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Proceedings of the 19th Biennial Southern Silvicultural Research Conference

Blacksburg, VA

March 14-16, 2017

[Faint, tilted text from a dictionary or glossary, likely related to silviculture, is visible in the background.]



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Proceedings of the 19th Biennial Southern Silvicultural Research Conference

**March 14–16, 2017
Virginia Tech Skelton Conference Center
Blacksburg, VA**

Compiled by

Julia E. Kirschman

Hosted by

Department of Forest Resources and Environmental Conservation, Virginia Tech

USDA Forest Service, Southern Research Station

Preface

The 19th Biennial Southern Silvicultural Research Conference was held March 14–16, 2017, at the Inn at Virginia Tech and Skelton Conference Center, Blacksburg, VA. This conference provided a forum for silviculturists, researchers, and practitioners to actively engage in the broad field of silviculture, to report their study results, to present new concepts and techniques, to discuss topics of mutual interest, to coordinate cooperative efforts, and to stay current on developments in the field. Scientists, foresters, landowners, and others interested in forest management have found the conferences and their proceedings to be valuable sources of information on current and developing trends in southern forest silviculture.

The three-day event started with keynote speakers Tom Fox, Callie Jo Schweitzer, and Scott Barrett giving an *Overview of the Region; History, Highlights, and Perspectives of Southern Upland Hardwood Silviculture Research*; and *Biomass Harvesting and Utilization Market Adaptations for Wood Energy in Virginia*.

The conference offered 70 oral presentations along with a 55-poster session. Presentations were grouped into concurrent sessions under research topics: Oak Silviculture, Invasive Species, Disturbances and Damaging Agents, Forest Mensuration and Modeling, Pine Bark Beetles, Loblolly Pine Fertilization, Afforestation, Long-Term Silvicultural Studies, Bottomland Silviculture, Forest Soils and Best Management Practices, Loblolly Pine: Density and Competition Control, Shortleaf Pine Silviculture, Longleaf Pine Silviculture, Ecophysiology, and Fire.

A field tour on the last day focused on silviculture of Appalachian hardwoods and was hosted by personnel from the Virginia Tech Department of Forest Resources and Environmental Conservation and USDA Forest Service George Washington and Jefferson National Forests. The tour highlighted five area forests of the Ridge and Valley Physiographic Province in Montgomery and Giles Counties, VA.

Acknowledgments

The Biennial Southern Silvicultural Research Conference (BSSRC) started with the first regional conference held in Atlanta, GA, November 6–7, 1980. This was a result from a decision made in 1978 to continue the in-house biennial meetings conducted at the Southern and Southeastern Forest Experimental Stations level and expand them on a regional level for researchers, universities, and State and private organizations to participate and provide a forum to present research and discussions on issues facing southern forests. After 37 years of research and discussions, the conference is still a viable avenue for expanding silviculture research. These conferences would not be available without the dedication and collaboration between USDA Forest Service Southern Research Station scientists, universities, and State and private research organizations. We gratefully acknowledge Virginia Tech Department of Forest Resources and Environmental Conservation faculty, staff, and students for handling the local arrangements and providing a space for the 19th BSSRC. Thank you to the co-chairs and the committee members for the months of planning, organizing, and facilitating the conference. We also want to acknowledge the volunteers and staff that were moderators, field trip guides, presentation judges, and poster assemblers. We thank every person who supported making this conference a success.

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■ March 14, 2017

Session A:

Oak Silviculture – Callie Schweitzer, USDA Forest Service, Southern Research Station
Invasive Species – Rod Will, Oklahoma State University

Session B:

Disturbance and Damaging Agents – Mohammed Bataineh, University of Arkansas-Monticello
Forest Mensuration and Modeling – Yuhui Weng, Stephen F. Austin State University

Session C:

Pine Bark Beetles – Joshua Adams, Louisiana Tech University
Loblolly Pine Fertilization – Jim Guldin, USDA Forest Service, Southern Research Station

Session D:

Afforestation – Don Bragg, USDA Forest Service, Southern Research Station
Long-Term Silvicultural Studies – Brian Oswald, Stephen F. Austin State University

■ March 15, 2017

Session A:

Bottomland Silviculture – Brian Lockhart, USDA Forest Service, Southern Research Station
Forest Soils and Best Management Practices – Andy Ezell, Mississippi State University

Session B:

Loblolly Pine: Density and Competition Control – Eric Jokela, University of Florida
Shortleaf Pine Silviculture – Wayne Clatterbuck, University of Tennessee
Longleaf Pine Silviculture – Jennifer Gagnon, Virginia Tech

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Ecophysiology – Chris Maier, USDA Forest Service, Southern Research Station
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Oak Silviculture

Moderator:

Callie Schweitzer

USDA Forest Service, Southern Research Station

RESIDUAL EFFECTS OF MECHANICAL SITE PREPARATION ON SOIL COMPACTION IN OAK (*QUERCUS* SPP.) PLANTINGS

Andrew B. Self, Andrew W. Ezell, and Emily B. Schultz

Abstract—Mechanical site preparation is often used to aid with amelioration of compacted soil conditions typically found on former agricultural areas. While immediate reduction in soil compaction through use of cultural treatments is well studied, less research is available regarding longer term mechanical treatment residual effects. Four mechanical site preparation treatments were employed across three Mississippi sites during the winter of 2007. Treatments were installed using 10-foot centers as follows: control, subsoiling, bedding, and combination plowing. Two years post-treatment, 216 paired reading locations were randomly selected within each mechanical treatment area to sample soil resistance difference between treatment and non-treatment areas. Mechanical soil resistance was measured to a depth of 18 inches with readings taken at 3-inch depth intervals. Analysis determined significant site interactions. Consequently, sites were analyzed independently for main effects and interactions. Soil resistance differences varied by treatment with more intensive treatments exhibiting greater residual differences compared to those observed in less intensive treatment areas.

INTRODUCTION

Afforestation in the Delta region of Mississippi has resulted in the establishment of approximately 477,000 acres of hardwood plantations.¹ Successful establishment of these plantations is the result of research and operational efforts over the past several decades. Continued Federal and State cost share funding is expected to sustain interest in planting additional acreage into the foreseeable future.

Mechanical site preparation is often prescribed to aid in amelioration of the compaction problems sometimes associated with former agricultural fields (Allen and others 2001, Russell and others 1997, Self and others 2012, Stanturf and others 2004). Many former fields have substantial levels of compaction due to past land use practices (Allen and others 2001, Gardiner and others 2002, Stanturf and others 2004). Treatment with various forms of mechanical site preparation treatments can have beneficial effects regarding compaction commonly found in these areas. Potential increases in growth and survival from mechanical site preparation can come from improvements in moisture and nutrient uptake, organic matter concentration, enhanced root formation, and better planting quality (Ezell and Shankle 2004, Fisher and Binkley 2000, Kabrick and others 2005,

Patterson and Adams 2003, Rathfon and others 1995, Russell and others 1997, Stanturf and others 2004). Multiple treatment options exist; however, subsoiling, bedding, and combination plowing are the most commonly prescribed forest site preparation practices. These treatments are designed to disturb surface layers of the soil profile and fracture restrictive layers often found in retired agricultural fields. These efforts can lead to increased planted seedling growth and survival (Ezell and Shankle 2004, Gardiner and others 2002, Russell and others 1997, Stanturf and others 2000, Stanturf and others 2004). Measurement of soil resistance in mechanical site preparation treatments can be very informative regarding the impact of individual treatments on physical soil characteristics. While the immediate benefits of mechanical site preparation in reducing soil compaction are well known, the longer term effect on soil compaction is a relatively unexplored topic in hardwood afforestation.

MATERIALS AND METHODS

Site Description

This study was located on three sites. Two sites were owned by the Mississippi Department of Wildlife, Fisheries, and Parks (MDWFP). One site was located on Copiah County Wildlife Management Area (WMA), and the other was located on Malmaison WMA. The third site was located near Arkabutla Lake on land owned by the U.S. Army Corps of Engineers.

¹ Personal communication. Dan Prevost. 2017. Forester/ Wildlife Biologist, Delta Wildlife, 433 Stoneville Road, Stoneville, MS 38776.

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Malmaison WMA site - The Malmaison WMA study area was located approximately 14 miles northeast of Greenwood, MS in Grenada County. The site was formerly used in row-crop production and retired from agricultural production in the late 1990s. It was maintained as an opening for wildlife using mowing and disking from agricultural retirement until the initiation of this study. Soils were silt loams, and 40-year average yearly precipitation was 53.8 inches (NOAA 2011c). Soil tests indicated onsite pH ranged from 6.3 to 7.0.

Copiah County WMA site - The Copiah County WMA study area was located approximately 16 miles northwest of Hazlehurst, MS in Copiah County and was retired from row crop production in the 1980s. It was maintained as an opening for wildlife using mowing and disking from agricultural retirement until the initiation of this study. Soils were silt loams, and 35-year average yearly precipitation was 59.2 inches (NOAA 2011a). Soil tests indicated that onsite pH is 5.2.

Arkabutla Lake site - The Arkabutla Lake study area was located approximately 5 miles northwest of Coldwater, MS in Desoto County. The site was in soybean [*Glycine max* (L.) Merr.] production until September 2007. Soil series were silt loams, and 40-year average precipitation was 56.1 inches (NOAA 2011b). Soil tests indicate that the site had an average pH of 6.2.

Experimental Design

The study was completely replicated at all three sites. Each site had its own unique installment of randomized mechanical site preparation treatments. A split-plot design was utilized with site preparation treatments in a randomized complete block. Three blocks were established with each block receiving four different mechanical site preparation treatments randomly applied as a group.

Mechanical Site Preparation Treatments

Four mechanical site preparation treatments were used in this study: control (no mechanical treatment), subsoiling, bedding, and combination plowing. All site preparation treatments were applied on 10-foot centers. Subsoiling was performed to a depth of 15 inches using the Case International eco-lo-til™ 2500 subsoiler system. Bedding was performed using a furrow plow with the blades set to pull a soil bed approximately 3 feet wide and between 8 and 10 inches deep. Combination plowing involved pulling a soil bed over the top of subsoiled trenches. Mechanical site preparation treatments were applied during the first week of November, 2007.

Soil Resistance Readings

Mechanical soil resistance was determined using a Field Scout SC900 Soil Compaction Meter. Paired-measurement readings were taken on each plot to a depth of 18 inches with resistance measured in pounds per square inch (PSI). Two hundred eighty-eight locations were randomly selected for soil resistance readings at each site, totaling 864 reading locations across all three sites (288 total per site = 72 per mechanical treatment = 24 per mechanical treatment per block). Readings were recorded at depths of 3, 6, 9, 12, 15, and 18 inches at each measurement location. Readings were performed approximately 22 months post-mechanical treatment when soil moisture conditions were slightly below field capacity during September, 2009.

Data Analysis

All statistical analyses were performed using Statistical Analysis System version 9.2 (SAS 9.2). Soil resistance readings were analyzed by taking the difference between readings within mechanical treatments and readings outside of mechanical treatments. General Linear Modeling (GLM) and analysis of variance (ANOVA) were used to test null hypotheses. Full model was significant and a significant site interaction was found. Analyses were then conducted for each site separately.

Proc GLM was used to test for main effects and interactions, and to estimate least square means (LSMEANS). Separate analyses were used to identify a treatment effect if a significant interaction occurred. The LSMEANS LINES option was used to identify differences among pairwise comparisons if a significant interaction occurred. Differences were considered significant at the $\alpha = 0.05$ level of significance.

RESULTS AND DISCUSSION

Analyses of soil resistance readings were performed on the difference between readings within the mechanical treatment and immediately adjacent to the mechanical treatment. Consequently, positive PSI values indicate that soil within the treatment exhibited less resistance compared to the area immediately out of the treatment area. Negative PSI values indicate that the soil within the treatment exhibited more resistance compared to the area immediately outside of the treatment area.

During analysis, a general trend was observed for 18-inch resistance readings at the Arkabutla Lake site. Readings in subsoiled and combination plowed areas showed substantially greater soil resistance at a depth of 18 inches compared to control areas. This anomaly was not observed at the Copiah County WMA or Malmaison WMA sites. The Arkabutla Lake site had a

band of clay soil that was observed in soil profile pits at around 16 to 17 inches depth. Down pressure from the subsoil plow foot being pulled at a depth of 15 inches is thought to have served to create an artificial compaction layer. This band of clay was not present at the Copiah County WMA or Malmaison WMA sites and increased soil resistance was not encountered at the 18-inch depth. Due to these artificially increased readings at 18 inches at the Arkabutla Lake site and the similarity of resistance readings of the other mechanical treatment/site combinations at the 18-inch depth, readings at this depth were not used for further analysis.

Soil Resistance Differences by Site and Mechanical Treatment

A significant main effect difference was detected among mechanical site preparation treatments for soil resistance difference at the Arkabutla Lake site ($p = <0.0001$, $F = 272.52$). Overall, difference in soil resistance for the subsoiling and combination plowing treatments (247.5 PSI and 222.9 PSI, respectively) did not differ at the Arkabutla Lake site (table 1). Both were greater than average difference in bedded areas (119.0 PSI) and all three were greater than the difference in control areas (7.9 PSI). This indicates that all three mechanical treatments were successful in reducing soil resistance compared to areas where no mechanical site preparation was performed.

Analysis detected a significant main effect difference among mechanical site preparation treatments for soil resistance difference at the Copiah County WMA site ($p = <0.0001$, $F = 176.86$). Statistical ranking for overall differences of soil resistance readings by mechanical site preparation treatment at the Copiah County WMA site was identical to that of the Arkabutla Lake site (table 1). Overall, soil resistance measures for the subsoiling and combination plowing treatments (104.5 PSI and 105.9 PSI, respectively) did not differ significantly. Both were greater than average difference in bedded areas (15.9 PSI) and all three were greater than the difference in control areas (-1.3 PSI). This indicates that all three

mechanical treatments were successful in reducing soil resistance compared to areas where no mechanical site preparation was performed. Overall, the Copiah County WMA site exhibited observably lower levels of soil resistance compared to either of the other two sites. An explanation for this occurrence might be that the Copiah County WMA site was in intensive deep cultivation immediately prior to the establishment of the study. It is possible that disturbance of this nature could require a significant amount of time to return to a state similar to soil conditions found in more traditionally cultivated areas. Neither of the other sites received this type of cultivation.

Another dissimilarity was observed between the Copiah County WMA and other sites. Soil resistance was appreciably lower to a depth of 15 inches in subsoiled and combination plowed areas compared to control areas for all three sites (table 2). Readings in bedded areas indicated that a substantial reduction in resistance existed to a depth of 9 inches at the Arkabutla Lake and Malmaison WMA sites. However, bedded areas at the Copiah County WMA site exhibited lower PSI readings only at the 3-inch depth. This divergence of the Copiah County WMA site from the other two sites is probably a result of the deep cultivation. It is thought that loose soil created by the bedding treatment filled in fissures created by deep cultivation. This siltation effect of loose soil could have negated the positive effects of bedding observed at the other two sites.

Analysis detected a significant main effect difference among mechanical site preparation treatments for soil resistance difference at the Malmaison WMA site ($p = <0.0001$, $F = 290.73$). Ranking for mechanical treatments was similar to ranking observed at the Arkabutla Lake and Copiah County WMA sites. Overall, difference in soil resistance for the subsoiling and combination plowing treatments (269.8 PSI and 252.5 PSI, respectively) did not differ at the Malmaison WMA site (table 1). Both were greater than average

Table 1—Overall soil resistance differences by site and mechanical treatment at two years post-treatment

	Arkabutla Lake	Copiah County WMA	Malmaison WMA
Mechanical treatment	soil resistance difference (PSI) ^a		
Subsoiling	247.5a	104.5a	268.8a
Combination plowing	229.9a	105.9a	252.5a
Bedding	119.0b	15.4b	88.6b
Control	7.9c	-1.3c	1.1c

^a Values followed by different letters within a column are significantly different at $\alpha = 0.05$.

Table 2—Two-year post-treatment soil resistance differences by site and mechanical treatment and depth interaction

Mechanical treatment	Depth of reading	Arkabutla Lake	Copiah County WMA	Malmaison WMA
	<i>inches</i>	<i>soil resistance difference (PSI)^a</i>		
Control	3	-0.6a	-4.9a	-5.0c
	6	8.1a	4.2a	17.9a
	9	7.5a	-3.1a	7.6b
	12	9.9a	0.7a	-9.6c
	15	14.8a	-3.1a	-5.2c
Subsoiled	3	15.1d	39.8c	26.3d
	6	234.8c	105.7b	252.0c
	9	371.5a	169.7a	383.8a
	12	310.6b	186.7a	372.0a
	15	305.5b	20.7c	310.1b
Bedded	3	72.0c	55.9a	55.1c
	6	302.5a	13.6b	287.4a
	9	246.7b	-1.1b	111.5b
	12	-32.3e	-38.7c	-22.8d
	15	6.0d	47.5a	11.9d
Combination plowed	3	59.5c	54.2c	52.1c
	6	290.6a	65.9c	284.7b
	9	248.9b	159.5a	374.7a
	12	232.3b	153.6a	349.0a
	15	318.1a	96.5b	302.1b

^a Values followed by different letters within a column are significantly different at $\alpha = 0.05$.

difference in bedded areas (88.6 PSI), and all three were greater than soil resistance difference in control areas (1.1 PSI). This indicates all three mechanical treatments were successful in reducing soil resistance compared to control areas.

Soil Resistance Differences by Site, Mechanical Treatment, and Depth

Analysis detected a significant interaction between mechanical treatment and depth for difference in soil resistance at the Arkabutla Lake site ($p < 0.0001$, $F = 98.89$). No difference in soil resistance was observed at any level in control areas (table 2). Resistance in subsoiled areas differed by depth with the greatest difference at a depth of 9 inches (371.5 PSI). Resistance at the 12- and 15-inch depths did not differ (310.6 PSI and 305.5 PSI, respectively), but resistance at both depths was greater than at the 6- or 3-inch

depths (234.8 PSI and 15.1 PSI, respectively). Greater differences in soil resistance at deeper locations in the soil were expected due to the deep fracturing qualities associated with subsoiling.

The greatest difference in soil resistance observed in bedded areas was at a depth of 6 inches (302.5 PSI), followed by a depth of 9 inches (246.7 PSI), then 3 inches (72.0 PSI) (table 2). As expected, the least differences in soil resistance were observed at the 12- and 15-inch depths (-32.3 PSI and 6.0 PSI, respectively) with the 12-inch depth exhibiting increased resistance within the treatment. The bedding treatment did not extend below 10 inches in depth and should not have had substantial influence on readings below that depth. A greater resistance of -32.3 PSI at the 12-inch depth within the bedding treatment was minor. Statistical ranking of soil resistance differences in

combination plowed areas was similar to that observed in subsoiled areas. Resistance differences at levels deeper than 3 inches were all significantly greater than resistance difference at 3 inches (59.5 PSI). Resistance differences at 6 and 15 inches (290.6 PSI and 318.1 PSI, respectively) did not differ, but were greater than those observed for the 9- and 12-inch depths (248.8 PSI and 232.3 PSI, respectively). Due to combination plowing including a subsoiling treatment, the similarity between the two treatments was expected.

Analysis detected a significant interaction between mechanical treatment and depth for difference in soil resistance at the Copiah County WMA site ($p = <0.0001$, $F = 27.77$). No difference in soil resistance was observed at any level in control areas (table 2). Similar soil resistance differences were observed at depths of 9 and 12 inches (169.7 PSI and 186.7 PSI, respectively) in subsoiled areas. Significantly less difference in soil resistance was noted at the 6-inch depth (105.7 PSI). Analysis detected the least soil resistance difference at the 3- and 15-inch depths (39.8 PSI and 20.7 PSI, respectively). No difference in soil resistance difference was noted at these two depths. As discussed earlier, greater differences in soil resistance readings were expected in the lower depths. It is possible that lower resistance levels observed at the 15-inch depth were influenced by the deep cultivation mentioned earlier.

The greatest differences in soil resistance observed in bedded areas were at depths of 3 and 15 inches (55.9 PSI and 47.5 PSI, respectively) (table 2). Resistance differences at these two depths were not significantly different. However, both were greater than differences observed at the 6- and 9-inch depths (13.6 PSI and -1.1 PSI, respectively). Soil resistance within treatment was found to be significantly higher at the 12-inch depth (-38.7 PSI). Readings at this depth indicated that there was a substantial increase in soil resistance with bedding compared to all other treatments. It is possible that siltation issues discussed earlier resulted in increased compaction at the 12-inch depth. While not statistically differentiated, soil resistance difference at the 9- and 12-inch depths (159.5 PSI and 153.6 PSI, respectively) was greater than resistance difference at the 15-inch depth (96.5 PSI). The least difference in soil resistance readings was observed in the 3- and 6-inch depths (54.2 PSI and 65.9 PSI, respectively).

Analysis detected a significant interaction between mechanical treatment and depth for difference in soil resistance at the Malmaison WMA site ($p = <0.0001$, $F = 39.31$). Unlike in Arkabutla Lake and Copiah County WMA site observations for difference in soil resistance differences, differences were noted in control areas (table 2). The greatest difference in soil resistance in control areas was observed at the 3-inch depth (17.9 PSI).

Less difference in soil resistance was detected at the 9-inch depth (7.6 PSI). Analysis detected similar ranking in soil resistance difference at the 3-, 12-, and 15-inch depths (-5.0 PSI, -9.6 PSI, and -5.2 PSI, respectively). Difference between readings was negative at these three depths indicating that there were slight increases in soil resistance within the treatment. It is improbable that the slightly increased soil resistance observed in control areas would have any negative influence on planted seedling growth or survival. While statistically different, resistance differences at varying depths throughout the soil column are very small, ranging from -9.6 PSI to 17.9 PSI. It is highly unlikely that changes of this level would have any significant real world impacts on growth or survival of any seedlings planted.

Soil resistance difference rankings and the overall numeric scale of differences were very similar to the Arkabutla Lake site in bedded areas. Resistance in subsoiled areas differed by depth with the greatest observed difference at depths of 9 and 12 inches (383.8 PSI and 372.0 PSI, respectively) (table 2). Resistance at these depths did not differ, but resistance at both depths was greater than observed at the 15-inch depth (310.1 PSI). Less resistance difference was found at the 6-inch depth (252.0 PSI), and the lowest overall difference was noted at the 3-inch depth (26.3 PSI). Again, greater differences in soil resistance at deeper locations in the soil were expected.

