classes were found. In 2006 and 2009, stem densities of seedlings >1 to ≤2 feet tall (P = 0.003) and >2 to ≤3 feet tall (P = 0.00164) were greater in the MSW than in the control, and in 2009 stems >3 to ≤4 feet tall were greater in the MSW than in the control (P = 0.0115). Total stem densities for all reproduction were also greater in the MSW than in the control in 2006 (P = 0.0049) and 2009 (P = 0032). Immediately after the final harvest, SPA of the MSW for stems >2 to ≤3 feet tall (P = 0.0329) and >3 to ≤4 feet tall (P = 0.0268) were greater than those in the control. In 2014, no differences in stem densities between treatments were found. Total stem densities for both treatments were similar to those found when the study began, with 9,329 SPA in the control and 9,386 in the MSW. A major difference, however, was that most of the stems were in the largest size class for both treatments (53 percent of the control and 59 percent of the MSW).

Table 3.—Seedling stems per acre (SPA) for all species, by five size classes and total seedling densities, for a midstory shelterwood treatment (MSW) and a control, over nine time periods, for upland hardwood stands located in Jackson County, AL. Times are as follows: 2001, pretreatment; 2002 through 2009, midstory herbicide post-treatment; 2011, 1 year after final harvest; 2014, 4 years after final harvest.

		Seedling size class					
Treatment	Year	≤1 foot	>1 foot to ≤2 feet	>2 feet to ≤3 feet	>3 feet to ≤4 feet	>4 feet to ≤1.5inches d.b.h.	Total
Control	2001	5,281 b	1,476	548	214	548	8,648
MSW		7,038 a	1,162	333	281	400	9,819
Control	2002	4,414	1,305	343	181	581	6,824
MSW		5,519	1,376	319	295	238	7,748
Control	2003	5,110 b	2,081	776	305	700	8,971
MSW		7,529 a	1,738	667	305	390	10,629
Control	2004	4,371 b	1,576	519	281	652	7,400
MSW		6,862 a	1,643	476	229	381	9,590
Control	2005	5,419 b	1,776	648	381	848	9,071
MSW		7,548 a	2,571	929	429	600	12,076
Control	2006	4,538	1,143 b	533 b	262	881	7,357 b
MSW		6,319	1,943 a	967 a	443	729	10,400 a
Control	2009	6,286	2,457 b	829 b	400 b	995	10,967 b
MSW		6,290	4,824 a	1,800 a	910 a	1,424	15,248 a
Control	2011	6,090	3,890	1,210 b	548 b	1,371	13,110
MSW		4,543	4,633	2,029 a	1,033 a	1,952	14,190
Control	2014	1,548	1,238	924	681	4,938	9,329
MSW		257	1,181	1,262	1,119	5,567	9,386

Different letters within columns indicate significant difference between treatments at α ≤ 0.05.

Before treatment, the MSW stands had more *Quercus* stems in the smallest height class than the control (P = 0.0269) (Table 4). One to five years post-treatment, this difference was not detected. In 2009, ≤1 foot tall *Quercus* stems were once again greater in the MSW than in the control (P = 0.0047). We did not detect a statistical difference in stem density in the other height classes in 2009 but noted that in the MSW, *Quercus* stems increased by 404 SPA in the >1 to ≤2 foot height class and stems in the other height classes increased from none before treatment to 62, 10, and 5 for the >2 to ≤3 foot tall, >3 to ≤4 foot tall, and >4 foot to ≤1.5 inch d.b.h. classes, respectively (Table 4). Immediately after the final harvest, the MSW had greater SPA in the >1 to ≤ 2 foot height class (P = 0.0352), and in 2014 there were more *Quercus* stems in the >3 to ≤4 foot (P = 0.0042) and >4 foot to ≤1.5 inch d.b.h. classes (P = 0.0211) compared to the control. Four years after the final harvest, total *Quercus* densities were 949 in the control and 914 in the MSW; 55 percent of these stems were in the smallest height class for the control and fewer than 5 percent for the MSW. Thirty-three percent of total *Quercus* stems in the MSW were in the largest size class, which represented a change from none before treatment to 300 SPA following 4 growing seasons after the final harvest.

Table 4.—Seedling stems per acre (SPA) for *Quercus,* by five size classes and total seedling densities, for a midstory shelterwood treatment (MSW) and a control, over 9 time periods, for upland hardwood stands located in Jackson County, AL. Times are as follows: 2001, pretreatment; 2002 through 2009 midstory herbicide post-treatment; 2011, 1 year after final harvest; 2014, 4 years after final harvest.

		Seedling size class					
Treatment	Year	≤1 foot	>1 foot to ≤2 feet	>2 feet to ≤3 feet	>3 feet to ≤4 feet	>4 feet to ≤1.5 inches d.b.h.	Total
Control	2001	776 b	129	14	5	14	938
MSW		1,524 a	29	0	0	0	1,552
Control	2002	557	105	19 a	0	14	695
MSW		1,086	24	0 b	0	0	1,110
Control	2003	490	129	14	0	10	643
MSW		1,095	71	0	0	0	1,167
Control	2004	486	124	24	0	14	648
MSW		995	76	24	10	0	1,105
Control	2005	748	81	29	5	14	876
MSW		986	90	10	0	5	1,090
Control	2006	595	76	14	0	10	695
MSW		748	86	19	0	0	852
Control	2009	752 b	114	29	5	5	905 b
MSW		1,143 a	433	62	10	5	1,652 a
Control	2011	795	143 b	25	14	33	1,010
MSW		395	348 a	105	52	14	914
Control	2014	519	229	90	24 b	86 b	948
MSW		48	233	205	129 a	300 a	914

Different letters within columns indicate significant difference between treatments at $\alpha \leq 0.05$.

In 2014, *L. tulipifera* was ranked first in the frequency of occurrence in both control and MSW plots and had the greatest number of SPA in the largest seedling size class; 995 SPA were in the MSW and 1,776 in the control. Before treatment, there were five SPA *L. tulipifera* in the control (all in the largest size class) and 64 SPA in the MSW (50 SPA ≤1 foot tall, and 9 SPA >1 to ≤2 feet tall). In 2003, 2004, 2005, 2006, and 2009, *L. tulipifera* total densities ($P < 0.001$) and ≤1 foot tall densities ($P < 0.001$) were greater in the MSW compared to the control; these stems were predominantly ≤1 feet tall. In 2009, we did not tally any *L. tulipifera* in the control. In 2009, we tallied 500 SPA of *L. tulipifera* in the MSW, with a gradient of stems across the seedling size classes—42 percent in the ≤1 foot tall size class, 37 percent were >1 to ≤2 feet tall, 12 percent were >2 to ≤3 feet tall, and 9 percent were taller than ≥3 feet to ≤1.5 inches d.b.h. In 2014, we found no differences for *L. tulipifera* tallies between treatments. In the control, 75 percent of the total 2,376 SPA of *L. tulipifera* were in the largest seedling size class, and in the MSW, 50 percent of the 1,986 SPA were in the largest size class. *Fraxinus americana* also ranked high as a frequent competitor and had high densities in the largest seedling size class compared to other species. In 2009, we observed more *F. americana* SPA in the MSW (961 SPA) compared to the control (371) ($P = 0.0228$). In 2014, the difference in *F. americana* stem densities did not differ between treatments, but 61 percent of the 625 SPA were in the largest seedling size class in the MSW, with 43 percent of the 250 SPA in the same size class for the control. *Acer saccharum* seedlings also ranked high in plot frequency and SPA. Before treatment, there were no differences for *A. saccharum* seedling densities between the control and MSW. One year after treatment (P = 0.0366), and in 2006 (P = 0.0473) and 2009 (P = 0.0135), there were higher densities of *A. saccharum* seedlings in the >1 to ≤2 foot size class for the MSW compared to the control. The number of seedlings in the MSW increased from 1,886 SPA pretreatment to 3,314 SPA 8 growing seasons post-treatment. In 2009, there were greater densities of *A. saccharum* seedlings in the >2 to ≤3 foot tall (P = 0.0099) and >3 to ≤4 foot size class (P = 0.0196) for MSW compared to the control. In 2011, only the MSW >2 to ≤3 foot height class still had greater SPA compared to the control (P = 0.0052). There were 943 total SPA of *A. saccharum* in the MSW in 2014; 38 percent of these were in the largest seedling size class.

We tagged 183 seedlings in the control and 166 seedlings in the MSW. Tagged seedlings represented 27 species in the control and 18 species in the MSW and included *Quercus* spp., *F. americana*, *A. saccharum*, *Carya* spp., and *P. serotina*. For all species, initial height and g.l.d. did not differ between the control and the MSW (P = 0.2527). Average height was 2.1 feet and average g.l.d. was 0.26 inch. In 2002, 88 percent of the MSW seedlings and control seedlings survived; stems of species that died were dispersed across all species. Control tagged seedling height (2.1 feet) (P = 0.0072) and diameter (0.28 inch) (P = 0.0322) were greater compared to MSW seedling height (1.2 feet) and diameter (0.20 inch). In 2009, 52.5 percent of the control seedlings and 62.7 percent of the MSW seedlings were alive. No differences in heights and diameters between treatments were found in 2009, although seedlings did grow, and control seedlings averaged 3.1 feet tall with 0.50 inch g.l.d., and MSW seedlings averaged 2.8 feet tall and had 0.43 inch g.l.d. In 2014 following the overstory harvest, 26.8 percent of the control seedlings were alive compared to 34.9 percent in the MSW treatment.

We tagged 66 *Quercus* seedlings in the MSW and 67 in the control. Survival after treatment was 85 percent in the control and 93 percent in the MSW. Survival after 8 growing seasons was 31 and 60 percent for the control and MSW, respectively, and after final harvest survival was 21 (control) and 29 (MSW) percent. There were no differences between treatments at any given time for tagged *Quercus* height and diameter. Control *Quercus* seedlings grew in height from 1.1 to 2.2 feet during the study, and MSW seedlings grew from 1.0 feet to 4.5 feet. In addition, g.l.d. increased for control (0.15 inch to 0.39 inch) and MSW (0.11 inch to 0.64 inch) seedlings.

DISCUSSION

Removing the midstory decreased density of stems 1.5 inches and greater compared to the control. The density of overstory and midstory *Quercus* stems did not change during phase I in the MSW and in the control over 8 growing seasons. After phase II—the final harvest—no residual *Quercus* stems were ≥1.5 inches d.b.h. in either treatment. Four years after the final harvest, the species composition of larger stems in the MSW is scant; only two species, *C. canadensis* and *F. americana*, provide any structure for stems greater than 1.5 inches.

Light availability in studies of midstory removal at phase I have been reported at 10-20 percent of full sun (Craig et al. 2014, Dey et al. 2012, Lhotka and Loewenstein 2009, Lorimer et al. 1994, Miller et al. 2004, Motsinger et al. 2010, Parrott et al. 2012, Schweitzer and Dey 2011). Deadening of the midstory provided an ephemeral increase in the amount of full sunlight reaching the seedling stratum (Lockhart et al. 2000, Miller et al. 2004, Schweitzer and Dey 2011). The highest light penetration was recorded in the first growing season post-treatment, and a diminishing percentage was recorded through 2009. Light availability in the MSW was only within a 10 to 20 percent range for 2 years post-treatment, although it was always higher than that measured in the control. In a central Appalachian hardwood stand in West Virginia, Miller et al. (2004) found slight reductions in light levels in their midstory removal treatment after 3 growing seasons and attributed this to the expansion of overstory trees into gaps created by the treatments. They expected, however, enhanced light under the most severe midstory treatment to continue for 10 to 12 years after treatment. On two studies on the northern Cumberland Plateau in Kentucky, Parrott et al. (2012) and Craig et al. (2014) removed midstory trees and reduced BA by 20 percent, which resulted in 14-18 percent of full light reaching the forest floor after 7 years.

On our mid-Cumberland Plateau escarpment study sites, we found increases in non-*Quercus* seedlings associated with increased light and growing space created by midstory removal. Our midstory herbicide treatment did not target stems ≤1.5 inches d.b.h. Our stands contained 400 SPA of seedlings >4 feet tall to ≤1.5 inches d.b.h. in 2001, and by 2009 this had increased to 1,425 SPA. *A. saccharum* stem density in the largest seedling size class increased from 110 SPA to 190 SPA, *L. tulipifera* from 0 SPA to 33 SPA, and *F. americana* from 4 SPA to 29 SPA. Forty percent of the occupancy of this stratum's growing space (taller than 4 feet to the midstory canopy) comprised shade-tolerant small-stature trees of *Viburnum* spp. and *C. canadensis* (571 SPA in 2009). Parrott et al. (2012) also reported that non-*Quercus* competitive species such as *A. rubrum* benefited from the midstory removal. Lhotka and Loewenstein (2009) implemented a midstory herbicide treatment in conjunction with the treatment of understory vegetation <4.5 feet tall, and they suggested that the small gain in *Quercus* height growth under the more severe treatment did not warrant the costs associated with treating the understory. This study did not consider the ground-layer understory vegetation in the treatment. On these sites, treating these ground-layer woody stems with herbicide may be prudent to prevent them from occupying the subsequent midstory growing space and decreasing the light created when the midstory dies.

Phase I of this shelterwood prescription was designed to develop and maintain *Quercus* species so that a critical density of competitive seedlings is present before the overstory is removed. Many have reported that a stocking goal should be 400 SPA of *Quercus* that are ≥4.5 feet tall (Janzen and Hodges 1987, Johnson et al. 2009, Lockhart et al. 2000, Sander 1979). Although we did not meet this minimum stocking level, we did have an increase from 0 to 300 SPA in the largest *Quercus* seedling size class in the MSW. In addition, we measured a 3.5-foot height growth and a 0.5 inch g.l.d. increase over the 8 growing seasons in the MSW treatment, compared to 1.1 feet height growth and 0.2 g.l.d. growth for *Quercus* in the control. Understory light levels of 20-50 percent full sunlight have been reported to promote *Quercus* growth (Ashton and Berlyn 1994, Gottschalk 1994, Hodges and Gardiner 1993, Rebbeck et al. 2012), although several studies have reported positive *Quercus* growth under a range of 3 to 18 percent full sunlight (Craig et al. 2014, Lhotka and Loewenstein 2009, Motsinger et al. 2010).

Four years after the final harvest, the MSW stands are devoid of vertical structure and appear shrubby, in contrast to the control stands, which had greater stump sprouting and higher diversity of woody stems. Landowners who are interested in pursuing such treatments to enhance or maintain *Quercus* in their stands should note the appearance of the MSW at this stage. The reproduction cohort continues to respond, and the MSW stands will continue to succeed toward a mixed upland hardwood forest, although the species composition may differ from that of the previous stand. The results of this study, which represent composition and structure dynamics at a stand level after the final harvest, indicated that although *Quercus* seedlings increased in densities and height, competing species also responded favorably to the MSW. An additional tending treatment to release competitive *Quercus* may be necessary to maintain its stocking in these stands.

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EARLY STUMP SPROUTING AFTER CLEARCUTTING IN A NORTHERN MISSOURI BOTTOMLAND HARDWOOD FOREST

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Abstract.—Midwestern bottomland hardwood forests are often composed of species that are capable of sprouting vigorously, yet relatively little is known about sprout development within these mixed-species systems. This study describes stump sprouting of midwestern bottomland hardwood species in the first 3 growing seasons after a clearcutting with reserves (~2.0 m^2/ha of residual basal area) treatment. Our results revealed that sprout development varied between species and between small and large parent trees over the 3-year study period. Stumps of American sycamore (*Platanus occidentalis* L.), American elm (*Ulmus americana* L.), and green ash (*Fraxinus pennsylvanica* Marsh.) maintained higher than 80 percent survival; the cumulative survival of river birch (*Betula nigra* L.) was 13 percent by the third growing season. Our results did not support a growth tradeoff between dominant sprout height and the number of live sprouts produced. The variation in sprouting ability that we observed in this study confirms the importance of factoring in pretreatment stand structure and composition when deciding on silvicultural regeneration methods in these bottomland hardwood forests.

INTRODUCTION

Many hardwood tree species are capable of sprouting (del Tredici 2001). Sprouts often have a competitive advantage over other sources of regeneration after disturbances (Dietze and Clark 2008, Vickers et al. 2011, White 1991). The rapid early growth of sprouts is supported by an established root system with stored carbohydrates (del Tredici 2001). A rapid flush of sprouts after a major disturbance can lead to the initiation of stands that are composed mainly of sprout-origin stems. As a result, sprouting allows forests to recover quickly from disturbances and can maintain predisturbance species composition (Dietze and Clark 2008).

Understanding the sprouting potential of a stand can help forest managers design silvicultural prescriptions for many hardwood forest types. Past research has shown that sprouting is related to the age, size, and condition of the parent tree before a disturbance (del Tredici 2001) and that site factors (e.g., site quality) can also influence sprouting (Gould et al. 2007, Johnson 1977, Weigel and Johnson 1998). Much of our understanding of sprouting is, however, based on research in upland forests.

Research on the regeneration of midwestern bottomland hardwood forests has been limited, and forest managers are often challenged to successfully achieve regeneration objectives in this forest type. Bottomland forest ecosystems support species that are capable of sprouting vigorously (Burns and Honkala 1990), but sprouting dynamics in these mixed-species systems are poorly understood. This investigation evaluated the effects of species and parent tree size on stump sprouting of midwestern bottomland hardwood species during the first three growing seasons after clearcutting with reserves. We also tested the relationship between stem density and height growth of sprout clumps to determine if there was a tradeoff between these sprouting characteristics.

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METHODS

Study Site

This investigation is based on a silviculture experiment in a bottomland hardwood forest in northeastern Missouri. The study site is located in the Deer Ridge Conservation Area (CA), a public land area that is managed by the Missouri Department of Conservation. The experiment was established in 1993 in a mature bottomland forest remnant that is adjacent to a channelized segment of the North Fabius River in Lewis County, MO.

The silviculture experiment at the Deer Ridge CA includes a clearcutting treatment applied as one of three treatment levels. Treatments were assigned to 2.6- to 3.2-ha stands after a completely randomized design. A clearcutting with reserves (CCR) treatment was randomly assigned to eight stands, which were treated between summer 1999 and spring 2000. Before treatment, stands that were designated for clearcutting had a mean stem density and basal area of 465 tree/ha and 28 m^2/ha, respectively (trees ≥11.4 cm diameter at breast height [d.b.h.]). The CCR treatment removed nearly all trees >2.5 cm d.b.h. On average, the CCR retained 20 trees/ha and 2 m^2/ha of basal area in reserves. Silver maple (*Acer saccharinum* L.), the dominant species before treatment, was the most abundant species retained after clearcutting followed by oak species (*Quercus* L.), which were retained for habitat value and to encourage oak regeneration (Olson et al. 2015).

Sampling Design and Analysis

After the harvest in 2000, stump sprouting of all trees ≥11.4 cm d.b.h. (large stumps) was tracked in two, 0.2-ha circular plots within each CCR stand. Stems <11.4 cm d.b.h. (small stumps) were also tracked. Before treatment, 15 small silver maple stumps and 15 small American elm (*Ulmus americana* L.) stumps were identified in each stand to track sprouting. Small stumps of other common species and groups, including American sycamore (*Platanus occidentalis* L.), boxelder (*A. negundo* L.), hackberry (*Celtis occidentalis* L.), eastern cottonwood (*Populus deltoides* Bartr. Ex Marsh. var. deltoides), hickory spp. (*Carya* Nutt.), river birch (*Betula nigra* L.), and green ash (*Fraxinus pennsylvanica* Marsh.), were also tracked but at lower sampling intensity. As many as 100 small stumps of each species were initially marked and tracked in all CCR stands combined.

Several attributes of each sprout clump were recorded, including the number of live sprouts and height of the dominant sprout. Sprouting information was collected following the first (2000), second (2001), and third (2002) growing seasons.

A repeated measures analysis of variance (ANOVA) was used to assess the effects of species and parent tree size (large versus small stumps) on the cumulative survival (i.e., survival based on the initial population) of stumps, number of live sprouts/stump, and height of the dominant sprout within a sprout clump over the first three growing seasons following clearcutting (2000–2002). ANOVA models were based on a completely randomized split-plot design. Species (whole-plot factor) and parent tree diameter class (split-plot factor) were included as fixed effects, and stand was included as a random effect. Response variables were summarized at the stand level before analysis. Low sample sizes for some species limited the number of species included in ANOVA models. ANOVA was performed using PROC MIXED in SAS (SAS Institute Inc., Cary, NC; version 9.2). Data were transformed to improve normality and homogeneity of residuals when warranted. Mean separation of significant sources of variation (SVs) was performed using Tukey's honestly significant difference. Significant interactions were examined using the SLICE option. Significance was assessed at $\alpha = 0.05$.

To test for a tradeoff between the height and stem density of sprout clumps, the relationship between dominant sprout height and the number of live sprouts after the first growing season was tested using simple linear regressions built individually for each species. After inspecting scatterplots and linear regression results, we refit models for species with positive slopes from linear regression using a power function:

$$y = a \times x^{b} \qquad (1)$$

Where

y = dominant sprout height,

x = number of live sprouts, and

a and b are estimated parameters.

For species with a negative slope in the linear regressions, models were refit with a negative exponential function:

$$y = a \times e^{(-b \times x)} \qquad (2)$$

All models were fit using PROC MODEL in SAS, and significance was assessed at $\alpha = 0.05$.

Table 1.—ANOVA table for analysis of cumulative stump survival, density of sprouts/stump, and dominant stump sprout height for sprouts originating off of large and small stumps (dbh ≥11.4 and <11.4 cm, respectively). Sources of variation (SV) are species (S), diameter class (D), year (Y), and interactions among the three.

	Stump survival				Sprouts/stump				Dominant height			
SV	Ndf	Ddf	F	P > F	Ndf	Ddf	F	P > F	Ndf	Ddf	F	P > F
S	7	62.3	6.27	<0.001	4	44.4	5.49	0.001	4	39.3	7.72	<0.001
D	1	14.1	12.55	0.003	1	15.1	23.60	0.002	1	13.5	0.09	0.773
S*D	7	62.3	1.60	0.1523	4	44.4	1.07	0.3840	4	39.3	0.95	0.444
Y	2	140	17.27	<0.001	2	101	33.77	<0.001	2	95.2	53.71	<0.001
S*Y	14	140	1.79	0.045	8	101	4.71	<0.001	8	95.2	4.70	<0.001
D*Y	2	140	1.44	0.241	2	101	1.26	0.288	2	95.2	0.48	0.620
S*D*Y	14	140	1.50	0.118	8	101	1.04	0.409	8	95.2	0.69	0.699

RESULTS

Species, year, and the interaction of species and year were significant SVs in ANOVA models for stump survival, sprouts per stump, and dominant sprout height for all stumps (large and small) in the CCR treatment (Table 1). Diameter class of stumps (large versus small stumps) was also a significant SV in models for stump survival and sprouts per stump.

Averaging across diameter classes, mean cumulative stump survival exceeded 80 percent for American elm, green ash, American sycamore, and hackberry after the first growing season and remained higher than 80 percent for American elm, green ash, and American sycamore by the end of the third season (Fig. 1A). In contrast, river birch survival was 22 percent after the first season and dropped to 13 percent by the end of the second and third seasons, which was significantly lower than several species in all years. Survival significantly decreased for hackberry (years 1 and 2 > year 3) and green ash (year 1 > years 2 and 3), but we saw no difference in survival for the other species. When averaged across species and time, mean cumulative survival of small stumps (80 percent) was significantly greater than large stumps (56 percent) (Fig. 2A).

The mean number of sprouts per stump, when averaged across diameter classes, differed between species only after the first growing season (Fig. 1B). American elm and hackberry stumps supported a significantly greater number of sprouts than silver maple, green ash, and American sycamore, with three, five, and four more sprouts per stump on average, respectively. Mean sprouts per stump for all species declined steadily and significantly over 3 years (i.e., year 1 > year 2 > year 3). Most notable were significant decreases in the number of sprouts for American elm (year 1 > years 2 and 3) and hackberry (year 1 > year 2 > year 3). When averaged across species and time, the mean number of sprouts/stumps was significantly greater for large stumps (nine sprouts) than small stumps (six sprouts) (Fig. 2B).

The mean dominant sprout height did not differ among species after the first growing season but did by the end of the second and third seasons (Fig. 1C). Dominant sprouts of American sycamore were significantly taller (3.8 m) than green ash and hackberry after the second season, and American sycamore was taller (5.3 m) than all species considered after the third season. By the end of the third season, dominant American sycamore sprouts had, on average, a height advantage over dominant sprouts of silver maple, American elm, green ash, and hackberry of 1.3 m, 1.2 m, 2.0 m, and 2.1 m, respectively.

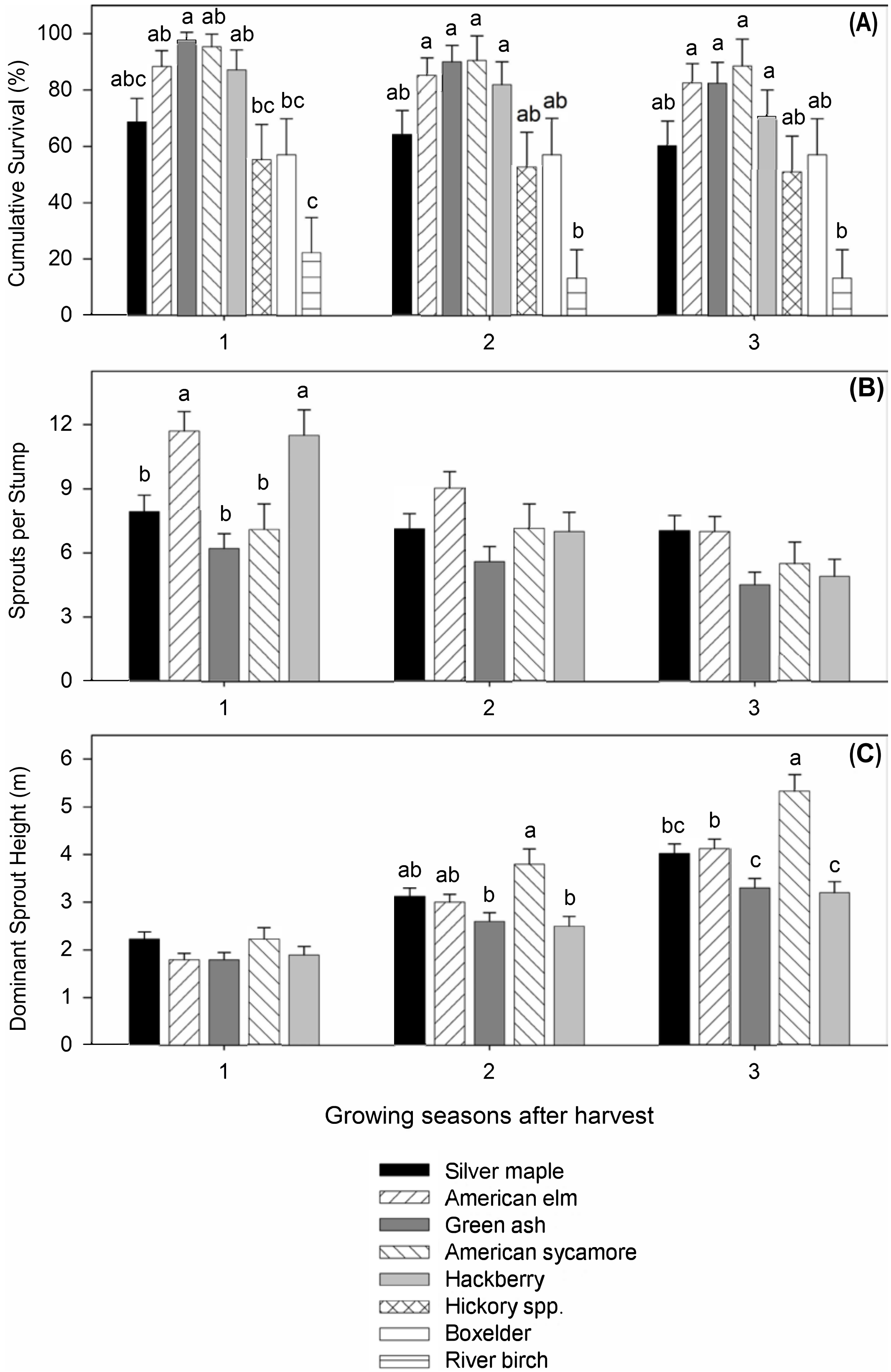

Figure 1.—Least squares means of cumulative stump survival (A), number of sprouts/stump (B), and dominant stump sprout height (C) for bottomland hardwood species over three growing seasons after harvest. Fewer species were included in analysis of variance models for sprouts/stump and dominant sprouts because of low sample size. Different letters indicate significant differences between species within each respective year ($p < 0.05$).

Table 2.—Regression results for number of sprouts at the end of year 1 and sprout height at the end of year 1

Species	*a*	*b*	p-value	r^2
American elm[a]	1.63	0.05	0.0326	0.0123
American sycamore[a]	1.74	0.17	<0.0001	0.2666
Boxelder[a]	1.91	0.16	0.0037	0.1539
Eastern cottonwood[b]	1.53	0.01	0.8552	0.0011
Green ash[b]	1.87	0.01	0.4520	0.0051
Hackberry[a]	1.53	0.10	0.0339	0.0442
Hickory spp.[a]	1.08	0.04	0.6108	0.0039
Oak spp.[b]	2.11	0.01	0.6773	0.0163
River birch[a]	1.60	0.15	0.2287	0.0840
Silver maple[a]	1.73	0.19	<0.0001	0.2208

[a]Model form $y = a \times x^b$

[b]Model form $y = a \times e^{(-b \times x)}$

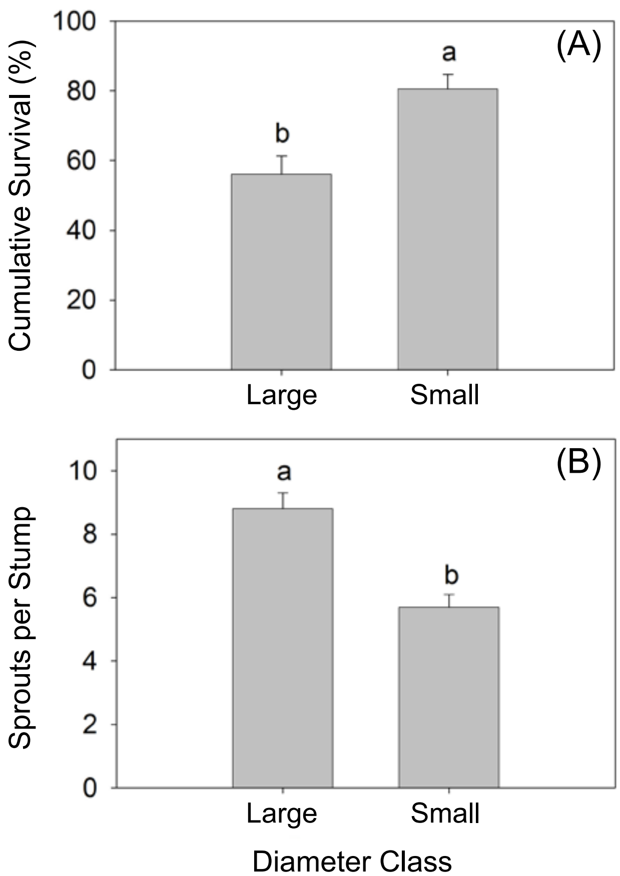

Figure 2.—Least squares means of cumulative stump survival (A) and number of sprouts/stump (B) for large and small stumps (d.b.h. ≥11.4 and <11.4 cm, respectively) of bottomland hardwood species over three growing seasons after harvest. Different letters indicate significant differences between diameter classes ($p < 0.05$).

Simple linear regression of first year dominant sprout height predicted from the first year number of live sprouts per stump produced species-level models with positive and negative slopes. The nonlinear regression models resulted in significant models for American elm, American sycamore, boxelder, hackberry, and silver maple (Table 2). The strongest relationships between number of sprouts and dominant sprout height from significant models were for American sycamore (Fig. 3B) and silver maple (Fig. 3E), with r^2 values of 0.2666 and 0.2208, respectively.

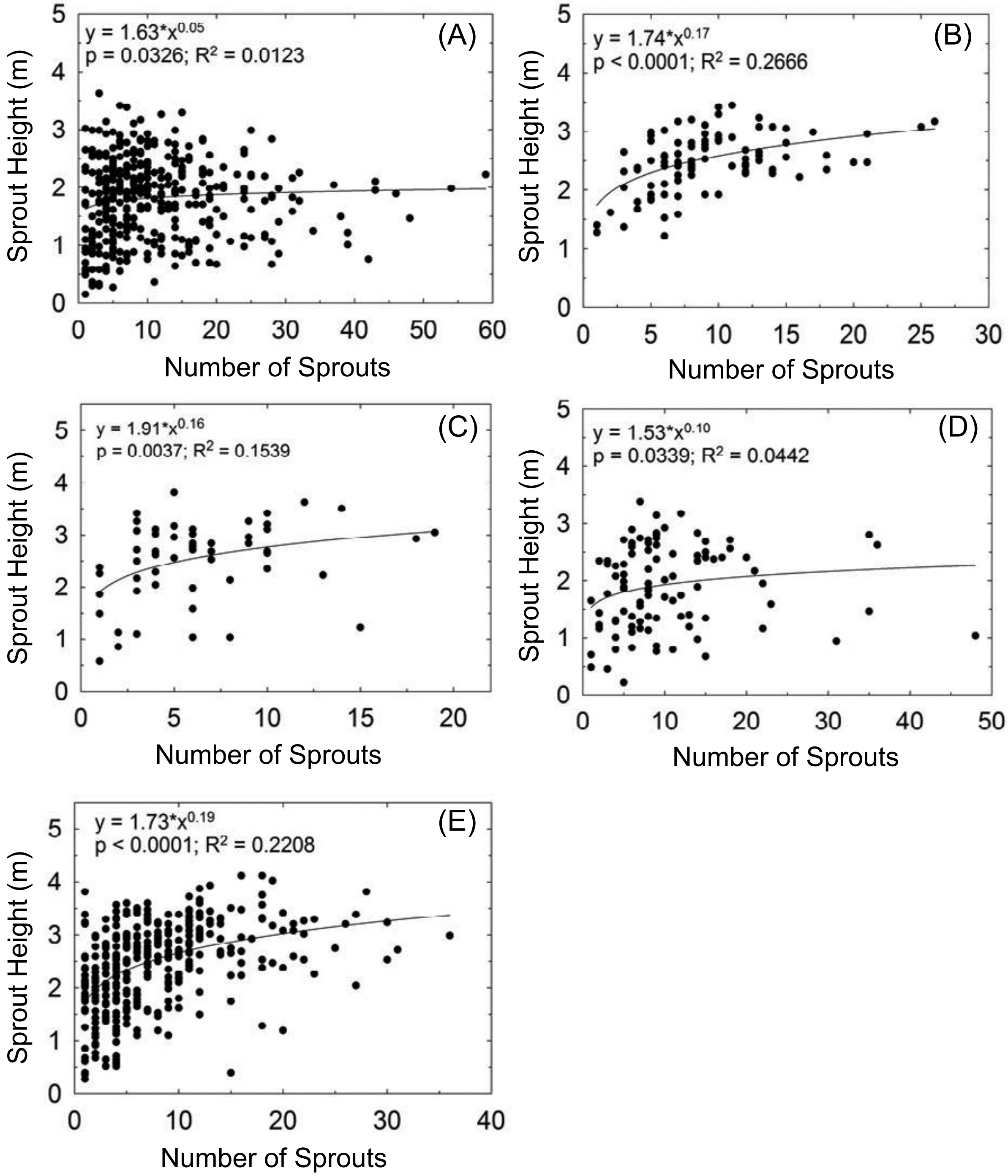

Figure 3.—Scatterplots and nonlinear regression models depicting the relationship between first-year dominant sprout height and the number of live sprouts for American elm (A), American sycamore (B), boxelder (C), hackberry (D), and silver maple (E).

DISCUSSION

Sprouting characteristics observed in this study varied by species. Three-year cumulative survival ranged from a low of 13 percent for river birch to greater than 80 percent for American elm, American sycamore, and green ash. Sprout clumps are generally expected to have high survival because of the initially rapid growth that enables sprouts to assert early dominance (Gould et al. 2007, McQuilkin 1975) that may persist for several decades after overstory removal (Johnson 1975, Roth and Hepting 1943, Wendel 1975). Low survival of river birch sprouts could be linked to this species' pioneer strategy. River birch frequently produces large crops of light, wind-dispersed seed, and it is capable of producing abundant seedlings, particularly on recently disturbed sites (Grelen 1990). Because of this ability to establish numerous seedlings after disturbance, river birch may be less dependent on vegetative reproduction than co-occurring species in these floodplain systems.

Interspecific variability in height growth during early stand development is a common pattern of mixed-species stand dynamics after a major disturbance (Oliver and Larson 1996), particularly under open conditions where species can express their growth potential (Vickers et al. 2014). In this study, dominant sprouts of American sycamore were significantly taller than all other species by the end of the third growing season and were, on average, more than 1 m taller than the next tallest species (American elm). American sycamore is one of the fastest growing tree species in North America and is a vigorous stump sprouter (Wells and Schmidtling 1990). Silver maple and American elm had similar dominant sprout heights, and both species had a height advantage over several other species by year 3. Rapid growth and vigorous sprouting are also attributes of silver maple and American elm (Bey 1990, Gabriel 1990). Rapid sprout growth enables species such as American sycamore, silver maple, and American elm to aggressively recolonize recently disturbed floodplain sites after top-killing disturbances.

Johnson (1975) and Roth and Hepting (1969) reported that sprout clump densities follow a pattern of rapid decline during the first decade followed by a gradual decline over subsequent decades. Stumps of American elm and hackberry supported more live sprouts than other species before the start of the second growing season, but their stem densities declined rapidly during year 2. This decline in density could be related to a variety of biotic and abiotic factors that affect these floodplain forests. From a stand development perspective, this pattern could represent the early transition from the stand initiation stage to the stem exclusion stage. For example, the large flush of American elm and hackberry sprouts initiated by clearcutting (i.e., stand initiation) was followed by early onset of intense competition within sprout clumps and with neighboring plants (i.e., the start of stem exclusion) in the open conditions created by the CCR treatment. Because stand development is a continuous process, transition from one stage to the next occurs continuously with rate changes driven by internal and external factors that either slow or hasten the process.

Within the data range of our study, our findings supported the expectation that parent tree size would influence sprouting characteristics. We observed more live sprouts on large stumps than on small stumps. A higher density of sprouts developing off large stumps was likely due to a larger bank of dormant buds accumulated by larger, and presumably older, trees before harvest (Johnson 1975, Kozlowski et al. 1991). The higher sprout density could also be related to a greater perimeter of cambium for adventitious bud development (Kozlowski et al. 1991). Cumulative survival of sprout clumps was higher for small stumps than for large stumps, which is similar to the commonly reported inverse relationship between sprouting probability and parent tree size (del Tredici 2001). Although we did not detect an effect of stump size on dominant sprout height, differences in sprout density and survival of sprout clumps between small and large stumps could convey competitive advantages onto each. For example, higher survival of small stumps could translate into a survival advantage over sprout clumps that develop from large stumps, whereas a higher density of sprouts from surviving large stumps could increase the likelihood that at least one sprout will secure a position in the developing canopy. The latter situation could result from a buffering effect of larger sprout clumps that protects interior sprouts from competition from neighboring sprout clumps.

Theoretically, stumps that produce many sprouts have less to invest in individual sprout growth than those that produce fewer sprouts. This scenario represents a growth tradeoff in how live stumps allocate finite resources to sprout production and would be supported by a negative relationship between sprout height and number of live sprouts. Regression relationships between dominant sprout height and the number of live sprouts after the first growing season, however, indicated that few species displayed patterns of a negative relationship, none of which were statistically significant. This result does not support a simple growth tradeoff between sprout height and density. A significant positive relationship was detected for several species, although the r-squared values were greater than 0.2 for only American sycamore and silver maple. The power function used to describe this relationship suggests that dominant sprout height reaches either a peak or a plateau with an increasing number of live sprouts. This pattern is biologically logical because there is likely a limit to maximum sprout height in a 3-year period. Moreover, stumps that produce low numbers of sprouts may be of low vigor, resulting in poor subsequent growth of the sprouts. Including additional covariates, such as size and vigor of the parent tree, may also help to shed light on growth tradeoffs in sprout production.

CONCLUSIONS

Our study provides evidence that sprouting is common to bottomland hardwood species, and the interspecific variation in sprouting ability suggests the importance of factoring in pretreatment composition when deciding on silvicultural regeneration methods in these hardwood forests. For example, even-aged regeneration methods applied to stands composed of species with a strong sprouting ability, such as American sycamore and American elm, are more likely to stimulate sprouts that capture a dominant position in the regenerating cohort than stands composed of species with relatively poor sprouting ability, such as river birch. In this study, however, we considered only sprout-origin regeneration. Additional research is needed to determine the contribution of sprouting to stand initiation in this mixed-species forest type.

ACKNOWLEDGMENTS

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DISTURBANCE

DISTURBANCE, SUCCESSION, AND STRUCTURAL DEVELOPMENT OF AN UPLAND HARDWOOD FOREST ON THE INTERIOR LOW PLATEAU, TENNESSEE

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Abstract.—We quantified species composition, stand structure, canopy disturbance history, and *Quercus* establishment and canopy accession patterns in an upland hardwood forest in Tennessee. The forest established in the mid-1800s and exhibited structural characteristics that were within the range of what has been reported from other late-successional forests in the region. The forest overstory was dominated by *Quercus prinus*, but *Acer saccharum* was the most abundant species. *Quercus* recruitment was continuous from stand initiation through the 1950s. Prior to 1880, the majority of *Quercus* trees established in closed canopy conditions, whereas after 1880 most *Quercus* trees established in high light environments. *Quercus* establishment in canopy gaps resulted in multi-aged *Quercus* populations in the forest. We documented three forest-wide disturbances during development (1922, 1945, and 1973). The 1922 event coincided with an establishment pulse of largely understory taxa, and the 1945 event corresponded with an establishment pulse of *A. saccharum*.

INTRODUCTION

Late-successional hardwood stands throughout the temperate zone are noted for their unique structural characteristics, which are often described as complex (Hale et al. 1999, Johnson 2004, Oliver and Larson 1996). Ironically, we know little about the disturbance processes that occurred during development to create the structures that characterize these late-successional stands. Furthermore, late-successional stands vary considerably in their structural attributes (Parker 1989, Tyrrell et al. 1998), and it is unknown to what extent that variability is controlled by a stand's unique disturbance history (Burrascano et al. 2013). The overarching goal of our study was to document long-term patterns of forest development and disturbance in a late-successional *Quercus*-dominated forest on the southern Interior Low Plateau in Tennessee. Our specific objectives were to: (1) quantify species composition and structural elements, (2) reconstruct the frequency, magnitude, and spatial extent of canopy disturbance events during forest development, and (3) elucidate relationships between disturbance processes and forest development.

METHODS

Our study was conducted on the Hill Forest State Natural Area (HFNA) located 20 km southwest of Nashville, TN. The 91 ha reserve is managed as a state natural area by the Tennessee Department of Environment and Conservation. Our study was specifically focused within a ca. 60 ha portion within the core section of the reserve. To quantify forest composition and structure, we established 27 separate 0.04 ha fixed-radius plots throughout the inner core of the HFNA. On each plot we recorded species, crown class, and diameter at breast height (d.b.h.;

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diameter measured ca. 1.4 m above the ground) of all stems ≥5 cm d.b.h. to quantify species composition and stand structural characteristics. We used increment borers to extract tree core samples from all trees ≥20 cm d.b.h. and the four trees ≥5 cm but <20 cm d.b.h. located nearest to plot center on each plot. To increase our sample size of trees to reconstruct disturbance history, we opportunistically collected increment cores from *Quercus* (*Q. prinus*, *Q. rubra*, *Q. alba*, and *Q. velutina*) canopy trees immediately adjacent to our inventory plots that exhibited characteristics associated with old trees such as low stem taper, high stem sinuosity, and low crown volume (Pederson 2010).

Once all tree rings on our *Quercus* core samples were visually dated, we measured raw ring widths of all overstory *Quercus* series (n = 56) to the nearest 0.001 mm using a Velmex measuring stage (Velmex, Inc., Bloomfield, NY) interfaced with Measure J2X® software (VoorTech Consulting, n.d.). The measurement series were visually compared and statistically analyzed using segmented time series correlation analysis in the COFECHA program to ensure each growth ring was assigned to the proper calendar year of formation (Grissino-Mayer 2001, Holmes 1983). We then analyzed changes in raw ring widths with respect to the running mean of the previous and subsequent 10 years (Nowacki and Abrams 1997). Release events were identified as periods in which raw ring width was ≥25 percent (minor) or ≥50 percent (major) of the 10-year preceding and subsequent mean, sustained for a minimum of 3 years (Hart and Grissino-Mayer 2008, Hart et al. 2008, Hart et al. 2012). We classified forest-wide canopy disturbance events as episodes when there was a synchronous release experienced by a minimum of one tree at least 10 years of age on 25 percent of the study plots (Hart and Grissino-Mayer 2008, Nowacki and Abrams 1997, Rubino and McCarthy 2004). Trees with mean radial growth rates of ≥2.00 mm/year for the first 10 years were considered to be of gap origin, and those with mean radial growth rates of <2.00 mm/year for the first 10 years were classed as understory origin (Hart and Grissino-Mayer 2008, McEwan and McCarthy 2008, Ruffner and Abrams 1998). We subsequently determined if and when the trees experienced growth releases identified by the 10-year running mean method. We combined the recruitment strategy classes with the detection of subsequent release event(s) to assign all *Quercus* trees ≥20 cm d.b.h. to an establishment and canopy accession strategy class: gap origin-no release, gap origin-gap release, or understory origin-gap release (Hart et al. 2012, Rentch et al. 2003).

RESULTS AND DISCUSSION

Species richness of woody stems ≥5 cm d.b.h. was 29, diversity (H') was 2.61, and evenness (J) was 0.78. Basal area was 24.0 m^2/ha and tree density was 426 stems/ha. The most important species (based on relative density and dominance) were *Acer saccharum* and *Quercus prinus,* and the most dominant species was *Quercus prinus*. The diameter structure of all trees in the forest revealed an inverse J-shape from small size classes to large size classes. The mean q-factor was 1.38. We documented 20 trees/ha ≥60 cm d.b.h. and 8 trees/ha ≥70 cm d.b.h. The oldest tree we documented was a *Q. prinus* with an inner date at breast height of 1841. *Quercus* establishment was continuous from the mid-1800s through the 1950s. However, establishment was greatest from 1890 through the 1910s, with 45 percent of all *Quercus* trees in our sample becoming established during this 30-year period.

Of the 56 overstory *Quercus* trees analyzed for disturbance history, 46 (82 percent) exhibited at least one release event. In total, 101 release events (89 percent minor and 11 percent major releases) were detected from these individuals. A total of 33 trees (59 percent of samples analyzed) experienced multiple releases, including 15 trees (27 percent) with three or more releases during their lifespans. The mean release duration was 5 years for all *Quercus* combined. The longest period between release initiation events after 1900 was 10 years. Release initiations often occurred in consecutive years; the longest periods of consecutive initiations were 1968-1975 and 1942-1947. The single year with the highest frequency of release initiations was 1945 (*n* = 14 release initiations). The mean release initiation return interval was 3 years.

We documented three forest-wide disturbances during stand development. These episodes were initiated in 1922, 1945, and 1973. Synchronous release initiations across broad areas indicated exogenous disturbances such as strong winds, ice storms, or selective timber harvesting prior to establishment as a natural area resulted in the removal of canopy trees throughout much of the forest. Although trees may have been removed individually or in small groups, these disturbances can be considered intermediate-stand scale events because trees were removed from throughout the forest at a rate well above that of background mortality. From the late-1800s to the end of the record, decadal release to sample size percentages (i.e., the percentage of living trees at least 10 years of age that exhibited a radial growth release) exhibited an oscillating pattern, with peaks occurring approximately every three decades. Peaks in release to sample size ratio values occurred in the 1910s, 1940s, 1970s, and 1990s. Of the 56 overstory *Quercus* trees analyzed, 31 (55 percent) established in a canopy gap and 25 (45 percent) established under a closed forest canopy. Early in stand development, most *Quercus* trees established in closed canopy environments. Prior to 1880, only two *Quercus* trees established in high light environments. In contrast, nine *Quercus* trees established in low light environments prior to 1880. Establishment of gap origin trees peaked from the 1890s to the 1910s. Of the *Quercus* trees that established in the understory of a closed canopy, the mean period of suppression was 28 years, and the longest residence time in the understory prior to overstory release was 62 years observed in a *Q. alba*. We did not observe differences in establishment or accession across *Leucobalanus* and *Erythrobalanus* groups.

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STRUCTURAL COMPLEXITY AND DEVELOPMENTAL STAGE AFTER AN INTERMEDIATE-SCALE WIND DISTURBANCE ON AN UPLAND *QUERCUS* STAND

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Abstract.—Promoting stand structural complexity is an increasingly popular silvicultural objective, as complex structures are hypothesized to be more resistant and resilient to perturbations. On April 20, 2011 in Lawrence County, Alabama, an EF1 tornado tracked 5 km, leaving a patchwork mosaic of disturbed areas. In summer 2014, we established a 100 m × 200 m (2 ha) rectangular plot perpendicular to the swath of the storm within an affected *Quercus alba* stand to document the effects of wind disturbance on stand structure. Stem mortality was clustered near the swath of the tornado. Within the 2 ha plot, 22 percent of the basal area was removed by the wind event. To compare structural attributes across areas of increasing disturbance severity, we divided the plot into disturbance classes (minimal, light, and moderate). Our results will improve our understanding of the structural attributes of upland *Quercus* stands after an intermediate scale wind event and may be used to determine types of silvicultural systems needed to enhance structural complexity and minimize the disparity between natural and managed stands.

INTRODUCTION

The enhancement and maintenance of stand structural complexity is an increasingly popular management objective because stand structure is often used as a proxy for managing stand function (Franklin et al. 2002, O'Hara 2014, Puettmann 2011). Relative to more homogeneous stand structures, complex structures with heterogeneity in stem size, canopy architecture, and presence of deadwood are hypothesized to be more resistant and resilient to perturbations and typically promote biodiversity (Hansen et al. 1991, O'Hara and Ramage 2013). One approach to achieving complex structures is to use natural disturbance-based silviculture because natural disturbances often enhance structural complexity. Natural disturbance-based silviculture attempts to emulate the biological legacies of disturbance regimes, and the complexities of these legacy structures vary by disturbance type, intensity, and severity. More information on intermediate scale disturbances, which may create multi-aged stands and more complex structures, is needed to refine silvicultural treatments intended to enhance structural heterogeneity or to emulate the biological legacies created by natural disturbances.

METHODS

The study site was located in an upland *Quercus alba* stand in the Sipsey Wilderness on the William B. Bankhead National Forest in north Alabama. On April 20, 2011, an EF1 tornado tracked 5 km through the Sipsey Wilderness, leaving a patchwork mosaic of disturbed areas. In summer 2014, we established a 100 m × 200 m rectangular plot perpendicular to the swath of the storm in an affected *Q. alba* stand. Within the plot we recorded the species, diameter at

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breast height (d.b.h.), and height of all living stems ≥5 cm d.b.h.; and the species, diameter at 1.37 cm above the root collar, and decay class of all dead woody stems. Decay classes assigned to each dead stem included: decay class 1 (least decayed; sound wood); decay class 2 (sound to somewhat rotten wood); decay class 3 (substantially rotten wood); or decay class 4 (most decayed; mostly rotten wood) (Fraver et al. 2002).

Using the basal area removed by the storm, we created a kriged surface to divide the plot into three approximately equal area disturbance severity classes: minimal, light, and moderate disturbance. The minimum and maximum contour values were 0.0-2.3 m^2 ha^{-1} (0-8 percent basal area removed) for minimal disturbance, 2.3-6.4 m^2 ha^{-1} (8-24 percent basal area removed) for light disturbance, and 6.4-20.0 m^2 ha^{-1} (24-75 percent basal area removed) for moderate disturbance. These values corresponded to the wind disturbance classification criteria used by Hanson and Lorimer (2007). To determine the structural complexity of each disturbance class, we used multiple spatial and nonspatial indices, including the Gini coefficient, the Clark-Evans aggregation index (R), the diameter differentiation index (DT), and the structural complexity index (SCI). The Gini coefficient describes how evenly basal area is distributed among stems. A Gini coefficient value of 0 indicates all stems have the same d.b.h., whereas a value of 1 indicates that all stems have unequal d.b.h. values. The Clark-Evans aggregation index indicates the spatial pattern of stems and ranges from 0 to 2.15. A value of 0 indicates complete clustering, 1 represents a random distribution, and 2.15 represents a perfectly uniform distribution. The diameter differentiation index is a spatially explicit index that uses the four nearest neighbors of each tree within a plot to describe the variation in d.b.h. among neighboring trees within a stand. A value of 0 represents neighborhoods within a plot that have no size differentiation, and a value of 1 represents neighborhoods within a plot that have maximum size differentiation: small variation (0.0-0.3); average variation (0.3-0.5); large variation (0.5-0.7); very large variation (0.7-1.0) (Pommerening 2002). The structural complexity index incorporates x and y coordinates of each stem and stem height to describe three-dimensional structural variability. The SCI has a minimum value of 1 (i.e., all stems have the same height) and has no maximum value. We also assigned each disturbance class to a stand-size class using guidelines by Lorimer and Halpin (2014) to describe the stand structure of each disturbance class (Hanson and Lorimer 2007). The stand size classes outlined by Lorimer and Halpin (2014) include sapling, pole, mature-sapling mosaic, mature, and old-growth classes, which respectively correspond to the stand initiation, stem exclusion, mixed stage, understory reinitiation, and complex stages of development described by Oliver and Larson (1996) and Johnson et al. (2009).

RESULTS AND DISCUSSION

Within the 2 ha plot, 22 percent of the basal area was removed by the storm, reducing basal area from 26.5 m^2 ha^{-1} to 20.5 m^2 ha^{-1}. Within the minimal, light, and moderate disturbance classes, 8, 17, and 44 percent, respectively, of the basal area was removed by the storm. All disturbance classes exhibited negative exponential diameter distributions for all stems ≥5 cm d.b.h., thus the storm did not affect the shape of the diameter distribution. The Gini coefficients for the minimal, light, and moderate disturbance classes were 0.68, 0.73, and 0.69, respectively. Although all disturbance classes exhibited an unequal distribution of basal area, the Gini coefficients indicated that basal area was the most unevenly distributed among stems (i.e., more heterogeneous structure) in the light disturbance class. Based on the Clark-Evans aggregation index, live stems were regularly distributed in the minimal (R = 1.14) and light disturbance classes (R = 1.07) and randomly distributed in the moderate disturbance class (R = 0.97). Decay class 1 stems were regularly distributed in the minimal disturbance class (R = 1.88), randomly distributed in the light disturbance class (R = 0.97), and clustered in the moderate disturbance class (R = 0.66). The patterns of stem mortality resulted in a more uniform distribution of living stems post-disturbance. The values for diameter differentiation for the minimal, light, and moderate disturbance classes were 0.46, 0.43, and 0.43, respectively, indicating average variation in neighboring stem size within each disturbance class. A higher proportion of larger stems was removed by the storm, so lower DT values in the light and moderate disturbance classes may be the result of having fewer living large d.b.h. stems. Based on the SCI, the light disturbance class was the most structurally complex (SCI = 3.70). The SCI of minimal and moderate disturbance classes were 2.96 and 2.76, respectively. When assigned stand-size classes, the minimal and light disturbance classes were considered a mature-sapling mosaic size class, whereas the moderate disturbance class was classified as a sapling size class. By relating these size classes to the stages of development (Johnson et al. 2009, Oliver and Larson 1996), we determined that this stand was most likely in the mixed stage of development, although the moderate disturbance neighborhood was classified as sapling size. The classification of the moderate disturbance class as a smaller size class than the minimal and light disturbance classes indicated the increase in intra-stand structural heterogeneity as a result of the wind event. Further analysis based on taxonomic groups will more fully describe the structural complexity of the stand and elucidate the spatial patterns of the biological legacies after the wind disturbance.

ACKNOWLEDGMENTS

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STORMWISE: INTEGRATING ARBORICULTURE AND SILVICULTURE TO CREATE STORM-RESILIENT ROADSIDE FORESTS

Jeffrey S. Ward, Thomas E. Worthley, Thomas J. Degnan, and Joseph P. Barsky[1]

Abstract.—The band of trees within 30 m of roads (i.e., roadside forests) is often left unmanaged during traditional forest management activities because of liability concerns about inadvertently causing a vehicular accident or damaging utility lines during harvests. The trees in these same neglected forests often cause extensive utility outages and road blockages during extreme weather. Building on the prescriptions developed in Connecticut , the authors, utility companies, state foresters, highway departments, and forest landowners initiated a collaborative project—Stormwise—that is developing and testing practical, cost-effective, and proactive protocols that integrate silvicultural and arboricultural practices. The goals are to reduce the risk of damage during extreme storms, increase habitat diversity, recover underused volume, and maintain aesthetic appeal. Immediately adjacent to utility and road corridors, trees are pruned using ANSI standards, and at-risk trees are removed. To the interior, crop tree management is used to develop trees with wind-firm, open-grown characteristics along with subcanopies of short-stature trees, native shrubs, and herbs. Seven study areas have been established along 2.3 km of roadside forests. Lessons learned about tree selection and coordination from implementation at three areas are being incorporated into treatments scheduled at the remaining sites. Results of treatments and monitoring will be used to inform communities and stakeholders about the management of roadside forests.

INTRODUCTION

The 30 m of forest adjacent to utility and transportation corridors, hereafter referred to as "roadside forests", is often left unmanaged because of understandable liability concerns about accidently felling a tree onto a power line or across a road in front of a moving vehicle. Management of roadside forests is typically limited to periodic pruning of branches over utility wires or branching extending into the traffic safety zone, removal of saplings and shrubs in those zones, and the occasional removal of hazard trees.

The relative importance of the roadside forest resource is unappreciated. We estimated that approximately 5 percent of Connecticut's forests are within 30 m of a public road, qualifying them as roadside forests (State Vegetation Management Task Force [SVMTF] 2012). This high proportion categorized as roadside forests might seem to be an outlier because of the unique combination of high forest cover (58 percent, Shifley et al. 2012) and high population density (fourth highest in the United States) in Connecticut; however, a recent study reported that 5 percent of forests across the nation were within 42 m of a road (Riitters and Wickham 2003). Road density does not have to be high for a substantial proportion to be considered roadside forest. For example, roadside forests would account for 4.2 percent of forested area in a 260 ha block bounded by an 8 m wide road.

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Because the expanse of roadside forests with trees that could affect utility infrastructure and transportation in the region is considerable, a worthwhile goal is moving from their benign neglect with the reactive removal of failed trees affecting public safety to fostering the development and implementation of proactive, cost-effective management practices. The challenges for forest managers are to maintain the aesthetic appeal and ecosystem function (Coffin 2007) of forested byways and mitigate the potential for tree-caused damage to infrastructure during extreme storms. Because the oak-hickory forest does not have a balanced age structure at the regional level (Shifley et al. 2012), and rarely does at the landscape level, creating storm-resistant roadside forests provides an opportunity to increase biodiversity by increasing the diversity of age classes, species, and stand structures.

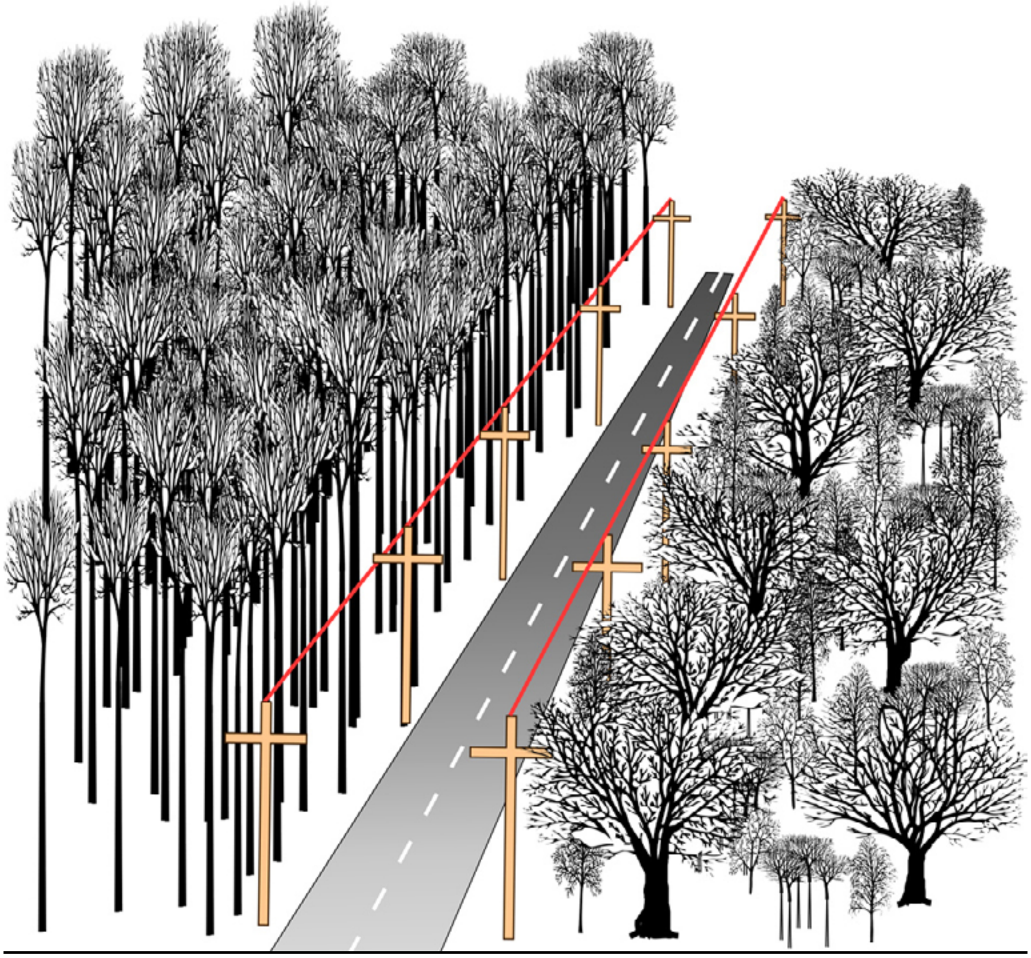
In response to several exceptionally extreme storms in 2011 and 2012, which caused massive tree failure that disrupted utility services and blocked roads for weeks in some areas, a private/public/nongovernmental organization collaborative called Stormwise was formed to characterize and develop management protocols for roadside forests. This partnership includes the private sector (Eversource Energy, United Illuminating, and Connecticut Water Company); public and quasi-public entities (the Connecticut Agricultural Experiment Station, the University of Connecticut, the Connecticut Department of Energy & Environmental Protection, the Connecticut Department of Transportation, the South Central Regional Water Authority, and the Town of Manchester); and nongovernmental organizations (Audubon Connecticut and the White Memorial Foundation).

The proactive approach of Stormwise is to develop healthy, storm-resistant roadside forests by melding arboricultural (individual tree care) and silvicultural (forest management) practices (SVMTF 2012). At the individual tree level, Stormwise vegetation management ideas are based in four very basic concepts:

1. Trees with room or space to grow are healthier trees.
2. Tree branches and twigs will grow and develop toward the sunlight.
3. Trees with the freedom to move in the wind will become more wind firm.
4. The right tree growing in the right place can be given a competitive advantage.

Our vision is to preserve the aesthetic and other benefits of forested roadsides while enhancing their structural and biological diversity. This is achieved by promoting the growth of open-grown trees with crowns that are wide rather than tall, such as those in open fields or orchards, compared to trees in forests, which tend to be tall with narrow crowns. Species with decurrent growth (*Acer*, *Fagus*, *Quercus*) will be favored; those with excurrent growth (*Betula*, *Liriodendron*, *Pinus*, *Populus*) will be discouraged.

Trees growing without dense competition typically display a thigmomorphogenetic (wind) response of stouter buttressed stems and branches and have well-anchored, widespread root systems (Kozlowski and Pallardy 1997). All these characteristics of open-grown trees make them more resistant to wind damage, especially to stem failure or becoming windthrown (Poulos and Camp 2010). Open-grown trees can still provide important aesthetic, hydrologic, and habitat benefits and remain wind firm and of lower risk to power and road infrastructure (Fig. 1). Surrounding these trees and on the immediate roadside edge we envision a matrix of short-stature trees (*Benthamidia*, *Carpinus*, *Ostrya*) and shrubs (*Cornus*, *Hamamelis*, *Viburnum*, *Vaccinium*).



A: Unmanaged roadside forest with tall trees susceptible to storm damage and with few small trees and shrubs.

B: Storm-resistant forest of wind-firm trees that are wide rather than tall and are interspersed with shrubs and small trees. Utility lines are shown on both sides of the street for illustrative purposes.

Figure 1.—Illustration of typical unmanaged roadside forests (A) and roadside forests after full implementation of Stormwise standards (B).

We face a number of challenges to widespread implementation and adoption of Stormwise roadside forest management. On the implementation side, we must develop practical expertise and explore various methods and logistics for implementing a novel vegetation management paradigm. Implementation questions are being examined at our demonstration areas where we are promoting cooperation and coordination of work done by utilities/arborists and foresters/loggers—two groups that are friendly but have not traditionally had the opportunity to work together on projects. Tree crews and forest practitioners will need to use their unique skill sets to develop methods and techniques that are specific to Stormwise roadside forest management.

Acceptance by the public and the arboricultural and forestry communities will be key to widespread adoption of Stormwise roadside forest management. This acceptance will not necessarily hinge only on demonstrating that these practices result in increased utility reliability in extreme weather, but also that the practices create forests that are aesthetically pleasing (Jones 1993, Wolf 2003) and management practices that are cost-effective for utilities, land managers, and landowners. One unique challenge being explored is whether wood produced during roadside forest management activities can be used for products and whether value can be recovered for towns or landowners to offset some of the cost of proactively removing hazard trees.

The proximity of utility lines would present a challenge for implementing these prescriptions if only traditional forest harvesting techniques were available. This challenge is met by partnering with utility tree crews (Eversource Energy and United Illuminating) to fell trees that, should they fail, could land on utility wires or travel corridors. Utility companies immediately benefit from the reduced costs of not having to chip branches (they stay in the woods). Landowners or towns can derive immediate benefits from trees that can be used or sold for higher-value lumber rather than for firewood. The environmental benefits of developing roadside forests with appropriate species include greater biodiversity, increased structural and age diversity, and healthier and more vigorous trees.

The objectives of this paper are to highlight the relative importance of roadside forests, to begin characterization of their composition and structure, and to suggest that roadside forests should be thought of not as an unmanageable liability, nor as a purely aesthetic screen, but as an opportunity to demonstrate proactive forest management. Managing these forests proactively could reduce damage to infrastructure caused by trees during extreme storms; increase age, species, and habitat diversity; recover underused volume; and maintain aesthetic appeal.

STORMWISE FOREST MANAGEMENT

The following paragraphs have been modified from the section "Roadside management in a forested landscape" in the Connecticut vegetation management task force report (SVMTF 2012).

Management Zone

Incorporate the area as large as 30 m on both sides of the infrastructure (e.g., utility lines, roads) passing through forested areas. This includes an infrastructure zone (within 10 m of wires or roads) and a side zone (extending out an additional 20 m) in stands scheduled for forest management (Fig. 2). Mature upper canopy trees are often 20 m tall; some white pine, yellow-poplar, and red oak reach 30 m or taller. Therefore, the management zone should be 30 m wide to include all mature trees that could damage utility infrastructure or block roadways during

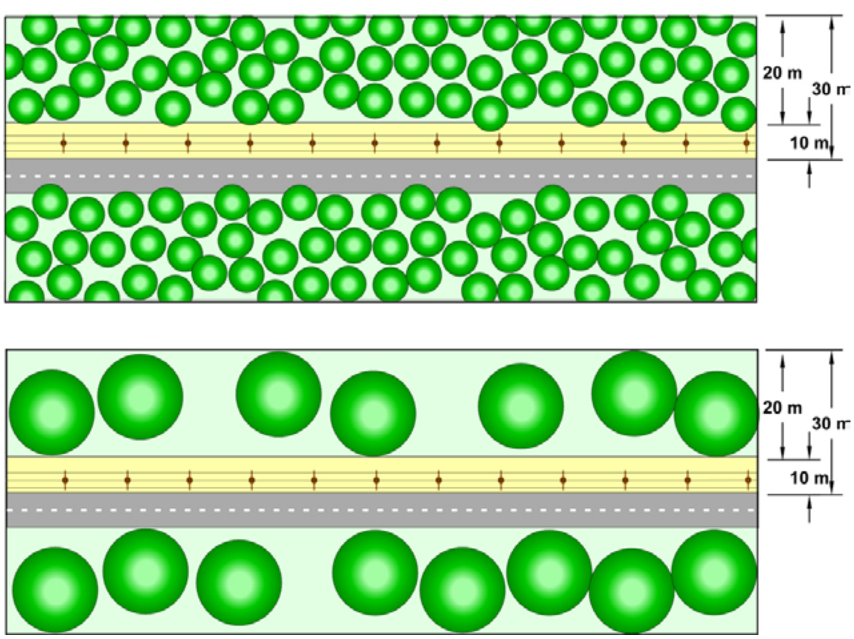


Figure 2.—Bird's-eye view of a typical roadside forest (top) with 1,000+ trees/km on each side of a road compared with a Stormwise forest (bottom) with fewer trees that have crowns that are wide instead of tall. Small trees and shrubs are not shown in either forest, but would occupy gaps between trees.

Tree crowns are shown as green circles and utility poles are illustrated as crossed brown circles connected by brown wires.

an extreme storm. Reducing the visual impact of any cutting operation is imperative because roadside forests by definition are in highly visible locations. Harvesting debris is inevitable; however, care should be taken to reduce the number of high stumps and reduce the height of unmerchantable tops (Jones 1993).

Infrastructure Zone

Combine traditional pruning practices (ANSI A300 Standards) (American National Standard 2005) and an enhanced hazard tree identification process with a long-term, selective tree removal program. Conduct an in-depth review of trees that looks at aboveground and belowground health symptoms and structural risks. This includes reviewing trees that may otherwise be aggressively pruned, to determine if excessive live wood (more than 25 percent) will be removed or the tree will have insufficient leaf area after pruning (typically equivalent to 40 percent of total tree height). Aggressive pruning after a rigid utility line clearance or sight-line standard may leave a tree with long-term structural and health risks that are also visually unappealing. Leaving some structurally sound large trees with well-balanced crowns near wires will improve the aesthetics of the residual stand. Smaller growing trees, shrubs, and grasses should be encouraged to emphasize the "right-tree-right-place" approach. Encourage protection of native species such as dogwood or hophornbeam and shrubs such as mountain laurel, witch hazel, and spicebush. Consider controlling invasive shrubs (e.g., honeysuckle) and vines (e.g., bittersweet). It will be important to consider sight lines when developing a denser understory adjacent to roads.