Overall, bedded area resistance differences and the numeric scale of those differences were similar to those observed for the Arkabutla Lake site. The greatest difference in soil resistance observed in bedded areas was at a depth of 6 inches (287.4 PSI), followed by readings at the 9-inch depth (111.5 PSI), then resistance difference at the 3-inch depth (55.1 PSI) (table 2). The least differences in soil resistance were found at the 12- and 15-inch depths (-22.3 PSI and 11.9 PSI, respectively) with the 12-inch depth exhibiting increased resistance within the treatment. Again, the bedding treatment did not extend below 10 inches in depth and should not have had substantial influence on readings below that depth. The small increase in soil resistance observed within the bedding treatment is minor.

Statistical ranking of soil resistance differences in combination plowed areas was similar to those observed at the Arkabutla Lake and Copiah County WMA sites. Resistance differences at levels deeper than 3 inches were all significantly greater than resistance difference at the 3-inch depth (52.1 PSI) (table 2). Resistance differences at 9 and 12 inches (374.7 PSI and 349.0 PSI, respectively) did not differ, but were greater than those observed for the 6- and 15-inch depths (284.7 PSI and 302.1 PSI, respectively).

CONCLUSIONS

As expected, two years after treatment, areas receiving deep ripping treatments (subsoiling and combination plowing) possessed reduced levels of soil compaction compared to those that did not receive similar treatment (bedding and control areas). Decreased soil compaction is likely beneficial in every hardwood planting, but becomes of paramount importance in former agricultural fields. Often these areas possess artificially increased compaction which may require mechanical treatment for successful plantation establishment to occur.

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CROP TOLERANCE OF OAK SEEDLINGS IN HERBACEOUS WEED CONTROL APPLICATIONS USING INDAZIFLAM

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Abstract—Planting bareroot oak seedlings continues to account for substantial acreage across the South, especially on retired agricultural lands. It is now well established that good seedlings, good planting, and herbaceous weed control (HWC) are the trilogy of factors needed for consistently successful establishment. Survival exceeding 90 percent is now common when all three factors are satisfied. Even though research on HWC for oaks (*Quercus* spp.) began more than 25 years ago, the list of effective materials available for such use is still very short. Cost efficacy and crop tolerance remain the critical elements of evaluation for any new application. Indaziflam has shown promise for HWC applications in longleaf pine (*Pinus palustris*). In order to evaluate its use for oaks, six treatments were applied over the top of two species of recently planted oak seedlings at two planting sites in south Mississippi. All treatments were replicated three times at each site in these plantings of 1-0 bareroot seedlings. Plots were evaluated at 30, 60, 90, 120, and 150 days after treatment. Results indicate that indaziflam could be a useful alternative in oak plantings. The study provided a comparison of the indaziflam treatments to the current operational standard of post-plant sulfometuron methyl applications.

INTRODUCTION

Herbaceous weed control (HWC) in hardwood plantings continues to be an important consideration. Compared to similar practices in pine plantations, the list of approved chemicals is much more limited. That lack of materials combined with the occurrence of most hardwood plantings on retired agricultural areas which can have a much more established and competitive weed complex creates a serious challenge for hardwood managers on many sites.

Improved survival is the goal of herbaceous weed control in hardwoods. Survival may be increased 20–44 percent after the first growing season depending on the situation (Ezell and others 2007). Oust® XP was established as the “gold standard” for HWC in hardwoods almost 20 years ago (Ezell and Catchot 1998). However, evaluating new materials is always a worthwhile effort in the search for improved cost efficiency.

OBJECTIVES

The objectives of this study were to (1) evaluate crop tolerance of planted oak seedlings during the first growing season to treatments containing indaziflam (Esplanade®) and (2) evaluate efficacy of treatments in different field situations.

MATERIALS AND METHODS

Study Sites

A total of four study sites were utilized in south Mississippi: two sites in George County, MS (near Lucedale, MS) and two sites in Stone County, MS (near Wiggins, MS). All areas had been cleared after Hurricane Katrina seriously damaged or eliminated previous forest cover on the sites. Initial conditions on all sites were very similar to retired agricultural areas. Vegetation complexes varied greatly among sites, but all had well established competition including both grasses and forbs.

Treatments – A complete list of treatments is found in table 1. Treatments 1–6 were applied in both 2015 and 2016 while Treatment 7 was applied only in 2016.

Table 1—List of treatments in 2015 and 2016 Bayer hardwood studies

Trt. no.	Herbicide and rate per acre
1	Untreated
2	Oust® XP (2 oz.)
3	Esplanade® (3.5 oz.)
4	Esplanade® (7 oz.)
5	Oust® XP (2 oz.) + Esplanade® (3.5 oz.)
6	Oust® XP (2 oz.) + Esplanade® (7 oz.)
7 (2016 only)	Method® (6 oz.) + Esplanade® (7 oz.)

Note: line above Trt. no. 7 is to emphasize that this treatment was applied in only one year.

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Plot Layout and Experimental Design – Each plot consisted of an area 6 feet wide and 100 feet long. This represented 100 feet of a planted row with a 6-foot-wide spray area. Each plot contained a minimum of 10 planted oak seedlings.

Three replications of each treatment were installed both years. A randomized complete block design was utilized in all applications.

Application – All treatments were applied using a CO₂-powered backpack sprayer with a hand-held wand and single TK 2.5 Floodjet nozzle. These were banded applications using a 6-foot swath over the top of the planted seedling. The 2015 treatments were applied April 2, 2015 due to a delay in arrival of materials, and the 2016 treatments were applied March 16, 2016. All sites had greater than 80 percent coverage of competing vegetation at the time of application.

Species/seedlings – Three oak species were utilized in this study: Nuttall oak (*Quercus nuttallii*), Shumard oak (*Q. shumardii*), and swamp chestnut oak (*Q. michauxii*). Two species were included in each installation, with Shumard being utilized both years. In all plantings, high-quality 1-0 bareroot seedlings were utilized. Seedlings were obtained from a forest industry nursery in Mississippi and satisfied the minimum order specification requirements of ≥ 20-inch height above the root collar and a minimum of 10 first order lateral roots (FOLR). Seedlings were hand-planted in February, 2015 and February, 2016.

Evaluations – Plots were evaluated at 30, 60, 90, 120, and 150 days after treatment. All seedlings were examined for any signs of phytotoxic impact. An ocular estimate of the percent cover by grass or broadleaves was completed for each plot at each evaluation timing.

RESULTS

Field data were analyzed and means separation tests were applied. Due the general lack of statistical significance, no means separation is presented. Tabular information clearly defines the control (or lack thereof) in the treatments. In the 2015 trials, no treatments were successful in controlling the grass at the George County site (table 2). Coverage was well established at the time of application and increased with time. This was not unexpected, as the principal grass species on the site were cogongrass (*Imperata cylindrica*) and *Andropogon*, and Oust® XP is not effective on these species. Broadleaf forb control was excellent in the 2015 George County plots with all treatments giving good results (table 3). It is possible that grass coverage helped keep forb coverage low, but treated areas had less forb cover than the untreated check. Grass was well controlled by all treatments containing Oust® XP at the Stone County site in 2015 (table 4). Grass coverage at the site decreased throughout the evaluation period, but the 2-ounce-per-acre Oust® XP treatment had best overall results. By contrast, no treatments were effective at controlling all the broadleaf species at the Stone County site in 2015 (table 5). The Stone County site had common ragweed (*Ambrosia artemisiifolia*), horseweed (marestail) (*Erigeron canadensis*), goldenrod (*Solidago* spp.), and sicklepod (*Senna obtusifolia*). Thus, what was not controlled quickly covered any open areas. For example, Oust® XP controlled the ragweed but not sicklepod; Esplanade® provided sicklepod control, but not for ragweed.

In the 2016 trials, none of the treatments provided good grass control at the George County site (table 6). This was an old field setting with a number of grass species, and while most species were controlled, brown-top millet was not and quickly covered all plots. Treatments with Oust® XP did a good job of controlling forbs on the

Table 2—Average percent grass cover by treatment and time of observation – George County site (2015)

Trt. ^a	30	60	90	120	150
-----percent-----					
Untrt.	90.0	93.7	96.3	96.3	96.7
O(2)	40.0	82.7	94.7	93.7	94.3
E(3.5)	80.0	90.0	95.0	94.3	95.7
E(7)	86.7	90.7	93.3	94.7	95.7
O(2) + E(3.5)	50.7	88.7	94.3	94.3	95.7
O(2) + E(7)	51.7	84.3	92.3	94.3	96.7

^a Trt. = treatment [herbicide and rate (in ounces) per acre]. O = Oust® XP; E = Esplanade®.

Table 3—Average percent forb cover by treatment and time of observation – George County site (2015)

Trt. ^a	30	60	90	120	150
-----percent-----					
Untrt.	15.0	13.3	13.3	13.3	13.3
O(2)	1.0	0.7	1.3	1.5	1.7
E(3.5)	7.3	1.3	2.0	3.0	3.0
E(7)	5.0	1.3	2.7	2.3	2.7
O(2) + E(3.5)	3.0	1.0	2.0	2.0	2.0
O(2) + E(7)	2.7	1.0	1.3	1.3	2.0

^a Trt. = treatment [herbicide and rate (in ounces) per acre]. O = Oust® XP; E = Esplanade®.

Note: Values in bold are different from untreated at alpha=0.05.

Table 4—Average percent grass cover by treatment and time of observation – Stone County site (2015)

Trt. ^a	30	60	90	120	150
-----percent-----					
Untrt.	80.0	46.3	13.7	16.7	19.3
O(2)	5.0	8.3	2.7	2.7	3.3
E(3.5)	77.0	16.7	3.7	3.0	4.0
E(7)	77.0	24.3	2.3	6.0	6.3
O(2) + E(3.5)	4.3	18.7	3.0	8.3	7.0
O(2) + E(7)	1.0	7.3	5.3	8.0	9.0

^a Trt. = treatment [herbicide and rate (in ounces) per acre]. O = Oust® XP; E = Esplanade®.

Note: Values in bold are different at alpha=0.05.

Table 5—Average percent forb cover by treatment and time of observation – Stone County site (2015)

Trt. ^a	30	60	90	120	150
-----percent-----					
Untrt.	15.3	75.0	100.0	92.3	95.0
O(2)	5.0	53.3	82.3	86.7	90.0
E(3.5)	10.0	76.3	88.3	92.3	93.3
E(7)	10.0	63.3	86.7	89.3	90.0
O(2) + E(3.5)	5.0	30.7	81.7	79.7	82.3
O(2) + E(7)	5.0	35.0	74.7	70.0	74.7

^a Trt. = treatment [herbicide and rate (in ounces) per acre]. O = Oust® XP; E = Esplanade®.

George County site in 2016 (table 7). The combination of Oust® XP with the lower rate of Esplanade® (3.5 ounces per acre) provided better control than the combination using the higher rate (7 ounces per acre) of Esplanade®. This result was due to the distribution of one difficult-to-control species, but it does indicate that 3.5 ounces per acre of Esplanade® may be a sufficient amount for herbaceous weed control in many situations. At the Stone County site in 2016, none of the treatments provided good control of the grass species (table 8). This site had a very strong complex of grass species including cogongrass and tetraploid bahiagrass (*Paspalum notatum* Flügge), neither of which is controlled by any of the treatments. Much of the plant diversity on this site may be attributed to various land

uses since being cleared after Hurricane Katrina which include pasture, wildlife food plots, and gardening. Only the treatments containing Oust® XP provided acceptable broadleaf control on this site (table 9). As compared to the George County site of 2016, the higher rate of Esplanade® (7 ounces per acre) provided better control than the lower rate during the latter part of the growing season when combined with Oust® XP. This is attributed to the weed complex on the site.

Crop Tolerance – A major emphasis of these studies was to evaluate the crop tolerance of planted oak seedlings to indaziflam using the product Esplanade®. More than 750 seedlings were evaluated during the course of the two years of field trials. Of that total, the only seedlings

Table 6—Average percent grass cover by treatment and time of observation – George County site (2016)

Trt. ^a	30	60	90	120	150
-----percent-----					
Untrt.	0.7	4.3	73.3	65.0	70.0
O(2)	0.0	4.0	96.7	93.3	80.0
E(3.5)	0.0	2.3	78.3	60.0	33.3
E(7)	0.0	3.0	31.7	31.7	18.3
O(2) + E(3.5)	0.0	0.7	90.0	91.7	85.0
O(2) + E(7)	0.0	1.0	53.3	76.7	50.0
M(7) + E(7)	0.0	6.0	63.3	65.0	80.0

^a Trt. = treatment [herbicide and rate (in ounces) per acre]. O = Oust® XP; E = Esplanade®; M = Method®.

Note: line above Trt. no. 7 is to emphasize that this treatment was applied in only one year.

Table 7—Average percent forb cover by treatment and time of observation – George County site (2016)

Trt. ^a	30	60	90	120	150
-----percent-----					
Untrt.	75.0	93.3	35.0	46.7	46.7
O(2)	1.7	5.0	3.7	6.7	16.7
E(3.5)	60.0	80.0	15.0	43.3	66.7
E(7)	65.7	70.0	41.7	66.7	35.0
O(2) + E(3.5)	4.0	11.0	6.3	10.0	25.0
O(2) + E(7)	0.0	1.7	13.3	25.0	33.3
M(7) + E(7)	0.0	4.7	13.3	28.3	10.0

^a Trt. = treatment [herbicide and rate (in ounces) per acre]. O = Oust® XP; E = Esplanade®; M = Method®.

Note: Values in bold are different at alpha=0.05.

Note: line above Trt. no. 7 is to emphasize that this treatment was applied in only one year.

Table 8—Average percent grass cover by treatment and time of observation – Stone County site (2016)

Trt. ^a	30	60	90	120	150
-----percent-----					
Untrt.	60.0	61.7	50.0	61.7	36.7
O(2)	23.3	71.7	100.0	95.0	94.3
E(3.5)	71.7	80.0	93.3	86.7	56.7
E(7)	66.7	70.0	66.7	76.7	60.0
O(2) + E(3.5)	43.3	70.0	96.7	71.7	85.0
O(2) + E(7)	25.0	56.7	98.3	88.3	86.7
M(7) + E(7)	91.7	100.0	100.0	96.7	86.7

^a Trt. = treatment [herbicide and rate (in ounces) per acre]. O = Oust® XP; E = Esplanade®; M = Method®.

Note: line above Trt. no. 7 is to emphasize that this treatment was applied in only one year.

Table 9—Average percent forb cover by treatment and time of observation – Stone County site (2016)

Trt. ^a	30	60	90	120	150
-----percent-----					
Untrt.	60.0	53.3	63.3	46.7	76.7
O(2)	20.0	6.7	10.0	5.7	5.7
E(3.5)	38.3	30.0	23.3	26.7	53.3
E(7)	41.7	36.7	46.7	28.3	43.3
O(2) + E(3.5)	20.0	0.7	10.7	33.3	25.0
O(2) + E(7)	8.3	0.3	9.0	12.3	13.3
M(7) + E(7)	8.3	0.0	8.3	11.7	10.0

^a Trt. = treatment [herbicide and rate (in ounces) per acre]. O = Oust® XP; E = Esplanade®; M = Method®.

Note: Values in bold are different at alpha=0.05.

Note: line above Trt. no. 7 is to emphasize that this treatment was applied in only one year.

exhibiting any phytotoxic symptoms were those in plots treated with Method® (treatment #7 in 2016). Both Shumard and swamp chestnut oak seedlings were affected with swamp chestnut seedlings exhibiting more severe symptoms. However, this did not result in any additional mortality.

SUMMARY

Indaziflam appears to be safe for use over planted oak seedlings. This material provided best competition control when mixed with sulfometuron methyl (Oust® XP) and the mixture did not create any negative impacts on the seedlings. Control of all vegetation was less than

desirable for most treatments, but the weed complexes on the study sites were more challenging than those usually encountered in oak plantings.

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IMPLICATIONS OF FREQUENT HIGH INTENSITY FIRE ON THE LONG-TERM STABILITY OF OAK BARRENS, WOODLANDS, AND SAVANNAS

Wayne K. Clatterbuck and Rebecca L. Stratton Rollins

Abstract—A study was initiated in 1963 to evaluate the stand dynamics associated with three fire frequency treatments (annual burning, 5-year periodic burning, and fire exclusion) on a pyric oak (*Quercus* spp.)-dominated site in south-central Tennessee. Controlled burns were conducted during the dormant season. The experimental design was a randomized block, blocked on location, with three replications of the three treatments. The purpose of this paper is to report on the overstory structural vegetation changes that accompany these burning treatments after 54 years. Overstory number of stems and basal area of the two burning treatments have gradually diminished through time and with little to no ingrowth. Presently, the overstory of both fire frequency treatments consists of sparsely populated, individual trees with <18 square feet per acre of basal area. Most of these trees are decrepit with fire scars and decay jeopardizing their longevity. Annual burns promoted an oak savanna-like structure dominated by herbaceous vegetation. The 5-year periodic burns promoted woody vegetation, which typically was top-killed and resprouted after each burn. The fire exclusion treatment had a closed overstory with basal areas greater than 78 square feet per acre and little midstory or understory. Our results suggest high intensity, small-scale, frequent fires in pyric oak systems do not support oak ingrowth and would be relatively unstable communities. To allow oak ingrowth to occur, land managers should cease burning for a greater period of time or conduct lower intensity burns.

INTRODUCTION

The stand structure and composition of many former oak barrens, woodlands, and savannas were shaped by frequent fire regimes from both natural and anthropogenic sources (Peterson and Reich 2001). With the advent of fire suppression, these open vegetation communities have diminished and are quite rare (Dey and others 2017, Noss 2013). Interest in the use of frequent prescribed burning to restore these pyric oak structures has increased (Keyser and others 2016). Typically, with frequent burning, a diverse herbaceous understory of grasses and forbs develops with sparse, open overstories of scattered trees or groups of trees. A midstory canopy layer is usually absent or limited (Burger and others 2016).

A topic not addressed well in the literature is the impact of frequent burning damage on overstory trees and the resulting long-term sustainability implications of these systems. To address this topic, we capitalize on a University of Tennessee 54-year long-term fire frequency and stand dynamics research study and evaluated the impact of frequent, high intensity fire on the present condition of overstory trees and ingrowth for an oak barrens site in Tennessee (DeSelm 1994).

METHODS

Study Site

The study was initiated in 1963 and is located at 35°30'N: 86°15'W on the Interior Low Plateau Province of the Eastern Highland Rim in middle Tennessee on the University of Tennessee Forest Resources Research and Education Center (FRREC) in Franklin County, near Tullahoma, TN. Although the Eastern Highland Rim is typically characterized as more rolling with hills and valleys, the Interior Low Plateau Province on the eastern edge of the Rim is an undulating flat plain derived from loess-derived soils with a fragipan that inhibits water movement (Fenneman 1938). Soils are mapped as the Dickson series (fine-silty, siliceous, thermic Glossic Fragiudults with slopes of 0 to 2 percent (USDA NRCS 2001). These soils are usually excessively dry during late summer and have a perched water table during late winter. The study site is on Landtype 12 (Broad Silty Uplands) of Smalley's (1983) site classification of the Eastern Highland Rim. The climate is characterized by long, moderately hot summers and short, mild winters (Thorntonwaite 1948). Annual average monthly temperatures range from 40 °F in December and January to 77 °F in July and August. Annual monthly precipitation ranges from 5.4 inches per month December–March, 4.4 inches per month April–July, and 3.5 inches per month

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August–November with a monthly low of 2.5 inches in October. Total average annual precipitation is 53 inches (Smalley 1983).

The area had a long history of low intensity, frequent fires for agriculture and grazing from Native American culture through European settlement. Other more modern sources of fire, prior to University acquisition of the property, included wildfires from an adjacent large U.S. Army military complex (presently Arnold Air Force Base or Arnold Engineering Development Center) that used fire for clearing the area for training and maneuvers during World War II, wildfires resulting from a primary north-south railroad line adjacent to the property, and wildfires from adjacent farmers and landowners who burned for agriculture and grazing (DeSelm and others 1991).

Known as the ‘oak barrens,’ the subclimax pyric community is characterized as a sparse overstory with basal areas from 5 to 20 square feet, little to no midstory, and an open grass and forb understory (Delcourt 1979, DeSelm 1994). At study inception in 1963, the forest was understocked and degraded due to previous (pre-1950) frequent fires, poor previous management practices (primarily high-grading), and overgrazing (Stratton 2007). The forest structure was at the early stages of a two-aged forest. The initial forest condition was a sparse, mature overstory of approximately 80-year-old oaks with approximately 20 square feet per acre average basal area and 20 trees per acre. Basal area of the midstory [5 to 11 inches diameter breast height (DBH) at 4.5 feet] was approximately 30 square feet per acre with 100 trees per acre. Nearly 100 percent of the midstory stems were oak species. Total stand basal area (overstory and midstory) was approximately 50 square feet per acre (Nichols 1971). The study area was in the stand initiation phase (Oliver and Larson 1996) with the understory composed of the same woody species present today. The present overstory is primarily oak species including post oak (*Q. stellata*), southern red oak (*Q. falcata*), scarlet oak (*Q. coccinea*), and blackjack oak (*Q. marilandica*). Other species include hickories (*Carya* spp.) and blackgum (*Nyssa sylvatica*). Willow oak (*Q. phellos*), water oak (*Q. nigra*), and red maple (*Acer rubrum*) occur along the first-order stream which divides the study sites. Two age classes currently represent the overstory trees: an older age class for white oaks, primarily post oak at an estimated 120 to 150 years old and a slightly younger age class for the red oaks from 80 to 100 years old. Site index for upland oaks averages 70 feet at 50 years (Smalley 1983).