Side Zone (Young Stands)

Ideally, creation of a storm-resistant forest within the side zone would begin in young stands by releasing 75-100 trees/ha (225-300/curb km, each side of the road) from competition using well-developed crop tree management prescriptions (Miller et al. 2007). Tree selection should be based on growth form (excurrent versus decurrent) and structural integrity and should include a diversity of species. All trees directly competing with selected residual trees should be removed when the canopy begins to close, about every 10-20 years. As trees mature and the crowns of other residual trees close in, the goal should be to reduce the number of residual trees from 75-100 to 30-50 large trees/ha (90-150/curb km) with wide-spreading crowns. These storm-strong trees should be well spaced and managed so that they develop stout trunks and healthy crowns similar to what might be found in an open park. Unlike a park, though, these trees would be surrounded by a diversity of smaller trees, shrubs, and wildflowers.

Side Zone (Mature Stands)

Creating a storm-resistant forest is more challenging in mature roadside forests. Removing all but 30-50 trees/ha in one step would increase the susceptibility of the remaining trees to wind damage for several years until they develop increased stem taper (more wood at the bottom of the tree) and better anchored roots. A more pragmatic approach is to begin with a heavy crown thinning that emphasizes removing trees with obvious structural defects (e.g., cavities or codominant stems with included bark); potential structural defects (e.g., frost cracks, fungal structures, heavy lean, offset crowns); small crowns; perennial diseases; or dieback in the upper crown (Poulos and Camp 2010). The thinning would allow residual trees to become more wind firm by increasing stem taper and developing stronger root systems.

The next step would be to conduct another thinning 10-15 years later, after the trees have become more wind firm. Proceeding immediately to the goal of 30-50 large trees/ha may be possible if sufficient candidate trees with balanced crowns and well-anchored root systems are present. Otherwise, multiple thinnings are recommended to achieve this goal. The thinning process will also encourage growth of native shrubs and small trees that will contribute to the natural look by creating a multilevel canopy reminiscent of an old-growth forest. Currently many, if not most, roadside forests in Connecticut are even-aged stratified mixtures. A more desirable condition would be a low-density, uneven-aged mixture. The specific combination of silvicultural treatments needed to achieve this condition will depend on a number of factors, including initial stand structure. Invasive vines and shrubs may be a problem in some areas after thinnings. Depending on the level of infestation by invasive species, mechanical or chemical control may be required.

The results of this management will be an infrastructure zone that is dominated by shorter right-tree-right-place trees and shrubs, and a side zone with widely spaced large trees in the overstory, younger trees of multiple age and size classes growing in the midstory, and native shrubs in the understory. In the wire zone, long-term pruning costs will be reduced by removing many side zone trees that would otherwise have encroached into the infrastructure zone. This more active, holistic management approach focuses on developing storm-resistant forests along utility line corridors and will reduce damage during the next extreme storm. This approach requires a more expansive vision of the roadside forest and utility line corridor management that looks at reducing immediate risks and at long-term individual-tree care and whole-forest care. For maximum effectiveness, this management regime should be combined with education and outreach to landowners to increase public awareness and achieve a safer, more reliable utility and transportation system.

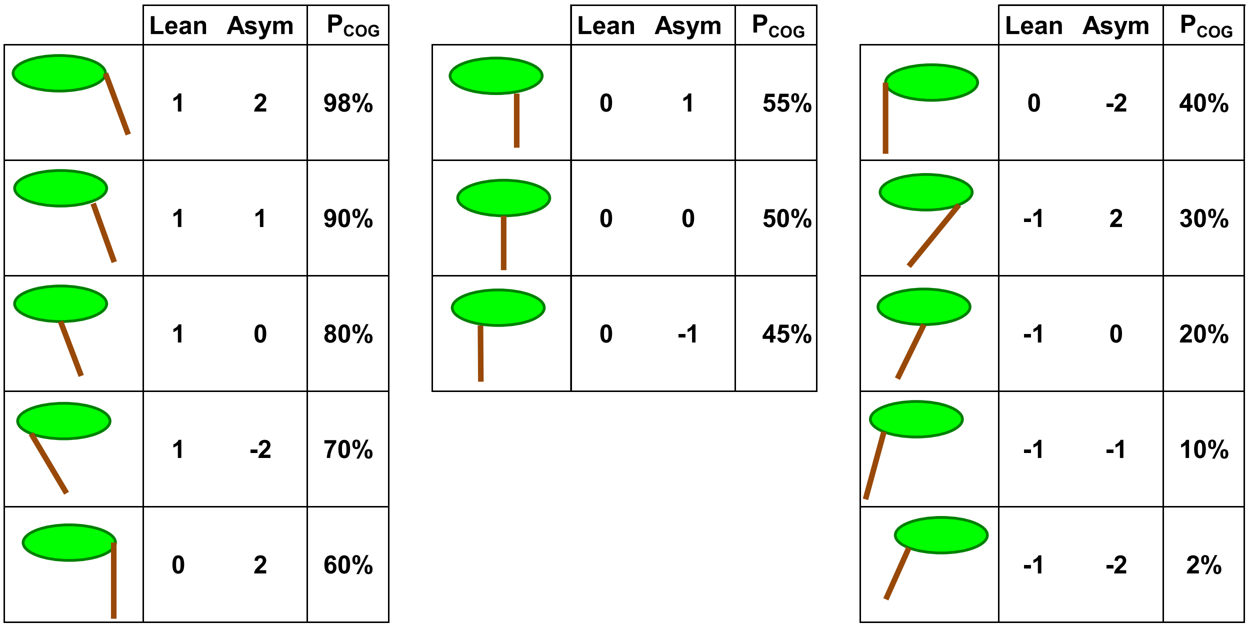
METHODS

Seven study areas were established in Connecticut roadside forests in 2014 and 2015 (Table 1). Once harvesting is completed in 2016, three of the study areas will be part of commercial harvest operations, two areas not scheduled for adjacent-stand management will be harvested to produce lumber in small volumes for the state park system, and one area will be a modification of scheduled highway vegetation management.

All stems with diameters >12.5 cm within 40 m of infrastructure edge (mowed road shoulder, centerline between utility poles if no road) were sampled along transects 200-600 m long (Fig. 2). Site constraints limited sampling to 25 m from forest edge at the Fairfield County site, which was a 50 m wide forested strip between limited access highway lanes. A number and diameter at breast height (d.b.h.) mark were painted on all trees before mapping to the nearest 0.1 m and recording the species, crown class, and diameter.

Table 1.—Description of roadside study areas established in Connecticut

County	Plot size (m)	Road type	Electric service
Fairfield	600 × 25	Limited access highway	None
Hartford	200 × 40	State highway	Backbone (3-wire)
Litchfield	200 × 40	State highway	Backbone (3-wire)
Middlesex	200 × 40	Town road	Backbone (3-wire)
New Haven	400 × 40	State highway	Backbone (3-wire)
New London	400 × 40	State highway	Backbone (3-wire)
Tolland	250 × 40	Town road	Lateral (1-wire)



Lean	Asym	P_{COG}
1	2	98%
1	1	90%
1	0	80%
1	-2	70%
0	2	60%
0	1	55%
0	0	50%
0	-1	45%
0	-2	40%
-1	2	30%
-1	0	20%
-1	-1	10%
-1	-2	2%

Figure 3.—Center of gravity weighting factor (P_{COG}) estimates using lean—whether a tree bole is leaning toward infrastructure (to left) and asymmetry (asym)—the position of the crown relative to the root collar.

The following data were collected for all trees within 20 m of utility poles and all trees with the potential to fall into the utility corridor: total height; pulpwood height (to 10 cm diameter); degree of root, stem, and crown damage; branch union type; and cracks. Trees were considered to have major defects if they had at least one of the following conditions: roots with advanced decay in or near root collar, a cavity at least one-third of tree diameter, more than one-third of canopy dead, codominant leaders with included bark, or large cracks with decay.

To examine the likelihood that a tree would fall toward the utility wires or road, information was collected about bole lean and crown asymmetry toward or away from the utility wires. Crown asymmetry was assigned one of five classes: all to road, two-thirds to road, centered over bole, one-third to road, or all away from road (Fig. 3). Because approximately 70 percent of a tree's biomass was found in the merchantable saw log (Swank and Reynolds 1986), boles leaning more than 7.5° toward utility poles were considered to have a positive center of gravity (COG). Conversely, those leaning more than 7.5° away from the road were considered to have a negative COG. Trees with neutral bole COGs were considered to have positive COGs if the asymmetry of the crowns was at least two-thirds toward the road.

At all sites, work was planned in coordination with a utility arborist, and trees within reach of the roadside or power line were identified for trimming, removal, or no treatment. Roadside trees (within the wire zone) of commercial size designated for removal were topped by the utility tree crew and the main trunk or bole was left standing. The remaining trees and roadside boles designated for harvesting were dropped using applied directional-felling techniques. Wood removal was by skidder or forwarder at locations where harvesting was under way in the adjacent stand. At locations where only the treatment site was harvested (small area, small volume), scale-appropriate equipment was used for wood removal and value-added operations. Equipment combinations included a rapid transit vehicle with an arch, a small four-wheel drive tractor with a three-point hitch-mounted winch, and a tractor with an arch. A portable band saw mill was incorporated for value-added product recovery at two locations.

ANALYSIS

To examine how density changed with distance from utility poles, we placed distances at 2 m intervals between 0 and 10 m (0-2, 2-4 … and 8-10 m from poles) and 4-m intervals thereafter (10-14, 14-18 …). Density and basal area values were standardized per hectare for ease of comparison.

Repeated measures with analysis of variance for density were used with distance from poles as the "within subject" factors (SYSTAT software, San Jose, CA). Because the number of within subject factors must be less than the number of replicates for repeated measures analysis, we first examined whether density varied 18-40 m from utility poles. Finding no difference (see Results), we examined whether density/basal area varied within 24 m of poles using the 2- to 4-, 6- to 8-, 10- to 14-, 14- to 18-, 18- to 22-, and 22- to 26-m intervals. This kept the number of within subject factors to less than the number of replicates. The 0- to 2-m interval was not used because few had 2 m of pole centers. Reported p-values are those after applying the conservative Greenhouse-Geisser Epsilon correction for deviations from compound symmetry (SYSTAT Software, San Jose, CA; 2009). Differences were considered significant at $p < 0.05$.

Because of limited funding, actual heights were measured on only 2,879 of the 3,743 trees included in the study. Heights were measured on at least 50 trees of each species at each study area or of all trees of a given species if a site had fewer than 50. These data were used to develop local allometric equations, HT = a + ln(d.b.h.), that were used to estimate the total and pulpwood height of the remaining trees.

The probability (P_{infra}) that a tree would strike infrastructure (utility wire, road pavement) if it fell randomly is

$$P_{infra} = \frac{cos^{-1}\left(\frac{D}{H}\right)}{\text{À}}, \qquad (1)$$

given a tree of height (H) at distance (D) from the infrastructure (derivation in Appendix). Because the smaller branches in the upper canopy are less likely to break utility wires or to impede traffic if they fall on the road, we used the pulpwood height rather than total height for calculating P_{infra}. We also calculated P_{risk} by weighting P_{infra} with a factor (P_{COG}) relative to whether a tree's COG was positive, neutral, or negative (Fig. 3). For simplicity, all calculations assume level terrain and homogeneous soil conditions.

RESULTS

A total of 3,743 trees were included on the seven study areas (Table 2). Stand types included oak (Middlesex, Tolland), oak-red maple (Hartford), red maple (New Haven), sugar maple-black cherry (Litchfield), and pine-oak (New London). All stands were fully stocked with a basal area of 24.5-36.3 m^2/ha. The mean height of measured upper canopy trees was 17.9-25.4 m across stands.

Not unexpectedly, density of both poletimber and sawtimber was very low immediately adjacent to the utility corridor because most trees that interfered with wires were removed to maintain service continuity (Fig. 4). For both size classes, but especially for poletimber, there was an apparent peak of density at approximately 10 m from utility pole centers and then a gradual decrease to densities consistent with the surrounding forest.

Table 2.—Summary of stand characteristics of roadside study areas established in Connecticut. N = number of trees in each study area.

County	N	Oak	Basal area distribution				m²/ha
			Maple	Birch	Other	White pine	
Fairfield	953	43%	19%	18%	13%	7%	26.4
Hartford	336	40%	39%	16%	6%	0%	29.9
Litchfield	382	2%	50%	3%	35%	10%	36.3
Middlesex	269	55%	26%	6%	13%	0%	24.5
New Haven	630	8%	70%	5%	14%	2%	27.3
New London	813	25%	18%	0%	0%	57%	35.7
Tolland	360	72%	19%	4%	5%	1%	25.1
Combined	3,743	31%	34%	7%	11%	16%	26.4

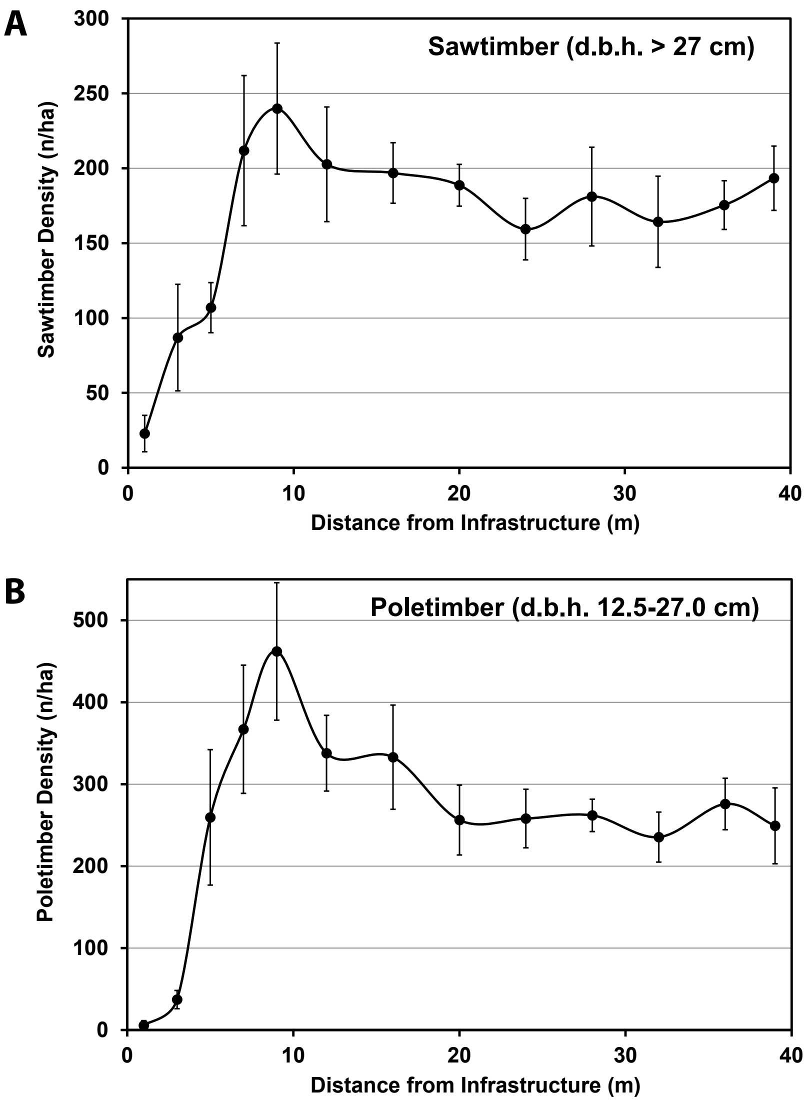


Figure 4.—Density (n/ha) of sawtimber (A) and poletimber (B) trees by distance to infrastructure (e.g., utility wires, roads).

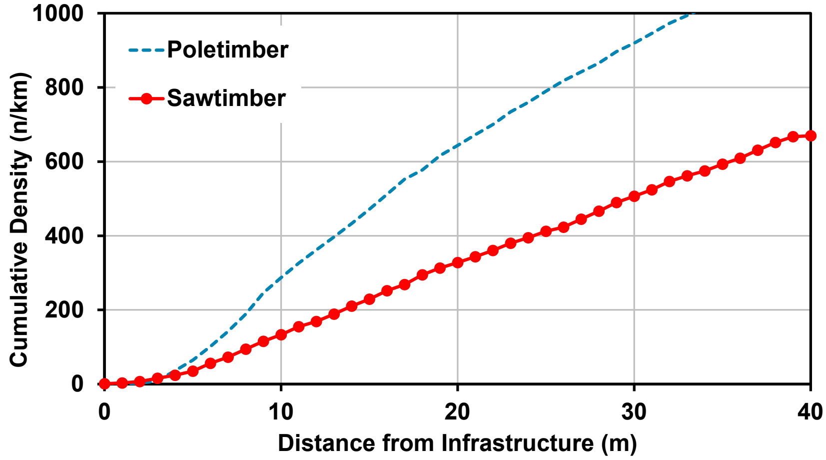


Figure 5.—Cumulative number of trees by size class and distance from infrastructure (e.g., utility wires, roads).

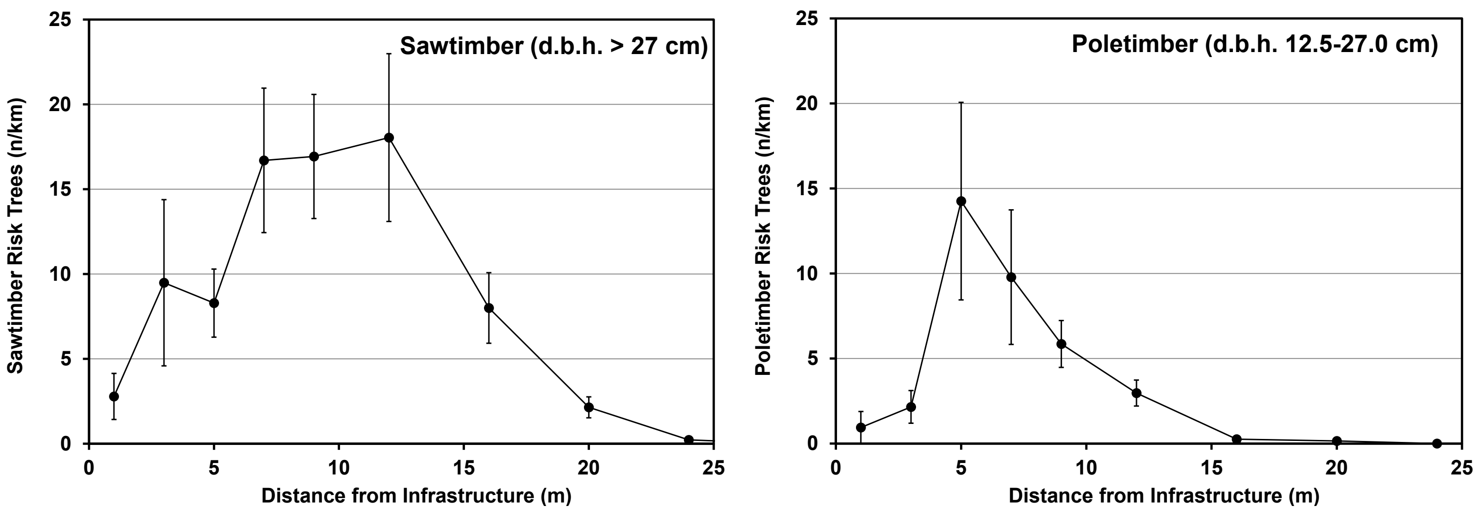


Figure 6.—Estimated number of risk sawtimber trees (left) and poletimber trees (right) that will eventually fall on infrastructure (e.g., on utility wires or across the road). Risk trees are those that will have a branch with a diameter of 10 cm or larger that would strike infrastructure when it falls.

Repeated measures with analysis of variance for density indicated a significant distance effect for poletimber density at distances less than 26 m from utility poles ($F_{5,30}$ = 9.71, P_{GG} = 0.003, where P_{GG} is after Greenhouse-Geisser Epsilon correction), but not for poletimber density in the intervals 18-40 m from utility poles ($F_{4,20}$ = 1.36, P_{GG} = 0.295). For sawtimber density there was only a weak effect at distances less than 26 m from utility poles ($F_{5,30}$ = 2.39, P_{GG} = 0.105) and none for 18-40 m from utility poles ($F_{4,20}$ = 0.53, P_{GG} = 0.612).

From a utility or road maintenance viewpoint, the number of trees per kilometer of forest/corridor interface (Fig. 5) is more pertinent than density per area. There were 313 sawtimber and 616 poletimber trees/km within 20 m of the utility corridors. We estimated that a surprisingly high number of sawtimber (83 ± 17/km) and poletimber (36 ± 12/km) risk trees would fall into the utility or travel corridor if they were windthrown (Fig. 6). Most of these risk trees are outside the utility protection zone and outside zones cleared along travel corridors, except for limited access highways.

DISCUSSION

Utility disruption and road obstruction caused by falling branches or windthrown trees are all too common along forested roadsides during extreme weather events (wind, ice, or snow). Except for removal of trees identified as being very high risk because of advanced decay or severe lean, the traditional reactive approach has been to deal with each tree after it has failed and has fallen onto utility wires or across roads.

Currently tree crews and arborists who work for public utilities in many states are the de facto managers of the roadside forest. In Connecticut, the Public Utility Regulatory Authority limits public utility tree crew activity to a clearly defined utility protection zone. With the concurrence of the abutting property owner, any and all branches may be trimmed within the utility protection zone as high as the equipment can reach. This activity, called *enhanced tree trimming*, addresses the bulk of branch failure issues directly above power lines that might occur during heavy wet snow or ice storms. Our study suggests, however, that potential damage resulting from wind is a broader forest management issue that demands a combination of arboricultural and silvicultural solutions to alter the conditions in roadside stands, as opposed to simply trimming roadside trees.

Although the reactive approach spreads the cost out over time, it often results in power outages, blocked roads limiting emergency access by first responders and the public, and the occasional tragedy when a falling tree strikes a moving vehicle. During the extreme storms that happen every several decades, extended power outages and road blockage caused by falling trees can last for days or weeks and result in economic losses that can reach into the billions of dollars (SVMTF 2012). In addition, because the focus is on removing material to restore power and open travel corridors, the potential economic value of the timber is at best converted to that of firewood.

Wood product volume and value recovery vary dramatically from location to location, and data collection on wood product volume remains incomplete. Two general observations are (1) some wood product volume can be recovered from trees removed at the immediate roadside; however, the value is insufficient to cover the cost of removal by arboricultural tree crews; and (2) at some locations where roadside tree volume has been combined with wood product volume yield from 30-m deep roadside stand management, the implementation cost may be largely offset by the value of products recovered. Much depends on initial timber quality, available markets, harvesting methods used, and opportunities for value-added operations.

Our estimate of nearly 120 risk trees/km (approximately 1 tree/8 m) that could be windthrown into the utility or travel corridor during extreme storms is likely low because it assumes that the trees will not grow taller or form branches with larger diameters. The perception that the number of risk trees is low is likely due to some of these trees dying and breaking apart in situ and because others are removed from wires as they fail over a period of several decades or longer. As Dave Goodson, manager of Vegetation Management at United Illuminating (CT), has repeatedly said, however, "gravity wins" and these many trees will eventually have to be dealt with. The choice the landowners, foresters, utility companies, and highway vegetation managers have to make is whether to deal with the problem and the expense reactively or proactively.

The consequences of neglecting roadside forests will only increase as trees become taller and more massive. This neglect has the potential to affect thousands to millions of people during extreme storms and makes the case that long-term solutions need to go beyond simple utility tree trimming and removal of obvious roadside hazard trees. Addressing this neglect will require adaptive management using innovative combinations of silvicultural and arboricultural techniques. Our study used a variety of operational practices. In some cases, harvesting was accomplished opportunistically as part of a commercial management operation that was already planned. In other cases, the treatment was deliberately planned and executed using small-scale equipment and methods that were appropriate to the small area and volume of potential wood-product material involved. The diverse nature of roadside forest conditions and ownership patterns demands that we examine an assortment of methods, logistics, skill-set combinations, and economic considerations to determine the optimum approaches to roadside forest management implementation. Further study is under way to examine these questions, as well as those associated with community attitudes, landowner opinions, and labor-force requirements. We plan to publish information from these studies in the future.

Like the Firewise program, which encourages local solutions for wildfire safety by involving homeowners, community leaders, planners, and others in an effort to protect people and property from wildfire risk,[2] Stormwise is designed to promote healthy, storm-resistant forests. The proactive Stormwise approach is a collaborative, coordinated effort by landowners, foresters, utility companies, and highway vegetation managers to manage the oft-neglected wooded strip between the point where shoulder mowing and tree pruning stops and the point where logging can be safely accomplished. This forest-road/utility interface is a surprisingly high proportion of the forest resource and is perhaps the most visible canvas of forest management.

We expect that forest owners and utility companies implementing Stormwise strategies will enjoy long-term benefits of fewer trees falling on wires during extreme weather and longer time spans between maintenance and management activities. Ultimately, storm-resistant roadside forests, once established, will require a different type of management and maintenance than the current regime of periodic tree-trimming and fallen tree removal. Rather than a 4- to 5-year trimming cycle, the Stormwise roadside forest will require some management activity every 10-15 years. And rather than being conducted from a bucket truck, much of the management activity would be at the ground level, cutting trees before they grow tall enough to fall on wires and maintaining the appropriate species mixture, density, and age structure in the roadside forest to allow it to remain wind firm.

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[2] See http://www.firewise.org/ for more information.

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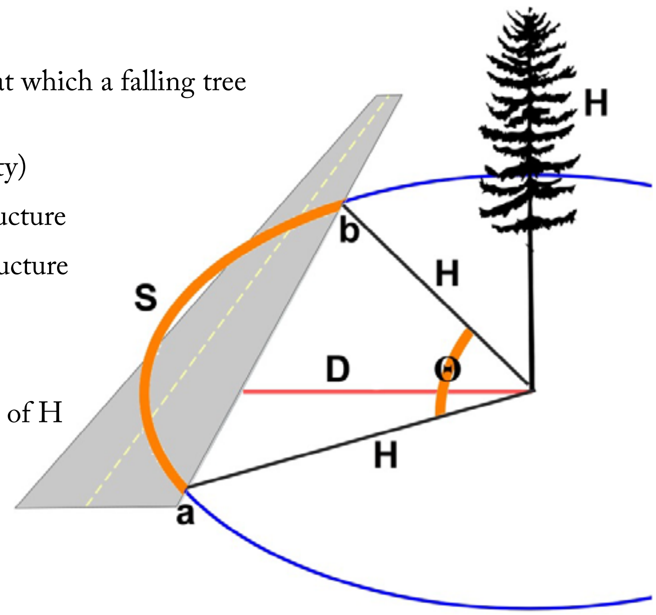
APPENDIX



H = tree height = radius of circle describing limit at which a falling tree damages infrastructure

D = distance of tree to infrastructure (road or utility)

a = point 1 at which falling tree intercepts infrastructure

b = point 2 at which falling tree intercepts infrastructure

S = length of arc ab

θ = angle of arc ab

C = perimeter length of circle with a radius length of H

From mensuration formulas

$C = 2\pi * H$

$S = H * \theta$

$\theta = 2 * \arccos(D/H)$

The probability that the tree falls within area prescribed by arc ab (P_{infra}) is

$P_{infra} = S / C$

$S/C = (H * \theta) / (2\pi * H)$

$= \theta/2\pi$

$= [2 * \arccos(D/H)]/(2\pi)$

$= \arccos(D/H)/\pi.$

The content of this paper reflects the views of the author(s), who are responsible for the facts and accuracy of the information presented herein.

FIRE DISTURBANCE

AN EVALUATION OF HARDWOOD FUEL MODELS FOR PLANNING PRESCRIBED FIRES IN OAK SHELTERWOOD STANDS

Patrick H. Brose[1]

Abstract.—The shelterwood burn technique is becoming more accepted and used as a means of regenerating eastern mixed-oak (*Quercus* spp.) forests on productive upland sites. Preparation is important to successfully implement this method; part of that preparation is selecting the proper fuel model (FM) for the prescribed fire. Because of the mix of leaf litter and logging slash, FMs 6, 8, 9, 10, and 11 may be appropriate representations of oak shelterwood stands for planning prescribed fires. This study compares fire behavior in oak shelterwood stands to BehavePlus-generated predictions for these FMs. BehavePlus most accurately predicted flame lengths and rates of spread with FM 11. When FM 6 was used, BehavePlus overestimated fire behavior. With FMs 8 and 9, BehavePlus consistently underestimated fire behavior, and with FM 10 BehavePlus produced overestimations and underestimations with no discernible pattern. When planning prescribed fires in oak-dominated shelterwood stands, resource managers should use either FM 11 to predict average fire behavior or FMs 6 and 9 to predict maximum and minimum fire behaviors.

INTRODUCTION

Throughout the eastern United States, resource managers increasingly use prescribed fire to regenerate, restore, and sustain upland oak (*Quercus* spp.) forests (Dickinson 2006, Hutchinson 2009, Van Lear and Brose 2002, Yaussy 2000). A popular regeneration method to use when oak seedlings are dense is to proceed with stand renewal via the shelterwood-burn technique (Brose et al. 1999, Van Lear and Brose 2002). This technique entails integrating a moderate- to high-intensity prescribed fire between the first and final removal cuts of a two-step shelterwood sequence (Brose et al.1999, Van Lear and Brose 2002). Because of the harvest – burn – harvest order of this method, identifying the fuel model (FM) that accurately portrays an oak shelterwood stand is necessary to plan a safe and successful prescribed fire.

An FM is a numerical description of the herbaceous and woody material available for burning in fire-prone ecosystems. These models provide fuel data for fire behavior prediction systems (Pyne et al. 1996). FMs contain parameters for loading the four fuel diameter classes (Fosberg 1970), height of the fuel bed, surface area to volume ratio, and moisture of extinction threshold. Rothermel (1972) developed the first 11 FMs (3 grass, 2 shrub, 3 timber, and 3 logging slash) for use in his fire spread model, and Albini (1976) added 2 additional shrub models. Anderson (1982) published a photo series describing these 13 FMs in detail. That series is widely used to help select the appropriate model for many common fire-prone ecosystems. Scott and Burgan (2005) developed 40 more FMs to supplement the original 13 FMs. These new FMs cover unrepresented fuel types and include dynamic moisture content for live herbaceous vegetation.

One purpose of FMs is to use them, in conjunction with topographic and weather data, in fire behavior prediction systems such as BEHAVE (Andrews 1986, Andrews and Chase 1989) and FARSITE (Finney 1998). These computer-based systems generate estimates of several fire-line characteristics (flame length [FL], rate of spread [ROS], and heat energy output) that are critical

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to successfully implementing a prescribed fire or safely suppressing a wildfire. These fire behavior prediction systems can also be used to determine which FM is the most accurate representation of a fire-prone ecosystem. Testing of the common hardwood FMs in this manner is limited.

Grabner et al. (2001) used Behave to generate ROS predictions for FMs 1 (short grass), 2 (timber and grass), 3 (tall grass), and 9 (loose leaf litter) in oak savannas in Missouri. They found that Behave generated reasonable ROS estimates for all FMs at wind speeds slower than 10 feet per minute, but at faster wind speeds Behave accurately predicted ROS for FM 2 only. In North Carolina, Phillips et al. (2006) used FARSITE to determine which FM portrayed mountain laurel (*Kalmia latifolia*), an ericaceous shrub that forms thickets in mixed-oak forests of the Appalachian Mountains. They found that none of the shrub FMs were viable representatives of mountain laurel thickets and concluded that a custom FM needed to be developed for that plant community.

Oak shelterwoods are another environment that may not be adequately characterized by any of the hardwood FMs, because two major fuel types are present: leaf litter and dry logging slash. This study analyzes data from Brose (1997) using BehavePlus (Andrews et al. 2005) to determine which of the 13 standard FMs (Anderson 1982) is the most accurate representation of oak shelterwood stands undergoing regeneration via the shelterwood-burn technique. I limited my pool of potential FMs to the original 13 because these are widely used by wildland fire professionals. I also focused only on FMs that are either commonly used to represent hardwood environments or were relatable to partly cut stands. Selected FMs include FM 6 (hardwood slash), FM 8 (compacted leaf litter), FM 9 (loose leaf litter), FM 10 (litter and timber), and FM 11 (light logging slash). Finally, I focused on FL and ROS because these are of utmost importance to wildland fire professionals in the region. Knowing which FM to use when planning prescribed fires in oak shelterwoods will help resource managers more safely and successfully use the shelterwood-burn technique when regenerating mixed-oak forests on productive upland sites.

METHODS

Study Site

The field portion of this study was conducted at the Horsepen Lake Wildlife Management Area in the Piedmont of central Virginia. This area consists of broad, gently rolling hills on sandy loam soils (Reber 1988). The climate is warm continental with 50 inches of precipitation distributed evenly throughout the year and an average growing season of 190 days. The area is presently owned and managed by the Virginia Department of Game and Inland Fisheries (VDGIF).

In 1994, three oak shelterwood stands on productive upland soils (oak site index$_{50}$ ranged from 70 to 80 feet) were selected for an oak regeneration study (Brose 1997). The VDGIF created these shelterwoods between 1990 and 1992 by salvaging trees that had been injured or killed by an ice storm in 1989. Harvesting reduced basal areas to an average of 50 square feet per acre in each shelterwood. Slash was left in place after the harvest. Each shelterwood was 20-50 acres and was dominated (67 percent of residual basal area) by four upland oak species: black (*Q. velutina*), northern red (*Q. rubra*), scarlet (*Q. coccinea*), and white (*Q. alba*). Other hardwoods present in these stands included American beech (*Fagus grandifolia*), blackgum (*Nyssa sylvatica*), flowering dogwood (*Cornus florida*), hickory (*Carya* spp.), red maple (*Acer rubrum*), and yellow-poplar (*Liriodendron tulipifera*).

Table 1.—Site attributes of the nine oak shelterwood burn units in central Virginia and the dates, times, and preburn weather conditions of the nine prescribed fires. These data were used in the BehavePlus program to predict flame length and rate of spread for each of the five hardwood fuel models examined in this study.

Attribute/condition	Prescription fire 1	Prescription fire 2	Prescription fire 3
Slope (%)	10	5	5
Aspect	NE	E	E
Burn date	25 Feb 1995	27 Feb 1995	27 Feb 1995
Time of burn	1300	1100	1430
Air temperature (°F)	46	43	48
Relative humidity (%)	26	52	44
Wind direction	NW	E	E
Wind speed (mph)	4	1	2
Cloud cover (%)	0	100	100
	Prescription Fire 4	Prescription Fire 5	Prescription Fire 6
Slope (%)	7	5	10
Aspect	E	SW	E
Burn date	26 Apr 1995	26 Apr 1995	26 Apr 1995
Time of burn	2000	1630	1830
Air temperature (°F)	68	73	70
Relative humidity (%)	28	20	20
Wind direction	SW	SW	SW
Wind speed (mph)	1	5	3
Cloud cover (%)	0	0	0
	Prescription Fire 7	Prescription Fire 8	Prescription Fire 9
Slope (%)	3	10	5
Aspect	NE	W	NW
Burn date	24 Aug 1995	24 Aug 1995	24 Aug 1995
Time of burn	1630	1430	1230
Air temperature (°F)	92	95	95
Relative humidity (%)	46	44	46
Wind direction	SW	SW	SW
Wind speed (mph)	0	5	8
Cloud cover (%)	0	0	0

Fuel Sampling Procedures

Each shelterwood stand was divided into four equally sized units; three units were designated for burning in different seasons (spring, summer, and winter). Near the center of each burn unit, a 100-foot × 150-foot plot, visually judged to represent average fuel conditions, was located for measuring the flaming front. The slope and aspect of each burn unit were determined with a clinometer and compass, respectively (Table 1).

Within each fire behavior plot, 11 fuel inventory transects were systematically installed to uniformly sample the area. Each transect was 50 feet long and was inventoried for woody fuels using the planar-intersect method (Brown 1974). Sound woody fuels (those not in an advanced state of decay) were tallied by the time-lag size classes of Fosberg (1970). One-hour fuels were smaller than 0.25 inch in diameter; 10-hour fuels were 0.25 to 1 inch in diameter. These two size classes were tallied along the proximal 6-foot section of each transect. Hundred-hour fuels were 1-3 inches in diameter and were tallied along the distal 12-foot section of each transect. Fuels larger than 3-inch diameter and rotten woody fuels of all sizes were ignored because these are not used in the standard FMs (Anderson 1982). For each transect, the woody fuel counts in each time-lag class were converted to tons per acre using established fuel equations (Brown 1974) and hardwood-specific gravities (Anderson 1978). For each fire behavior plot, a mean fuel loading for each time lag class was calculated by summing the appropriate loading of each transect and dividing by the number of transects.

Litter loading was measured by collecting two 1.36-square-foot samples per transect of the Oi and Oe horizons. Samples were collected near the midpoint and distal end of each transect. The samples included the leaf litter, dead herbaceous plants, and woody material smaller than 1 inch in diameter found among the leaves. Live herbaceous matter was excluded because it is not part of the selected FMs (Anderson 1982). Samples were dried at 122 °F for 72 hours then weighed on an electronic scale to the nearest 0.1 ounce. The masses of the two leaf litter samples from each transect were averaged, and the average was converted to tons per acre. Transects in the winter and spring burn units were inventoried for litter fuels and woody fuels in February 1995; those in the summer burn units were inventoried in August 1995.

Several weeks before each prescribed fire, I placed a 10-hour fuel moisture stick in each fire behavior plot so it could acclimate to the environmental conditions. Within 1 hour before each prescribed fire, the fuel moisture stick was weighed with a hand scale to determine the percent moisture content. One-hour fuel moisture content was calculated from the 10-hour fuel moisture content using conversion tables in the Fireline Handbook (National Wildfire Coordinating Group 1989). Fuel moisture for the 100-hour fuels was measured using a moisture probe by sampling twelve 100-hour fuels suspended within 1 foot of the ground.

The Prescribed Fires

VDGIF personnel conducted each prescribed fire in accordance with department policy and state law. The fires occurred on February 25 and 27 (winter), April 26 (spring), and August 24 (summer), 1995. Weather was monitored with a belt weather kit. Air temperature, cloud cover, relative humidity, wind direction, and wind speed were recorded at the beginning of each prescribed fire (Table 1).

All prescribed fires were lighted using drip torches in a strip head-fire ignition pattern beginning at the downwind side of the burn block. Initial strips were 5-10 feet apart, but these widened to 150 feet once the control lines were secured. As ignitors neared the fire behavior plot, a strip was lighted 20-30 feet outside the plot on the downhill or upwind side and allowed to burn through the plot. FLs were measured (in feet) by photographing the flaming front as it passed five trees previously marked with paint at 1-foot intervals to a height of 5 feet (Rothermel and Deeming 1980). ROS was calculated by marking, timing, and measuring five 2-minute runs with a stopwatch. The five FL measurements were combined to calculate a mean FL for each prescribed fire. Similarly, a mean ROS was calculated for each prescribed fire based on the five ROS measurements of that burn.

Statistical Analysis

For the initial comparison of FMs 6, 8, 9, 10, and 11 to the oak shelterwoods, I used T-tests with unequal variances (Ott 1993) to test whether published fuel loadings and fuel-bed depths (Anderson 1982) were different from the mean fuel loadings and fuel-bed depths found in the oak shelterwoods. I further compared the five FMs by using analytical techniques developed by Smith and Rose (1995) and Sneeuwjagt and Frandsen (1977). I used BehavePlus (Andrews et al. 2005) to predict the average FL and ROS for each of the nine prescribed fires based on each of the five FMs and the weather conditions, site characteristics, and fuel moistures recorded just before each prescribed fire. The nine predicted FL and ROS outputs of each FM were then regressed on the mean FL and ROS measurements of the nine prescribed fires. Scatter plots were used to graphically represent the deviation of the BehavePlus-generated FL and ROS predictions for each FM from the actual measured values. Perfect agreement between predicted and measured FL and ROS values resulted in a diagonal line with an intercept of zero (0) and a slope of one (1). Deviation from this optimal line showed where a FM overestimated or underestimated FL and ROS.

I used regression on the observed and predicted FL and ROS values to determine their linear equations and coefficients of variation (r^2). This value provides a measure of consistency; a higher r^2 indicates more consistency among the predicted outputs than a lower r^2 (Smith and Rose 1995, Sneeuwjagt and Frandsen 1977). Finally, I calculated the mean relative difference between the observed and predicted FL and ROS for each FM (Smith and Rose 1995, Sneeuwjagt and Frandsen 1977). This metric quantifies how much an overestimation or underestimation differs from the observed on a percent scale. It is calculated by subtracting the predicted FL or ROS from the observed FL or ROS, dividing that difference by the observed FL or ROS, then multiplying that quotient by 100.

RESULTS

The nine burn units were quite similar to each other (Table 2). Each had a complete covering of leaf litter that contributed 2.1-3.3 tons per acre to the total fuel loading. The summer burn units had leaf litter loadings that were approximately 25 percent lower than those of the winter and spring burn units. Each burn unit had 80-85 percent of its woody fuels in the 100-hour size class. Total fuel loadings for the nine burn units were 9.9-16.0 tons per acre. Fuel moisture content varied among the nine burn units from 5-20 percent.

The fuels data from the nine burn units were combined to calculate the mean fuel loadings and fuel-bed depth for an oak shelterwood, and those means were compared to fuel characteristic values of the five FMs (Table 3). The oak shelterwood had a fuel-bed depth of 0.9 feet. This was shallower than that of FM 6 (2.5 feet), deeper than that of FMs 8 and 9 (0.2 feet for each), and the same as that of FMs 10 and 11 (1.0 foot each). The oak shelterwood contained 10.84 tons per acre of litter and woody fuels. The fuel loading was concentrated in the 100-hour and 1-hour size classes, which averaged 6.74 and 2.98 tons per acre, respectively. The balance, 1.12 tons per acre, was in the 10-hour size class. In the 1-hour size class, FMs 9 (2.92 tons per acre) and 10 (3.01 tons per acre) did not differ from the oak shelterwood mean of 2.98 tons per acre; the other three FMs had considerably lighter loadings at 1.50 tons per acre for each. In the 10-hour size class, FM 9 had lower loading (0.41 tons per acre) and FMs 6, 10, and 11 had higher loading (2 to 4.51 tons per acre) than oak shelterwood mean of 1.12 tons per acre. FM 8, at 1 ton per acre, did not differ from the oak shelterwood. In the 100-hour size class, all FMs had loadings that were lower than the 6.74 tons per acre calculated for the oak shelterwood, but only those of FMs 6, 8, and 9 were statistically different.

Table 2.—Fuel loadings and moistures of the nine oak shelterwood burn units in central Virginia

Fuel or site attribute	Prescription fire 1	Prescription fire 2	Prescription fire 3
Litter cover (%)	100	100	100
Leaf litter (tons/acre)	3.3 ± 0.3	3.3 ± 0.3	3.1 ± 0.3
1-hour fuels (tons/acre)	0.3 ± 0.1	0.4 ± 0.1	0.4 ± 0.1
10-hour fuels (tons/acre)	0.6 ± 0.1	1.7 ± 0.2	0.8 ± 0.2
100-hour fuels (tons/acre)	4.7 ± 0.8	7.7 ± 1.3	5.2 ± 1.0
Total fuels (tons/acre)	9.9	14.1	10.5
1-hour fuel moisture (%)	6	12	12
10-hour fuel moisture (%)	10	15	15
100-hour fuel moisture (%)	18	20	20
	Prescription fire 4	Prescription fire 5	Prescription fire 6
Litter cover (%)	100	100	100
Leaf litter (tons/acre)	3.1 ± 0.3	3.3 ± 0.3	3 ± 0.3
1-hour fuels (tons/acre)	0.4 ± 0.1	0.6 ± 0.1	0.5 ± 0.1
10-hour fuels (tons/acre)	0.7 ± 0.1	1.7 ± 0.2	0.8 ± 0.2
100-hour fuels (tons/acre)	5.2 ± 0.8	9.4 ± 1.3	6 ± 1
Total fuels (tons/acre)	10.4	16	11.3
1-hour fuel moisture (%)	9	5	7
10-hour fuel moisture (%)	10	10	10
100-hour fuel moisture (%)	12	12	12
	Prescription fire 7	Prescription fire 8	Prescription fire 9
Litter cover (%)	100	100	100
Leaf litter (tons/acre)	2.4 ± 0.3	2.6 ± 0.3	2.1 ± 0.3
1-hour fuels (tons/acre)	0.4 ± 0.1	0.2 ± 0.1	0.4 ± 0.1
10-hour fuels (tons/acre)	0.7 ± 0.1	2.2 ± 0.2	0.9 ± 0.2
100-hour fuels (tons/acre)	6.4 ± 0.8	9.3 ± 1.3	6.8 ± 1
Total fuels (tons/acre)	9.9	14.3	10.2
1-hour fuel moisture (%)	11	9	9
10-hour fuel moisture (%)	10	15	15
100-hour fuel moisture (%)	14	14	14

Table 3.—Fuel-bed characteristics of oak shelterwoods (mean ± 1 standard error) in central Virginia, along with potentially suitable hardwood fuel models. Fuel model values marked by an asterisk (*) are different from the corresponding oak shelterwood mean at the 0.05 level.

	Fuel loading in tons/acre				
Fuel model	1-hour	10-hour	100-hour	Total	Fuel-bed depth (feet)
Oak shelterwood	2.98 ± 0.32	1.12 ± 0.19	6.74 ± 1.58	10.84 ± 0.77	0.9 ± 0.2
6 (dormant brush)	1.50*	2.50*	2.00*	6.00*	2.5*
8 (compact litter)	1.50*	1.00	2.50*	5.00*	0.2*
9 (loose litter)	2.92	0.41*	0.15*	3.48*	0.2*
10 (timber and litter)	3.01	2.00*	5.01	10.02	1.0
11 (light slash)	1.50*	4.51*	5.51	11.52	1.0

Table 4.—Linear equations and associated r^2 values created by regressing the BehavePlus-generated flame length (FL) and rate of spread (ROS) predictions for the five hardwood fuel models on the measured fire behavior during the nine prescribed fires. Over/under refers to BehavePlus's overprediction (+) or underprediction (–) of the FL and ROS relative to the measured fire behavior during the prescribed fires.

Fuel model number and description	FL equation	FL r^2	Over/under	ROS equation	ROS r^2	Over/under
06 (hardwood slash)	1.151 + 0.823x	0.791	+32%	1.857 + 1.378x	0.695	+50%
08 (compacted litter)	0.141 + 0.188x	0.799	–281%	0.206 + 0.123x	0.757	–420%
09 (loose litter)	0.706 + 0.271x	0.737	–52%	0.346 + 0.453x	0.917	–72%
10 (litter and timber)	1.611 + 0.711x	0.405	+34%	1.100 + 0.707x	0.430	–22%
11 (light slash)	0.625 + 0.851x	0.745	+15%	0.200 + 0.839x	0.892	–18%

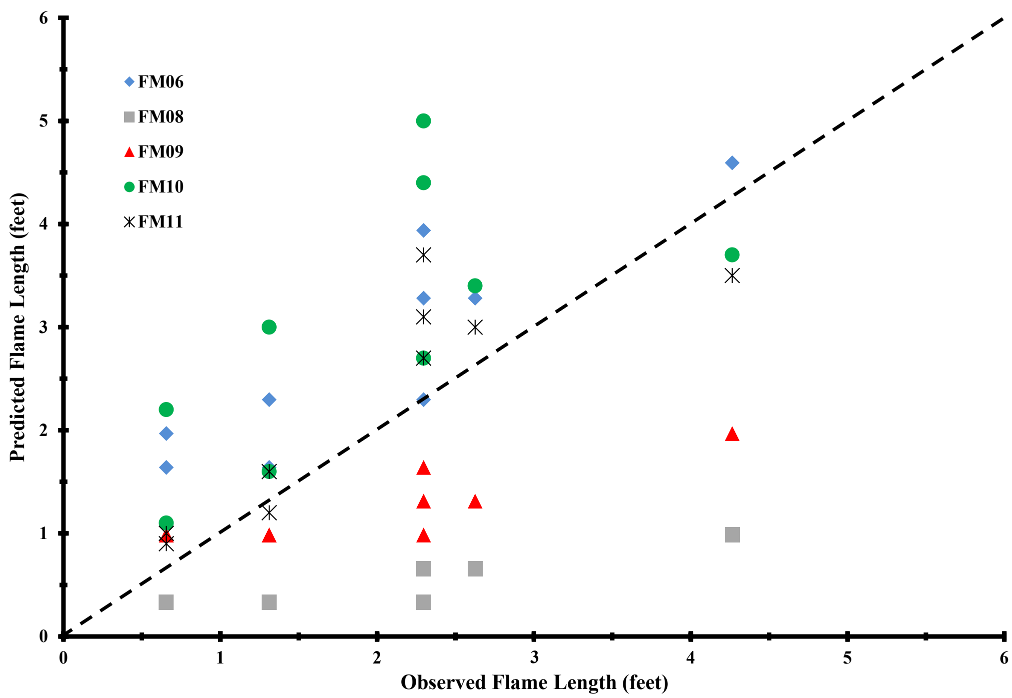

Figure 1.—The predicted flame lengths of fuel models 6, 8, 9, 10, and 11 regressed onto the flame lengths observed in nine prescribed fires conducted in oak shelterwood stands in central Virginia. The black diagonal line marks perfect agreement of the observed and predicted flame lengths. Fuel model predictions of flame length above and below this line are overestimations and underestimations, respectively.

Of the five FMs, FM 10 was the most similar to the oak shelterwood. It differed only in the 10-hour fuel loading; 2.0 tons per acre compared to 1.12 tons per acre for the oak shelterwood. FM 11 was also quite similar to the oak shelterwood; it differed in the 1-hour size class at 1.50 versus 2.98 tons per acre, and in the 10-hour size class at 4.51 versus 1.12 tons per acre. FM 6 was the most dissimilar fuel model relative to the oak shelterwood. It differed in loading in all fuel size classes, in total fuel loading, and in fuel-bed depth. FM 8 agreed with the oak shelterwood only in terms of 10-hour fuel loading; FM 9 agreed only in terms of the 1-hour fuel loading.

Regressing the BehavePlus-generated FL prediction derived from each FM onto the mean observed FL of each prescribed fire showed that BehavePlus outputs for FM 11 most closely match those of the oak shelterwood (Fig. 1, Table 4). For FM 11, BehavePlus overestimated FL by an average of 15 percent. Furthermore, the FL predictions consistently tracked changes in observed FL as evidenced by an r^2 value of 0.745. BehavePlus also overestimated FL when

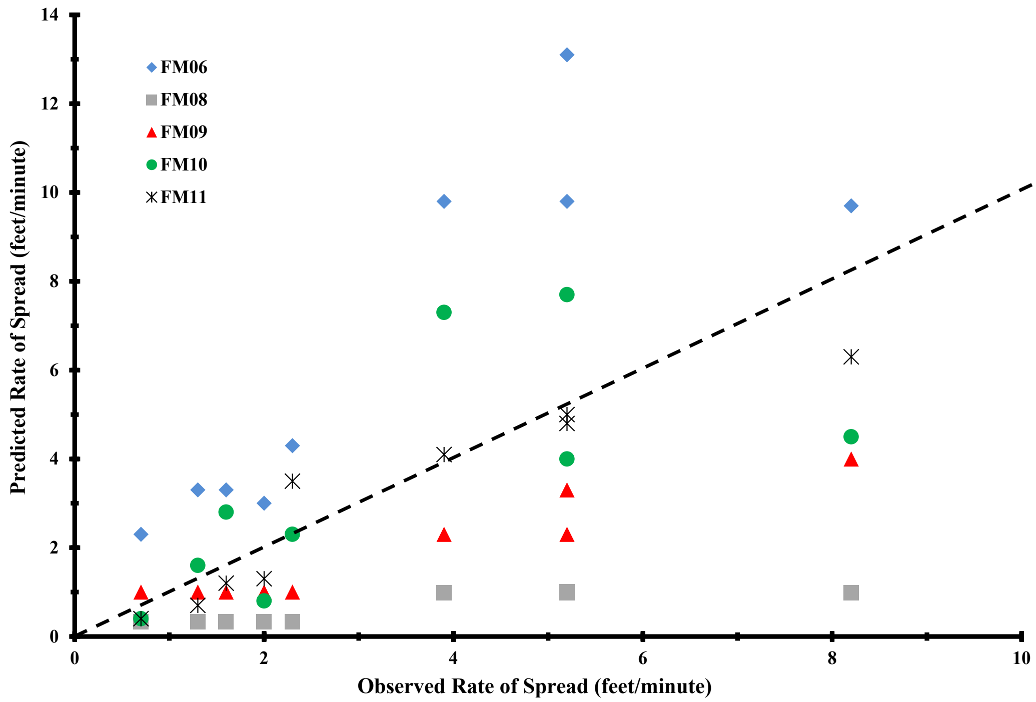

Figure 2.—The predicted rates of spread of fuel models 6, 8, 9, 10, and 11 regressed onto the rates of spread observed in nine prescribed fires conducted in oak shelterwood stands in central Virginia. The black diagonal line marks perfect agreement of the observed and predicted rates of spread. Fuel model predictions of rate of spread above and below this line are overestimations and underestimations, respectively.

using FMs 6 and 10, by averages of 32 percent and 34 percent, respectively. Of these two FMs, FL estimates based on FM 6 were more consistent than those of FM 10 (r^2 of 0.791 and 0.405, respectively). Conversely, BehavePlus underestimated FL by an average of 52 percent when using FM 9 and by approximately 280 percent when using FM 8. The predictions became especially erroneous for both when the actual FL exceeded 2 feet. The consistency of the BehavePlus FL predictions using FMs 8 and 9 was quite strong with respective r^2 values of 0.737 and 0.799.

Like FL, BehavePlus estimates of ROS from FM 11 most closely matched those measured in the oak shelterwood burn units (Fig. 2, Table 4). ROS predictions derived from FM 11 were, on average, underestimated by 18 percent. These predictions were also strongly consistent with an r^2 value of 0.892. BehavePlus produced underestimates of ROS when using FM 8 and 9; those of FM 9 were more accurate than those of FM 8 at -72 percent versus -420 percent, respectively. Both sets of predictions were strongly consistent, however, with an r^2 of 0.757 for FM 8 and 0.917 for FM 9. When using FM 6, BehavePlus consistently overestimated ROS by 2-8 feet per minute relative to those measured in the oak shelterwood burn units. This resulted in an average overestimation of 50 percent and an r^2 of 0.695. BehavePlus estimates of ROS were inconsistent when using FM 10. Three times BehavePlus estimates of ROS were well matched to those of the oak shelterwood, but three times it overestimated ROS and three times it underestimated ROS. Consequently, the underestimation was only 22 percent, but the r^2 value was 0.43.

DISCUSSION

Of the 13 standard FMs, five (FMs 6, 8, 9, 10, and 11) have attributes that make them potentially suitable for modeling fire behavior in oak shelterwoods on productive upland sites. Of these five, FM 11 (light slash) appears to best represent the fuel conditions that occur in an oak shelterwood based on fuel-bed characteristics and agreement between BehavePlus estimates of FL and ROS and measured fire behavior. FM 11's 100-hour fuel loading, total fuel loading, and fuel-bed depth were quite similar to those found in an oak shelterwood. The oak shelterwood had more 1-hour fuel loading and less 10-hour fuel loading than FM 11. These are likely due to the combining the leaf litter loading with the 1-hour fuel loading and the fact that the oak shelterwoods were 2-4 years old and some of the small branches of the logging slash would have decayed. The agreement of the observed and predicted FLs and ROSs further indicate the appropriateness of FM 11 as a representation of an oak shelterwood. On average, FL was overestimated by 15 percent and ROS was underestimated by 18 percent. The r^2 values for both parameters (0.745 for FL and 0.892 for ROS) indicate that the predictions were quite consistent. The FM 11 predictions closely matched the measurements in the fire behavior plots.

FM 6 represents dormant shrubs but is also used for hardwood slash (Anderson 1982). In this regard it is a potential FM for oak shelterwoods. In 17 of 18 comparisons, BehavePlus overestimated FL by 32 percent and ROS by 50 percent when using FM 6, even though it has only about half the total FL of the oak shelterwoods (6.00 tons per acre versus 10.84 tons per acre). For FL, these overestimations were fairly consistent across all nine prescribed fires (r^2 = 0.791), and the differences between observed ROS and predicted ROS tended to increase as observed ROS increased (r^2 = 0.695). These results are likely due to the differences in fuel-bed depth and orientation between FM 6 and the oak shelterwoods. FM 6 has a vertically oriented 2.5-foot deep fuel bed (Anderson 1982); an oak shelterwood has a horizontally oriented < 1-foot deep fuel bed. Fires burn hotter and move faster through vertically oriented fuel beds than through horizontally oriented fuel beds (Pyne et al. 1996). Consequently, BehavePlus produced overestimations for both fire behavior parameters when using FM 6.

FM 10 (litter and timber) represents a broad range of forest conditions (insect- or disease-ridden stands, windthrown stands, overmature stands with deadfall, and aged partial harvests) that have accumulated 100-hour and larger fuels in their understories (Anderson 1982). This FM appears to be a good representation of oak shelterwoods because the published fuel-bed characteristics are quite similar to those measured in the oak shelterwoods. For example, FM 10 has 3 tons per acre of 1-hour fuels, 5 tons per acre of 100-hour fuels, 10 tons per acre of total fuels, and a fuel-bed depth of 1 foot. The oak shelterwoods averaged 2.98, 6.74, 10.84, and 0.90 feet, respectively, for these same fuel-bed attributes. BEHAVE-generated predictions of FLs and ROSs for FM 10, however, did not match up well with observed fire behavior in the oak shelterwoods. In four of the nine comparisons, BehavePlus substantially overestimated FLs when using FM 10, and its ROS predictions had no discernible trend (three overestimations, three underestimations, and three close matches). These inconsistencies are apparent in the r^2 values: 0.405 for FL and 0.430 for ROS. The disagreement between FM 10 and the oak shelterwoods in terms of FL may result from their differences in the 10-hour fuel loading; 2.00 tons per acre for FM 10 and 1.12 tons per acre for the shelterwood. Similarly, the differences in ROS between the two may be due to FM 10 containing a live fuel component of 2 tons per acre (Anderson 1982). No such live fuel component was present in the oak shelterwoods.

FMs 8 and 9 represent litter-dominated fuel beds (Anderson 1982). Generally, FM 8 is associated with forest types such as northern hardwoods whose leaves form a compact mat on the forest floor. FM 9 refers to mixed-oak forests whose leaves create and maintain a loose litter bed. This difference in litter composition and arrangement is manifested in their 1-hour fuel loadings: FM 8 has 1.50 tons per acre and FM 9 has 2.92 tons per acre. Also in play is how well they compare to the observed fire behavior in the oak shelterwoods. When using either fuel model, BehavePlus underestimated FL and ROS relative to what was measured, but its predictions based on FM 8 were closely accurate at only the lowest FLs and ROSs (< 1 foot and < 1 foot per min, respectively). As FL and ROS increased, BehavePlus predictions based on FM 8 became increasingly inaccurate resulting in mean relative differences of -281 percent and -420 percent. When using FM 9, BehavePlus estimated FLs and ROSs that were consistently lower than what was measured (-52 percent and -72 percent), but not to the degree of FM 8. These estimations are low because these two FMs lack the woody fuel component in the 10-hour and 100-hour size classes, and the fuel-bed depth is less than 25 percent of what was measured.

Forest managers who intend to use the shelterwood-burn technique in mixed-oak stands should use FM 11 to represent fuel-bed conditions. Of the five FMs examined in this study, BehavePlus most closely predicted FLs and ROSs when using FM 11 and did so consistently in all nine prescribed fires. Conversely, FM 8 should not be used. It is a poor match for an oak shelterwood as evidenced by the model's chronic and increasing underestimation of FL and ROS as both of these fire behavior parameters increased. Similarly, FM 10 should be avoided when modeling anticipated fire behavior in oak shelterwoods. The FL and ROS estimations generated by BehavePlus when using FM 10 are either too high (FL) or too inconsistent (ROS) to be usable.

FM 6 and 9 can be used in preparing for prescribed fires in oak shelterwoods if they are used together. BehavePlus consistently predicted high FLs and ROSs when using FM 6 and low FLs and ROSs when using FM 9. But, if used in tandem, they create an upper and lower bound of probable fire behavior. Some forest managers may find that knowing the minimum and maximum expected fire behaviors is more valuable in prescribed fire planning than knowing the average estimated fire behavior generated by FM 11.

This study has several limitations. The results are most usable in oak shelterwood stands similar to the ones used in this study with upland sandy loam soils, oak site $index_{50}$ of 70-80 feet, 50-percent basal area reduction 2-4 years earlier, and comparable weather and site conditions. They may be applicable outside these conditions, but with caution. The reported FL and ROS data from the prescribed fires were coarse because of the simplistic methodologies for obtaining them. Similarly, the sample size was small, with only nine prescribed fires, and the average observed FL and ROS of each prescribed fire were based on only five observations per fire. Consequently, the ranges of observed FLs and ROSs were limited to less than 4 feet for FL and 8 feet per minute for ROS, and the fire behavior prediction trends of FMs 6, 9, and 11 may not be as accurate at higher FLs and faster ROSs. Repeating and enlarging this study, employing more precise fire-behavior-measuring methodologies, and broadening the sites and conditions would address these limitations and refine the conclusions of this paper.

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TREE-QUALITY IMPACTS ASSOCIATED WITH USE OF THE SHELTERWOOD-FIRE TECHNIQUE IN A CENTRAL APPALACHIAN FOREST

Janice K. Wiedenbeck, John P. Brown, Thomas M. Schuler, and Melissa Thomas-Van Gundy[1]

Abstract.—Wounding from prescribed fires and forest harvest operations creates concerns about the future health, grade, volume, and value recovery potential of affected trees. The wounds, regardless of origin, may compartmentalize and heal over. Or they may be slower to heal or too significant to defend against pathogens that invade the wound zone and promote decay formation and spread. Even tree species that are good at compartmentalization after being wounded can succumb after a series of wounding events. We often create this scenario when conducting prescribed fires in conjunction with thinning and regeneration operations. A combination prescribed fire–shelterwood treatment study to evaluate oak regeneration (*Quercus* spp.) and establishment in a mesic mixed-oak forest was conducted in 2000 in West Virginia. Before and after each of two prescribed fires that were intended to eliminate a shade-tolerant understory, a shelterwood harvest to open the canopy to promote oak regeneration, and a subsequent prescribed fire designed to further cull less fire tolerant non-oak species, tree-quality conditions were evaluated for all stems 5-inch diameter at breast height and larger. The initiation and development of wounds and broken tops were tracked and correlated with silvicultural activities and weather events. The cumulative and interaction effects of repeated mechanical stressors on these stems are significant factors in long-term research that seeks to determine the costs and benefits of prescribed fire treatments to promote oak regeneration.

INTRODUCTION

Many forestry, wildlife, and ecology professionals accept the use of prescribed fires in eastern hardwood forest management. Resource managers use fire to shape regeneration, habitat, and ecological restoration outcomes and to reduce fuel buildup to lower the risk of wildfires. Managers of numerous national forests in the eastern region have revised their planning documents to include fire as a restoration tool (Nowacki et al. 2009). In the Appalachian region, a post-shelterwood prescribed fire treatment has been cited as another means of promoting oak regeneration (Brose et al. 1999). Eastern hardwood tree mortality caused by fire has been evaluated in multiple studies. Factors assessed in these fire-caused mortality studies include fuel types and loadings (Brose and Van Lear 1999, Wendel and Smith 1986, Yaussy and Waldrop 2010); bark thickness (Harmon 1984, Yaussy and Waldrop 2010); tree diameter (Harmon 1984, Hutchinson et al. 2005, McCarthy and Sims 1935); season in which fire occurred (Brose and Van Lear 1999); fire severity (Regelbrugge and Smith 1994, Yaussy and Waldrop 2010); and tree vigor before fire exposure (Yaussy and Waldrop 2010). Fire damage severity based on the degree of bole damage and the crown condition has also been widely studied (Brose and Van Lear 1999, Pomp et al. 2008, Wendel and Smith 1986). The process of wound formation after fire injury has been studied by Smith and Sutherland (2001, 2006) and Sutherland and Smith (2000).

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These studies, though, lack repeated measurements over time of the effects of the prescribed fire on tree condition. Only one study evaluated the effect of a post-shelterwood establishment prescribed fire on the change in bole condition after the fire (Brose and Van Lear 1999). After a prescribed fire conducted in the spring, the proportion of the undamaged oak boles dropped from 73 percent to 50 percent and the proportion of dead oaks increased by 18 percent. More than 90 percent of the oaks that died or showed severe bole damage had accumulated logging slash at their bases (Brose and Van Lear 1999). Their study incorporated a cumulative effects component (shelterwood cut + fire) but only evaluated the preburn and postburn tree conditions at a single point in time.

OBJECTIVE

The main objective of this study was to assess over an extended period (15 years) the cumulative and interaction effects of repeated mechanical stressors on trees of the Central Appalachian broadleaf forest. The hypotheses under investigation were as follows:

H_1: The quality condition of trees does not change between treatments over time.

H_2: The quality condition of trees does not change between treatments.

H_3: The quality condition of trees does not change over time.

METHODOLOGY

A study of the combined effects of prescribed fire and shelterwood harvest on oak regeneration has been under way in the Fernow Experimental Forest (Schuler et al. 2013) since 2000. The research site is in the Canoe Run watershed of the Fernow Experimental Forest (39.03°N, 79.67°W) in West Virginia. The elevation of the study site is 1,920-2,200 feet with a western aspect and a mean slope of 39 percent. It is described as a mixed mesophytic site with overstory dominated by northern red oak (*Quercus rubra* L.), chestnut oak (*Q. prinus* L.), and white oak (*Q. alba* L.) (Schuler et al. 2013). The study site is a second-growth forest that is about 100 years old.

Two prescribed fire treatments have been applied to the study site. The first was conducted in April 2002 and 2003—in 2002 the prescribed fire was interrupted because of bad weather, so the second half of the site was not burned until the following April. The second fire treatment was conducted in April 2005. The maximum temperature probe readings recorded during these two burns were 576 °F in 2002 and 621 °F in 2005, and the associated rates of spread were 30 feet per minute and 144 feet per minute, respectively (Schuler et al. 2013). These prescribed fires were characterized as "moderate to low intensity with flame lengths shorter than 3 feet that resulted from the combustion of leaf litter and 1-hour surface fuels" (Schuler et al. 2013, p. 432).

During winter 2009-2010, a shelterwood cut removed 50 percent of the overstory trees in 20 of the 24 plots (each plot is 0.5 acres). The other four plots were untreated reference (referred to as *control*) plots on which neither prescribed fires nor shelterwood operations were conducted. The residual "leave" tree basal area was 45-50 square feet per acre and comprised largely oaks. Only trees of 11 inches diameter at breast height (d.b.h.) or larger were marked for cut.

In spring 2014 a third prescribed fire was applied to half (10) of the treated plots. The maximum probe reading recorded in this burn was higher than for the earlier burns (929 °F), and the mean maximum temperature recorded on the 48 thermocouples was 272 °F.

The 20 plots that were burned twice and received the shelterwood cut are referred to collectively as the *manipulative treatments.* Of these, the 10 plots burned again after the shelterwood cut in 2014 are referred to as the *post-shelterwood burn* treatment. The 10 plots burned before the shelterwood harvest only are referred to as the *no post-shelterwood burn* treatment. Figure 1 shows the treatment areas and plot layout.

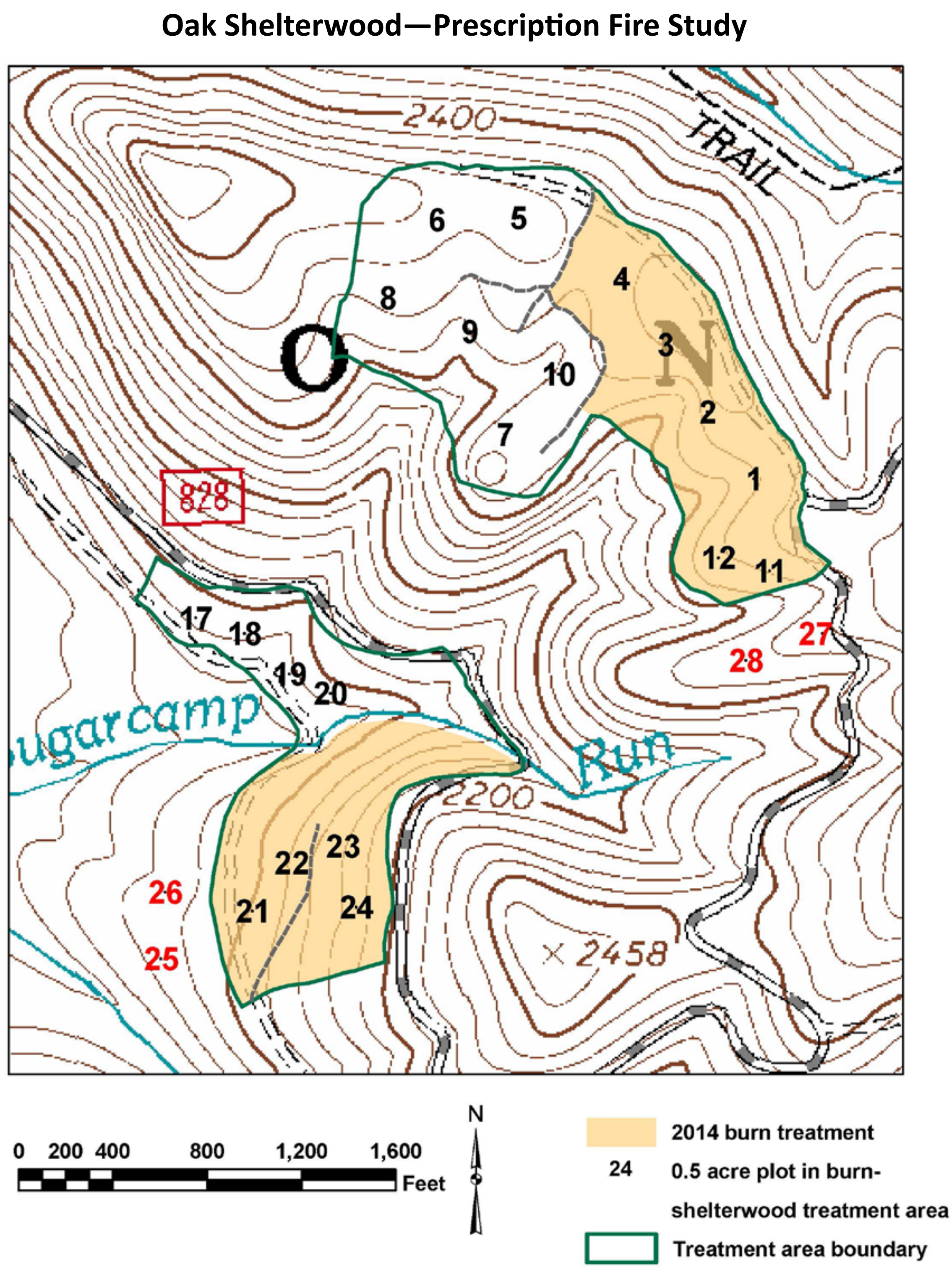

Figure 1.—Treatment map with plot locations.

The experimental unit is a 0.5-acre growth plot surrounded by a buffer that is approximately 100 feet wide. Within the 0.5-acre plot, all trees ≥5 inches d.b.h. were individually numbered, tagged, and tracked. On seven occasions since the study began, growth plot data have been collected (only four times for the four reference plots) for each tree. Each remeasurement included d.b.h., crown class (CC), and nonquantitative quality conditions. Quality assessments were binary—the given characteristic was either present or not. Live tree characteristics conditions included (1) stem skinned, (2) top broken, (3) bent/lean, and (4) vines in crown. Trees that died since the previous measurement cycle were assigned a cause of death of (1) unknown, (2) cut, (3) destroyed, or (4) fire killed. For the live tree assessments, multiple characteristics could be present. For the dead tree assessments, only one cause of death was assigned. The same team of forestry technicians conducted all of the assessments.

In addition to the silvicultural treatments, Superstorm Sandy caused varying amounts of tree damage throughout the forests of the region in 2012. This early-season storm (October 30) dumped large amounts of wet, heavy snow over a multicounty region of West Virginia, Maryland, and southwest Pennsylvania at a time when many trees had not yet dropped their leaves. The storm resulted in significant tree damage and woody debris. This damage is partially reflected in a comparison of the May 2010 and May 2013 growth plot measurements.

Statistical Methodology

To begin, one-way analysis of variance was performed on the plot-level data from 1999 (before treatments were applied to the stands) to ascertain if the number of trees per plot and mean d.b.h. of all trees 5 inches and larger was the same among plots assigned to treatments. Because the number of trees per plot (plot size = 0.5 acres) varied by treatment at the beginning of the study, stand density index (SDI) was used as a stand density measure and was treated as a covariate in the analysis. CC (dominant, codominant, intermediate, suppressed) (Smith et al. 1997) was included as an explanatory variable in the models.

Given the repeated measures over time, factors affecting quality conditions were tested using logistic regression mixed models in SAS (PROC GLIMMIX) (SAS Institute, Cary, NC). Several possible covariance structures were selected to account for the repeated measures: variance components, compound symmetry, spatial exponential (SP(EXP)), spatial power (SP(POW)), and spatial Gaussian (SP(GAU)). Significance levels were set at $\alpha = 0.05$. The interaction term treatment*time was included in the model, and the remaining terms were tested as main effects. The corrected Akaike information criterion (AICc) was used in model selection. Multiple comparisons (MCs) were conducted using the Tukey-Kramer adjustment. For the MC test of the treatment*time interaction, the continuous variable time was fixed at the four time points in the study where all treatments were measured. The significance level was Bonferroni adjusted to account for the four sets of MC tests. The mixed logistic model is of the form

$$g(p)=\boldsymbol{X\beta}+\boldsymbol{Z\gamma}+\boldsymbol{\varepsilon}$$

Where

$\boldsymbol{X}$ = a matrix consisting of values for treatment, time, stand density index, and crown class,

$\boldsymbol{\beta}$ = the vector of the unknown fixed effects parameters,

$\boldsymbol{Z}$ = the known design matrix for the random effects,

$\boldsymbol{\gamma}$ = the vector of the unknown random effects parameters,

p = the probability of the quality condition occurring,

$g(p)$ = the logit function= $\frac{p}{1-p}$, and

ε = the unobserved vector of random errors.

Plot averages for the treatments were calculated to gain insight into the statistical results. Tree survival (mortality) results are evaluated here using simple summary statistics to inform the discussion of tree condition. Survival analysis for trees among treatments will be evaluated in a subsequent paper.

RESULTS AND DISCUSSION

Tree Mortality

Figure 2 shows the percentage change in the number of live trees per plot at key points in this study. The baseline for each treatment is set to the mean number of trees present in 1999 (T_0). The percentage-based decline in live trees for both manipulative treatments is 13 percent after 7 years and two moderate- to low-intensity prescribed fires, notably more than for the control plots in which the number of 5-inch and larger stems declined only 4 percent (Fig. 2). In 2009, the decline in live tree stems in the manipulative treatment plots compared to the number present in 1999 was 20 percent; for the control plots the reduction was only 7 percent. These reductions in tree numbers occurred after two prescribed fires but before the shelterwood harvest, Superstorm Sandy, and post-shelterwood prescribed fire affected the plots.

Once a tree dies, it is removed from consideration in future surveys. As a result, the listed numbers of dead trees during each assessment are only those trees that died since the previous plot survey. In both 2003 and 2006 most trees that died on the treatment plots were killed by fire (73 percent in 2003 and 83 percent in 2006). In 2009, the tally that was conducted before the shelterwood operation found another 93 trees newly dead in the treatment plots, but the cause

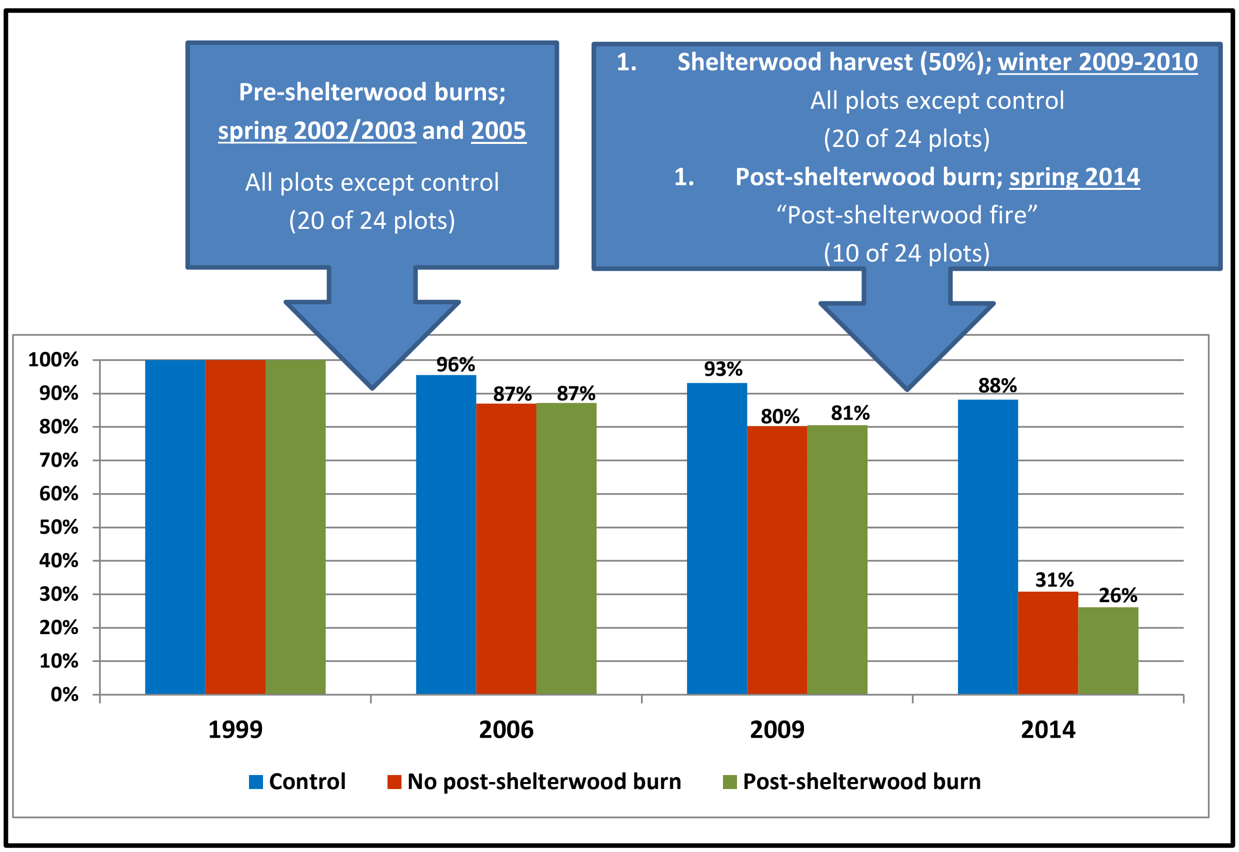

Figure 2.—Proportional change in the number of live trees greater than 5 inches d.b.h. from study installation in 1999 through post-shelterwood, post-prescribed fire treatments in 2014 by treatment.

Table 1.—Final models for all quality conditions

Effect	Number DF[c]	Denominator DF	F value	Pr > F
	Quality condition = stem skinned			
Treatment*time	4	3596	14.47	<0.0001
Treatment	4	3596	9.42	<0.0001
Time	1	3596	406.93	<0.0001
CC[a]	3	3596	25.09	<0.0001
	Quality condition = top broken			
Treatment	4	2878	7.51	<0.0001
Time	1	2878	159.93	<0.0001
CC	2	2878	41.49	<0.0001
	Quality condition = bent/lean			
Treatment	4	2799	3.14	0.0137
Time	1	2799	7.14	0.0076
CC	2	2799	18.96	<0.0001
	Quality condition = vines in crown			
Time	1	3576	50.14	<0.0001
SDI[b]	1	3576	19.42	<0.0001
CC	3	3576	2.84	0.0367
	Quality condition = fire damage			
Treatment	4	3215	12.4	<.0001
Time	1	3215	49.5	<.0001
SDI	1	3215	13.62	0.0002
CC	3	3215	24.85	<.0001

[a]CC = crown class
[b]SDI = stand density index
[c]DF= degrees of freedom

of death for these was not determined. Again in 2013, in the second survey after the shelterwood harvest, the manipulative treatment plots had a substantial number of newly dead trees (64) with unknown causes of death; the control plots had no newly dead trees. Tree mortality at this juncture can be attributed to the cumulative and delayed effects of the silvicultural manipulations in combination with higher vulnerability of these lower-density stands to Superstorm Sandy's snow and wind loads.

Statistical Results for Tree Condition

For all quality conditions, the AICc value for the full model was the lowest when using the variance components covariance structure. The full model consisted of the treatment*time interaction term, SDI, along with CC. For all models, the AICc value was lowest for the variance components covariance structure. Therefore, variance component was maintained as the covariance structure for testing the significance of the independent variables. Nonsignificant variables were dropped from the model, and the model was refit until all terms were significant (Table 1).

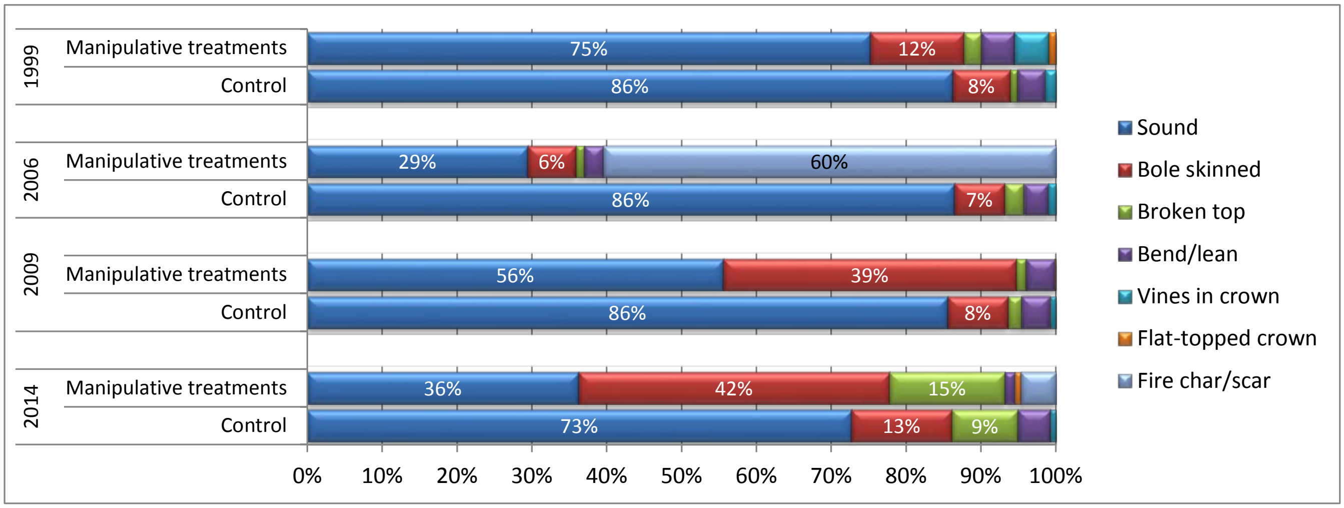

Figure 3.—Condition of live trees tallied at the time of study installation and three subsequent points in time: (A) 2006 after all plots except control plots subjected to two prescribed fires, (B) 2009 before shelterwood harvest on all but the control plots, and (C) 2014 after post-shelterwood prescribed fire on 10 of 24 plots.

MC tests were conducted for treatment. Fixed time points had to be selected to conduct the paired comparisons for the stem-skinned quality condition because the treatment*time term was significant (Table 1). Four years had measurements that included all treatments, so these four time points were used. In general, only the control treatment differed significantly from the others through time. Skinned stems were commonly associated with trees that showed significant fire scars or char in previous inventories of the plots that were treated with prescribed burns (Fig. 3).

Fixing time was not required for the MC tests for treatment for the top broken, bent/lean, and fire damage quality conditions because the interaction term was not significant. For the top broken condition, the control treatment differed significantly from the manipulative treatments. For the bent/lean condition and the fire damage condition, the MC test results were convoluted, suggesting that special contrasts may be needed to identify and explain the differences indicated by the models.

For CC comparisons, we observed a significant difference between codominant trees and any other CC for the stem-skinned condition (Table 2). For both the top broken and bent/lean conditions, codominants were significantly different (occurring less frequently) than intermediate and suppressed trees. No dominant trees exhibited these quality conditions. Only the intermediate versus suppressed tree comparison was different for the vines to crown quality condition. For fire damage, the codominant, intermediate, and suppressed classes all differed. The dominant class showed no differences between the others, but there were so few dominant trees that this is likely a sample size/statistical power issue.

Discussion of Tree Condition Results

When the study began in 1999, 75 percent of the trees tallied in the plots assigned to the manipulative treatments were rated as sound compared to 86 percent of the trees in the control plots (Fig. 3). The percentage of sound trees in the four control plots remained steady through the 2009 measurement cycle but dropped to 73 percent in the 2014 tally. The sound percentage of trees in the plots that were treated with two prescribed fires was only 29 percent in 2006 and rose back up to 56 percent in the 2009 assessment. In the most recent survey, which was after

Table 2.—Multiple paired comparisons for crown class for all quality conditions

Quality condition	Crown class	Paired comparison*
Stem skinned	Dominant	a,b
	Codominant	a
	Intermediate	b
	Suppressed	b
Top broken	Codominant	a
	Intermediate	b
	Suppressed	b
Bent/lean	Codominant	a
	Intermediate	b
	Suppressed	b
Vines in crown	Dominant	a,b
	Codominant	a,b
	Intermediate	a
	Suppressed	b
Fire damage	Dominant	a,b,c
	Codominant	a
	Intermediate	b
	Suppressed	c

*Same letters indicate no significant differences between crown classes.

the shelterwood harvest, Superstorm Sandy, and a post-shelterwood burn on 10 plots, the sound percentage of trees in the plots was 36 percent (Fig. 3).

The 20 treatment plots were subjected to two prescribed fires in the 2002–2005 time frame and had substantial amounts of charred bark on 60 percent of the living tree stems in 2006. This average varied among the 20 treatment plots from 20 percent to 93 percent (standard deviation = 18 percent). When evaluated again in 2009 after 4 years without any manipulative operations, none of the trees in either the control or treatment plots showed signs of remnant fire char. The incidence of skinned boles tallied in the 20 treatment plots rose significantly in the 2009 assessment, however ($\bar{x}$ = 39 percent, s = 11 percent). The skinned boles were the next phase in the progression of the fire damage as the charred bark sloughed off and a cat-face type wound began to form. Conversely, some of the trees that had large areas of black bark in 2006 did not lose bark or appear wounded in 2009 (Fig. 3).

The occurrence of broken tops in these plots was consistently lower than 5 percent during plot surveys conducted through 2009 on the control and treatment plots. The 2014 results (Fig. 3) indicate that the mean percentage of live trees with broken tops jumped to 15 percent on the treatment plots and to 9 percent on the four control plots. These data were collected after the shelterwood harvest and after Superstorm Sandy.

The increase in broken tops was investigated more closely by looking at the results from two growth plot surveys that were conducted on treatment plots in 2010 and 2013. This focused look at tree condition classes during the 2009-2014 time frame showed us that the plots on which the shelterwood harvest was conducted in winter 2009-2010 had an increase in the relative occurrence of broken tops in the survey conducted immediately after the harvest, changing

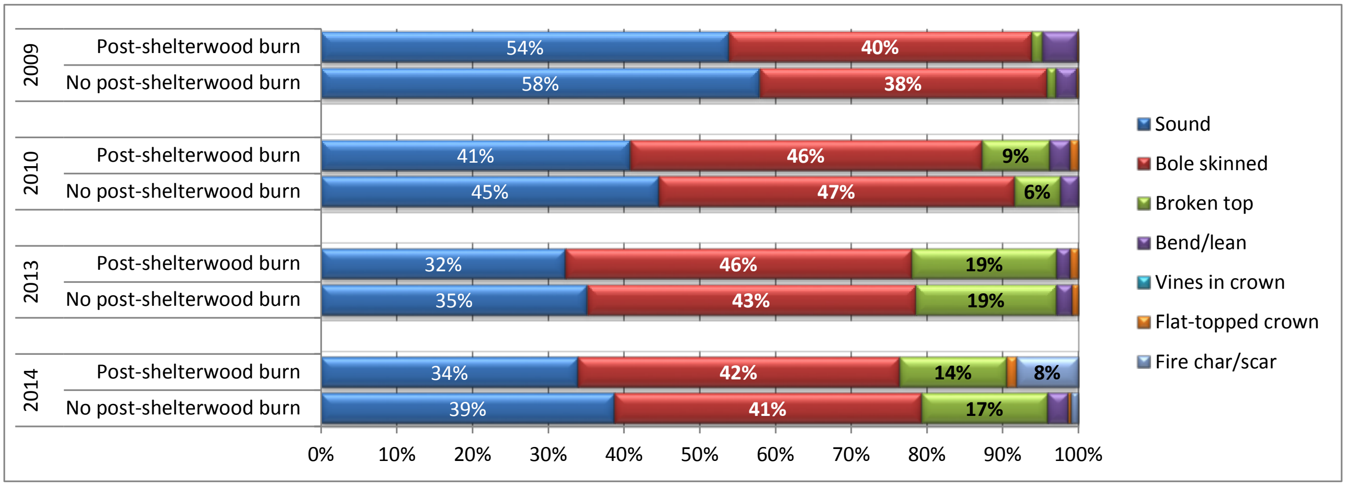

Figure 4.—Tree condition proportions for the two manipulative treatments based on all trees. Includes results of four plot assessments conducted between 2009 and 2014 including the 2010 and 2013 partial assessments (control plots not remeasured).

from 1 percent of trees before the harvest to almost 8 percent of trees after. The 2013 plot survey, however, again showed a large increase in the relative incidence of broken tops: the rate increased from 8 percent in 2010 to almost 19 percent 3 years later (Fig. 4). The opening of the canopy resulting from the shelterwood operation made a higher percentage of the trees in the manipulative plots more susceptible to the heavy snow and wind loads that occurred in 2012 during Superstorm Sandy.

The multiple comparisons conducted on CC appear to be illogical in that the dominant trees are grouped with the suppressed trees for the stem-skinned, vines in crown, and fire damage conditions (Table 2). Less than 1 percent of all trees during each tally period were classed as dominant, however. The codominant MC results compared to the intermediate and suppressed results indicate that skinned stems, broken tops, tree lean, and fire damage affect intermediate and suppressed trees more than codominant trees. The prescribed fires and harvest damage from the shelterwood operation are affecting significant percentages of trees in the treatment plots, but the affected trees are understory trees. Forest managers seek to eliminate many of these intermediate and suppressed trees over time to improve oak regeneration.

The proportion of hardwood stems in a central Appalachian forest exhibiting skinned boles, broken tops, lean, vines in crown, and fire damage changed over 15 years, with treatment being a factor. Stem-skinned (bole) proportions were different among treatments and the differences varied over time, so hypothesis 1 (H1: the quality condition of trees does not change between treatments over time) is rejected for the stem-skinned condition class. The other quality conditions showed significant main effects for time and for treatment, so hypothesis 3 (H3: the quality condition of trees does not change over time) is rejected, and hypothesis 2 (H2: the quality condition of trees does not change between treatments) is rejected for all but the vines in crown quality condition class. Quality condition varied among treatments, with the reference/control treatment having lower proportions of the non-normal (less desirable) conditions noted than for the manipulative treatments that included prescribed fires and a shelterwood harvest operation.

Parsing out the cumulative effects of the combination of treatments—preharvest prescribed fires, a shelterwood harvest of 50 percent of overstory trees, and a postharvest prescribed fire—on the condition of residual trees is a multifaceted challenge. The prescribed fires and harvest damage from the shelterwood operation are affecting significant percentages of trees in the manipulative treatment plots, but these are not the favored dominant or codominant trees. Instead, the affected trees are predominantly shade-tolerant species in intermediate and suppressed CCs, which the shelterwood-prescribed fire management regime seeks to eliminate over time to maintain the dominance of oak. The proportion of trees with fire char and scar in the post-shelterwood fire treatment was low (8 percent) compared to the fire char and scar proportion found in 2006 (60 percent), even though the maximum temperatures recorded on the thermocouples were generally higher. This outcome supports the supposition by Brose and Van Lear (1999, p. 88) that although "fire may be intense due to the presence of slash and increased exposure to sunlight and wind …conversely, the open nature of the shelterwood stand coupled with the thick bark of a mature tree may limit damage."

Predictably, the opening of the canopy resulting from the shelterwood operation made a higher percentage of the trees in the manipulative plots more susceptible to the heavy snow and wind loads that occurred during Superstorm Sandy.

Tree mortality summaries indicate the need to further this investigation by conducting a survival analysis to learn more about the influence of silvicultural activities and tree condition over time on the survival of trees of different species having different CCs.

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ARTIFICIAL REGENERATION

ESTABLISHING NORTHERN RED OAK ON A DEGRADED UPLAND SITE IN NORTHEASTERN PENNSYLVANIA: INFLUENCE OF SEEDLING PEDIGREE AND QUALITY

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Abstract.—Enrichment plantings using large oak seedlings of regional sources may promote superior survival and growth compared to direct seeding or standard nursery seedling material. This study evaluated the survival and growth of planted 1-0 northern red oak (*Quercus rubra* L.) seedlings among 11 families and 3 seedling size classes (small, average, and premium). Seedlings were planted in April 2005 in a deer exclosure in a failed clearcut in Pike County, PA. After 7 years, survival averaged 84 percent and was greater in the premium than either the small or average size classes in 2 of the 11 families. Total height and ground-level diameter of premium seedlings were greater in five and eight of the families, respectively, than seedlings in either the average or small size classes. Results suggest that selecting premium seedlings from certain mother trees can improve planted seedling survival and growth and have ramifications for seed orchard construction.

INTRODUCTION

The reduction in oak regeneration throughout much of the eastern United States can be attributed to changes to disturbance regimes (Abrams 2003, Crow 1988), browsing by overpopulated deer herds (Rooney and Waller 2003), increased competition by fire-intolerant species (Abrams 1992), interference from invasive species such as hay-scented fern (*Dennstaedtia punctilobula* [Michx.] T. Moore) (Horsley 1988, McWilliams et al. 1995), and mortality from the nonnative gypsy moth (*Lymantria dispar dispar* L.; Kegg 1971). Advance regeneration of desirable timber and wildlife hardwood species, including oaks, is inadequate throughout most of Pennsylvania (McCaskill et al. 2013; McWilliams et al. 1995, 2007), and recruitment of oaks after harvest has been very poor (Marquis et al. 1976).

When natural regeneration of oaks is inadequate, enrichment planting can be a useful tool to produce desired levels of oak stocking. Success of oak plantings is a function of genetic factors, site quality, site conditions at the time of planting, competition, planting methods, and stock quality (Burdett 1990, Dey and Parker 1997, Dey et al. 2008, Kormanik et al. 1995). Indicators of high quality stock include number of 1st-order lateral roots, root collar diameter (RCD), root volume, and stem height (Dey and Parker 1997, Jacobs et al. 2005, Kormanik et al. 2002). High quality oak seedlings generally grow faster than smaller seedlings, can better compete with other vegetation, and can better survive damage and dieback. These advantages give large seedlings a better chance of surviving repeated browsing by white-tailed deer (*Odocoileus virginianus Miller*). Northern red oak (*Quercus rubra* L.) seedlings with RCDs larger than 8-10 mm (c.f. Dey and Parker 1997) and stem height taller than 50 cm (Johnson 1981) have been recommended for successful establishment on productive sites.

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Previous work to evaluate success of oak-enrichment plantings in forested settings has focused on establishment on highly productive sites (e.g., Kormanik et al. 2002, Morrissey et al. 2010) or sites in the South or Midwest (e.g., Kormanik et al. 2002, Morrissey et al. 2010, Schuler and Robison 2010, Spetich et al. 2000, Thompson and Schultz 1995). Highly productive sites, which generally have greater soil moisture and nutrient availability than xeric sites, have been particularly challenging for regenerating oak because of the abundance of faster-growing shade-intolerant species such as yelow-poplar (*Liriodendron tulipifera* L.) (Morrissey et al. 2010). Few studies have evaluated planted oak establishment with little or no preplanting site preparation on xeric or poor quality sites in the Northeast. Although oak is easier to regenerate on poor quality sites because severe competition is lacking, regeneration is still questionable if the area has a high deer population. The interaction between deer browsing and competing vegetation (e.g., hay-scented fern) limits the establishment and advancement of oak seedling cohorts (de la Cretaz and Kelty 2002). Deer exclosure fences and nonselective herbicide application to remove competing vegetation are effective management options for promoting oak regeneration. The cost of deer fencing is extremely high, however; woven wire fencing costs $2 or more per linear foot (Penn State Extension 2006) and requires fairly regular maintenance. Therefore, for most private landowners deer fencing lends itself to only small acreage.

Most studies evaluating success of oak-enrichment plantings focus on early survival and growth, rarely following the trees beyond the first 5 years. Here we present 7-year results of a long-term study to compare survival, height, and diameter of 1-0 northern red oak seedlings of three seedling size classes from 11 families. The objective is to understand the interaction among seedling genetics and seedling quality in an enrichment planting of northern red oak on a low-productivity upland site.

METHODS

Experimental Material

Seeds from 11 open-pollinated northern red oak mother trees were used in this study. Acorns were harvested from mother trees located in natural forested stands at the U.S. Military Academy reservation, West Point, NY, and proximal area in fall 2003. Mother trees were located at least 0.40 km apart to avoid collecting closely related material. The acorns were planted at the Georgia Forestry Commission's Flint River Nursery in Byromville, GA, in December 2003 at a density of 65 seeds/m^2. Fertilization and irrigation of the seedlings followed guidelines developed by Kormanik et al. (1994). The 1-0 seedlings were lifted in late January 2005 and transported to Knoxville, TN, where they were stored in a cold room (~1 °C). Total height and root collar diameter of each seedling were measured and seedlings were individually tagged. Finally, seedlings were visually sorted into three size classes within each family: small, average, and premium, according to height and RCD (Clark et al. 2000) (Table 1). Before starting the grading process, we chose several seedlings that appeared to represent small, average, and premium sizes for each family and used them as model seedlings for each size group by the planting crew (Clark et al. 2000). We used the minimum height (50 cm) (Johnson 1981) and RCD (8-10 mm) (Dey and Parker 1997) recommended for northern red oak seedlings as the standard for our average seedling size class. One family (family 1) was divided into average and premium size classes only, because small seedlings were lacking.

Table 1.—Height (± standard error) and ground-level diameter (g.l.d.) of seedlings at planting and after seven growing seasons. N total at planting was 759 seedlings.

Family	Quality	Initial height (cm)	7-year height (cm)	Initial g.l.d. (mm)	7-year g.l.d. (mm)
1*	Average	66 ± 4	163 ± 24	8.1 ± 0.4	24.4 ± 2.9 b
	Premium	89 ± 4	200 ± 22	10.8 ± 0.6	33.1 ± 2.6 a
6	Small	50 ± 6	177 ± 28	7.2 ± 0.4	27.0 ± 3.8
	Average	71 ± 5	221 ± 27	8.6 ± 0.4	30.8 ± 3.5
	Premium	68 ± 4	210 ± 22	10.1 ± 0.6	33.4 ± 2.6
7*,†	Small	54 ± 6	97 ± 27 b	6.9 ± 0.4	17.1 ± 3.5 b
	Average	79 ± 5	156 ± 27 b	8.9 ± 0.6	22.1 ± 3.4 ab
	Premium	95 ± 4	235 ± 21 a	12.1 ± 0.5	30.9 ± 2.3 a
8	Small	39 ± 5	181 ± 22	6.9 ± 0.5	31.0 ± 2.7
	Average	52 ± 3	195 ± 19	8.6 ± 0.4	32.0 ± 1.9
	Premium	62 ± 3	217 ± 19	10.2± 0.4	34.1 ± 1.9
9*,†	Small	37 ± 2	130 ± 16 b	6.9 ± 0.3	19.9 ± 1.1 b
	Average	44 ± 2	154 ± 17 ab	8.1 ± 0.3	23.3 ± 1.4 ab
	Premium	45 ± 3	162 ± 19 a	9.4 ± 0.4	24.9 ± 1.9 a
10*,†	Small	39 ± 2	141 ± 16 b	7.1 ± 0.2	21.3 ± 1.2 b
	Average	44 ± 2	140 ± 17 b	8.4 ± 0.2	21.3 ± 1.4 b
	Premium	51 ± 2	173 ± 17 a	9.5 ± 0.2	25.6 ± 1.4 a
11*,†	Small	48 ± 2	129 ± 18 b	6.9 ± 0.4	17.3 ± 1.7 c
	Average	53 ± 2	151 ± 17 b	8.1 ± 0.3	23.0 ± 1.6 b
	Premium	71 ± 2	202 ± 16 a	10.1 ± 0.4	29.6 ± 1.3 a
12*	Small	36 ± 3	134 ± 19	7.0 ± 0.4	20.1 ± 1.9 b
	Average	43 ± 3	143 ± 20	8.5 ± 0.4	22.0 ± 2.1 ab
	Premium	49 ± 3	150 ± 18	9.6 ± 0.4	24.0 ± 1.7 a
14*	Small	40 ± 3	122 ± 21	7.2 ± 0.4	18.7 ± 2.4 b
	Average	44 ± 2	128 ± 19	7.9 ± 0.3	20.6 ± 2.0 ab
	Premium	55 ± 4	158 ± 22	9.5 ± 0.4	25.6 ± 2.5 a
15*,†	Small	60 ± 7	117 ± 33 b	8.1 ± 1.0	16.7 ± 4.9 b
	Average	63 ± 3	185 ± 20 a	9.3 ± 0.4	29.6 ± 2.1 a
	Premium	76 ± 3	185 ± 19 ab	11.5 ± 0.5	27.8 ± 1.9 a
16	Small	45 ± 4	160± 22	7.2 ± 0.5	24.2 ± 2.7
	Average	63 ± 4	196 ± 21	8.6 ± 0.4	28.4 ± 2.4
	Premium	84 ± 3	185 ± 21	10.1 ± 0.4	31.1 ± 2.4

* Families with differences in 7-year g.l.d. among quality classes ($\alpha = 0.05$).
† Families with differences in 7-year height among quality classes.
Different letters within column and family indicate significant differences.

Study Area

This study was established on the Delaware State Forest in Blooming Grove, PA (41°25'N, 75°03'W, elevation 420 m). This area is part of the glaciated low plateau section province of northeastern Pennsylvania and is dominated by oaks, primarily white oak (*Quercus alba* L.), northern red oak and chestnut oak (*Q. prinus* L.), hickory (*Carya* spp.), white pine (*Pinus strobus* L.), pitch pine (*P. rigida* Mill.), and red maple (*Acer rubrum* L.). The soils at the site are of the Manlius Series characterized as strongly acidic, rocky-silt loam with low soil moisture retention. The stand was clearcut in 1975 as part of a commercial harvest to regenerate oak and other economically desirable hardwood species. Preferential browsing by overabundant white-tailed deer inhibited hardwood seedling regeneration and instead facilitated the establishment of a thick understory of sweet fern (*Comptonia peregrine* [L.] J.M. Coult) and ericaceous shrubs (primarily *Vaccinium* spp.). An 8-ha 2.4-m tall woven-wire deer fence was erected on the site in 2005 before planting to protect the experimental material from deer browsing.

Experimental Design

The study was designed to examine the effects of family and seedling quality on seedling survival and growth. Seedlings were planted in two plots approximately 150 m apart. Within the plots, seedlings were planted in an incomplete block design with four seedlings in each block: one small, one average, and one premium size class of 11 families, along with one bulked seed lot. The bulked seed lot seedlings were not included in the analysis presented in this paper. A total of 1,067 seedlings (759 included in the results presented here) were planted in a 2.4-m × 2.4-m grid on each experimental site on April 12 and 13, 2005. Seedlings were planted with a Jim Gem KBC© bar, which was modified by adding 5 cm to each side of the blade. This created a blade that was 15 cm at the top and tapered to the tip.

Measurements

Height and ground-level diameter (g.l.d.) of seedlings were measured at the time of planting and annually thereafter. Survival was tallied at the end of the 7th growing season.

Statistical Analysis

All analyses for this study were processed using SAS 9.3 software (SAS Institute 2011). Seedling response was analyzed using a mixed model analysis of variance to determine significant effects of family and seedling size class on height and g.l.d. after 7 years. Residuals were tested for normality and equal variance. Using a binomial distribution, PROC GLIMMIX was used to evaluate differences in survival among families and quality classes. Least squares means with Fisher's least significant difference mean separation are reported using a 5 percent significance level.

RESULTS

After 7 years 84 percent of the oak seedlings were alive and averaged 166 cm in height and 25.4 mm in g.l.d., 114.0 cm taller and 16.9 mm larger in g.l.d. than they were when planted. Compared to family 7, the family with the highest average 7-year survival (94 percent), seven families were statistically similar in survival and three families had lower survival ($P = 0.0005$, $F = 4.59$; Fig. 1). Within families, the premium seedlings showed higher survival over the average or small seedling size classes in 2 of the 11 families ($P = 0.02$; $F = 2.03$; Table 2).

Compared to family 6, the family with the greatest height after 7 years (202.5 cm), three other families were statistically similar and seven were smaller (Fig. 2; $P < 0.0001$, $F = 5.26$). Compared to family 8, the family with the greatest 7-year g.l.d. (32.3 mm), two of the families were similar and eight were smaller ($P < 0.0001$, $F = 7.73$; Fig. 3). Seven year height and g.l.d. were greater in the premium size class than in either the average or small size class in five and eight of the pedigreed families, respectively ($P < 0.0001$ for each, $F = 5.02$, 4.59, respectively; Table 1).

DISCUSSION

Overall, survival among the seedlings was high (84 percent, Table 2 and Fig. 1) and comparable to the only other 7 year northern red oak enrichment studies we found; 83 percent across three size classes in a study in North Carolina (Kormanik et al. 2002), and 72 percent across six browse protection treatments in a study in Connecticut (Ward et al. 2000).

In families 12 and 14, survival differed among size classes (Table 2 and Fig. 1); these were among the families with the lowest height and diameter after 7 years (Figs. 2 and 3). Family 12 and 14 seedlings averaged 141 and 140 cm in height and 22.2 and 21.9 mm in g.l.d., respectively, compared to 171 cm (height) and 26.2 mm (g.l.d.) averages for the remaining families. Families 12 and 14 were also among the shorter families at planting. For these families, the size advantage of the premium size class may have been more important for competing with other vegetation than for families with larger height and diameter at the time of planting.

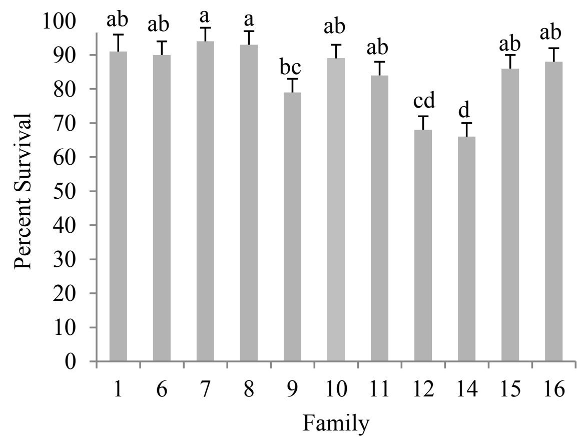

Figure 1.—Percent survival (± standard error) among families 7 years after planting. Bars with the same letter are not significantly different (α = 0.05).

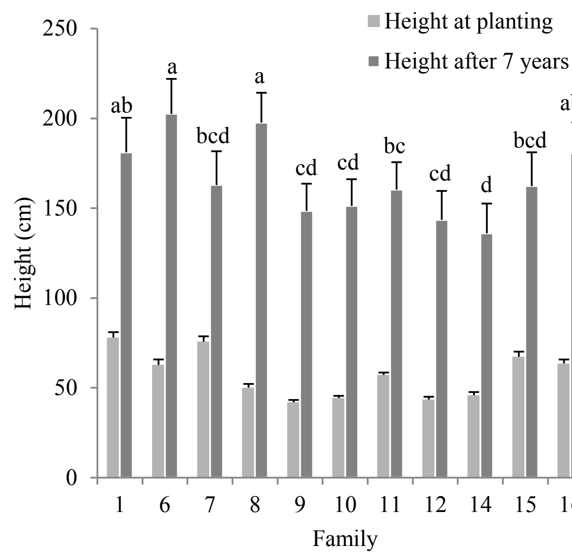

Figure 2.—Planting and 7-year height (± standard error) among families. Seven-year height bars with the same letter are not significantly different (α = 0.05).

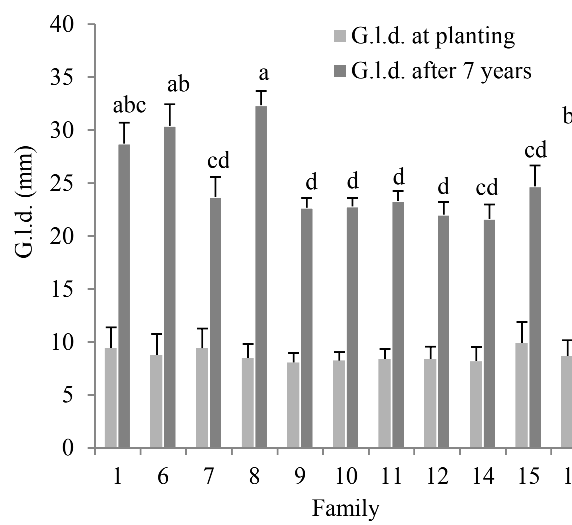

Figure 3.—Planting and 7-year ground collar diameter (g.l.d.; ± standard error) among families. Seven-year g.l.d. bars with the same letter are not significantly different (α = 0.05).

Differences in 7-year diameter among size classes were found in 8 of the 11 families, and size class was important to height in five of the families. Many studies have found that larger northern red oak seedlings, when correctly handled and planted, survive and grow better than smaller seedlings on productive sites (Dey and Parker 1997; Jacobs et al. 2005; Kormanik et al. 1995, 2002; Thomson and Schultz 1995). Taller seedlings impart a height advantage important for competing with fast-growing species on highly productive forest sites (Dey and Parker 1997). Few studies have evaluated the effect of seedling size on less productive sites, such as our study site. Seedlings with larger diameters at planting tend to have larger root systems and more fine roots, which are vital for regaining root-to-soil contact after transplanting (Burdett 1990). Our study shows that initial seedling diameter is also important for subsequent seedling growth on less productive sites. Initial seedling height did not appear to be as important as initial diameter. This may be partly due to the large initial average height of our seedlings (45 cm). This is close to the 50-cm height recommended for oak plantings (Johnson 1981) and therefore may not have been representative of small seedlings that are produced by many tree nurseries. Also, the fast-growing shade-intolerant species that can outcompete oaks on mesic sites, such as red maple, serviceberry (*Amelanchier* spp.), and yellow-poplar, tend to grow less abundantly and rapidly on xeric sites; therefore, height may be less important to planted seedlings on these sites. Furthermore, sweet fern became established on our planting sites in response to the canopy removal in the mid-1970s and may have hindered establishment, survival, and growth of naturally regenerating tree seedlings, thereby conferring an advantage to the planted oak.

The differences in height and diameter among families are not surprising because of the substantial genetic variation for height growth within and between northern red oak seedling populations (Kolb and Steiner 1989). Variation in survival and size among the families over seven growing seasons and two plots suggest that collections from certain mother trees should be avoided because they produce poor quality seedlings. In several years we will evaluate their competitive ability based on growth and height relative to competing vegetation and will subsequently rogue inferior families. Superior families will be left and maintained as a seed orchard to provide acorns for the production of high quality seedlings for regional enrichment plantings.

Table 2.—Percent survival (± standard error) of size classes within each family 7 years after planting

Family	Quality	Percent survival
1	Average	82 ± 8
	Premium	100 ± 8
6	Small	88 ± 8
	Average	83 ± 8
	Premium	100 ± 8
7	Small	100 ± 8
	Average	89 ± 8
	Premium	95 ± 8
8	Small	100 ± 8
	Average	94 ± 8
	Premium	85 ± 8
9	Small	81 ± 8
	Average	83 ± 8
	Premium	75 ± 8
10	Small	84 ± 8
	Average	94 ± 8
	Premium	87 ± 8
11	Small	76 ± 8
	Average	84 ± 8
	Premium	91 ± 8
12*	Small	66 ± 8 b
	Average	51 ± 8 b
	Premium	86 ± 8 a
14*	Small	52 ± 8 b
	Average	51 ± 8 b
	Premium	94 ± 8 a
15	Small	83 ± 8
	Average	79 ± 8
	Premium	94 ± 8
16	Small	93 ± 8
	Average	92 ± 8
	Premium	78 ± 8

* Families with differences in 7-year survival among size classes within family ($\alpha = 0.05$).
Different letters within column and family indicate significant differences.

MANAGEMENT IMPLICATIONS

On low productivity sites where natural oak regeneration is lacking, enrichment planting using high-quality seedlings within deer exclosures is a feasible way to establish northern red oak. The Pennsylvania Bureau of Forestry maintains approximately 16,200 ha of deer fencing on state forest lands, providing ample opportunities for enrichment plantings throughout the state. Fencing and using tree shelters to prevent deer damage to seedlings is not economically feasible for most landowners. Some National Resource Conservation Service programs offer cost-sharing opportunities to help landowners plant and protect seedlings. Planting oak on sites with an expansive sweet fern layer may actually benefit the planted seedlings by reducing growth of natural regeneration. Seedlings of at least 60 cm in height and 8 mm in RCD are recommended for plantings on such marginal or poor sites. Selecting certain mother trees can improve overall seedling quality in the nursery and have ramifications for future seed orchard construction. Our results substantiate the importance of tree improvement programs such as the Tennessee Tree Improvement Program and the Hardwood Tree Improvement and Regeneration Center. In a period of increasing challenges stemming from the introduction and spread of nonnative pests and pathogens, and predicted range changes caused by climate change, the availability of locally adapted, high quality hardwood seedlings will become more important.

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CULTIVAR IDENTIFICATION AND GENETIC RELATEDNESS AMONG 25 BLACK WALNUT (*JUGLANS NIGRA*) CLONES BASED ON MICROSATELLITE MARKERS

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Abstract.—A set of eight microsatellite markers was used to genotype 25 black walnut (*Juglans nigra* L.) clones within the Purdue University germplasm repository. The identities of 212 ramets were verified using the same eight microsatellite markers. Some trees were mislabeled and corrected as to clone using analysis of microsatellite markers. A genetic dendrogram was constructed to show the degree of genetic relatedness between clones. Two additional dendrograms, one based on crown architecture traits and the other on tree size and form traits, were also built and compared with the genetic dendrogram. The genetic dendrogram showed that these eight molecular markers had the ability to distinguish genetically related clones from less related ones. Crown architecture traits and tree size and form traits were able to group genetically related clones together, but less accurately than the genetic matrix.

INTRODUCTION

Black walnut is a highly valuable timber species that is planted and grown widely in the eastern United States. A genetic improvement program for black walnut at Purdue University was initiated in 1967, and a clone bank containing black walnut timber genotypes was established (Beineke 1983, 1989). This clone bank contained genotypes from various parts of Indiana and other states.

Accurate and fast cultivar identification is particularly important for vegetatively propagated species to improve the efficiency of breeding and to protect property rights (Nicese et al. 1998). Numerous molecular techniques have been developed to verify cultivar identity, conduct parentage analysis, and evaluate genetic correlation among walnut cultivars (Dangl et al. 2005). These techniques include isozymes (Arulsekar et al. 1985; Rink et al. 1989, 1994; Solar et al. 1994); restriction fragment length polymorphism (Fjellstrom and Parfitt 1994); randomly amplified polymorphic DNA (Nicese et al. 1998); and microsatellite (simple sequence repeats [SSR]) markers (Woeste et al. 2002). Microsatellite markers developed by Woeste et al. (2002) have been widely used in genetic studies of walnut (*Juglans* spp.) (Dangl et al. 2005, Parks et al. 2014, Zhao et al. 2013). Molecular techniques genetically characterize commercial plant species more accurately than traditional methods that rely on the evaluation of phenological and morphological traits that are usually time consuming and subject to large errors because of environmental factors (Nicese et al. 1998, Weising et al. 1994).

The goals of this study were to (1) use microsatellite markers to verify the clonal identities of 212 black walnut grafted ramets believed to represent 25 clones; (2) examine the genetic relatedness of 25 black walnut clones; and (3) determine the level of correspondence between genotypic and phenotypic dendrograms of 25 black walnut clonal selections.

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MATERIALS AND METHODS

Plant Material

We genotyped samples of 25 grafted clones of black walnut that have been growing in a plantation in West Point, IN (Table 1) since 2002. The clones were either mass selected from wild populations, from the Purdue University Black Walnut Improvement Program, or from a commercial company. Five to 10 grafted ramets were randomly sampled from each clone, for 212 trees in total. Some trees were excluded from the crown architecture and tree size analyses because of wind damage that occurred in summer 2009, 2010, and 2011.

Table 1.—Origin of the 25 black walnut (*Juglans nigra*) clones investigated in this study and documented paternal relationship

Clone	Mother	Origin[a]
C55	Unknown	Darlington, IN
C130	Unknown	wild
C715	C130	West Lafayette, IN
C720	Fayette-1	West Lafayette, IN
C702	BW95	West Lafayette, IN
C703	BW249	South Raub, IN
C707	BW95	South Raub, IN
C710	C55	South Raub, IN
C714	C55	South Raub, IN
C705	BW205[b]	West Lafayette, IN
C701	BW41	West Lafayette, IN
C717	BW36	West Lafayette, IN
C718	C55	wild
C730	C55	wild
C700	Unknown	wild
C708	Unknown	wild
C709	Unknown	West Lafayette, IN
C712	Unknown	wild
C713	Unknown	wild
C716	Unknown	wild
C719	Unknown	wild
C726	Unknown	wild
C728	Unknown	wild
C729	Unknown	wild
C777	Unknown	wild

[a] The places where the selections were found, usually from the progeny test of a previous elite black walnut selection; some, however, were found from the wild populations (wild).
[b] The grandmother of BW205 was BW97.

Study Area

The local annual average temperature in West Point, IN (40.4°N, 87°W) is 11.1 °C, and the annual average precipitation is 92.2 cm (U.S. Climate Data 2012). The soil type is well-drained Elston loam (NRCS 2016); site index is approximately 29 m on a 50 year basis (Zellers et al. 2012). All black walnut trees were planted at a spacing of 4.57 m × 6.10 m and were intensively managed.

Genomic DNA Extraction and Polymerase Chain Reaction Amplification

Fresh leaf samples were collected from each tree, put in plastic bags, and placed in a cooler in the field. All leaves were stored at 4 °C in a refrigerator before the DNA was extracted and quantified as described in Zhao and Woeste (2011). For polymerase chain reaction (PCR) amplification, 12 pairs of primers (Integrated DNA Technologies®, Table 2) from Woeste et al. (2002) were selected to fingerprint these 25 black walnut clones. Genomic DNA was diluted to 50-100 ng·μL^{-1}. PCRs were performed in a volume of 15 μL in 96-well plates, containing 1.5 uL of 2 mM deoxynucleotide solution (GeneMate), 1.5 μL of 1 mg·mL^{-1} bovine serum albumin, 1.5 μL of 1× *Taq* buffer, 1.5 μL of 1 mM magnesium chloride, 1.5 μL of 10 μM reverse primer, 0.3 μL of 10 μM forward primer, 1.5 μL of 10 μM M13 tag with 5' 6-Carboxyfluorescein or 6-Hexachlorofluorescein fluorescent (Schuelke 2000), 0.5 μL of 5 units·μL^{-1} *Taq* polymerase, 1 μL of genomic DNA, and 4 μL of nanopure water. The thermal cycle procedure for all primers was 3 minutes at 94 °C, 35 cycles of 45 seconds at 94 °C, 1 minute at 55 °C, and 45 seconds at 72 °C, and 1 cycle of 5 minutes at 72 °C at the end. Size multiplexing, which combines different markers with nonoverlapping size ranges (Dangl et al. 2005), was used with color multiplexing (5' 6-carboxyfluorescein and 6-hexachlorofluorescein). PCR products were diluted using nanopure water to 1/20 of its original concentration. One μL of diluted PCR product with 14 μL of formamide: rox (67:3) was added. Next, the DNA was denatured by heating at 95 °C for 5 minutes, snap cooled, and submitted to Purdue Genomic Center for analysis with an ABI 3700 sequencer (Applied BioSystems, Foster City, CA).

Table 2.—Characteristics of 12 microsatellite markers used to genotype 25 black walnut (*Juglans nigra*) clones

	Microsatellite Loci	Primer sequence (5′- 3′)	Allele size range
1	WAG 06	F: CCATGAAACTTCATGCGTTG	134–172
		R: CATCCCAAGCGAAGGTTG	
2	WAG 32	F: CTCGGTAAGCCACACCAATT	163–217
		R: ACGGGCAGTGTATGCATGTA	
3	WAG 72	F: AAACCACCTAAAACCCTGCA	135–159
		R: ACCCATCCATGATCTTCCAA	
4	WAG 27	F: AACCCTACAACGCCTTGATG	199–245
		R: TGCTCAGGCTCCACTTCC	
5	WAG 69	F: TTAGTTAGCAAACCCACCCG	164–188
		R: AGATGCACAGACCAACCCTC	
6	WAG 82	F: TGCCGACACTCCTCACTTC	140–234
		R: CGTGATGTACGACGGCTG	
7	WAG 76	F: AGGGCACTCCCTTATGAGGT	228–254
		R: CAGTCTCATTCCCTTTTTCC	
8	WAG 90	F: CTTGTAATCGCCCTCTGCTC	142–178
		R: TACCTGCAACCCGTTACACA	
9	WAG 24	F: TCCCCCTGAAATCTTCTCCT	222–248
		R: TTCTCGTGGTGCTTGTTGAG	
10	WAG 86	F: ATGCCTCATCTCCATTCTGG	208–250
		R: TGAGTGGCAATCACAAGGAA	
11	WAG 89	F: ACCCATCTTTCACGTGTGTG	179–233
		R: TGCCTAATTAGCAATTTCCA	
12	WAG 97	F: GGAGAGGAAAGGAATCCAAA	149–189
		R: TTGAACAAAAGGCCGTTTTC	

Note: Sequence of M13 tag: AGTAAAACGACGGCCAGT; F: forward; R: reverse.

Data Analysis

Allele peaks were marked using GeneMapper v3.7.1 (Applied BioSystems, Foster City, CA). Reference samples from genotype C55 were used as positive controls in each plate. Errors with two base pairs' difference in allele size were allowed when binning and labeling the allele peaks. Allele frequency, probability of identity, probability of exclusion, and pair-wise genetic distance were calculated using the software GenAlEx (Peakall and Smouse 2006, 2012). Allele frequency calculation followed the method in Hartl and Clark (1997), using the equation for codominant data. Two types of probability of identity were calculated: (1) the unbiased probability of identity (P_I), i.e., the probability that two unrelated, randomly sampled trees would have identical genotypes, estimated following Paetkau et al. (1998); and (2) the probability of identity of full siblings (P_{Isib}), i.e., the probability that two randomly selected full siblings would have identical genotypes, calculated using the formula in Waits et al. (2001). The probability of exclusion of these loci for paternity analysis (P_1), i.e., to exclude a putative parent when the genotype of the mother is known, calculated following Jamieson and Taylor (1997) equations 1a and 4; the possibility of excluding a putative parent when the genotype of another parent is unavailable for test (P_2), estimated following Jamieson and Taylor (1997) equations 2a and 4; the probability of excluding a pair of putative parents when the genotypes of both parents were unknown (P_3), calculated using equations 3a and 4 (Jamieson and Taylor 1997).

A pair-wise genetic distance (D_s, a form of Euclidean distance) matrix was generated following equation 1 in Smouse and Peakall (1999). Then a cluster dendrogram based on D_s was prepared by NTSYSpc 2.0 (Rohlf 1997) using the unweighted pair-group method, which uses arithmetic averages (Sokal and Michener 1958). Two phenotypic dendrograms were constructed based on Euclidean distance using PROC CLUSTER and PROC TREE in SAS 9.3 (SAS Institute., Cary, NC), also following the unweighted pair-group algorithm. One phenotypic dendrogram was based on crown architecture traits; a second was based on tree size and form traits. All phenotypic data were collected between 2009 and 2012 (Pang 2014). The crown architecture traits were clonal means of each tree's average branch angle, branch frequency, branch diameter, and branch basal area per meter of stem; the tree size and form traits included clonal mean of d.b.h. (diameter at breast height, 1.37 m), tree height, crown radii, the ratio of tree height to d.b.h. (height ratio), and stem straightness (Tables 3 and 4). Mantel tests (Mantel 1967) were performed following Smouse and Long (1992) to evaluate the correspondence between the genetic matrix and each of the two phenotypic matrices using GenAlEx (Peakall and Smouse 2006, 2012).

Table 3.—Clone means of four crown architecture traits of 25 black walnut (*Juglans nigra*) clones (standard deviation)

Clone	Branch angle[a]	Branch diameter[b]	Branch frequency[c]	Branch basal area[d]
	degrees	*cm*	*per meter*	*cm²*
C130	69.7(5.1)	2.8(0.3)	13.0(1.3)	79.0(10.8)
C55	65.3(3.7)	2.5(0.2)	14.6(2.5)	57.2(11.6)
C700	70.3(5.4)	2.8(0.3)	13.1(1.8)	55.1(7.2)
C701	65.5(2.4)	2.8(0.1)	11.9(1.8)	61.5(7.6)
C702	55.3(2.9)	2.6(0.2)	16.9(1.7)	63.3(7.1)
C703	70.3(2.7)	2.8(0.3)	16.2(1.8)	79.6(12.3)
C705	66.0(2.3)	2.9(0.2)	13.8(1.3)	70.9(15.4)
C707	61.9(3.8)	2.7(0.3)	15.2(1.7)	62.9(15.2)
C708	65.7(6.6)	2.9(0.4)	18.4(1.1)	71.9(10.7)
C709	65.7(3.7)	2.4(0.3)	13.8(0.6)	54.3(9.9)
C710	57.5(3.6)	2.8(0.2)	13.7(1.5)	75.2(9.6)
C712	68.4(1.8)	2.9(0.2)	13.1(1.6)	76.5(12.1)
C713	53.2(2.4)	2.9(0.2)	14.6(1)	88.9(11.3)
C714	66.6(4.2)	3.1(0.2)	15.5(1.2)	74.0(17.3)
C715	57.0(4.9)	2.8(0.3)	16.5(1.9)	81.9(12.6)
C716	67.7(7.7)	2.8(0.2)	15.3(1.4)	65.5(11.7)
C717	66.4(6.9)	2.7(0.1)	13.1(1)	80.8(16.1)
C718	70.3(3.3)	3.2(0.1)	13.3(1.3)	69.1(11.3)
C719	67.5(0.7)	2.7(0)	12.2(3)	66.7(22.9)
C720	60.8(3.5)	3.0(0.3)	12.6(1.5)	66.2(9.7)
C726	70.0(3.3)	2.9(0.2)	8.9(1.1)	74.3(8)
C728[e]	67.7	2.9	14.5	70.9
C729[e]	79.6	2.4	12.6	55.2
C730	61.7(4.4)	2.2(0.2)	14.4(1.2)	66.2(7.3)
C777[e]	66.4	2.8	16	94.6

[a] The average of the insertion angle of all living branches
[b] The average diameter of all living branches
[c] The number of branches per meter of stem
[d] The accumulated branch basal area per meter of stem
[e] Clones that had only one tree left because of wind damage, making standard deviation unavailable

Table 4.—Clone means for tree size and form traits for 25 black walnut (*Juglans nigra*) clones (standard deviation)

Clone	d.b.h. 2009	d.b.h. 2010	d.b.h. 2011	Tree height 2009	Tree height 2010	Tree height 2011	Crown radius 2009	Crown radius 2010	Crown radius 2011	Height: d.b.h. 2009	Height: d.b.h. 2010	Height: d.b.h. 2011	Straightness[a] 2008
	------	------	------	------	------	cm	------	------	------				
C130	13.1(1.2)	15.2(1)	16.7(0.9)	717(51)	887(41)	1023(33)	288(26)	286(20)	237(27)	55.1(3)	58.6(2.3)	63(4.8)	3.2(0.9)
C55	12.3(0.8)	14.5(1)	16.1(1)	850(62)	1009(52)	1113(56)	280(15)	266(17)	245(27)	69(5.1)	69.8(4.1)	69.3(3)	4.2(0.7)
C700	10.9(0.8)	13(0.7)	14.8(0.9)	807(41)	902(86)	975(120)	273(16)	271(34)	220(54)	74.6(6.2)	69.7(7)	66.1(8)	3.2(0.5)
C701	10.5(0.7)	12.8(0.5)	14.6(0.5)	784(43)	909(24)	1046(41)	304(24)	273(27)	199(25)	74.6(6)	71.2(1.8)	71.7(2.1)	3.5(0.5)
C702	12.8(0.7)	14.8(0.6)	16.4(0.5)	815(52)	986(41)	1085(43)	291(21)	248(27)	251(25)	63.9(4.4)	66.6(2.8)	66.2(1.9)	3.9(0.7)
C703	12.1(1.1)	14.5(0.9)	16.3(0.9)	755(52)	936(30)	1055(31)	284(17)	263(26)	264(26)	62.6(2.9)	64.7(4.2)	64.8(2.6)	3(0.6)
C705	11.3(0.9)	13.5(1)	15.2(1.2)	832(117)	963(53)	1061(69)	300(19)	308(25)	254(23)	73.6(7.2)	71.6(2.9)	69.1(4.1)	2.9(0.4)
C707	12.2(1.2)	14.2(1)	15.4(1.2)	831(54)	952(78)	1059(51)	289(29)	249(33)	258(30)	68.7(6.8)	67.2(6.8)	68.9(4.5)	3.1(0.5)
C708	12.4(0.8)	14.9(0.9)	16.3(0.9)	838(49)	1002(31)	1106(35)	291(25)	269(23)	253(18)	67.7(5.6)	67.4(3.9)	68.1(5.3)	2.9(0.7)
C709	11.3(0.6)	13.2(0.5)	14.9(0.5)	797(23)	921(9)	1031(29)	274(22)	250(15)	247(16)	70.9(3.1)	69.7(2.6)	69.1(2)	2.4(0.9)
C710	12.6(0.5)	14.7(0.4)	16.2(0.4)	880(33)	1020(48)	1134(25)	291(21)	279(24)	282(30)	70.1(1.7)	69.4(3.5)	69.8(1.8)	4.3(0.8)
C712	13(0.9)	15.3(1)	17.1(1.1)	827(39)	1000(73)	1156(41)	302(24)	269(17)	252(30)	63.8(3.9)	65.3(3.8)	67.9(4.3)	4(0.8)
C713	13.1(1.1)	15.4(1)	17(1.1)	811(50)	972(60)	1105(51)	306(15)	284(30)	247(31)	62(3.2)	63.3(3.6)	65.2(3.3)	3.7(0.5)
C714	14.1(1.1)	16.4(0.9)	18.1(0.8)	938(48)	1098(40)	1233(51)	304(20)	295(25)	281(19)	66.6(3.9)	67.1(2.4)	68.2(2.5)	4.4(0.5)
C715	14.1(1.2)	16.2(1.1)	17.6(1.1)	774(59)	911(60)	1021(54)	308(18)	285(24)	274(27)	55.1(2.9)	56.4(2.4)	58.4(3.7)	4.5(0.9)
C716	11.9(0.7)	14.2(0.9)	15.7(0.9)	859(35)	955(68)	1101(73)	297(29)	267(28)	269(30)	72(2.2)	67.4(3.3)	70.2(1.6)	4.5(0.8)
C717	11.9(1)	13.8(0.6)	15.2(0.2)	754(48)	780(56)	953(86)	300(20)	288(25)	250(29)	63.3(3.9)	56.5(6)	62.6(6.2)	2.7(0.6)
C718	13(0.3)	15.3(0.2)	16.9(0.3)	851(32)	1032(29)	1130(19)	275(17)	258(21)	249(33)	65.6(2.4)	67.3(2)	66.7(1.6)	3.8(1.1)
C719	10.4(1)	12.9(0.8)	14.4(0.5)	693(13)	747(37)	920(35)	304(4)	265(35)	254(13)	67.3(7.7)	58.2(6.5)	64.2(4.9)	3(0)
C720	11.5(1.5)	13.9(1.3)	15.5(1.4)	772(45)	942(60)	1082(25)	284(24)	294(15)	267(29)	67.8(7.7)	67.8(3.8)	67.6(2.9)	3.2(1.2)
C726	11.2(0.5)	13.2(0.5)	14.7(0.5)	797(75)	886(24)	1016(39)	298(14)	279(21)	262(23)	71.3(5.5)	67.1(2.1)	69.1(1.9)	2.8(0.7)
C728[b]	12.2	14.2	15.8	777	867	1035	299	264	248	63.7	60.9	65.7	3
C729[b]	8.5	9.9	11.1	584	729	771	259	252	240	68.6	73.6	69.8	1
C730	11.7(0.7)	13.9(0.9)	15.3(1)	739(42)	893(16)	1047(27)	283(23)	258(13)	229(12)	63.3(4.5)	64.5(4.1)	68.5(4.4)	4.2(0.8)
C777[b]	12.4	14.9	16.8	789	915	/	312	322	315	63.4	61.6	/	3

[a]1 to 5 with 5 as the straightest
[b]Clones that had only one tree left because of wind damage, making standard deviation unavailable

RESULTS AND DISCUSSION

Allele Frequency, Probability of Identity, and Probability of Exclusion

Although the 12 markers we used to characterize the clones in our study were published SSRs with successful applications (Robichaud et al. 2006, Woeste et al. 2002), the visualization of allele peaks at loci WAG06, WAG24, WAG69, and WAG90 was not consistent or clear, so these loci were dropped. Among them, WAG06, WAG24, and WAG90 often presented multiple (more than three) allele peaks; thus, it was difficult to make allele calls. We omitted WAG69 because of the likely presence of a null allele. In all, then, only eight markers yielded clear results with regard to allele sizes (Table 5).

Based on the allele profile (size and frequency) produced by the eight SSRs for the 25 black walnut clones (Tables 5 and 6), WAG32, WAG27, WAG86, WAG89, and WAG97 were more polymorphic than WAG72 and WAG76, with only eight and five alleles observed for the latter two loci, respectively. Most markers had strong power of exclusion, which means they could precisely identify relatives and determine paternity/parentage reliably. The exceptions,

Table 5.—Allele sizes (in base pairs) at eight microsatellite loci for 25 black walnut (*Juglans nigra*) clones

Clone	WAG32		WAG86		WAG72		WAG82[a]		WAG27		WAG89		WAG76		WAG97	
C130	183	185	222	222	146	146	172	182	217	227	199	209	230	238	171	189
C55	169	191	222	240	146	146	182	190	225	229	197	197	232	236	155	161
C700	181	181	220	232	144	144	168	178	223	227	209	211	232	236	163	169
C701	179	189	216	226	146	146	194	196	221	227	185	219	230	236	155	161
C702	189	191	228	240	146	146	162	190	225	229	191	197	232	236	159	161
C703	171	171	222	236	144	146	184	196	221	225	189	197	236	236	155	173
C705	181	187	216	238	146	148	178	182	213	213	201	213	232	232	157	161
C707	181	181	222	224	146	148	190	196	221	233	209	211	232	236	155	169
C708	187	195	232	238	146	146	178	180	221	221	211	217	232	238	159	161
C709[a]	177	199	212	232	144	144	–	–	219	221	189	209	232	234	163	171
C710	181	191	214	240	146	152	170	184	221	229	197	209	230	236	161	173
C712	179	183	214	222	146	146	176	194	241	241	195	207	230	236	161	161
C713	183	193	216	224	144	144	194	200	211	219	191	209	236	236	157	159
C714	169	173	234	240	146	146	176	182	221	229	187	197	232	232	155	161
C715	181	183	222	230	146	146	182	194	211	217	199	209	232	238	161	189
C716	169	169	222	222	146	146	182	206	223	225	197	201	230	236	155	173
C717	169	181	222	230	146	158	178	200	221	221	187	187	230	236	167	173
C718	171	191	220	222	146	146	168	182	223	229	197	207	232	234	155	163
C719[a]	183	214	222	250	146	156	–	–	211	241	187	199	234	234	163	167
C720	175	207	214	238	146	146	166	180	211	223	201	215	230	232	153	155
C726[a]	179	189	214	216	146	154	–	–	219	219	199	211	230	236	159	167
C728	181	181	216	228	146	146	178	194	213	213	197	213	232	232	161	173
C729	183	197	214	222	144	150	168	188	225	225	205	205	232	236	161	169
C730	169	187	216	222	146	152	182	190	221	229	197	209	232	236	155	175
C777	181	212	220	236	144	152	162	178	211	225	169	191	232	232	133	149

[a] C709, C719, and C726 were not successfully amplified by primer WAG82. As a result, their allele sizes were missing.

Table 6.— Frequencies of observed alleles at eight microsatellite loci based on 25 black walnut (*Juglans nigra*) clones (number of alleles observed for each locus)

Locus	Allele size (base pair)	Frequency	Locus	Allele size (base pair)	Frequency
WAG32	169	0.120	WAG89	169	0.020
(19)	171	0.060	(17)	185	0.020
	173	0.020		187	0.080
	175	0.020		189	0.040
	177	0.020		191	0.060
	179	0.060		195	0.020
	181	0.220		197	0.200
	183	0.120		199	0.080
	185	0.020		201	0.060
	187	0.060		205	0.040
	189	0.060		207	0.040
	191	0.080		209	0.160
	193	0.020		211	0.080
	195	0.020		213	0.040
	197	0.020		215	0.020
	199	0.020		217	0.020
	207	0.020		219	0.020
	212	0.020	WAG76	230	0.160
	214	0.020	(5)	232	0.380
WAG86	212	0.020		234	0.080
(15)	214	0.100		236	0.320
	216	0.120		238	0.060
	220	0.060	WAG97	133	0.020
	222	0.280	(14)	149	0.020
	224	0.040		153	0.020
	226	0.020		155	0.180
	228	0.040		157	0.040
	230	0.040		159	0.080
	232	0.060		161	0.240
	234	0.020		163	0.080
	236	0.040		167	0.060
	238	0.060		169	0.060
	240	0.080		171	0.040
	250	0.020		173	0.100
WAG72	144	0.180		175	0.020
(8)	146	0.640		189	0.040
	148	0.040	WAG82	162	0.045
	150	0.020	(16)	166	0.023
	152	0.060		168	0.068
	154	0.020		170	0.023
	156	0.020		172	0.023
	158	0.020		176	0.045
WAG27	211	0.100		178	0.136
(11)	213	0.080		180	0.045
	217	0.040		182	0.182
	219	0.080		184	0.045
	221	0.220		188	0.023
	223	0.080		190	0.091
	225	0.140		194	0.114
	227	0.060		196	0.068
	229	0.120		200	0.045
	233	0.020		206	0.023
	241	0.060			

Table 7.—Discrimination power of the black walnut (*Juglans nigra*) microsatellites used in this study

Locus	P_I[a]	P_{ISibs}[b]	P_1[c]	P_2[d]	P_3[e]
WAG32	0.0181	0.3057	0.7986	0.6651	0.9381
WAG86	0.0262	0.3206	0.7552	0.6056	0.9147
WAG72	0.2340	0.5329	0.3441	0.1743	0.5347
WAG82	0.0163	0.3016	0.8098	0.6804	0.9428
WAG27	0.0260	0.3169	0.7585	0.6091	0.9114
WAG89	0.0179	0.3049	0.8003	0.6673	0.9384
WAG76	0.1275	0.4231	0.4764	0.3036	0.6583
WAG97	0.0274	0.3200	0.7522	0.6019	0.9094
Combined	0.0000	0.0002	1.0000	0.9987	1.0000

[a]The probability that two unrelated, randomly sampled trees would have identical genotypes
[b]The probability that two randomly selected full siblings would have identical genotypes
[c]The possibility of excluding a putative parent when the genotype of mother is known
[d]The possibility of excluding a putative parent when the genotype of another parent is unavailable for test
[e]The probability of excluding a pair of putative parents when the genotypes of both parents were unknown

WAG72 and WAG76, were less powerful than the other markers (Table 7) because they were less polymorphic. Although usable markers were reduced from 12 to 8, combined together, these eight markers had a probability of exclusion (P_I) that was less than 2.9×10^{-12} and a P_{Isibs} of 0.0002 (Table 7). These data indicate that the SSRs provided high power of discrimination for cultivar identification, paternity determination, parentage analysis, and relatedness. The likelihood that individuals would share genotypes simply by chance was essentially zero.