Study design

Nine 1.8-acre rectangular (200 x 400 feet) plots were established, three plots of each treatment: annual burn, 5-year periodic burn, and control (no burns). The original study design was a randomized block, blocked on

location with three replications of the three treatments. Controlled burns (henceforth burns) were conducted during the dormant season from late February to early April when burning conditions were favorable. A plowed fire line separated each plot. Each plot was burned individually in a ring pattern. Each burn was distributed evenly across the plot, but as a result of the fire ring pattern, the intensity of the fire was greater in the center than the edge of the plot. The annual treatments started in 1963, and 5-year periodic treatments started in 1964. A tornado went through the study area in March 2011 disrupting the continuity of the annual burn regimes for 2 years. According to FRREC records, the annual burn plots have been burned 50 times in 54 years and the 5-year periodic burn plots have been burned 11 times.

Sampling

The 2016 data collected for this report were obtained after the fourth growing season of the last 5-year periodic burn regime. Each treatment plot was bisected lengthwise with a line transect where 1/10-acre measurement plots were located at one-chain intervals yielding five measurement plots (15 total plots per treatment). The sampled area represented 0.5 acres of the 1.8-acre treatment plot or a sampling intensity of about 28 percent. Only stems >2 inches DBH were measured. Species, DBH, and fire damage of the bole were recorded. Bole fire damage was observed on the lower 2 feet of the bole and classified as superficial, moderate, or severe. All the stems received some superficial charring in the burn plots. Superficial damage was classified as fully intact bark completely encircling the bole and the cambium was not visually damaged. Moderate bole damage classification was that a fire scar had occurred and no visible wood decay was evident. Severe bole damage was that wood decay was visible inside the tree beyond the cambium.

Stand structural data and understory composition data were collected at different intervals for each burning treatment from earlier study reports and are available from DeSelm and others (1991), Nichols (1971), and Stratton (2007). This study reports only on the presence and condition of overstory trees after 54 years of different burning regimes.

RESULTS

Mean basal area of both annual burn (18.1 square feet per acre) and periodic 5-year burn (13.9 square feet per acre) treatments decreased from approximately 50 square feet per acre during the 54-year period since study inception in 1963 (table 1). Basal area of the control treatment increased from 50 square feet per acre to 78.7 square feet per acre in the absence of fire. The mean number of trees >2 inches DBH also decreased from an average of 120 trees per acre to 25.3 trees per acre in the annual burn treatment and to 15.3 trees

Table 1—Mean basal area, number of trees, and diameter (DBH) of trees >2 inches in diameter with standard errors for each treatment after 54 years for the long-term prescribed fire study in the Tennessee Oak Barrens, Tullahoma, TN^b

Treatment	Mean basal area (std error) ----ft ² /acre----	Mean number of trees (std error) ----trees/acre----	DBH of tree of mean basal area -----inches-----
Annual burn	18.1 (3.2)	25.3 (4.4)	11.4
Periodic 5-yr burn	13.9 (2.8)	15.3 (3.4)	12.9
Control (no burn) ^c	78.7 (5.2)	---	---

--- = Data not collected.

^a DBH = diameter breast height at 4.5 feet.

^b Based on fifteen 0.1-acre plots per treatment.

Table 2—Bole damage classification for each tallied tree for each burn treatment and both treatments combined after 54 years for the long-term prescribed fire study in the Tennessee Oak Barrens, Tullahoma, TN

Bole damage classification	Annual burn	Periodic 5-yr burn	Combined treatments
Superficial	2 (5%)	0	2 (3%)
Moderate	9 (22%)	5 (22%)	14 (22%)
Severe	29 (73%)	18 (78%)	47 (75%)
Total	40	23	63

per acre in the periodic burn treatment (table 1). The diameters of trees in both burn treatments ranged from 7 to 18 inches with a tree diameter of mean basal area of 11 to 13 inches (table 1). The burn treatments had very few trees with diameters between 2 and 6 inches DBH.

The lower boles of all sample trees for both burning treatments combined (n=63 trees) incurred fire damage from the regimes of repeated burning (table 2). More than 74 percent of the tree boles were classified as severe with wood decay visible. Moderate damage was observed on 22 percent of the sample trees. Only two trees had superficial damage; both were in the annual burn treatment. Both of these trees were located in wet depressions where fire intensity was less severe. None of the trees in the control plots had any visible fire damage.

DISCUSSION

With the burning regimes imposed in this study, the basal area and number of overstory trees gradually decreased throughout the 54 years. The gradual decrease in the overstory can be attributed to three sources: (1) bole damage and structural decay associated with the repeated burning, (2) blowdown from the 2011 tornado, and (3) tree senescence. The burn treatments accelerated the mortality rate as evidenced by the overall decrease in total basal area of the burn plots and the increased total basal area of the control

plots, both before and after the tornado event. All of the overstory trees sampled in the burn treatments were damaged by the fires with 74 percent of them classified as severe damage with internal wood decay (table 2). The 2011 tornado caused an increase in down woody debris and uprooting of some overstory trees; however, we did not quantify the amount or proportion in this study. Saturated soils from storms just prior to the tornado followed by 111–135 mile-per-hour winds (EF 2 tornado) (Edwards 2017) caused some trees in the control and burning treatments to uproot and blow over. The impact of the tornado winds appeared to be patchy across the study area with no tree blowdowns in several of the treatment plots. A few dead, standing trees also were observed in the treatment plots after the tornado. Many of the older trees were 80+ years at study inception in 1963, which would put them in excess of 130 years of age and in the later range of their life cycles, especially the red oaks. The longevity of oaks in the red oak family is much less than those in the white oak family (Dey and Schweitzer 2015, Johnson and others 2009).

The periodic burn treatment plots had fewer overstory trees than the annual treatment plots (table 1). The 5 years of growth between periodic burns resulted in greater fire intensity due to greater accumulation of fuels. The ring fire pattern would have also increased

the intensity of the burn in the center of the plot. The fuels buildup in the annual treatments was much less than the periodic treatment as a result of fire frequency. The differences in fuel buildup and resulting fire intensity differences created more intense fires in the 5-year periodic treatment, which accelerated tree damage and mortality.

The annual burning regime created a barrens or savanna structure with few or no woody stems in the understory (Stratton 2007). The periodic 5-year burn regime created a woody shrublike structure. The burning regime results are similar to that reported by Knapp and others (2015) in Missouri. The striking impact of these treatments is that few stems became ingrowth or tree recruitment for the overstory. None of the sample trees in either of the burn treatments were between 2 and 7 inches DBH. We did observe a few stems on the margin of the periodic burn treatment that could be future ingrowth, but intensity of burns on the margin of the plots was less than the center of the plots. Upon closer observation of these stems, they did incur fire scars from previous burns that could jeopardize their recruitment into the overstory.

The fire behavior and fire effects associated with the small scale of these plots, the implementation of the ring pattern for the burning treatments, and constant fire frequency regimes are quite different from the fire behavior and fire effects that would occur on larger scales. Large-scale burning results in more variable fire behavior and effects from differing amounts and distribution of fuels as well as other stand, weather, and topographic variables (Stambaugh and others 2016). Although it is impossible to separate the effects of the scale of the ring pattern, some broader fire implications are evident and would occur regardless of scale. Some stems escape burning coverage to become future ingrowth, especially in surface depressions or near riparian areas where soil moisture is greater. However, these conditions also tend to accumulate fuels which potentially create more intense future burns that would damage trees. Burning damage, whether from one or repeated fires, increases the potential mortality of overstory trees. Recruitment and ingrowth may prove challenging regardless of scale, particularly if fire intensity is not managed.

Under fire regimes imposed in this study, fire frequency and intensity are impeding ingrowth and therefore not replacing these senescent overstory trees. If a sparse overstory is desired for the future and ingrowth is not occurring, we recommend omitting fire from the area for a longer time interval to allow stems to become large enough to survive burning and/or decrease fire intensity (Dey and others 2017, Knapp and others 2015). Although the length of time required to allow ingrowth into the

overstory is unknown (Arthur and others 2012), that time period will be a function of fire intensity, build-up of fuels present, and the diameter of stems to survive the subsequent burning events. Instituting a longer burn interval until ingrowth occurs will promote more woody growth which will alter the community structure during that interval. The desired community structure can then be restored once more frequent burning is reinitiated. A different fire frequency regime or burn pattern(s) may be necessary and periodically adjusted to achieve desired results.

Frequent, high intensity burning to sustain barrens, woodlands, and savanna structures neither maintains the existing overstory stems nor promotes recruitment or ingrowth of young stems into the canopy. The stability of the overstory is at risk because of frequent, high intensity burning regimes that tend to damage these trees. The accumulation of damage from repeated fires gradually increases stem mortality and diminishes tree longevity. To promote trees to the overstory, burning should cease until ingrowth from existing stems is large enough to survive future burns. However, fire scars and decay associated with prescribed burning will always be a factor in the perpetuation and longevity of the sparse overstory in these systems suggesting that these disturbance-dependent communities may be more transitional and less enduring over time.

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PRELIMINARY COMPARISONS OF HERBICIDES AND APPLICATION PROCEDURES TO PROMOTE SIZE OF ADVANCED OAK AND YELLOW-POPLAR REPRODUCTION AFTER HARVEST

Stephen E. Peairs and Wayne K. Clatterbuck

Abstract—The repetitious use of diameter-limit harvesting in upland hardwoods has led to low-valued stands with heavy midstory and understory canopy layers containing mostly shade-tolerant species. Limited documentation is available on the means to successfully regenerate these impoverished areas into stands of more desirable, shade-intolerant species. The objective of this study is to evaluate the effectiveness of various herbicides and application methods to accelerate the growth (size) of small natural oak (*Quercus* spp.) and yellow-poplar (*Liriodendron tulipifera*) reproduction after harvest that otherwise would probably be outgrown by other undesirable species. The study area is in west-central Tennessee on the Western Highland Rim on nonindustrial forest land that has been cutover several times. Site index is 70 to 75 feet for upland oaks at 50 years, typical of many upland hardwood sites. Three 10-acre harvest blocks, each containing the following six treatments were established: banded foliar spray, banded foliar spray plus pre-emergent broadcast spray, radial release spray, radial release plus pre-emergent broadcast spray, pre-emergent broadcast spray only, and untreated control. Individual oak and poplar seedlings were measured after treatment applications in the fall of 2014 and again in January/February of 2017. Initial findings after two complete growing seasons indicate there was no difference in the natural reproduction growth response between treatments.

INTRODUCTION

Sustainable forestry requires that desirable timber species, typically shade-intolerant species, are successfully regenerated to form future stands. Mismanagement of forest land thru diameter-limit harvesting or high-grading leads to stand impoverishment when repetitiously implemented. Noss and others (1995) proposed that high-quality oak/hickory forests are on the decline in areas across the Central and Southern Appalachians. This degrade in quality is primarily attributed to species composition shifts resulting from diameter-limit harvesting. Trimble (1973) found that repeated single-tree selection and diameter-limit harvesting lead to a higher proportion of shade-tolerant species in the overall stand species composition along with a general reduction in species diversity. In such stands, foresters must take appropriate management actions to ensure the future stand will consist of higher proportions of shade-intolerant timber species. Establishment of adequate oak regeneration is also problematic in undisturbed and properly managed hardwood stands. Oaks (*Quercus* spp.) are some of the most difficult tree species to regenerate using common silvicultural practices and natural seed stock (Hannah 1987), even in favorably

stocked stands. Oaks are a highly preferred species but have proven difficult to successfully establish in future stands after a disturbance. The primary reason that oak regeneration fails to form dominance in the future stand is usually attributed to a lack of adequate advanced regeneration or the slow growth yielding oak less competitive compared to other plants to capture available growing space. Various silvicultural practices are typically required to enable oak to develop into competitive size classes. Silvicultural regeneration practices have been scientifically studied in order to formulate methodology to routinely establish competitive oak reproduction. Loftis (1990) advocates the removal of subcanopy basal area by the shelterwood method or herbicide application to enable a greater establishment of oak in the understory. In ideal instances, advance oak regeneration would be established in oak-dominated stands prior to overstory removal in such fashion. The advance regeneration should be of adequate size as suggested by Sander (1971) whom advised that advance reproduction is an imperative for new oak sprout growth following clearcutting. His study shows that oak sprout growth was related to ground line diameter of the old stem in that larger stems resprouted and grew at faster rates. The most optimal size were stems between 0.5

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to 1 inch in diameter as these stems were able to attain a position in the dominant canopy and produced fewer sprouts per stem compared to regeneration >1 inch. For mismanaged stands that lack adequate parent oak stems to supply reproduction, managers must seek other options to successfully establish oak regeneration.

Early chemical release treatments have shown some potential to assist in improving oak regeneration success. A study by Thompson and Nix (1992) found that early crop tree release within a four year old clearcut using various herbicides significantly decreased herbaceous and woody plant competition. This reduction in competition resulted in increased seedling ground line diameters but did not improve height growth compared to control treatments. Nix (2004) remeasured the released natural oak in the clearcut ten years after the initial chemical release treatment. The follow up study found that four herbicide treatments significantly increased diameter growth of released oak seedlings. The researcher suggests that applying herbicide release treatments assists in enabling desirable oak to form dominance in the overstory canopy. This study investigates if herbicide applications applied during the initial growing season after a silvicultural clearcut applied to an impoverished stand can be implemented to enhance natural regeneration.

METHODS

Study site

The study site is located on former Mead-Westvaco land situated along the Humphreys and Houston County boundary line in west central Tennessee. One of the replicated blocks is located in Humphreys County, and the other two are in Houston County. All blocks are positioned on north and northeast facing aspects beginning near the top of the ridge and extended down to the midslope position. Soil types present in each block are Bodine cherty silt loams with estimated site index values of 70–75, base age 50 for upland oak species. The stand prior to the harvesting disturbance was considered as degraded due to multiple diameter-limit harvesting entries. Species composition primarily included white oak (*Q. alba*), hickory (*Carya* spp.), red oak (*Q. rubra*), blackgum (*Nyssa sylvatica*), yellow-poplar (*Liriodendron tulipifera*), chestnut oak (*Q. montana*), ash (*Fraxinus* spp.), sugar maple (*Acer saccharum*), and black cherry (*Prunus serotina*). A well-established understory and midstory of moderate to large saplings were present. Common species recorded during pre-harvest inventory in the larger size reproduction classes included eastern hophornbeam (*Ostrya virginiana*), yellow-poplar, blackgum, ash, hickory, American beech (*Fagus grandifolia*), and flowering dogwood (*Cornus florida*).

Study design

The study incorporates a randomized complete block sampling design. Three individual blocks were replicated on sites with uniform site productivity. These blocks received silvicultural clearcuts in early spring of 2014. Six individual treatment units approximately ¾-acre in size were implemented within each block:

1. Chemical seedling release treatments utilizing sulfometuron methyl (SFM 75® by Alligare LLC) only
2. Alternating banded strip treatment utilizing foliar sprays of glyphosate
3. Radial spray release utilizing foliar sprays of glyphosate
4. Alternating banded strips plus release using sulfometuron methyl
5. Radial sprays plus release using sulfometuron methyl
6. Untreated control

The three units that received sulfometuron methyl treatments were applied in May–June of 2014. Glyphosate applications were conducted between July–August of 2014. Banded sprays were approximately 4 feet in width and alternated treated (foliar sprayed) strips and untreated strips across the unit. Radial foliar sprays were implemented around individual oak and yellow-poplar seedlings using glyphosate only. Radial sprays treated vegetation in the surrounding area of approximately a 5-foot radius from the sample seedling. A stove pipe apparatus covered the seedling being released to protect foliage from incidental contact with herbicide solution. The banded and radial spray methods were also applied to two of the blocks that were previously sprayed with sulfometuron methyl. The final treatment unit did not receive any herbicide applications. These treatment units are labeled as SFM 75 only, radial, banded, radial plus SFM 75, banded plus SFM 75, and untreated control.

Approximately 150 naturally regenerated seedlings (approximately half of the population were oak species and half were yellow-poplar) for all treatment units over the three replicated blocks were measured in the fall of 2014 for ground line diameter and overall height. Ground line diameter was measured with digital calipers with accuracy to one-hundredth of an inch. Height measurements were taken with a standard retractable ruler to the nearest half an inch. All measurements were taken after completion of treatment applications during the dormant season. A total of 2,653 seedlings were measured on the site. Seedlings were measured again

after two full growing seasons had elapsed in January/February of 2017. Only 1,563 seedlings were able to be relocated for measurement due to the robust response of warm-season grass vegetation. Measurement procedures were repeated in similar fashion as were performed during the initial measurements. The difference from the 2017 measurements minus the 2014 measurements was statistically analyzed for both diameter and height growth.

Statistical Analyses

Statistical analyses were performed for analysis of variance (ANOVA) using mixed models (SAS Institute Inc., Cary, NC version 9.4). Data tests indicated satisfactory normality and equal variances. No transformations were utilized in the analyses. Tukey's significant difference test was incorporated to separate the least squares means. The significance level was set at $\alpha = 0.05$. All sampled seedlings, regardless of species, were combined by treatment for analysis to compare treatment means.

RESULTS

Ground line diameter growth means (combined oak and yellow-poplar seedlings) following two growing seasons for individual herbicide treatments were as follows: 0.360 inches for banded, 0.318 inches for banded plus sulfometuron methyl, 0.310 inches for control, 0.355 inches for sulfometuron methyl only, 0.303 inches for radial release, and 0.339 inches for radial plus sulfometuron methyl. No significance difference was revealed for change in diameter growth between treatments ($p = 0.74$) (table 1).

Height growth means for the banded, banded plus sulfometuron methyl, untreated control, sulfometuron methyl only, radial release, and radial plus sulfometuron methyl were 24.4, 21.5, 23.5, 27.1, 17.2, and 21.7 inches, respectively. Findings show a non-significant difference between treatments ($p = 0.058$) (table 2).

Competing plant competition was substantial after two complete growing seasons (table 3). Ocular estimations revealed that all but three out of eighteen treatment units had 50 percent or greater coverage by broomsedge bluestem (*Andropogon virginicus*) and Nepalese browntop (*Microstegium vimineum*). All treatments displayed high levels of grass establishment on at least one of the three replications. Banded glyphosate with sulfometuron methyl applications received the greatest amount of herbicide active ingredient but had the highest rates of grass coverage.

DISCUSSION

Results indicate that initially there was no apparent response between herbicide treatments and the untreated control after two growing seasons. The lack of response may be attributed to below average

precipitation over the course of active plant growth since herbicide applications were completed. Seven of the 12 months (April–September) showed below average rainfall for the 2015 and 2016 growing seasons according to National Oceanic and Atmospheric Administration (NOAA) data. Of particular interest is the period during April and May of the 2016 growing season. Monthly rainfall for these months was 4.05 inches and 3.16 inches below the average (dating back to the year 2000). This season of the year is when tree diameter growth or the early wood should be growing at optimal levels. The reduction in moisture available for uptake by trees probably affected plant growth. If these spring months had a normal or above average rainfall, seedling growth response may likely have been different amongst treatments.

Another plausible explanation may be increased competition from grasses following the harvest disturbance. The high establishment rate of grasses suggests that the reduction of forbs through herbicide applications essentially released grasses from suppression. The tremendous emergence of warm season grasses, primarily broomsedge and Nepalese browntop likely hindered tree seedling development. Grasses have dense, fibrous root systems which compete for space in the same soil horizon as new tree seedlings. Available soil moisture is likely intercepted by the fibrous grass roots limiting growth potential for tree seedlings. The ability for a given tree to expand its root system is also impacted by the abundance of grass roots. The result of this competition by grass is a reduction in above ground height growth with growth instead allocated more towards root biomass (Collet and others 2006, Harmer and Robertson 2003). The study also suggests that total root length and the number of root tips decrease with increasing competition. High-density assemblages of grass root contributed to nutrient and water depletion. The diminution of resources directly leads to condensed seedling root growth (Collet and others 2006). Thus, seedling growth may have been directly influenced by the dense establishment of grasses on the study site. Treatments that received higher intensity herbicide applications using both sulfometuron methyl and glyphosate had a higher percentage of coverage by grasses based on ocular estimations. A secondary treatment during the growing season of 2015 to control the emerging grass may be beneficial to enhance seedling growth. Additional treatments however may prove unfavorable economically to private landowners although sufficient control of broomsedge can be achieved by applying bromacil, diuron, tebuthuron, buthidazole (Griffin and others 1988), and glyphosate (Butler and others 2002). Perhaps delaying applications until after a full growing season may yield a contradictory outcome as revealed by this study. The probability is high however that grass would re-emerge from seed stock even with a change

Table 1—Tukey mean separation results amongst combined treatments for seedling ground line diameter growth

Treatment	Observations	Mean diameter (inches)	Standard error	Letter group
Banded	265	0.360	0.015	A
Banded + SFM 75	263	0.318	0.015	A
Control	294	0.310	0.014	A
SFM 75 only	271	0.355	0.014	A
Radial	248	0.303	0.015	A
Radial + SFM 75	222	0.339	0.015	A

Table 2—Tukey mean separation results amongst combined treatments for seedling height growth

Treatment	Observations	Mean height (inches)	Standard error	Letter group
Banded	265	24.4	1.042	A
Banded + SFM 75	263	21.5	1.048	AB
Control	294	23.5	0.996	A
SFM 75 only	271	27.1	1.073	A
Radial	248	17.2	0.889	B
Radial + SFM 75	222	21.7	0.936	AB

in application timing. Given this conclusion, it may be best to accept the coexistence of both vegetation types. Eventually, the released oak and yellow-poplar will develop and create shaded environments that diminish the presence of the grasses. The projected enhanced rate of growth after two growing seasons did not occur based on current findings.

CONCLUSION

This study initially suggests that an early precommercial thinning to newly established reproduction after two growing seasons may be unnecessary to ensure successful establishment of desirable oak and yellow-poplar seedlings in the future overstory. The initial herbicide treatments did assist in controlling the establishment of shade-tolerant species regeneration, but the consequence was promotion of thick grasses rather than increased growth of oak and yellow-poplar seedlings. However, in subsequent years, we hypothesize that these early silvicultural release treatments to promote oak and yellow-poplar through the use of herbicides may create a greater abundance of these species in the overstory at crown closure. The abundant shade-tolerant species with the potential

to become part of the overstory or influence the development of more desirable, shade-intolerant species were controlled by these herbicide treatments, even though these treatments did promote the proliferation of grasses. This result will be attributed to the increased amount of growing space available for crown development created from the early thinning. In addition, should adequate or surplus rainfall occur over the next couple of growing seasons, a difference may be noticeable between the treatments. Future data to support or renounce these theories will be acquired in upcoming years.