Several trees were found to be wrongly labeled based on their allele sizes at the eight loci. Sample BD111 was labeled as C55, and BM111 was marked as C703; however, both of their genotypes matched clone C707. AS115 was labeled as C728, but its genotype did not match any other trees of C728 or any other clones. AY138 and AZ130 were both recorded as C716, but their genotypes were different from other trees of C716. The allele sizes indicated that AY138 and AZ130 shared the same genotype, so they were temporarily named C777, a clone that was not on the original list. Ramets of C704 were found to be identical to those of C715. These results were in accordance with the phenological observations recorded for each clone from 2009 and 2010 (Pang 2014). As a result, there were still 25 genotypes in total, and we corrected the mislabeled trees before conducting further analysis.

Table 8.—Pair-wise genetic distance among 25 black walnut (Juglans nigra) clones based on the allele information at eight microsatellite loci, calculated by GenAlEx 6.41

130[a]	55	700	701	702	703	705	707	708	709	710	712	713	714	715	716	717	718	719	720	726	728	729	730	777	
0																									130
13	0																								55
18	18	0																							700
13	12	17	0																						701
15	5	17	11	0																					702
16	11	16	12	13	0																				703
17	14	15	15	14	19	0																			705
14	12	10	12	13	12	13	0																		707
15	14	16	12	12	16	12	13	0																	708
18	19	12	18	18	16	18	16	16	0																709
14	9	14	11	9	11	15	11	13	16	0															710
13	13	20	10	13	16	17	16	15	21	12	0														712
18	18	13	15	15	14	18	15	19	14	15	17	0													713
15	6	18	12	9	15	12	13	11	17	11	14	20	0												714
6	11	15	12	12	16	12	11	12	17	12	11	15	11	0											715
11	7	19	13	13	11	17	14	17	21	13	14	19	12	13	0										716
15	15	16	13	16	13	17	12	13	18	11	16	17	13	14	12	0									717
12	7	14	13	10	12	14	13	14	16	12	14	19	9	11	10	16	0								718
14	17	19	17	17	18	19	17	18	13	17	15	18	17	13	17	15	13	0							719
14	13	17	12	13	16	13	14	12	18	13	14	18	12	12	12	16	11	16	0						720
16	17	18	12	14	17	18	16	16	14	14	15	14	18	16	17	16	17	13	15	0					726
18	13	15	14	12	18	6	13	14	20	13	16	20	12	11	17	16	14	20	15	19	0				728
17	14	14	17	14	14	17	15	18	17	15	15	15	17	15	16	18	16	18	17	18	19	0			729
12	6	15	11	10	11	12	9	12	15	9	15	14	8	11	9	12	9	16	13	15	14	15	0		730
19	16	11	18	13	16	13	14	16	15	15	20	15	15	14	19	17	15	18	15	19	14	14	14	0	777

[a]Letter "C" was omitted from all clone names to save space

Genetic Relatedness between 25 Black Walnut Clones

A genetic dendrogram can summarize microsatellite data and reveal genetic relationships among tested cultivars (Dangl et al. 2001). Three pairs of clones were grouped into three clades: C702 and C55 had the smallest genetic distance (five); there were two pairs with a genetic distance of six, C130/C715 and C705/C728 (Table 8, Fig. 1). The proximity of C130 and C715 was not surprising because C130 was believed to be the female parent of C715. C55 and C702 shared one allele at all eight loci (Table 5), indicating that C55 may be the sire of C702, because C702's mother was believed to be BW95, a clone not included in this study. C705 and C728 also shared one allele at each locus, indicating close consanguinity, almost certainly as parent-offspring, but possibly full siblings. Breeding records indicate that C55 was the mother of C710, C714, C718, and C730. All these relationships were supported by the close genetic distance between them (Table 8) and the dendrogram based on genetic distance (Fig. 1). The genetic dendrogram and relatedness matrix also disclosed that C702 is closely related to C710, C714, C716, C718, and C730, almost certainly as a full sibling. According to previous breeding records, C702 and C707 are maternal half-siblings from BW95, but this was not reflected in the dendrogram as were the half siblings of C55 mentioned earlier. C702 and C707 may be half-siblings, but the SSR genotype profiles failed to show it because they inherited, by random segregation, different alleles from their mother at many of the eight SSR loci. At least one of the two clones may also have been incorrectly assigned as an offspring of BW95. The genotype of BW95 will need to be compared with both C702 and C707 to obtain more information.

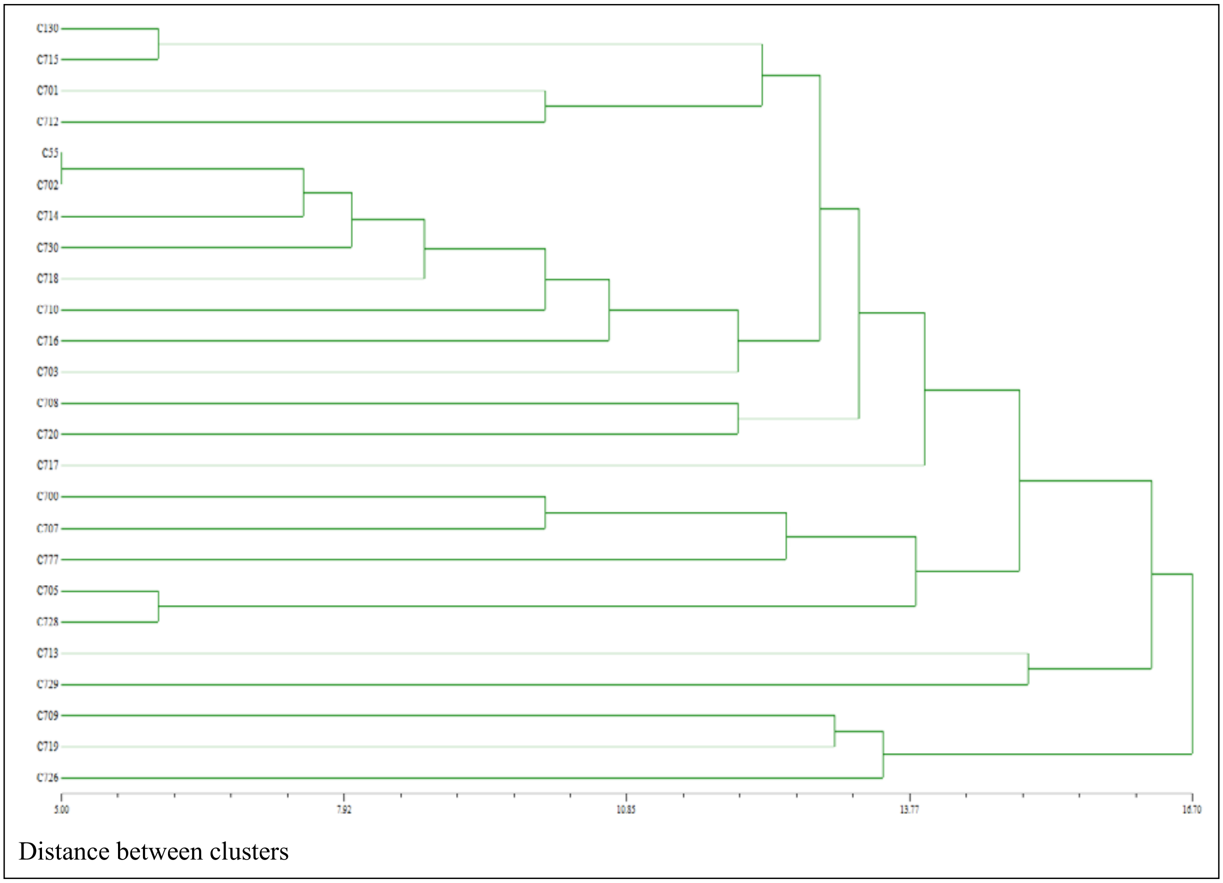

Figure 1.—Dendrogram generated using cluster analysis based on the proportion of shared alleles among 25 black walnut (*Juglans nigra*) genotypes by NTSYSpc 2.0.

The Concordance and Discordance between Genetic Relatedness and Phenotypic Relatedness

The crown architecture dendrogram (Fig. 2) revealed some phenotypic similarities between clones. C705 and C728 were closely grouped based on cluster analysis of crown architecture traits, just as they were in the genetic dendrogram (Fig. 1). C702 and C707 were half-siblings and were grouped in one clade at a distance of 0.52. Half-siblings C718, C710, and C714 were close to each other; C718 and C714 were more closely clustered. C55, as the presumed female parent of C730, and possible male parent of C702, was located near C730 and C702 in the crown architecture dendrogram as well. C716 was near C714, C718, and C710, as it was in the genetic dendrogram. The positions of these clones in the dendrogram make sense given their known genetic relationship. Not all clones were located within the dendrograms where they were expected, however; for example, C715, believed to be an offspring of C130, was far from C130 based on the branch attributes that comprised the crown architecture (Fig. 2). This disparity between genetic relatedness and crown architecture similarity was probably because C715 had larger branch frequency than C130 (16.5 ± 1.9 versus 13 ± 1.3, respectively), and smaller branch angle than its supposed female parent (57 ± 4.9 versus 69.7 ± 5.1) (Table 3).

The tree size and form dendrogram (Fig. 3) clustered clones with similar size and form together. C716 was located close to C710, C718, and C55, as it was in the genetics dendrogram. Again, C728 and C705, which are most likely a parent-offspring pair or full siblings, were not far apart in the dendrogram. Although C715 and C130 were unlike each other in branch characteristics,

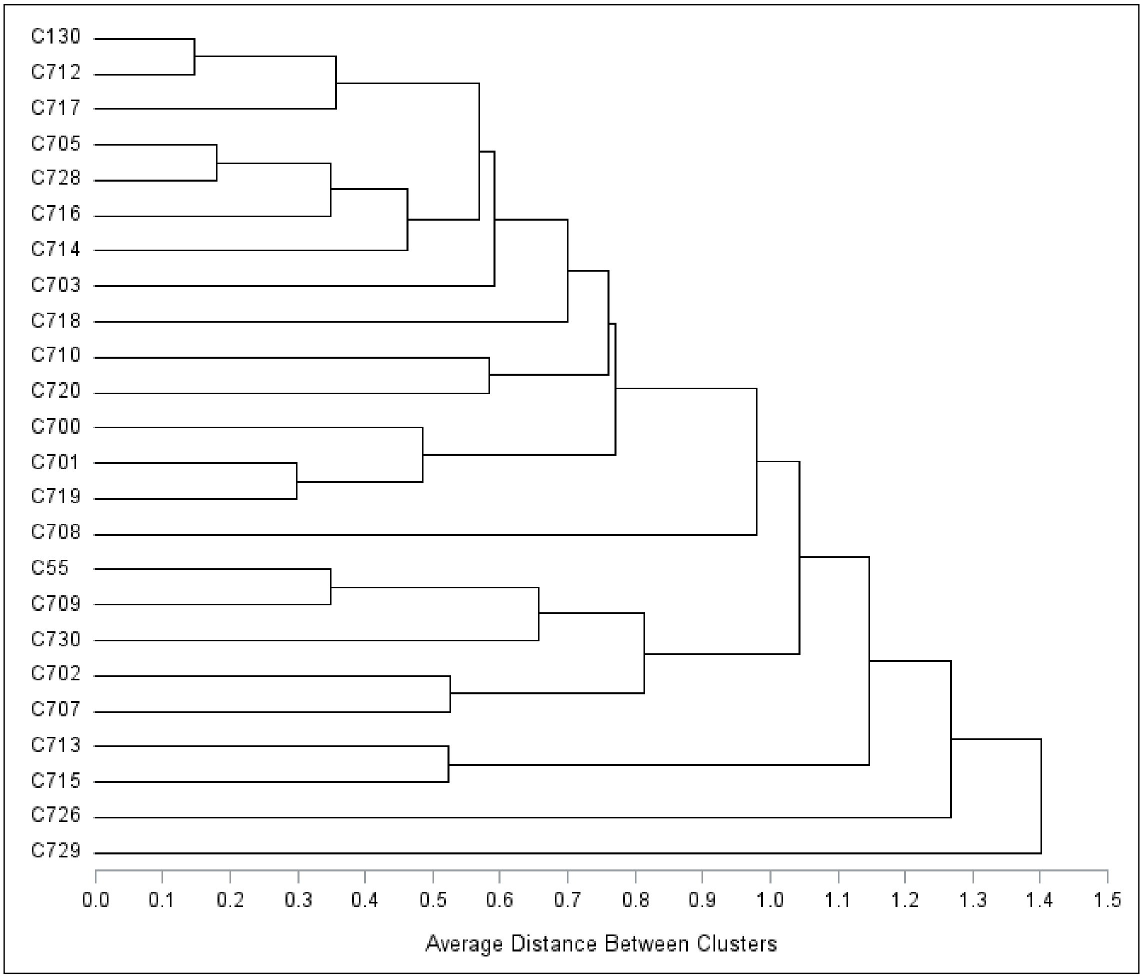

Figure 2.—Dendrogram generated using cluster analysis based on crown architecture traits of 25 black walnut (*Juglans nigra*) genotypes.

as shown in the crown architecture dendrogram, they were grouped in one cluster in this dendrogram, indicating that C715 possibly inherited size and form from its female parent, but not branch structure. C702 and its presumed half-sibling C707 clustered closely. C55 was close to its presumed offspring C718, C710, and its possible offspring C702. Among all the presumed offspring of C55, C714 was furthest from C55. In fact, C55 was closer to the unrelated C130 than it was to C714. It was probably because C714 was much larger in d.b.h., tree height, and crown radius than C55, its siblings, and C130 (Table 4). The most phenotypically divergent clone based on tree size and form traits was C729, which was also most divergent in terms of crown architecture. C729 was not closely related genetically to any other clone (Fig. 1).

Overall, the genetic dendrogram showed that the eight microsatellites we used had the ability to distinguish genetically related clones from unrelated ones, which was reflected by the high probabilities of exclusion (>0.9987, Table 7). Each of the two phenotypic dendrograms grouped some genetically related clones together, but less accurately than the genetic distance dendrogram. Phenotypic similarity as reflected in architecture, tree size, and form traits, was not accidental but a consequence of genetic relatedness. Phenotypes imperfectly reflect genotypic relatedness in part because growth is a deviation-amplifying process (Stage 1987) because of the accumulated effects of local environments over time and genotype × environment interactions over time. Variability in important traits increases as trees grow. In this way, even clones, which are genetically identical, can begin to look different over time. The accumulated effects of local

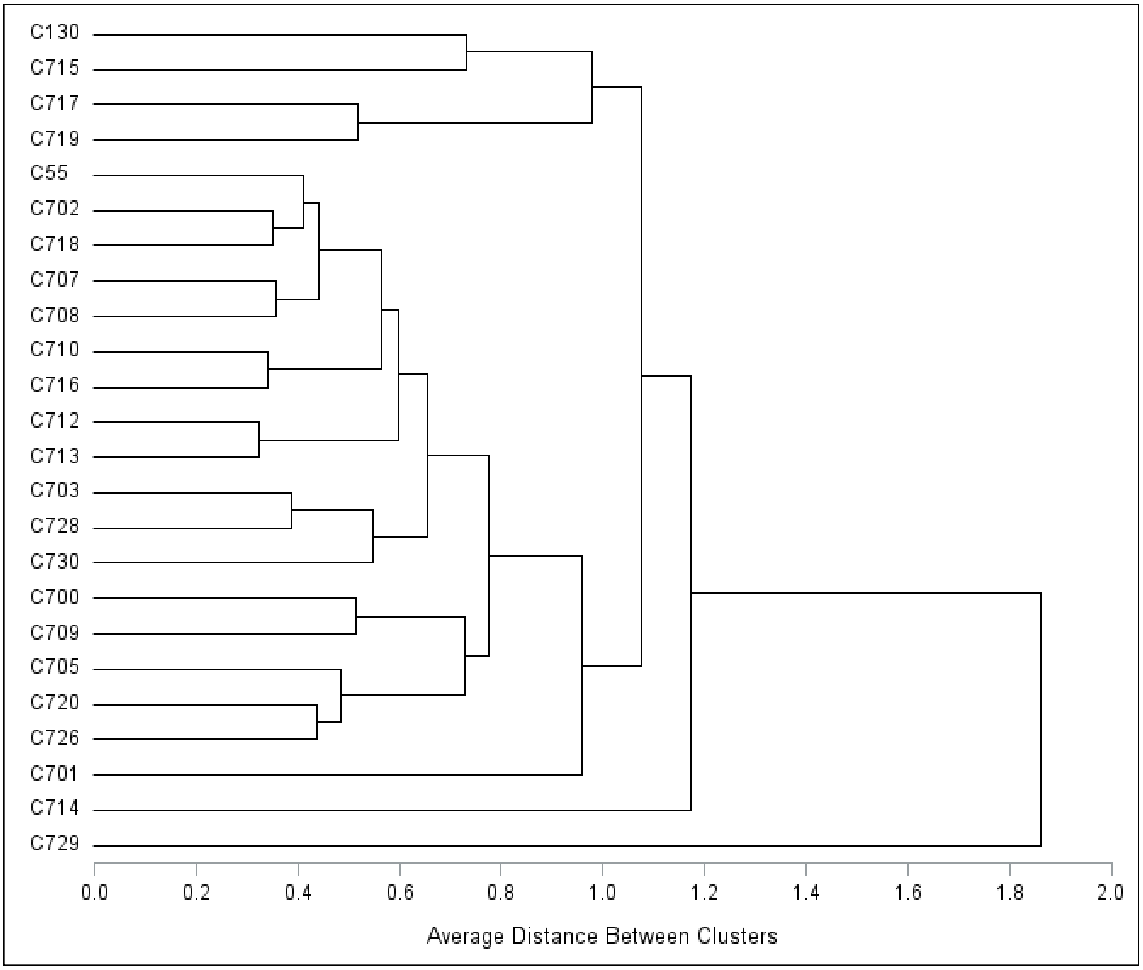

Figure 3.—Dendrogram generated using cluster analysis based on tree size and form traits of 25 black walnut (*Juglans nigra*) genotypes.

environmental differences on a tree's growth can make it difficult to untangle which genes control the growth and form of trees and to what extent. Some confusing associations, such as C55 being closer to C130 than its presumed offspring C714 in the size and form dendrogram, and the unexpected placement of C715 far away from its maternal parent C130 in the crown architecture dendrogram, may in part be the result of such deviation-amplifying processes. The lack of correspondence between the phenotypic and genotypic dendrograms can also be attributed to the relatively small sample size of trees and the high level of genetic relatedness among these trees. If there were more clones like C729 that are more distinct from the rest of the clones, we would have a better perspective. The discordance between genetic relatedness and phenotypic relatedness, however, implies that desirable crown architecture can be obtained from genetically unrelated parents within the breeding program, which is positive because it reduces the necessity for inbreeding. The dendrograms of crown traits and tree size and form traits are useful for understanding how trees can be grouped by architecture and form, which may ultimately be useful in understanding whether certain tree characteristics are physiologically more desirable for efficient growth in plantations.

The attempt to associate phenotypic and genetic similarity is the basis for considerable research in modern quantitative genetics. The correspondence between the genetic distance measures and the phenotypic matrices can be either concordant (Atchley et al. 1982, Villani et al. 1992), or discordant (Atchley et al. 1988, Hampl et al. 2001, Woeste et al. 1998). In our case, the Mantel correlation between the matrix of genetic relatedness and the matrix of similarity in crown architecture was low (0.105), as was the correlation between the genetic relatedness matrix and tree form and size matrix of similarity (0.141). Complex traits usually have a complex genetic basis, and when a large number of genes are involved in a trait, it often has a wide variance that is strongly affected by environment. The phenotypes entered into the matrix were complex—a mathematical combination of several distinct phenotypes, each with its own error of estimation, reducing its connection to common underlying genetic causes. Stated another way, in general, the more complex the phenotype, the more genes involved in its expression, and the lower its heritability, which was shown in this case by the low correspondence between the genotypic and phenotypic matrices. The process of reducing multidimensional reality to a single number facilitates comparison, but a lot of information is lost in the process. Having a more detailed genotype can increase the connection between genotype and phenotype, but even if the number of DNA markers were increased to thousands and a full quantitative trait loci (QTL) analysis performed, such an analysis is often able to explain only a small percentage of the total phenotypic variance for quantitative traits such as yield or plant height. Despite their drawbacks, our results have value because they represent the first systematic attempt to measure and categorize phenotypic variability for important growth and form traits in black walnut. By adding more microsatellite markers and more black walnut clones from the genetic improvement program, more accurate and more comprehensive pedigree information will be available to black walnut breeders in the future.

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MODELING AND UTILIZATION

INVESTMENT ACTIVITIES IN THE U.S. SECONDARY WOODWORKING INDUSTRY

Matthew Bumgardner, Urs Buehlmann, and Karen Koenig[1]

Abstract.—The U.S. secondary woodworking industry has shown signs of improvement after the steep losses in sales volume associated with the housing crisis that began in 2007. Employment in several sectors has begun to increase, suggesting that companies that survived the downturn are positioning to increase sales growth. It is likely that investment plans to improve firm-level productivity or capabilities are also gaining momentum. The objective of this paper was to ascertain investment activities in the secondary woodworking industry based on a survey of "Woodworking Network" magazine subscribers conducted in 2015. The survey was the latest of a 6-year series that tracked firm performance and activities annually since 2008, with an added recent focus on investment plans. Respondents provided information regarding their planned investment levels, areas in which they planned to invest, and the most important current drivers of investment plans. While responses showed a general increase in planned investment levels over the next 3 years for large firms (65 percent planned to invest more in 2015 than in 2014), a plurality of smaller firms (42 percent) were uncertain of their investment plans. Furthermore, a higher percentage of large firms indicated they planned to invest compared to small firms in nearly every category investigated. There were some differences between small and large firms in what was driving company investment plans. The results suggest that overall investment activity might be expected to increase in the near term, but the business environment still remains somewhat uncertain for many smaller firms.

INTRODUCTION

Construction-based markets continued an overall growth trend in 2014. U.S. spending on single family housing, multi-family housing, and nonresidential construction increased in 2014, but repair and remodeling (i.e., residential improvements) declined (Fig. 1) (U.S. Census Bureau 2015). The largest proportional increase was in multi-family housing, which increased by 33 percent from 2013 to 2014, while single family housing and nonresidential construction increased by 13 and 11 percent, respectively.

Correspondingly, the secondary woodworking industry also has seen improving business conditions associated with the overall improvement in construction markets. For example, during the worst of the housing crisis from 2008 to 2009, approximately 81 percent of secondary woodworking companies reported losing sales volume; for 2012 to 2103, this percentage had declined to 26 percent (Buehlmann et al. 2014). Employment trends in secondary woodworking industries also have improved, with the number of employees in the kitchen cabinet industry (North American Industry Classification System [NAICS] code 33711) increasing by 11 percent from 2012 (employment low point) to 2014 and the millwork industry (NAICS 32191) realizing an 8 percent increase in employment for the same period (U.S. Bureau of Labor Statistics 2015). Given these improvements, it might be expected that the secondary

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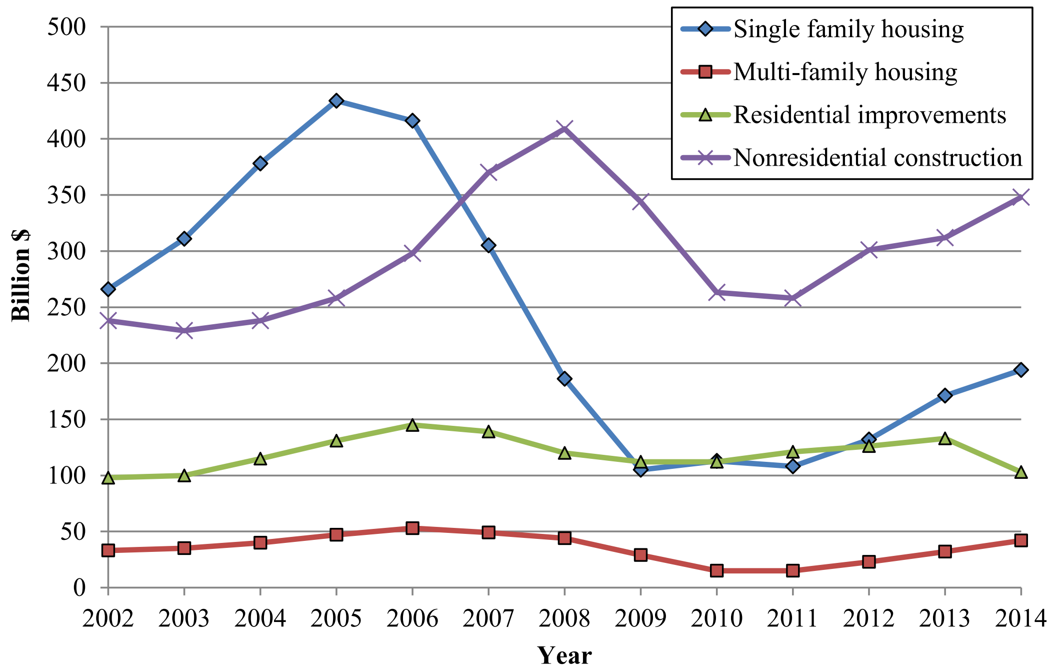

Figure 1.—Value of private U.S. construction put in place, 2002-2014 (U.S. Census Bureau 2015).

woodworking industry would be considering increasing business investments to increase capacity, decrease per unit manufacturing costs, and improve quality, or might consider offering new features or new product lines.

In economic terms, investment means business spending for capital goods to be used to provide goods and services (Boyes and Melvin 1991). Broadly, the determinants of the level of investment made by businesses include the interest rate, cost of capital goods, and profit expectations, all of which influence the potential return on investment. Other factors, such as the rate of technological change within an industry and capacity utilization, can also influence investment levels (Boyes and Melvin 1991). In the present study it was assumed that firm size played a role. According to the resource-based view of firms, resources internal to the firm are the primary source of competitive advantage (Hoopes et al. 2003). Under this scenario, large firms will have relatively greater financial means to make investments. For example, previous research has shown that small firms struggle with issues surrounding external financing (Huang and Brown 1999), and that being a large company within an industry requires the ability to make durable and irreversible investments (Ghemawat 1986). So while it might not be surprising to learn that small firms plan to spend less on investments than large firms, even less is known about the specific areas that small firms might be interesting in investing in or the current drivers of their investment decisions.

The objective of the present study was to assess business investment activities of small and large firms in the secondary woodworking industry by contrasting their planned investment levels, areas in which they plan to invest, and the most important current drivers of investment plans. A secondary objective was to determine changes in sales volume performance for secondary woodworking companies to assess the business environment for making investment decisions.

METHODS AND SAMPLE DESCRIPTION

Since February 2010, "Wood & Wood Products" or "Wood Products" magazine (now called "Woodworking Network" magazine) has conducted an online survey of their secondary wood working subscribers in February and March of each year to assess the previous year's performance, behaviors, and perceptions of market conditions in housing and related construction markets. Several of the questions have remained the same from year to year to help track industry activities over time. The 2015 study included new questions to focus on investment plans and activities, which are the focus of this paper. The number of responses received each year has ranged from 359 in 2010 to 193 in 2014, with response rates generally ranging from 1 to 3 percent. A higher response rate was achieved in 2010 (46 percent), but a more targeted list was available for use in that year. For the 2015 study there were 228 usable responses received. Although formal checks for nonresponse bias are not possible, the summary data in Table 1 suggest that samples from each year have similar firm characteristics.

Each year, most respondents were either company owners or in positions of corporate/operating management (ranging from 64 to 72 percent of the sample) and represented firms in at least 40 states. For 2015, the states of California, North Carolina, Indiana, Pennsylvania, Illinois, Michigan, New York, Texas, and Ohio each accounted for at least 4 percent of the total responses. Similar to years past, kitchen/bath cabinet producers made up the largest percentage of the 2015 sample, representing 32 percent of respondents (the lowest percentage for cabinets in the 6 years of the survey). Nearly 18 percent were household furniture producers (representing the largest percentage for household furniture in the 6 years of the study), 12 percent were millwork manufacturers, 7 percent were architectural fixtures firms, 7 percent were producers

Table 1.—Selected firm characteristics for respondents by study year

	Year					
Firm characteristics	2010	2011	2012	2013	2014	2015
Number of states represented in sample	46	46	42	41	42	40
Main products produced[a]	----------	----------	----------	*Percent*	----------	----------
Kitchen/bath cabinets	36	44	41	42	35	32
Household furniture	8	7	13	14	12	18
Architectural fixtures	8	7	10	8	11	7
Molding/millwork	13	11	11	11	15	12
Other	35	30	25	25	28	32
Respondent position						
Corporate/Management/Owners	72	67	71	67	72	64
Number of employees[a]						
1-19	--[b]	61	68	67	66	65
20-49	--	12	11	7	6	11
50+	--	27	21	26	27	24
Price point of primary product[a]						
Low to medium[c]	36	35	29	32	36	32
Medium-high	54	54	56	56	50	50
High	10	11	16	11	14	18

[a] Yearly totals may not sum to 100 percent due to rounding.
[b] --indicates question not asked in 2010.
[c] Low, low-medium, and medium categories combined.

of dimension or components, and 5 percent manufactured office/hospitality/contract furniture. While an additional 20 percent indicated their production was in "other" categories, most could reasonably be classified into one of the aforementioned categories (especially millwork). Closets were also a somewhat common product area. Similar to past years, most responding firms were small, with 49 percent having sales of less than $1 million in 2014 and another 28 percent having sales of $1-10 million. Furthermore, 65 percent of respondents had 1-19 employees. Lastly, respondents' production was domestically focused, with 84 percent indicating that more than 60 percent of their sales in 2015 would result from domestically produced and/or sourced products; this percentage was similar to previous years' studies.

Comparative analyses were conducted by categorizing respondents as either a small firm (1-19 employees) or large firm (20 or more employees). For these analyses, there were 125 small firms and 68 large firms; 35 firms did not indicate their firm size so were only included in aggregate measures. Respondents provided information regarding their planned investment levels, areas in which they planned to invest, and the most important current drivers of investment plans. For frequency count data, chi-square tests were used. For two-group comparisons of interval-level data (i.e., comparing small and large firms on scaled responses), two-tailed t tests were used. When there was a difference in variances between the groups (based on a folded F test), the Satterthwaite method was used. An α level of 0.10 was used for all tests. Statistical analyses were carried out using SAS® Enterprise Guide 6.1 (SAS Institute, Cary, NC).

RESULTS AND DISCUSSION

Changes in Performance

Analysis of year-over-year sales performance from 2009 to 2014 revealed continued gradual improvement in the percentage of firms reporting positive changes in sales volume. In 2009, 81 percent of respondents reported losing sales volume from the previous year, and 60 percent reported losing a volume of 20 percent or more. By 2014 (the current study), this proportion had declined to 21 percent (Fig. 2). Furthermore, the proportions of respondents in the Somewhat Better (sales up by 10 percent) and Much Better (sales up by 20 percent or more) categories have been increasing or holding steady each year. Nearly a quarter of respondents indicated that their sales were up by 20 percent or more from 2013 to 2014. This year's results also showed an increasing number of respondents indicating sales were unchanged year over year, suggesting more firms are seeing a stabilizing marketplace. Overall, these changes in sales volume suggest a business environment conducive to firms considering investments to improve productivity and capabilities.

Planned Investment Activities

While 65 percent of large firms planned to invest more in 2015 than 2014 to improve productivity and capabilities, just over a third of small firms indicated they planned to increase investments in 2015 (Fig. 3). For small firms, many (42 percent) were uncertain about their investment plans for 2015 at the time of the study. In fact, this was the most common response from small firms. The "no" response was the lowest category for both small and large firms, however, suggesting a generally favorable environment for investments in the secondary wood industry.

A majority of small firms (67 percent) indicated they planned to invest less than $250,000 over the next 3 years, and nearly a quarter had no investment plans for the period. Of the small firms that responded, none had plans to spend more than $1 million in investments over the next 3 years. Conversely, large firms were somewhat equally distributed across the investment categories, and only 4 percent indicated their firms had no investment plans (Fig. 4).

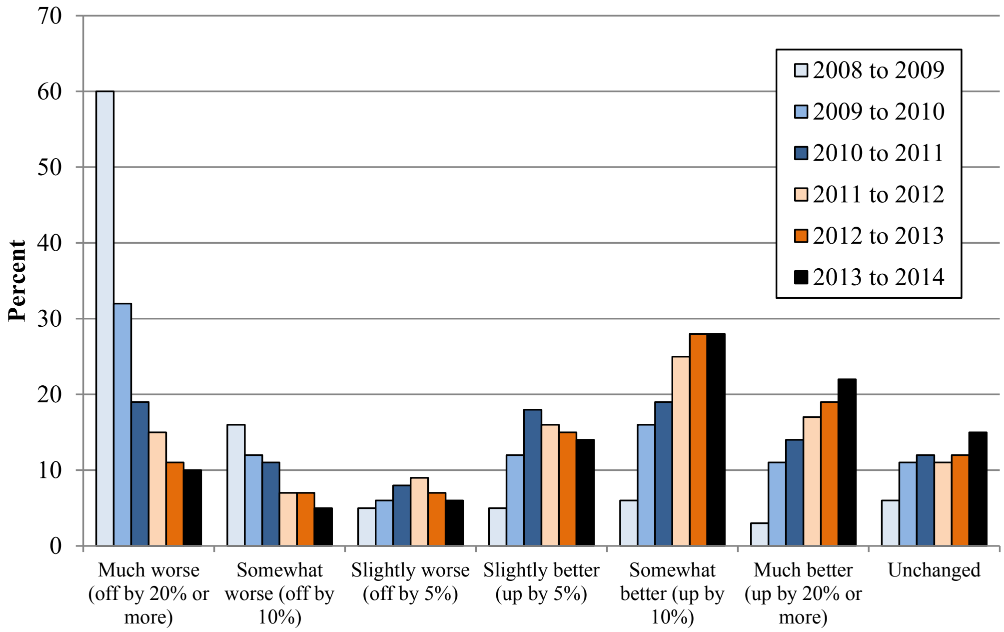

Figure 2.—Responses to the question, "Compared to the previous year, last year's sales volume was . . ." (e.g., 2015 study asked for change from 2013 to 2014), by study year.

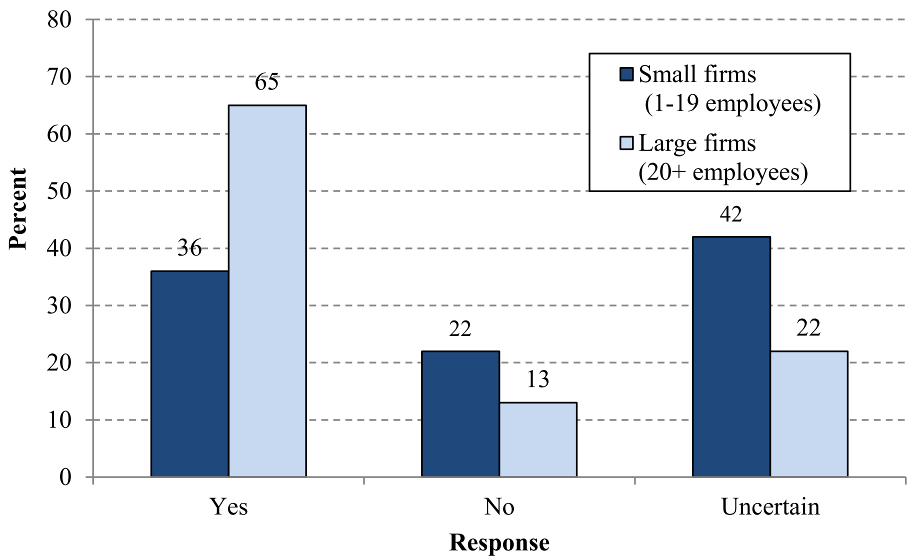

Figure 3.—Categorical responses to the question, "Does your company plan to invest more in productivity and capability improvements in 2015 than it did in 2014?" Chi-square statistic for associated frequency counts = 14.6 (P=0.001).

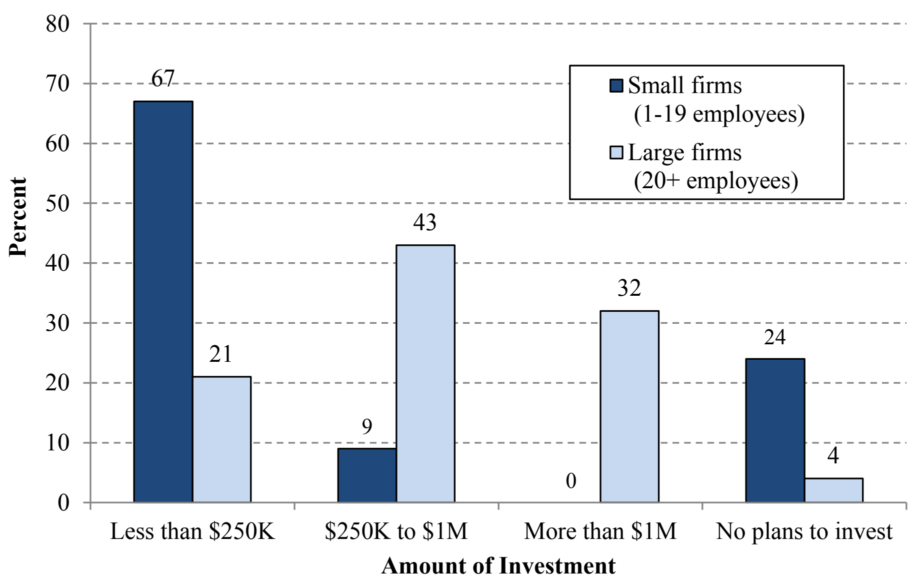

Figure 4.—Categorical responses to the question, "Over the next 3 years, about how much does your company plan to spend on investments to improve productivity or capabilities?" Chi-square statistic for associated frequency counts = 93.5 (P<0.001).

Table 2.—Areas where companies will invest significantly to improve productivity or capabilities within the next 3 years. Results ordered by large firm percentages, with greatest areas of investment at the top.

Investment areas	Small firms (*n*=125)	Large firms (*n*=68)	Chi-square statistic[a]	P[b]
	---------*Percent* ------------			
Sales force expansion/development	16.8	50.0	23.82	**<0.001**
Employee training	24.0	45.6	9.50	**0.002**
Advertising/marketing communications	30.4	41.2	2.27	0.132
Panel processing	16.8	33.8	7.25	**0.007**
Assembly	11.2	30.9	11.49	**0.001**
Inventory reduction	9.6	30.9	14.07	**<0.001**
Design/manufacturing software	22.4	29.4	1.16	0.282
Finishing	19.2	29.4	2.61	0.106
Solid wood processing	21.6	26.5	0.58	0.445
E-commerce	16.0	17.6	0.09	0.769
Decorative laminating/veneering	7.2	16.2	3.82	**0.051**
Component outsourcing	13.6	11.8	0.13	0.717
Rough mill	7.2	10.3	0.56	0.456
Certification/green initiatives	4.8	8.8	1.22	0.269

[a] Chi-square tests based on associated frequency counts in 2 x 2 tables for each investment area.
[b] Bold *P* values indicate significant differences between small and large firms at α=0.10.

This study also assessed the general categories or areas where investments were planned within the next 3 years (Table 2). For small firms, this included advertising/marketing communications (30.4 percent), employee training (24.0 percent), design or manufacturing software (22.4 percent), and solid wood processing (21.6 percent). For large firms, the areas with the most planned investment activity were sales force expansion (50.0 percent), employee training (45.6 percent), and advertising/marketing communications (41.2 percent). Thus, marketing and employee training were areas important to both small and large firms.

In every category except component outsourcing, a higher percentage of large firms indicated planned investments than did small firms (Table 2). This was especially true for sales force expansion/development, employee training, panel processing, assembly, inventory reduction, and decorative laminating/veneering, all of which were statistically significantly different between small and large firms. These results show areas especially important to larger firms, including becoming leaner (inventory reduction) and improving manufacturing capabilities. They also suggest that many large firms are expecting to expand sales in the next 3 years with the addition of more sales capabilities, which might be an important indicator of economic confidence. Three of these areas (employee training, sales force expansion, and inventory reduction) were found to be significant in a previous study conducted during the economic downturn associated with the housing crisis (Buehlmann et al. 2013). The consistency across studies for these factors suggests these are areas of enduring interest to large firms.

Factors Influencing Investment Plans

Lastly, respondents assessed the most important current drivers of investment plans (Table 3), which offered a similar pattern to the results from Table 2. Small firms rated nearly every factor as less important than did large firms, and several of the differences were statistically significant, including to improve productivity, to reduce labor costs, to replace aging equipment, and to

Table 3.—Importance values for factors driving company investment plans over the next 3 years. Results ordered by large firm means, from least important to most important.

Factors	Mean importance value[a] Small firms (*n*=125)	Large firms (*n*=68)	*t* value	*P*[b]
To improve design capabilities	3.5	3.3	0.97	0.332
To replace aging equipment	2.8	3.3	-2.62	**0.009**
To increase capability to customize products	3.4	3.5	-0.29	0.775
To reduce labor costs	3.2	3.6	-1.94	**0.054**
To enter new product markets	3.3	3.6	-1.27	0.207
To maintain market share	3.7	3.7	-0.11	0.915
To improve product quality	3.7	3.9	-0.93	0.353
To increase market share	3.7	4.1	-2.61	**0.010**
To improve productivity	3.9	4.3	-2.23	**0.027**

[a] Scale anchored by 1 = *Not at all important* to 5 = *Very important*; no other scale points were labeled.
[b] Bold *P* values indicate significant differences between small and large firms at α=0.10.

increase market share. Improving design capabilities seemed relatively important to small firms. Given that most small firms are either uncertain of their near-term investment plans or have no plans to invest, small firms also perceive a lesser importance to investing than do large firms that invest more frequently and heavily than do small firms.

SUMMARY AND CONCLUSION

Results of the 2015 survey showed a continuation of previous trends of more firms indicating year-over-year sales increases or unchanged sales. Overall, most respondents perceived a stable or improving market, which suggests an environment where companies might be looking to invest to improve their productivity and capabilities.

Overall, 76 percent of small firms and 96 percent of large firms indicated they planned to invest to improve productivity or capabilities over the next 3 years. Large firms showed a tendency toward greater investment activity overall, as well as less uncertainty about their investment plans for the very near term (2014 to 2015). Planned investment amounts varied for larger firms, but three-quarters indicated they would spend at least $250,000. For small firms, most indicated that their investments over the next 3 years would be $250,000 or less. However, 42 percent of small firms were uncertain of their year-over-year investment plans for 2014-2015. When this 42 percent is coupled with the 76 percent of small firms indicating they would invest over the next 3 years, it suggests more comfort with planning over the longer-term (3 year) planning horizon.

For most investment areas surveyed, large firms showed a tendency to plan a greater level of investment than did small firms. Investment plans for large firms also appeared to be broad, with nearly a third planning to invest in areas ranging from sales force expansion to inventory reduction to panel processing and assembly.

Similar to previous research (Buehlmann et al. 2013), advertising and marketing communications was an especially important investment area for small firms. This is likely related to the importance of maintaining a consistent cash flow to smaller firms (Huang and Brown 1999), especially when facing some uncertainty regarding the current business environment. Furthermore, Ballantine et al. (1993) claim that small firms face inherent uncertainty regarding profitability in general (i.e., regardless of overall business conditions).

Small firms showed a tendency to rate most potential drivers of investment plans as less important than did large firms, suggesting small firms perceive fewer possible benefits to investment (especially investments related to replacing aging equipment, improving productivity, and reducing labor costs). However, the resource constraints under which small firms operate might be contributing to a link between investment plans for small firms and the perceived benefits of investment.

Overall, the results suggest that investment activity in the secondary woodworking industry might be expected to increase in the near term. However, the study also showed a business environment that remains somewhat uncertain for many smaller firms. It is important to understand investment patterns for secondary woodworkers since investments can improve productivity and ultimately contribute to the competitiveness of the industry.

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Use of trade names in the manuscript does not constitute endorsement of any product or service.

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FOREST DYNAMICS

CROWN CLASS DYNAMICS OF OAKS AFTER COMMERCIAL THINNING IN WEST VIRGINIA: 30-YEAR RESULTS

Gary W. Miller, Jamie L. Schuler, and James S. Rentch[1]

Abstract.—Commercial thinning in hardwood stands is generally applied to reduce overcrowding and to favor the development of desired residual species until the stand is mature. In mixed hardwood stands, commercial thinning also provides an opportunity to promote vigorous overstory oaks (*Quercus* spp.) that will serve as sources of acorns and advanced seedlings needed to regenerate oaks in the next stand. Forest managers need information on sustaining and increasing the number of overstory oaks at mid-rotation when the stand is still several decades from maturity. In this study, crown class dynamics of 897 northern red (*Quercus rubra*), chestnut (*Q. montana*), and scarlet (*Q. coccinea*) oaks were monitored for 30 years after commercial thinning in 53-year-old central Appalachian mixed hardwood forests. Twenty 3-acre treatment plots were included in the study, and individual trees were examined immediately after thinning in 1983 and again in 2013. Fisher's exact test was used to compare the distributions of ending canopy position for thinned and control plots. In general, thinning enhanced crown class stability and survival rates of oaks that began in the upper canopy. For oaks that began in the intermediate crown class, the transition rates to the upper canopy after thinning were 33, 25, and 0 percent for northern red oak, chestnut oak, and scarlet oak, respectively. In control plots, oaks that began in the lower canopy had greater mortality rates, and very few trees ascended to the upper canopy. Forest managers can use this information to plan mid-rotation thinning treatments to enhance upper canopy species composition in the latter stages of stand development.

INTRODUCTION

If an adequate seed source is present in a mature oak (*Quercus* spp.) stand, the forest manager at least has the basic resource needed to regenerate and sustain oak species in the next stand. With careful management of seedling cohorts that result from periodic acorn crops, the forest manager can promote advanced reproduction that is large enough and abundant enough to compete successfully once the new stand develops (Dey 2014). For example, decades of research has shown that successful oak regeneration depends on the presence of competitive oak seedlings, sprout sources, or both at the time of a stand-replacing disturbance (Loftis 2004). In most cases, preparatory treatments are needed to develop competitive seedlings by focusing on the oak regeneration process 10 to 20 years before a final overstory harvest (Brose et al. 2008). These preparatory treatments include shelterwood methods to manage sunlight levels, herbicides and prescribed fires to control interfering plants, and other measures to mitigate deer browsing impact. All of these treatments focus on enhancing the growth and survival of advanced oak seedlings in preparation for an overstory harvest. Once an adequate cohort of competitive advanced oak seedlings is present, the forest manger has a reasonable assurance that oak will compete successfully in the next stand. It follows that sustaining or restoring oak

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species composition in mixed hardwood stands depends on the presence of acorn-producing trees 10 to 20 years before overstory trees are removed.

What can the forest manager do to restore an adequate oak seed source if the seed source has been depleted by previous harvests? What can the forest manager do to enhance a marginal seed source that may or may not produce enough acorns and seedlings to even begin the regeneration process described above? More attention is needed on the difficult task of enhancing the number and vigor of upper canopy oaks that are present several decades before the regeneration process begins. Commercial thinning at mid-rotation provides an opportunity to restore and promote vigorous overstory oaks that will later serve as seed trees and sources of advanced seedlings necessary to regenerate oaks in the next stand. Thinning treatments provide individual oaks with a partial or full crown release, thus allowing for faster growth, crown expansion, and increased vigor. In some cases, the crown release allows subordinate oaks to ascend to the upper canopy where over time they can become the vigorous seed trees needed for abundant acorn crops in the future.

This report describes crown class dynamics of 897 northern red (*Q. rubra*), chestnut (*Q. montana*), and scarlet (*Q. coccinea*) oaks that were monitored for 30 years after commercial thinning in 53-year-old central Appalachian mixed hardwood forests in West Virginia. Forest managers may find this information useful as they prepare mixed hardwood stands at mid-rotation to produce more abundant acorn crops many years later near the end of the rotation. Without mid-rotation thinning, subordinate oaks tend to remain so. Ward and Stevens (1994) reported 60-year crown class dynamics for northern red oaks beginning at age 25 in stands where thinning was not applied, and they found that oaks in the lower canopy exhibited greater mortality and crown class regression than those in the upper canopy. Crown release, a benefit of applying thinning treatments to individual trees, increases the probability of oaks reaching and persisting in a dominant or codominant crown class, with response varying depending on their initial crown class and competitive status (Miller et al. 2007, Rentch et al. 2009). Perkey et al. (2011) reported that growth and crown expansion of codominant oaks was related to the degree of crown release over a 20-year period. More important, within the long-term strategy of preparing stands to produce more acorns, there is a positive relationship between canopy position and the capacity of individual oaks to maintain vigor, grow faster, and produce more abundant acorn crops (Healy 1997, Johnson et al. 2009, Trimble 1969). In short, vigorous oaks in the upper canopy produce more acorns than those in the lower canopy. Crown class dynamics reported here provide the forest manger with a basis for predicting the benefits of thinning in terms of retaining oaks in the upper canopy and stimulating oaks in the lower canopy to ascend to the upper canopy where they can become better seed producers.

STUDY SITE

The study area is a second-growth mixed hardwood forest located on the West Virginia University Research Forest, near Morgantown, WV. The overstory composition included yellow-poplar (*Liriodendron tulipifera*), northern red oak, chestnut oak, white oak (*Q. alba*), red maple (*Acer rubrum*), and black cherry (*Prunus serotina*). The species composition was typical of the mixed mesophytic forest described by Braun (1950), with a site index of 75 feet for northern red oak, base age 50 years. Elevation on the study site averaged 2,100 feet, and annual precipitation averages 51 inches and is evenly distributed throughout the year (Carvell 1983). Soils are characterized as Dekalb series (loamy-skeletal, mixed, mesic Typic Dystrochrept) which are moderately deep, well-drained soils of low natural fertility on slopes and ridgetops (Soil Conservation Service 1959). The forest in the study area had its origin in 1930 after heavy logging for charcoal and lumber, thus producing an even-aged stand that was 53 years old when this study was installed.

METHODS

Twenty 3-acre square treatment plots were established in 1982 (Brock et al. 1986). A ½-acre square measurement plot was established within each treatment plot, thus allowing for a similarly treated buffer area around each measurement plot. Five plots were randomly assigned to each of three thinning treatments and a control. The treatments represented light, medium, and heavy thinning defined as 75, 60, and 45 percent residual relative density (RD) as defined by Gingrich (1967). The treatments were a form of crown thinning described by Smith (1962), with removals distributed to release desirable intermediate and upper canopy trees by removing low vigor and low value species until the desired residual density was reached. The residual stand generally included crop trees containing high-quality wood products of preferred species such as yellow-poplar, northern red oak, and black cherry. All trees within the ½-acre measurement plots were assessed before and immediately after commercial thinning in the spring of 1983. Initial crown class for each tree was defined as the crown class observed in 1983 when the stand was being marked for treatment prior to thinning. Subsequent survival and crown class assessments took place in 1993, 2003, and 2013.

Crown class was based on definitions proposed by Smith (1962). Dominant trees have crowns that rise above the general canopy and receive full light from above and partial light from the sides. Codominant trees have crowns in the upper canopy but are largely blocked by neighboring crowns from receiving light from the sides. Intermediate crowns receive a little light from above and no light from the sides, and suppressed trees have completely overtopped crowns and receive only diffuse light beneath the overstory.

Crown class transition rates were analyzed for northern red oak, chestnut oak, and scarlet oak located in thinned and control plots. Each tree was assigned an initial crown class and was then assigned to one of three outcomes at the end of the 30-year study: upper canopy (dominant or codominant), lower canopy (intermediate or suppressed), or dead. Count data from initial crown class to one of the three outcomes were converted to transition rates, defined as the proportion of trees in a given initial crown class that finished in the upper canopy, lower canopy, or died at the end of the study period. The distribution of outcomes among treatments was compared using Fisher's exact test (SYSTAT 2008). For each species and initial crown class, the distribution of outcomes in control plots was compared to that found in all thinning levels combined, followed by pairwise comparisons among the three thinning levels. The significance level for comparisons was $\alpha = 0.05$.

RESULTS

Crown class changes of residual oaks in this study were examined after 20 years and results are reported in Rentch et al. (2009). The 30-year results reported here exhibit similar trends, although in control plots the fate of oaks has gotten worse in the third decade for all species and all initial crown classes. As the untreated stand aged from 73 to 83 years, a few dominant and codominant oaks slipped to the lower canopy, and mortality continued among the intermediate and suppressed oaks. In the thinned plots, upper canopy oaks were relatively stable in the third decade, while those in the lower canopy suffered additional mortality. Canopy gaps created by the removal of merchantable trees in the thinning treatments were present for 15 to 20 years after thinning, gradually closing as the crowns of the residual trees expanded. Most canopy gaps in the thinned plots had closed by the 20^{th} year, thus the increasingly dense overhead canopy continued to have a negative impact on the shorter, lower canopy oaks during the final decade.

Response of Upper Canopy Oaks

There was no significant difference in the response of dominant trees in thinned plots compared to those in control plots. For northern red oaks, more than 90 percent of dominant trees remained in the upper canopy, only one tree died after thinning, and two trees in control plots slipped to an intermediate crown class (Table 1). For chestnut oaks (Table 2) and scarlet oaks (Table 3), more than 65 percent of dominant trees remained in the upper canopy in both thinned and control plots.

Thinning significantly improved both survival and crown class stability of codominant trees for all three oak species (Tables 1-3). In thinned plots, only 15 percent of codominant northern red oaks descended to the lower canopy or died, while 85 percent remained in the upper canopy. In control plots, 45 percent of codominant northern red oaks descended to the lower canopy or died. Chestnut oaks and scarlet oaks exhibited similar responses to thinning. In control plots, about 80 percent of codominant chestnut oaks and scarlet oaks descended to the lower canopy

Table 1.—Thirty-year crown class transition rates for northern red oaks in a 53-year-old central Appalachian hardwood stand after thinning

Initial crown class	Initial canopy status	Treatment	n	30-year canopy status: Upper canopy	Lower canopy	Dead	Pairwise comparisons[a]	
				---------------- percent ----------------				
Dominant	Upper	Control	28	93	7	0	a	a
		75% RD[b]	10	100	0	0	a	
		60% RD	12	92	0	8	a	
		45% RD	3	100	0	0	a	
		Thinned[c]	25	96	0	4		a
Codominant	Upper	Control	49	55	27	18	a	a
		75% RD	39	85	5	10	b	
		60% RD	31	77	13	10	ab	
		45% RD	36	92	3	5	b	
		Thinned	106	85	7	8		b
Intermediate	Lower	Control	34	0	32	68	a	a
		75% RD	30	17	50	33	ab	
		60% RD	20	45	45	10	b	
		45% RD	10	60	20	20	b	
		Thinned	60	33	44	23		b
Suppressed	Lower	Control	99	0	29	71	a	a
		75% RD	38	0	29	71	ab	
		60% RD	36	0	58	42	b	
		45% RD	7	0	43	57	ab	
		Thinned	81	0	43	57		b

[a] For pairwise comparisons within each category of initial crown class, rows with different letters are significantly different from expected values.
[b] RD=relative density
[c] All thinned combined.

or died, while about 20 percent remained in the upper canopy. In thinned plots, 79 percent of codominant chestnut oaks and 67 percent of codominant scarlet oaks remained in the upper canopy 30 years after thinning.

Response of Lower Canopy Oaks

Among oaks in the suppressed crown class, thinning generally increased their survival (Tables 1-3). In control plots, more than 70 percent of suppressed oaks died during the study. Mortality dropped to 57 percent for northern red oaks and 32 percent for chestnut oaks in thinned plots. The sample size for suppressed scarlet oaks was too small to draw any meaningful conclusions; two trees in a control plot died, while one tree in a thinned plot survived. Of 256 suppressed oaks observed in this study, none ascended to the upper canopy in either thinned or control plots after 30 years.

Table 2.—Thirty-year crown class transition rates for chestnut oaks in a 53-year-old central Appalachian hardwood stand after thinning

				30-year canopy status				
Initial crown class	Initial canopy status	Treatment	n	Upper canopy	Lower canopy	Dead	Pairwise comparisons[a]	
				---------------- percent	----------------			
Dominant	Upper	Control	14	71	29	0	a	a
		75% RD[b]	5	80	20	0	a	
		60% RD	5	80	0	20	a	
		45% RD	9	78	22	0	a	
		Thinned[c]	19	79	16	5		a
Codominant	Upper	Control	45	20	53	27	a	a
		75% RD	34	79	21	0	b	
		60% RD	18	94	6	0	c	
		45% RD	25	68	20	12	c	
		Thinned	77	79	17	4		b
Intermediate	Lower	Control	54	6	31	63	a	a
		75% RD	8	12	63	25	ab	
		60% RD	14	29	71	0	b	
		45% RD	2	50	50	12	b	
		Thinned	24	25	67	8		b
Suppressed	Lower	Control	42	0	26	74	a	a
		75% RD	12	0	67	33	b	
		60% RD	14	0	71	29	ab	
		45% RD	5	0	60	40	ab	
		Thinned	31	0	68	32		b

[a] For pairwise comparisons within each category of initial crown class, rows with different letters are significantly different from expected values.
[b] RD=relative density.
[c] All thinned combined.

Thinning increased survival of oaks in the intermediate crown class as well. In control plots, mortality was 68, 63, and 89 percent of intermediate northern red oaks, chestnut oaks, and scarlet oaks, respectively (Tables 1-3). In thinned plots, mortality of intermediate oaks was less than 25 percent for all three species. One of the key benefits of thinning oak stands at mid-rotation is the opportunity to increase the number of upper canopy oaks, primarily by stimulating intermediate oaks to ascend to codominant status. None of the intermediate northern red oaks or scarlet oaks ascended to codominant status in control plots, and only 3 of 54 chestnut oaks moved up in control plots. In thinned plots, 33 percent of northern red oaks and 25 percent of chestnut oaks ascended from intermediate to codominant status after 30 years. Among the northern red oaks that moved to the upper canopy, 65 percent did so within the first 10 years after thinning, 20 percent moved up in the second decade after thinning, and another 15 percent moved up in the third decade. Among the chestnut oaks that moved to the upper canopy, 83 percent did so in the first 10 years after thinning, the remaining 17 percent moved up in the second decade, and no chestnut oaks moved up in the third decade. None of the intermediate scarlet oaks ascended to codominant status in the thinned plots; however, the sample size was relatively small.

Table 3.—Thirty-year crown class transition rates for scarlet oaks in a 53-year-old central Appalachian hardwood stand after thinning

Initial crown class	Initial canopy status	Treatment	n	30-year canopy status: Upper canopy	Lower canopy	Dead	Pairwise comparisons[a]	
				-------- percent --------				
Dominant	Upper	Control	19	78	11	11	a	a
		75% RD[b]	4	50	0	50	a	
		60% RD	1	100	0	0	a	
		45% RD	7	72	14	14	a	
		Thinned[c]	12	67	8	25		a
Codominant	Upper	Control	28	18	36	46	a	a
		75% RD	30	67	23	10	b	
		60% RD	0	0	0	0		
		45% RD	3	67	0	33	b	
		Thinned	33	67	21	12		b
Intermediate	Lower	Control	9	0	11	89	a	a
		75% RD	4	0	75	25	a	
		60% RD	0	0	0	0		
		45% RD	1	0	100	0	b	
		Thinned	5	0	80	20		b
Suppressed	Lower	Control	2	0	0	100	a	a
		75% RD	1	0	100	0	b	
		60% RD	0	0	0	0		
		45% RD	0	0	0	0		
		Thinned	1	0	100	0		b

[a] For pairwise comparisons within each category of initial crown class, rows with different letters are significantly different from expected values.

[b] RD=relative density.

[c] All thinned combined.

DISCUSSION

Forest management objectives for sustaining or increasing the number of overstory oaks might include seed production for wildlife food, accumulation of seed trees for future regeneration, increasing species diversity, and enhancing high quality wood production. The basal area threshold for adequate acorn production and associated advanced oak reproduction is approximately 40 square feet per acre of overstory oaks in the mid-Atlantic region (Brose et al. 2008). Depending on the average diameter of overstory trees, forest managers need to promote 30 to 40 upper canopy oaks per acre as the stand nears maturity to assure an adequate seed source is present at the beginning of the oak regeneration process. Thinning treatments at key stages allow oaks to maintain or ascend to an upper canopy position, particularly on high-quality sites where numerous competing species exhibit faster early height growth (Dey 2014, Miller et al. 2007, Oliver 1980). Residual stand goals for the number of upper canopy oaks can guide "cut-or-leave" criteria when the stand is marked for thinning. In general, enough oaks in both upper and lower canopy positions should receive an adequate crown release after thinning to achieve the desired overstory species composition.

To apply the crown class dynamics presented here, forest managers first need a tally of both upper canopy oaks and those in the intermediate crown class. Because none of the suppressed oaks ascended to the upper canopy after thinning, they lack the potential to be reliable seed trees in the future. Suppressed oaks could be retained after thinning to serve as potential sprout sources at final harvest, but they promise little in the way of future acorns or advance seedlings during the regeneration process. If the stand contains an adequate number of upper canopy oaks before thinning, then the thinning treatment should be applied to favor their crown development and retain them in the upper canopy. In this study, retention of upper canopy status 30 years after thinning was about 90 percent for northern red oak, 80 percent for chestnut oak, and 70 percent scarlet oak. Crown class retention of codominant oaks dropped significantly in control plots. If ascension of intermediate oaks is required to achieve residual stand goals for upper canopy oaks, the thinning treatment must release several intermediate oaks in order to promote one to the upper canopy. In this study, 33 percent of intermediate northern red oaks and 25 percent of intermediate chestnut oaks ascended to the upper canopy after thinning. Marking guidelines could include "release 3 to get 1" for northern red oak and "release 4 to get 1" for chestnut oak if the thinning is planned about 30 years before final harvest. No intermediate scarlet oaks ascended to the upper canopy after thinning.

The timing of the first commercial thinning is a function of site quality and local wood markets. On relatively high quality sites, upper canopy trees may reach merchantable size classes sooner than on lower quality sites. In addition, local wood markets and general economic conditions often affect the merchantability of the relatively low volumes associated with thinning harvests. If sustaining and promoting upper canopy oaks for future acorn production is a priority, forest managers should consider applying the first thinning as early as possible in the rotation. In cases where the number of upper canopy oaks is a concern, it might be advantageous to apply a precommercial crown release to selected oaks before the first commercial thinning, thus enhancing their status until commercial operations are feasible.

Traditional silvicultural criteria should be used to decide which oaks to favor as residual trees in the upper or lower canopy. Residual trees should have relatively good vigor and full crowns with little evidence of excess branching or crown dieback. Residual oaks with good initial stem quality tend to retain good quality after a thinning treatment (Miller et al. 2008). Likewise, trees with excess branching or evidence of poor quality before thinning tend to become worse after thinning. In making cut-or-leave decisions among intermediate trees before thinning, it is recommended that preference be given to retaining oaks that exhibit better-than-average initial stem quality. Visible stem quality may indicate good genetic characteristics, as well as favorable microsite conditions, that would allow the tree to respond well to the thinning treatment.

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DEVELOPMENT, SUCCESSION, AND STAND DYNAMICS OF UPLAND OAK FORESTS IN THE WISCONSIN DRIFTLESS AREA: IMPLICATIONS FOR OAK REGENERATION AND MANAGEMENT

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Abstract.–Throughout the deciduous forests of the eastern United States, oak (*Quercus*) regeneration has declined in stands historically dominated by oak species. In the Wisconsin Driftless Area, the level of decline in oak regeneration is variable and influenced by stand structural development, historical disturbance regime, abiotic site characteristics, and historical land-use legacies. This research aims to elucidate the relative importance and site-specific nature of the factors influencing oak regeneration and identify interactions between drivers. To document stand succession, development, and factors influencing oak regeneration, we collected forest inventory and dendrochronological data from 11 sites distributed throughout the region. The stages of structural development identified across sites included pole, mature-sapling mosaic, mature, early-transition old-growth, and steady-state old-growth. Structural complexity was variable across the region but generally increased with advanced structural development. At all sites, oak regeneration was negligible, with significantly lower relative densities of oak species in the understory than in the overstory. Future analyses will use multivariate techniques to investigate the relationship between stand development, land-use history, disturbance history, abiotic conditions, and oak regeneration.

INTRODUCTION

Oak species (*Quercus* spp.) are a major component of forested ecosystems of the eastern United States, and the ecological, commercial, and societal importance of these oak forests represents a distinctive long-term legacy. However, throughout the Central Hardwood Forest region, a majority of oak species are failing to regenerate, and oak abundance is exhibiting a marked decline (Abrams 1992, Fei et al. 2011, Lorimer 1993). Coupled with this decline, shade-tolerant mesophytic species such as sugar maple (*Acer saccharum* Marsh.) and red maple (*Acer rubrum* L.) are increasing in abundance, particularly in the understory. This compositional shift has been documented across multiple site types and has led many researchers to predict a pervasive transition from oak-dominated systems to forests with overstories dominated by maple and other shade-tolerant taxa. Active fire suppression beginning in the early 20th century is the most often cited explanation for the oak replacement pattern. Additional drivers of the transition have been proposed and include alterations of canopy disturbance characteristics, changes in moisture regime, and the influence of differential historical land uses (Lorimer 1993, McEwan et al. 2011).

In the Driftless Area of southwestern Wisconsin, declines in the abundance of oak species regeneration have been documented throughout the past ca. 40 years (Hix and Lorimer 1991, Hix et al. 2005, Lorimer et al. 1994, Martin et al. 1992, Povak et al. 2008). The mosaic pattern of both the presettlement and modern vegetation represents the multitude of developmental

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pathways that have occurred in the region. These stand developmental pathways are dependent on factors that influence the initial stand composition and structure (e.g., previous land use, seed source availability, local and regional climate, microenvironmental abiotic conditions) as well as dynamic processes that operate throughout stand development (e.g., stand disturbance events, extreme weather events, silvicultural operations). While the decline in oak species regeneration has been documented throughout the Wisconsin Driftless Area (WDA), the degree of oak regeneration decline is variable and influenced by a given site's historical forest development, disturbance regime, moisture regime, and abiotic site characteristics. Here, we present results that quantify structural development, structural complexity, and the status of oak regeneration throughout the Wisconsin Driftless Area.

METHODS

To examine the development, succession, and stand dynamics of upland oak stands in the region, we collected forest inventory and dendrochronological data from 11 study sites distributed throughout the Driftless Area in southwestern Wisconsin (Fig. 1). The Wisconsin Driftless Area, located in the northern portion of the Central Hardwood Forest region, represents a unique physiographic province in the upper Great Lakes region because it was not glaciated during the last glacial maximum (Martin 1916). The disturbance regime in southwestern Wisconsin prior to widespread EuroAmerican settlement included frequent, low-intensity ground fires of lightning and Native American origin and wind events (Curtis 1959, Johnson 1976). However, the modern disturbance regime has changed considerably, caused most notably by the onset of active fire suppression in the early 20th century and the conversion of prairie and savanna landscapes to agriculture or closed-canopy forests (Curtis 1956, Rhemtulla et al. 2007). These changes in land use and land management have not only influenced the composition and structure of the WDA forested landscape, but have also affected the frequency and intensity of

Figure 1.—Map of Wisconsin showing the Wisconsin Driftless Area (shaded gray) and site locations (black dots).

the modern disturbance regime. Because oak species are only moderately tolerant of shade, the frequency and intensity of modern disturbances (whether caused by natural or anthropogenic means) have a large influence on the regeneration success of oak species in the WDA by removing portions of the canopy and increasing light levels in the forest understory.

Across the 11 study sites, we established 34 circular plots 0.04-ha in size and recorded the species, diameter at breast height (d.b.h.), and crown class for all trees (stems ≥10 cm d.b.h. and ≥1.37 m height) and saplings (stems ≤10 cm d.b.h. and ≥1.37 m height). All plots were located within stands that the Wisconsin Department of Natural Resources classified as being dominated by oak or central hardwood species and that established prior to 1940. To analyze stand age structure, disturbance history, and climate-growth relationships, we collected one increment core sample from all stems ≥10 cm d.b.h. within each plot. After the tree-ring samples were processed in the laboratory, tree-ring widths were measured to the nearest 0.001 mm and the measurement series were statistically compared to ensure accurate crossdating. Sampled stands were multi-aged, so the stage of structural development was determined using the median age of canopy trees and a modified version of the decision tree developed by Lorimer and Halpin (2014). We constructed diameter distributions with 5 cm size-class intervals and determined the general trend of each distribution by fitting polynomial functions to the semilogarithmic-transformed diameter distributions using all possible combinations of d.b.h., d.b.h.^{2}, and d.b.h.^{3}. The best-fitting model was selected from significant models (p < 0.05) based on the highest adjusted r^2 and the lowest root mean square error. Distribution shape was determined by examining the model variables that were present and the sign of the model coefficients. We calculated several indices that quantify structural diversity: standard deviation$_{dbh}$, the coefficient of variation$_{dbh}$, the Gini coefficient$_{dbh}$, and the Shannon diversity index (H'_{dbh}). Indices of oak species regeneration and maple species regeneration were calculated for each site as the difference between the relative density of understory stems and the relative density of overstory stems.

RESULTS

In the Wisconsin Driftless Area, site-level density values of all saplings and trees (i.e., all stems ≥1.37 m height) ranged between 919 and 2,058 stems/ha and site-level basal area values ranged between 20.5 and 47.1 m^2/ha (Table 1). Across the region as a whole, forest canopies were dominated by oak species (Table 2), while the sapling layer was dominated by more mesic species including sugar maple and red maple (Table 3). This pattern of a predicted compositional shift was further evidenced by negative regeneration indices for oak and positive regeneration indices for maple throughout the region (Table 4). The oldest documented oak tree established at breast height in 1825 while the youngest sampled oak tree established at breast height in 1992. Across the 11 sites, stages of stand structural development varied, with stands in the pole, mature-sapling mosaic, mature, early-transition old-growth, and steady-state old-growth stages represented. The progression of structural stages mirrored the progression of increasing density of large trees (≥45 cm d.b.h.) (Table 1). Four diameter distribution shapes were identified across the region: concave, increasing-*q*, negative exponential, and rotated sigmoid (Fig. 2). Two sites did not yield significant regression models. The standard deviation$_{dbh}$, coefficient of variation$_{dbh}$, Gini coefficient$_{dbh}$, and Shannon diversity index (H'_{dbh}) all generally increased with increasing structural development (Table 5).

Table 1.—Site characteristics for 11 upland oak sites within the Wisconsin Driftless Area. Sites are arranged in ascending order by the density of trees ≥45 cm d.b.h. Unless otherwise noted, values were calculated from all sampled saplings (stems ≤10 cm d.b.h. and ≥1.37 m height) and trees (stems ≥10 cm d.b.h.).

Site	Plot n	Density	Basal area	Density of trees ≥45 cm d.b.h.	Basal area of trees ≥45 cm d.b.h.	Quadratic mean d.b.h.[a]	Developmental stage[b]
		stems/ha	*m^2/ha*	*stems/ha*	*m^2/ha*	*cm*	
New Glarus	2	1,675.0	20.5	12.5	4.2	18.7	P
Dunville	1	1,875.0	23.1	25.0	5.2	24.9	MSM
Perrot	4	993.8	23.5	25.0	5.8	24.0	MSM
Nelson Dewey	1	1,200.0	21.4	50.0	11.8	23.9	MSM
Dell Creek	2	1,337.5	38.9	50.0	12.9	25.0	M
Mirror Lake	4	9,18.75	31.1	50.0	9.5	26.4	M
Blue Mound	3	2,058.3	37.1	50.0	9.2	28.8	M
Borst Valley	3	1,341.7	33.8	66.7	16.0	25.7	M
Devils Lake	4	1,512.5	44.4	81.3	17.7	29.0	M
Kickapoo Valley	7	1,796.4	47.1	82.3	25.3	31.0	OG-ET
Wyalusing	2	950.0	43.3	100.0	21.7	29.1	OG-SS

[a] Includes all stems ≥8 cm d.b.h.
[b] P = pole; MSM = mature-sapling mosaic; M = mature; OG-ET = early transition old-growth; OG-SS = steady-state old-growth.

Table 2.—Density, relative density, dominance, relative dominance, and importance value (average of relative density and relative dominance) for all trees (stems ≥10 cm d.b.h.) sampled on upland oak stands in the Wisconsin Driftless Area, sorted by importance value

Species	Density	Relative density	Dominance	Relative dominance	Importance value
	stems/ha		*m^2/ha*		
Quercus rubra L.	95.31	18.77	14.90	42.12	30.44
Quercus alba L.	75.00	14.77	6.84	19.34	17.06
Acer rubrum L.	92.19	18.15	2.19	6.19	12.17
Acer saccharum Marshall	64.06	12.62	2.47	6.97	9.79
Tilia americana L.	27.34	5.38	2.53	7.14	6.26
Ostrya virginiana (Mill.) K. Koch	37.50	7.38	0.51	1.45	4.42
Pinus strobus L.	17.97	3.54	1.45	4.11	3.82
Carya cordiformis (Wangenh.) K. Koch	16.41	3.23	0.75	2.11	2.67
Quercus velutina Lam.	9.38	1.85	1.11	3.15	2.50
Ulmus rubra Muhl.	16.41	3.23	0.59	1.66	2.44
Fraxinus americana L.	14.06	2.77	0.70	1.98	2.37
Juglans nigra L.	15.63	3.08	0.44	1.25	2.16
Carya ovata (Mill.) K. Koch	9.38	1.85	0.32	0.91	1.38
Quercus bicolor Willd.	3.91	0.77	0.06	0.17	0.47
Prunus serotina Ehrh.	3.13	0.62	0.05	0.15	0.38
Acer negundo L.	2.34	0.46	0.11	0.30	0.38
Pinus resinosa Aiton	1.56	0.31	0.11	0.30	0.31
Quercus macrocarpa Michx.	0.78	0.15	0.16	0.46	0.31
Betula papyrifera Marshall	2.34	0.46	0.04	0.13	0.29
Celtis occidentalis L.	1.56	0.31	0.01	0.04	0.17
Robinia pseudoacacia L.	0.78	0.15	0.02	0.06	0.11
Prunus pensylvanica L. f.	0.78	0.15	0.01	0.03	0.09
Total	507.81	100.00	35.38	100.00	100.00

Table 3.—Density, relative density, dominance, relative dominance, and importance value (average of relative density and relative dominance) for all saplings (stems ≤10 cm d.b.h. and ≥1.37 m height) sampled on upland oak stands in the Wisconsin Driftless Area, sorted by importance value

Species	Density	Relative density	Dominance	Relative dominance	Importance value
	stems/ha		*m^2/ha*		
Acer rubrum L.	256.25	26.28	0.64	36.28	31.28
Acer saccharum Marshall	241.41	24.76	0.34	19.60	22.18
Ostrya virginiana (Mill.) K. Koch	146.88	15.06	0.31	17.56	16.31
Ulmus rubra Muhl.	57.03	5.85	0.11	6.53	6.19
Fraxinus americana L.	44.53	4.57	0.05	2.76	3.66
Prunus serotina Ehrh.	39.84	4.09	0.03	1.98	3.03
Quercus alba L.	18.75	1.92	0.05	2.66	2.29
Tilia americana L.	16.41	1.68	0.04	2.27	1.98
Carya cordiformis (Wangenh.) K. Koch	17.97	1.84	0.03	1.82	1.83
Carya ovata (Mill.) K. Koch	12.50	1.28	0.03	1.92	1.60
Zanthoxylum americanum Mill.	28.13	2.88	0.01	0.32	1.60
Celtis occidentalis L.	21.09	2.16	0.02	0.92	1.54
Prunus pensylvanica L. f.	17.97	1.84	0.02	0.92	1.38
Juglans nigra L.	5.47	0.56	0.02	1.02	0.79
Pinus strobus L.	7.81	0.80	0.01	0.66	0.73
Hamamelis virginiana L.	10.16	1.04	0.00	0.24	0.64
Carpinus caroliniana Walter	10.16	1.04	0.00	0.21	0.63
Acer negundo L.	4.69	0.48	0.01	0.64	0.56
Crataegus spp.	4.69	0.48	0.01	0.61	0.54
Euonymus atropurpureus Jacq.	7.81	0.80	0.00	0.07	0.44
Quercus rubra L.	1.56	0.16	0.01	0.51	0.33
Rhamnus cathartica L.	1.56	0.16	0.00	0.27	0.21
Quercus bicolor Willd.	0.78	0.08	0.00	0.25	0.16
Cornus alternifolia L. f.	0.78	0.08	0.00	0.00	0.04
Salix purpurea L.	0.78	0.08	0.00	0.00	0.04
Total	975.00	100.00	1.76	100.00	100.00

Table 4.—Regeneration indices for all species with sapling or tree importance values ≥1, sorted in ascending order by index value

Species	Index value
Quercus rubra L.	-18.61
Quercus alba L.	-12.85
Tilia americana L.	-3.70
Pinus strobus L.	-2.74
Juglans nigra L.	-2.52
Quercus velutina Lam.	-1.85
Carya cordiformis (Wangenh.) K. Koch	-1.39
Carya ovata (Mill.) K. Koch	-0.56
Prunus pensylvanica L. f.	1.69
Fraxinus americana L.	1.80
Celtis occidentalis L.	1.86
Ulmus rubra Muhl.	2.62
Zanthoxylum americanum Mill.	2.88
Prunus serotina Ehrh.	3.47
Ostrya virginiana (Mill.) K. Koch	7.68
Acer rubrum L.	8.13
Acer saccharum Marshall	12.14

Table 5.—Indices quantifying diameter diversity of all stems ≥3 cm diameter at breast height (d.b.h.) for 11 upland oak sites within the Wisconsin Driftless Area. Sites are arranged in ascending order by the density of trees ≥45 cm d.b.h.

Site	Developmental stage[a]	Standard deviation	Coefficient of variation	Gini coefficient[b]	Shannon diversity index[c]	Average *q*-factor (± SE)[d]
New Glarus	P	9.0	0.7	0.3	1.6	1.1 (± 0.6)
Dunville	MSM	12.6	0.9	0.4	1.7	1.4 (± 0.4)
Perrot	MSM	12.1	0.8	0.4	1.8	1.2 (± 0.3)
Nelson Dewey	MSM	12.4	1.1	0.4	1.4	0.9 (± 0.4)
Dell Creek	M	12.8	0.8	0.4	1.9	1.1 (± 0.3)
Mirror Lake	M	13.8	0.7	0.4	2.1	1.5 (± 0.3)
Blue Mound	M	13.9	0.9	0.5	1.7	1.7 (± 0.4)
Borst Valley	M	14.1	0.9	0.4	1.9	1.4 (± 0.4)
Devils Lake	M	14.8	1.0	0.5	1.6	1.6 (± 0.4)
Kickapoo Valley	OG-ET	16.2	1.1	0.5	1.7	1.4 (± 0.3)
Wyalusing	OG-SS	15.7	0.9	0.4	2.0	1.5 (± 0.5)

[a]P = pole; MSM = mature-sapling mosaic; M = mature; OG-ET = early transition old-growth; OG-SS = steady-state old-growth.

[b]Values range from 0 to 1, with 0 corresponding to sites with perfect size equality and 1 representing maximum size inequality.

[c]Values range from 0 to ln(number of diameter classes), with 0 corresponding to all trees being in the same diameter class and maximum values representing an even distribution of trees within diameter classes.

[d]Calculated from 5-cm diameter classes beginning with the 5 cm class.

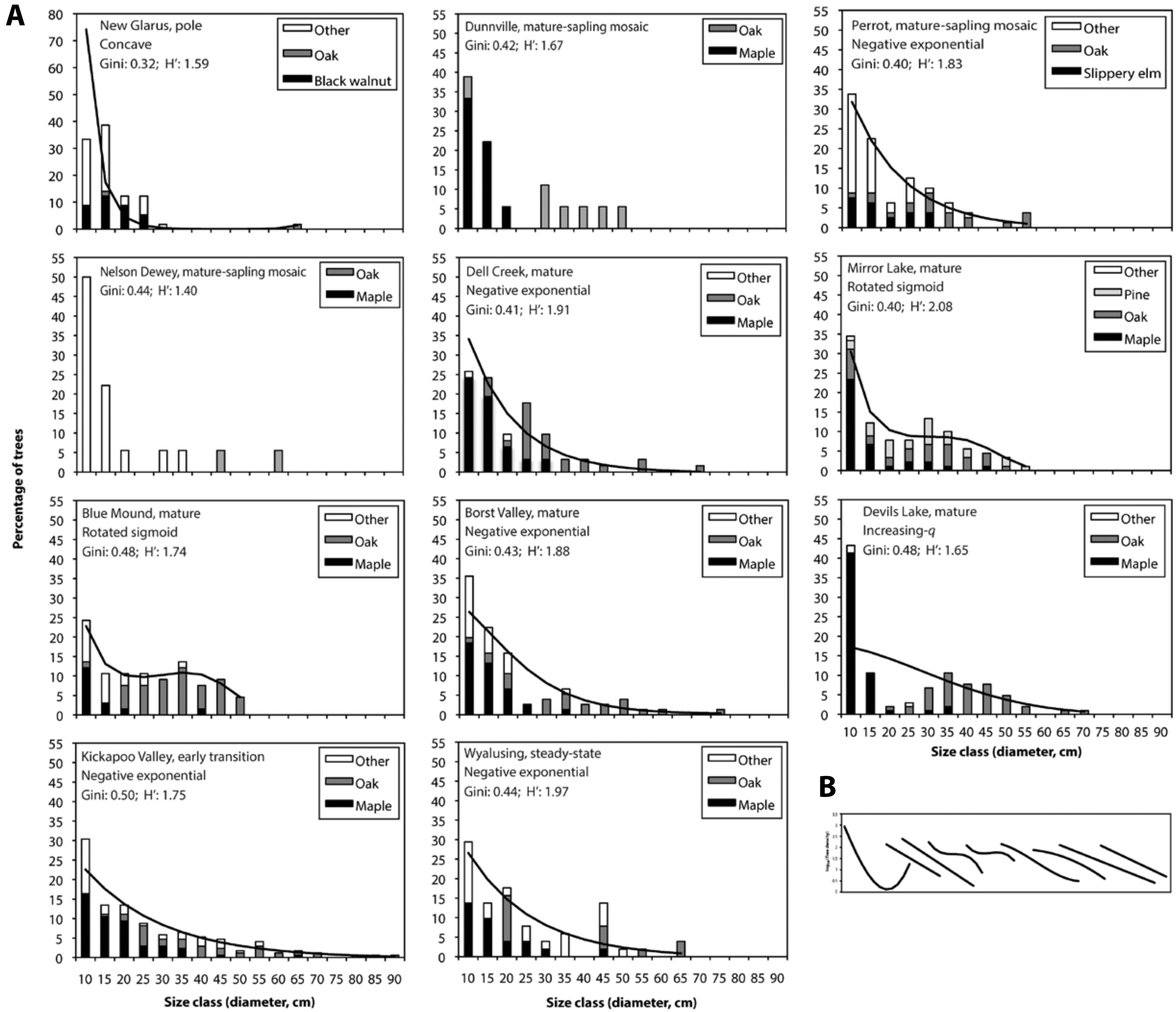

Figure 2.—(A) Size-class distributions (based on relative frequency) for all tree species in the 11 upland oak sites in the Wisconsin Driftless Area. Sites are arranged in ascending order by the density of trees ≥45 cm d.b.h. and regression curves are superimposed on the diameter distributions. (B) Change in the shape of the regression curves for all sites from pole (left) to steady state (right) ordered by density of trees >45 cm d.b.h.

CONCLUSIONS AND FUTURE ANALYSES

The results of this study indicate that the upland oak forest stands of the Wisconsin Driftless Area are transitioning from oak dominance to dominance by more shade-tolerant species such as sugar maple and red maple. Stands throughout the region are in different stages of structural development, and structural diversity generally increased with advancing structural development. Future research will use historical land survey analyses and dendrochronological methods to investigate the influence of historical land-use legacies, disturbance history, and extreme weather events on the composition, structure, and function of the upland oak stands.

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FOREST DYNAMICS AT THE MISSOURI OZARK FOREST ECOSYSTEM PROJECT VIEWED THROUGH STOCKING DIAGRAMS

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Abstract.—Stocking diagrams come in two forms, the Gingrich diagram and the density management diagram. While they both present the same information about a forest stand, they each provide a different perspective on the data being displayed. Density management diagrams have been around since the 1930s and the Gingrich diagram has been around since the 1960s, but applications of the diagrams to investigate long-term forest dynamics have been rare. We apply both types of stocking diagrams to stands at the Missouri Ozark Forest Ecosystem Project to examine change over a 20-year period and to graphically illustrate the temporal trends.

INTRODUCTION

Stocking is a concept used in forestry to describe the space occupied by trees. This concept is similar to carrying capacity applied to animal populations. In a forest management context, stocking provides a tool for measuring and managing the growing space occupied by the trees in a stand. Stocking accounts for the number and size of trees of a given forest type that can partially fill, completely fill, or over-fill the available growing space.

There have been two prevailing approaches to examining stocking graphically. The first graphical approach published in the United States was by Reineke (1933). It used a log-log plot of the number of trees per acre (TPA) by their mean size (Fig. 1A). Reineke started by using mean tree diameter as the measure of tree size, although many other metrics of mean size have been tried. Today, quadratic mean tree diameter (QMD) is the commonly used metric of mean tree size because it is easy to apply to forest inventory data that are commonly available. In Reineke's original publication, the TPA variable was on the y-axis with QMD on the x-axis. It has become convention (Long 1985) to reverse these axes as it makes comparisons with other stocking charts more convenient. Besides the major axes, a Reineke stocking diagram includes a maximum stocking line defined as the maximum size density index (SDI) for the forest type in question (Larsen 2001). By convention, the maximum SDI applies for a given species or forest type, and it indicates the number of 10-inch diameter trees per acre that would occupy the entire growing space (or some given proportion of the growing space). The Reineke size density diagram translates the index for other combinations of mean QMD and TPA. Reineke published SDI parameters for species across the United States. The accepted SDI number for upland oaks in the Central Hardwood region is 230 10-inch diameter trees per acre (Schnur 1937).

The second graphical approach is a distillation of a sequence of papers by Gingrich (1967). In the Gingrich stocking diagram, the graph displays TPA on the x-axis and basal area per acre (BAa) on the y-axis. QMD is simply the diameter of the tree of average BAa which can be computed for any combination of basal area and number of trees per unit area (Fig. 1B).

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Included in the Gingrich stocking diagram are lines indicating threshold stocking levels. The basis for these thresholds originated from the tree area equation of Chisman and Schumacker (1940). In the Chisman and Schumacker paper, the authors proposed an equation to estimate area occupied by a tree, given its diameter at breast height (d.b.h.). They found that a quadratic equation predicted the relationship best. Because the independent variable in the stocking equation is diameter, this means the equation contains both d.b.h. and d.b.h.^{2}, which can be thought of as d.b.h. and basal area because basal area is simply the squared d.b.h. multiplied by a constant. At the time the Chisman and Schumacker paper was developed, people still used normal yield tables, which are tables showing change over time in basal area, volume, and number of trees for undisturbed stands. By using data from normal yield tables, Gingrich could determine the number of trees per acre of a given d.b.h. that would utilize all the growing space on a site. This information can be used to define the line of 100 percent stocking (sometimes called the A-line) on a Gingrich Diagram (see Gingrich 1967 or Larsen 2010 for details) (Fig. 1B).

Krajicek, a coworker of Gingrich in southern Illinois, wanted to develop an index of crowding in a forest. Kriajicek's solution was to measure open-grown trees, which represented trees with the largest possible crown area for a given d.b.h. This led to the concept of crown competition factor (CCF), a metric of the area that would be occupied if all trees on a site had maximum crown area (Krajicek et al. 1961). Crown competition factor is scaled so that a number of 100 indicates that the available space would be fully occupied if all trees were free to grow with their crowns at maximum area. If CCF is less than 100, there is unoccupied growing space on the site. If CCF is greater than 100, the tree crowns are competing with one another, and trees have smaller crowns than open-grown trees with an equivalent d.b.h.

Gingrich used the concept of crown competition factor to identify what is commonly referred to as the B-line on the Gingrich stocking chart. The B-line defines a threshold in stocking where stocking values above the B-line fully occupy the physical space and indicate crowding to some extent (i.e., CCF > 100). Stocking values below the B-line represent conditions where trees do not fully occupy the available growing space (i.e., CCF < 100).

In Gingrich (1967), a third line known as the C-line is displayed on the stocking diagram. The C-line represents stocking levels from which a stand could grow to reach the B-line in 10 years. Nowhere in the Gingrich's papers does he provide an equation or any statistics on this line. We view this as a bit of expert opinion.

STOCKING DYNAMICS

Stocking diagrams are routinely used to assess stand conditions at a single point in time, but most of us have not used stocking diagrams to examine changes over time for repeat-measurement data. With repeat-measurement data for large inventories, a stocking diagram can be produced for each stand or plot, resulting in hundreds or even thousands of individual stocking charts. Most people find viewing that many stocking charts to be tedious. Consequently, this approach is not well suited for a printed publication. This may be why there are so few papers on this subject. Additionally, two-dimensional descriptive statistics have not been well developed to describe subtle differences in stocking trajectories over time.

In general, repeat measurements for undisturbed plots show a trajectory over time of increasing quadratic mean diameter, increasing basal area, and decreasing number of trees per acre. In this paper we illustrate the utility of stocking diagrams for displaying stand development trajectories using repeat-measurements data sets.

METHODS

The data used in this study are from the Missouri Ozark Forest Ecosystem Project (MOFEP), which was implemented by the Missouri Department of Conservation, an agency responsible for natural resource management on state-owned lands as well as for providing landowner assistance. The MOFEP study was established on about 9,000 acres of state forest land along the Current River in Shannon, Reynolds, and Carter counties of Missouri (Shifley and Brookshire 2000). The study included nine sites ranging from 700 acres to 1,300 acres and grouped into three replicated blocks with three component treatments per block. The three treatments were no harvest management (NH), uneven-aged management (UAM), and even-aged management (EAM). On MOFEP sites, an overstory inventory was implemented on 648 1/2-acre, fixed-area plots. These plots were measured in 1992, 1995, 1997, 2001, 2005, 2009, and 2012.

In this paper, a sequence of stocking diagrams illustrates the effects on stocking for each treatment during the 20-year-period of the measurements. Each data set was graphed using both Reineke and Gingrich diagrams to compare the dynamics revealed by each graph type.

RESULTS AND DISSCUSSION

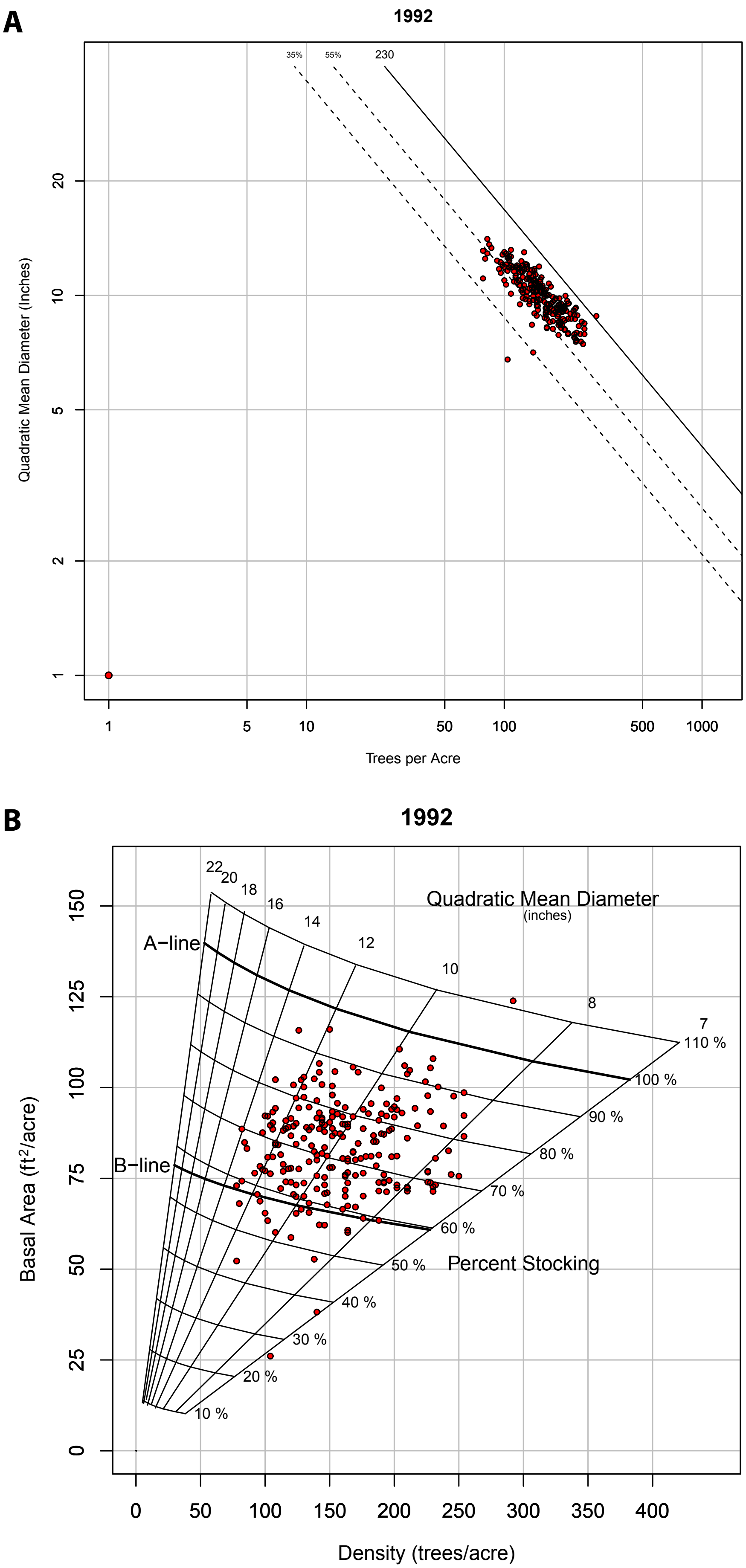
In general, the effect of the NH treatment was to reduce the range of the trees per acre and increase the range of basal area (Figs. 1 and 2), but the center of the plot cloud was at about 80 percent stocking. If we consider the Reineke diagrams (Figs. 1A and 2A), we have changed the trees per acre from about 120 to 150, and the quadratic mean diameter has increased from about 10 to about 12 inches.

For the UAM sites (Figs. 3 and 4), the magnitude of the changes was less than that observed for the EAM sites. Uneven-aged management increased the range in basal area but decreased the range in trees per acre. It also reduced the range in quadratic mean diameter. It is important to note that prior to treatment, these sites were generally fully stocked forests and the densities were reduced by the treatments. This occurred because prior to treatment, the stands were mostly even-aged and harvesting was required to begin to convert them to uneven-aged stands, a process that will require five to six entries to be fully achieved.

On the EAM sites (Figs. 5 and 6), the plots exhibited a greater range of basal area than the other sites, largely because of the number of plots with a low basal area after regeneration harvests were implemented. The range in trees per acre was reduced by a small amount and the quadratic mean diameter increased because of growth following treatment.

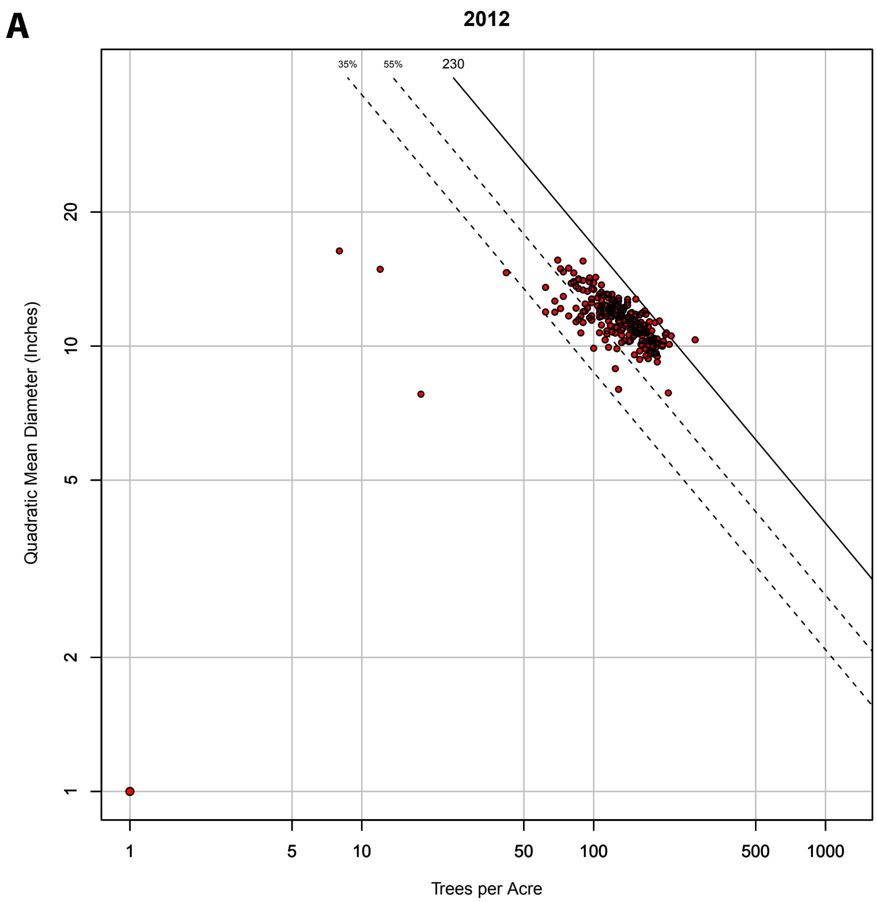
Each of the treatments are readily identifiable in the pattern in each of the stocking diagrams. These patterns provide a visual means for understanding stand development patterns. Another trend more easily observed in the full temporal sequence is that the harvests are not removing all the forest growth, so the stands are accumulating growing stock.

A



B

Figure 1.—Reineke (A) and Gingrich (B) stocking diagrams for the Missouri Ozark Forest Ecosystem Project no harvest (NH) sites in 1992.



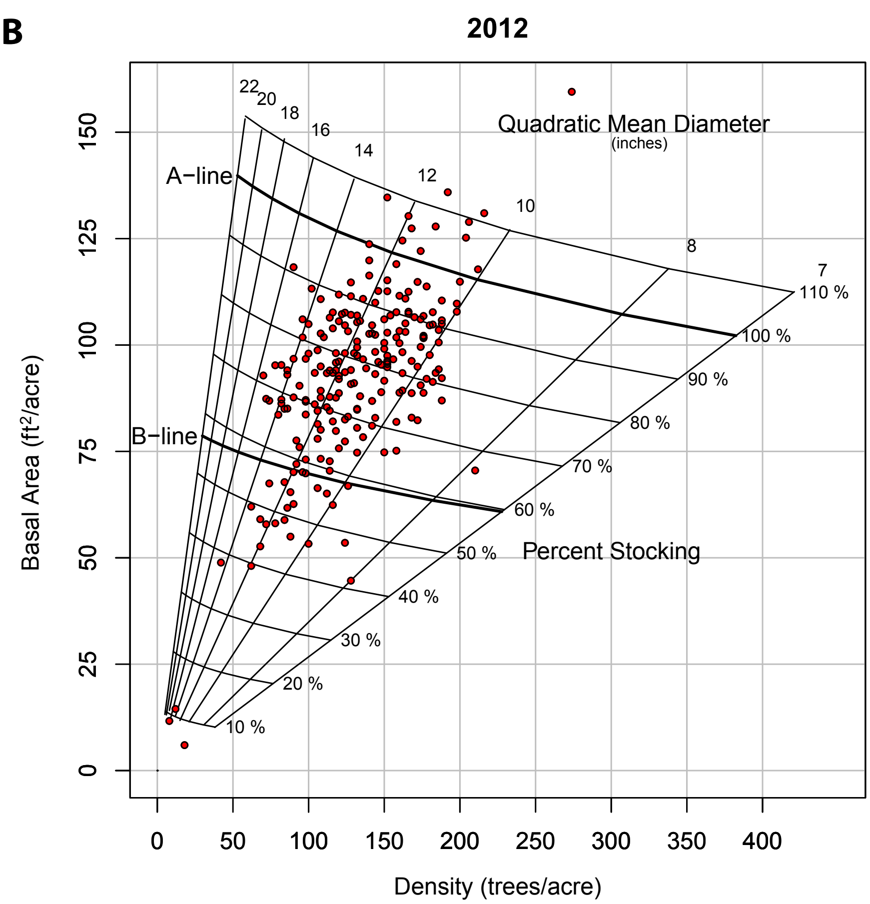
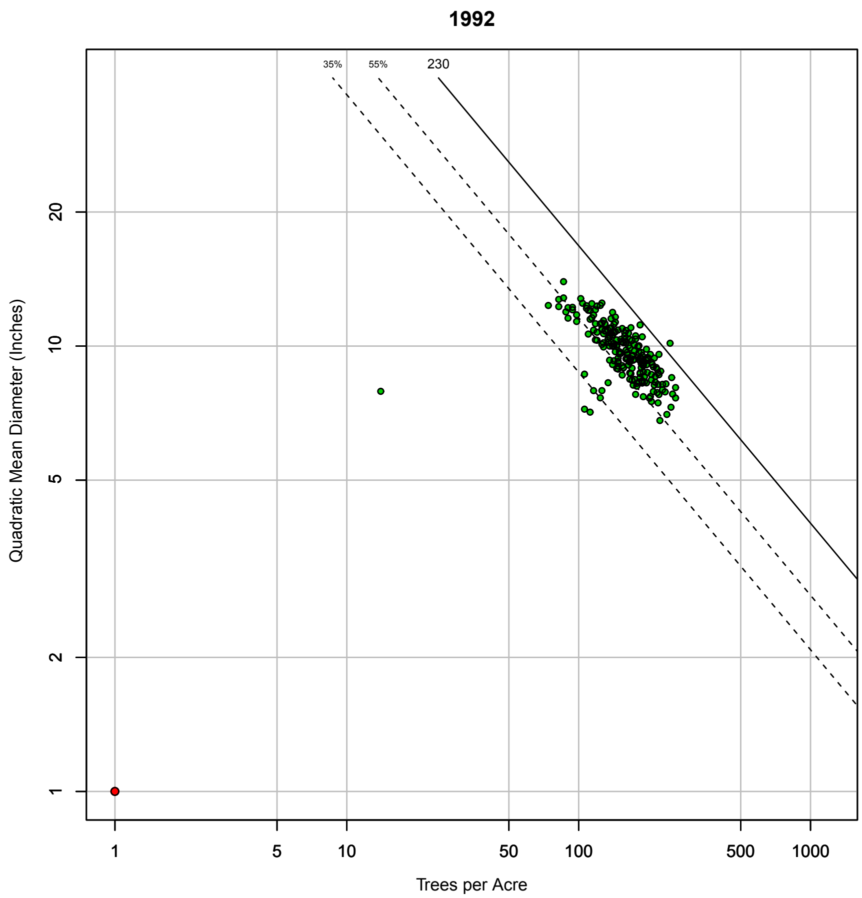


Figure 2.—Reineke (A) and Gingrich (B) stocking diagrams for the Missouri Ozark Forest Ecosystem Project no harvest (NH) sites in 2012.

A



B

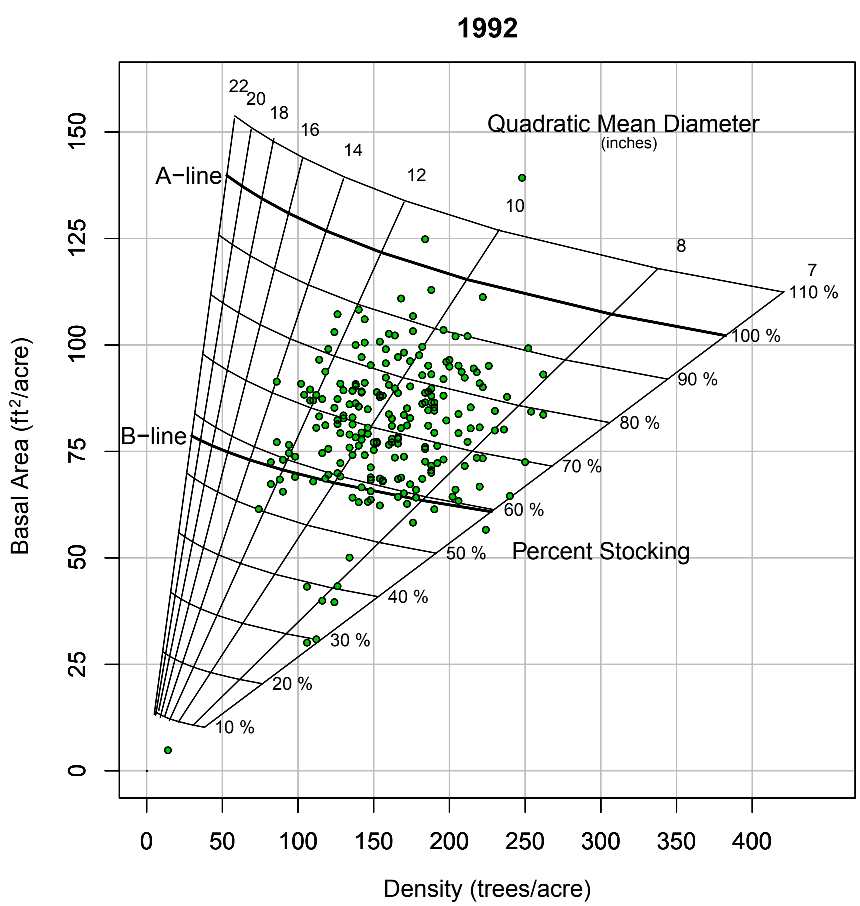
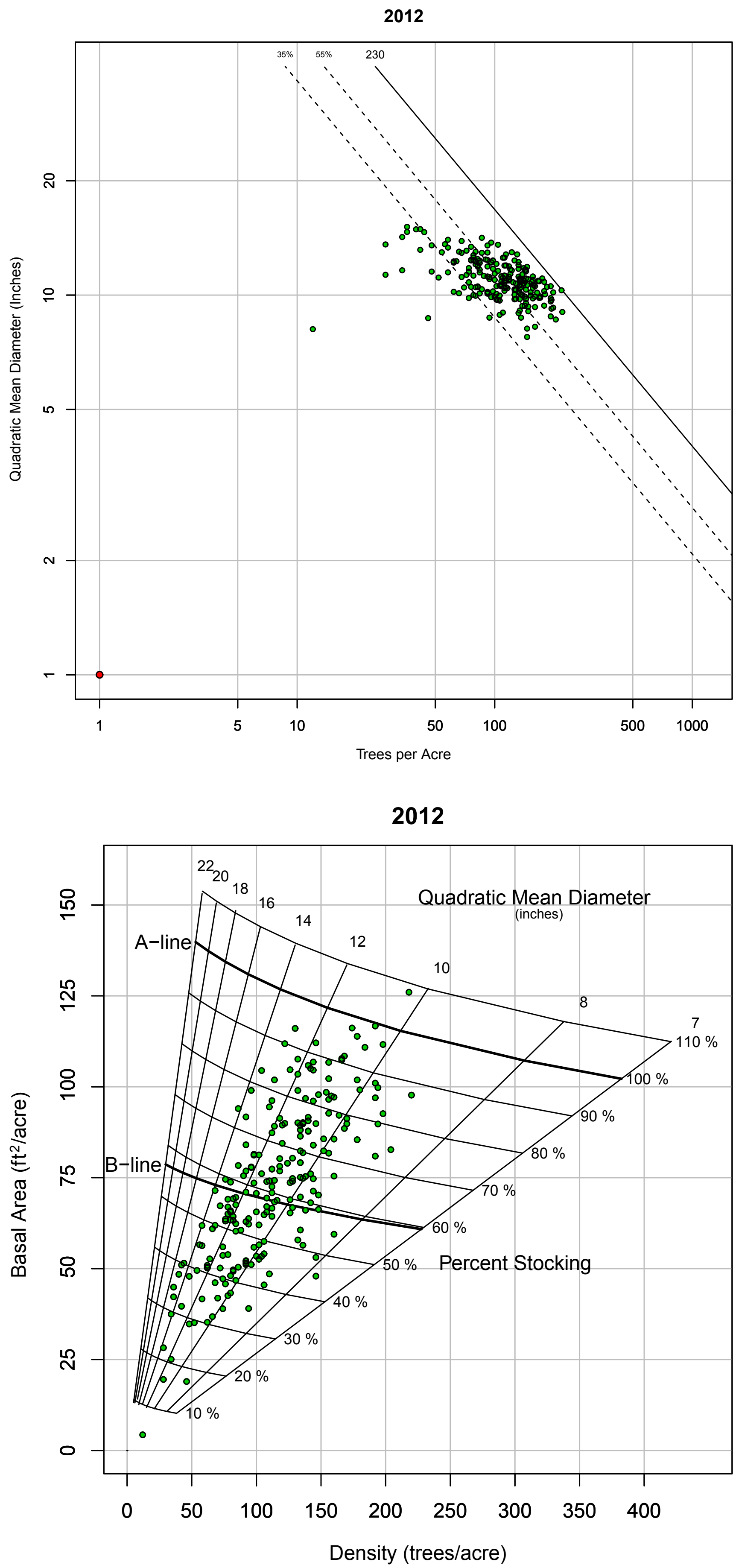


Figure 3.—Reineke (A) and Gingrich (B) stocking diagrams for the Missouri Ozark Forest Ecosystem Project uneven-aged management (UAM) sites in 1992.

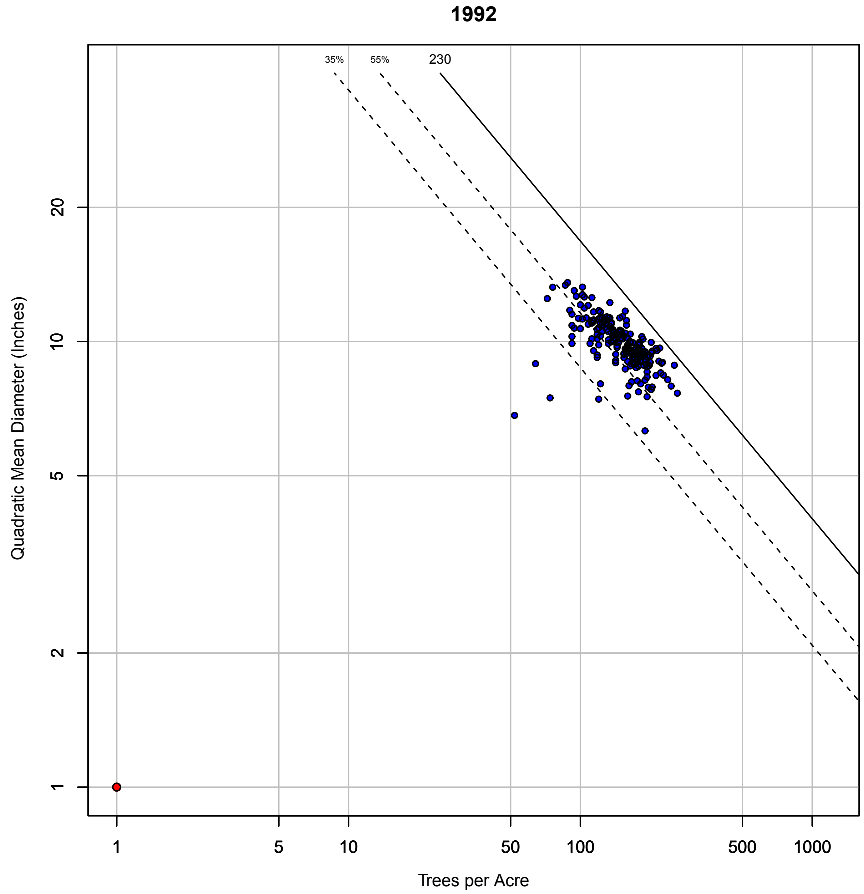
A



B

Figure 4.—Reineke (A) and Gingrich (B) stocking diagrams for the Missouri Ozark Forest Ecosystem Project uneven-aged management (UAM) sites in 2012.

A



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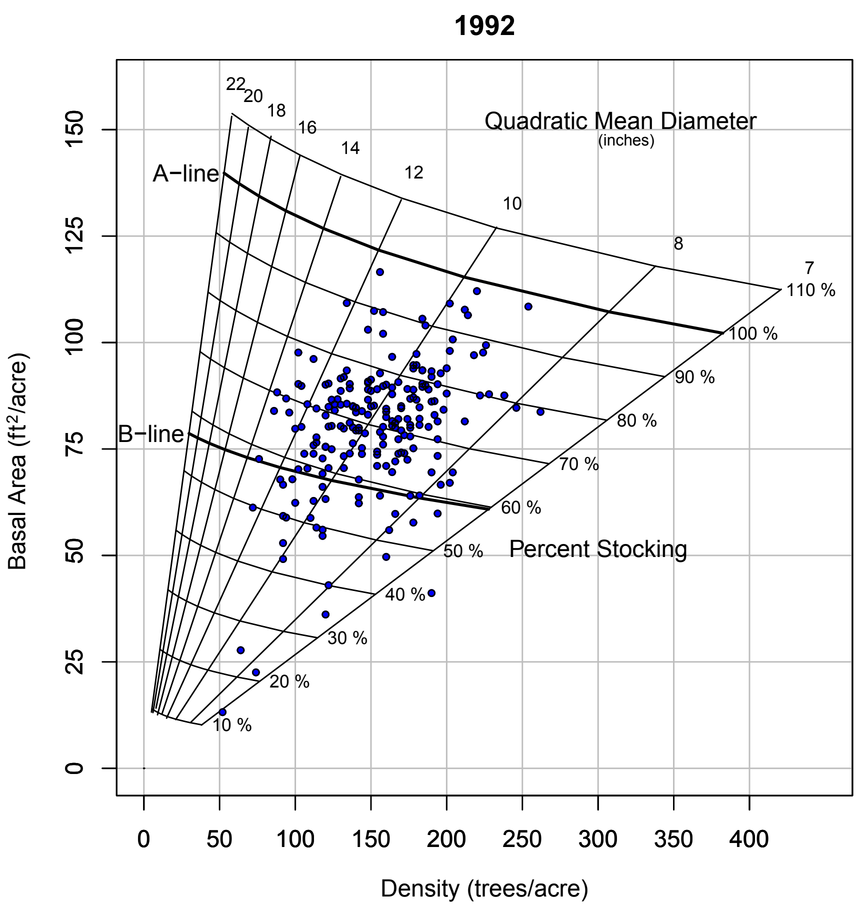

Figure 5.—Reineke (A) and Gingrich (B) stocking diagrams for the Missouri Ozark Forest Ecosystem Project even-aged management (EAM) sites in 1992.

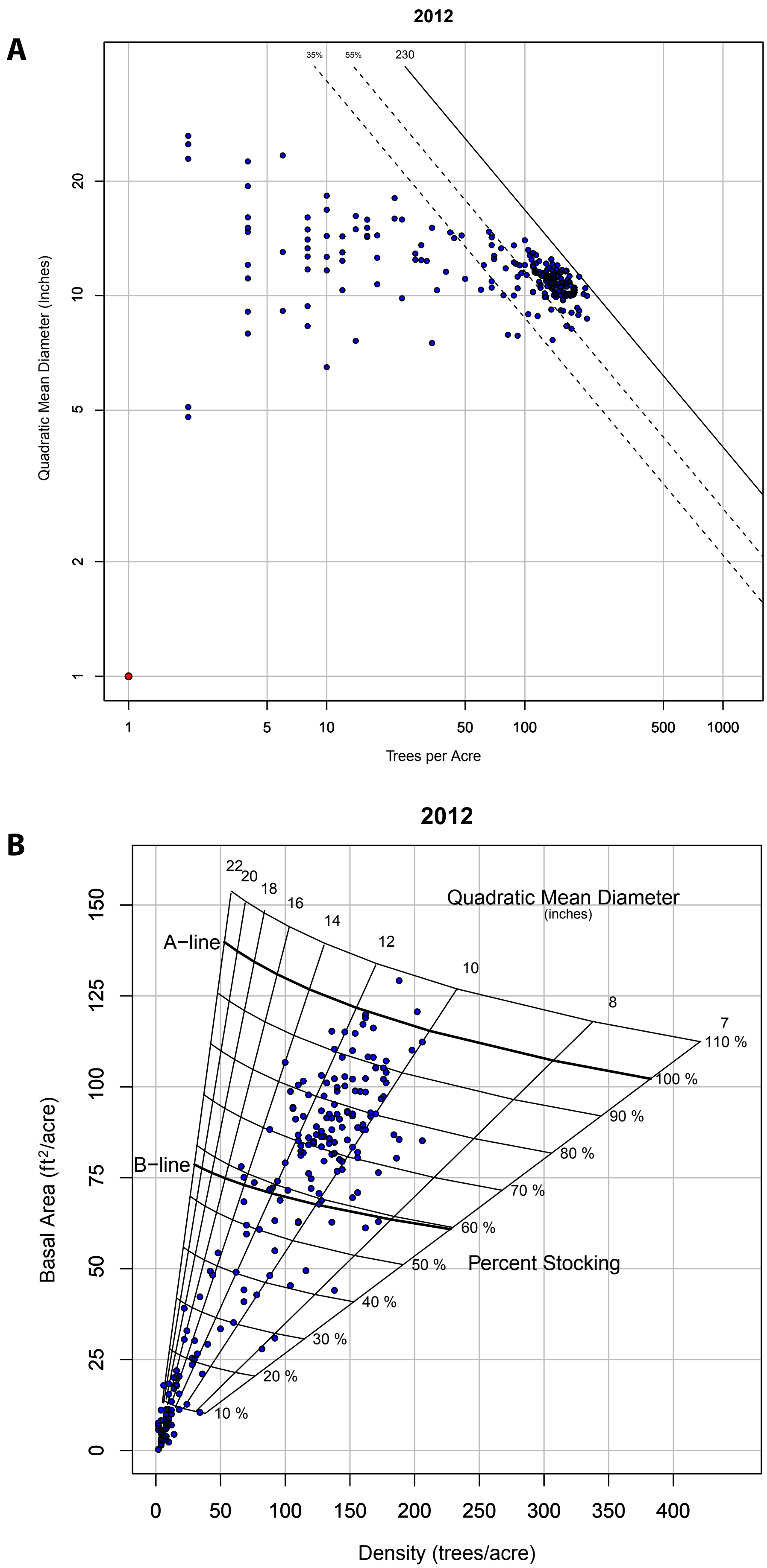

Figure 6.—Reineke (A) and Gingrich (B) stocking diagrams for the Missouri Ozark Forest Ecosystem Project even-aged management (EAM) sites in 2012.

CONCLUSIONS

Stand development patterns can be tracked on stocking diagrams using repeat-measurement plots. Stocking diagrams are helpful for interpreting changes caused by management treatments or other disturbances. The diagrams presented here provide a picture of the relative impact of three forest management practices applied over two decades for a large, forested landscape.

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FOREST HEALTH

MORTALITY, EARLY GROWTH, AND BLIGHT OCCURRENCE IN HYBRID, CHINESE, AND AMERICAN CHESTNUT SEEDLINGS IN WEST VIRGINIA

Melissa Thomas-Van Gundy, Jane Bard, Jeff Kochenderfer, and Paul Berrang[1]

Abstract.—Two plantings of second (BC_3F_2) and third (BC_3F_3) backcross generations of hybrid American chestnuts established in east-central West Virginia were assessed after 4 years to determine family effects on growth and survival. Pure American and pure Chinese chestnut seedlings were also planted as controls for height growth, form, blight occurrence, and blight resistance. Overall mortality after four growing seasons totaled 12 percent and 41 percent on the two sites. In 2014, the mean height of live stems differed by site and by seedling type. Pure American and BC_3F_3 hybrids were the tallest on both sites. Changes in height from time of planting to 2014 were similar or the same for American seedlings and BC_3F_3 hybrids on both sites. A similar trend among seedling types occurred for total height; American seedlings and BC_3F_3 hybrids performed similarly and showed the greatest mean total height on each site compared to the other seedling types. Two BC_3F_3 families had greater than 25 percent of their stems considered poor form on the Cheat site; 45 percent of the stems in one BC_3F_3 family were considered poor form on the Gauley site in 2015. Considering seedling types only, the greatest percentage of stems with a canker rating of 4 (blight-killed stems) was found for the BC_3F_3 hybrids on the Gauley site. There is little incidence of stems with a rating of 3 (blight present, sunken surface of cankers, large area covered, fruiting bodies present) or 4 on the Cheat site compared to the Gauley site. Family differences in height, form, dieback/resprouting, and canker occurrence are becoming apparent and are expected to continue.

INTRODUCTION

The story of the American chestnut (*Castanea dentata*) with its past abundance and removal from the overstory because of chestnut blight (*Cryphonectria parasitica*) and ink disease (*Phytophthora cinnamomi*) is well documented (Crandall et al. 1945, Smith 2000, Youngs 2000). American chestnut now persists in the eastern forest as understory stump sprouts that sometimes reach reproductive age (Paillet 2002, Woods and Shanks 1959). Sites where American chestnut now grows may reflect a contraction of the species' niche having shifted to dry, southern- to western-facing slopes from more mesic sites as a result of the blight (Burke 2012). The loss of American chestnut from the overstory affected species composition. Northern red oak (*Quercus rubra*), chestnut oak (*Q. prinus*), and white oak (*Q. alba*) increased in overstory importance after the loss of American chestnut (Keever 1953), oaks and red maple (*Acer rubrum*) replaced American chestnut in gaps (Woods and Shanks 1959), and hemlock (*Tsuga canadensis*) and yellow-poplar (*Liriodendron tulipifera*) replaced American chestnut on more mesic sites (Elliott and Swank 2008). If diseased trees were not removed by logging, individual trees could take 2-10 years or longer to die from the blight, so in some areas, adjacent nonchestnut trees gradually filled in the resultant gaps (Woods and Shanks 1959).

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West Virginia is in the center of the species' former range (Little 1977), and American chestnut was noted to occur across the state, comprising 12 percent of the overstory by some estimates (Brooks 1910). Based on witness trees in land surveys, American chestnut was a relatively minor component of European-contact forests in north-central West Virginia; the importance values were 1.5, 1.7, and 2.4 in three counties (Rentch and Hicks 2005). Witness trees for the area that later became the Monongahela National Forest (MNF), also in West Virginia, show that about 6 percent of the trees listed in deeds were American chestnut; however, they were not evenly distributed, ranging from 4 percent to 18 percent of witness trees depending on ecological subsection (Thomas-Van Gundy and Strager 2012). American chestnuts were associated with moderate elevations, sites of low moisture, high topographic roughness, ridge landforms, and soils weathered from acid sandstone (Lily soil series); few American chestnuts grow in the highest elevations (Thomas-Van Gundy and Strager 2012).

Efforts to create blight-resistant or blight-tolerant American chestnut hybrids or clones started soon after the disease effects were felt (Anagnostakis 2012). The U.S. Department of Agriculture Bureau of Plant Industry began crossing American chestnut with Asian species to produce trees with American chestnut form and Asian resistance (Clapper 1954, Graves 1942). These efforts later became part of the Connecticut Agricultural Experiment Station's work on restoring American chestnut (Anagnostakis 2012). The backcross method of breeding was introduced into these efforts in 1986 (Burnham et al. 1986). The American Chestnut Foundation (TACF), founded in 1983, began with the mission to backcross blight resistance from the Chinese chestnut (*C. mollissima*) into the American chestnut (Ellingboe 1994, Hebard 2001). The latest hybrid available from TACF (BC_3F_3) is predicted to be approximately 94 percent American chestnut (Ellingboe 1994, Hebard 2001). The BC_3F_3 hybrids are the third generation of the third backcross to produce trees that are predicted to have blight resistance similar to the Asian chestnut and the "timber" form of the American chestnut (Hebard 2001). Resistance testing of the parents of these hybrids suggests that blight resistance is likely to be intermediate to high (Hebard et al. 2014). Seedlings of the recent cross, BC_3F_3 "restoration" chestnut, are showing levels of blight resistance that are intermediate between the resistance of pure American chestnut and the Chinese breeding stock (Hebard 2012).

Before the blight, American chestnut in the southern Appalachian Mountains grew faster than most other associated species, reaching half their final height by age 20. Only white pine (*Pinus strobus*) and yellow-poplar grew faster (Ashe 1912). Stump sprouts in Connecticut reached 10 feet by age 4 and were the tallest sprouts in the hardwood coppice forest (Schwartz 1907). Reproduction through the formation of seedling-sprouts was described for one area as sprouts occurring at the base of the stem or just below the soil surface on 12- to 18-year-old seedlings; these could persist for 30–60 years with intervening cycles of dieback from shading or browse (Schwartz 1907). Based on stump sprouting ability (American chestnut is not known to root sprout [Schwartz 1907]) the creation of even-aged stands through coppice cutting was recommended for managing American chestnut (Zon 1904).

In 2011, the Northern Research Station of the U.S. Forest Service, the MNF, and TACF began cooperating on a long-term test planting of BC_3F_3 hybrids on national forest lands as part of a broader effort (Clark et al. 2014a). Along with outplantings, the hybrids are being screened for blight resistance by TACF at their seed orchards, and families that do not show resistance are removed from further breeding schemes (Hebard 2012). Hybrid seedlings on the MNF were planted to determine: (1) the degree to which the BC_3F_3 seedlings resemble American chestnuts in a natural forest setting and the degree to which Chinese characteristics (other than blight resistance) remain (Hebard et al. 2014), (2) the degree to which the BC_3F_3 seedlings are resistant to blight, and (3) how long resistance persists in these hybrids. In this analysis, we report the 4-year results on mortality, height growth, occurrence of resprouting, and occurrence of blight.

METHODS

Study Area

Two sites on the MNF were chosen for planting: one on the Cheat Ranger District and one on the Gauley Ranger District. The Cheat site is 7 acres that were part of a commercial timber sale, fenced to prevent deer browsing after clearcut harvest in 2008, and located in Tucker County near the town of Saint George, WV (39°08'56.7" N, 79°41' 50.1" W). Site preparation after harvest included severing all stems shorter than 2 inches diameter at breast height except desirable tree regeneration. Some red maple stump sprouts were treated with 2 percent RazorPro® herbicide in fall 2010. Herbicide was released on the planted seedlings in fall 2011. Individual planted seedlings were released from woody stem competition within a 2- to 3-foot radius by application of RazorPro. Blackberry (*Rubus* spp.) stems were treated only if they completely overtopped the seedling.

The Gauley site is a 2-acre portion of a larger clearcut located in Webster County above Sawyer Run, a tributary to the Williams River near the town of Cowen, WV (38°23'57.8" N, 80°26' 28.4" W). The planted area was harvested and fenced in 2010. Treetops and logging slash resulting from timber harvest were removed from the site before planting. The remainder of the 37-acre clearcut was harvested in 2011. Vis-pore® mats were used to control competing vegetation as trees were planted. Planted seedlings were released from overtopping competition by hand cutting selected stems in spring 2013. American chestnut saplings were noted on the site before harvest and planting.

The Gauley plantation site is higher in elevation (3,280-3,300 feet) than the Cheat site (2,460-2,500 feet). Both sites have moderate site indices (index species of northern red oak) at 72 for the Gauley site and 75 for the Cheat. The Gauley site faces mainly southwest and the Cheat site faces northwest. Soils on the Gauley site are mapped as the Clifftop-Laidig association, and the Cheat site is classified as Gilpin channery silt loam in the respective county soil surveys. Surface soil on the Gauley planting site appears higher in sand content than the Cheat site, and the site includes sandstone surface boulders.

Both sites were tested for infection by *Phytophthora* root rot (*P. cinnamomi*), and samples showed no infection at the time of planting. *P. cinnamomi* has caused American chestnut mortality, particularly at low elevations, starting in the 1820s when it was accidently imported from Asia (Anagnostakis 2012). American chestnut is highly susceptible to *Phytophthora* root rot even on moderately wet soils with little compaction (Clark et al. 2014b, Rhodes et al. 2003); therefore, this pathogen must not be present at the start of the study.

Study Design and Data Collection

Nursery stock was planted by contractors (Gauley) or MNF staff (Cheat) using standard hand planting techniques (planting bars) and power augers. Seedlings were planted in March and April 2011 in rows about 8 feet apart with about 8 feet between seedlings as site conditions allowed. The seedlings were 1-year-old bare-root stock that had been grown at the Virginia State nursery near Waynesboro, VA. The BC_3F_3 seedlings were grown from nuts produced in seed orchards at TACF's farms near Meadowview, VA. The hybrid seedlings were open pollinated; family designations are from the TACF breeding scheme where families share a single mother tree of known genetics. Most of the American chestnut ancestors of the trees in the seed orchard were in the vicinity of Mount Rogers National Recreation Area, VA, but some originated further south in the Appalachian Mountains. Mount Rogers National Recreation Area is approximately

140 miles from the Gauley planting site, about 200 miles from the Cheat planting site, and 2°-3° of latitude south of the two planting sites; therefore, we assumed the experimental material from the BC_3F_3 and American chestnut were locally adapted to the MNF study areas.

Plantation studies require that mortality of individuals be considered when setting up the experimental design. The planting design needs to include enough experimental units such that data collected can be assessed with statistical power. Resolvable incomplete block design (Patterson and Williams 1976) was chosen for this study to address the concerns of expected mortality, the number of families to test, and differences in planting site conditions. Each incomplete block contained 35 trees.

Seedlings from 23 families of BC_3F_3 seedlings were planted at each site (Table 1), but not all families were planted at both sites. Thirty-one families of BC_3F_3 seedlings were planted across

Table 1.—Family names, numbers of trees, and the type of seedlings planted on each site. Gray highlights show families planted at both sites.

Cheat		Gauley		
Family	Count	Family	Count	Type
BLWPureAmerican	25	BLWPureAmerican	25	Pure American
CliffAcc	25	CliffAcc	25	Pure American
HaunAmerican	25	HaunAmerican	25	Pure American
CH297	25	CH297	25	BC_3F_2
SC80	25	SC80	25	BC_3F_2
D1-28-138	50	D1-28-138	50	BC_3F_3
D2-26-72	25	D2-26-72	25	BC_3F_3
D2-28-52	25	D2-28-125	25	BC_3F_3
D2-29-27	25	D2-28-42	25	BC_3F_3
D2-50-15	21	D2-28-52	26	BC_3F_3
D3-27-105	24	D2-50-12	25	BC_3F_3
D3-27-46	24	D2-50-15	24	BC_3F_3
D4-18-30	25	D3-27-105	25	BC_3F_3
D4-20-65	25	D3-27-46	25	BC_3F_3
D4-26-63	25	D4-18-30	25	BC_3F_3
D4-27-103	25	D4-20-65	25	BC_3F_3
D5-17-130	50	D4-26-63	25	BC_3F_3
D5-17-61	50	D5-17-130	50	BC_3F_3
D5-17-89	25	D5-17-61	50	BC_3F_3
D5-25-49	26	D5-17-89	25	BC_3F_3
D5-26-54	49	D5-25-147	25	BC_3F_3
D5-27-101	24	D5-26-54	50	BC_3F_3
D7-26-86	50	D6-26-29	25	BC_3F_3
D8-26-104	25	D7-26-20	25	BC_3F_3
D9-26-36	25	D7-26-86	50	BC_3F_3
W1-29-8	24	D7-28-83	25	BC_3F_3
W1-31-63	25	D8-26-15	25	BC_3F_3
W3-32-49	25	W1-31-63	25	BC_3F_3
opCD	49	opCD	50	Pure Chinese
Grand total	866		875	

both sites. Seedlings of pure American chestnut (three families at both sites), Chinese chestnut (one family at both sites), and intermediate hybrids (BC_3F_2; two families on both sites) were planted as control trees. Approximately 875 trees were planted at each site (Table 1). Each seedling was assigned an individual numbered tag to ensure that family identification and initial seedling characteristics could be followed throughout the study.

Initial heights were recorded within 2 weeks of planting in 2011. Total height (base of the tree to the top of the tallest live bud) was measured in early spring 2012, 2013, 2014, and 2015 on the Cheat site and early spring 2012, 2013, and 2015 on the Gauley site. Survival, damage, and form were noted for each seedling using qualitative descriptions when heights were taken. Damage remarks included dead top resprouted, dead top, resprouted, insect damage, bear damage, broken top, herbicide damage, top cut at nursery, overtopped, live/dead, yellow leaves (not related to fall color change), bent stem, grapevine on stem, overtopped by slash, and greenbrier on stem. Stems were considered to have poor form if any of the following were noted in 2015: epicormic branching, forking, low forking, lean (not related to storm damage), multistem, flat top, or crook. By 2015, blight cankers were evident on some stems and severity class was noted. Classes used were 1 (no blight); 2 (blight present but superficial or swollen, no fruiting bodies); 3 (blight present, sunken surface of cankers, large area covered, fruiting bodies present; and 4 (blight-killed stem). These ratings were adopted from a similar study conducted by the Southern Research Station (S. Clark, personal communication).

Data Analysis

Differences in performance among families were analyzed as generalized linear mixed models with live seedling height assessed three ways: (1) mean height in 2014 (no resprouting individuals included), (2) mean change in height from the time of planting to 2014 (no resprouting individuals included), and (3) mean total height from 2011 through 2014 (resprouts included and time as repeated measure). Sites were assessed separately. For the first two measures, differences in performance among families were analyzed as generalized linear mixed models (PROC GLIMMIX, SAS 2012) with the intercept as a random variable and the family as the fixed variable. These models were run as incomplete block design using a gamma distribution with log link function on the Cheat site and Gaussian response distribution with the identity link on the Gauley site. For both sites, denominator degrees of freedom were calculated with the Kenward-Rodger method. For these analyses, significance was tested at $\alpha = 0.05$, pairwise comparisons were adjusted using the Tukey Kramer method, and back transformed least square means for family are reported.

Calculation of mean total height (the third height measure) did include stems that had resprouted during the 4 years. The differences among families were analyzed as generalized linear mixed models with repeated measures (PROC GLIMMIX, SAS 2012) with initial height (2011) as covariate following methods in Littell et al. (1996). Models were run as incomplete block design using a gamma distribution with log link function with family and year and family*year interaction tested; year and intercept were random variables. For both sites, denominator degrees of freedom were calculated with the Kenward-Rodger method. For these analyses, significance was tested at $\alpha = 0.05$, pairwise comparisons were adjusted using the Tukey Kramer method, and back transformed least square means are reported.

Descriptive statistics were used for percentage of mortality at 2015, percentage of resprouting in any year, and percentage of canker rating class at 2015, all by family and site.

RESULTS

Mortality

Mortality after the first growing season (seedlings surveyed in October 2011) differed greatly by site at about 1.6 percent for the Cheat sites and about 12.7 percent for the Gauley site (data not shown) for all seedling types. By the end of the 2013 growing season, mortality on the Gauley site had increased to 23.9 percent and by only 7.3 percent on the Cheat site. Mortality after 4 seasons and across all families totaled 12 percent on the Cheat site and 41 percent on the Gauley site (Fig. 1). About 13 percent of the BC_3F_3 hybrids planted on the Cheat site had

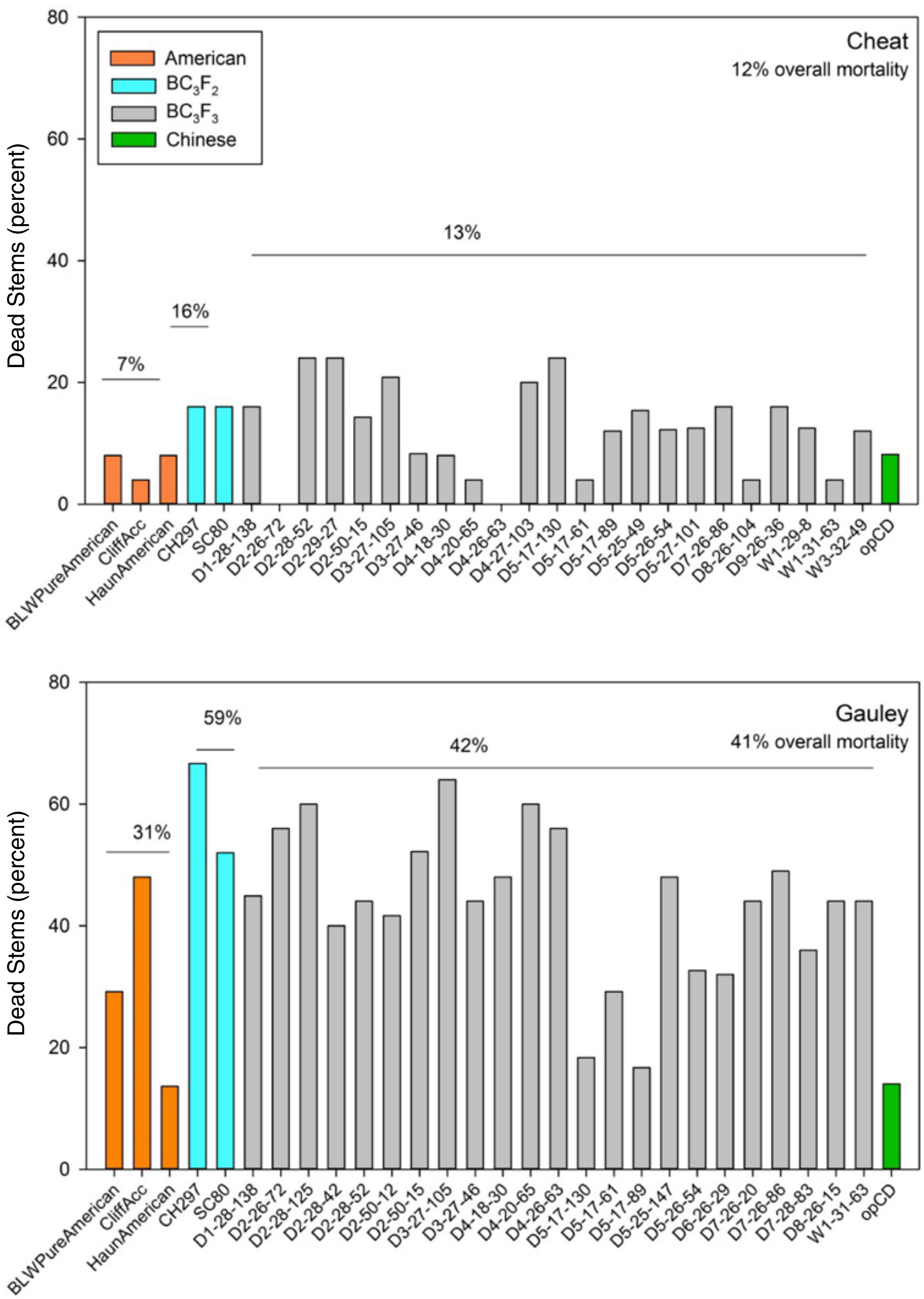


Figure 1.—Mortality by family as of spring 2015. Percentages given above vertical lines are the mean mortality by seedling type.

died compared to 42 percent on the Gauley site. No families experienced greater than 50 percent mortality on the Cheat site, and 10 families experienced that level or higher on the Gauley site. On the Cheat site, the highest mortality occurred in the BC_3F_3 families (D2-28-52, D2-29-27, and D5-17-130) at 24 percent of seedlings for each family. As a group, pure American chestnut families had the lowest mortality at 7 percent compared to 16 percent for the BC_3F_2 hybrids and 8 percent for pure Chinese on the Cheat site. On the Cheat site, 2 families, D2-26-72 and D4-26-63, showed no mortality. Mortality was higher in all families and groups on the Gauley site; 31 percent of the pure American chestnut were dead by 2015, as were 59 percent of the BC_3F_2, 42 percent of the BC_3F_3, and 14 percent of the pure Chinese seedlings.

Height

In 2014, the mean height of live stems (no resprouting stems included) differed by site and by seedling type. Pure American and BC_3F_3 hybrids were tallest on both sites (Table 2). BC_3F_3 hybrids averaged 9.3 ± 0.4 feet (mean ± standard error) on the Cheat site and 7.6 feet (± 0.2 feet) on the Gauley site. Seedlings of pure Chinese origin were shortest on both sites. The greatest mean heights in 2014 were found in three BC_3F_3 families on the Cheat site: D2-26-72 (10.9 feet ± 0.8 feet), D4-18-30 (10.1 feet ± 0.8 feet), and D5-26-54 (10 feet ± 0.6 feet), although their mean heights differ significantly from only one BC_3F_3 family (D5-17-130, 6.7 feet ± 0.4 feet) and the Chinese chestnut family (Fig. 2). On the Gauley site no significant differences were found among families; however, the greatest mean height was 8.7 ± 0.8 feet for seedlings of the D2-28-125 family (Fig. 2).