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Table 3—Ocular estimates of vegetative ground cover percentages by individual treatment units

Block	Treatment	Vegetative cover type				
		Broomsedge	Microstegium	<i>Rubus</i> sp.	Leaf litter/ saplings	Herbaceous/ sparse grass
A	Control	***	40%	60%	***	***
B	Control	95%	***	5%	***	***
C	Control	80%	***	5%	***	15%
A	Control + SFM 75	40%	10%	50%	***	***
B	Control + SFM 75	60%	***	25%	***	15%
C	Control + SFM 75	95%	***	***	***	5%
A	Radial	10%	40%	30%	20%	***
B	Radial	15%	***	85%	***	***
C	Radial	90%	***	10%	***	***
A	Radial + SFM 75	60%	***	***	20%	20%
B	Radial + SFM 75	50%	***	5%	45%	***
C	Radial + SFM 75	85%	***	15%	***	***
A	Banded	10%	75%	15%	***	***
B	Banded	15%	***	70%	15%	***
C	Banded	85%	***	15%	***	***
A	Banded + SFM 75	95%	***	5%	***	***
B	Banded + SFM 75	95%	***	5%	***	***
C	Banded + SFM 75	70%	***	***	***	30%

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ERRATA

The author found a significant difference among treatments after submission of this article. These findings will be made known to the scientific community at a later date.

RADIAL GROWTH RESPONSES OF UPLAND OAKS FOLLOWING RECURRENT RESTORATION TREATMENTS IN NORTHERN MISSISSIPPI

Kathryn R. Kidd, J. Morgan Varner, and J. Stephen Brewer

Abstract—Fire exclusion over the past century has substantially altered composition, structure, and fuel dynamics in upland oak-hickory (*Quercus-Carya*) forests in the Southeastern United States. Numerous restoration efforts have been made to re-establish historical disturbance regimes into these altered forests. However, our understanding of the implications of restorative disturbances on stand dynamics has primarily been limited to shifts in species composition and post-disturbance regeneration. Therefore, we examined annual radial growth responses of dominant upland oaks following a combination of prescribed fires (2004, 2006, 2008, 2010, 2012, and 2014) and thinning (starting in 2004) treatments (thin+burn) in stands which had previously been unburned since the early 1900s. Radial stem growth rates were quantified using tree cores from 22 post oak (*Q. stellata*) and southern red oak (*Q. falcata*) in a 2.5-acre thin+burn and control stand at the Strawberry Plains Audubon Center in northern Mississippi. Radial growth rates were not significantly greater following repeated thinning and prescribed burning than prior to treatment initiation for either post oak or southern red oak. For the first 6 years after the initial thin, the annual ring width for southern red oak was identical in the thin+burn ($1.9 \pm 0.1 \text{ mm year}^{-1}$) and control ($2.0 \pm 0.2 \text{ mm year}^{-1}$) stands. However, in 2010 radial growth for southern red oak in the thin+burn increased such that the annual ring width for 2010 was 22 percent greater in the thin+burn than in the control stands. In contrast to the positive growth response in southern red oak (2 percent), post oak demonstrated a significantly different ($p = 0.014$) negative response (-19 percent) in the relative percent change in total radial growth for the 11-year period post-treatment initiation when compared to the 11-year period prior to treatment initiation. Radial growth for both species was negatively impacted by a severe drought in 2007 with southern red oak exhibiting the greatest decrease in radial growth. Results from this study highlight the underlying role of climatic factors and species life history characteristics in evaluating radial growth patterns following forest disturbances.

INTRODUCTION

Long-term fire exclusion coupled with the absence of significant harvesting disturbances has created contemporary forested conditions in upland oak-hickory (*Quercus-Carya*) forests in the Eastern (Nowacki and Abrams 2008) and Western United States (Cocking and others 2012). Such contemporary forests in the Southeastern United States are characterized by altered composition (i.e., an increase in frequency of fire-sensitive, shade-tolerant mesophytic species), forest stand structure (i.e., increased stem densities), and fuel dynamics (i.e., fuel beds less conducive to facilitating fire disturbances) (Cocking and others 2012, Kreye and others 2013, Nowacki and Abrams 2008). The reintroduction of disturbance into these now contemporary forested conditions leaves many questions unanswered relative to the impacts on forest structure and composition, radial stem growth, and forest health, particularly where there is strong potential

for interactions with climatic factors (e.g., extreme fluctuations in temperature and precipitation).

Radial stem growth patterns recorded in tree-ring records can be analyzed to reflect environmental growth conditions and thus can be used to evaluate trees' responses to the reintroduction of disturbance along with climatic influences on growth (Fraver and White 2005, Fritts and Swetnam 1989, Rentch and others 2002). Annual radial stemwood growth (i.e., annual ring width) is restricted by the most limiting factor (Fritts 1976). For instance, increased competition from high stem densities can reduce light, water, and nutrient availability, thus reducing radial stemwood growth for the duration of such stressed conditions. When conditions improve (e.g., increased light, water, or nutrient levels), an increase in radial stemwood growth or a growth release will occur in response to the most limiting factor no longer limiting radial growth.

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In 2003, an oak woodland restoration project was established at the Strawberry Plains Audubon Center (SPAC) in northern Mississippi (Brewer 2014). Recurrent thinning and prescribed burning treatments were implemented to restore historic oak woodland conditions in a long-undisturbed (since early 1900s) upland oak-hickory forest dominated by southern red oak (*Q. falcata*) and post oak (*Q. stellata*). Restoration treatments were designed to increase light availability by decreasing overstory and midstory stem densities and to promote regeneration by more shade-intolerant, fire-tolerant species by reducing the competition from more fire-sensitive, shade-tolerant mesophytic species. The reintroduction of disturbance into this long-disturbed contemporary forest provided the opportunity to examine radial stemwood growth responses to the recurrent disturbances. Therefore, the objectives of this study were to 1) determine if southern red oak and post oak demonstrated a radial stemwood growth response to repeated thinning and prescribed fire treatments and 2) determine if radial stem growth responses differed between the two upland oak species.

MATERIALS AND METHODS

Study Area

This study was conducted at the SPAC in northern Mississippi. Specifically, the SPAC is located in Marshall County, MS, approximately 10 km north of Holly Springs, MS and 80 km southeast of Memphis, TN (34°49' N, 89°28' W). The study site consisted of two adjacent 2.5-acre upland oak-hickory stands situated on a Providence silt loam soil. From the mid-1800s to the early 1900s the area was intensively managed for cotton but since, these upland oak-hickory stands have not been burned or received any significant disturbance (Surrette and others 2008). Upland sites in this region were characterized as oak woodlands before conversion for agriculture (Brewer 2001, Surrette and others 2008). However, since previous agricultural use, long-term effects of fire exclusion and absence of harvesting disturbance created contemporary conditions which were no longer conducive to frequent, low-intensity disturbance (Nowacki and Abrams 2008).

Treatments

The two adjacent stands represented a paired design; one stand was designated as a control (no treatment) and one as a thin+burn (recurrent thinning and burning). This pair of oak-hickory stands was part of a larger study established in 2003 aimed at restoring historic oak woodland conditions (Brewer 2014, Brewer and Menzel 2009). The original study included a total of two replicated pairs (one control and one treatment stand). Our current study utilized only one of the pairs due to differences in species composition, soils, and site hydrology between the pairs. In 2003, all stems with a diameter at breast height (DBH, 4.5 feet above

ground) >4 inches were tagged and inventoried. The overstory in both the control and thin+burn stand was initially dominated by post oak and southern red oak. In 2004, an initial thinning was conducted in the thin+burn treatment stand. A combination of girdling via Pathway® (picloram and 2,4-D) and felling techniques were used to primarily target the fire-sensitive, mesophytic non-oak species such as red maple (*Acer rubrum*), sweetgum (*Liquidambar styraciflua*), winged elm (*Ulmus alata*), blackgum (*Nyssa sylvatica*), and dogwood (*Cornus florida*). To further increase canopy gaps and amount of light reaching the forest floor, both of which are characteristics of fire-prone oak woodlands, smaller-scale thinning disturbances were implemented in 2008, 2010, and 2012 in which an additional 10 percent of the canopy cover was thinned via girdling (Brewer 2014). By 2015, the basal area decreased from 110 to 75 square feet per acre and stand density decreased from 123 to 54 stems per acre in the thin+burn. Post oak and southern red oak composed the majority of the overstory with mockernut hickory (*Carya tomentosa*) as a minor component. In combination with thinning disturbances, prescribed fires were conducted in September 2004, October 2006, July 2008, April 2010, March 2012, and March 2014 to aid in reducing competition from more fire-sensitive species and to promote native grasses and fuel dynamics associated with historic oak woodland conditions. In the absence of thinning and burning disturbances, the control stand was representative of a typical eastern deciduous contemporary forest, in which the midstory composition consisted of mesic shade-tolerant, fire-intolerant species with little to no seedlings or shrubs (Nowacki and Abrams 2008). In the control stand, basal area increased from 100 to 110 square feet per acre and stand density increased from 122 to 143 stems per acre between 2003 and 2015. Thus, in 2016, when our sampling occurred, the control stand was characterized by a greater component of red maple, sweetgum, winged elm, blackgum, and dogwood in the mid- and overstory strata than in the thin+burn stand. In contrast, the thin+burn stand contained more shade-intolerant and fire-tolerant species (i.e., *Quercus* spp.) in the midstory and seedling regeneration layers.

Field Methods

During the summer of 2016, 11 southern red oak and post oak trees were selected within each of the 2.5-acre thin+burn and control stands. Two tree cores were extracted from each selected tree at DBH using a manual increment borer. Cores were taken 90° apart from each other around the circumference of the tree. Two cores were collected rather than one to reduce within-tree variation (Copenheaver and others 2009). Trees located at stand boundaries were not selected in order to avoid potential edge effects. Trees that displayed wounds or broken tops were omitted.

Laboratory Methods

Tree cores were air dried and glued to wooden mounts. Progressively finer sand paper was used to surface cores until cellular structures became visible in the cross-sectional view under magnification (Phipps 1985). Two tree-ring chronologies were developed: 1) southern red oak and 2) post oak. Each chronology was developed using all 22 trees (44 cores) from across both stands (5 acres). This was done to ensure the standard minimum of 20 trees was used to develop each chronology (Copenheaver and others 2009). Prior to treatment implementation there was no significant disturbance in either stand, therefore the trees should have been exposed to the same environmental conditions during the formation of the majority of annual growth rings. Cores were visually cross-dated using the list method in which narrow growth rings common among samples were identified and used as signature years to ensure proper alignment of dating (Yamaguchi 1991). Annual tree-ring widths were measured under stereoscopic magnification to the nearest 0.01 mm using the Velmex Measuring System and J2X software (v.3.2.1, 2004). Dated tree-ring width measurement values were verified to ensure quality of visual cross-dating using COFECHA software (Holmes 1983). Dating errors detected by COFECHA were corrected.

Data Analysis

To determine if changes in radial stem growth occurred following implementation of recurrent thinning and burning treatments, mean annual ring width and total radial growth increment for the 11 years prior to and following treatments were compared. Comparisons were made between pre- and post-treatment values (mean ring width and total 11-year radial growth) for each individual species using Wilcoxon-Mann-Whitney tests within the NPAR1WAY procedure in SAS 9.3 (SAS 2012). To determine if radial growth responses differed between species, the relative percent growth change in total 11-year radial growth increment from pre- to post-treatment implementation was calculated for southern red oak and post oak (adapted from Nowacki and Abrams 1997). Relative percent change in growth was compared between southern red oak and post oak in both the control and thin+burn stands using a Wilcoxon-Mann-Whitney test. All tests were performed at a significance level of $\alpha = 0.05$.

RESULTS AND DISCUSSION

Tree-Ring Chronologies

The southern red oak tree-ring chronology included years of 1910 to 2015 (mean length of series = 68.4 years) while the post oak chronology spanned from 1853 to 2015 (mean length of series = 110.5 years). Series intercorrelation values were 0.700 for the southern red oak and 0.669 for the post oak chronologies indicating a relatively strong degree of correlation (ranging from

0-weak to 1-strong) for interannual ring widths and thus cross-dating among samples (series). Mean sensitivity values, which provide a year-to-year measure of variability in ring width, were 0.207 for southern red oak and 0.205 for post oak indicating little variability in ring width for the length of the chronologies.

Pre- and Post-Treatment Radial Growth Responses Compared

Prior to implementation of recurrent thinning and burning treatments, mean annual ring width for southern red oak for the time period 1992 to 2003 was similar ($p = 0.922$) between the thin+burn ($2.2 \pm 0.1 \text{ mm year}^{-1}$; mean $\pm$ SE) and control ($2.3 \pm 0.3 \text{ mm year}^{-1}$) stands (fig. 1). Radial growth rates for post oak were also relatively similar ($p = 0.309$) in the thin+ burn ($1.2 \pm 0.2 \text{ mm year}^{-1}$) and control ($1.5 \pm 0.2 \text{ mm year}^{-1}$) stands prior to treatments (1992 to 2003). An increase in post oak radial growth occurred in the control stand around 2001, after which the mean annual ring width was approximately 25 percent greater for post oak in the control than in the thin+burn stand. For the first six years (2004 to 2009) after the initial heavy thin, the mean ring width for southern red oak remained nearly identical in the thin+burn ($1.9 \pm 0.1 \text{ mm year}^{-1}$) and control ($2.0 \pm 0.2 \text{ mm year}^{-1}$) stands despite the additional 10 percent basal area thinning in 2008 and prescribed fires in 2004, 2006, and 2008. However, in 2010 radial growth for southern red oak in the thin+burn increased such that the mean annual ring width for 2010 was 22 percent greater in the thin+burn than in the control stands. Radial growth rates for southern red oak continued to be greater ($p = 0.140$) in the thin+burn ($2.6 \pm 0.2 \text{ mm year}^{-1}$) than in the control ($2.0 \pm 0.2 \text{ mm year}^{-1}$) from 2010 through 2013. Post oak radial growth was consistently greater ($p = 0.033$) in the control ($1.4 \pm 0.2 \text{ mm year}^{-1}$) than in the thin+burn ($1.0 \pm 0.1 \text{ mm year}^{-1}$) stand during the 2004 to 2015 time period (fig. 1). The relative percent change in the total radial growth for the 11-year period prior to (1992 to 2003) and following (2004 to 2015) initiation of treatments was not significantly different between the control and thin+burn stands for southern red oak ($p = 0.224$) or post oak ($p = 0.053$) (fig. 2).

Although changes occurred in radial stemwood growth, we were unable to detect statistically significant growth responses for southern red oak or post oak through analysis of mean annual ring width or total radial growth increments between pre- and post-initiation of thinning and burning treatments. The most likely reason for the lack of significant response to treatment is the underlying role of climate on radial growth responses. Radial growth is limited by the most limiting factor, and in our study this may have been water availability rather than light. Climate records indicate a severe drought occurred in 2007 (NOAA 2017). Coincidentally, annual ring width decreased in both the thin+burn and control stands likely masking any immediate growth response in the

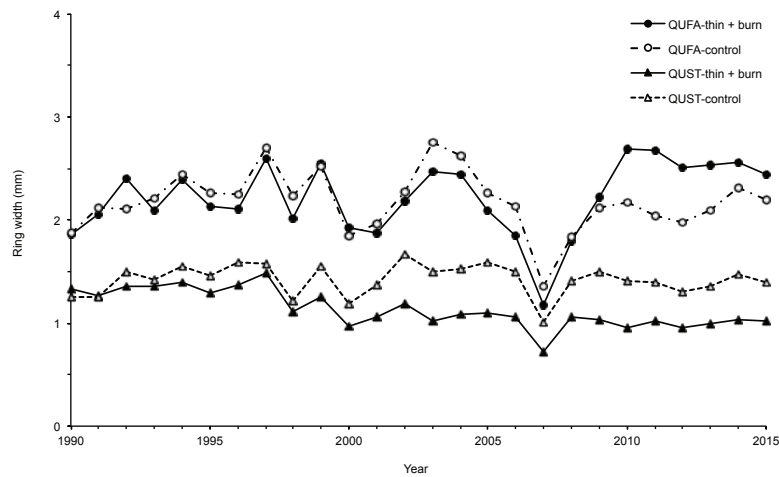


Figure 1—Mean annual tree-ring width (mm) for southern red oak (QUFA) and post oak (QUST) within the thin+burn and control stands for the 1990 to 2015 time period at the Strawberry Plains Audubon Center in northern Mississippi. Twenty-two QUFA and QUST tree cores (11 trees) were analyzed from within each stand. Recurrent thinning and prescribed fire treatments in the thin+burn were initiated in 2004.

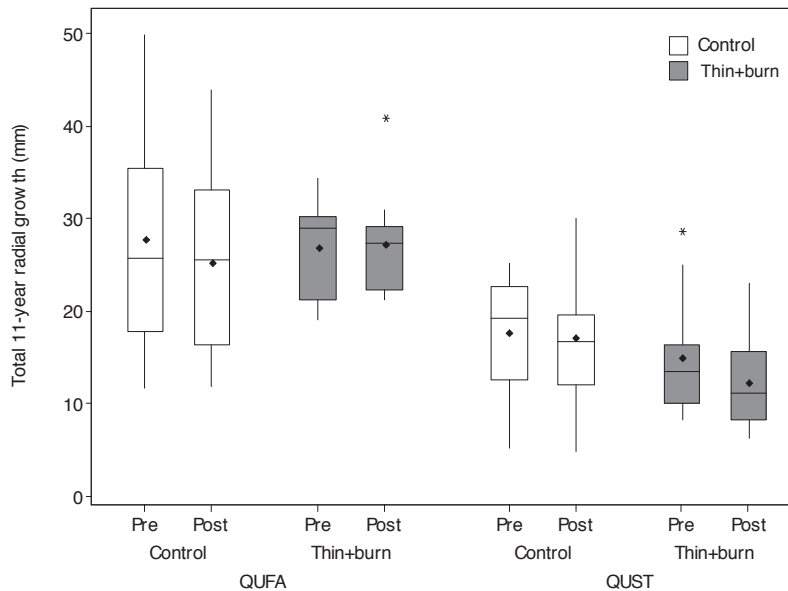


Figure 2—Boxplot of total radial growth (mm) for southern red oak (QUFA) and post oak (QUST) within the thin+burn and control stands for each 11-year period prior to (Pre; 1992 to 2003) and following (Post; 2004 to 2015) implementation of recurrent thinning and prescribed fire treatments at the Strawberry Plains Audubon Center in northern Mississippi. Twenty-two QUFA and QUST tree cores (11 trees) were analyzed within the thin+burn and control stands. Diamond markers indicate mean values.

thin+burn stand. In 2007, the monthly Palmer Drought Severity Index was -3.34 for May, -3.24 for June, -2.35 for July, and -2.93 for August on a scale of +6 (extreme wet spell) to -6 (extreme drought) (NOAA 2017). Another reason for the lack of significant growth responses could be related to changes in physiology as trees mature in age and increase in diameter. Such changes in physiology may make mature trees more resistant to short-term alterations in environmental conditions than

younger trees (Liñán and others 2012, Voelker 2011). Further, the relatively low intensity and spatial pattern (asynchronous) of the additional 10 percent thinning treatments in 2008 and 2010 may have contributed to the lack of a significant radial growth response following initiation of restoration treatments. Gap sizes adjacent to selected trees were measured; however, data was not analyzed in this preliminary report.

Radial Growth Responses Compared Between Species
The relative percent growth change in total radial growth for the 11-year period prior (1992 to 2003) compared to following treatment initiation was significantly different ($p = 0.014$) between southern red oak (increased by 2 percent) and post oak (decreased by 19 percent) in the thin+burn stand (fig. 2). In the control stand, total radial growth during 2004 to 2015 decreased for both ($p = 0.450$) southern red oak (6 percent) and post oak (2 percent) when compared to the total radial growth during 1992 to 2003.

Southern red oak exhibited a slightly positive radial growth response in the thin+burn stand whereas post oak demonstrated a negative growth response. Southern red oak also demonstrated a greater growth response to the 2007 drought (fig. 1). Post oak appeared to be more resistant to thinning, burning, and climatic disturbances in our study. Differences in growth responses are most likely attributed to species' life history strategies and characteristics. Longevity has been shown to be approximately twice as long (320 years) for post oak than southern red oak (150 years) (Guyette and others 2004). Other researchers have identified slower growth rates as tradeoffs to ensure longevity in post oak (Guyette and others 2004). Differences in species responses could have also been due to physiological effects of tree age and size. On average, post oaks analyzed in this study were older and larger in diameter than the southern red oaks. The increased age and size of post oak may have reduced the radial growth response identified. Ring widths were not standardized prior to analysis in our current study, as we were directly comparing the 11 years of growth prior to and following treatment implementation and the percent change in growth between the two time periods (adapted from Nowacki and Abrams 1997). Further, standardization of radial growth may have decreased the tree-level response and thus we may have reduced the variation to the point where possible responses were not detected. To account for diameter size-related growth trends in future analyses, we will use the basal area increment, which takes into account the diameter rather than only changes in raw ring width and 11-year radial growth increments.

CONCLUSION

Identifying growth releases following known disturbances will aid future use of radial growth patterns to detect the occurrences and impacts of canopy disturbances, particularly smaller-scale disturbances. Our study recognizes the role that stand disturbances (e.g., repeated thinning and burning), climate, tree age and size, and species life history strategies play in influencing radial growth responses.

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Invasive Species

Moderator:

Rod Will

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PLANT COMMUNITY RESPONSE TO THE MANAGEMENT OF AN INVASIVE TREE

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Abstract—We designed several treatments to directly control an invasive tree, Chinese tallow [*Triadica sebifera* (L.) Small] on Parris Island Marine Corps Recruit Depot located in Beaufort, SC. We examined the response of the plant community to four treatment series: 1) control (C), 2) mastication (M), 3) fire (F), and 4) combination of mastication and fire (MF). We found that mastication significantly reduced midstory density of all species. However, without fire, midstory density increased to levels similar to those without mastication within 2 years. The MF treatment reduced midstory density over the course of the study, resulting in an increase in important oak species. The MF treatment also increased ground flora richness, without reducing the richness of regenerating tree species. Our results show that MF resulted in a positive response from the native community.

INTRODUCTION

Forest ecosystems are under increasing stress from environmental change, including invasion by nonnative species that can disrupt important ecological functions and services. Nonnative species invasions may interfere with maintaining desired vegetation types by altering post-disturbance succession and keeping communities in persistent undesirable states (D'Antonio and Chambers 2006), and the capacity of an ecological community to resist invasion is an important ecosystem service (Foster and others 2015). Invasive species management approaches should be designed and implemented with consideration of the processes that have historically characterized the ecological system as these may be important to building resistance in the community. Often, maintaining or restoring the structure and ecosystem processes known to favor resident species can be used to increase community resilience and resistance and thereby reduce invasion potential (D'Antonio and Chambers 2006). However, it is important to understand how invasive species management that includes changing community structure and re-establishing ecological processes impacts native species diversity.

The southeastern maritime forest has been historically characterized by a fire return interval from between 2 and 26 years, depending on the topographic situation and ignition sources (Frost 2005). On islands, the combination of lightning and burning by Native

Americans may have produced a fire return interval of 5 to 7 years (Frost 2004). Increased fire frequency in maritime forests also increases herbaceous diversity (Frost 2005). Fire suppression and logging have led to the conversion of two-layered forests with open understories to nearly impenetrable thickets of dense multistoried woody vegetation with essentially no herbaceous plants (Frost 2005), similar to the conditions seen in this study prior to treatment. Fire is also an important component of managing southern U.S. pine forests with open-stand structures and herbaceous understories. At Parris Island Marine Corps Recruit Depot (referred to as “Parris Island” hereafter), slash pine plantations were established in the 1970s and managed infrequently with fire resulting in dense, shrub-dominated understories that have been readily invaded by Chinese tallow [*Triadica sebifera* (L.) Small] (Pile and others 2017a), an aggressive invader of coastal forests that is considered sensitive to repeated growing season prescribed fire (Bruce and others 1997, Grace and others 2005).

The objective of our study was to determine the community response to treatments for the control of Chinese tallow. Specifically, we tested the response of native species to treatment effects by measuring their abundance and establishment prior to and following each treatment as well as the cumulative treatment effect. Treatments were designed to reduce the density of Chinese tallow and to create open-stand forest

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structure with the use of prescribed fire to enhance and promote native species adapted to frequent surface fire regimes. We applied four treatment types, including: 1) a control (C) with herbicide applied only to target Chinese tallow replicating prior management actions at Parris Island; 2) a fire (F) treatment with herbicide applied to Chinese tallow, followed by a growing season prescribed fire; 3) a mastication (M) treatment to reduce midstory density followed by an application of herbicide to target regeneration of Chinese tallow; and 4) a combined mastication and fire (MF) treatment which was similar to the M treatment, but with the addition of a growing season prescribed fire. We hypothesized that: 1) mastication would effectively reduce shrub density, resulting in increased ground layer richness and woody species regeneration by increasing available resources; 2) mastication would promote fire behavior and spread by increasing surface fuel loading and air flow; 3) fire alone would reduce shrub density, but not to the extent of the mastication treatment; and 4) the combination of mastication and fire would have longer-lasting effects on reducing shrub density while also favoring native, fire-adapted herbaceous species.

METHODS

Study Site and Background

The study was conducted at Parris Island located in Beaufort County, SC. Parris Island began management to control Chinese tallow in 2001 using herbicide. In 2011, we designed a research project to investigate several treatments to control Chinese tallow with an emphasis on its physiology (Pile and others 2017b). Specifically, we were interested in testing Chinese tallow response to treatments that included prescribed fire, as this species is suspected to be intolerant to repeated growing season prescribed fires (Grace and others 2005).

Parris Island is located on the Southeastern Coastal Plain ecoregion (EPA 2013), with relatively flat topography ranging from 0 to 7 m above mean sea level. The area is characterized by mild winters and hot summers, with soils that are generally described as fine sands to fine loamy sands. The island is comprised of 3257 ha, of which 608 are managed forest lands that are dominated by mature slash pine (*Pinus elliotii* Engelm.) plantations and natural mixed maritime hardwood forests. The hardwood forest component includes live oak (*Quercus virginiana* Mill.), Darlington oak (*Q. hemisphaerica* W. Bartam ex Willd.), cherrybark oak (*Q. pagoda* Raf.), water oak (*Q. nigra* L.), green ash (*Fraxinus pennsylvanica* Marshall), sweetgum (*Liquidambar styraciflua* L.), and red maple (*Acer rubrum* L.). Prior to treatment, a dense midstory of wax myrtle [*Morella cerifera* (L.) Small] and yaupon (*Ilex vomitoria* Aiton.) with a sparse understory of herbaceous plants represented the forest understory. Nonnative, invasive

species other than Chinese tallow were also present on Parris Island prior to treatment [i.e., tree of heaven [*Ailanthus altissima* (Mill.) Swingle], chinaberry [*Melia azedarach* L.], lantana [*Lantana camara* L.], Japanese honeysuckle [*Lonicera japonica* Thunb.], and Japanese stiltgrass [*Microstegium vimineum* (Trin.) A. Camus]] but were not in significant densities to require specific control efforts. However, by reducing understory density and restoring fire as an ecological process we believed these treatments would indirectly reduce establishment and further recruitment of these nonnative invasive species.

Study Design

This study was conducted with a randomized, complete block design, blocked by forest stand with eight replicates and four treatment combinations, resulting in 32 experimental units. Treatment areas ranged from 0.5 to 2 ha in size. Mastication was applied with a Caterpillar model HM315 roller chopper with carbide teeth in the M and MF treatment areas in May of 2013. Mastication was followed in late summer of the same year by a targeted foliar application of 2.5 percent v/v Garlon 4 Ultra ® herbicide and water to kill any regrowth (i.e., basal sprouts, stump sprouts, root sprouts) or seedling recruitment of Chinese tallow. For the treatments that did not have mastication (F and C), in the winter of 2013, a basal bark herbicide application of 25 percent Garlon 4 Ultra ® v/v with basal paraffin oil was applied to the lower portion of all stems <15 cm diameter at breast height (DBH) of Chinese tallow. Stems >15 cm DBH received the same application rate but applied by the “hack and squirt” injection method. Prescribed fire treatments (F and MF) were conducted in May of 2015. Fires were conducted as backing fires with strip head fires approximately every 15 m burning into the backing fire. The average air temperature during prescribed burns was 28 ± 0.7 °C with average relative humidity of 66 ± 2.5 percent. The average dead ground fuel moisture was 17.5 ± 1 percent. The average percent area burned was 71 ± 6 percent within the sample plot and average fire temperature was 196 ± 17 °C.

Sampling Units

We established a 20 m x 40 m sample plot in each treatment area to determine the effect of treatments on the forest plant community. All trees >3 cm DBH were measured and recorded to species (referred to hereafter as “trees”). In four randomly selected 10 m x 10 m subplots, all saplings or shrubs >1.4 m tall but <3 cm DBH were measured and recorded to species (referred to hereafter as “midstory”). In the middle of each 10 m x 10 m subplot, a 1 m x 1 m quadrat was established (eight total), and all tree regeneration and ground flora were identified and recorded by Braun-Blanquet’s cover class (referred to hereafter as “regeneration” and “ground flora,” respectively) (Westhoff and Van Der Maarel 1978).

A pretreatment survey was conducted in the summer of 2012 and two post-treatment resurveys were conducted in the summers of 2014 and 2015 (3 months post-fire). Ground flora were divided into habit functional types as forbs, graminoids, subshrubs, and vines.

Data Analysis

Trees, midstory, regeneration, and ground flora were analyzed and reported separately to determine the effect of treatments on forest structure and composition. The effect of treatment combination on native species density (stems/ha), basal area (m²/ha), coverage of woody regeneration (percent), coverage of ground flora by species and habit functional type (percent), and species richness of woody regeneration and ground flora, were compared across survey years. Percent coverage was determined as the mid-point between each cover class and was averaged over the entire experimental unit. Mixed model analysis of variance (ANOVA) was conducted using treatment, block, and year as main effects. Year and treatment were fixed effects, block was included as a random effect in the model, and all interactions were tested. In addition, we made within-year comparisons by developing a model to test the effect on the same metrics by containing terms for treatment and block, with block as a random effect. Follow up comparisons between treatments using Tukey's HSD when the ANOVA models were significant.

Density (stems/ha), basal area (m²/ha), and percent coverage of regeneration and ground flora were used to determine the relative effect of treatment on each species or habit functional type. We calculated relative abundance indices (relative density, relative basal area, and relative percent coverage) by plot for each species in 2012 (prior to treatment) and 2015 (final survey year). We then determined percent change (% by subtracting the relative abundance in 2015 from the relative abundance index in 2012 for each species:

$$[\% \Delta = (XSp_i 15 / X TOT_{15}) - (XSp_i 12 / X TOT_{12})] \quad (1)$$

where

$XSp_i 15$ = the index of interest for the i th species in 2015

$XTOT_{15}$ = the total of that index value for all species in the plot in 2015 less the values in 2012.

The percent change by species was averaged by treatment type. We developed a model to determine the relative change by including treatment and block as a random factor. We used linear contrasts to determine the effects of the mastication treatment (MF and M versus F and C), the fire treatment (MF and F versus M and C), and the combined mastication and fire treatment (MF versus all other treatment types).

Data were analyzed using JMP[®], Version 11 SAS Institute Inc., Cary, NC and SAS[®] 9.1.3 SAS Institute Inc., Cary, NC. Data are reported as means and standard errors of the mean. Where appropriate, data were transformed to meet the assumptions of hypothesis testing, but values are reported in original scale to ease interpretation. A p -value ≤ 0.05 was considered significant.

RESULTS AND DISCUSSION

Mastication

The mastication treatment was effective in two ways: it reduced the density of the Chinese tallow, and it reduced the density of understory shrubs. While mastication is not a natural disturbance, its application aids in the transition to a different disturbance type, frequent fire. In recent years, mastication has become a widely used tool to relocate vertical fuels to the ground prior to prescribed fire (Kreye and others 2014). However, plant response and recovery from mastication may depend on a range of factors (Kreye and others 2014). Mastication treatments in our study significantly reduced tree density, midstory basal area, and midstory density (table 1). Mastication also had a negative impact on tree-sized wax myrtle (table 2) and midstory slash pine (table 3) and increased the number of tree-sized oaks, which is probably attributed to more growing space for these species following the reduction in shrubs (table 2). The regeneration of slash pine increased following mastication, which is consistent with enhanced germination from exposed mineral soil for this species (Lohrey and Kossuth 1990). However, the regeneration coverage of yaupon and water oak decreased.

Prescribed Fire

Our goal was to reduce the density of Chinese tallow, a species believed to be sensitive to growing season prescribed fires (Grace 1998), while promoting the native species that are adapted to this disturbance, and thereby creating open-stand structures and a fire-dependent feedback loop. However, prescribed burning alone did not have a discernable impact on native species density or abundance. Prescribed burning did not affect midstory basal area or midstory density (table 1). However, the regeneration coverage of slash pine (table 4) and subshrubs (table 5) decreased following treatment. Based on our findings, a single prescribed fire will not effectively reduce shrub densities and promote understory diversity. However, desired results may not have been achieved due to wet conditions causing the fire not to burn as hot as intended (Pile and others 2017b). Also, the final survey was conducted 3 months following the prescribed fire limiting our ability to make accurate predictions of longer-term community response following this treatment.

Table 1—Treatment means, standard errors, and linear contrasts of overall treatment effects on basal area and density of trees and midstory stems recorded at Parris Island, SC

Treatment	Trees ^a		Midstory ^b	
	BA (m ² /ha)	Density (trees/ha)	BA (m ² /ha)	Density (trees/ha)
C	32.8 ± 2.8	2000 ± 236	1.7 ± 0.3	8510 ± 1510
F	31.0 ± 3.1	1387 ± 246	1.7 ± 0.3	7137 ± 1567
M	26.7 ± 2.9	1026 ± 237	0.8 ± 0.3	6099 ± 1523
MF	30.7 ± 2.9	1043 ± 237	0.9 ± 0.3	5135 ± 1523
	p = 0.49	p < 0.001	p = 0.01	p < 0.01
2012	32.2 ± 1.7	1807 ± 212	2.0 ± 0.2	10513 ± 1425
2014	29.0 ± 1.7	1155 ± 212	1.1 ± 0.2	4606 ± 1425
2015	29.6 ± 1.7	1131 ± 212	0.8 ± 0.2	5042 ± 5042
	p < 0.01	p < 0.001	p < 0.001	p < 0.001
<i>Overall treatment effect</i>				
M (M + MF)	p = 0.28	p < 0.001	p = 0.001	p < 0.01
F (F + MF)	p = 0.71	p = 0.06	p = 0.93	p = 0.06
MF	p = 0.87	p = 0.02	p = 0.046	p < 0.01

P-values in **bold** indicate a significant result based on an alpha of 0.05.

BA = basal area; C = control; F = fire; M = mastication; MF = combination of mastication and fire.

^a Trees are <3 cm DBH.

^b Midstory stems are <3 cm DBH and >1.4 m tall.

Table 2—Overall treatment effect based on linear contrasts for means, standard errors, and test statistics of the percent change in relative density (trees/ha) from 2012 to 2015 for the most commonly recorded tree species^a

Species	M		F		MF	
<i>Acer rubrum</i>	0.07 ± 0.05	0.23	-0.03 ± 0.04	0.41	-0.00 ± 0.06	0.99
<i>Fraxinus pennsylvanica</i>	1.11 ± 0.97	0.27	-0.97 ± 0.98	0.34	-0.47 ± 1.12	0.68
<i>Ilex vomitoria</i>	-3.86 ± 3.78	0.32	-3.40 ± 3.83	0.39	-0.80 ± 4.36	0.86
<i>Juniperus virginiana</i> var. <i>silicicola</i>	0.52 ± 0.41	0.21	-0.37 ± 0.41	0.37	-0.08 ± 0.47	0.87
<i>Liquidambar styraciflua</i>	-2.47 ± 1.64	0.15	-2.50 ± 1.65	0.14	-3.02 ± 1.89	0.12
<i>Morella cerifera</i>	-10.2 ± 4.46	0.03	0.99 ± 4.50	0.83	-6.15 ± 5.13	0.24
<i>Pinus ellottii</i>	13.2 ± 8.19	0.12	-2.65 ± 8.29	0.75	-1.26 ± 9.43	0.90
<i>Quercus hemisphaerica</i>	1.68 ± 0.90	0.08	-0.17 ± 0.91	0.85	1.03 ± 1.04	0.33
<i>Quercus nigra</i>	4.00 ± 1.86	0.04	3.03 ± 1.88	0.12	6.65 ± 2.14	0.01
<i>Quercus pagoda</i>	1.45 ± 0.67	0.04	-0.02 ± 0.67	0.98	1.05 ± 0.77	0.19
<i>Quercus virginiana</i>	3.21 ± 1.39	0.03	0.65 ± 1.40	0.65	3.13 ± 1.60	0.06

Values in **bold** indicate significant differences based on an alpha of 0.05.

M = mastication; F = fire; MF = combination of mastication and fire.

^a Trees are >3 cm DBH.

Table 3—Overall treatment effect based on linear contrasts for the means, standard errors, and test statistics of the percent change in relative density (trees/ha) from 2012 to 2015 for midstory species^a

Species	M		F		MF	
<i>Baccharis halimifolia</i>	0.01 ± 0.27	0.96	-0.42 ± 0.21	0.06	-0.47 ± 0.32	0.15
<i>Callicarpa americana</i>	0.67 ± 1.53	0.67	0.72 ± 1.54	0.65	0.50 ± 1.76	0.78
<i>Ilex vomitoria</i>	-2.69 ± 15.4	0.86	-9.08 ± 15.58	0.57	-14.2 ± 17.7	0.43
<i>Juniperus virginiana</i> var. <i>silicicola</i>	-0.26 ± 0.20	0.19	0.22 ± 0.20	0.27	-0.17 ± 0.23	0.45
<i>Lantana camara</i>	0.81 ± 1.32	0.55	0.86 ± 1.34	0.52	2.20 ± 1.53	0.16
<i>Liquidambar styraciflua</i>	1.46 ± 3.10	0.64	1.17 ± 3.13	0.71	4.72 ± 3.57	0.20
<i>Morella cerifera</i>	2.88 ± 8.32	0.73	2.98 ± 8.42	0.73	-11.4 ± 9.58	0.25
<i>Pinus elliotii</i>	-18.5 ± 9.66	0.04	20.15 ± 9.76	0.05	8.11 ± 11.1	0.47
<i>Quercus hemisphaerica</i>	-0.03 ± 0.02	0.23	-0.00 ± 0.02	0.69	-0.00 ± 0.03	0.82
<i>Quercus nigra</i>	0.27 ± 0.43	0.55	0.37 ± 0.44	0.40	0.62 ± 0.50	0.23
<i>Quercus pagoda</i>	-0.04 ± 0.09	0.64	-0.13 ± 0.09	0.17	-0.00 ± 0.11	0.94
<i>Quercus virginiana</i>	-0.22 ± 0.14	0.13	-0.07 ± 0.14	0.64	-0.19 ± 0.16	0.24

Values in **bold** indicate significant differences based on an alpha of 0.05.

M = mastication; F = fire; MF = combination of mastication and fire.

^a Midstory stems are <3 cm DBH and >1.4 m tall.

Table 4—Change in relative cover of regenerating woody species from 2012 to 2015 survey years by overall treatment effect

Species	Treatment effects					
	M		F		MF	
<i>Acer rubrum</i>	-0.7 ± 0.8	0.43	-0.9 ± 0.8	0.29	-0.6 ± 0.9	0.54
<i>Baccharis halimifolia</i>	0.3 ± 0.3	0.33	-0.3 ± 0.3	0.33	0.2 ± 0.3	0.57
<i>Callicarpa americana</i>	1.7 ± 1.8	0.36	-1.4 ± 1.9	0.44	-1.2 ± 2.1	0.59
<i>Celtis laevigata</i>	0.2 ± 0.4	0.66	0.6 ± 0.4	0.16	0.5 ± 0.5	0.27
<i>Fraxinus pennsylvanica</i>	-0.1 ± 0.1	0.61	0.2 ± 0.1	0.09	0.1 ± 0.1	0.48
<i>Ilex vomitoria</i>	-15.5 ± 7.4	0.05	6.6 ± 7.5	0.39	-5.4 ± 8.6	0.54
<i>Juniperus virginiana</i> var. <i>silicicola</i>	-	0.99	-	0.71	-	0.90
<i>Lantana camara</i>	-0.1 ± 0.1	0.66	0.1 ± 0.1	0.40	-	0.91
<i>Liquidambar styraciflua</i>	0.5 ± 0.7	0.52	-0.6 ± 0.7	0.43	0.4 ± 0.8	0.65
<i>Morella cerifera</i>	-0.3 ± 6.7	0.97	5.6 ± 6.8	0.42	2.3 ± 7.8	0.77
<i>Pinus elliotii</i>	20.3 ± 6.8	0.01	-17.8 ± 6.9	0.02	1.7 ± 7.9	0.83
<i>Prunus serotina</i>	-0.4 ± 0.6	0.55	0.7 ± 0.6	0.28	-0.5 ± 0.7	0.49
<i>Quercus hemisphaerica</i>	-	0.39	-	0.39	-	0.61
<i>Quercus nigra</i>	-1.2 ± 0.5	0.05	0.3 ± 0.6	0.65	-0.6 ± 0.6	0.32
<i>Quercus pagoda</i>	-	0.39	-	0.39	-	0.61
<i>Quercus virginiana</i>	0.7 ± 1.0	0.54	0.7 ± 1.0	0.50	1.8 ± 1.2	0.14

Relative percent cover values were calculated as the proportion of each species by plot and by year (2012 and 2015), with percent change determined as the difference in proportions between 2015 and 2012 survey years by species and averaged across treatments by species.

Differences in relative distributions were analyzed with an ANOVA and Tukey's HSD with significant differences indicated in **bold** based on an alpha of 0.05.

M = mastication; F = fire; MF = combination of mastication and fire.

Table 5—Mean ($\pm$ SE) overall treatment effects on percent change in relative cover of ground flora habit functional type between 2012 and 2015

	Treatment effects					
	M		F		MF	
Forbs	7.4 $\pm$ 6.1	0.24	3.2 $\pm$ 6.1	0.61	4.1 $\pm$ 7.0	0.57
Graminoids	3.4 $\pm$ 5.9	0.57	3.0 $\pm$ 5.9	0.61	5.9 $\pm$ 6.8	0.40
Subshrubs	-10.4 $\pm$ 7.9	0.20	-18.2 $\pm$ 8.0	0.03	-27.1 $\pm$ 9.1	<0.01
Vines	-8.1 $\pm$ 8.6	0.36	16.0 $\pm$ 8.6	0.08	9.2 $\pm$ 9.9	0.36

Values in **bold** indicate significant differences based on an alpha of 0.05.

M = mastication; F = fire; MF = combination of mastication and fire.

Mastication and Prescribed Fire Combination

The purpose of our combined mastication and fire treatment was to quickly alter forest structure to an open canopy with an intention to create a herbaceous understory, reminiscent of historical maritime forest structure and processes. Adding frequent fire was intended to increase species diversity and function while also increasing community resistance to invasion by species not well-adapted to frequent fires. However, our approach is not characterized as restoration because these sites have a long history of human disturbance (Pile and others 2017a) with the majority of stands represented by mature slash pine plantations. Restoration to pre-European settlement would be nearly impossible and outside the scope of management goals. Instead, we intended to promote the historic disturbance (i.e., frequent surface fires) that inhibits the invader (Funk and others 2008) while also increasing desirable native species, including native oak species, and thereby potentially increasing the community resistance to future invasion.

We found that when mastication and fire were combined (MF) there was a significant reduction in tree density (-428 ± 173 stems/ha; $p = 0.02$), midstory density (-2113 ± 1003 ; $p = 0.04$), and midstory basal area (-0.6 ± 0.3 m²/ha; $p = 0.04$). There was also an increase in tree-sized water oak density (table 2), ground flora richness (1.5 ± 0.6 ; $p = 0.03$; fig. 1), and a decrease in subshrubs (table 5). Prescribed fire with mastication as an intermediary treatment was effective at maintaining reduced shrub density when compared to mastication alone. The increase in rapid-growing, long-lived, and fire-tolerant species, such as slash pine and water oak, may also help to reduce the competitiveness of Chinese tallow by occupying growing space. When the mastication treatment was not followed by prescribed

fire, shrub density increased and ground layer richness remained constant.