Table 2.—Arithmetic means and standard errors (SE) for the three measures of height by site and seedling type

	Cheat		Gauley	
Type	Mean	SE	Mean	SE
	Height in 2014 (feet)			
Pure American	9.3	0.2	7.5	0.2
BC_3F_2	8.0	0.4	6.1	0.3
BC_3F_3	9.3	0.2	7.6	0.2
Pure Chinese	7.1		5.9	
	Change in height from 2011 to 2014 (feet)			
Pure American	6.0	0.0	4.8	0.3
BC_3F_2	5.0	0.0	5.1	0.6
BC_3F_3	6.0	0.1	4.6	0.2
Pure Chinese	4.6		3.7	
	Total height growth (feet)			
Pure American	5.8	0.0	5.2	0.2
BC_3F_2	4.9	0.1	4.3	0.3
BC_3F_3	5.8	0.1	5.5	0.1
Pure Chinese	4.3		3.8	

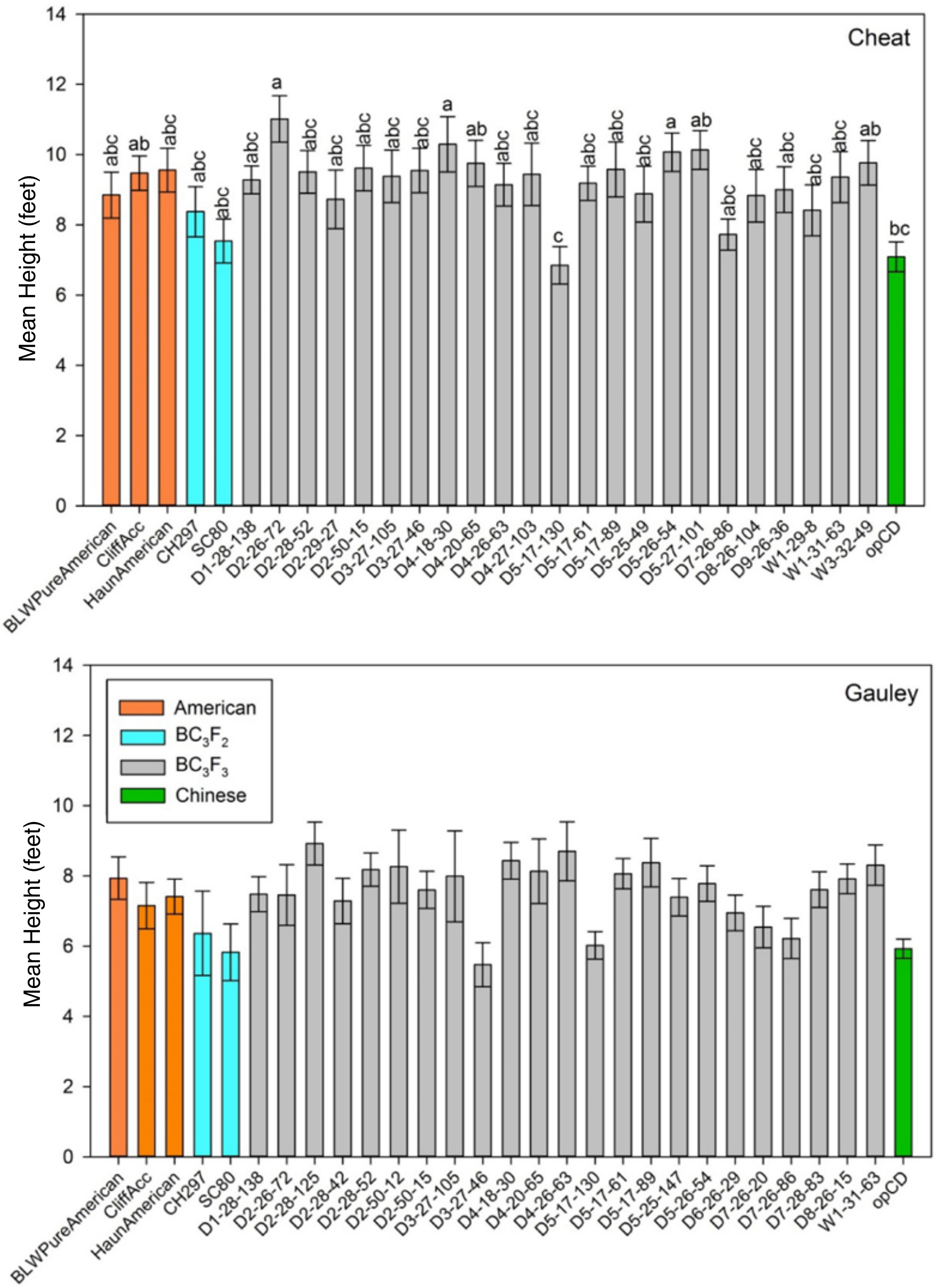


Figure 2.—Least square mean heights (± SE) by family in 2014. Means with different letters are significantly different (α = 0.5); no significant differences were found for the seedlings on the Gauley site.

As found for mean height in 2014, the change in height from time of planting to 2014 (no resprouting stems included) was similar or the same for pure American seedlings and BC_3F_3 hybrids on both sites (Table 2). Greater variability in this measurement on the Gauley site resulted in no statistically significant differences among families (Fig. 3). On the Gauley site, seedlings in the D2-28-125 family (6.3 ± 0.6 feet) were significantly taller than those in the D3-27-46 (3 ± 0.6 feet) and D5-17-130 (3.1 ± 0.3 feet) families (all BC_3F_3). The greatest mean change in height for the Cheat site was found in the BC_3F_3 family D2-26-72 (7.3 ± 1.5 feet), which also showed no mortality. On the Gauley site, the greatest mean change in height also occurred in a BC_3F_3 family, D2-28-125, which also had the greatest mean height in 2014. On both sites, one BC_3F_3 family had mean height growth at or lower than the pure Chinese chestnut trees, D5-17-130.

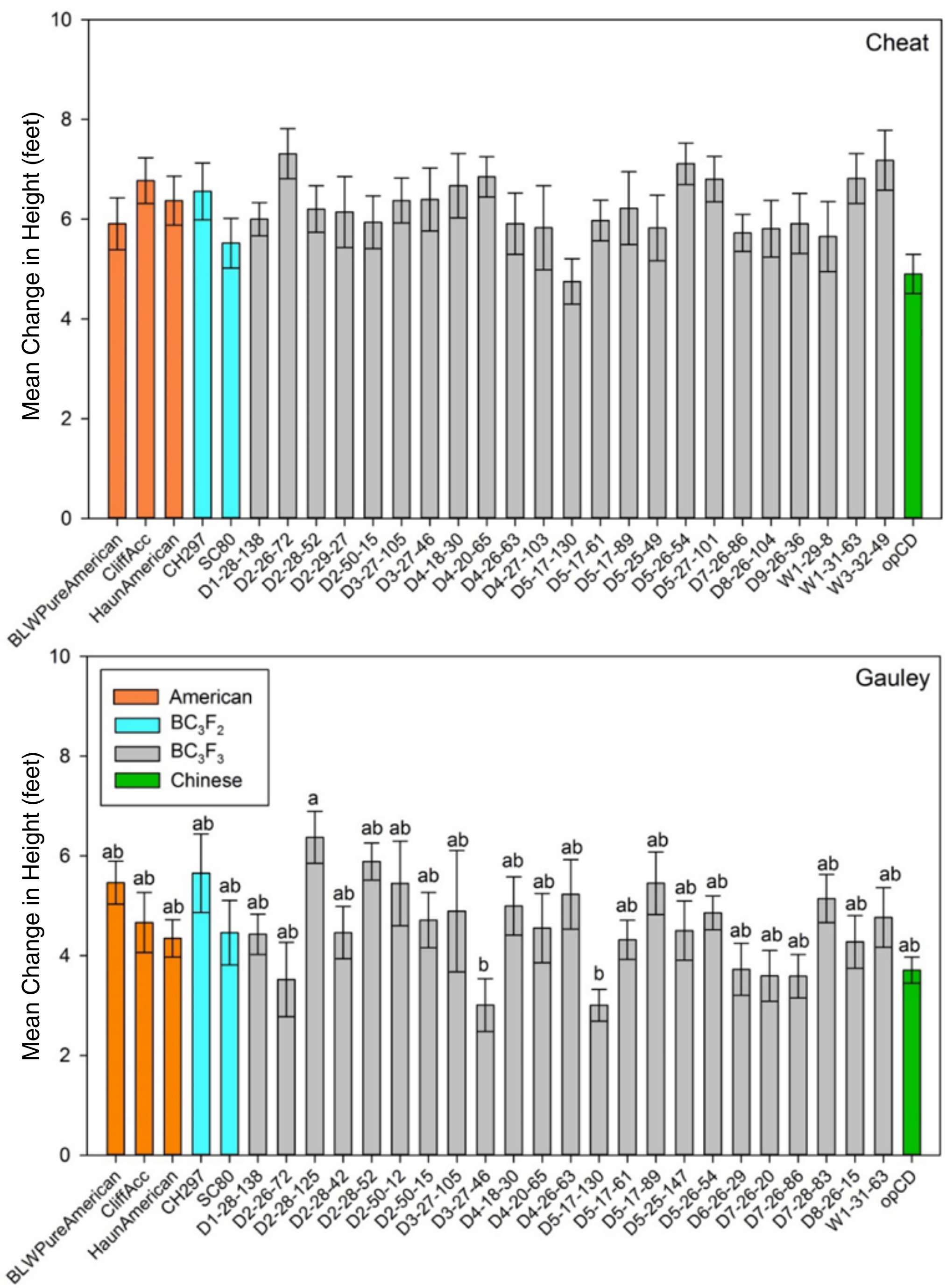
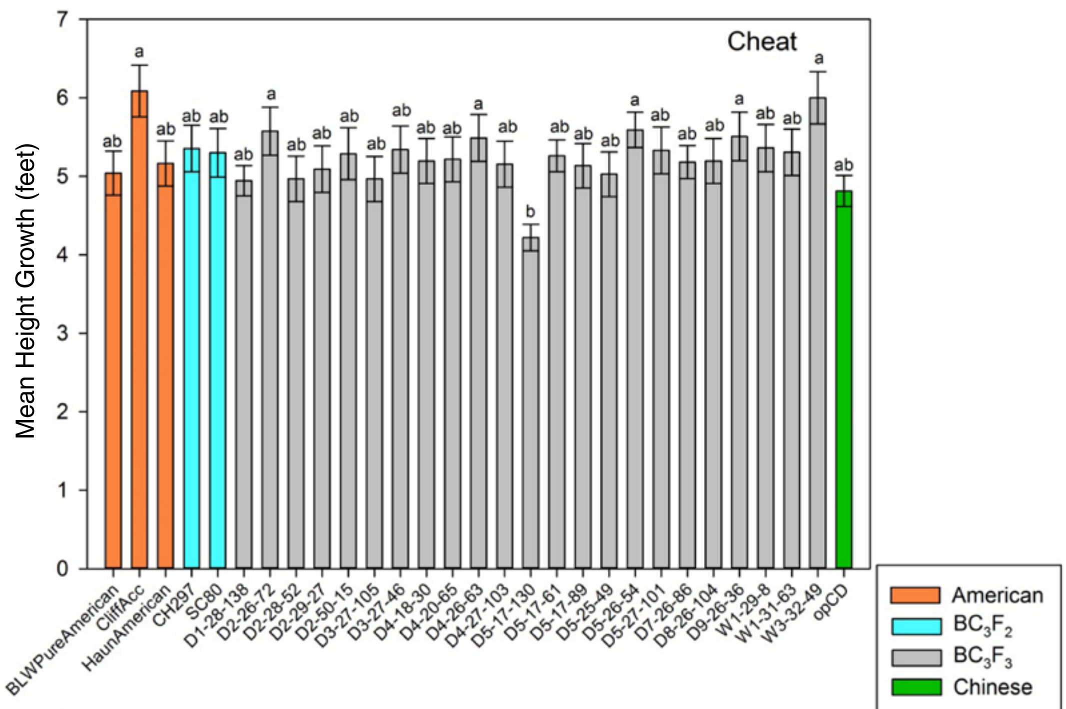


Figure 3.—Least square mean change in height from 2011 to 2014 (± SE) by family. Means with different letters are significantly different (α = 0.5); no significant differences were found for seedlings on the Cheat site.

A similar trend among seedling types is found for total height growth. Pure American seedlings and BC_3F_3 hybrids performed similarly and showed greatest mean total height compared to the other seedling types (Table 2). When total height growth over time is tested, including resprouting individuals, family differences are statistically significant (Fig. 4). On the Cheat site, six families (one pure American and five BC_3F_3) had greater growth than one family (BC_3F_3), and on the Gauley site, one family (BC_3F_3) showed significantly greater growth over four families (two BC_3F_3 families, one BC_3F_2 family, and one pure Chinese family). Seedlings of the D5-17-130 hybrid family showed generally lower total growth at both sites (lowest on the Cheat site), although this was not statistically different than many other families. Similarly, pure Chinese chestnut seedlings on the Gauley site showed generally less total height growth, significantly less than 14 out of 29 families.



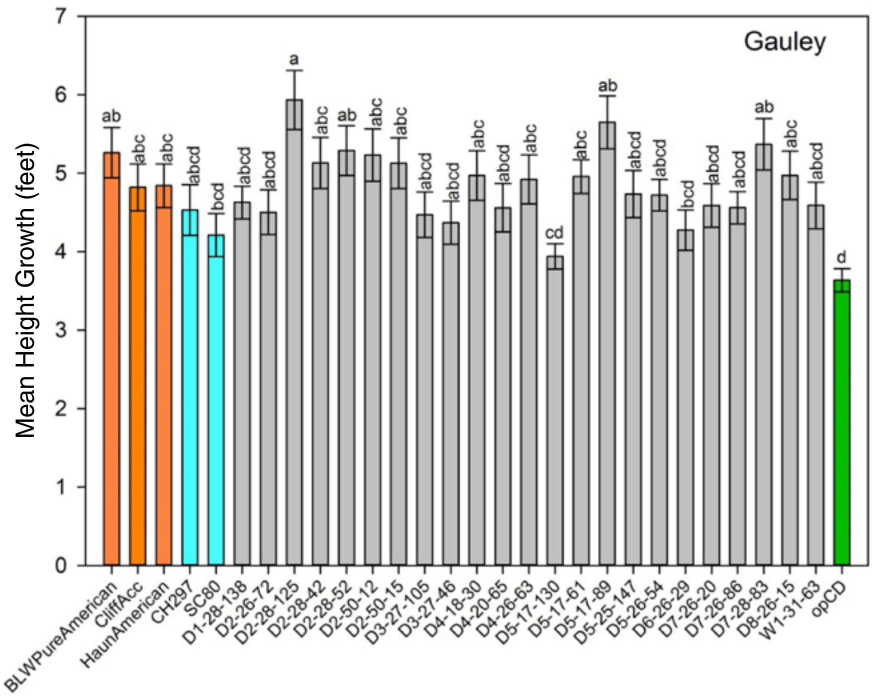


Figure 4.—Least square mean total height growth (± SE) by family. Means with different letters are significantly different (α = 0.5).

Form and Canker Incidence

The number of stems considered to have poor form in 2015 was generally higher for all seedling types on the Gauley site compared to the Cheat site (Table 3). Two BC_3F_3 families had greater than 25 percent of their stems considered poor form on the Cheat site (D4-26-63 and D5-17-89); 45 percent of the stems in one BC_3F_3 family (D5-17-61) were considered poor form on the Gauley site in 2015 (Fig. 5). One family on the Cheat site (D2-28-52) and two on the Gauley site (D2-26-72 and D3-27-105) showed no stems considered poor form in 2015. The lowest incidence of poor form was found in the pure Chinese seedlings on the Cheat site.

Table 3.—Mean percentage of stems with poor form (in 2015) by site and seedling type

	Cheat	Gauley
Type	% of stems	% of stems
Pure American	9.3	16.7
BC_3F_2	6.0	22.9
BC_3F_3	9.2	18.2
Pure Chinese	2.0	23.3

Pure Chinese seedlings have the greatest incidence of dieback and resprout at both sites (Table 4; about 25 percent of stems on the Cheat site and 54 percent on the Gauley site). The incidence of main stem dieback with resprouting, noted in the remarks for each seedling, was also assessed by family (Fig. 6). The BC_3F_3 hybrid with low height growth (D5-17-130) showed resprouting on about 20 percent (Cheat site) and 28 percent (Gauley site) of stems.

Table 4.—Mean percentage of stems that resprouted from dieback any year by site and seedling type

	Cheat	Gauley
Type	% of stems	% of stems
Pure American	6.7	12.2
BC_3F_2	6.0	8.0
BC_3F_3	8.9	11.4
Pure Chinese	24.5	54.0

Only the percentages of stems with ratings of 2, 3, or 4 are reported here because a 1 rating meant no cankers noted. Considering seedling types only, the highest percentage of stems with canker rating of 4 (blight-killed stems) was found for the BC_3F_3 hybrids on the Gauley site (Table 5). Few stems had a rating of 3 (blight present, sunken surface of cankers, large area covered, fruiting bodies present) or 4 on the Cheat site compared to the Gauley site (Fig. 7). Two BC_3F_3 families on the Cheat site show 17 percent and 19 percent of stems with a canker rating of 2 (D4-18-30 and W1-29-8, respectively), and these are the highest percentages of 2 ratings on this site. Three BC_3F_3 families on the Gauley site show 20 percent, 20 percent, and 17 percent of stems with canker rating of 2 (D3-27-105, D5-17-89, and D5-26-54, respectively). Only 5 BC_3F_3 families on the Cheat show stems that were considered killed by blight (4 rating) ranging from about 3 percent to 9 percent of stems, and no American chestnuts had rankings higher than 3. In comparison, on the Gauley site 24 families had stems killed by blight, ranging from about 2 percent to 29 percent of stems.

Table 5.—Mean percentage of stems with canker rankings of 2, 3, or 4 by site and seedling type

	Cheat			Gauley		
	% of stems			% of stems		
	Canker rating			Canker rating		
Type	2	3	4	2	3	4
Pure American	4.3	1.4	0.0	5.4	5.2	7.9
BC_3F_2	4.8	0.0	0.0	9.4	3.8	9.4
BC_3F_3	4.9	0.8	1.1	7.9	2.9	10.7
Pure Chinese	11.1	0.0	0.0	11.6	7.0	0.0

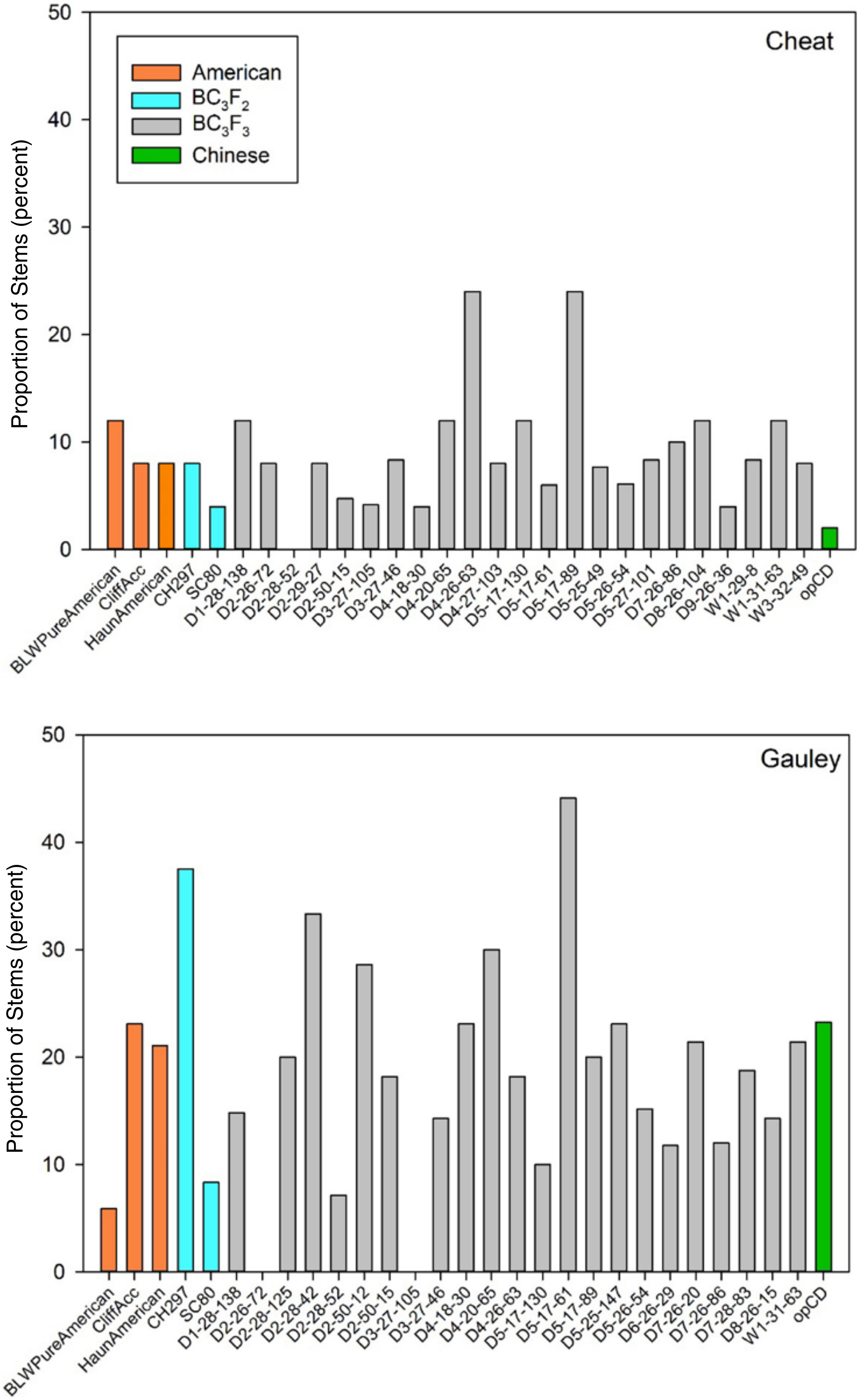


Figure 5.—Percentage of stems with poor form (in 2015) by family.

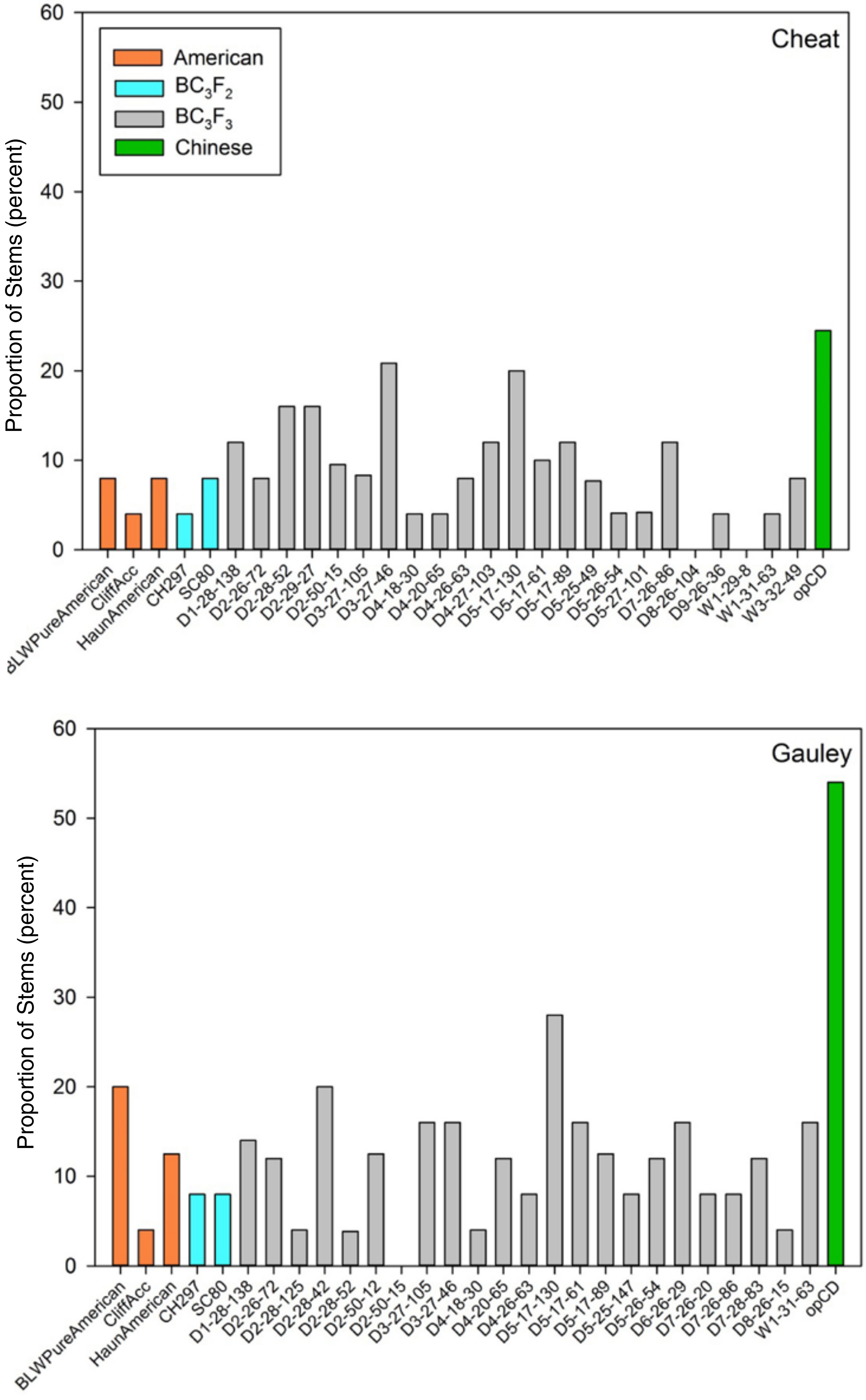


Figure 6.—Incidence of resprouting in any year by family.

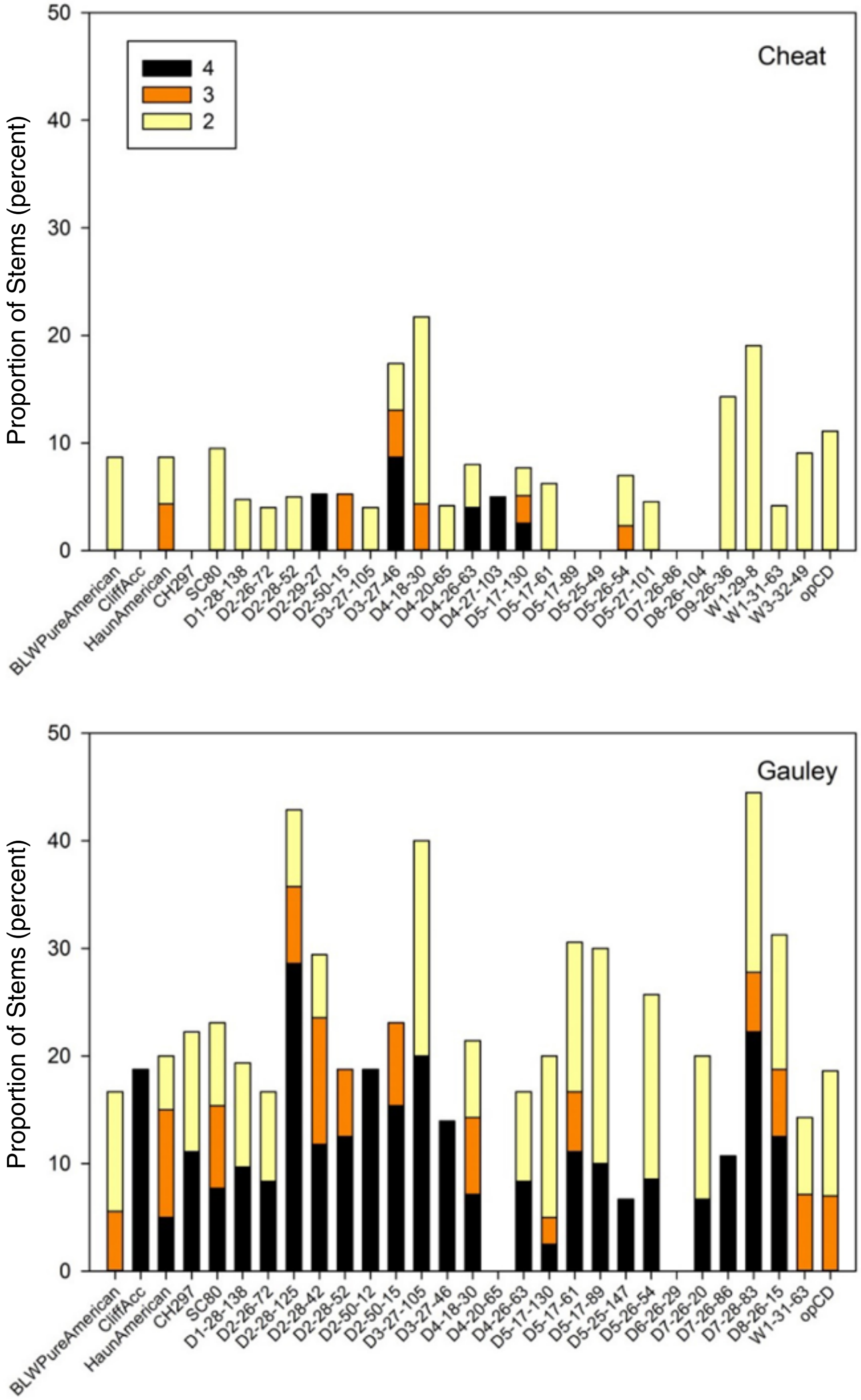


Figure 7.—Canker ratings (for ratings 2-4 only) in spring 2015 by family. Ratings are 2: blight present, superficial, swollen, no fruiting bodies; 3: blight present, sunken surface of cankers, large area covered, fruiting bodies present; 4: blight-killed stem.

DISCUSSION

As Clark et al. (2015) found in plantings in North Carolina, Tennessee, and Virginia, the BC_3F_3 seedlings planted in West Virginia responded more like American chestnut than Chinese chestnut. Progeny and provenance studies on other Fagaceae species show that growth rates by family change as the individuals move from juvenile to adult stages (Kriebel et al. 1988, Míguez-Soto and Fernández-López 2015). We anticipate changes in the rankings of families over time at our sites for many measures, especially as the stands reach the stem exclusion stage of development.

The spatial patterns of mortality on the Gauley site, especially within 1 year of planting, suggest that some combination of compaction, soil moisture, and perhaps *P. cinnamomi* compromised seedling growth. In 2012, an unknown species of *Phytophthora* was found on the Gauley site; further testing of soils and dead seedlings is needed to confirm the presence of *P. cinnamomi*. We do know, however, that ink disease is present at the nursery that produced the seedlings we used.

Treetops were skidded off the Gauley site immediately before planting. Skidder paths were visible as lanes through the site. Most seedlings planted in these lanes did not survive to age 4. In a greenhouse study, American chestnut seedling mortality was highest on wet and compacted soils regardless of fungicide treatment (Rhoades et al. 2003). In a study similar to ours with hybrid seedlings planted in forested conditions, higher mortality occurred on sites with poor drainage (Clark et al. 2014b). Although the numbers of seedlings in the greenhouse study were low, the results emphasize the findings of others that American chestnut seedlings are not well suited to wet and compacted soils. Based on our field observations, this also holds true for the advanced hybrid seedlings.

Another possible contributing factor to mortality on the Gauley site is soil texture. Mortality in a planted stand of American chestnuts in eastern Kentucky was negatively related to sand and coarse fragment content in the soil (Rhoades et al. 2009), where the percentage of sand and coarse fragments (measured and assessed separately) was negatively related to seedling survival. The Clifftop-Laidig soil association underlying the Gauley site is described as extremely stony, which applies to the surface, and a channery silt loam in the A horizon (1-3 inches). By definition, in a channery silt loam, 15–35 percent of soil volume is in channers, flat rock fragments up to 6 inches long. Using that composition range and the relationship between coarse fragments and seedling survival described by Rhoades et al. (2009), survival on the Gauley site is predicted as approximately 50-74 percent; year 4 survival on the Gauley site was 59 percent.

The poor form category included low forks, epicormic branching, and lean (not related to storm damage). Given storm damage on the Cheat site (Hurricane Sandy, snow-on-leaves event in October 2012), it was surprising that the percentage of stems with poor form was not higher. In this storm, blackberry thickets that still had leaves trapped snow and bent or broke many seedlings that were growing underneath the blackberry. Some stems did break off completely; however, those would not have been in the poor form category. About 450 stems were staked after the storm; many stakes and twine had rotted off by 2015. Temporary staking may have confounded the use of this qualitative measure for determining desirable families to continue in the breeding program. The staking of at least some of the worst-affected stems was considered necessary to salvage the study.

Sprouting is part of the reproduction strategy of American chestnut (Paillet 2002, Russell 1987, Schwartz 1907, Wang et al. 2013) but can confound height growth measurements. Resprouting could be in response to planting shock, dieback from blight or another disease, or damage from small mammals. For these reasons, height growth was expressed in three ways. Only the total height growth over the entire 4 years (Fig. 4) included individuals that resprouted; for the other two measurements, resprouted individuals were removed from calculations. Although the incidence of resprouting after dieback (Fig. 6) may not be directly used to determine successful families for restoration planting, it will be interesting to see if any families are consistent resprouters. Others have found dieback to be greater in seedlings of larger ground-line diameter and for Chinese chestnuts, although no differences among hybrid families were found (Clark et al. 2015).

These results represent only 1 year of rating stems for the occurrence of chestnut blight cankers. It is likely too early to start removing families based on this criterion alone, although these families are also being screened at TACF orchards. Preliminary results of TACF screening efforts show that the D5-17-130 family shows resistance to the blight (J. Westbrook, personal communication); however, it is one of the shortest families in the West Virginia plantings at 6.7 feet on the Cheat site and 6.1 feet on the Gauley site in 2014. In terms of canker ratings in 2015 on these sites, family D4-18-30 shows a high percentage of stems with a canker rating of 2 on the Cheat site (Fig. 7), and TACF trials have also found high resistance to the blight fungus in this family (J. Westbrook, personal communication). TACF trials have also found seedlings of the D2-28-125 and D3-27-46 families to be low in resistance (J. Westbrook, personal communication). Our results show that seedlings of D2-28-125 have high mortality from blight on the Gauley site (Fig. 7) and the D3-27-46 family has higher mortality on the Cheat site.

This study is part of the larger goal of restoring American chestnut, as a self-regenerating hybrid, to forested settings where the species can continue to evolve (Hebard 2012). The number of backcross families under evaluation at our study sites is unique. These plantings will continue to be monitored for health and growth with the goal of informing the breeding program.

ACKNOWLEDGMENTS

We thank TACF for the planting stock used in this research and for its continued involvement in restoration of American chestnut. Specifically, we thank Fred Hebard for his work on establishing these plantings and Jared Westbrook for continuing the cooperation between the MNF and TACF. John Stanovick was instrumental in the design of the plantings and analysis of the data. We thank Richard Hovatter and Donald Lowther for help with data collection. Two anonymous reviewers provided helpful comments to improve this manuscript.

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CHARACTERISTICS OF SITES AND TREES AFFECTED BY RAPID WHITE OAK MORTALITY AS REPORTED BY FORESTRY PROFESSIONALS IN MISSOURI

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Abstract.—A new syndrome was named rapid white oak mortality in 2011 to describe the rapid death of white oak trees (*Quercus alba* L.) within one growing season. A survey with 24 questions about stand and site characteristics, site history, and symptoms was distributed to forestry professionals to gather information about the new syndrome. Sixty-three reports were received. Most of the mortality was reported in the southeastern quadrant of Missouri. Site characteristics and the affected trees species were atypical of those usually associated with oak mortality in Missouri. The survey confirmed that rapid white oak mortality should be considered a new syndrome.

INTRODUCTION

Mortality of oak species in Missouri is typically attributed to oak decline and oak wilt. Both maladies disproportionately affect trees in the red oak group. An upsurge of white oak (*Quercus alba* L.) mortality that did not fit the descriptions of either malady was reported in 2011. Reports of new locations with mortality spiked during 2012 and declined during the following years. Most reports in 2015 included previously undetected pockets of mortality, along with those that continue to grow as declining trees die. Mortality patterns included the deaths of entire tree crowns followed by rapid bronzing of foliage at the beginning of the growing season or during late summer and an association with lower slopes and drainages. White oak and post oak (*Quercus stellata* Wangenh.) were the only species in the white oak group that were reported to be dead and dying; most dead trees were white oak. The phenomenon was named rapid white oak mortality (RWOM) to set it apart from oak decline until a regional study of causal agents could be performed.

A 1-year study was conducted during 2014 to investigate the distribution of RWOM and the site and stand characteristics associated with the disease. As part of this study, we compiled data from a survey form that agency employers had distributed to their forestry staff. The survey contained questions about locations of mortality pockets, land use, site and stand characteristics, symptoms, and site history.

METHODS

The Missouri Department of Conservation (MDC) forest health program requested that all MDC foresters fill out survey forms for any reported RWOM locations. Forms were also shared with and distributed by U.S. Forest Service (USFS) employees to its staff. The 1-page survey consisted of 24 questions grouped into five subject areas: location and description of land use, site characteristics, stand characteristics, visual symptoms observed on declining white oak trees, and known site history. Questions were categorical except for location information, elevation,

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slope aspect, and steepness. Foresters and forestry staff members were instructed to walk an area of approximately 0.2 ha that had white oak mortality before they answered the questions. (Throughout the remainder of this paper, these foresters and forestry staff members are referred to as respondents.)

The survey included five questions about location. Respondents provided the name of the surveyed location, the county, the geographic coordinates or the legal description, and the soil map unit from the soil survey. They also described how the public or private land was being used (e.g., forest).

The survey contained eight questions about site characteristics associated with dead and declining white oak trees. Respondents identified the aspects, elevations, average percent slopes, slope shapes, and positions of dead and declining white oak trees on the slope. They also indicated what they perceived as general soil moisture content, the presence of pooled water or drainages, and whether the water was seasonal or permanent. Information about soil characteristics at the site was later gathered from the Natural Resources Conservation Service Web soil survey (NRCS 2016) using the soil map unit information or geographic coordinates provided. A soil profile depth of 150 cm was used when gathering data from the Web soil survey for the following soil features: available water storage; the percentages of clay, sand, and silt; and the depth to restrictive layer. A soil depth of 80 cm was used to obtain coarse fragment data from the Web soil survey. The overall percentage of coarse fragments was calculated by multiplying the relative percentage of total coarse fragments for each soil horizon by the depth of each horizon and then dividing the sum for all horizons by the total soil depth.

Four questions were asked about stand characteristics. Respondents identified the dominant tree type, the relative percentage of the stand that was in the white oak group, the types of dead and dying trees, and whether white oak regeneration was evident. Respondents answered an additional five questions about dead and dying white oak trees in the stand. These questions included an estimate of the percentage of the dead and dying white oak trees, as well as the crown position, the severity of the crown dieback, and the size of dead and declining trees.

Respondents were asked eight questions about symptoms or signs associated with declining white oak trees. They were also asked to indicate if holes made by wood-boring insects were present and if so, the shapes and relative sizes of the holes.

The site history category included 10 questions about management (e.g., timber stand improvement and fire severity), exceptional weather events, and insect outbreaks that had occurred since 2005. Respondents were encouraged to further describe the events and list the year or years of the events. Respondents were contacted with questions about their answers and to complete unanswered questions. Frequencies were calculated for answers to the 24 questions. The 55 reports that described the new pattern of mortality (e.g., lower slopes and drainages) were emphasized. Geographic coordinates supplied by forestry professionals or approximate coordinates from legal descriptions were used to generate maps in ArcGIS version 10.3.1 (Esri, Redlands, CA).

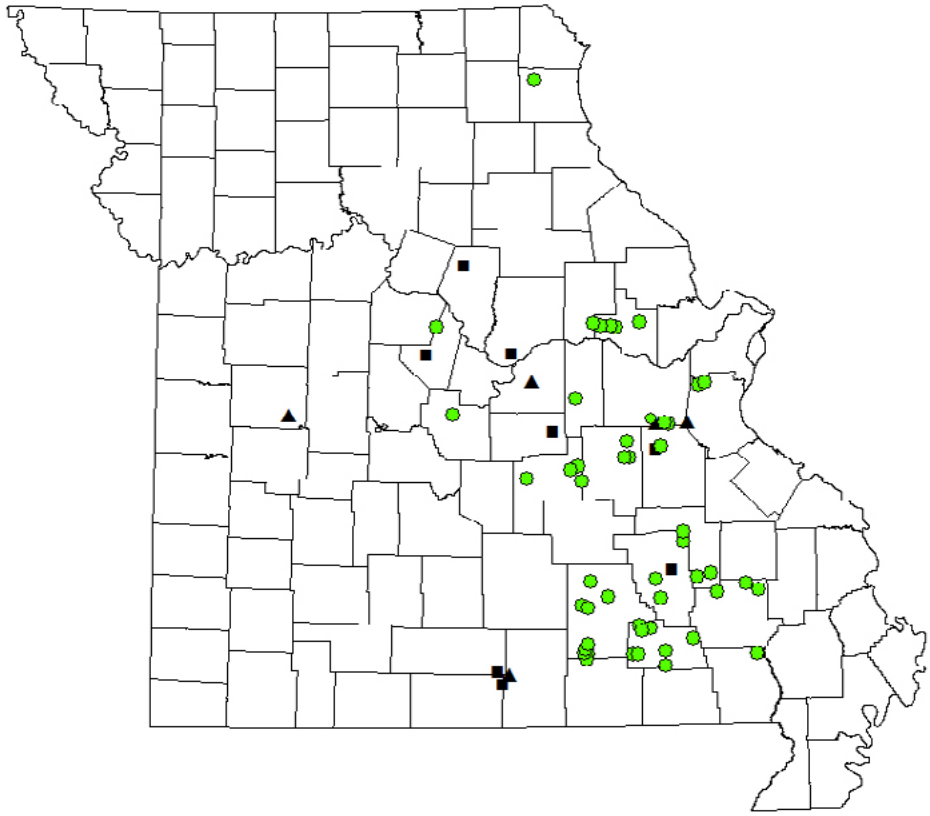

Figure 1.—Locations in Missouri that were surveyed between May and September 2014 and reported to have dead and dying oaks on an upper slope (black square), lower slope and drainage (green circles), or on the upper and lower slopes (black triangles)

RESULTS

Forestry professionals (the respondents) returned 63 survey forms, each for a separate location. Foresters from the MDC completed 94 percent of the survey forms, U.S. Forest Service employees completed 5 percent of the forms, and private industry completed 1 percent of the forms. Respondents reported white oak mortality at locations as far north as Lewis County, MO, and as far west as Henry County, MO (Fig. 1). Most surveys were completed for locations in the southeastern quadrant of Missouri and the counties immediately next to that quadrant. Forty-three of the described locations were public lands and 20 were private lands. Of the 43 public locations, 34 were state owned and 9 were federally owned. Respondents described one or more land uses for each location: 89 percent were listed as forest, 16 percent as riparian, 8 percent as roadside, 5 percent as pasture, and 2 percent as yard. No locations were park or urban, and no land use was reported for four locations.

Respondents reported white oak mortality mostly on lower slopes and in drainages throughout the southeastern quadrant of Missouri and in one location in northern Missouri (Fig. 1). In a few cases, respondents reported white oak mortality on lower and upper slopes. These few reports of mortality on lower and upper slopes tended to be in an east-to-west band just below the center dividing line of the state (Fig. 1). Respondents also reported white oak mortality only on upper slopes at eight locations. These reports were in the southeastern quadrant of the state and in nearby counties (Fig. 1). The eight reports of white oak mortality on upper slopes only were excluded from compilation of survey results because they fit the typical pattern of oak decline. The remaining 55 reports of RWOM on lower slopes and in drainages or on lower and upper slopes were used to develop a description of RWOM.

Table 1.—Surface soil textures typical of the surveyed areas at each of the 55 surveyed locations with white oak mortality on lower slopes and in drainages

Surface soil texture	No. of locations
Silt loams	
Extremely gravelly silt loam	1
Very gravelly silt loam	10
Gravelly silt loam	5
Silt loam	3
Loams	
Very gravelly loam	4
Gravelly loam	4
Loam	4
Sandy loams	
Very gravelly sandy loam	5
Gravelly sandy loam	1
Very cobbly fine sandy loam	2
Fine sandy loam	2
Silt clay	
Very channery silt clay	1
Other	
Slightly decomposed plant material	13

Forestry professionals reported that RWOM was occurring on 13 slopes with protected aspects (316°-134° azimuth) and 21 slopes with exposed aspects (135°-315° azimuth). Aspects for three locations with protected slopes and aspects for nine locations with exposed slopes were within a few degrees of the neutral line between protected and exposed aspects (315°-316° and 134°-135°, respectively). No slope aspect was reported for the other 21 locations, because either the mortality was in a flat area at the base of the hill or no information was reported. RWOM that was associated with hillslopes was reported most often on gentle and strong slopes. The average slope steepness was 12 ± 2 percent. Respondents stated that RWOM areas were associated with drainages 65 percent of the time, and those drainages had intermittent or ephemeral water flow 95 percent of the time. Forestry professionals reported no drainages in the surveyed area for the other 19 reports.

Soil information from the Web soil survey for the 55 locations indicated that surface soil textures included loams, silty loams, and sandy loams that typically had low water-holding capacities (12.6 ± 1 cm/cm) because of a high percentage of coarse fragments (Table 1). Approximately one-third of locations had soils with a restrictive layer of 31-150 cm depth.

Respondents described RWOM in a variety of stand types. Slightly more than half of RWOM-affected stands were dominated by trees in the white oak group or the white and red oak groups. The other 44 percent of RWOM-affected stands were a mixture of oak species with pine or a mixture of white oak with other non-oak hardwoods. These RWOM-affected stands had white oak group regeneration 75 percent of the time. Respondents reported that mortality ranged from fewer than 25 percent of white oak trees in the surveyed portion of stands to complete or nearly complete mortality. Stands with complete or nearly complete mortality were scattered

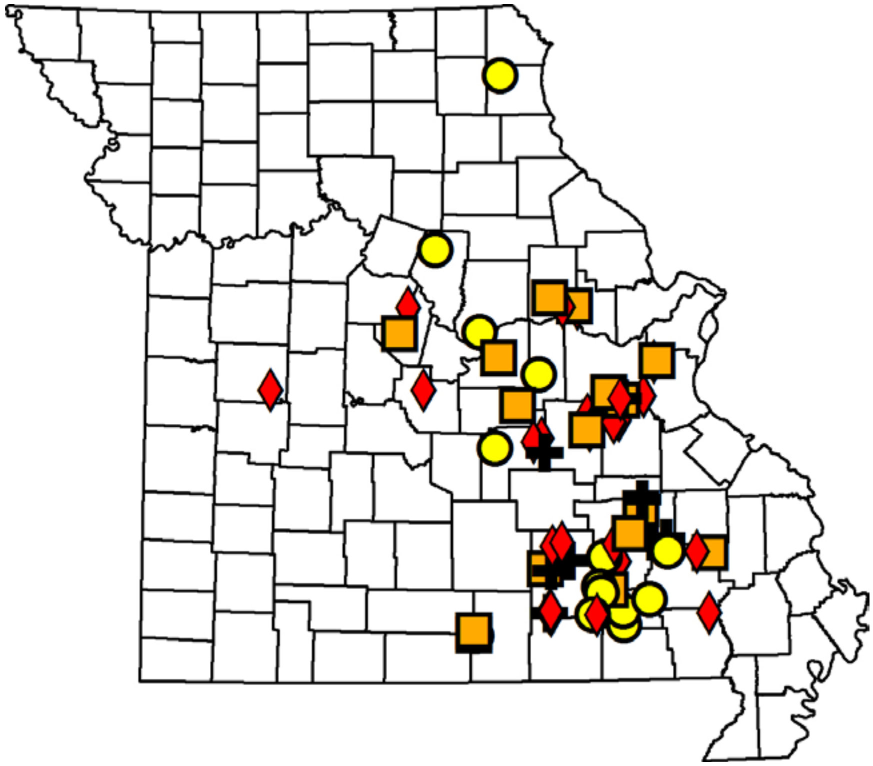

Figure 2.—Severity ratings for white oak mortality at each location surveyed between May and September 2014. Forestry professionals rated the severity as slight with <25 percent of white oak trees dead (yellow circle), moderate with 26–50 percent of white oak trees dead (orange square), severe with 51–75 percent of white oak trees dead (red diamond), or very severe with 76–100 percent of white oak trees dead (black plus sign). Map by Robbie Doerhoff, Missouri Department of Conservation, Forestry Division.

throughout southeastern Missouri (Fig. 2). These respondents also reported other dead and declining tree species in RWOM-affected stands. Trees in the red oak group were dead and declining at 20 percent of the locations and post oak trees were dead and declining at 22 percent of the locations. Respondents indicated that codominant and dominant white oaks larger than 25 cm diameter at breast height were dead and declining at nearly all locations, and that smaller codominant and dominant white oaks were dead and dying at one location. Fewer respondents stated that intermediate and suppressed trees were dead or declining at 40 percent and 16 percent of locations, respectively. Individuals also indicated that 71 percent of the surveyed locations still had declining white oak trees.

The four disease symptoms or signs most often associated with RWOM stands were upper branch dieback (80 percent of stands), Biscogniauxia (Hypoxylon) canker (56 percent of stands), fallen bark on the ground (55 percent of stands), and wilted leaves attached to trees (42 percent of stands). Respondents reported insect holes on declining trees at 13 locations. Round insect holes ≤ 2 mm wide were reported at 11 locations, and holes approximately 5 mm wide were reported at 6 locations.

Respondents described a number of weather and management events at RWOM locations since 2005. Respondents indicated that 95 percent of the RWOM locations experienced severe drought, 67 percent experienced the 2012 severe frost or the 2007 late spring freeze, and 45 percent experienced the exceptionally wet growing seasons of 2008 and 2009. Flooding was reported at 31 percent and severe winds at 25 percent of the locations. Respondents indicated that insect defoliation had occurred at 35 percent of the locations and often cited a looper

(Lepidoptera: Geometridae) outbreak during 2014. A jumping oak gall outbreak in 2010 was reported only 11 percent of the time. Respondents indicated that some sites had undergone active management since 2005, and timber stand improvement was reported at 31 percent of locations. Most timber stand improvement was performed to thin the understory and remove undesirable trees. Other timber stand improvements included timber sales and sometimes salvage sales. Respondents also indicated that 11 percent of the locations had been exposed to a moderate to severe burn.

DISCUSSION

This is the first summary of the distribution of RWOM in Missouri. The large number of reports of dead and declining white oaks on lower slopes and in drainages confirmed a large scale die-off of white oaks and showed that it is associated predominantly with the Salem Plateau of the Ozark Highlands. The Salem Plateau is a hilly, forested region formed mostly by the erosion of sedimentary, Ordovician rocks. The plateau also includes a smaller, rugged region of weathered igneous Cambrian and pre-Cambrian outcroppings associated with the St. Francois Mountains (Unklesbay and Vineyard 1992). The Ultisols and Alfisols in the region support vigorous white oak growth.

Six RWOM locations were also in the transition zone between the glaciated plains and the Salem Plateau in an area north of the Missouri River. Parts of this zone were not glaciated, and these locations may share some characteristics with the Salem Plateau and some with the glaciated plains.

Two survey locations with white oak mortality on lower slopes or in drainages were distant from other RWOM locations and should be investigated further. One location in Lewis County was in the glaciated plains and the other was in the Osage Plains. The soils and topographies of these two regions are very different from those of the Salem Plateau. If these two reports of RWOM are similar to those in southeastern Missouri, soils and some site characteristics might contribute less to this syndrome than other factors.

RWOM was associated with lower slopes and drainages in valleys with ephemeral and intermittent streams. The soils typical of these areas were sandy loams, silty loams, and loams derived from cherty dolomite and cherty sandstone residua that are characteristics of the Salem Plateau (Unklesbay and Vineyard 1992). The deaths of so many trees on these sites present a conundrum because white oaks are particularly well suited to these dry and mesic uplands, especially protected aspects on lower slopes (Johnson et al. 2009). Restrictive layers in soils can limit root growth and water availability, but these were present in soils at just one-third of the locations. One possibility is that trees affected by RWOM occurred in pockets with soils that are atypical of the general area. Trees in ephemeral or intermittent stream areas may also have been affected by root pathogens, such as *Phytophthora*, that are favored by these conditions. *Phytophthora* root rot is most severe when infected trees are exposed to too-wet and too-dry cycles because of abundant zoospore production and infection followed by host stress (Corcobado et al. 2013).

Respondents most frequently mentioned the 2012 drought (52 locations), when asked about exceptional weather events at RWOM locations. This historic statewide drought first developed in the southeastern and west-central sections of Missouri in April 2012 and spread through the rest of the state during May, with little relief until September. The hydrologic deficit was not made up until the following year. Only a small part of the RWOM affected area was affected by a short period of drought during late summer 2011. Some respondents also noted a late spring frost during 2012 and an April freeze during 2007. The spring frost during 2012 occurred in late April in the Ozarks, when vegetative growth began 3 weeks earlier than normal. Most of the early growth was observed in low-lying areas within steep valleys. The freeze during 2007 was an 82-hour event that occurred in early April and affected branch tips and leaves that had emerged 2-4 weeks earlier than normal. Fewer than half the respondents listed the record-breaking years of 2008 and 2009, when Missouri received more than 106 inches of precipitation in a 2-year time period. All these events likely decreased tree health; however, the role of drought reported during 2012 is particularly unclear. RWOM was first reported in 2011, a year before the drought entered southeast Missouri. Although white oak is considered a drought-tolerant species, the peak in reports during the spring and early summer of 2012 may indicate that many white oaks were of low vigor before the drought and succumbed quickly to the extreme heat and lack of water.

Respondents indicated that timber stand improvement (i.e., thinning) had been performed at nearly one-third of the locations between 2005 and 2014. Some of the reported management likely had little effect on RWOM because it occurred after or during the mortality event. For example, six respondents mentioned that mortality was already present in stands at the time of management. One question was whether active management would reduce or preclude mortality. Unfortunately, too few of the reported locations had active management before RWOM to assess this question. This would likely be worth pursuing in the future. RWOM was reported in only three locations despite thinning or harvesting 4 or more years earlier.

RWOM appears to be a new syndrome in Missouri, with a unique combination of characteristics shared across the Salem Plateau. Unique characteristics include high white oak mortality, especially in the absence of increased mortality of other oak species, as well as high numbers of dead trees on lower slopes and in drainages. These characteristics are the reverse of those associated with oak decline, which has been ongoing in Missouri for decades (Fan et al. 2012, Kabrick et al. 2008). The timing and time course of RWOM are also unique in relation to drought. RWOM was reported just before or during the early development of a severe drought at many of the surveyed locations. In contrast, oak wilt often peaks 2 to 3 years after a drought (Fan et al. 2012).

CONCLUSION

This description of RWOM was developed from 55 of 63 surveys that describe white oak mortality on the lower slopes and in drainages. RWOM is occurring on federal, state, and private lands and appears to be a regional phenomenon. It occurs on lower slope positions and in drainages. Although the die-off varies by location, it can be very severe. Soils typically found at these locations are gravelly and well drained and little available water is stored. RWOM occurs in stands that are dominated solely by the white oak group, as well as in more diverse stands. White oak trees of all crown positions can be affected, but most are large dominant and codominant trees. At some locations, dead and declining post oak trees and trees in the red oak group may be present. At any specific location, trees may die rapidly with leaves still attached within a single growing season; others decline over a longer time period. Biscogniauxia canker may be associated with some declining trees and may be the cause of bark piles on the ground and rapid deterioration of branches. Locations with RWOM have experienced severe drought, late spring freezes and frosts, and exceptionally wet weather that may have further weakened trees. Future research should aim to confirm the survey reports taken at each location and to scientifically test portions of the description in this paper to see if the description correctly describes this new syndrome.

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PLANNING FOR AND IMPLEMENTING AN EMERALD ASH BORER-INDUCED FOREST RESTORATION PROGRAM IN MUNICIPAL WOODLANDS IN OAKVILLE, ONTARIO

Peter A. Williams and Candace Karandiuk

Abstract.—Oakville is an urban municipality with 846 ha of woodland. Management priorities are to maintain forest health, environmental health, and safety; wood production is a minor objective. The town developed a comprehensive strategy to plan for emerald ash borer (EAB; *Agrilus planipennis*) induced ash mortality and forest restoration. Oakville has begun implementing its forest management program focusing on removing and salvaging dead and dying ash from its woodlands and supporting forest regeneration. This management plan is described in this paper along with some of the challenges of its implementation.

INTRODUCTION

The town of Oakville is located on the coastal plain of the north shore of Lake Ontario west of Toronto and east of Hamilton (Fig. 1). Oakville has been a leader in urban forest health management in Ontario since dealing with catastrophic oak decline in several of the town's woodland parks (Williams and Schwan 2013). Since then, threats from other insects such as gypsy moth (*Lymantria dispar*), emerald ash borer (*Agrilus plannipenis*), and Asian longhorned beetle (*Anaplophora glabrapennis*) have required monitoring and action.

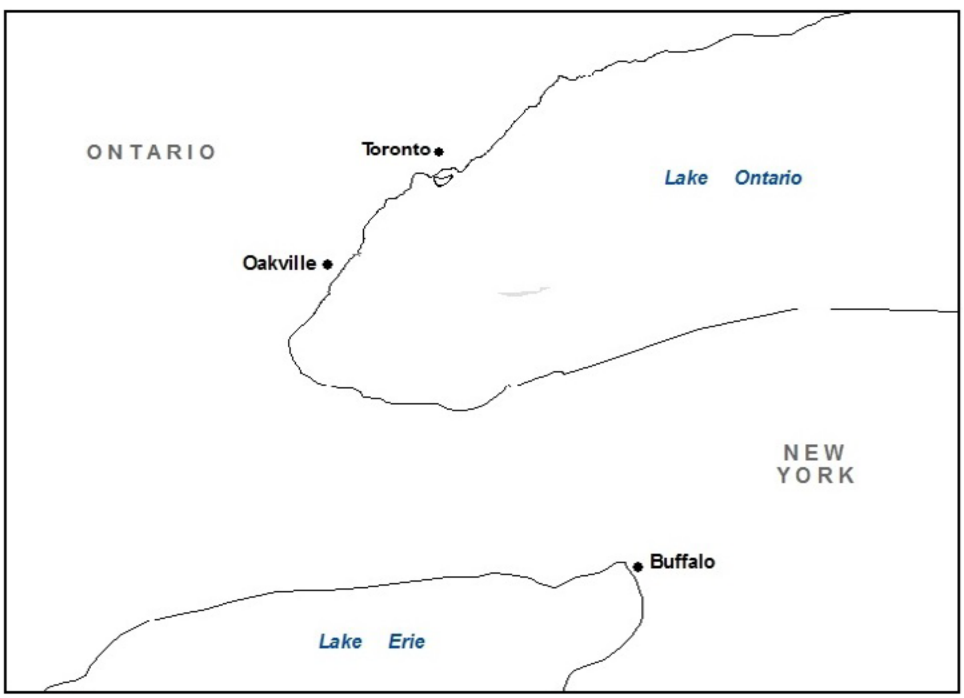

Figure 1.—Oakville, Ontario location map.

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Emerald ash borer (EAB) was discovered in Oakville in 2008 and since then the town, like many, has worked diligently to manage the EAB crisis. While the town has protected most ash in developed parks and along streets, currently (2015) most ash in Oakville woodlands are dead or heavily infested with EAB. The town is required to mitigate hazardous situations on their property. So while forest health management and planning was underway in Oakville prior to 2008, EAB presented an urgent need to develop and implement plans to deal with the crisis and try to maintain healthy forests and a safe environment.

Forest Management Policy

Oakville has had a progressive policy of woodland acquisition since the 1980s, and its woodlands were acquired as part of the development process where hazard lands, parks, or other areas were conveyed to the town. As an area developed, forests on these lands were subsequently planted or developed naturally. The town currently owns 821 hectares (2029 acres) of woodland on 154 properties (larger than 0.5 ha) and 25 hectares of smaller groups of trees on 118 parcels. Other woodlands may have been acquired as a purchase or transfer from another public agency.

Oakville was one of the first municipalities in Ontario to employ professional foresters as urban forest managers starting in 1978. Management of the town's urban and natural forests has benefitted greatly by employing four registered professional foresters, in part because of the amount of private and public forest in the town, but primarily because of the training in tree and forest health and the strategic approach to forest management that are foci of the forestry profession.

While the town was successful in retaining a significant urban forest complex, the forest has been lightly managed as the primary management objectives were environmental protection, outdoor recreation, and wildlife habitat. This changed in 2002 when several significant woodland parks were affected by catastrophic oak decline (Williams et al. 2013). This situation focused attention on the need to increase forest and tree health and take action to keep it healthy. Management since then has focused on restoration after the oak decline, gypsy moth survey and management, and activities associated with EAB and other invasive pests.

After review of forest management procedures used to deal with the oak decline problem, Oakville made the commitment to manage its forests according to internationally recognized guidelines developed by the Forest Stewardship Council (FSC) and to certify its woodlands as well-managed through the FSC/Smartwood Certification program. The FSC has developed the most rigorous criteria for ensuring that other values are considered in forest management. Although wood production and sale are not important management objectives for the town, forest certification recognizes that wood products generated as a by-product of management activities come from well-managed woodlands. Wood generated by activities such as salvaging wood from trees cut for safety or forest health reasons will be responsibly harvested and marketed as a principle of sustainable forest management, to help maintain aesthetic qualities and to recover some of the cost of management activities.

Policy formulation, planning, and implementation sometimes proceeded concurrently because of the rapidly developing problems with EAB infestation. Strategic treatment of street and park ash with TreeAzin® systemic insecticide (azadirachtin; Canadian Forest Service and BioForest Technologies, Sault Ste. Marie, ON, Canada) was initiated to protect trees from EAB; this action started immediately after EAB was found, but in the meantime, broader strategies and policies were developed.

EAB MANAGEMENT PLANNING

Oakville began developing an EAB Management Strategy in 2008. The EAB Management Strategy (Bioforest 2009) was never officially adopted by the town but most of its strategic recommendations for EAB and ash management were implemented. These recommendations included an inventory of the ash resources in the town, the development of an EAB management plan, and a multi-year budget allocation to effectively carry out defined management strategies and tactics. EAB management plan recommendations include:

- Surveys
  - Street and park tree inventory
  - Forest resource inventory
  - EAB delineation survey
  - Annual EAB monitoring surveys (branch sampling, etc.)

- Ash Reduction
  - Removing infested trees
  - Establish EAB sinks to attract and then kill EAB
  - Treatment with TreeAzin® to protect high value ash
  - Phloem reduction by removing ash prior to infestation

- Collaborative Research and Development
  - Work with agencies regarding wood movement restrictions
  - Collaborate with Natural Resources Canada (NRC) and Ontario Ministry of Natural Resources (OMNR) experts and researchers

- Effective communications plan to the public and staff
  - Annual review of EAB research
  - Annual review of EAB management plans and refinement
  - Development of a decision support model (DSM)

Most of the recommendations were effectively implemented. For the Ash Reduction strategy (listed above), only the TreeAzin treatment was effectively implemented. Proactive ash removal and other interventions to reduce EAB and ash populations provided too many communication challenges. A strategic choice was made to protect as many ash street and park trees over 20 cm d.b.h. as possible. Effective implementation of the treatment program required accurate EAB delimitation information, proactive ash protection, and an accurate street and park tree inventory.

Oakville partnered with Natural Resources Canada to participate in the development of the EAB branch-sampling methodology (Ryall 2013), which showed greater reliability than EAB survey procedures.

Oakville's current EAB management program has six components:

- **Canopy Conservation**: treatments to save 75 percent of the municipal ash canopy on the street and in parks using TreeAzin® for trees 20 cm d.b.h. and greater
- **Roads and Active Parks Hazard Abatement**: removal of dead and dying ash along roads and in active parks (i.e., those trees not in the canopy conservation program)

- **Woodlands Hazard Abatement**: removing dead and dying ash throughout woodlands over a 10-year period
- **Canopy Replacement**: replanting of trees lost due to EAB
- **Quality Assurance**: strategic analysis, review, and guidance through mapping, treatment strategy, monitoring of insect populations, and efficacy reporting
- **Community Engagement**: public relations, advertising campaigns, and a communications advisor

FOREST MANAGEMENT PLANNING

Many of the town's forest health management activities have been in response to critical forest issues such as oak decline (2002) (Williams et. al. 2013), gypsy moth (2007), EAB (2008 and ongoing), and invasive plant management. The town developed a strategic urban forest management plan in 2008 (Urban Forest Innovations and Kenney 2008), prior to the discovery of EAB infestation. However the strategic planning process was partially supplanted by the EAB find and associated planning and management.

Draft forest management plans were initially developed for properties affected by oak decline and some closest to EAB infestations. However, these plans were never finalized but instead were incorporated into a new planning process where a general Forest Management Plan (FMP) (Oakville 2013) was developed that outlined forest management strategies, procedures to assess and monitor the forests, and methods to implement site-specific prescriptions for forest management and restoration activities. The FMP applied to forested sites, not to small patches of trees and the urban forest in developed areas.

Forest management planning was underway before EAB was discovered in 2008 as a requirement for FSC certification. However, EAB imparted urgency to the process because significant ash mortality was expected over the next 5 to 10 years in the area.

A forest resource inventory (FRI) of town woodland properties began in 2010. The FRI, including all GIS mapping, compartment delineation, and layer creation, was completed in 2012. This provided data to define compartments or stands within woodlands. The woodland management pilot project (WMPP) assessed methods of removing dead, dying, and hazardous ash in 2012.

A general forest management plan (FMP) described the Oakville's woodland management goals, and objectives. It outlined procedures for 5-year operating plans (OPs) and annual operating plans (AOPs); introduced critical issues associated with EAB infestation; and included ash management and forest regeneration strategies for ash stands in Oakville.

Woodland Management Pilot Project

The woodland management pilot project (WMPP) assessed methods of removing ash from 11 woodland properties where there were imminent safety concerns. This experience helped to refine operational strategies into four approaches: natural processes, arboricultural strategy, and harvest strategies (two levels). In practice, the strategies are often combined on a property. The approaches are as follows:

- **Natural Processes**: Recommended in areas where the dead ash are left to break down by natural processes. This strategy can be used when the ash are very small, where they are not near trails, property lines, other public use areas and/or otherwise do not pose a moderate or high risk to the public.

- **Arboricultural Strategy**: Recommended in locations where the ash are near trails, property lines, or other public use areas; are small and can easily be handled manually; where there is little usable wood from the ash; or where access to the property is not appropriate for the harvest approach.
- **Harvest Strategy:** Recommended where there are significant ash numbers and sufficient wood volume to warrant salvaging for lumber or firewood. Salvaging helps expedite the work, save costs, results in less debris/waste, and provides revenue to offset the cost of removal. The salvage strategy requires a reasonable volume of wood, and a work location (landing) with access for large trucks and harvesting equipment and an area for log piling. Various equipment (e.g., conventional skidders/forwarders, small-scale equipment and horses) will be considered for use in each property, and the approach used on a particular property will depend on site characteristics, access, weather, public use, and contractor availability.
- **Modified Harvest Strategy:** Recommended for areas with larger trees and limited access for equipment or trucks. This strategy may be used for cutting trees and moving wood within the forest, but the wood will not be removed from the property. Generally, the harvesting equipment will be used to help fell trees and move the wood and debris away from the trails or more visible areas, but left in the forest to break down.

Ash Management and Forest Regeneration Strategies

The FMP outlined several ash removal strategies to be used in forest operations. These were based primarily on safety and the regeneration strategies to be used following EAB infestation. Stand assessments conducted for the silvicultural prescriptions (SP) provide current and more detailed information on stand composition and structure, existing regeneration, and competition potential from invasive plants (e.g., buckthorn). The SP considers the viability of natural regeneration and its implications for forest development considering stand disturbance from EAB-induced ash mortality and/or ash removals. Based on this consideration, the SP identifies potential regeneration sites and strategies for stands or parts of stands.

The FMP suggests up to 15 percent of the forest area affected by ash removals will be actively regenerated and that forest development in the remaining area will be left to natural processes. Two active regeneration strategies were developed for allocation to woodland sites considered appropriate for the approach. The regeneration strategies are:

- **Natural Processes:** Specified where no active regeneration will be implemented and the forest will rely on natural processes for woodland development. This option is to be used in areas where the residual stand and regeneration is sufficient to continue as, or develop into a healthy woodland; and where poor access, dominance of invasive shrubs, public considerations, and/or limited resources precluded active regeneration efforts.
- **Regeneration Enhancement:** Encourages the development of desirable forest trees where a stand has some good natural regeneration or is not entirely dominated by buckthorn. This strategy includes strategically planting trees where they are most likely to survive and controlling buckthorn that is competing with existing natural regeneration or planted seedlings.
- **Prime Sites:** For sites that are accessible, where regeneration efforts are likely to succeed with acceptable costs and efforts, and where planting and tending can be supported by community participation. Prime Sites include more intensive site management and,

where appropriate, site preparation (e.g., soil tillage to remove competing weeds and provide a clean site for planting; control of buckthorn and other invasive undesirable plants; and tending and replanting to ensure adequate stocking for up to 5 years).

Twenty regeneration sites were established in the eleven forest properties involved in the WMPP using strategies and assessment procedures described below.

WOODLAND OPERATIONS PLANNING

Woodland Property Prioritization

Oakville selects and prioritizes its woodlands for management by using the decision support model (DSM) developed through a combination of staff expertise and a consultant/contractor. The DSM uses the FRI and information on public use, local EAB populations, and other factors to suggest which woodland parks are more likely to present public safety concerns associated with the dying ash, and these sites are given a higher priority.

Branch sampling surveys and active trapping of EAB using green volatile pheromone traps (Ryall 2013) are used to assess where insects have reached peak or are past peak and not declining as the EAB moves to alternate locations. Both survey methods have been used to quantify densities of the insects to create delimitation maps for planning purposes. The areas where ash removals are currently being implemented are where EAB was first discovered, and these areas coincide with the delimitation mapping.

Tree Removals

Removal strategies are chosen based on the size and amount of ash in the stand, access, and its proximity to a maintained trail or woodland edge. Arborist strategies are incorporated near trails and woodland edges (i.e., where the woodland edge is near public use areas such as playgrounds, maintained park or private property). All ash and other trees that represent potential hazards over 10 cm (4 inches) d.b.h. and within a tree length of a maintained trail or woodland edge are marked for removal using a harvest strategy or for pruning using an arboriculture strategy. Using criteria in the Forest Operations Manual (Oakville 2013), ash and other trees near trails, property lines, or other public use areas that are judged to be unsafe to be removed by Certified Arborists and forest technicians are marked with orange paint for removal or pruning by an arborist contractor. Those that are judged to be safe to be removed by harvesting crews are marked for removal with yellow paint.

Trees to be removed to improve forest health include trees suppressing good oaks and thinning some trees in denser patches. Trees to be removed using arboricultural methods (e.g., climbing with ropes) to minimize damage to property or other trees were marked in orange as required in the Forest Operations Manual.

Monitoring

Once contractual arrangements have been made, work will be initiated within the acceptable time frames outlined in the prescriptions and contract. Regular monitoring will be conducted by Oakville forestry staff and contractors to ensure that work is done according to requirements and that variances from the specifications are approved and/or documented. From time to time, alterations may be made to alter the work specifications in response to changes in weather, equipment availability, and/or new instructions from the town.