Our results are similar to those of Kane and others (2010), where mastication treatments alone reduced midstory density but did not increase ground layer richness. However, when mastication was combined with prescribed fire, native species richness increased by 150 percent compared to the control (Kane and others 2010). Increased ground layer richness may also help to impede establishment of Chinese tallow, by reducing available resources and may aid in carrying repeated surface fires. Siemann and Rogers (2003) reported reduced above-ground competition in grasslands increased the survival and growth of Chinese tallow. In addition, mastication reduced the height of the very dense wax myrtle and yaupon midstory to levels below 1.4 m tall, and the application of prescribed fire kept these fast growing species near the ground, also increasing important surface fuels. When yaupon remains near the ground surface it is known to promote surface fire spread in communities with frequent fire regimes (Mann and Fischer 1987, Villarrubia and Chambers 1978,) and may be important for maintaining this type of disturbance regime especially in areas where fine fuels are sparse.

While frequent surface fires may keep Chinese tallow at juvenile stages, resulting in a reduction of invader density over the long term, a short, fire-free period will be necessary to recruit the regeneration of species such as slash pine to sapling-sized class when it is less sensitive to fire. Planting of native tree species, such as longleaf pine (*P. palustris* Mill.) that are adapted to frequent surface fire regimes, even at juvenile stages, would allow for the continued application of frequent prescribed burning, thus increasing the competitiveness of native species adapted to this disturbance regime while

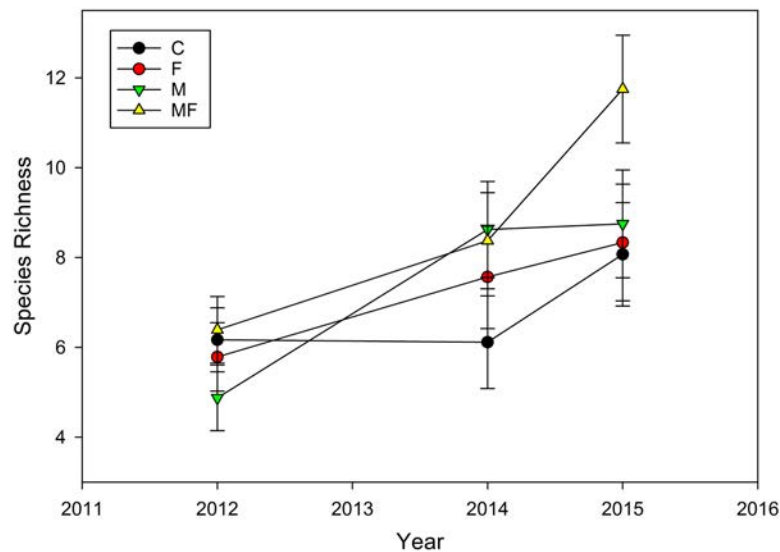


Figure 1—Change in mean ($\pm$ SE) ground layer herbaceous richness by treatment type (C = control; F = fire; M = mastication; MF = combination of mastication and fire) and survey year. Mastication occurred in the spring of 2013 and prescribed fire in the spring of 2015.

ensuring Chinese tallow does not reach reproductive maturity or reduced sensitivity to fire during fire-free periods. Additionally, enrichment planting with native herbaceous species may be necessary to increase the functional diversity of frequent fire-evolved species that may be absent from the seed bank after a long history of human manipulation.

CONCLUSION

We found that, by using a management approach that included both mastication and prescribed fire, along with herbicide to target the invader, we were able to maintain a reduced midstory density, increase ground flora richness, and create a forest structure that is reminiscent of what historically characterized this ecological community. However, our results are limited to the immediate effects of the prescribed fire treatment. Repeated prescribed burning and continued monitoring are needed to determine if frequent burning after the MF treatment will continue to reduce Chinese tallow and positively impact native species. Future studies will also be needed to test for increased community resistance to Chinese tallow invasion under these treatment regimes.

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PRELIMINARY FINANCIAL EVALUATION OF MANAGEMENT REGIMES CONTROLLING CHINESE PRIVET IN LOBLOLLY PINE STANDS

Fabio J. Benez-Secanho, Donald L. Grebner, Andrew W. Ezell, and Robert K. Grala

Abstract—Chinese privet (*Ligustrum sinense* Lour.) is the most common invasive shrub species in the Southern United States, causing biodiversity and economic losses. This study evaluated several treatments found in the literature and conducted a financial analysis to identify the most cost-effective management regimes for controlling this species in loblolly pine (*Pinus taeda* L.) stands. Simulated scenarios were created to assess these management regimes. Three components were used: infestation levels, herbicide application methods, and herbicides. For each simulated scenario, the financial impact on land expectation values (LEV) was analyzed. Results indicated that the most cost-effective management regime controlling Chinese privet in loblolly pine stands is to aerial spray with Arsenal AC, followed by a backpack spray 2 years later with the same herbicide. Chinese privet control is economically feasible, and a positive LEV could be achieved for all scenarios. Further research of the same nature with more components and variables is being conducted.

INTRODUCTION

Components of the Earth's biodiversity are being altered by humans' activities, leading to an increase in species invasions and extinction of endemic species. These changes in composition of communities and ecosystems can affect the provision of ecosystem services, which are essential for society (Estrada and Flory 2014, Hooper and others 2005). Competition among plants can cause environmental changes, especially when an exotic species spreads over a nonnative range, competes with native species, and becomes an invasive species (Brooker 2006). About 50,000 species were introduced in the United States for many reasons, such as landscaping, biological control, packing materials, and food production. Currently, exotic species account for more than 98 percent of the U.S. whole food system, and are responsible for an estimated annual value of US\$800 billion (Pimentel and others 2005). However, due to favorable conditions some of these species escaped from cultivation and became invasive species, spreading over 133 million acres in the United States and causing an estimated annual loss of US\$120 billion (Pimentel and others 2005).

Chinese privet (*Ligustrum sinense* Lour.) is a shrub native to China, and was introduced in the United States in 1852 for landscaping purposes (Dirr 1998, Maddox and others 2010). Due to its ability to grow and reproduce rapidly, it spread over the Southern United States and became the most common invasive shrub species in

this region (Oswalt and Oswalt 2011, Urbatsch and Skinner 2000). Other invasive species belonging to the same genus, such as European privet (*Ligustrum vulgare* L.), mostly found in the Northeastern United States (BONAP 2015), and Japanese privet (*Ligustrum japonicum* Thunb.), mostly occurring in the Southern United States, are less abundant than Chinese privet (BONAP 2015, Maddox and others 2010). Chinese privet is an effective colonizer, growing by seed dispersal and vegetatively, and is normally introduced into new areas carried by wildlife, especially birds (Maddox and others 2010). Chinese privet can grow in the understory and suppress growth of tree seedlings, compromising overstory regeneration and causing a shift within the ecosystem from forest to shrub land (Loewenstein and Loewenstein 2005).

Although some species have potential to harm or feed on Chinese privet, there is no widespread use of any biological control (Maddox and others 2010). Chinese privet can sprout vigorously after burning, so fire is not recommended as a stand-alone treatment (Urbatsch and Skinner 2000). Mechanical removal is more effective if combined with herbicide applications because Chinese privet can vegetatively spread (Hanula and others 2009, Klepac and others 2007). Finally, herbicide applications alone can effectively control privet (Maddox and others 2010, Miller 2003).

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Only a few studies in the literature evaluated the financial tradeoffs of invasive species control measures. A common financial criteria land expectation value (LEV) can be used to compare alternative forest investments (Bettinger and others 2009). Grebner and others (2011) utilized LEV to analyze alternative management regimes controlling kudzu [*Pueraria montana* var. *lobata* (Willd.)], and Prevost and others (2007) assessed financial tradeoffs controlling cogongrass [*Imperata cylindrica*, (L.) P. Beauv.] by using LEV. Therefore, the objective of this study was to compare the financial effectiveness of common silvicultural treatments used to control different levels of Chinese privet infestations in loblolly pine stands.

MATERIALS AND METHODS

Area Conditions

This study evaluated Chinese privet treatments in loblolly pine (*Pinus taeda* L.) stands. These stands were assumed to be 20 years old and having 251 trees per acre at this age. Growth and yield were simulated using the software acquired from the U.S. Department of Agriculture Forest Service's Forest Vegetation Simulator (FVS). An initial planting density of 538 seedlings per acre (9 feet x 9 feet) was used in this simulation, and a rotation length of 33 years was adopted. First thinning was to occur at age 13 and a second thinning at age 21. This rotation length and thinning intervals were consistent with previous literature (Davis 2013). This study also assumed that the area will be artificially regenerated after harvesting and the same regime adopted.

Chinese privet can grow dense thickets even in the understory of forests, which can adversely impact operational efficiency of ground-based herbicide applications (Klepac and others 2007, Maddox and others 2010). Therefore, three different Chinese privet infestation levels were evaluated. Each level was defined by the number of Chinese privet stems per acre and was adapted from Hanula and others (2009), Hart and Holmes (2013), and Merriam and Feil (2002): low density

(<3,496 stems per acre), medium density (ranging between 3,497 and 8,742 stems per acre), and high density (>8,743 stems per acre).

Management Regimes

Some of the most effective treatments for Chinese privet were collected, reviewed, and used to calculate costs under different levels of Chinese privet density and herbicide control options. These treatments were herbicide-based, and had a minimum effectiveness of 90 percent when controlling for this species to decrease the chance of recolonization after treatment (Miller 2005). This study evaluated two different herbicide application methods: aerial (helicopter) and skidder equipped with broadcast sprayer. In addition, two herbicide brands were included in this evaluation: Arsenal AC (imazapyr) and Accord XRT II (glyphosate). Aerial applications are recommended to be conducted during the winter to limit native species damage, and herbicides containing glyphosate should be avoided even during the winter due to their potential for harming pine trees (Dow AgroSciences 2012). Rates of active ingredient per acre were estimated based on studies from the literature, and they varied according to the Chinese privet infestation level. Table 1 provides the quantity of each herbicide per acre based on rates of active ingredient and their respective concentration. Chinese privet is not properly controlled with one single herbicide application (Johnson and others 2010, Klepac and others 2007, Miller 2005). Therefore, all management regimes include a second application 2 years later, using a backpack to spray on resprouts. This second application used the same herbicide of the first application and the same rate used in low infestation level treatments.

Financial Analysis

Land expectation value was the method chosen to compare economic impact of controlling Chinese privet in loblolly pine stands, and cash flows were discounted at a rate of 6 percent. Net present value (NPV) of current stands was first calculated using revenues from thinning and final harvesting,

Table 1—Rate of herbicide per acre used in the management regimes for controlling Chinese privet (*Ligustrum sinense* Lour.)

Commercial name	Active ingredient	Herbicides (product/ac) ^a			Chinese privet infestation levels		
		Concentration (active ingredient/gal)	Unit		Low	Medium	High
Accord XRTII	Glyphosate	5.07 lb/gal	gal/ac		0.592	0.888	1.183
Arsenal AC	Imazapyr	4.9 lb/gal	gal/ac		0.128	0.153	0.179

^a Rates are calculated based on the concentration of active ingredient and rate of active ingredient per acre.

and costs of management regimes controlling Chinese privet. In addition, LEV for a loblolly pine reforestation was calculated using revenues from thinning and final harvesting and costs of reforestation. This LEV calculation did not include costs for controlling Chinese privet because it was assumed that this species population was effectively controlled at this point (Hudson and others 2013). Finally, this study merged NPV of current stands and LEV of future reforestation to calculate the combined LEV. Therefore, this combined LEV accounts for costs and revenues from current and future stands. Since this study assumed that current stands are 20 and 22 years old when Chinese privet treatments occur, these are years 0 and 2 for financial analysis purpose. LEV for an area free of Chinese privet was also calculated for comparison purpose.

Costs and revenues

Costs used in this study represent an average of southern States (Alabama, Arkansas, Florida, Georgia, Louisiana, Mississippi, North Carolina, South Carolina, Tennessee, Texas, and Virginia), and were obtained from vendors, literature, and personal communication. Table 2 displays costs for herbicides and surfactant. Costs related to herbicide application are reported in table 3, and they vary according to Chinese privet infestation level. An average cost of US\$144.89 per acre for loblolly pine regeneration in the Southern United States was calculated by accounting for chemical site preparation, planting stock, and hand planting.¹ Revenues from timber harvesting were calculated using prices from TimberMart-South. The average price for pulpwood used in this study was US\$10.23 per short ton, US\$17.31 per short ton for chip-n-saw, and US\$25.39 per short ton for sawtimber (TimberMart-South 2016).

RESULTS AND DISCUSSION

A land expectation value of US\$895.18 per acre, with a rate of return of 12.35 percent, was first calculated for a loblolly pine stand free of Chinese privet. This LEV was incorporated with the NPVs of all simulated area conditions, generating a combined LEV for each management regime. The maximum possible combined LEV that can be obtained for this loblolly pine stand was US\$3,238.41 per acre when no Chinese privet is present. Table 4 reports the results for all area conditions, displaying the maximum possible LEV for this loblolly pine stand, total cost per acre for each management regime (this cost includes a first and second application), regime

Table 2—Average costs of herbicides and surfactant used in the management regimes for controlling Chinese privet (*Ligustrum sinense* Lour.)

Product	Unit	Cost/unit (US\$)
Accord XRT II (herbicide)	Gallon	32.64
Arsenal AC (herbicide)	Gallon	182.38
Cide-Kick II (surfactant)	Gallon	36.95

Source: Local and online vendors.

Table 3—Costs^a for herbicide application for controlling Chinese privet (*Ligustrum sinense* Lour.)

Application method	Chinese privet infestation level		
	Low (US\$/ac)	Medium (US\$/ac)	High (US\$/ac)
Aerial	30.00	30.00	30.00
Skidder	55.00	65.00	70.00
Backpack	80.00	-	-

Sources: Aerial: Personal communication. Michael McCool. 2016. Provine Helicopter Service, 308 Airport Road, Greenwood, MS 38930.

Skidder and backpack: Costs were estimated based on Miller (1988) and personal communication, Gueth Braddock. 2016. Silviculturist, 524 Natchez Dr. NE, Brookhaven, MS 39601.

^a Costs for herbicide and surfactant are not included.

effectiveness, LEV when regime costs are included, and the percentage that these LEVs represent when compared to the maximum LEV for this area.

Results depicted in table 4 suggested that aerial application with Arsenal AC followed by a second application using the same herbicide, but spraying with backpack sprayer, was the most cost-effective management regime when controlling for Chinese privet when the infestation level is low, and has a cost of US\$151.21 per acre. This regime has an effectiveness rate of 94 percent, and yielded a LEV of US\$3,087.20 per acre, which represents 95.33 percent of the maximum LEV. The most cost-effective management regime for controlling a medium infestation of Chinese privet is aerial spray with Arsenal AC followed by an application with the same herbicide using backpack sprayer two years later (table 4). This regime has a cost of US\$155.88 per acre, and yielded a LEV of US\$3,082.53 per acre.

¹ Personal communication. Randall J. Rousseau. 2016. Associate Extension/Research Professor, Room: 369 Thompson Hall, Department of Forestry, Mississippi State, MS 39762-9681.

Table 4—Results for controlling low, medium, and high infestation levels of Chinese privet (*Ligustrum sinense* Lour.) in a loblolly pine (*Pinus taeda* L.) stand

Infestation level	Code ^a	Chinese privet control			
		Total cost ^b	Effectiveness	Financial analysis	
		US\$/acre	%	LEV ^c (US\$/acre)	% ^d
No Chinese privet	No control	0.00	-	3,238.41	100
Low	A-Ar	151.21	94	3,087.20	95.33
	S-Ar	177.13	94	3,061.28	94.53
	S-Ac	162.70	99	3,075.71	94.98
Medium	A-Ar	155.88	94	3,082.53	95.19
	S-Ar	192.72	94	3,045.68	94.05
	S-Ac	182.36	99	3,056.05	94.37
High	A-Ar	160.53	94	3,077.88	95.04
	S-Ar	194.36	94	3,044.05	94.00
	S-Ac	197.01	99	3,041.39	93.92

^a Codes depict characteristics of management regimes. No control = no management regime is used. Application method: A = aerial; S = skidder. Herbicide: Ar = Arsenal AC; Ac = Accord XRTII.

^b Total cost includes two applications.

^c LEV (land expectation value) when control costs are included.

^d Percentage this LEV represents of the maximum possible LEV for each area.

When the infestation level is high, table 4 shows that spraying aerially with Arsenal AC followed by a second application using backpack sprayers with the same herbicide was also the most cost-effective management regime. However, under this condition, costs of controlling Chinese privet were higher due to higher rates of herbicides, and this treatment costs US\$160.53 per acre. The LEV of this regime is US\$3,077.88 per acre, and represents 95.04 percent of the maximum LEV.

Overall within treatments, costs for controlling Chinese privet increased as the infestation level increased. Due to a higher number of plants per unit area, ground-based herbicide application operations were more expensive for higher infestation levels. Aerial applications were less costly when compared to ground-based, and for this reason the most cost-effective management regimes in this study used aerial applications. Even though aerial application costs were constant, regardless of the infestation level, higher herbicide rates were necessary to control higher infestation levels of this species, which made these applications more costly under these conditions. When determining whether aerial applications are feasible or not, an important consideration is target area size. Normally, contractors only spray areas larger than 20 acres.

However, some contractors offer their services for a group of landowners of small areas from the same region, making these applications economically feasible.² Although aerial applications had better financial performance, due to site restrictions, landowner preferences, or other reasons, these applications sometimes cannot be used. For these cases, this study included ground-based applications conducted by a skidder as possible management regimes. Controlling invasive species can aid in recapturing underutilized or degraded sites, which provides wildlife habitat and can generate extra revenues for landowners (Grebner and others 2011).

CONCLUSIONS

The study objective was to evaluate and examine the financial tradeoffs of controlling Chinese privet in loblolly pine stands with different levels of infestation. Regardless of the infestation level, an aerial application and followup backpack spray with Arsenal AC was the most cost-effective control method for Chinese privet. Results indicated that controlling any level of Chinese privet infestation in a loblolly pine stand with these specific characteristics

² Personal communication. Michael McCool. 2016. Provine Helicopter Service, 308 Airport Road, Greenwood, MS 38930.

was economically feasible. Revenues generated by this stand were sufficient to offset management regime costs, and the financial impact of these regimes on LEVs was relatively small. Further economic analysis is being conducted to evaluate management regimes controlling for Chinese privet in other forest types, accounting for stand density and including a greater number of control methods.

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Disturbance and Damaging Agents

Moderator:

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AGALINIS – A ROOT PARASITE ON LOBLOLLY PINE

Alan Byron Wilson and Lytton John Musselman

Abstract—Loblolly pine (*Pinus taeda* L.) is the most widely planted pine species in the Southern United States due to its ability to grow well on many different sites. After a tree is planted, insects and disease can have an impact by reducing growth that sometimes leads to mortality. Parasitic plants can also reduce the growth of loblolly pine by attaching to the roots. Over the years, several publications have documented the devastating impact *Seymeria* (*Seymeria cassioides* Orobanchaceae) can have on loblolly pine. The related fascicled gerardia or purple false foxglove (*Agalinis fasciculata* Orobanchaceae) has recently been found in numerous locations in young loblolly pine stands. This paper provides information on its impact on loblolly pine, how to identify *A. fasciculata*, where the plant has been found, and ways to control it.

Landowners in the Southeastern United States are planting loblolly pine more than any other pine species because it can be grown on many different soil types and drainages. Also, it responds well to silvicultural inputs such as fertilization. It was estimated that in 2013, over 756 million seedlings of loblolly pine were grown in forest tree nurseries (South and Harper 2016), more than any other pine species.

After a tree is planted, diseases and insects can reduce growth. Fusiform rust (*Cronartium fusiforme*), pitch canker (*Fusarium circinatum*), and Nantucket pine tip moth (*Rhyacionia frustrana*) can negatively impact loblolly pine growth. Recently, *A. fasciculata* was observed in some 3-year-old loblolly pine plantations causing loss of growth, and occasionally, mortality. In many cases, where *A. fasciculata* is growing adjacent to the planted pine tree, height growth is greatly reduced and needles are brown. Often the tree's lower limbs are dead, resulting in crown recession which further reduces the tree's growth (fig. 1).

Agalinis fasciculata is an annual, herbaceous plant that parasitizes several pine species (Musselman and Mann 1978) including loblolly pine, longleaf pine (*P. palustris* Mill.), sand pine (*P. clausa* Chapm. ex Engelm.), slash pine (*P. elliotii* Engelm.), and shortleaf pine (*P. echinata* Mill.). Root parasites have highly specialized roots known as haustoria which attach to the host's roots. The haustorium is the conduit which allows moisture and nutrients to move from the host to the parasite. *Agalinis fasciculata* is native to the Southeastern United States (Miller and Miller 1999), has pink or purple flowers

lasting one day, and grows 5 to 6 feet tall by the end of the growing season. In the late fall and early winter, the plant may be identified by the numerous seed capsules located at the top of the plant (fig. 1).

After a tract is harvested and before planting, it is common practice to prepare the site using herbicides to control vegetation that will compete with the



Figure 1—Loblolly pine impacted by *A. fasciculata* growing between two trees. (photo by Alan Wilson)

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seedling. It is thought that herbicide rates used to control hardwoods and shrubs are sufficient to control *A. fasciculata*. Since the parasitic plant is an annual, applying the herbicide before flowering, or no later than late August in the Southeastern United States, is crucial for the treatment to be effective. Additional work is planned to better understand how to control *A. fasciculata* in loblolly pine plantations.

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Forest Mensuration and Modeling

Moderator:

Yuhui Weng

Stephen F. Austin State University

A COMPARISON OF TWO GROUPS OF YIELD PLOTS REPRESENTATIVE OF LOBLOLLY PINE PLANTATIONS IN THE SOUTHEASTERN UNITED STATES

Ralph L. Amateis and Harold E. Burkhart

Abstract—Stand growth and productivity during the first decade for two groups of yield plots established in managed loblolly pine plantations growing in the Southeastern United States were compared. Both groups were established on site-prepared areas in operational stands growing under intensive management. Group 1 plots represented stands established during the 1990s with open pollinated single family or multi-family stock and received silvicultural treatments typical of the period. Group 2 plots were established approximately 10 years later in the same physiographic region with clones and subjected to site-suitable silvicultural treatments. After accounting for the difference in age, average height growth of dominant and codominant trees (site trees) was not significantly different between the groups. Individual tree diameter and height growth were different resulting in stem form characteristics that were significantly different for the two groups. Characteristics that negatively affect stem quality were generally reduced in the clonal plantings suggesting a greater proportion of large trees will be suitable for sawtimber. Simulations with a growth and yield model showed increased productivity and future potential value for plantations established with clones.

INTRODUCTION

Over the past 60 years loblolly pine plantations in the Southern United States have become some of the most productive forests in the world (Fox and others 2004, 2007). The reasons for this are many. Prudent harvesting practices and effective site preparation methods have reduced damage to the site and suppressed competing vegetation resulting in improved seedling survival and faster early growth. Where needed, nutrient amelioration and more effective early chemical release treatments have allowed young stands to develop quickly with reduced competition. With each new generation, genetically enhanced planting stock selected for faster growth, better stem form and resistance to disease has been a large contributor to this incremental improvement in productivity (Schmidtling and others 2004).

To test hypotheses about the individual effects of any subset of these factors on stand growth requires designed experiments where the treatment of interest is assessed using ANOVA methods while other factors are held constant. This is often not possible due to the large size of such experiments and the limited resources available. An alternative approach uses growth and yield data from permanent long-term remeasurement plots in existing plantations to compare stand conditions at desired stages of stand development. This approach acknowledges the obvious confounding of all factors

contributing to growth and utilizes regression methods to test for differences in existing conditions (ANCOVA) and to make projections for future conditions. While lacking the inferential rigor of a designed study, it is well suited for applications where the objective is to assess the overall growth performance of a population of stands across a broad range of climatic, edaphic influences and management practices (Burkhart and Amateis 2012).

The purpose of this study was to assess the early growth of two groups of growth and yield plots in existing managed operational loblolly pine plantations, one established with open pollinated stock and the other with clones, and to make comparisons at similar ages. A further goal is to provide a glimpse into the future productivity of both groups to assist managers tasked with planning for the future.

METHODS

Data

Over the past 20 years, the Forest Modeling Research Cooperative (FMRC) has installed permanent growth and yield plots in two populations of intensively managed loblolly pine plantations on cutover sites established with improved genetic material. The newer group (G2) consists of 42 plots in stands established during the period 2005-2010 with varietal material and the latest

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silvicultural treatments. Although understory brush and herbaceous vegetation is prevalent, non-pine competition in the main canopy is negligible. These stands lie in the Coastal Plain areas of AL, GA, FL, SC, and NC between latitudes 30.5°N and 35.0°N and longitudes -78.0°W and -87°W (fig. 1). Ages at time of plot establishment were between 2 and 6 years with measurements at 2-year intervals. This schedule resulted in 125 plot measurements with 14,503 trees (tables 1 and 2).

The older group of plots (G1) is a subset of a study established across the natural loblolly pine growing region in stands established during the 1990s. Ages at time of plot establishment ranged from 3 to 8 years (Russell and others 2010). These stands received operational silvicultural treatments common for the era and suited for the site, climate, and management objectives of the owner. Some competition with hardwoods in the main canopy exists for some of the plots. All were established with open-pollinated single family or multi-family genetic material. Forty-six of these G1 plots lie in the same Coastal Plain corridor as the G2 plots and were used for this comparative study (fig. 1). Measurement data at ages 3 to 10 years were used from the G1 plots to correspond to the measurement ages of the G2 plots. Intervals between measurements were 2 years. The G1 data resulted in 133 plot measurements with 12,604 tree measurements (tables 1 and 2). Stem quality codes based on a tiered hierarchical classification system were collected at each measurement for both G1 and G2 datasets. First, the stem was classified as single-stemmed or forked, then having a normal top or broken top. Condition of the bole was assessed as straight, bole or butt sweep, or short crook. Finally, the health of the tree was categorized as having or not having insect or disease damage.

Height-Age Relationship

An analysis of covariance (ANCOVA) approach was used to examine the height-age relationship for the two populations. The response variable was 2-year dominant and codominant height growth and the covariate was stand age at the midpoint of the 2-year growth period:

$$HDg_{ij} = \mu + \tau_i + B(\tau_{ij} - \bar{\tau}_i) + \varepsilon_{ij} \quad (1)$$

where

HDg_{ij} is the 2-year average height growth of dominant and codominant trees for the i th dataset (G2 or G1), μ is the grand mean, τ_i is the effect of the i th dataset, τ_{ij} is the j th observation of the covariate in the i th dataset, $\bar{\tau}_i$ is the i th population mean and ε_{ij} is the error term. Equation (1) assumes equal slopes between the two populations. Table 3 summarizes the height-age data used for the evaluation. To test the validity of the assumption of equal slopes, the interaction term between the covariate and the two categorical independent variables (populations) was included in equation (1). The Chapman-Richards equation was fitted to the dominant height-age data for both datasets:

$$HD = \beta_1(1 - \exp(-\beta_2 A))^{\beta_3} \quad (2)$$

where

HD is the average height of the dominant and codominant trees (feet), A is stand age and $\beta_1 - \beta_3$ are parameters. Equation (2) was fitted to each dataset using nonlinear least squares. Due to the limited range of the height-age data, the asymptote parameter (β_1) was set to 100 feet for both datasets and β_2 (the rate parameter) and β_3 (the shape parameter) were estimated for each dataset.

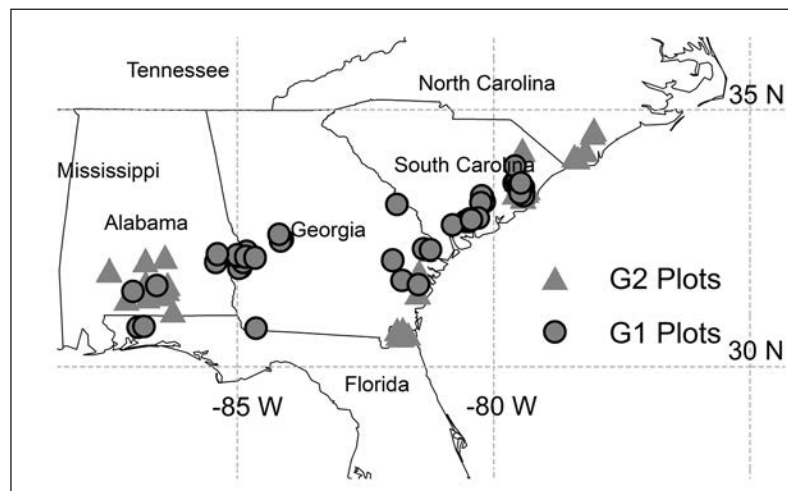


Figure 1 — Location of 42 G2 and 46 G1 growth and yield plots in operational loblolly pine plantations in the Southeastern United States.

Table 1—Summary statistics for dbh, total height and 2-year growth increment of two populations of loblolly pine trees growing in the Coastal Plain areas of AL, GA, FL, SC, and NC

Variable	Mean	Std. Dev.	Coef. Var.	Minimum	Maximum
-----G2 (obs=14503)-----					
D.b.h. (inches)	3.1	1.76	56.7	0.1	8.3
DGro(2-yr.) (inches)	1.8	0.7	35.4	-0.1	5.7
H (feet)	17.3	9.2	53.0	0.6	45.7
HGro(2-yr.) (feet)	9.0	3.2	35.4	-10.3	27.5
H/d.b.h. (feet per inches)	6.9	3.2	46.3	1.5	50.0
-----G1 (obs=12604)-----					
D.b.h. (inches)	4.2	1.7	39.5	0.1	10.0
DGro(2-yr.) (inches)	1.0	0.52	52.2	-0.2	3.9
H (feet)	26.0	9.9	38.0	1.8	55.6
HGro(2-yr.) (feet)	7.8	2.5	33.8	-15.0	19.7
H/d.b.h. (feet per inches)	6.4	1.6	25.0	1.7	53.0

Std. dev. = standard deviation; Coef. Var. = coefficient of variation.

Table 2—Summary statistics for per unit area plot-level data representing two populations of loblolly pine stands growing in the Coastal Plain areas of AL, GA, FL, NC, and SC

Variable	Mean	Std. Dev.	Coef. Var.	Min.	Max.
-----G2 (obs=125)-----					
Age (years)	4.6	1.9	41.0	2	9
Hd	18.0	8.8	49.0	4.3	38.5
Trees (acre)	461	89.7	19.4	344	704
BA	30.5	26.9	88.3	0.17	110.8
Dq (inches)	3.0	1.7	55.8	0.28	6.4
PercNPComp ^a	0	0	0	0	0
-----G1 (obs=133)-----					
Age (years)	7.4	1.9	25.7	3	10
Hd	26.8	9.1	34.0	8.6	49.3
Trees (acre)	636	92.8	15.1	437	875
BA	71.8	36.4	50.8	2.9	147.7
Dq (inches)	4.4	1.4	31.8	0.82	7.4
PercNPComp ^a	3.5	3.0	86.2	0.4	14.2

Std. dev. = standard deviation; Coef. Var. = coefficient of variation; Hd= the average height of dominant and codominant trees (feet); BA = basal area (square feet per acre).

^a Percent of total stand basal area in non-planted pine overstory competition.

Table 3—Two-year height growth for the dominant and codominant trees and the age at the midpoint of that growth period for the G2 and G1 plots

Variable	N	Mean	Std. Dev.	Coef. Var.	Min.	Max.
-----G2-----						
Age (year)	83	4.6	1.35	29.7	3	8
HD Growth	83	9.4	2.6	23.9	4.4	15.0
-----G1-----						
Age (year)	91	7.3	1.4	19.8	4	9
HD Growth	91	8.4	1.8	21.8	3.7	12.4

Std. dev. = standard deviation; Coef. Var. = coefficient of variation; HD = the average height of dominant and codominant trees (feet).

Growth and Allometry of D.b.h. and Height

ANCOVA methods were used to compare d.b.h. and height relationships between the G1 and G2 populations over the 4-year period. A comparison of the sample group means for height and d.b.h. growth was conducted using age as a covariate. The models were:

$$Hg_{ij} = \mu + \tau_i + B(\tau_{ij} - \bar{\tau}_i) + \varepsilon_{ij} \quad (3)$$

$$Dg_{ij} = \mu + \tau_i + B(\tau_{ij} - \bar{\tau}_i) + \varepsilon_{ij} \quad (4)$$

$$(H / D)_{ij} = \mu + \tau_i + B(\tau_{ij} - \bar{\tau}_i) + \varepsilon_{ij} \quad (5)$$

where

HDg_{ij} and Dg_{ij} are the 2-year average height and d.b.h. growth, respectively, of trees for the i th population (G2 or G1) and $(H/D)_{ij}$ is the height over d.b.h. The overall mean is μ , τ_i is the effect of the i th population, τ_{ij} is the j th observation of the covariate, age, in the i th population, $\bar{\tau}_i$ is the i th population mean, and ε_{ij} is the error term.

Stem Quality Assessments

Three assessments of stem quality were evaluated at three measurements for the G2 and G1 populations: forking, disease or insect damage, and stem sweep. Trees with either bole or butt sweep were considered as having a defect of the main stem that would preclude them from being considered sawtimber quality. Frequencies by population for each of these defects were tested with Chi-square and Odds Ratio tests of significance. In this comparison, the closer an odds ratio is to 1 the more likely it is that the defect will occur in

both groups. An odds ratio > 1 indicates the defect will be more likely to occur in the G1 plots. An odds ratio < 1 indicates the defect will be more likely to occur in the G2 plots.

Simulation with PTAEDA

For comparison purposes the PTAEDA growth and yield model was used to project current stand conditions to age 25 for the G2 and G1 plots. PTAEDA was chosen over other growth and yield models because it carries a long and well-vetted history (Amateis and Burkhart 2016), was not parameterized with data from either of these datasets, and allows an existing diameter distribution that accounts for stem size and stem quality to be inputted. Yield predictions for total tons and three product classes are outputs.

The number of trees by d.b.h. class and the percent of defective stems for each d.b.h. class along with stand age, site index and percent of total stand basal area in non-planted loblolly pine for each plot at the third measurement were inputted to PTAEDA and projected to age 25. Per acre output included total tons, large sawtimber (defined as 12 inches d.b.h. class and above to a top diameter limit of 8 inches), small sawtimber (defined as 8 inches d.b.h. class to the 12 inches d.b.h. class to a top diameter limit of 6 inches) and pulpwood (5 inches d.b.h. class and above to a top diameter limit of 4 inches including large trees not qualified to make sawtimber quality) tons. Topwood from each sawtimber tree to a 4 inch top limit was added to the pulpwood component. Topwood above 4 inches was considered unmerchantable. To be classified as sawtimber a tree had to meet both the size requirements and stem quality requirements of no forking, no excessive sweep, no broken top, and no disease or insect damage on the main stem at the third measurement.

RESULTS

Height-Age Relationship

The Type I SS for equation (1), which disregards the covariate, was significant ($Pr > F = 0.0008$). The Type III SS, which accounts for the age covariate, was not significant ($Pr > F = 0.4623$). Thus the hypothesis that dominant height growth is not different between the two datasets could not be rejected at this early age of stand development. Table 4 presents the results of fitting equation (2) to the height-age data. The 95 percent confidence limits on and overlap for the two datasets. At these ages, the slope of the height growth trajectory is not significantly different between the two populations but the G2 plots are growing toward a higher level than the G1 plots (fig 2).

Growth and Allometry of D.b.h. and Height

Results of fitting equations (3), (4) and (5) by 2-year measurement periods are summarized in table 5. These results suggest that the G2 plots have generally been

growing faster in height and dbh resulting in a stem form that is significantly different than the G1 population.

Stem Quality Assessment

Table 6 summarizes the frequency analyses for stem defects between the two populations. Generally the G1 plots exhibit greater frequencies of defects than the G2 plots. This is particularly evident for the disease and insect damage category. They also noted that faster height growth was weakly associated with more forking and straightness with less forking.

Results of the simulations with PTAEDA (table 7) suggest that productivity of the varietal stands of loblolly pine should be significantly greater than the open pollinated ones. Mean site index for the G2 plots was about 8 percent higher than the G1 plots. Total tons for the G2 plots were about 25 percent greater than the G1 plots. When total tons are merchandized into products, sawtimber tons were on the order of 2 times as much

Table 4—Parameter estimates and fit statistics for fitting equation (2) to the G2 and G1 datasets using nonlinear least squares

Parameter	Estimate	Std. error	Approximate 95 percent confidence limits	
-----G2 (N=98; MSE=12.9)-----				
β_2	0.0746	0.00832	0.0581	0.0911
β_3	1.3835	0.1161	1.1531	1.6139
-----G1 (N=117; MSE=32.8)-----				
β_2	0.0740	0.0110	0.0522	0.0957
β_3	1.4841	0.1899	1.1080	1.8601

MSE=mean squared error.

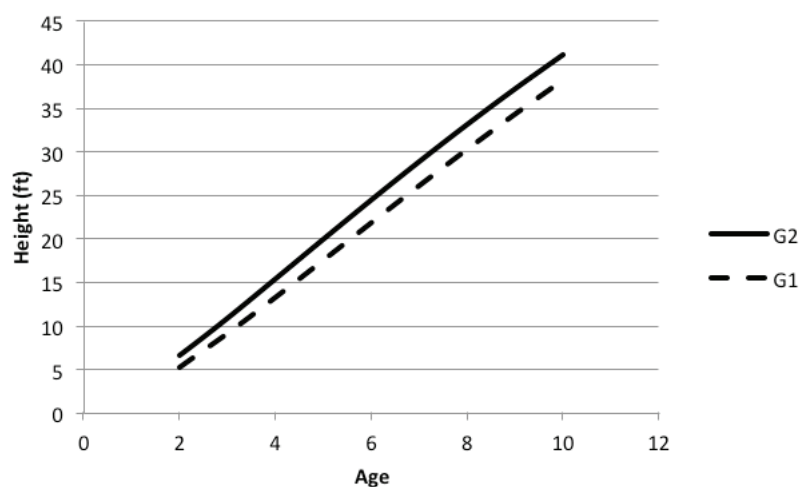


Figure 2—Plot of the Chapman-Richards equation fitted to the G2 and G1 datasets (common asymptote of 100 feet assumed).

Table 5— Analysis of covariance (ANCOVA) results of fitting equations (3) and (4) to 2-year height and d.b.h. growth increment data and equation (5) to three measures of height over d.b.h. for the G1 and G2 groups

Measurement	Mean		Coef. Var.		Type III SS
	G2	G1	G2	G1	Pr>F
-----2-year height growth (feet)-----					
1 - 2	9.0	8.3	30.7	31.8	<.0001
2 - 3	9.0	7.9	39.7	29.0	0.6550
-----2-year diameter growth (inches)-----					
1 - 2	2.0	1.3	34.4	44.1	<.0001
2 - 3	1.7	0.9	33.3	46.2	<.0001
-----Height over d.b.h. ratio-----					
1	9.3	6.5	47.1	33.1	0.0738
2	6.0	6.3	31.3	19.7	0.0020
3	5.7	6.5	24.5	17.4	<.0001

Coef. Var. = coefficient of variation.

Table 6—Frequencies (number of trees) for three types of stem defects (forking, disease and insect damage, and bole or butt sweep) by population and measurement period with Chi-square test of significance and the odds ratio^a

	Measurement 1		Measurement 2		Measurement 3	
	No	Yes	No	Yes	No	Yes
-----Forking-----						
G2	4894	55	4696	155	4266	438
G1	4174	231	4022	330	3527	341
Chi-square	134.3 (Pr<.0001)		88.46 (Pr<.0001)		0.6303 (Pr=0.4273)	
Odds ratio	4.925 (3.659, 6.627)		2.486 (2.045, 3.022)		0.942 (0.812, 1.092)	
-----Disease and insect damage-----						
G2	4888	61	4775	76	4313	391
G1	4021	384	3718	636	3156	713
Chi-square	288.2 (Pr<.0001)		546.8 (Pr<.0001)		193.6 (Pr<.0001)	
Odds ratio	7.652 (5.582, 10.06)		10.75 (8.44, 13.69)		2.492 (2.185,2.843)	
-----Bole or butt sweep-----						
G2	4826	120	4185	617	3550	699
G1	4069	278	3843	411	3091	709
Chi-square	88.9 (Pr<.0001)		22.77 (Pr<.0001)		6.770 (Pr=.0093)	
Odds ratio	2.748 (2.209, 3.417)		0.725 (0.636, 0.828)		1.165 (1.038, 1.3070)	

^a Ninety-five percent lower and upper confidence limits in parentheses.

Table 7—Predicted site index and tons per acre at age 25 for unthinned Coastal Plain G2 and G1 growth and yield plot projections using the PTAEDA model

Variable	Mean	Std. Dev.	Min.	Max.
-----G2-----				
Site index (feet) ^a	79.4	10.4	50.9	100.2
Total tons per acre	213	49	96	343
Pulpwood tons per acre	90	57	14	224
Chip and saw tons per acre	57	32	3	141
Sawtimber tons per acre	43	36	1	154
-----G1-----				
Site index (feet) ^a	73.5	11.5	47.2	99.6
Total tons per acre	168	47	66	276
Pulpwood tons per acre	76	30	30	151
Chip and saw tons per acre	56	33	3	117
Sawtimber tons per acre	12	14	0	48

Std. dev.=standard deviation.

^aSite index estimated using the model of Diéguez-Aranda and others (2006).

(table 7). A t-test of the mean predicted tons at age 25 indicates there is a significant difference in total weight and sawtimber weight between the two populations but no significant difference for pulpwood and chip and saw (all comparisons done at the alpha=0.05 level).

DISCUSSION

Comparing the early growth from the G1 and G2 growth and yield plots showed the varietal plantations growing faster than the open pollinated generation planted in the same general area. It should be noted that the G1 plantations were established with open-pollinated stock available in the late 1980s to early 1990s, while the G2 varietal plantings used were clones available over a roughly 5-year period spanning 2003-2008. The varietal stands have almost no hardwood competition in the main canopy and generally exhibit better form and less disease incidence. At the first and second measurements forking was more likely to occur in the G1 plots than the G2 plots. Xiong and others (2010) found the forking defect to be partially related to family but at the individual level to be mostly determined by environment.

The simulations with PTAEDA are not meant to quantify, in an absolute sense, productivity differences between the two populations. Rather, the intent is to preview, in a relative sense, the difference in productivity that might be expected from predictions made using inventory plot data where climatic, edaphic, planting stock, and

management treatment factors are confounded. While no mid-rotation treatments have been applied in the G2 stands to this point, it's likely that treatments such as thinning and fertilization will be just as effective or even more effective than similar treatments applied to previous generations of plantations. Environmental factors that affect productivity over time may come into play including elevated levels of CO₂, changing amounts and patterns of rainfall and number and intensity of storms.

The implications from this study for managers are considerable. Due to a higher proportion of sawtimber quality trees reaching merchantable size sooner means earlier thinnings and generally shorter rotations will be possible. Nutrient amelioration, where needed, will likely occur sooner. Although the genetic component of variation has been eliminated with varietal planting stock there is still considerable variation in height and diameter growth due to environmental variation.

It is not possible to determine the extent that any particular factor has had on the results found in this study. The effects of site conditions, management treatments, and the planting stock selected for each site are confounded. What can be said from this observational study is that newer plantations established with more advanced silvicultural treatments and planted with clones will likely be more productive than previous populations.

ACKNOWLEDGMENTS

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COMPARISON OF TWO DIAMETER-BASED MEASURES FOR ESTIMATION OF STAND CARRYING CAPACITY

Sheng-I Yang and Harold E. Burkhart

Abstract—Diameter-based maximum size-density measures are useful to estimate stand basal area carrying capacity. In this study, we used an alternate formulation of relative spacing by replacing average dominant tree height with quadratic mean diameter, denoted diameter-based relative spacing (RSD). We compared the efficacy of RSD and Reineke's (1933) self-thinning rule for estimation of stand basal area carrying capacity. Data from a loblolly pine spacing trial, with planting density ranging from 6727 to 747 trees/ha, were used to estimate the coefficients of both measures. RSD largely eliminated the undesirable property of Reineke's self-thinning rule, which overestimated the maximum stand basal area when stands were young.