FOREST REGENERATION

Forest Regeneration activities were started at 18 sites covering 10.6 ha (26.5 acres) on the 11 properties. The procedures for regeneration prescriptions, planting, competition control, testing, and assessments were tested/developed through this experience.

Active forest regeneration activities are implemented on up to 15 percent of the forest area subject to ash removal operations. Active regeneration includes regeneration enhancement, and prime site strategies and practices are documented in a regeneration prescription developed for each property.

Opportunities for forest restoration are reassessed during operations planning and implementation. Where active regeneration strategies are identified (e.g., Enhancement or Prime Site), a regeneration prescription (RP) will be developed that describes site preparation, planting, and tending activities. Once completed, the recommendations in the prescriptions will be used to adjust the list of properties for operations in each AOP.

Regeneration Prescriptions

Regeneration prescriptions include a map showing regeneration strategies for each woodland. This identifies parts of properties or compartments where an Enhancement or Prime Site strategy will be used to facilitate forest regeneration and where natural processes will be allowed to occur on property. The RP will describe site conditions (i.e., soils and vegetation) and will recommend site-specific procedures for regeneration and plans to achieve "free to grow" (FTG) status of acceptable seedling numbers through an establishment/maintenance period of 5 years.

A RP using a regeneration Enhancement Strategy would include limited competition control and/or site preparation, tree planting, and perhaps limited follow-up planting or tending. The Enhancement Strategy will be used to augment existing seedlings where there are short-term planting opportunities caused by ash mortality, limited natural regeneration, and shrub density where it may be possible to help the forest regeneration by planting trees that will increase the number and variety of young trees, with limited tending and investment.

A RP using a Prime Site strategy would include more intensive invasive plant control (e.g., buckthorn), site preparation (e.g., brush or debris removal, soil amendment), replanting, tending, and assessment. These sites will assessed and tended annually for 5 years with FTG assessments at years 3 and 5. These assessments will determine the success of the forest regeneration efforts.

Stocking and Free To Grow Assessments

The RP describes activities on active regeneration sites to achieve an "acceptable" stocking level of 700 to 1000 FTG seedlings per ha (280 to 400/acre) that have reached FTG status within 5 years. A tree that meets FTG criteria is a well-established desirable tree with little competition free from nondesirable species; is at least 1.0 m from other FTG trees; and has a good likelihood of developing into a mature tree. Desirable seedling and FTG stocking classes and downstream tending and replanting are determined using adjusted stocking levels that add 250 seedlings/ha to the stocking for each of 4 (0 to 3) canopy closure classes for two layers of canopy: 2 to 5 m and >5 m height.

Sites are monitored annually to plan tending activities and each site is subject to FTG assessments in years 3 and 5. The need for significant tending or replanting efforts is identified in the 3-year FTG assessment. The 5-year FTG assessment will determine whether the stocking objectives have been met. After the 5-year assessment, work may be discontinued whether or not the objectives have been met; however a supplemental planting may be recommended if that alone planting will help meet objectives.

The 18 established regeneration sites received their 3-year stocking and FTG assessments in October 2015. The assessment revealed that 16 of the 18 properties had acceptable stocking or better of desirable seedlings and that four sites had achieved acceptable stocking of trees that had reached FTG status. The assessment suggested that buckthorn control, other tending, and additional planting were recommended at 7, 13, and 4 sites respectively. It is anticipated that most sites should achieve acceptable FTG Stocking levels in 2017.

Regeneration Contracting

Contracts were developed for regeneration efforts; these contracts are then modified for each tendering process. The regeneration contract included buckthorn control, site preparation, supplying genetically-appropriate plant materials, planting, and tending (including watering where required). The initial contract was for a single year to get started, and it is hoped that a multi-year contract format can be developed.

FOREST HEALTH AND HAZARD MONITORING

Forest health monitoring work began in 2014 when 90 woodlands were surveyed and 30 areas had forest monitoring plots installed. Forest monitoring plots generally used the same plot centers as FRI plots. Pest detection surveys will be conducted in all town woodlands over a 3-year cycle (approximately 90 per year) and reported annually. Oakville has 127 km of trails through forest compartments and 277 km of woodland perimeters that bound adjacent properties or town land used for other purposes (e.g., open parks, parking). Tree hazard assessment along trails and property lines are scheduled so that 81 km of trails and woodland perimeters are assessed each year on a 5-year cycle, in concert with other forest operations. That being said, assessment findings or contingencies often cause schedules to be changed in response to issues like forest health problems (e.g., gypsy moth infestations) or other critical issues such as storms (e.g., ice or wind damage to trees).

SUMMARY

Oakville has begun implementing its forest management program in earnest, mainly focusing on removing and salvaging dead and dying ash from its woodlands and supporting forest regeneration. Ash removals on 19 properties were completed by arboricultural and harvest contractors during the fall/winter of 2015. Additional properties are targeted for 2016. Tree removal costs were lower and regeneration costs higher than anticipated. A good understanding of the challenges associated with removal operations has been achieved, but challenges remain in the economical implementation of regeneration programs. Particular challenges include the costs associated with buckthorn control and brush cleanup, browsing by rabbits and deer, and the complex planting arrangements that use many species and sizes.

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BIOMASS AND CARBON

GUIDELINES FOR SAMPLING ABOVEGROUND BIOMASS AND CARBON IN MATURE CENTRAL HARDWOOD FORESTS

Martin A. Spetich and Stephen R. Shifley[1]

Abstract.—As impacts of climate change expand, determining accurate measures of forest biomass and associated carbon storage in forests is critical. We present sampling guidance for 12 combinations of percent error, plot size, and alpha levels by disturbance regime to help determine the optimal size of plots to estimate aboveground biomass and carbon in an old-growth Central Hardwood forest. The analyses are based on five 100-percent inventories covering a 66-year time period. Disturbance regimes during that time included periods of grazing, low tree mortality, and high tree mortality. The size and number of plots recommended for estimating biomass and carbon changed with the type of disturbance. This information can be used to help design inventories of biomass and carbon in older forests.

INTRODUCTION

Millions of acres of forest in the northern United States are expected to grow into age classes older than 100 years over the next two decades (Shifley et al. 2014), and most of those old forests will have experienced some degree of prior partial disturbance. Determining the current and future significance of these forests in biomass and carbon storage is important to resource managers, climate scientists, resource scientists, and policy makers. With limited forest monitoring resources, managers need efficient inventory methods to estimate biomass and carbon in old forests. Efficient inventory taking and biomass and carbon monitoring will improve understanding of the role of old forests in carbon sequestration and bioenergy production, both of which are relevant to managing climate change.

Spetich and Parker (1998b) published plot size recommendations for biomass estimation in old-growth hardwood forests; however, the tabular data presented in that publication were not sufficient to meet the current need for a wide range of inventory options. Accurate estimates of biomass and associated carbon estimates in hardwood forests have become increasingly important in climate-change-related policy and management decisions. Our objective in this paper is to provide information that is complete enough to support inventory designs for monitoring biomass and carbon in old midwestern hardwood forests. Analyses improve guidance on the interaction of plot size, sample size, and precision to support well-informed decisions about designing inventories for maturing upland hardwood forests.

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Table 1.—Physical characteristics of the 7.92-ha core area of the Davis Research Forest, Randolph County, Indiana

Year	Basal area (m^2/ha)	Density (number of trees/ha)	Average d.b.h. (cm)	Total biomass (Mg/ha)
1926	24	165	38	154
1976	31	320	27	207
1981	34	338	27	220
1986	33	329	27	216
1992	33	312	28	211

METHODS

Study Site

The study site is Davis Research Forest, a 21 ha old-growth deciduous forest in Randolph County, IN. Physiographically, the site is mainly mesic with inclusions of wet-mesic to hydric areas (Spetich et al. 1999). In 1926 a 100 percent census was done in which every tree 10.2 cm or larger was measured and tagged with a unique identification number. To facilitate the inventory, the forest was divided into a mapped grid of 55 contiguous plots. After field measurements were completed, all trees were plotted on a large-scale map.[2] These trees continued to be measured throughout their lives including inventories in 1976, 1981, 1986, and 1992. This paper focuses on the core 7.92 ha of the Davis Research Forest where data are most consistent across all inventories and influences of edge effects are minimized. Parker et al. (1985) and Spetich and Parker (1998a) report additional study details.

The forest overstory is dominated by oak-hickory (*Quercus-Carya*) species, and the understory is dominated by maple (*Acer*) and elm (*Ulmus*) species. Between 1926 and 1992 the average diameter at breast height (d.b.h.) decreased while density increased because of a developing understory. Tree density was greater in 1992 than in 1926 because of the ingrowth of small diameter trees, which resulted in a lower arithmetic average d.b.h. (Table 1). Biomass of all trees in the core area of the forest larger than 10 cm d.b.h. was 154 Mg/ha in 1926 and increased to 211 Mg/ha by 1992. The biomass of trees in forest understory (trees from 10 to 25 cm d.b.h.) more than quadrupled during that time, however, increasing from 4 Mg/ha in 1926 to 17 Mg/ha by 1992 (Spetich and Parker 1998a). Previous disturbances on this site included livestock grazing between the mid-1800s and 1917 that likely created the sparse understory observed during the 1926 inventory.

Analysis

Biomass dry weight of each tree was calculated for each of the five inventory periods: 1926, 1976, 1981, 1986, and 1992. Tree bole biomass was calculated using Hahn and Hansen (1991) equations; tree-top and branch biomass was determined using Smith (1985) equations. Species-specific site index values for use with these equations were determined from Neely (1987), Carmean (1979), and the Natural Resources Conservation Service (data from the Soil Conservation Service integrated resource information system, based on a statewide inventory for site index). For further details, see Spetich and Parker (1998a).

[2] Bur M. Prentice. 1927. Forest survey No. 1 Herbert Davis Forestry Farm unpublished report to the Department of Forestry and Conservation, Purdue University, West Lafayette, IN.

Table 2.—The 16 plot sizes used within the 7.92-ha core area of the Davis Research Forest for comparisons of sample size

Plot dimensions (m)	Number of plots in core area	Plot area (ha)	Total area of plots (ha)
10 × 10	792	0.01	7.92
20 × 20	198	0.04	7.92
30 × 30	84	0.09	7.56
40 × 40	44	0.16	7.04
50 × 50	24	0.25	6.00
60 × 60	21	0.36	7.56
70 × 70	12	0.49	5.88
80 × 80	10	0.64	6.40
90 × 90	8	0.81	6.48
50 × 180	8	0.90	7.20
60 × 180	7	1.08	7.56
70 × 180	6	1.26	7.56
80 × 180	5	1.44	7.20
90 × 180	4	1.62	6.48
100 × 180	4	1.80	7.20
110 × 180	4	1.98	7.92

Sixteen plot sizes were used to estimate biomass based on sample size comparisons (Table 2). The mean and standard deviations of biomass were calculated for each plot size. These results were used to estimate sample sizes required to estimate biomass within 5, 10, and 20 percent of the mean. Specifically (Freese 1962)

$$n = \frac{1}{\frac{E^2}{t^2 C^2} + \frac{1}{N}}$$

Where

n = the number of units in the sample,

N = the total number of sample units in the entire population,

E = allowable error (percent),

t = student's statistic for a specified α and n −1 degrees of freedom, and

C = the coefficient of variation for a particular plot size.

The sample size, n, was determined for all 16 plot sizes at allowable error levels of 5, 10, and 20 percent. For each error level, n was determined at α levels of 0.01, 0.05, 0.10, and 0.20. Sample size values were computed for each of the five inventory dates to develop recommendations for sampling biomass under the various disturbance regimes.

In each case, the most efficient plot size and sample size combination was designated as that which required measuring the least total area (plot size × n) at a specified allowable error and α. To simplify comparison of alternatives, we assumed that sampling efficiency was directly related to total area sampled to achieve a given allowable error (e.g., travel costs and plot establishment costs were zero).

Table 3.—Size and number of plots necessary to inventory total biomass while measuring the least total area for each percent error and α level combination (based on data from Davis Research Forest, Randolph County, Indiana)

		Disturbance regime														
		Grazing			Low mortality [a]						High mortality [b]					
		Year of Measurement														
		1926 [c]			1976			1981			1986			1992		
% error	Alpha	Plot size (ha)	Number of plots [d]	Total area (size x number)	Plot size (ha)	Number of plots	Total area (size x number)	Plot size (ha)	Number of plots	Total area (size x number)	Plot size (ha)	Number of plots	Total area (size x number)	Plot size (ha)	Number of plots	Total area (size x number)
5	0.01	0.01	642	6.42	0.09	62	5.58	0.09	62	5.58	0.09	58	5.22	0.64	8	5.12
5	0.05	0.01	564	5.64	0.09	50	4.50	0.09	50	4.50	0.09	47	4.23	0.64	6	3.84
5	0.10	0.01	504	5.04	0.09	43	3.87	0.09	42	3.78	0.09	39	3.51	0.64	5	3.20
5	0.20	0.01	408	4.08	0.09	32	2.88	0.09	32	2.88	0.09	29	2.61	0.64	4	2.56
10	0.01	0.01	410	4.10	0.09	34	3.06	0.09	34	3.06	0.09	30	2.70	0.09	33	2.97
10	0.05	0.01	303	3.03	0.09	23	2.07	0.09	23	2.07	0.09	21	1.89	0.09	22	1.98
10	0.10	0.01	241	2.41	0.09	18	1.62	0.09	18	1.62	0.09	16	1.44	0.09	17	1.53
10	0.20	0.01	166	1.66	0.09	12	1.08	0.09	12	1.08	0.09	11	0.99	0.09	12	1.08
20	0.01	0.01	167	1.67	0.09	14	1.26	0.09	14	1.26	0.09	13	1.17	0.09	14	1.26
20	0.05	0.01	109	1.09	0.09	9	0.81	0.09	9	0.81	0.09	8	0.72	0.09	9	0.81
20	0.10	0.01	80	0.80	0.09	7	0.63	0.09	7	0.63	0.09	6	0.54	0.09	7	0.63
20	0.20	0.01	51	0.51	0.04	10	0.40	0.04	10	0.40	0.09	4	0.36	0.04	11	0.44

[a] Low mortality = 1,299 kg/ha/year from 1977 to 1981.
[b] High mortality = 3,534 kg/ha/year from 1982 to 1992.
[c] Note: In 1926 even a small difference in cost of travel between plots would favor the 0.04-ha plot size. The 0.04-ha plots required measuring no more than 0.09 ha greater total area than the 0.01-ha plot size (see Table 4).
[d] Number of plots of a given size required to estimate mean biomass (or carbon) for live trees >10 cm d.b.h. within the indicated percent error (5, 10, or 20 percent) while accepting a probability of 1 – α that the error associated with the sample mean will be greater than the indicated percent error.

RESULTS

Recommended plot sizes for biomass estimation range from 0.04 to 0.64 ha and differ by disturbance regime (Tables 3 and 4). The smallest plot size was the most efficient in 1926 shortly after the grazing disturbance and the largest size was the most efficient for the 1992 measurement when the acceptable error was 5 percent.

Grazing during decades before 1926 reduced the biomass of small diameter trees. The 0.01-ha plot size (Table 3) was the most efficient in 1926, but it required only slightly less total area than the 0.04-ha plot size (Table 4). For the 1926 data set, samples based on the 0.04-ha plot size required measuring only slightly more total area than the 0.01-ha plot size. Estimated differences in the total area that needed to be measured when using the 0.01 versus the 0.04-ha plot sizes were small: 0.08 ha for the 5 percent error with an α of 0.20, 0.06 ha for the 10-percent error with an α of 0.20, and 0.05 ha for the 20 percent error with an α of 0.20. Conversely, using the 0.01-ha plot size required 3.6 to 4.0 times more plots (depending on E and α level) than the 0.04-ha size. Even a small difference in cost of travel between plots would favor the 0.04-ha plot size over the 0.01-ha plot size in this case (Tables 3 and 4).

For inventory years 1976, 1981, and 1986, the most efficient plot size for aboveground tree biomass estimation was usually 0.09 ha (Table 3). The only two exceptions were for 1976 and 1981 when the most efficient plot size was 0.04 ha at 20-percent error with α of 0.2. In those two cases, however, the total area to be sampled differed by only 0.05 ha between the 0.04-ha and 0.09-ha plot sizes.

Table 4.—Size and number of plots necessary to inventory 1926 total biomass while measuring the least total area for a 0.04 ha plot

		1926		
% error	Alpha	Plot size (ha)	Number of plots	Total area (size × number)
5	0.01	0.04	161	6.44
5	0.05	0.04	142	5.68
5	0.10	0.04	127	5.08
5	0.20	0.04	104	4.16
10	0.01	0.04	105	4.20
10	0.05	0.04	78	3.12
10	0.10	0.04	62	2.48
10	0.20	0.04	43	1.72
20	0.01	0.04	46	1.84
20	0.05	0.04	29	1.16
20	0.10	0.04	22	0.88
20	0.20	0.04	14	0.56

Tree mortality was relatively high between 1986 and 1992. Consequently, for some situations, the most efficient plot size increased to 0.64 ha. The 0.64-ha plot, however, was most efficient only at an allowable error rate of 5 percent. At 10- and 20-percent allowable error rates the 0.09-ha plot size was still the recommended size. At the 20-percent error and 0.2 α level, the 0.04-ha plot size was recommended (Table 3), but the difference in total area sampled for the 0.04-ha plots versus the 0.09 ha plots was only 0.01 ha.

For practical purposes, efficient strategies for measuring live tree biomass apply equally to measurement of carbon in live trees. The quantity of carbon in hardwoods is approximately half the biomass dry weight (Lamlom and Savidge 2003), and converting biomass estimates to carbon estimates by a constant multiplier of 0.5 has no influence on sample size estimates. When comparing estimates of carbon sequestered in trees with estimates of atmospheric carbon emissions, it is important to remember that by convention atmospheric carbon emissions are typically reported in terms of equivalent carbon dioxide (CO_2; i.e., two oxygen atoms joined with each carbon atom) rather than as carbon per se. Given that the atomic weight of a CO_2 molecule is 44 (two oxygen atoms at 16 each and one carbon atom at 12), 1 ton of carbon sequestered in biomass corresponds to 3.67 tons of atmospheric CO_2.

DISCUSSION

In this analysis we simplified the comparison of alternatives by assuming that efficiency was directly related to total area sampled to achieve a given allowable error. In real-world applications, however, we must consider numerous other factors, including travel costs to sites and between plots, plot establishment costs, plot perimeter versus borderline trees, and topography. All factors can vary between locations. The tables we provide can be used as a starting point to incorporate the considerations that are unique to particular circumstances into the decision-making process (e.g., see Husch et al. 2003). The plot size and number in Table 3 can assist in developing a biomass sampling scheme for aging upland hardwood forests. Table 3 also gives insight into how various types of disturbances may influence sample design (Cochran 1963, 1977; Freese 1962; Johnson 2000).

In 1926, it would have been necessary to inventory a greater area at each α level and percent error combination than in later measurement years. Grazing that ended by 1917 had reduced much of the understory biomass, resulting in high coefficient of variation values for biomass. This also resulted in a much patchier distribution of understory biomass (Ward et al. 1996). In 1926, sampling with the 0.01-ha plot size would have required up to four times more plots than the 0.04-ha plot size, which would have considerably increased the plot perimeter when these sampling options were compared at a 5-percent error and an α of 0.05. Other factors being equal, larger plots are more desirable because they have a smaller total perimeter for a given cumulative sampled area. Smaller plot sizes result in more borderline trees along plot perimeters. When dealing with borderline trees, determining whether they are in or out of the plot boundary is time consuming, and imprecise determinations can introduce a considerable source of error to sample estimates (Loetsch et al. 1973). For instance, the perimeter of 564 0.01-ha plots is nearly twice that of the perimeter for 142 0.04-ha plots.

In most cases a plot size of 0.01, 0.04, or 0.09 ha was appropriate for sampling aboveground biomass in this population. Plot sizes of 0.01 or 0.04 were the best options soon after understory disturbance, such as the grazing in the Davis Research Forest just before the 1926 inventory. A plot size of 0.09 ha was appropriate in most other inventories from 1976 to 1992. The most significant exception was after high mortality with a 5-percent error for the 1992 inventory where the most efficient plot size was 0.64 ha when the allowable error was 5 percent.

In practice the plot size and number of plots may need to be adjusted when live-tree biomass is inventoried simultaneously with other attributes of old forests such as tree density, dead trees, or coarse woody debris. Other studies that have investigated sampling for old-growth midwestern forests (e.g., Shifley and Schlesinger 1994) have shown that estimates of tree density are efficient with small plots sizes (e.g., 0.01 ha). Larger plot sizes (e.g., 0.1 ha) are more efficient for taking inventory of basal area and presumably for biomass and carbon estimates that are typically highly correlated with basal area. For a given plot size, forest attributes such as downed wood or density of standing dead trees generally have higher coefficients of variation than biomass, basal area, or volume and thus require larger plot sizes or larger sample sizes to estimate mean values with comparable levels of precision (Lombardi et al. 2014, Shifley and Schlesinger 1994).

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SIMPLE TAPER: TAPER EQUATIONS FOR THE FIELD FORESTER

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Abstract.—"Simple taper" is set of linear equations that are based on stem taper rates; the intent is to provide taper equation functionality to field foresters. The equation parameters are two taper rates based on differences in diameter outside bark at two points on a tree. The simple taper equations are statistically equivalent to more complex equations. The linear equations in simple taper were developed with data from 11,610 stem analysis trees of 34 species from across the southern United States. A Visual Basic program (Microsoft®) is also provided in the appendix.

INTRODUCTION

Taper equations are used to predict the diameter of a tree stem at any height of interest. These can be useful for predicting unmeasured stem diameters, for making volume estimates with variable merchantability limits (stump height, minimum top diameter, log lengths, etc.), and for creating log size tables from sampled tree measurements. Forest biometricians know the value of these equations in forest management; however, as currently formulated, taper equations appear to be very complex. Although standard taper equations are flexible and produce appropriate estimates, they can be difficult for many foresters to calibrate or use. Most foresters would not use taper equations unless they are embedded within a computer program. Additionally, very few would consider collecting the data to parameterize the equations for their local conditions.

The typical taper equation formulations used today are a splined set of two or three polynomials that are usually quadratic (2^{nd}-order polynomial) or cubic (3^{rd}-order polynomial) (Max and Burkhart 1976). These polynomials are usually constrained to be equal at the join points (i.e., at specific heights on the tree stem). This is accomplished by constraining the first derivatives of the polynomial equations. If the second derivatives are constrained to be equal at the join points, the lines will be smooth through the join points. This process often generates equations that are quite complex and difficult to use. Some formulations can be solved as closed-form equations, but not all functions can be solved in this form. The complexity of taper equations has likely limited their use.

This paper proposes an approach to taper estimation that maintains many of the advantages and concepts of taper equations, yet uses a simpler quantitative approach that is easier to parameterize. This approach invokes assumptions from forest stand dynamics (Oliver and Larson 1996) and a number of ideas from Jensen's matchacurve papers (Jensen 1973, Jensen and Homeyer 1971). When tested against equations that were developed using the standard methods of splined sets of polynomials to predict volume from large regional stem analysis data sets, the simple taper equations produced statistically equivalent results.

BACKGROUND

There is a large body of literature on taper equations, and they have been parameterized for most commercially important tree species in many regions worldwide. Taper equations have many advantages over volume equations in that they allow the user to choose the merchantability limits for the tree stem.

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The shape of a tree stem is such that a single parameter-based equation is difficult to fit to the shape. The exterior profile of a tree usually bends sharply near its base, is linear along the central portion of the bole, and is variable along the upper stem (Shaw et al. 2003). The shape can vary greatly by species. To accommodate this shape variation, biometricians have used splining equations, which use separate equations in different parts of a curve. The equations are constrained to pass through a common point by requiring the first derivatives to be equal at those points. Most authors also constrain the second derivatives to be equal, which makes the modeled taper line smooth at the join point. Generally two join points are used that are commonly forced to be a point of information (e.g., diameter at breast height [d.b.h.] and diameter in the upper stem) or adjusted iteratively to a point that produces the best fit to the observed data.

When developing a growth model, I wanted a taper equation that could be used easily with very simple data requirements for the end user but in a format that could use available data. The central portion of the stem is very nearly a linear equation, and the slope of the line could be described as an average taper rate; this information could come from either a rate of change between two points on the tree or an assumed taper rate selected by the end user. These are basic measurements that are often already taken and can provide the data needed to calibrate the equation for local conditions. To produce a function that approximates the shape of the tree stem, I followed the work of Jensen (1973) and Jensen and Homeyer (1971) for combining multiple-component models to develop a curve that can reproduce the same results of the standard modern taper equations. Jensen's works suggested that data could be standardized, graphed, and compared to standardized curves, and he provided papers on the exponential and sigmoidal equations and a paper about combining equations to produce compound curves. These methods were cumbersome to use in the 1970s because they required tracing graphs on paper using a standard curve. They are quite easy to use today with graphical computer programs. These papers provided the inspiration for the following approach.

METHODS

Equations

As stated previously, the simple taper relies on some assumptions from stand dynamics. The shape of the central part of the stem (between breast height and crown base) can be described as a tapered cylinder or the frustum of a cone (Fig. 1). The volume of the frustum can be calculated with equation 1:

$$V = \frac{L}{3}\left(A_l + \sqrt{A_l A_s} + A_s\right) \qquad (1)$$

Where

V = the cubic volume in the units of L and $\sqrt{A}$,

L = the length of the stem section,

A_l = the area of the large end of the log, and

A_s = the area of the small end of the log.

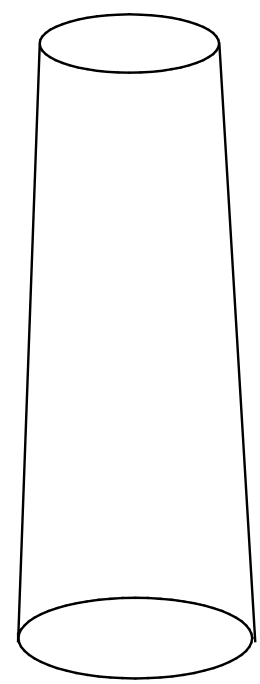

Figure 1.—A frustum of a cone using equation 1.

Using this idea, a tree profile is described as three parts: (1) below breast height, (2) breast height to crown base, and (3) stem within the crown.

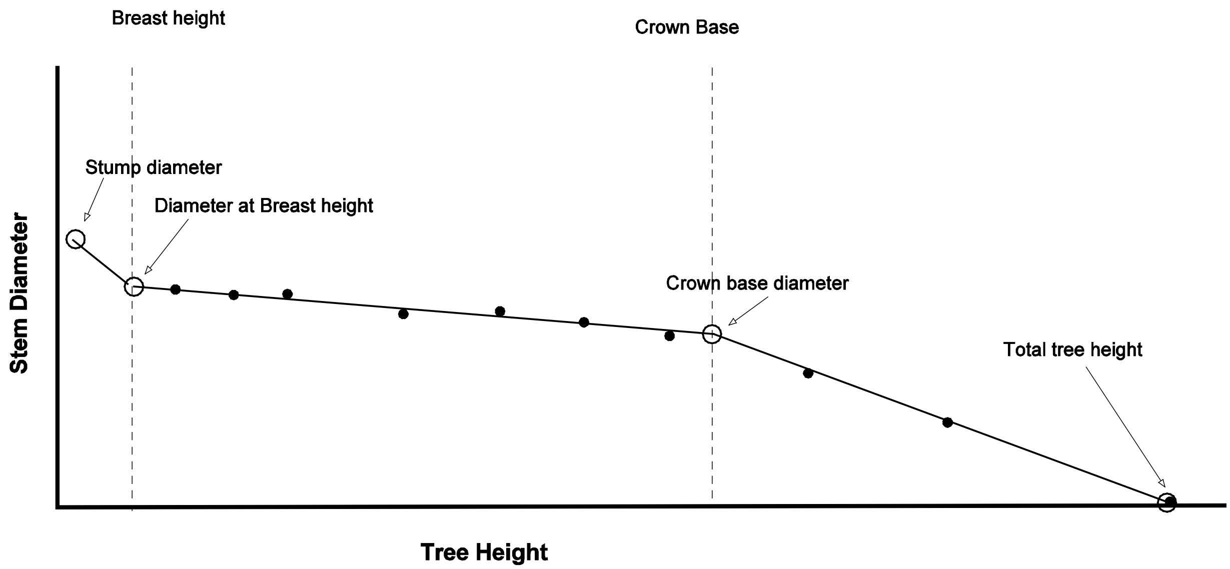

Figure 2.—Illustration of a simple taper profile. Diameter at breast height and diameter at crown base are used as the step between equations.

The typical method of illustrating taper equations is to describe the stem profile relative to the center of the stem (Fig. 2). In this diagram, y dimension is the radius of the stem and the x dimension is the height of the tree. When interpreting taper equations one can think of revolving the taper line around the stem center to estimate stem volume. This is simply the integral of the taper line with respect to height.

In most trees, the portion of the stem from breast height to the base of the crown produces a linear or nearly linear stem profile. This is supported by the long history and wide application of simple taper rates to describe trees and logs. Taper rates describe the change in the stem diameter per one unit of height (height and diameter units are the same). This can be derived from the slope parameter in a simple linear regression. The taper rate on trees is relatively easy to estimate from either standing or recently harvested trees. It requires two diameters measured on the stem at a known distance apart. These data can be collected with a dendrometer, a d-tape and a ladder, a caliper or log-scale stick on a cut log, or stem analysis on felled trees.

In all trees examined in this study, the taper rate below breast height differed from the taper rate above breast height, so a taper rate below breast height is also estimated. The taper rate for the crown portion of the tree does not need to be estimated from data because the diameter at the base of the crown is presumably predicted with the above breast height taper rate estimated with the diameter at crown base, and the diameter at the top of the tree is assumed to be 0. These are assumed to form a cone.

To use the simple taper equation, the user needs only two parameters: a taper rate below breast height and a taper rate above breast height.

The functions assume that at breast height the tree should equal diameter at breast height. For the section below breast height the equation is

$$d_h = dbh + p_b * \left(h - bh\right) \qquad (2)$$

Where

d_h = the diameter at height h on the tree,

dbh = the diameter at breast height,

p_b = the stump taper rate from Table 1 for the tree species,

h = the height at which the user desires a diameter prediction, and

bh = the breast height.

Note: all dimensions are in the same units.

The parameters are negative and are defined as the decrease in diameter for one unit increase in tree height with all dimensions in the same units. In equation 2, (h–bh) will be negative and will add to the breast height diameter.

Table 1.—Average taper rates by species (unitless)

Name	Code	n	Stump taper rate	Stem taper rate
Sand pine	107	237	–0.0257	–0.0115
Shortleaf pine	110	430	–0.0390	–0.0132
Slash pine	111	2867	–0.0460	–0.0128
Spruce pine	115	18	–0.0354	–0.0112
Longleaf pine	121	1104	–0.0373	–0.0103
Table Mountain pine	123	74	–0.0326	–0.0097
Pitch pine	126	30	–0.0375	–0.0132
Pond pine	128	488	–0.0414	–0.0124
White pine	129	170	–0.0209	–0.0107
Loblolly pine	131	1706	–0.0496	–0.0142
Virginia pine	132	203	–0.0268	–0.0117
Bald cypress	221	103	–0.1208	–0.0173
Pond cypress	222	351	–0.1645	–0.0199
Hemlock	260	25	–0.0170	–0.0149
Red maple	316	385	–0.0422	–0.0142
Hickory	400	380	–0.0518	–0.0158
Beech	531	18	–0.0529	–0.0172
Sweetgum	611	728	–0.0427	–0.0146
Yellow-poplar	621	212	–0.0453	–0.0121
Sycamore	731	97	–0.0568	–0.0140
Cherry	762	79	–0.0210	–0.0098
White oak	802	181	–0.0713	–0.0169
Scarlet oak	806	57	–0.0682	–0.0153
Southern red oak	812	274	–0.0624	–0.0211
Cherrybark oak	813	96	–0.0635	–0.0152
Laurel oak	820	450	–0.0722	–0.0172
Overcup oak	822	5	–0.1688	–0.0212
Water oak	827	446	–0.0531	–0.0168
Willow oak	831	57	–0.0625	–0.0202
Chestnut oak	832	81	–0.0307	–0.0125
Northern red oak	833	22	–0.0603	–0.0122
Post oak	835	154	–0.0565	–0.0196
Black oak	837	36	–0.0762	–0.0165
Black locust	901	46	–0.0216	–0.0100

For the section above breast height and below crown base the equation is:

$$d_h = dbh + p_s * (h - bh) \qquad (3)$$

Where all values are the same except for p_s, which is the stem taper rate from Table 1 for the tree species.

For the stem above crown base, a cone is assumed with the following equation:

$$d_h = (ht - h) * top\_ht\_rate \qquad (4)$$

Where

d_h = the diameter at height h on the tree,

ht = total tree height,

h = the height at which the user desires a diameter prediction, and

top_ht_rate = the implicit parameter calculated in the following steps:

1. Determine the diameter at crown base with equation 3.
2. Determine crown length as *total height – height to crown base.*
3. Calculate the *top_ht_rate* as *crown base diameter/crown length.*

The additions to the linear equation between breast height and the stump can presumably be described as the frustum of a cone as well, but with a different taper rate. To describe the volume between stump height and breast height, a frustum of a neiloid with top diameter d.b.h. and bottom diameter stump diameter works very well (Walters and Hann 1986). The differences between the frustum of a cone and the frustum of a neiliod are small, and the formula for a neiliod is more complex. A continuation of the main stem cone to stump height will capture most of the usable volume within this space.

Upper stem shape varies more than the other parts of the tree, and the author left three options for the user: (1) model the upper stem as a cone; (2) model the upper stem as a paraboloid, which adds volume to the upper stem compared to using a cone; or (3) model the upper stem as a neiloid, which removes volume from the upper stem compared to using a cone. The reasons for selecting a cone, paraboloid, or neiloid are covered in the discussion.

Data and Approaches

This study used the U.S. Forest Service, Southern Research Station, Forest Inventory and Analysis stem analysis data set. Cost (1978) described data collection. This study used only a subset of the data to demonstrate the methods (Table 2). Tree measurements on 11,610 trees and 34 species included d.b.h., total tree height, and crown ratio. The trees were sectioned along the stem at 4- or 5-foot intervals. At the ends of each section, diameter outside and inside of the bark and the section length were recorded.

From the stem profile data, outside bark values (i.e., the diameters at various heights on the stem) were calculated to demonstrate the methods. The taper rate, which is the change in diameter per unit of height, was calculated for each section of each tree. Mean taper rates were calculated for the two lower sections. For the top section the appropriate taper required to

Table 2.—Summary of the data used in this study including diameters (cm), heights (m), and height to crown base (m)

Name	Code	n	Minimum diameter	Maximum diameter	Minimum height	Maximum height	Minimum crown base	Maximum crown base
Sand pine	107	156	4.1	18.4	28	80	6.2	60.3
Shortleaf pine	110	418	5.1	21.7	24	96	6.4	59.2
Slash pine	111	2458	1.1	23.8	13	95	6.2	64.0
Spruce pine	115	17	7.7	22.2	58	83	21.6	52.5
Longleaf pine	121	1048	2.0	26.4	13	89	6.5	60.0
Table Mountain pine	123	73	4.9	13.6	35	64	21.5	53.1
Pitch pine	126	25	4.1	15.3	27	64	13.2	40.8
Pond pine	128	467	4.4	20.4	24	82	7.0	57.4
White pine	129	144	1.2	15.4	11	87	6.4	46.9
Loblolly pine	131	1648	5.1	34.7	19	98	6.2	64.0
Virginia pine	132	109	1.8	21.6	20	78	7.8	49.5
Bald cypress	221	97	1.1	20.1	14	97	10.5	64.4
Pond cypress	222	254	4.0	22.3	20	93	8.8	74.4
Hemlock	260	11	1.9	10.6	17	60	6.4	15.0
Red maple	316	204	1.7	26.0	25	90	7.2	56.0
Hickory	400	244	1.7	27.8	19	101	7.0	60.3
Beech	531	6	3.5	16.4	30	72	10.8	21.3
Sweetgum	611	508	1.6	23.9	14	105	6.3	70.4
Yellow-poplar	621	166	1.1	29.3	17	92	6.8	60.2
Sycamore	731	97	5.0	26.6	28	96	14.4	65.1
Cherry	762	38	1.4	16.6	19	81	7.6	48.0
White oak	802	179	5.1	27.6	25	101	7.5	60.6
Scarlet oak	806	55	5.1	27.7	33	87	11.7	52.2
Southern red oak	812	216	1.9	32.2	18	92	6.4	46.2
Cherrybark oak	813	96	5.1	26.0	42	93	7.2	51.1
Laurel oak	820	242	1.2	30.7	18	95	6.6	58.8
Overcup oak	822	5	9.6	33.2	52	98	15.6	46.9
Water oak	827	261	1.3	30.1	17	91	6.4	58.8
Willow oak	831	48	3.5	37.9	23	105	6.9	45.5
Chestnut oak	832	48	1.6	21.1	16	92	7.8	46.0
Northern red oak	833	22	5.1	15.4	42	75	20.4	39.0
Post oak	835	108	1.6	24.4	16	80	7.0	40.0
Black oak	837	36	5.1	24.3	28	82	14.7	45.5
Black locust	901	42	2.2	13.5	24	75	13.2	49.7

connect the crown base diameter to the top of the tree was selected (Fig. 2). From the data set, volumetric calculations using a frustum of a cone for each tree section were generated for each tree and considered to be the true volume.

Four approaches for estimating the taper rate were compared. The first approach was to calculate two taper rates for each tree by averaging the appropriate sections to create the two required taper rates. This approach requires stem analysis data for the tree being predicted; it is usually useful only for developing taper equations, but it could also be very useful for collecting optical dendrometry data. This method is labeled "tree taper" in Table 3. Figure 3 illustrates the difference between the volume calculated by the stem sections and the volume by the tree taper approach.

The second approach is for people who would like to use locally collected taper data without a formal stem analysis study. This method uses data from only four points on a tree: stump diameter, d.b.h, crown base diameter, and total tree height. This method could be implemented very easily during timber harvest to develop a local taper data set for a forest. This method is labeled "four point taper" in Table 3. Figure 4 illustrates the difference between the volume calculated by the stem sections and the volume derived using the four point taper approach. This approach works almost as well as the tree taper approach without the need to conduct a stem analysis study.

The third approach is to average the per-tree taper rates by species. This is the most practical use of these data because two taper rates are used: one below breast height and one above breast height for each species. This method is labeled as "average taper" in Table 3, and Table 1 provides a list of the values for the data set used in this study. Figure 5 illustrates the difference between the volume calculated by the stem sections and the volume by the average taper approach. This approach is the most practical for most foresters because only two taper rates per species are needed. The predictions from this approach are still very similar to those produced by the other methods.

The fourth approach is to use the stem profile equations published by Clark et al. (1991). This data set was developed from the same data set and used six parameters to predict the same trees. This method is labeled "Clark et al." in Table 3. Figure 6 illustrates the difference between the volume calculated by the stem sections and the volume by the approach described by Clark et al. (1991).

In all methods, the taper equation was used to predict the two end diameters for each of the three sections, and the cubic volume outside bark was calculated using the frustum of a cone. For these four approaches, the predicted cubic volume was compared to the volume from observed stem sections. The statistic for comparing this difference is root mean squared error (RMSE; see Table 3).

Table 3.—Root mean square error (RMSE) (m^3) by four ways of fitting the data including tree taper, the four point taper, average taper, and the six-parameter equations by Clark et al. (1991)

		RMSE			
Name	Code	Tree taper	Four point taper	Average taper	Clark et al.
Sand pine	107	0.0205	0.0224	0.0334	0.0206
Shortleaf pine	110	0.0310	0.0333	0.0542	0.0850
Slash pine	111	0.0385	0.0401	0.0670	0.0358
Spruce pine	115	0.0516	0.0524	0.0388	0.1346
Longleaf pine	121	0.0435	0.0460	0.0563	0.0526
Table Mountain pine	123	0.0408	0.0421	0.0446	0.0579
Pitch pine	126	0.0060	0.0135	0.0387	0.0359
Pond pine	128	0.0394	0.0406	0.0641	0.0224
White pine	129	0.0243	0.0248	0.0267	0.0160
Loblolly pine	131	0.0278	0.0336	0.0640	0.0221
Virginia pine	132	0.0147	0.0194	0.0221	0.0065
Bald cypress	221	0.0100	0.0146	0.0658	0.1784
Pond cypress	222	0.0403	0.0414	0.0947	0.1562
Hemlock	260	0.0171	0.0171	0.0205	0.0126
Red maple	316	0.0108	0.0213	0.0071	0.0944
Hickory	400	0.0063	0.0233	0.0441	0.1951
Beech	531	0.0003	0.0003	0.0089	0.2415
Sweetgum	611	0.0249	0.0359	0.0451	0.0752
Yellow-poplar	621	0.0247	0.0316	0.0121	0.1392
Sycamore	731	0.0038	0.0243	0.0039	0.0383
Cherry	762	0.0013	0.0089	0.0109	0.0393
White oak	802	0.0048	0.0453	0.0679	0.2523
Scarlet oak	806	0.0033	0.0130	0.0250	0.2908
Southern red oak	812	0.0007	0.0176	0.0482	0.1366
Cherrybark oak	813	0.0484	0.0652	0.0684	0.2063
Laurel oak	820	0.0136	0.0285	0.0136	0.1943
Overcup oak	822	0.3224	0.3224	0.5133	0.4014
Water oak	827	0.0113	0.0304	0.0365	0.1353
Willow oak	831	0.0048	0.0572	0.0369	0.2134
Chestnut oak	832	0.0183	0.0047	0.0129	0.1023
Northern red oak	833	0.0238	0.0006	0.0224	0.1181
Post oak	835	0.0356	0.0413	0.0556	0.1242
Black oak	837	0.0109	0.0207	0.0209	0.1908
Black locust	901	0.0085	0.0152	0.0010	0.0594

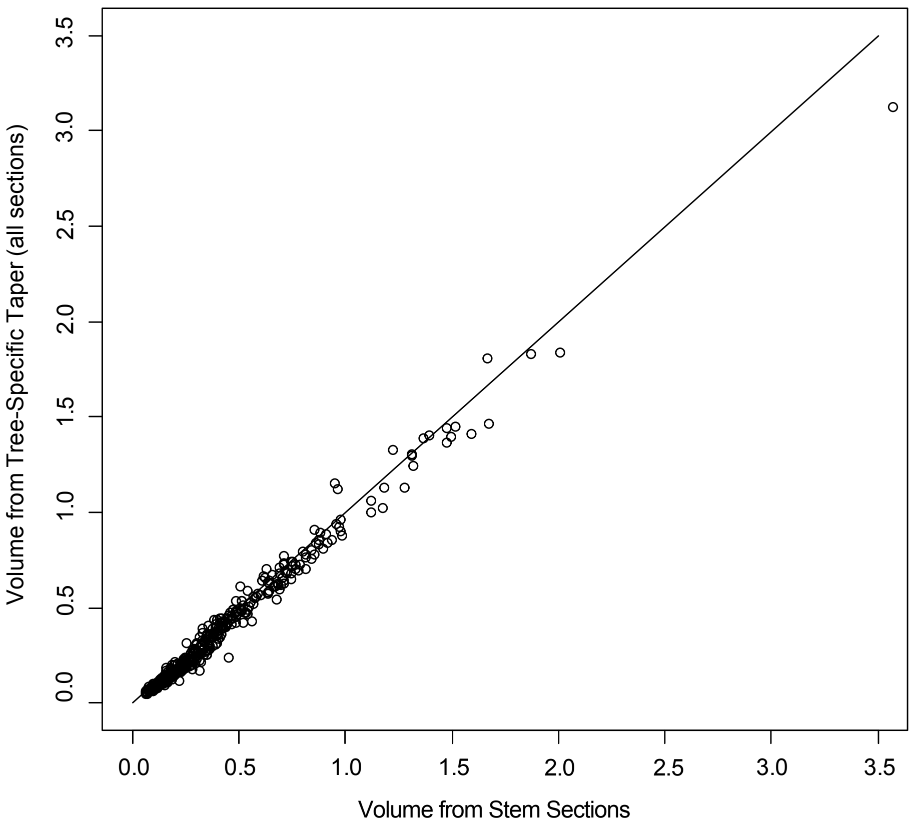

Figure 3.—A comparison of the volume (m^3) calculated from observed stem sections and the tree taper approach in Table 3. The data were from 418 shortleaf pine trees.

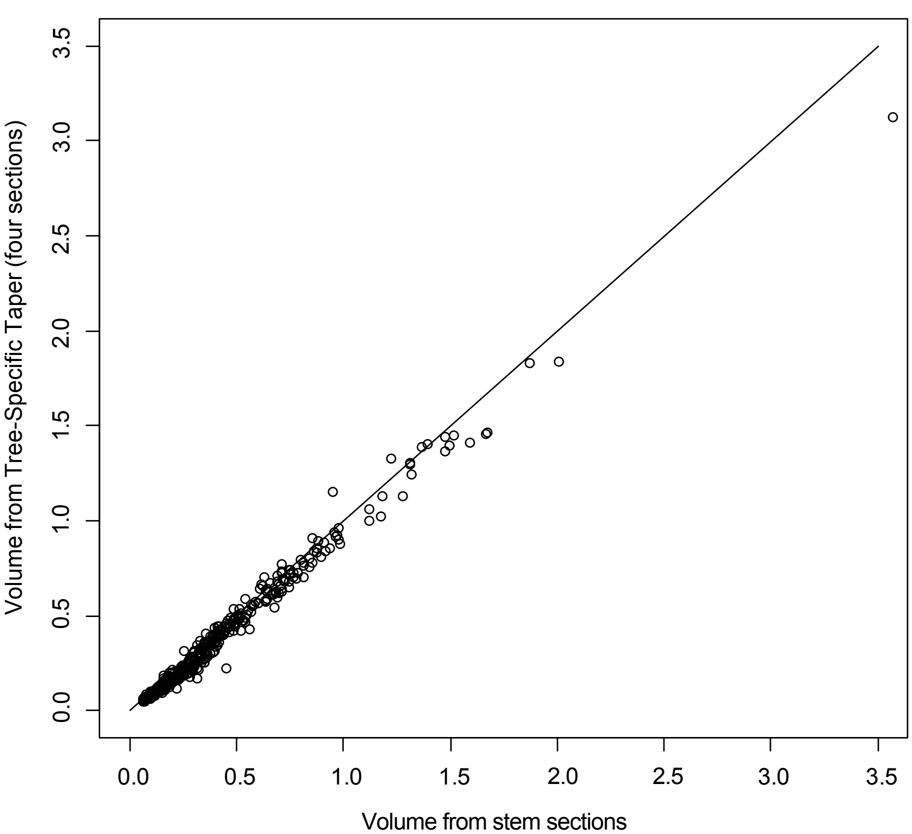

Figure 4.—A comparison of the volume (m^3) calculated from observed stem sections and the four point taper approach in Table 3. The data were from 418 shortleaf pine trees.

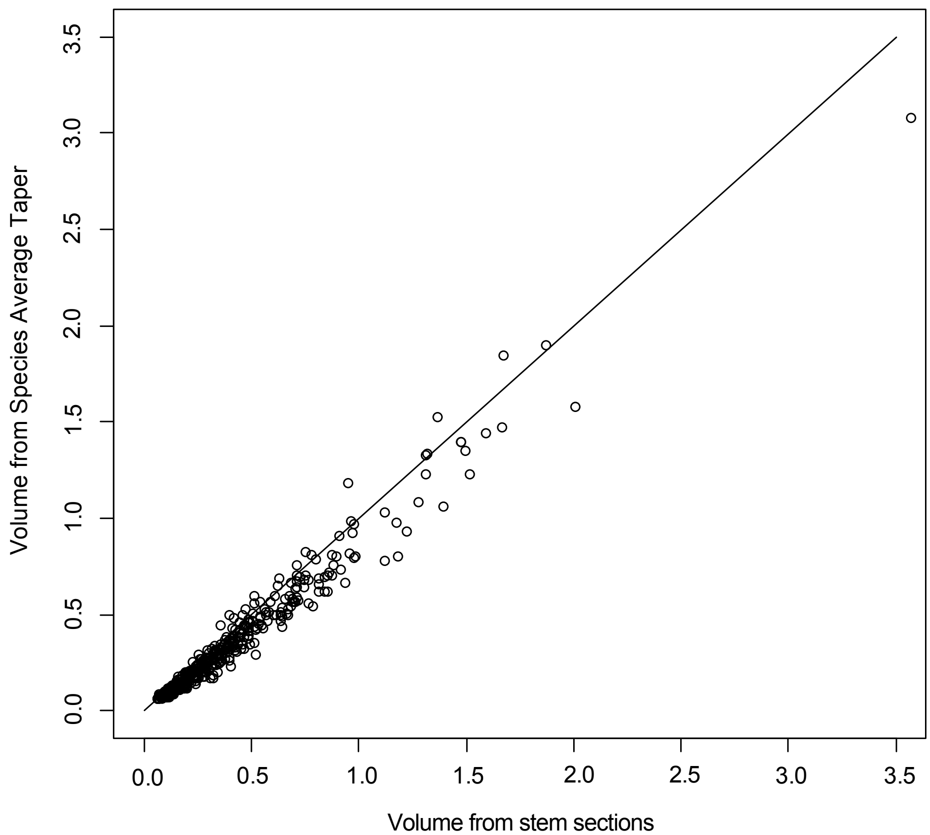

Figure 5.—A comparison of the volume (m^3) calculated from observed stem sections and the average taper approach in Table 3. The data were from 418 shortleaf pine trees.

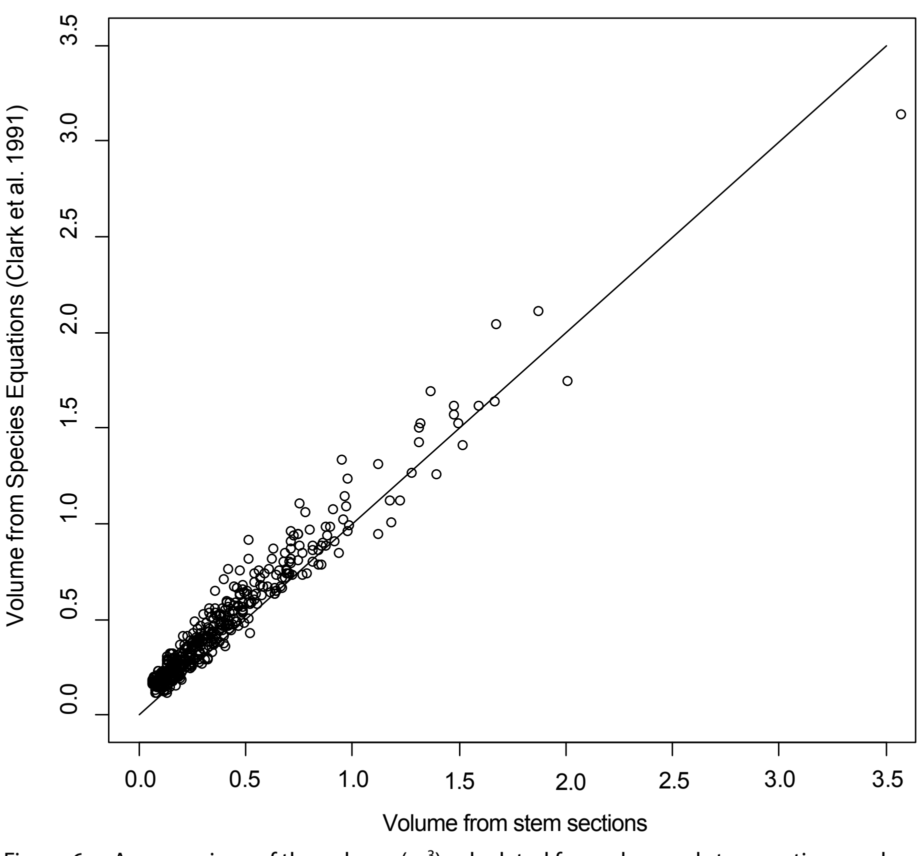

Figure 6.—A comparison of the volume (m^3) calculated from observed stem sections and the Clark et al. approach in Table 3. The data were from 418 shortleaf pine trees.

RESULTS

On average, for the 34 species in this study the tree taper method produced a 0.028 m^3 difference, the four point taper method produced a 0.051 m^3 difference, the average taper produced a 0.037 m^3 difference, and the Clark et al. (1991) equations produced a 0.120 m^3 difference. The Clark method works much better on most of the pines, and the methods proposed in this paper work better on the hardwoods. In general, these methods all work very well and could be considered statistically equivalent. The poor estimates seem to be from small sample sizes (e.g., overcup oak with only five trees).

Overall, the tree taper method generally provided the best fit of the simple taper approaches, but the data needs are greater than most people are willing to collect. In terms of bias, both the tree taper and the four point taper methods produce nearly identical patterns. The average taper approach tends to underpredict volume, and the Clark et al. (1991) equations tend to overpredict volume.

DISCUSSION

The beauty of the simple taper approach is its simplicity. It requires parameters that are simple to determine and logical for practicing foresters to collect. The rates are unitless and based on outside bark, so the same rate can be applied to all unit systems.

In Figs. 3 through 6, the tree taper approach (Fig. 3) works the best but requires stem diameter data on the tree of interest. Also, this approach slightly underpredicts the true volume. The four point taper (Fig. 4) works nearly as well but requires fewer measurements along the stem. This approach also slightly underpredicts the true volume. The average taper (Fig. 5) is the most practical approach and is presented in Table 3. Estimates made with this approach are similar to those made with the previous two methods, but are slightly underpredicted. All of the first three approaches are basically equivalent to the Clark et al. (1991) method, which slightly overpredicts the true volume (Fig. 6).

One of the important aspects of this approach is the estimation of crown base. This study used the crown base provided in the data set. Based on the stem profiles, there is usually a bend in the stem profile on any given tree that has experienced crown recession (Fig. 7). The crown base in this way of thinking is the base of the functional crown and usually does not include the residual branches on the stem below the crown. Many studies have shown that within the crown, the diameter increment is proportional to the leaf area above the point of interest (Jensen 1983). This means that the point of maximum diameter increment is at the crown base and the increment on the stem below the crown base is smaller and proportional to the size of the crown and the distance below the crown base. These observations are consistent with pipe model theory and observation (Valentine 1988).

This study showed that most of the differences were in the portions above the crown base. This portion of the stem is usually of low value or left in the woods as residue. The pine profiles within the crown tended to be slightly parabolic and hardwood profiles tended to be slightly neiloid. Some of these differences come from the fact that pines tend to exhibit excurrent branching and hardwoods tend to exhibit decurrent branching. These differences did not seem to be the same across the range of tree sizes of the various species. The current data set is insufficient to explore relationships between volume prediction and assumptions about shape for these species. The volume differences, however, are smaller than a few hundredths of a cubic meter if a paraboloid or neiloid shape is assumed.

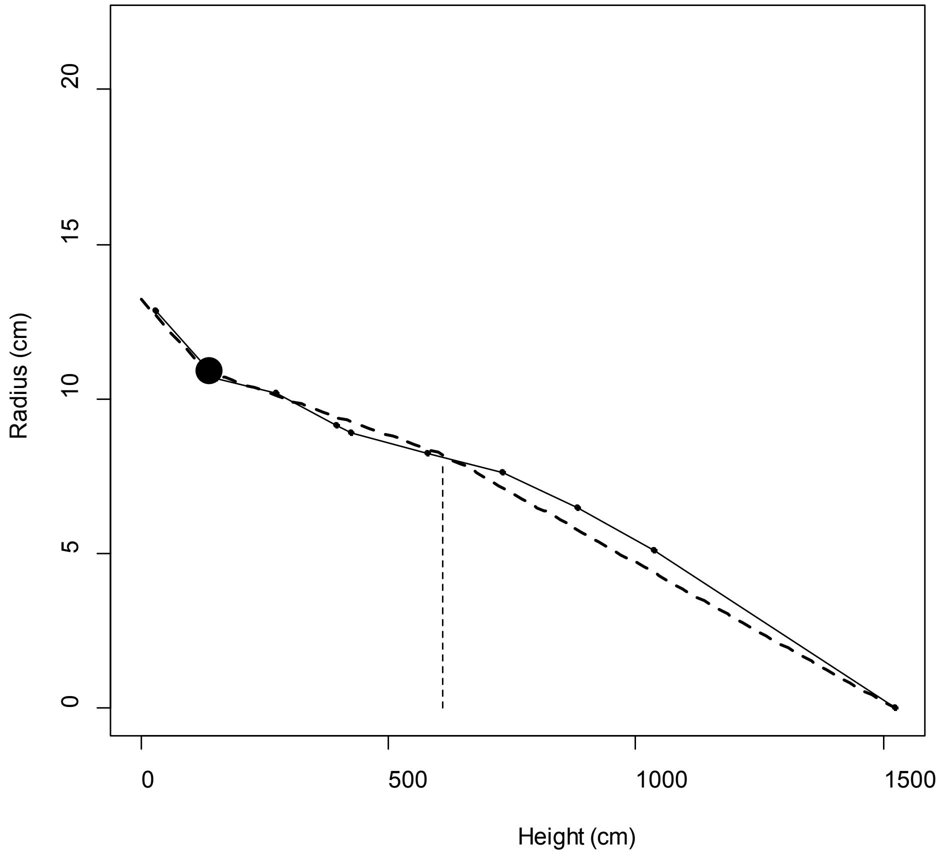

Figure 7.—Illustration of simple taper applied to a shortleaf pine tree. The solid thin line and points are the observed data; the thicker dashed line is the simple taper prediction. The large dot at breast height is the observed d.b.h. and the thin dashed vertical line indicated the crown base. All dimensions are in centimeters.

CONCLUSION

The simple taper equations are simple to use, require relatively few tree measurements, and provide very accurate volume predictions. The approach presented here allows managers to collect data on trees in their forest and can be used to develop local taper equations.

ACKNOWLEDGMENTS

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APPENDIX

A Microsoft Visual Basic function that can be imported into Excel to calculate simple taper

```
Function simpleTaper(h As Double, dbh As Double, ht As Double,
htcb As Double, stumpR As Double, stemR As Double, bh As Double)
As Double
 ' Function to calculate a simple taper equation
 ' Copyright by David R. Larsen, August 14, 2012
 '
 diameterCrownBase = dbh + stemR * (htcb - bh)
 crownLength = ht - htcb
 topRate = diameterCrownBase / crownLength

 simpleTaper = 0#

 If (h < bh) Then
 simpleTaper = dbh + stumpR * (h - bh)
 ElseIf ((h >= bh) And (h < htcb)) Then
 simpleTaper = dbh + stemR * (h - bh)
 Else
 simpleTaper = (ht - h) * topRate
 End If

End Function
```

LONG-TERM RESEARCH

AFTER 25 YEARS, WHAT DOES THE PENNSYLVANIA REGENERATION STUDY TELL US ABOUT OAK/HICKORY FORESTS UNDER STRESS?

William H. McWilliams, James A. Westfall, Patrick H. Brose, Shawn L. Lehman, Randall S. Morin, Todd E. Ristau, Alejandro A. Royo, and Susan L. Stout[1]

Abstract.—The Pennsylvania Regeneration Study was initiated in 1989 because of concerns about a long history of stress on oak/hickory (*Quercus/Carya*) forests from herbivory and other factors. The study, which addresses the need for landscape-level information about regeneration quality and abundance, comprises a suite of regeneration indicator measurements installed on a subset of Forest Inventory and Analysis monitoring plots. The State's oak/hickory forests have been under increasing stress because aging stands that originate from large-scale disturbances from more than 100 years ago are inundated by herbivory of preferred plants and invasion of native and nonnative invasive plants. Maintaining oaks in young stands is difficult because of herbivory, invasive plants, climate change, lack of fire, and other factors. This paper summarizes the Pennsylvania Regeneration Study results, offering a look at likely challenges faced by managers and policy makers, as well as by inventory specialists who design forest inventories for stressed forests.

INTRODUCTION

Pennsylvania is well known for its oak/hickory (*Quercus/Carya*) forest, which accounts for more than half the State's forest land, or 9.1 million acres. Oaks deliver more income from timber products than any other genus and are by far the most important source of mast for wildlife. A lack of major disturbances such as wildfire has led to conditions in which a dearth of available light limits the establishment of young oak seedlings. Competing vegetation is a critical factor for oak seedling establishment during the stand initiation phase because of competition for light and growing space (Jackson and Finley 2011). Concurrent with the lack of disturbance, white-tailed deer (*Odocoileus virginianus*) have emerged as the keystone herbivore that drives understory composition and structure (Horsley et al. 2003, Waller and Alverson 1997). Dey (2014) paraphrases these conditions as "not enough light and too many deer." This paper has two goals: (1) to describe how the results of a landscape-level study of regeneration address the need for inventorying and monitoring the condition and health of forest regeneration; and (2) to discuss lessons learned about monitoring, management, and policy that are pertinent to the oak/hickory forests of Pennsylvania and other regions.

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Pennsylvania Regeneration Study

In preparation for the Northern Research Station, Forest Inventory and Analysis (NRS-FIA) 1989 inventory of Pennsylvania's forest land, the Pennsylvania Bureau of Forestry convened a summit of environmental organizations to assist in guiding research toward issues that are pertinent to oak/hickory forests. It became clear that the overriding issue was the impact that deer were having on advance regeneration, and the perception was that favorable regeneration was lacking over large areas. As a result, a set of regeneration measurement protocols based on the silvicultural findings published by Marquis (1994) were implemented on all NRS-FIA plots measured during the leaf-on season. This marked the beginning of the Pennsylvania Regeneration Study (PRS).

By 1995, the NRS-FIA findings demonstrated that more than half the samples in stands that had adequate light for regeneration did not have adequate stocking of tree seedlings or saplings, a condition described as "impoverished" (McWilliams et al. 1995). In 2000, the NRS-FIA program changed from periodic inventories that were conducted about every 10 years to an annual approach in which some sample plots are measured each year. A pilot study was conducted to adapt the regeneration measurements from the periodic inventory to the new national plot design (McWilliams et al. 2001). In 2001, the new plot design was adopted as a standard feature of the inventory (McWilliams et al. 2015). Lessons learned from the PRS eventually led to the implementation of these regeneration indicator measurements across all 24 states served by the NRS-FIA beginning in 2012.

METHODS

Sample Design

The national FIA program's three-phase inventory and monitoring system was the source of field sample data for the study region (Bechtold and Patterson 2005). In Phase 1, classified remote sensing imagery was used to prestratify sample plots, allowing stratified estimation to reduce uncertainty (variance) for estimates of population totals. Phase 2 consisted of field samples at an intensity of roughly one plot per 6,000 acres. The sample of adult trees and other forest vegetation comprised a cluster of four 24-foot fixed-radius subplots. Each subplot contained a 6.8-foot radius microplot offset from the subplot center to tally saplings and seedlings. The standard Phase 2 inventory provided data and information at the plot, subplot, microplot, and tree levels using a mapped design that considers different site conditions to reduce bias and improve classification. The Phase 3 sample consisted of a suite of forest health indicators that were measured during the leaf-on summer season on a subset of plots. The PRS is a component of the Phase 3 sample with a sample size that fluctuates from year to year because it is a subset of the total number of forested Phase 2 annual inventory plots. It usually, though, consists of about 400 samples.

Primary Regeneration Indicator Measurements

Browse Impact

A browse impact code was used to indicate the pressure that herbivores are exerting on tree seedlings and other understory flora for the area surrounding the sample plot. Browse is defined as "the consumption of tender shoots, twigs, and leaves of trees and shrubs used by animals for food." This approach is distinguished from other browse studies that assess the amount of material removed from specific plants (Latham et al. 2005). The reason for assessing overall impact is that it is difficult to identify browse if no seedlings or other vegetation are present to be browsed (Latham et al. 2009). If the site has no understory vegetation, it can still be difficult to estimate browse impact. The following five codes were used:

1. Very low: plot is inside a well-maintained exclosure.
2. Low: no browsing is observed or vigorous seedlings are present (no exclosure is present).
3. Medium: browsing evidence is observed but uncommon; seedlings are common.
4. High: browsing evidence is common.
5. Very high: browsing evidence is omnipresent or a severe browse line exists.

Tree Seedling Counts

Seedling height is an important factor for estimating survival likelihood for naturally and artificially established seedlings, especially in areas with competing vegetation and herbivory (Brose et al. 2008, Sander 1972). Tree seedling counts are needed to generate estimates of the number of seedlings per unit area for populations of interest and to evaluate the adequacy of advance regeneration. Tree seedlings were counted on forested microplot conditions by species, source, and height class. All seedlings that have survived for 1 year and are at least 2 inches tall were included. In the case of large-seeded species, such as oak, hickory, and walnut (*Juglans* spp.), the level of establishment was determined by evaluating root collar diameter (RCD). An RCD of at least 0.25 inch defined a stem that is established. Seedlings with an RCD of at least 0.75 inch and a diameter at breast height smaller than 1 inch were coded as competitive. Large-seeded species at least 3 feet tall were also included as competitive.

Spatial Analysis

To create continuous maps of browse impact, indicator kriging (Isaaks and Srivastava 1989) within Esri's ArcMap 10.2.2 geostatistical analyst package (Esri, Inc., Redlands, CA) was used to create an interpolated surface from the sample data. Indicator kriging, a geostatistical technique that models autocorrelations between observations as a function of distance, yielded a visualization of the probability of exceeding the low browse impact class.

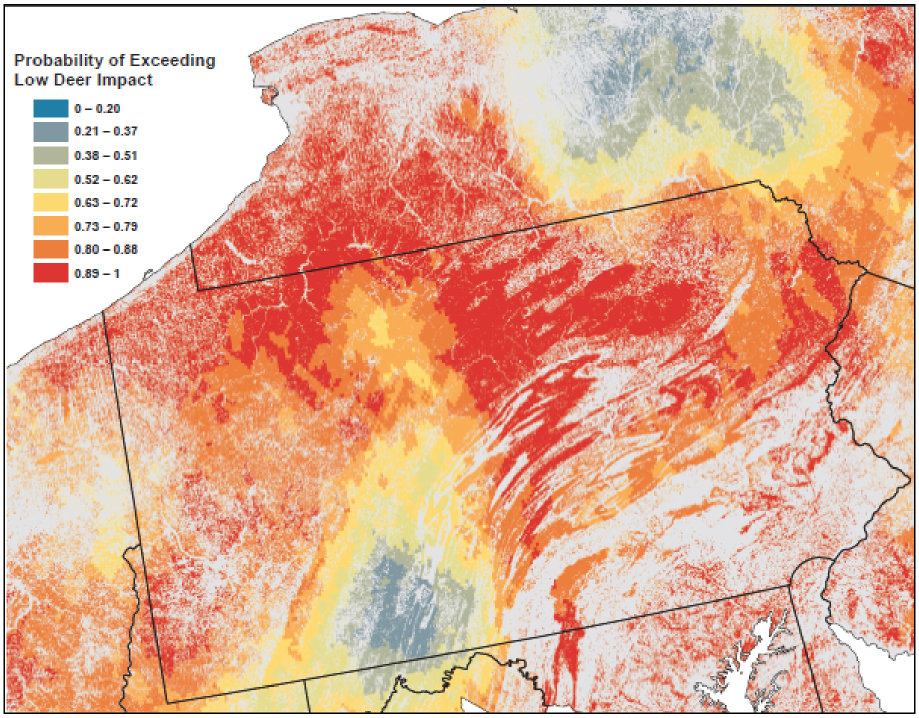

Figure 1.—Map showing the probability of exceeding low ungulate browse impact, throughout Pennsylvania, 2014.

RESULTS

PRS data and results have been used for a variety of assessments since the study began. We describe four representative case studies to highlight applications so far. Topics covered are herbivory impacts, overstory and understory composition and structure, regeneration adequacy, and habitat trends.

Herbivory Impacts

Sources of information that show the spatial distribution of herbivory impact are almost completely lacking. Sources have used population-based approaches that have proven difficult to develop and maintain over time. Some of the major challenges are estimating deer density, developing useful models of population based on stand conditions, and tracking deer movement. In addition, the lack of empirical data means that estimates of sampling error are not available.

Results from the PRS estimate the percentage of samples by impact level as 19 percent very low/low, 60 percent medium, and 21 percent high/very high. A geographic visualization was constructed to represent variations in occurrence based on the probability of exceeding low impact, which is the level where regeneration management options should be considered (Fig. 1). Areas of Pennsylvania with higher probability are apparent in the northwestern, northeastern, and southeastern regions. Areas with lower probabilities are in the mountains of the south-central region.

Overstory and Understory Composition and Structure

Pennsylvania's oak/hickory forests are maturing and will need to be able to regenerate and retain competitive tree seedlings and saplings in the stand-initiation phase of development. The relationship between overstory and understory tree composition and structure gives us a

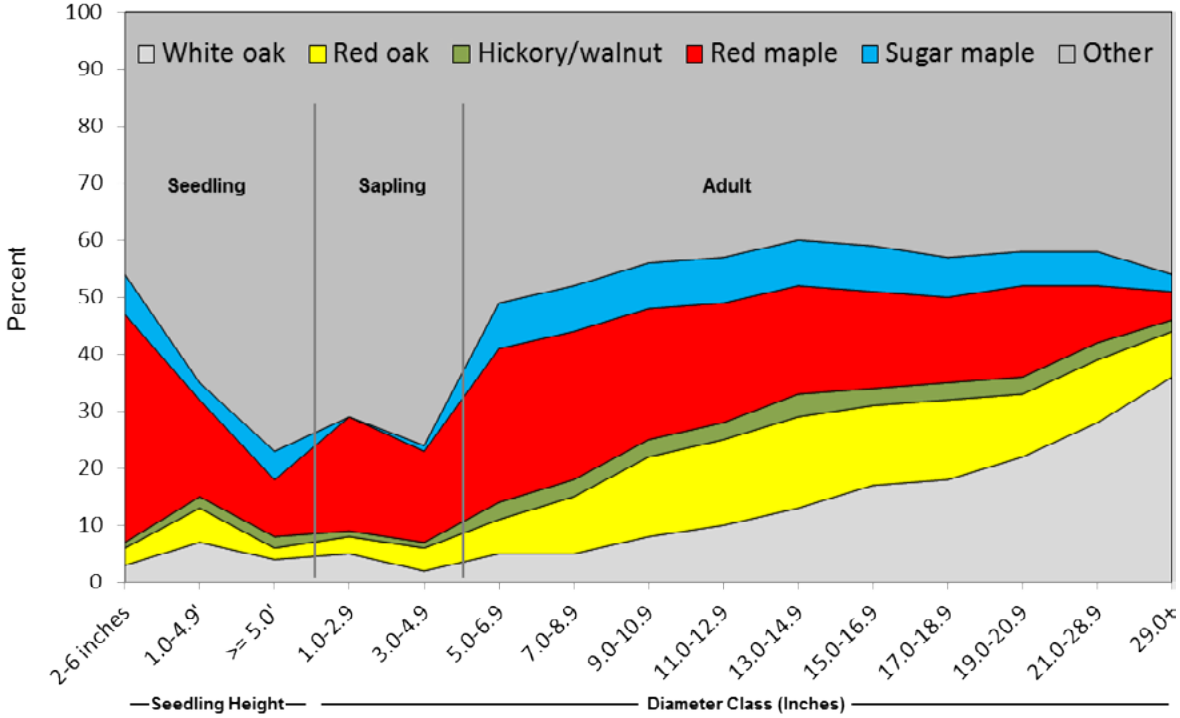

Figure 2.—Proportion of the number of seedlings, saplings, and dominant/codominant growing-stock trees (Adult) on forest land for select species or species group by size. Seedling estimates are for 2012–2014. Sapling and tree estimates are for 2010–2014. Black walnut is included in hickory species.

basis for understanding future forest conditions. Adding PRS seedling height results to stand profiles for larger trees extends the region of inference for regeneration to newly established and older seedlings (Fig. 2). Relative differences between the percentage of stems for seedlings and saplings versus adults yield information about prospective winners and losers in landscape-level stand development that in turn imply likely shifts in future composition and structure. The most prominent seedling and sapling winner that has the potential to reach high canopy was red maple (*Acer rubrum*). Other possible winners not shown are American beech (*Fagus grandifolia*) and birch (*Betula* spp.). High-canopy taxa that are under-represented as seedlings and saplings are white oak (*Q. alba*), red oak (*Q. rubra*), hickory/walnut (*Carya/Juglans*), and sugar maple (*A. saccharum*). Some of the more common understory and midcanopy taxa are serviceberry (*Amelanchier* spp.), striped maple (*A. pensylvanicum*), sassafras (*Sassafras albidum*), blackgum (*Nyssa silvatica*), hawthorn (*Crataegus* spp.), and eastern hophornbeam (*Ostrya virginiana*).

Regeneration Adequacy

Regeneration adequacy metrics developed for Pennsylvania's forest-type groups were used to compare the number of seedlings in the FIA sample by height class to published regeneration guidelines (McWilliams et al. 2015). Before applying these metrics, we screened samples for favorable light conditions for seedling establishment and development by limiting the sample to plots from 40 percent to 75 percent stocked with all-live trees based on the findings of Marquis (1994). The samples were deemed adequate if the regeneration sample exceeded the regeneration guidelines.

Oak/hickory samples with fewer than 70 percent of the microplots adequately stocked with regeneration represent conditions that may need some form of treatment to increase oak stocking. The current results show that more than 60 percent of the samples had fewer than 70

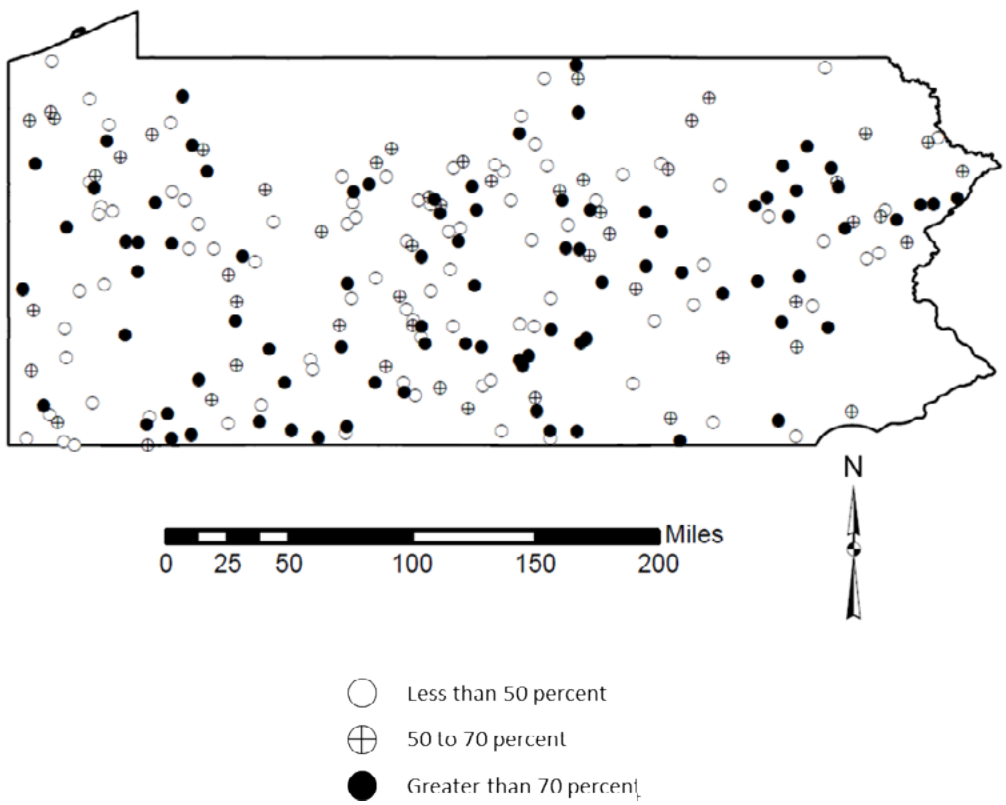

Figure 3.—Distribution of samples on forest land (in the mixed oak forest-type group with live tree stocking levels of 40 to 75 percent) by percentage of microplots with adequate advance regeneration of canopy replacement species, Pennsylvania, 2009–2013.

percent of the microplots with adequate advance regeneration. The geographic findings do not indicate a particular pattern (Fig. 3). The metrics used here include species other than oak that have the potential to achieve high canopy. This means that some of the samples with adequate stocking of advance regeneration may not have restocked with oaks, but rather with other species (e.g., red maple regeneration that commonly occurs under an oak canopy).

Habitat Trends

Balancing the need for healthy regeneration and wildlife habitat in new forests requires careful monitoring. The last case study covers the use of PRS data as part of a large-scale wildlife management planning process. The Pennsylvania Game Commission uses PRS information to help evaluate deer habitat health for determining the number of deer to harvest for Pennsylvania's wildlife management units (WMUs) shown in Fig. 4 (Rosenberry et al. 2009). A pivotal question for long-term planning has been: Are habitat conditions improving over time?

The PRS approach described in the previous section has been used to estimate the percentage of samples classified as oak/hickory with adequate stocking of species that can achieve a high canopy position. For contrast, this case study also evaluates composition for commercially desirable species. The basic difference is that the commercially desirable species list excludes beech and birch that are considered high canopy species.

Estimates of the percentage of samples classified as adequate over time for two WMUs in the oak/hickory region that have a history of high browse pressure illustrate habitat trends. WMU 2G is in the north-central region and has shown statistically significant improvement for high canopy species since 2005 (Fig. 4). In 2005, 36 percent of the 2G samples were adequately stocked with regeneration of high canopy species compared to 41 percent in 2013. Commercial species showed a significant decrease from 35 percent to 30 percent since 2005. WMU 4D lies to the south of 2G and is characterized by a higher concentration of oak/hickory compared to maple/beech/birch (*Acer/Betula/Fagus*), which is more common in WMU 2G. The trends show improvement for both high canopy and commercial species between 2005 and 2013. This case study uses a subset of the overall sample population yielding higher sampling errors that can limit the number of inferences that can be drawn. This means that when error rates are high, improvements in regeneration may remain undetected (Westfall and McWilliams 2012).

DISCUSSION

Twenty-five years of experience in monitoring tree regeneration in Pennsylvania offers guidance through lessons learned about monitoring, management, and policy. The tenet that all forest management activity has reciprocal impacts on wildlife habitat and other ecosystem services underlies forest management planning in Pennsylvania. This means that inventory specialists, managers, and policy makers must adjust goals and objectives to implement practices and prescriptions that balance the need for regeneration with healthy habitat for deer and other wildlife. Implications for broad-scale forest management of oak/hickory forests under stress begin with understory development before and after a stand-initiating disturbance. Currently only 4 percent of Pennsylvania's oak/hickory forest is classified as 20 years or younger. This dearth of young stands is a product of traditional use of partial harvests and passive management on private forest land. Regenerating oak/hickory forests requires both the development of competitive oak advance regeneration before harvest and near-complete overstory removal to allow for light. For most private oak/hickory forest in need of replacement, neither strategy is common. Private owners control 7 of every 10 acres of Pennsylvania's oak/hickory forest.

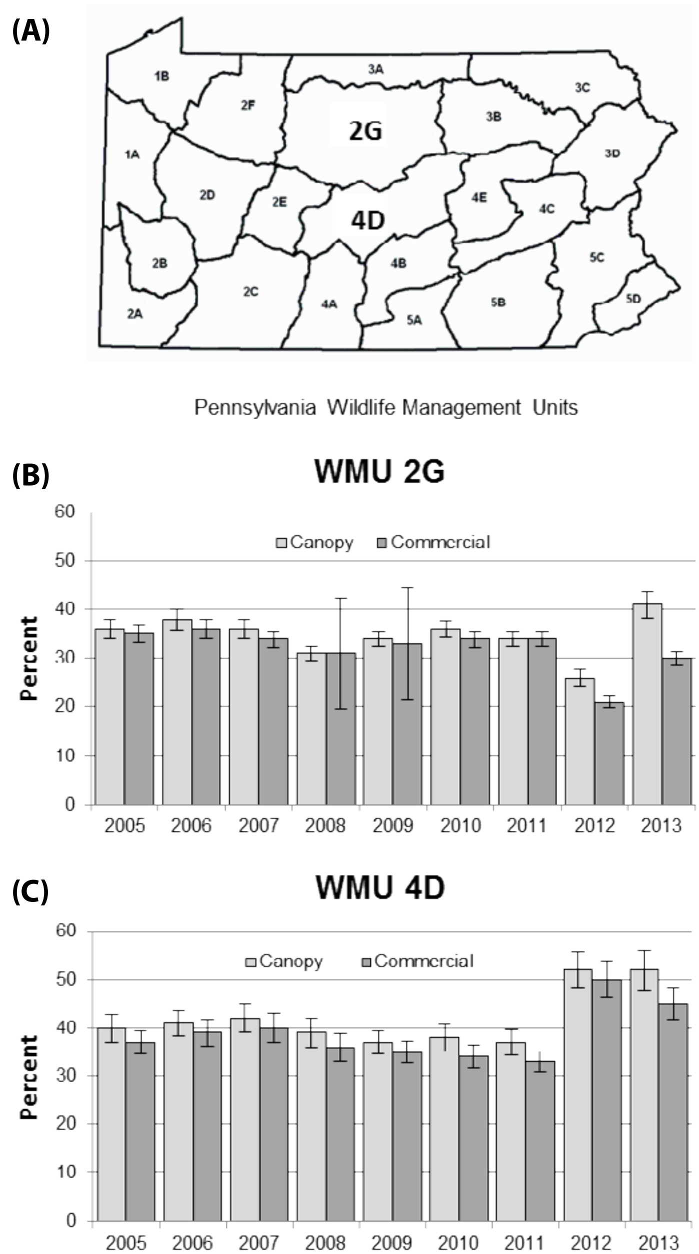

Figure 4.—Pennsylvania WMU boundaries (A) and percentage of samples adequately stocked with advance tree seedling and sapling regeneration for canopy replacement species for WMU 2G (B), and WMU 4D (C), 2005–2013. Sampling error bars are shown for the 68-percent confidence level.

Monitoring

Some basic techniques for regeneration inventory and monitoring have been delivered and vetted in the literature (McWilliams et al. 2015):

- The PRS sample facilitates useful statistical inference for baseline and trend analysis of herbivory levels and seedling attributes.
- Regeneration indicator data support a wide range of regeneration evaluation metrics to address taxa-level research questions.
- Preliminary work assessing regeneration adequacy for oak/hickory of the mid-Atlantic region shows promise for modifying the approach for use in other areas of the NRS region.
- Including regeneration status in FIA reports fills a critical gap in understanding the formation and development of young forests under stress from multiple interacting stressors.
- The regeneration indicator database satisfies the need for publicly available information about tree seedlings and herbivory impacts for subcontinental-scale studies.
- Incorporating regeneration indicator data will improve the interaction of ecological process models and other models with seedling recruitment modules.

Management

Another tenet is that regeneration potential is set during the understory reinitiation and stand initiation phases of early stand development (Dey 2014, Oliver and Larson 1996). Management options for oak/hickory forests under stress from herbivory and other factors will require a mix of silvicultural prescriptions. Managers will need to consider current methods and keep abreast of new research to make fully informed decisions. Most of the lessons learned for managing forests under stress are not new, but in Pennsylvania the full suite of stressors and their complex interactions must be considered. Some prominent examples follow:

- Prescriptions that establish, recruit, and retain competitive oak seedlings from the mature stand before harvest through the stem exclusion phase after harvest will need to pay particular attention to contemporary drivers and stressors (e.g., invasive plants and climate change).
- Herbivory by white-tailed deer needs to be considered before harvest in oak/hickory stands scheduled for stand initiation in medium-pressure and high-pressure subregions.
- The most successful technique for regenerating oaks stands is the shelterwood system and related management prescriptions described by Brose et al. (2008).
- Fencing is often needed to reduce deer impacts and facilitate application of herbicides and other controls.
- Controlling composition and structure of understory vegetation in stands with entrenched legacy vegetation as described by Royo et al. (2010) typically requires prescriptions that are often too costly for private forest landowners.
- Developing adequate stocking of competitive oak and associate species requires vigilant stewardship through the sapling stage in many cases.
- Information evaluated so far indicates a slow restorative process for understories with sparse regeneration.

Policy

Pennsylvania policy makers, planners, and legislators have used PRS results for guiding forest and wildlife management projects (Frye 2006). Prominent activities that may also be relevant for other areas of the mid-Atlantic region are as follows:

- Inform the Pennsylvania Bureau of Forestry planning efforts for state-owned forest land and support strategies for state-level private forest stewardship.
- Inform other state agencies and nongovernmental organizations grappling with contemporary forest issues such as forest health.
- Inform the forestry community by supplying content for 5-year state forest resource reports, technical articles, annual bulletins, and other outlets.
- Help the Pennsylvania Game Commission continue to improve the methods for deriving habitat health estimates for deer and other wildlife.
- Develop and distribute guidance for budgetary decisions on allocating funds among forest research projects.

CONCLUSIONS

The connections between humans, wildlife, forest health, and forest regeneration are important for sound forest management and policy formulation. Cumulative effects of these and other factors make for complicated policy and management decisions. As pointed out by McShea and Healy (2002), plans for maintaining the oak component are needed to slow the potential loss of oak trees over the next 100 years.

Oak regeneration has become a major forest-policy issue, even though excellent guides are available for regenerating oak (Brose et al. 2008, Johnson et al. 2009, Steiner et al. 2008). The heart of the issue is that the conditions for regenerating oaks and associate species are not favorable, forcing very high costs for management activities such as fencing, herbicide application, and weeding, to control competing vegetation, available light, and deer, (Jackson and Finley 2011). The cost of investing in future forest benefits is challenging for the diverse mix of private forest landowners who control most of the oak/hickory forest.

The PRS has confirmed that deer and the widespread application of partial harvest are drivers and barriers to young oak/hickory forest formation across Pennsylvania. Over the 25-year history of this study, the balance between regeneration and available browse has shifted in a positive direction, primarily because of reductions in the deer herd. A rough measure for the balance or imbalance may be the percent of forest by broad browse impact level. About 20 percent of the forest is estimated to have very low/low impact, or a positive balance. An estimate of 60 percent for medium impact means that large areas of oak/hickory forest will need management that considers browse impact along with local conditions. The estimate for high/very high impact is roughly 20 percent, which implies conditions that are out of balance. The PRS adds utility for monitoring, managing, and developing policy strategies for improving the health and resiliency of Pennsylvania's oak/hickory forest.

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POSTER PRESENTATIONS

MAXIMUM CROWN AREA EQUATION FOR OPEN-GROWN BUR OAK

M.C. Demchik, S.M. Virden, Z.L. Buchanan, and A.M. Johnson[1]

Abstract.—Bur oak (*Quercus macrocarpa* Michx.) is a classic savanna species with a range that covers much of the eastern United States. Because savannas are an endangered habitat in North America, significant restoration efforts are in progress across much of the range of bur oak. For open sites being planted with bur oaks as well as fully stocked sites that are being thinned to create savanna conditions, an understanding of potential maximum crown area is of great utility. Between 2013 and 2015, 45 open-grown bur oaks ranging in size from 3.6 to 36.1 inches diameter at breast height (d.b.h.) were measured across central and southern Wisconsin. The equation $Crown\ Area = 3.253DBH^{2.004}$ was used to predict crown area with an r^2 of 0.93 ($p < 0.001$). This strong relationship has substantial utility for modeling crown cover to guide management of savanna restoration sites.

INTRODUCTION

Savanna is one of the most endangered habitats in North America. The majority of oak savannas have either grown into closed canopy conditions (creating two-cohort forests) or been converted into agricultural land. Research from Nuzzo (1986) suggests that current savanna remnants represent only 0.02 percent of former cover. With the loss of open conditions, plant species diversity changes dramatically (Bray 1958). Wilcox et al. (2005) determined that 40 percent crown cover appeared to result in the highest level of species diversity. In addition to crown cover, other factors, most notably fire (Dorney and Dorney 1989, Haney et al. 2008), impact species diversity; however, during savanna restoration, crown cover is one of the easiest factors to manipulate using logging. Nielsen et al. (2003) demonstrated that restoring both the structure and processes for savannas (logging and added fire) resulted in a better response than restoring processes only (solely reintroducing fire). While restoration of fire also tends to reduce crown cover (Bowles et al. 2011), fire alone does not readily regulate canopy cover at levels desired for savanna restoration or maintenance.

Because logging is a valuable tool for managing savanna structure, tools to aid timber marking crews are essential. For this reason, Law et al. (1994) developed a savanna crown cover chart. This crown cover chart follows a similar approach to stocking charts piloted by Gingrich (1967) using crown competition factors based on tree area equations developed by Krajicek et al. (1961). Ek (as cited in Johnson et al. 2002) developed an equation for bur oak (*Quercus macrocarpa* Michx.) in Wisconsin but did not apply it to savanna restoration. However, crown competition factors that use tree diameter to estimate corresponding crown diameter and area of open-grown trees do not consider tree height, only spread of the crowns. Because the majority of savanna restoration will be occurring in either existing closed canopy forest cover (where the crowns have been influenced by competition for years) or in completely open conditions (in the case of conversion of past agricultural land to conservation covers), the impact of tree height may also be important. Additionally, the crown cover chart from Law et al. (1994) is focused at a stand level and does not include bur oak, a common savanna tree in the Upper Midwest; however, a tool which allows a forester to specifically consider individual trees would be of use in both newly

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planted savannas (targeting future stand conditions) as well as overstory reductions (in the case of restorations in existing woodlands and forests).

Inclusion of tree height in the modeling of crown cover can improve the accuracy of the predictions. Many studies have shown that height and crown geometry are quite important when determining the impact of wind on diameter growth (Meng et al. 2008, Sellier and Fourcaud 2009, Sharma and Parton 2007). For this reason, Seymour and Smith (1987) included height in their creation of a new stocking guide for white pine (*Pinus strobus* L.), which reduced the b-line (the minimum level at which full stocking can be achieved).

The objective of this study was to develop an equation for maximum crown area for bur oak that can be used by managers in both marking existing forests for savanna restorations as well as in decisions on planting densities for savanna restorations in former agricultural land.

METHODS

Between 2013 and 2015, 45 open-grown bur oaks ranging in size from 3.6 to 36.1 inches diameter at breast height (d.b.h.) were measured across central and southern Wisconsin. For each tree, d.b.h., crown radius (in the four cardinal directions), and total height were measured. Using the approach of Krajicek et al. (1961), a quadratic regression equation was used to predict measured crown area from tree diameter:

$$MCA = b_0 + b_1D + b_2D^2 \qquad (1)$$

Where

MCA = maximum crown area (square feet), and

D = d.b.h. (inches).

Additionally, to incorporate height (using the approach of Seymour and Smith 1987), the following equation was fitted using Minitab 10.5.1.1.0^{TM} (Minitab, Inc., State College, PA):

$$lnCA = b_0 + b_1 lnD = + b_2HT \qquad (2)$$

Where

CA = crown area (square feet), and

HT = total tree height (feet).

RESULTS

In the quadratic regression (Eq. 1), D was not significant ($p = 0.498$); however, D^2 was significant ($p < 0.001$). The following equation predicted crown area with an r^2 of 0.84 ($p < 0.001$):

$$CA = -54.80 + 3.539(D^2) \qquad (3)$$

When modeled using Eq. 2, no improved prediction was found by incorporating height into the model ($p = 0.415$ for lnHT). With height removed, the following equation with $r^2 = 0.93$ was generated (presented here in its multiplicative form):

$$CA = 3.253D^{2.004} \qquad (4)$$

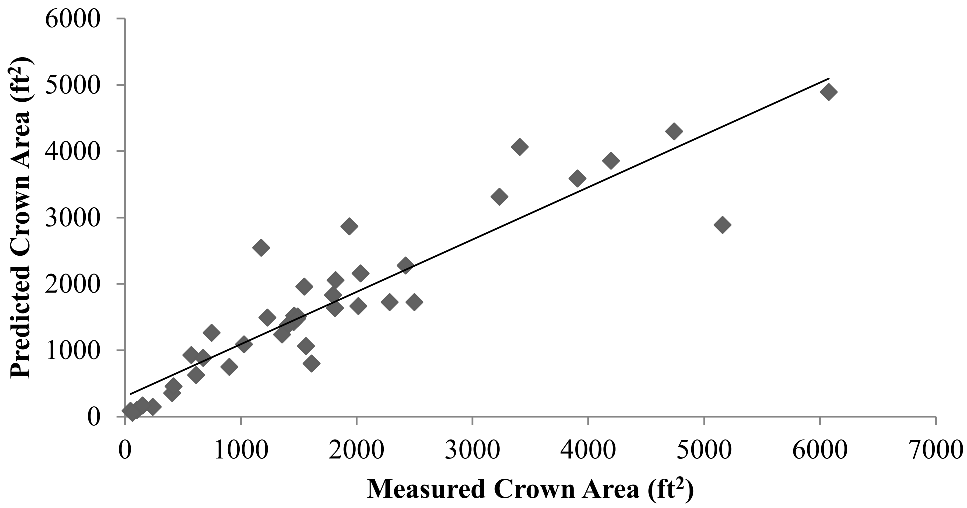

Figure 1.—Predicted crown area versus measured crown area for open-grown bur oaks in Wisconsin.

Table 1.—Number of open-grown bur oak trees of varying diameters needed per acre to meet different crown cover goals

			Crown cover			
D.b.h.	Crown radius	Crown area	100%	50%	25%	10%
inches	*feet*	*square feet*	----------trees/acre----------			
10	10.2	327	133	66	33	13
14	14.3	642	68	34	17	7
18	18.4	1064	41	20	10	4
22	22.5	1590	27	14	7	3
26	26.6	2223	20	10	5	2
30	30.7	2961	15	7	4	1
34	34.8	3805	11	6	3	1

Because the relationship in Equation 4 was extremely strong, this was the crown area model we accepted (Fig. 1). Using this equation, the number of open-grown bur oak trees needed per acre to reach different crown cover goals was calculated using the predicted radius for different diameter bur oak trees (Table 1).

DISCUSSION

It was surprising that height did not improve the prediction within the model because unpublished results by Demchik et al.[2] using height and crown area predicted crop tree d.b.h. very well for trees in a forest condition. The most likely explanation for this difference is that the selected bur oak trees were all open-grown. For this reason, these trees had probably maximized crown and diameter growth without limitations on height growth. The difference that we found between these open-grown trees and the forest trees suggests that using open-grown trees to model the lowest level of full stocking may not be sufficient for modeling maximum crown area

[2] Demchik, M.C.; Conrad, J.; Vokoun, M. Unpublished oak thinning data. Available from M.C. Demchik at: TNR Building, 800 Reserve Street, University of Wisconsin Stevens Point, Stevens Point, WI 54481.

under forest conditions. The use of height as a factor in the prediction of maximum crown area will require additional research.

For those involved in savanna restoration activities, decisions on what level of crown cover to leave will strongly impact the woody and herbaceous species diversity after restoration. When converting fully stocked stands to savanna conditions using logging, some windthrow and mortality should be expected. In ponderosa pine (*Pinus ponderosa* Dougl. Ex Laws.) restoration, Covington et al. (1997) left three times as many trees as they hoped to retain in an effort to allow sufficient trees for wind damage. However, the number of trees that need to be left to maintain target crown cover in spite of wind damage may vary by species. For example, Mujuri and Demchik (2009) found white oak (*Quercus alba* L.) reserve trees (which were growing in similar conditions to newly released savanna trees) had little mortality for 15 years after release, while northern pin oak (*Quercus ellipsoidalis* E. J. Hill) experienced 5.5 percent mortality after release. Additionally, the ability of trees to respond with increased growth as well as increased crown development may also vary by species. Brudvig et al. (2011) found increased growth of remaining trees (white oaks) with minimal mortality of the released trees after encroaching trees were removed to recreate a savanna. Similarly, Mujuri and Demchik (2009) found white oak increased in tree vigor and volume growth after release; however, northern pin oak tended to decline in vigor, although their predicted volume growth rate did increase. Because bur oak is a similarly long-lived tree when compared to white oak, survival after release will likely be similar to the white oaks in the previous two studies, suggesting that fewer trees need to be left for "redundancy" against mortality.

Values in Table 1 can be used to model the number of trees per acre (by diameter) that are needed to meet specific crown cover goals after the trees have expanded their crowns. This table could also be used to model the release of bur oaks in both oak woodland and forest conditions using the crown radius. With increasing interest in oak savanna restoration, tools similar to Table 1 may be of use in making informed management decisions.

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SITEQUAL V2.0—A FORTRAN PROGRAM TO DETERMINE BOTTOMLAND HARDWOOD SITE QUALITY

Don C. Bragg[1]

Abstract.—SITEQUAL is a computerized expert system that uses a number of easily determined soil conditions associated with physical structure, available moisture, available nutrients, and aeration to estimate site index for 14 southern hardwood species. The original program was written in the Basic language by Harrington and Casson (1986) based on the field methods for site evaluation developed by Baker and Broadfoot (1979). Unfortunately, this version of SITEQUAL does not operate as a stand-alone application, but rather requires a compiler that no longer works with modern operating systems. I have reprogrammed SITEQUAL (now version 2.0) in Fortran with a more user-friendly interface and a number of other minor improvements. To demonstrate SITEQUAL2.0's utility, I present two examples of output from bottomland hardwood sites in southeastern Arkansas. Further improvements to SITEQUAL2.0 are being considered, including the development of a graphical user interface and the addition of more species and environmental conditions.

INTRODUCTION

Bottomland hardwoods represent a significant forest resource in the eastern United States, including the Central Hardwoods region. While these low, seasonally flooded forests are generally considered to be some of the most productive, numerous factors influence the productivity of any given bottomland site, resulting in a dramatic range in species performance and possibilities.

In the 1970s, James B. Baker and Walter M. Broadfoot developed an expert system to predict bottomland hardwood site index using conventionally available (or easily derivable) topoedaphic attributes. As a soil scientist, Broadfoot had spent years developing the foundations of this system (Broadfoot 1964, 1969, 1976); Baker later contributed his experience in both soils and silviculture. Their first approximation included eight species (Baker and Broadfoot 1977) followed shortly thereafter by an update that included an additional six species (Baker and Broadfoot 1979). The Baker and Broadfoot system operates under the fundamental assumption that four factors are the primary determinants of hardwood growth performance in bottomlands: soil physical condition, growing season moisture availability, nutrient availability, and aeration (Baker and Broadfoot 1979).

Since the 1979 publication, others have refined certain aspects of this system, including an adaptation for loblolly pine (*Pinus taeda* L.) in southern Arkansas (Guldin et al. 1989) and a computerized version (SITEQUAL). The original SITEQUAL program was written by Harrington and Casson (1986) in the Basic language following the field methods and taxa of Baker and Broadfoot (1979). Unfortunately, because of how SITEQUAL was coded, it did not operate as a stand-alone application, but rather required an old Basic compiler which no longer runs on modern operating systems.

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To address this shortcoming, I have reprogrammed SITEQUAL (now version 2.0) in a simple, user-friendly interface with a number of minor processing and output improvements. To demonstrate SITEQUAL's utility, in this paper I present two examples of output from a range of bottomland hardwood sites in southeastern Arkansas and discuss plans for future improvements to SITEQUAL. These upgrades may include a graphical user interface, the addition of more species, and a broader range of environmental conditions.

METHODS

The Program

SITEQUAL version 2.0 (hereafter referred to as SITEQUAL2.0 to distinguish it from the original of Harrington and Casson) is currently available by request. SITEQUAL2.0 was written as a stand-alone Fortran[2] program that operates in a MS-DOS® shell available in the Windows® operating system (up to at least Windows 7). Since I did not have a working copy of the original SITEQUAL and only have the flow diagram from Harrington and Casson (1986) to work from, SITEQUAL2.0 represents an approximation of their program.

In addition to the new stand-alone interface, the ability to run multiple scenarios without exiting the program has been added to SITEQUAL2.0, as has a limited capacity to make some default adjustments. For example, the user can change the values of species-based input parameters by editing SPP_ATTRIBUTES.CSV in a spreadsheet or text editor. However, caution is advised in making these changes, as the default values were calibrated by the original authors and any departures from these may significantly impact the results (or cause the program to crash).

Program Operation

When executed, SITEQUAL2.0 opens to an introductory screen describing the current version of the program, including a quotation from Harrington and Casson's (1986) user guide (Fig. 1). Once the user has confirmed their intent to run the program, SITEQUAL2.0 reads a species attributes file to upload the default model settings and species parameters. This species attributes file (a comma-delimited ASCII data file called SPP_ATTRIBUTES.CSV, with a fixed data structure) must be located in the same directory as the SITEQUAL2.0 executable file (SITEQUAL.EXE).

In this example, the filename hungerrun.out has been assigned. The next screen in SITEQUAL2.0 continues the data input process (Fig. 2). This stage includes a new feature, the ability to process multiple datasets without restarting the program. Each dataset can be given a specific identifier code (up to 25 characters long) that can be any alphanumeric character or symbol/mathematical operator available on the keyboard (CHERRYBARK_OAK is used in this example). After this dataset is named, SITEQUAL2.0 proceeds through a list of inquiries regarding site conditions. In the current command-line version of this program, the user must choose one of the provided options, with the exception of pH where the actual pH value is entered. Typing something other than the choices provided will invoke an error message, and

[2] SITEQUAL2.0 was developed using Absoft Pro Fortran® v13.0.4, which is a Fortran 95 compiler that fully supports FORTRAN 77 and F2003 and F2008 features. Hence, as written, SITEQUAL2.0 should be portable to other standard versions of Fortran 95, including those Linux-based systems following this standard. However, it has not been compiled in any other environment and has only been run in a Windows 7 DOS® shell to date. Some elements may not be backward-compatible with older versions (for example, FORTRAN 77) of this programming language.

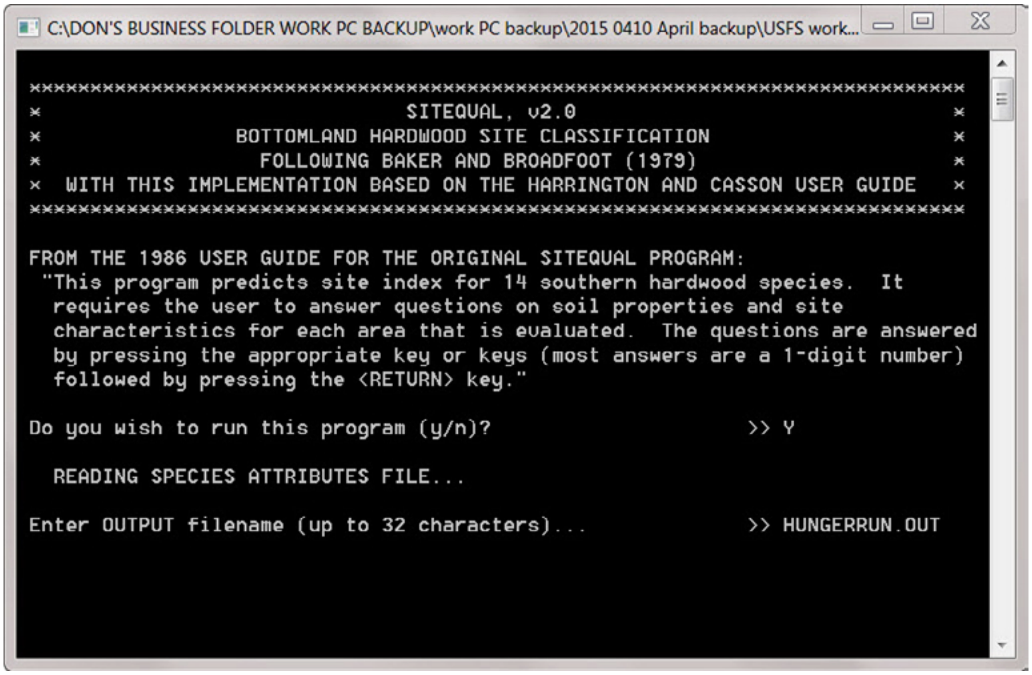

Figure 1.—Introductory screen of SITEQUAL2.0, including the initiating steps (reading the species attribute table from a file) and user entry of the output filename.

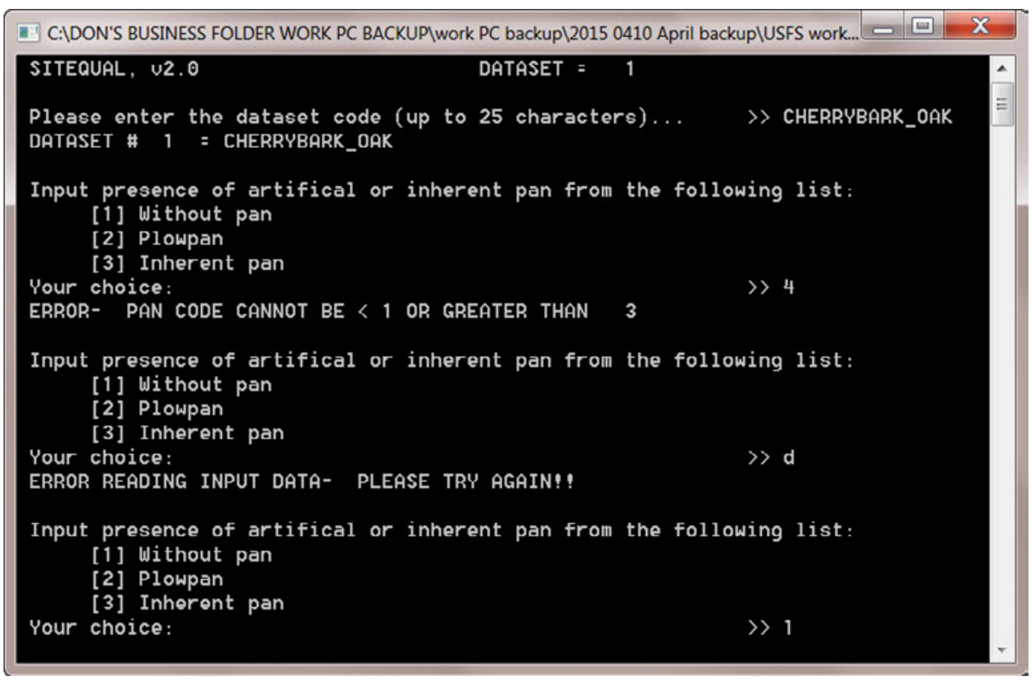

Figure 2.—The second data entry screen of SITEQUAL2.0, with examples of how incorrect data entries prompt the user to make corrections prior to advancing.

the user will be asked to enter an acceptable value. Figure 2 provides a couple of examples of incorrect entries and their resultant error messages. In the first instance, the value "4" has been entered, which is not one of the acceptable options for this particular question. The error message reiterates the acceptable range of values. In the second instance, the character "d" has been entered, which is not an integer at all. In the third instance, the valid entry "1" has been chosen and SITEQUAL2.0 then proceeds to the next question.

The user continues, answering all of the questions that appear until SITEQUAL2.0 has sufficient information to begin processing. After a final confirmation from the user that they are satisfied with their answers, SITEQUAL2.0 determines the site index estimates for all 14 species and writes to the file created by the user at the beginning of this process. The output of SITEQUAL2.0 is in the form of an ASCII text file that can be uploaded into any program (e.g., text editors, word processors, spreadsheets) capable of reading such files.

RESULTS AND DISCUSSION

More detailed examinations of the Baker and Broadfoot system are warranted. While the preliminary analyses of this system have been promising (Aust and Hodges 1988, Belli et al. 1998, Blackmon 1979), consideration of a wider range of species is still needed (Lockhart 2013). Such an evaluation is beyond the scope of this paper, however, and this effort focuses on demonstrating the capabilities of SITEQUAL2.0. The following examples of SITEQUAL2.0 represent output files generated from two different stands in southeastern Arkansas.

Example 1: Hunger Run Creek

Table 1 is an output of the CHERRYBARK_OAK dataset for the hungerrun.out file based on a site along Hunger Run Creek in Drew County (data adapted from Lockhart et al. [1999]). This productive site yielded high site index values for all species; however, some species obviously fared better under the conditions provided. While the aeration-based factors were maximized (100 percent) and greater than 92 percent of the physical condition points were achieved for all 14 species (Table 1), cottonwood (*Populus deltoides*) had less than half of the nutrient availability and not quite three-quarters of the moisture availability possible for that species. Yellow-poplar (*Liriodendron tulipifera*) had the highest value for the Hunger Run Creek location; however, this species is very uncommon in southern Arkansas. Cherrybark oak (*Quercus pagoda*) is considerably more prevalent and was predicted to have a site index of 112 feet in 50 years (Table 1). This value is slightly less than predicted by Lockhart et al. (1999), who gave a value of 114 feet for cherrybark. This modest difference arose because Lockhart et al. (1999) interpolated some of the default values of the Baker and Broadfoot system in the analog, tabular format (a paper datasheet) based on their own experience. SITEQUAL2.0 currently does not allow for that sort of modification.

Example 2: Nuttall Oak

Table 2 gives the output of a run of SITEQUAL2.0 using information adapted from Lockhart (2013). This site near Dermott in Drew County, Arkansas, is located on the Mississippi River Alluvial Plain and consists of an old-field stand that had been originally cleared in the 1970s, farmed for some years, and then replanted under the Wetlands Reserve Program (more details on the specific site can be found in Lockhart [2013]). According to SITEQUAL2.0, Nuttall oak (*Quercus nuttallii*), given the input parameters shown at the top of Table 2, would have a 50-year site index of 81 feet on this site. As noted in the previous example, the Nuttall oak site index in Table 2 is slightly different than the value (80 feet) given by Lockhart (2013). Again, this modest discrepancy arose because Lockhart interpolated between scoring categories as he evaluated the site conditions.

Table 1.—Cherrybark oak data set and output from the hungerrun.out file, based on data from a site along Hunger Run Creek in Drew County, Arkansas (adapted from Lockhart et al. [1999])

SITEQUAL, v2.0--Site evaluation for 14 southern hardwoods
OUTPUT FILE NAME >> hungerrun.out DATE & TIME FINISHED: FEB 18, 2015 13:38:18:964

Input values for each soil site property: DATASET # 1 = cherrybarkoak

Presence of pan:	1	Soil depth:	1
Stratification:	2	Annual fertilization:	0
Soil structure:	1	Soil texture:	2
Compaction:	1	Present cover:	1
Water table depth:	4	Topographic position:	1
Microsite:	3	Flooding times:	2
Geologic source:	3	Organic matter:	1
Topsoil depth:	1	Soil age:	1
pH:	5.25	Swampiness:	1
Mottling:	1	Soil color:	1

Species	Number of points and percentage of total possible by factor				Total site index
	Physical condition	Moisture availability	Nutrient availability	Aeration	
Cottonwood	44 (95.7%)	33 (71.7%)	12 (46.2%)	12 (100.0%)	101
Green ash	20 (95.2%)	34 (72.3%)	17 (65.4%)	10 (100.0%)	81
Hackberry, sugarberry	24 (96.0%)	18 (72.0%)	19 (76.0%)	25 (100.0%)	86
Cherrybark oak	29 (93.5%)	30 (78.9%)	22 (88.0%)	31 (100.0%)	112
Nuttall oak	23 (95.8%)	32 (76.2%)	26 (86.7%)	24 (100.0%)	105
Shumard oak	29 (93.5%)	24 (75.0%)	26 (89.7%)	30 (100.0%)	109
Swamp chestnut oak	26 (92.9%)	27 (87.1%)	21 (87.5%)	26 (100.0%)	100
Water oak, willow oak	27 (93.1%)	24 (70.6%)	21 (91.3%)	29 (100.0%)	101
Pecan	26 (96.3%)	24 (80.0%)	27 (90.0%)	28 (100.0%)	105
Sweetgum	28 (93.3%)	27 (75.0%)	20 (83.3%)	30 (100.0%)	105
Sycamore	30 (93.8%)	11 (55.0%)	29 (74.4%)	39 (100.0%)	109
Yellow-poplar	38 (95.0%)	25 (83.3%)	23 (92.0%)	30 (100.0%)	116

Table 2.—Nuttall oak data set and output from the nuttall.out file, based on data from a site near Dermott, Arkansas (adapted from Lockhart [2013])

SITEQUAL, v2.0--Site evaluation for 14 southern hardwoods
OUTPUT FILE NAME >> nuttall.out DATE & TIME FINISHED: FEB 10, 2015 8:40:41:158

Input values for each soil site property: DATASET # 1 = nuttall

Presence of pan:	1	Soil depth:	1
Stratification:	2	Annual fertilization:	2
Soil structure:	7	Soil texture:	1
Compaction:	2	Present cover:	3
Water table depth:	4	Topographic position:	1
Microsite:	2	Flooding times:	1
Geologic source:	1	Organic matter:	3
Topsoil depth:	2	Soil age:	1
pH:	5.25	Swampiness:	1
Mottling:	3	Soil color:	1

Species	Number of points and percentage of total possible by factor				Total site index
	Physical condition	Moisture availability	Nutrient availability	Aeration	
Cottonwood	31 (67.4%)	34 (73.9%)	9 (34.6%)	6 (50.0%)	80
Green ash	14 (66.7%)	36 (76.6%)	11 (42.3%)	7 (70.0%)	68
Hackberry, sugarberry	15 (60.0%)	18 (72.0%)	13 (52.0%)	18 (72.0%)	64
Cherrybark oak	18 (58.1%)	28 (73.7%)	13 (52.0%)	17 (54.8%)	76
Nuttall oak	15 (62.5%)	33 (78.6%)	15 (50.0%)	18 (75.0%)	81
Shumard oak	18 (58.1%)	22 (68.8%)	17 (58.6%)	17 (56.7%)	74
Swamp chestnut oak	16 (57.1%)	21 (67.7%)	13 (54.2%)	17 (65.4%)	67
Water oak, willow oak	17 (58.6%)	26 (76.5%)	12 (52.2%)	18 (62.1%)	73
Pecan	17 (63.0%)	21 (70.0%)	18 (60.0%)	16 (57.1%)	72
Sweetgum	18 (60.0%)	25 (69.4%)	10 (41.7%)	17 (56.7%)	70
Sycamore	19 (59.4%)	12 (60.0%)	20 (51.3%)	19 (48.7%)	70
Yellow-poplar	23 (57.5%)	22 (73.3%)	13 (52.0%)	15 (50.0%)	73

SUMMARY AND FUTURE PLANS

As currently implemented, SITEQUAL2.0 is a basic tool to evaluate site conditions for a handful of species in a limited geographic region. Previous versions of this program have been used in research and management. For example, the Baker and Broadfoot approach has been shown to be one of the better tools for site index prediction in bottomland hardwood and pine-hardwood sites in Arkansas and Mississippi (Aust and Hodges 1988, Belli et al. 1998, Guldin et al. 1989). More recently, Lockhart (2013) demonstrated the utility of this system for site quality assessments. Although not presented in this paper, it is also possible to use SITEQUAL2.0 to conduct a type of sensitivity analysis for a given site by adjusting one or more factors and examining how the predicted outcomes differ. SITEQUAL2.0's simple design based on an expert system also lends itself to helping students learn about the relationship between site conditions and tree performance.

The addition of more features to SITEQUAL2.0 is being contemplated, including a version that gives the user the option to export more details on how individual species scores were determined. As suggested by both examples presented, allowing the user some flexibility to modify the default scoring system for species is also being contemplated. Although achieving some limited capacity to modify default settings may be possible with only some minor code revisions, this particular change may require a considerably more sophisticated user interface than currently possible. Once sufficiently developed, future versions of SITEQUAL may also be created for portable devices (e.g., smart phones, tablets, or field computers).

The addition of more species and a wider range of site conditions presents a different suite of challenges. There are dozens of other possible species and many other potential site conditions that could be incorporated, if sufficient knowledge behind the relationships between species and site exists. Expanding the number of species and breadth of site conditions may be possible if spatially registered data on individual trees (including accurate species identifications as well as age relationships) can be linked to detailed and compatible site information. For example, it may be feasible to get species and age information for site trees used by the U.S. Forest Service, Forest Inventory and Analysis (FIA) program mapped to sufficiently described soil polygons. Such an effort would require the close cooperation of the FIA program, which carefully guards the exact locations of its sample trees. Likewise, it may also be possible to develop the necessary background data from existing studies or other inventory plots; however, such information may be restricted to a limited geographic area.

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EFFECTS OF LONG-TERM PRESCRIBED BURNING ON TIMBER VALUE IN HARDWOOD FORESTS OF THE MISSOURI OZARKS

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Abstract.—Prescribed fire is commonly used for restoring and managing oak woodlands but raises concern over the risk of value loss to timber products. We used a long-term prescribed burning study to quantify standing timber volume and stumpage value, fire scar presence and size, and timber value loss in comparison to unburned stands. Three study treatments were initiated in 1949: annual burning (Annual; 1-year fire return interval), periodic burning (Periodic; 4-year fire return interval), and no burning (Control). In 2013, we measured the diameter at breast height (d.b.h.) and merchantable height of each overstory tree of sawtimber size (≥ 9.5 inches d.b.h.), from which standing volume and stumpage value were calculated. We measured the dimensions of each fire scar that was present and determined percent value loss based on previously published equations. We found that 4.8 percent of the overstory trees were scarred in the Annual plots compared to 54.8 percent in Periodic plots. At the stand level, percent value loss from fire damage was estimated to be less than 1 percent on Annual plots and less than 3 percent on Periodic plots. However, the stumpage values of Annual plots and Periodic plots were 29.9 percent and 34.3 percent lower than that of the Control plots, respectively, due to lower standing volume and greater prevalence of low-value species (i.e., post oak [*Quercus stellata* Wangenh.]). These results suggest that long-term, frequent prescribed burning affects stand-level timber value primarily through effects on stand structure and composition rather than fire damage.

INTRODUCTION

The restoration of oak woodlands has become an important management objective for many public and private landowners in the Central Hardwoods Forest region (Kabrick et al. 2014). In comparison to forests, woodlands are characterized by lower stocking, higher light levels beneath the canopy, relatively open midstory layers, and greater abundance of herbaceous vegetation in the ground layer (Hanberry et al. 2014a, Nelson 2004). Motivation for woodland restoration is driven, in part, by evidence that woodlands were more common on the landscape historically than they are today (Batek et al. 1999, Hanberry et al. 2014b, Nuzzo 1986), as well as recognition of the high levels of floristic diversity (Knapp et al. 2015, Peterson and Reich 2008) and unique wildlife habitats (Reidy et al. 2014, Starbuck et al. 2015) associated with these ecosystems.

The structure and composition of oak woodlands can be maintained with frequent, low-intensity surface fires (Kabrick et al. 2014), which were an important historical disturbance in the region (Guyette et al. 2002). Frequent fire kills or top kills small-diameter woody vegetation (Dey and Hartman 2005, Hutchinson et al. 2005b, Waldrop et al. 1992) and thus creates open stand structure (Knapp et al. 2015). Greater growing space and light availability at the forest floor and the reduction in litter depth by consumption from burning provide opportunities for herbaceous plants to establish and flourish in frequent-fire ecosystems (Hiers et al. 2007, Veldman et al. 2014). As a result, the richness, diversity, and abundance of herbaceous vegetation have been

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found to increase following prescribed burning in oak woodlands (Hutchinson et al. 2005a, Kinkead et al. 2013).

Although the probability of direct woody stem mortality due to fire decreases as stem diameter increases, prescribed burning can have complex and long-lasting effects on individual trees. Damage to cambial tissue from prescribed burning commonly results in scar formation on the lower portion of tree boles (Guyette and Stambaugh 2004, Shigo 1984), which can lead to fungal infection and the development of decay (Smith and Sutherland 1999). The likelihood of fire damage and subsequent scar formation is variable and is affected by factors such as fire intensity, tree size, and tree species (Dey and Schweitzer 2015, Guyette and Stambaugh 2004). Moreover, fire-induced defects may increase in size through time or following additional fires (McEwan et al. 2007), resulting in potentially negative impacts on timber value (Loomis 1974, Marschall et al. 2014).

Land managers are commonly challenged with balancing multiple objectives, some of which may not be compatible (Bradford and D'Amato 2012). Throughout the eastern United States, uncertainty and concern exist regarding how the use of prescribed fire may affect timber value in hardwood forests. A recent study from the Missouri Ozarks quantified value loss of fire scarred red oaks and reported that much of the damage was located in slab material cut away at the mill, with an average product value loss of 10.3 percent among fire-scarred butt logs (Marschall et al. 2014). However, longer-term prescribed burning (i.e., multiple decades) in forests may also affect timber values by changing the structure (e.g., tree density or standing volume) and composition at the stand level.

A long-term (>60 year) prescribed burning study provides the opportunity to assess effects of fire on timber quality and value. Our specific objectives were to: (1) determine effects of long-term burning at different intervals on standing timber volume; (2) determine effects of long-term burning at different intervals on scar presence and scar size; and (3) estimate timber value loss after more than 60 years of prescribed burning due to stand-level effects (i.e., structure and composition) and individual tree effects (i.e., damage from scarring).

METHODS

This study was established in 1949 at the University Forest Conservation Area in the Ozark Highlands of southeastern Missouri. Soils were moderately well-drained, upland silt loams with a fragipan at moderate depth (Graves 1984) and slopes of 3 to 8 percent in the study area. Since the study was initiated, mean annual temperature has been 57 °F and mean annual precipitation has been 44.5 inches. The study used a randomized complete block design with two blocks (replicates) located approximately 1 mile apart. Both blocks have six study plots, each measuring 131.2 feet × 131.2 feet with a 32.8 foot buffer around all sides. A prescribed fire treatment of unburned control (Control), annual burning (Annual; 1-year fire return interval), or periodic burning (Periodic; 4-year fire return interval) were randomly assigned to plots, resulting in two plots of each treatment per block. All burns were conducted between March and May, and all units that were to be burned in a given year were typically burned on the same day. Fire behavior records kept since 1997 indicate that burns were of low to moderate intensity with generally good consumption of hardwood litter fuels. The most recent burns occurred in Periodic plots in 2012 for one block and 2013 for the other block.

In summer 2013, all overstory trees of sawtimber size (≥ 9.5 inches diameter at breast height [d.b.h.]) were tagged. Species, d.b.h., and merchantable height were recorded for each tree. We determined merchantable height of each tree to the nearest full 4-foot length by measuring the height to the first fork or by calculating the height to an 8-inch diameter inside bark (d.i.b.), assuming a Girard form class of 78. Standing volume was calculated for white oak (*Quercus alba*), post oak (*Q. stellata*), red oaks (*Q. coccinea, Q. falcata, Q. velutina*), hickories (*Carya glabra, C. texana, C. tomentosa*), and for all species together using International ¼-inch log rule. Although there were other species present (e.g., *Cornus florida, Nyssa sylvatica, Ulmus alata*), they did not contribute merchantable volume. We determined standing stumpage value by applying recent Missouri timber prices (Morris and Treiman 2015) (Table 1) to each tree in the study and calculating stumpage value at the stand level. White oak and post oak were analyzed separately because of the difference in stumpage price between the species.

Table 1.—Timber stumpage prices for January-March, 2015, published by Missouri Department of Conservation (Morris and Treiman 2015)

Species group	Stumpage price per board foot
Hickories	\$0.21
Post oak	\$0.13
Red oaks	\$0.29
White oaks	\$0.26
Mixed hardwoods	\$0.22

The presence of external damage was noted for each tree in the burned plots only, and the height, width (at the widest point), and depth of each wound (fire scar) were recorded. We assumed that each wound encountered below breast height (4.5 feet) was due to fire damage. We calculated scar volume as the product of scar height, width, and depth, and the proportion of the circumference of each tree that was scarred (circumference ratio) was calculated as scar width divided by tree circumference at breast height. We calculated the estimated timber value loss to the butt log using equations developed from red oaks by Marschall et al. (2014):

$$PVL = 0.51 + (13.5 * FDI) \quad (1)$$

$$FDI = \frac{(SH * SD)}{TBA} \quad (2)$$

Where

PVL = percent value loss to the butt log,

FDI = Fire damage index,

SH = scar height (inches),

SD = scar depth (inches), and

TBA = tree basal area (inches^2).

We summarized data at the stand level within each block (i.e., the experimental unit was two plots per treatment in each block). We calculated the number of trees per acre, the basal area (square feet per acre), volume (board feet per acre), and stumpage value (\$ per acre) by species group for all trees ≥9.5 inches d.b.h. Variables that described scar size (scar width, scar height, scar depth, scar volume) and scar ratios (circumference ratio, FDI) were calculated at the treatment level using only scarred trees (mean per tree) and using all trees (mean per acre). We applied the calculated PVL to each butt log and then determined the value loss for the entire merchantable stem.

We used mixed-model analysis of variance to test for treatment effects on stand-level response variables, with block included as a random effect. Pair-wise comparisons among treatments were tested using Fisher's protected Least Significant Different (LSD). Because fire scars were only present on Annual and Periodic treatments, we used t-tests to compare scar size, scar ratio variables, and percent value loss. Analyses on scar size and ratios were conducted using only scarred trees (mean per tree) and using all trees (mean per acre). Treatment effects were determined to be statistical significance when $p < 0.05$.

RESULTS

There were no significant treatment effects on the number of trees per acre, basal area per acre, merchantable volume per acre, quadratic mean diameter (QMD), or standing stumpage for any species group or for all trees combined (Table 2), although there were too few hickories and white oaks for statistical analysis of QMD. Although not statistically significant, merchantable volume ranged from less than 6,000 board feet per acre in the Periodic plots to greater than 7,000 board feet per acre in the Control plots. Red oaks contributed the majority of the volume in the Control plots (56 percent standing volume), and post oaks dominated volume in the Annual (62 percent standing volume) and Periodic plots (60 percent standing volume).

Table 2.—Means and one standard error (SE) for number of trees per acre, quadratic mean diameter, basal area, merchantable volume, and stumpage value by treatment for each species group tested, and p-values from ANOVA tests for treatment effects

	Control		Annual		Periodic		
Species group	Mean	SE	Mean	SE	Mean	SE	p-value
			Trees per acre				
Hickories	5.69	5.69	0.63	0.63	0.00	0.00	--[a]
Post oak	33.53	5.69	48.08	6.33	46.82	26.57	0.685
Red oaks	27.84	0.00	17.71	0.00	12.65	11.39	0.420
White oaks	3.16	1.90	3.16	3.16	1.27	1.27	0.809
Total	70.22	1.90	69.59	2.53	60.73	16.45	0.765
			Quadratic mean diameter (inches)				
Hickories	12.44	--	10.43	--	--	--	--
Post oak	13.52	0.01	13.52	0.30	13.98	1.34	0.897
Red oaks	16.65	0.37	14.70	0.83	17.17	0.75	0.218
White oaks	15.10	0.41	16.97	--	16.88	--	--
Total	14.85	0.22	14.00	0.26	14.83	1.95	0.848
			Basal area (square feet per acre)				
Hickories	4.80	4.80	0.38	0.38	0.00	0.00	--
Post oak	33.42	5.73	48.25	8.41	44.97	19.06	0.450
Red oaks	42.12	1.85	20.93	2.36	21.97	20.11	0.491
White oaks	4.06	2.58	4.97	4.97	1.97	1.97	0.832
Total	84.41	0.20	74.52	5.44	68.91	0.92	0.116
			Volume (board feet per acre)				
Hickories	367.30	367.30	25.51	25.51	0.00	0.00	--
Post oak	2539.06	337.99	3984.11	909.07	3510.23	1382.42	0.335
Red oaks	4172.66	233.04	1886.17	299.36	2132.91	1952.62	0.456
White oaks	349.18	203.60	491.12	491.12	178.32	178.32	0.810
Total	7428.21	0.13	6386.91	691.80	5821.46	391.89	0.241
			Stumpage value ($ per acre)				
Hickories	77.13	77.13	5.36	5.36	0.00	0.00	--
Post oak	330.08	43.94	517.23	118.86	456.34	179.75	0.337
Red oaks	1210.06	67.58	546.99	86.81	618.52	566.24	0.456
White oaks	90.79	52.93	127.69	127.69	46.36	46.36	0.810
Total	1708.06	18.54	1197.27	72.62	1121.22	340.14	0.284

[a]-- indicates sample size too small.

Table 3.—Means and one standard error (SE) for percentage of trees scarred and scar width, height, and depth measurements by treatment for the most common species groups, and p-values from t-tests for treatment effects. Sample sizes (n) represent all trees for percentage of trees scarred and only scarred trees for the scar characteristics.

	Annual			Periodic			
Species group	n	Mean	SE	n	Mean	SE	p-value
			Trees scarred (percent)				
Post oak	71	5.83	1.07	71	47.95	22.95	0.207
Red oaks	27	5.00	5.00	20	73.68	26.32	0.122
All trees	104	4.77	0.69	93	54.78	17.64	0.214
			Scar width (inches)				
Post oak	4	4.82	2.26	42	7.35	0.26	0.383
Red oaks	1	4.33	4.33	10	4.23	3.12	0.986
All trees	5	5.84	1.25	54	7.30	0.03	0.451
			Scar height (inches)				
Post oak	4	18.25	9.78	42	10.69	4.39	0.554
Red oaks	1	20.89	20.89	10	8.10	6.21	0.617
All trees	5	20.54	12.07	54	13.08	1.24	0.602
			Scar depth (inches)				
Post oak	4	1.62	1.03	42	1.86	0.11	0.843
Red oaks	1	7.34	7.34	10	0.64	0.37	0.458
All trees	5	3.63	3.04	54	1.48	0.18	0.553

Although the percentage of trees scarred was not significantly different between burn treatments, 54.8 percent of the trees were scarred in Periodic plots and 4.8 percent of the trees were scarred in Annual plots (Table 3). When analyzing only trees with fire scars, scar size measurements were not significantly different between treatments (Table 3). Mean scar volume was high in the Annual plots due to one red oak with a particularly large scar, which resulted in high variability between blocks (Fig. 1). Averaged at the stand level, however, scar size and scar ratios decreased in Annual plots due to the low number of trees that were scarred, resulting in <1.0 percent of total tree circumference being scarred in Annual plots compared to 9.8 percent in Periodic plots.

Percent value loss due to fire damage was low in both treatments. For trees with scars, the butt logs in Annual plots had an average value loss of 9.0 percent compared to 3.2 percent value loss in Periodic plots, although this was not significantly different (Fig. 2). However, at the stand level (analyzing all trees) the value loss was 0.5 percent in the Annual plots compared to 2.0 percent in the Periodic plots. The butt log commonly made up the majority of merchantable volume in the trees on these sites; however, the percent value loss was nearly halved when accounting for the value of the entire merchantable tree rather than only the butt log (Fig. 2).

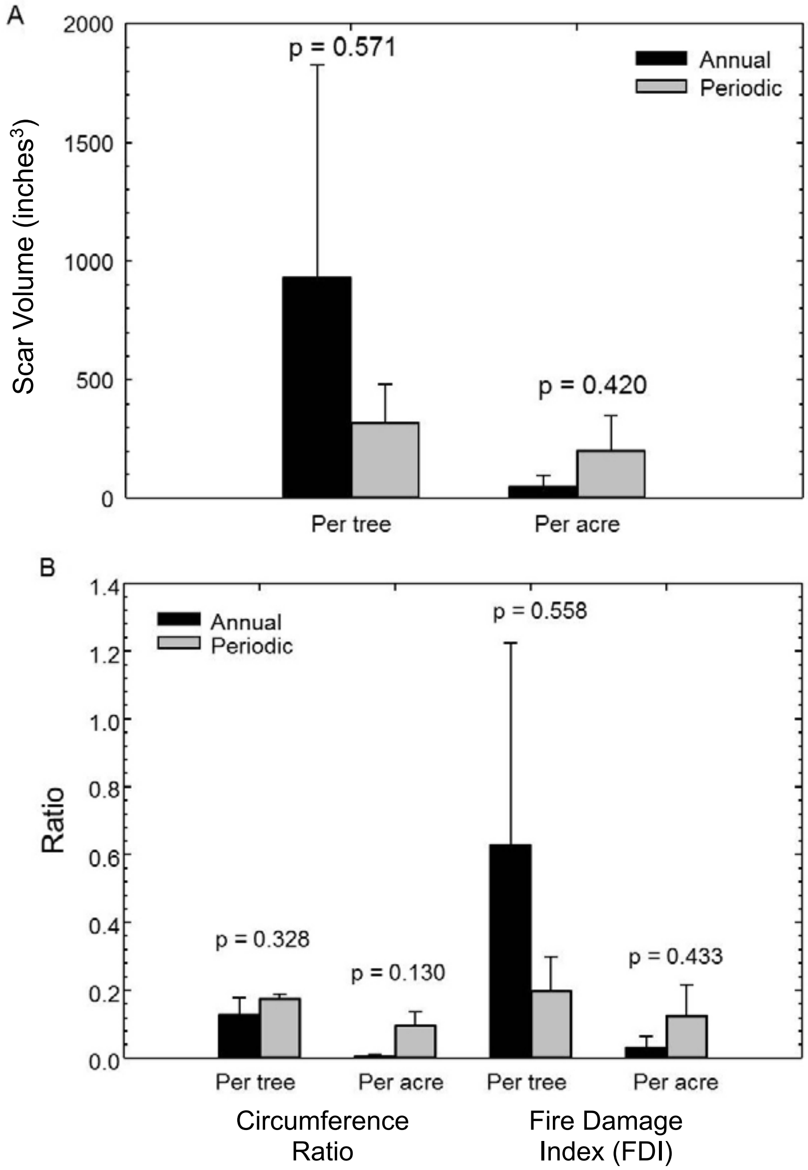

Figure 1.—Means and one standard error (error bars) for (A) scar volume, and (B) the proportion of tree circumference scarred and the Fire damage index. Each variable was calculated at the tree level (using only scarred trees) and at the stand level (using all trees), and p-values are from t-tests comparing burn treatments for each test.

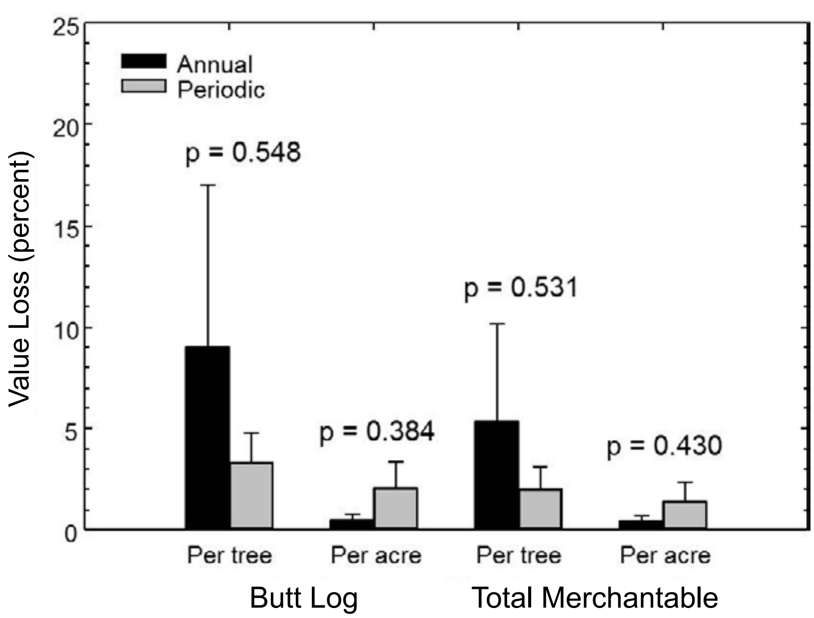

Figure 2.—Means and one standard error (error bars) for percent value loss of the butt log and of the total merchantable volume. Each variable was calculated at the tree level (using only scarred trees) and at the stand level (using all trees), and p-values are from t-tests comparing burn treatments for each test.

DISCUSSION

Our results showed no statistical effects of the burn treatments on stand structure. In a previous publication from this study, Knapp et al. (2015) reported that the burn treatments reduced stand basal area compared to the unburned control when considering trees >4 inches d.b.h. Because the probability of fire-induced mortality decreases as tree size increases (Dey and Hartman 2005), analyzing only sawtimber trees ≥ 9.5 inches d.b.h. in the current study likely reduced the impact of treatment effects. With no evidence of ingrowth of new trees into the canopy of burned plots (data not shown), the apparent shift in composition to post oak dominance on burned plots reflects the lower fire tolerance of mature red oak trees. It is not yet clear how the reduction in small-diameter stems previously reported (Knapp et al. 2015) would affect stand development and the prospect of future timber production with continued fire management.

For trees that were scarred, there was little indication that damage differed between the two burn treatments based on scar dimensions and tree size. However, few trees were scarred in the Annual plots but more than half the trees were scarred in the Periodic plots, resulting in relatively little damage at the stand level in the Annual plots. After the first 8 years of this study, Paulsell (1957) reported that 25 percent of trees on Annual plots and 40 percent of trees on Periodic plots had fire scars. It is likely that fire intensity differed between the treatments. Hardwood litter was the primary fuel type, and we measured greater litter fuel loads in Periodic plots (3.23 tons per acre) than in Annual plots (2.04 tons per acre), which followed patterns commonly observed with time-since-fire in Central Hardwood forests (Stambaugh et al. 2006). The likelihood of damage and the degree of damage from fire are both related to fire intensity (Abbot and Loneragan 1983, Bova and Dickinson 2005, Smith and Sutherland 1999). As a result, the higher fuel loads in Periodic plots likely resulted in greater fire intensity, and consequently more damage, despite lower fire frequency.

Our results indicated that value loss due to damage from prescribed burning was low for both treatments. Marschall et al. (2014) found that fire scar heights <20 inches, similar to those observed in our study, resulted in little value loss to red oak butt logs in the Missouri Ozarks. In contrast, the stumpage value of Annual plots ($1197 per acre) and Periodic plots ($1121 per acre) were 29.9 percent and 34.3 percent less than the unburned Control stumpage value ($1708 per acre) (Fig. 3). This difference was due to the higher volume and the value of red oak ($0.29 per board foot) on Control plots compared to post oak ($0.13 per board foot) on the burned plots.

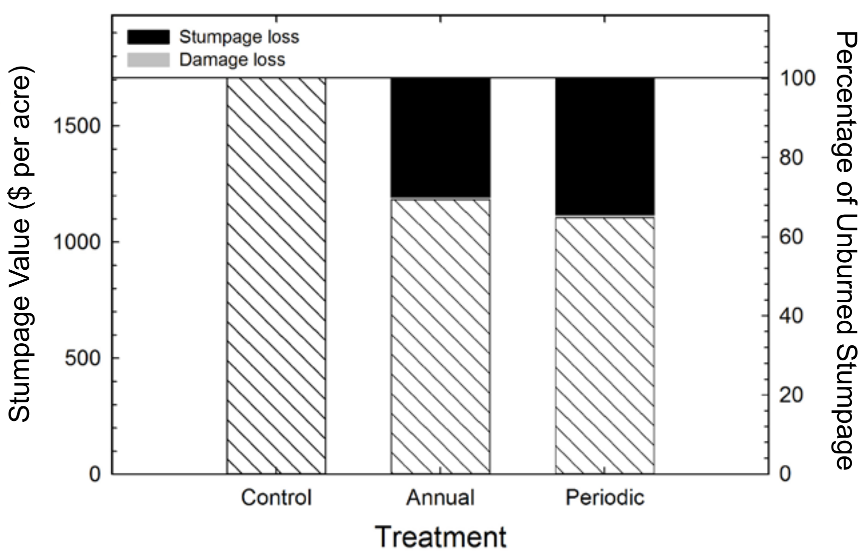

Figure 3.—Stumpage value (dollars per acre on left axis and percentage on right axis) for each treatment (hatched bars). The difference in value between each burned treatment and the control ($1708.06 stumpage value per acre) is attributed to changes in composition/structure (stumpage loss; black bars) and to scarring (damage loss; grey bars).

This study offers unique insight into the long-term effects of repeated prescribed burning on tree damage and potential value loss, despite several shortcomings. For example, statistical inference was constrained by replication in only two blocks. In addition, we applied the equations developed by Marschall et al. (2014) to estimate tree damage and value loss but recognize that the equations were developed for red oaks and for trees with no longer than 15 years of fire history. It is not clear how the development of damage would differ over longer timescales; for example, it is possible that trees in our study had internal damage or rot that was not accounted for by the models. Instances of past damage (see Paulsell 1957) may have healed over and not been identified as external scars in the current study (Smith and Sutherland 1999).

Despite these shortcomings, there were several important findings. The effects of long-term, frequent prescribed burning on stand density and composition may result in greater losses in timber value than value loss associated with fire scars on sawtimber trees. During the transition from forest to woodland structure, tree scarring from low- to moderate-intensity fires may cause relatively small decreases in standing sawtimber value. Over a period of decades, burning results in open-structured woodland conditions with an increased dominance of fire tolerant tree species, such as post oaks. Thus, decisions regarding where to target woodland restoration should consider both the existing stand composition and the overall management objectives. However, changes in stand-level value loss associated with stand density and species changes are highly dependent on value differences among species. Future changes in red oak versus post oak markets could have large influences on value losses or gains. These results may differ further when fire intensities exceed those of our study or in stands that contain trees with higher (e.g., stave, veneer) or lower (e.g., firewood, blocking) value potential.

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Proceedings from the 2016 Central Hardwood Forest Conference in Columbia, MO. The published proceedings include 31 papers pertaining to research conducted on artificial and natural regeneration, biomass and carbon, forest dynamics, forest health, modeling and utilization, prescribed fire, soils and nutrients, and wind disturbance.

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