INTRODUCTION

For plant populations, carrying capacity can be regarded as the maximum number or biomass of a species that a certain environment can support. In forestry, stand carrying capacity can be referred to as the maximum possible stocking of a stand, which is similar in concept to stockability or stocking capacity (DeBell and others 1989, Hall 1983). Stand basal area, the sum of cross-sectional areas of all stem diameters at breast height in a unit area, is an informative expression of stand carrying capacity (Burkhart and Tomé 2012, p. 175-177). Diameter-based maximum size-density measures can be used to estimate stand basal area carrying capacity. Yang and Burkhart (2017) indicated that stand basal area carrying capacity implied by three well-known diameter-based measures (Reineke's self-thinning rule, competition-density rule, and Nilson's sparsity index) were close to the actual observations.

Relative spacing (Hart 1926) is defined as the ratio of the average distance between trees to the mean dominant tree height of stands. The expression of relative spacing can be written as:

$$RS = \frac{\sqrt{\frac{10000}{N}}}{\bar{H}_d} = f(A)$$

where

N = number of trees per ha

$\bar{H}_d$ = average dominant tree height (m)

A = stand age (yrs).

Relative spacing is a commonly used measure of stand stocking, which has been widely applied to develop stand density management diagrams (Barrio-Anta and González 2005, López-Sánchez and Rodríguez-Soalleiro 2009). However, unlike diameter-based measures, relative spacing cannot be easily converted to diameter-related stand variables (e.g., stand basal area) by simple transformation or substitution of the equation.

Therefore, we used an alternate formulation of relative spacing by replacing mean dominant tree height with quadratic mean diameter, denoted diameter-based relative spacing (RSD). The purpose of this study was to compare and evaluate the efficacy of RSD and Reineke's (1933) self-thinning rule for estimation of stand basal area carrying capacity.

MATERIALS AND METHODS

Spacing Trials

Measurements were obtained from a loblolly pine spacing trial that employs a nonsystematic design introduced by Lin and Morse (1975). In 1983, the experiment was established at four sites, two in the upper Coastal Plain and two in the Piedmont. At each site, three nearly contiguous factorial blocks were established. A spacing factor (F) of 1.2 m (4 feet) was chosen in each block. Four levels of the factor (1F, 1.5F, 2F, and 3F) were randomly assigned to row and column positions. Accordingly, 16 plots made up a compact block: four square plots (1.2×1.2, 1.8×1.8, 2.4×2.4, 3.6×3.6 m), and 12 rectangular plots (1.2×1.8, 1.2×2.4, 1.2×3.6, 1.8×1.2, 1.8×2.4, 1.8×3.6, 2.4×1.2, 2.4×1.8, 2.4×3.6, 3.6×1.2, 3.6×1.8, 3.6×2.4 m). Forty-nine trees

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were planted in each plot. The initial planting densities ranged from 747 to 6727 trees/ha. Other than controlling competing vegetation with chemical herbicides in the first 3 years, no other management treatments were applied. An overview of the study design and field procedures is presented in Amateis and Burkhart (2012).

Groundline diameter (GLD) was measured at ages 1-5 and diameter at breast height (DBH) was recorded annually for ages 5-25. Total tree height was measured annually for ages 1-10 and biennially for ages 12-25. Due to more than half of the plots being damaged by mortality agents other than competition-induced tree mortality after age 18, only the measurements from ages 5-18 were included in the analyses. Quadratic mean diameter, average dominant tree height, number of trees per unit area, and BA per unit area (stand basal area) were computed for annual measurements taken at ages 5-18.

BA Implied by Diameter-based Relative Spacing

Diameter-based relative spacing (RSD) was defined as the ratio of the mean distance between trees to quadratic mean diameter ($\bar{D}_q$) expressed in cm. By letting RSD be a function of stand age (A), the equation becomes:

$$RSD = \frac{\sqrt{\frac{10000}{N}}}{\bar{D}_q} = f(A)$$

Stand basal area carrying capacity can be obtained by squaring both sides of the preceding equation and then multiplying by the coefficient k (≈ 0.00007854). That is,

$$G_{RSD} = \frac{10000k}{[f(A)]^2}$$

where

G_{RSD} is the stand basal area carrying capacity implied by RSD. The details of derivation can be found in Yang and Burkhart (2018). G_{RSD} was estimated using nonlinear quantile regression (quantile=0.01) in R.

BA Implied By Reineke's Self-thinning Rule

Reineke's self-thinning rule is a well-known maximum size-density relationship measure to describe potential stand density. Reineke (1933) found that the maximum

stand density (N) in trees per unit area and quadratic mean diameter ($\bar{D}_q$) follows a linear relationship on a log-log scale. Reineke's self-thinning rule is expressed in the form:

$$\ln N = a_0 + a_1 \ln \bar{D}_q$$

where

$\ln$ = natural logarithm; a_0 , a_1 = coefficients. Reineke (1933) indicated that the slope (a_1), close to -1.6, was generally consistent among species and regions but that the value of the intercept (a_0) varied. However, because subsequent studies questioned the assumption of constant slope (Binkley 1984, Zeide 1985), the slope coefficient was estimated in this study.

Rearranging equation 1 gives:

$$N = e^{a_0} \bar{D}_q^{a_1}$$

Stand basal area carrying capacity implied by Reineke's self-thinning rule can be obtained by inserting N in the equation for stand basal area. That is,

$$G_R = N \bar{D}_q^2 k = e^{a_0} \bar{D}_q^{a_1+2} k$$

where

G_R is the stand basal area carrying capacity implied by Reineke's self-thinning rule. G_R was estimated using quantile regression (quantile=0.01) in R.

BA Reference Curve

Stand basal area carrying capacity (G) was estimated directly by fitting the Chapman-Richards equation using nonlinear quantile regression (quantile=0.01) in R:

$$G = b_0 (1 - e^{b_1 A})^{b_2}$$

where

b_0 , b_1 , b_2 = coefficients. G functioned as a reference curve when comparing the stand basal area carrying capacity implied by RSD and Reineke's self-thinning rule.

RESULTS AND DISCUSSION

As shown in figure 1, in the older stands, stand basal area carrying capacity implied by RSD and Reineke's self-thinning rule are close to the observed values and values estimated by the Chapman-Richards equation. When stands were young, however, the maximum values implied by Reineke's self-thinning rule were much higher than the implied values of RSD. Consequently, Reineke's self-thinning rule overestimated the maximum stand basal area in the young stands. In contrast, RSD mitigated the undesirable property shown by Reineke's self-thinning rule and described the dynamics of basal area stocking more accurately.

CONCLUSIONS

Stand basal area carrying capacity implied by diameter-based relative spacing (RSD) followed closely the reference curve fitted by the Chapman-Richards equation. RSD is a more appropriate representation of the dynamics of basal area stocking than Reineke's self-thinning rule. Although these initial analyses showed that RSD is a reliable diameter-based measure, further investigations of the properties of RSD are warranted.

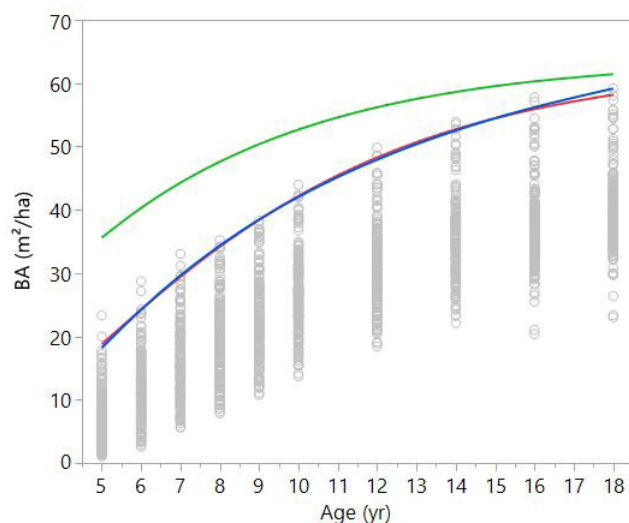


Figure 1 — Comparison of stand basal area carrying capacity estimated by diameter-based relative spacing (RSD) (blue line), Reineke's self-thinning rule (green line), and Chapman-Richards equation (red line).

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IF SURVIVAL MATTERS, SHOULD REGENERATION STUDIES HAVE MORE REPLICATIONS?

David B. South and Curtis L. VanderSchaaf

Abstract—When it comes to testing for differences in seedling survival, researchers sometimes make Type II statistical errors due to the inherent variability associated with survival in tree planting studies. For example, in one trial (with five replications) first-year survival of seedlings planted in October (42 percent) was not significantly different ($\alpha=0.05$) from those planted in December (69 percent). Did planting in a dry October truly have no effect on survival? Authors who make a Type II error might not be aware that as seedling survival decreases (up to an overall average of 50 percent survival), statistical power declines. As a result, the ability to declare an 8 percentage point difference as “significant” is very difficult when survival averages 90 percent or less. We estimate that about half of regeneration trials (average survival of pines <90 percent) cannot declare a 12 percent difference as statistically significant ($\alpha=0.05$). When researchers realize their tree planting trials have low statistical power, they should consider using more replications. Other ways to increase power include: (1) use a 0.1 α value which increases the Type I error (2) use a potentially more powerful contrast test (instead of an overall treatment F-test) and (3) conduct survival trials under a roof. Alternative methods include modeling survival data (instead of applying statistics to separate mean values) and simply estimating the treatment effect size with confidence intervals.

INTRODUCTION

Although researchers often fail to reject the null hypothesis, they should never accept a null hypothesis. Even so, often researchers conclude that various treatments did not affect seedling survival. This may be true when the difference between means is very small. However, in several studies, an examined factor was not significant ($\alpha=0.05$) but the treatment increased survival of pine seedlings by at least 20 percentage points (table 1). Could this type of increase be both biologically significant and statistically insignificant? This certainly can happen when the study design has low statistical power.

There is always a chance that some of our conclusions about treatment effects are wrong. In some trials researchers might conclude the treatment worked, but in reality it didn't (Type I error). More commonly we say the treatment had no effect on seedling survival, but there really was a treatment effect (Type II error). Examples of possible Type II errors are provided in table 1. Because of a combination of limited resources and tradition, the design of most seedling survival trials has insufficient replication to detect a “true” 8 percent difference in survival. This is because most researchers (80 percent) use four replications or less. Installing additional replications will cost more, but it will also improve the power of survival tests (i.e., $1 - \beta$ value).

We have noticed that statistically, it may be relatively easy to declare a 30 cm increase in height as significant, but hard to declare an 8 percent difference in seedling survival as statistically significant. This is partly because when average survival becomes < 90 percent, the standard errors increase. There are various ways to increase the power of our tests (table 2), but many choose to use less than 5 replications and some analyze trials with a simple multiple-range test (see various examples in volumes 1 to 14 of the Biennial Southern Silvicultural Research Conferences).

We and others have pointed out the importance of replications in both nursery (VanderSchaaf and others 2003) and field trials (Foster 2001, Zedeker and others 1993). In a previous paper, we examined the decline in power as plantations get older and increase in biomass (South and VanderSchaaf 2006). In this paper, we examine the impact of replication on the (a priori) power of establishment trials that report survival. In several cases, the power of the statistical test was so low that it was not able to declare a 20 percent difference as significant. Perhaps one reason authors don't typically report statistical power (Peterman 1990) is because the statistical power is low or because they only care about making Type I statistical errors. This paper examines various ways to increase the power of regeneration trials.

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Table 1—Examples of regeneration studies published in Biennial Southern Silvicultural Research Conference Proceedings where the increase in survival exceeded 8 percent but was not statistically significant ($\alpha = 0.05$). A Type II error is likely (see comments) since the power of the test was too low to declare the increase in survival as statistically significant

Treatment	Treatment survival (%)	Treatment increased survival (%)	Comment	Year	Page
Container stock	89	9	10 percent increase was significant at 0.05	1983	129
Shear and pile	82	10	Height and diameter growth increased	1983	43
September herbicide	90	13	Weed control was significant at 0.05	2008	220
High site preparation	71	15	Was significant at 0.05 4 years earlier	1988	166
Lifting in January	80	19	Was significant at 0.01 the previous year	1994	360
March planting-LA	99	20	LSD = 8 percent for Arkansas planting dates	1980	22
Wide spacing	76	21	High stocking increases mortality	2003	434
Nursery #19	56	22	27 percent increase was significant at 0.05	1988	154
December planting	69	27	29 percent increase was significant at 0.05	1984	126
Control	75	32	Fire killed seedlings $P > F = 0.067$	2013	139
Herbicides	58	38	Greatest level of weed control	1988	345

Year = year of meeting; Page = location of treatment means.

Table 2—The probability of correctly rejecting a null hypothesis (when the null hypothesis is false) is influenced by several factors

Factor	Statistical power increases with	Comment	Examples
Number of replications	More replications	Increases error degrees of freedom	Figures 2 and 3
Number of treatments	More treatments	Increases error degrees of freedom	Land and others 2004
Alpha value	A larger alpha value	Increases Type I errors	Xydias 1983
Type of test	A one-tailed test	Test only for a detrimental effect	Paquette and others 2011
Contrast statements	Use of contrast tests statements	Reduces P-values	Blake and South 1991
Variability among plots	Less experimental noise	Use trained hand-planters	Rowan 1987
Difference in means	Larger mean difference	Withhold rainfall to increase mortality	South and others 2012
Blocking on variable sites	Correct blocking	Typically reduces error term	South and Miller 2007

METHODS

Simulation Studies

Computer simulations were created to examine the effects of replication on standard error, standard deviation, least significant difference (LSD) and power. Simulation #1 assumed a completely randomized design (CRD) with two treatments. Plot size for experimental units did not vary (i.e. the experimental area doubled when the number of replications doubled). Simulation #2 involved a randomized complete block design (RCB) with two treatments and a total of 200 seedlings per

treatment. The numbers of replications simulated were 4, 8, 10 and 20, with 50, 25, 20 and 10 seedlings planted per experimental unit, respectively.

Literature Survey

We selected 50 papers that were published in the proceedings of the Biennial Southern Silvicultural Research Conference (BSSRC). Only papers that involved pine seedlings and detected a significant difference are included in table 3. Papers that did not report either F-test or mean comparison test results

Table 3—Examples of survival means (A:B) reported from 50 selected papers published in various volumes of the proceedings of the Biennial Southern Silvicultural Research Conferences. The mean separation test detected a significant ($\alpha = 0.05$) difference between A and B. In most of these trials, B-A is slightly larger than a least significant difference (LSD) value.

A	B	B-A (%)	Species	Year	Page	Replications	A	B	B-A (%)	Species	Year	Page	Replications
95	98	3	Loblolly	1999	337	15-RCB	73	84	11	Loblolly	1986	69	3-RCB
79	82	3	Loblolly	2003	415	4-RCB	87	98	11	loblolly	1986	215	5-RCB
95	99	4	Loblolly	2005	137	3-RCB	67	79	12	Longleaf	1986	223	4-RCB
95	99	4	Shortleaf	1992	266	4-RCB	77	89	12	White	1986	246	3-RCB
92	96	4	Shortleaf	2003	421	6-RCB	21	33	12	Loblolly	1990	158	8-RCB
92	96	4	Loblolly	2011	262	4-RCB	69	81	12	Longleaf	2001	107	3-RCB
90	95	5	Shortleaf	1986	362	3-RCB	64	77	13	Loblolly	1988	131	3-RCB
74	79	5	Slash	1984	84	4-RCB	74	87	13	Loblolly	2001	263	4-RCB
91	96	5	Slash	1984	323	3-RCB	64	77	13	Loblolly	1988	131	3-RCB
72	77	5	Loblolly	1994	360	4-RCB	74	88	14	Loblolly	1982	16	3-RCB
83	88	5	Slash	2005	187	4-RCB	65	79	14	Loblolly	1982	21	15-RCB
92	98	6	Loblolly	1990	150	5-RCB	56	70	14	Pit-loblolly	1984	372	4-RCB
91	98	7	Loblolly + Virginia	1982	143	3-RCB	56	70	14	Longleaf	1988	166	4-RCB
88	95	7	Longleaf	2005	96	6-RCB	62	76	14	Loblolly	1984	413	5-RCB
73	80	7	Longleaf	2005	87	3-RCB	22	37	15	Loblolly	1984	264	4-RCB
89	97	8	Loblolly	2001	53	3-RCB	61	77	16	Loblolly	2001	258	3-RCB
92	100	8	Loblolly	2003	334	4-RCB	74	91	17	Loblolly	1988	345	3-RCB
84	92	8	Loblolly	2005	169	3-RCB	42	59	17	Loblolly	1988	143	12-???
66	75	9	Longleaf	1984	398	5-RCB	53	71	18	Longleaf	1982	72	3-CRD
89	98	9	Loblolly	1990	105	4-RCB	78	97	19	Loblolly	1990	42	3-RCB
82	91	9	Loblolly	1994	238	3-RCB	69	89	20	Sand	1982	99	3-RCB
74	84	10	Loblolly	1980	93	3-RCB	43	64	21	Loblolly	2003	434	4-RCB
79	89	10	White	1982	129	6-RCB	63	86	23	Loblolly	2008	220	4-RCB
80	90	10	Loblolly	2005	122	3-strip-block	29	52	23	Loblolly	1988	154	8-RCB
75	86	11	Slash	1986	46	2-RCB	42	71	29	Loblolly	1984	126	5-RCB

Year = year of meeting; Page = location of means A and B; RCB = Randomized complete block; CRD = completely randomized design.

(e.g., Hassan and Silva 1999) were not included in the survey. In cases where multiple pairs of means were declared different, the pair with the smallest difference was selected. Various mean comparison tests were included and in some cases, the type of test used was not reported. In theory, most values listed in table 3 are slightly greater than an LSD value. The difference between the two means (B-A) was plotted on the y-axis and the lower mean (A) was plotted on the x-axis (fig. 1).

RESULTS

Simulations that varied replications revealed contrasting results. The standard error decreased when doubling replications involved doubling the area planted (fig. 2) but the standard error increased when doubling replications resulted in smaller (i.e. fewer seedlings per plot) experimental units (fig. 3).

The BSSRC survey suggests that an 8-percent difference in survival may be declared significant in less than 3 out of 10 trials. More than 1 out of 5 trials cannot detect a 15 percent difference in survival (fig. 1). Due to low statistical power (1-beta), many trials are not able to detect a significant difference in survival, even when the treatment caused a “real” increase in survival.

The survey data also indicate that statistical power declines as seedling mortality increases. When survival (of the lowest mean) is more than 70 percent, an 8 percent increase in survival might be declared significant perhaps 4 out of 10 times. However, the chance is near zero when survival (of the lowest mean) is less than 70 percent survival (fig. 1). The results from the simulated data are similar to those obtained from published results (fig. 4).

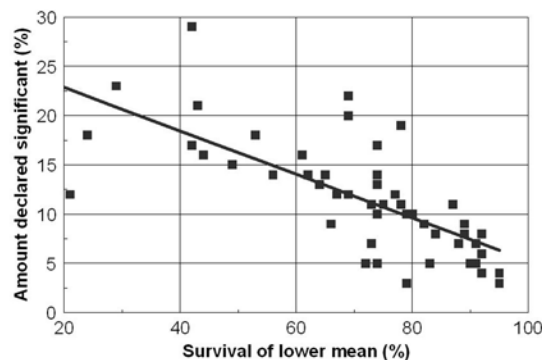


Figure 1—The relationship between the smallest significant difference (B-A from table 3) and lower survival value (A from table 3). Detecting a 10 percent difference in survival is likely when average seedling survival is > 90 percent but unlikely when average survival is < 70 percent. For this dataset, the LSD values appear to increase as the survival of the lowest treatment mean decreases.

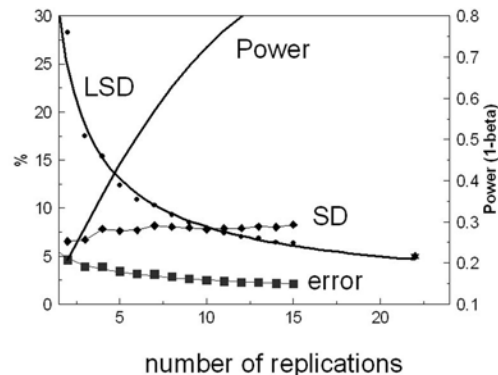


Figure 2—Simulation #1 used a completely randomized design with two treatment means (85 percent and 90 percent). Each data point in the graph represents the average value from 100 simulations. The size of each experimental unit was the same for each data point. Then average values for the standard error, standard deviation (SD) and least significant difference (LSD) vary with the number of replications. A 15 percent LSD ($\alpha = 0.05$) can be expected using four replications while a 10 percent LSD requires seven replications. The star represents an LSD value from a spacing study that contained 22 replications. The standard error of the mean (squares) and standard deviations (diamonds) are also plotted. An *a priori* power line (1-beta) is plotted assuming a constant standard deviation of 7, $\alpha = 0.05$ and an 8 percent survival difference between two means.

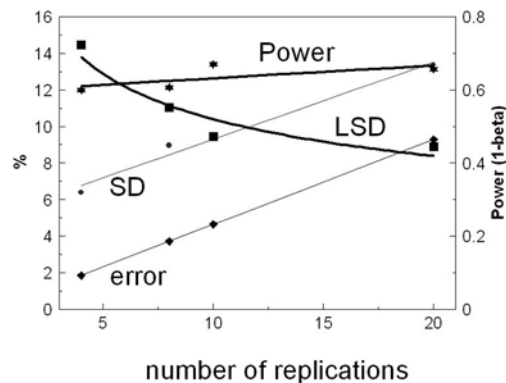


Figure 3—Simulation #2 involved randomized complete block design with two treatment means (70 percent and 80 percent). Each data point in the graph represents the average value from 10 simulations. The relationship between the least significant difference (LSD) and number of replications in a simulated study where the size of each experimental unit decreased as the number of replications increased. This simulation involved planting 200 simulated seedlings per treatment, regardless of the number of replications (i.e., a total of 400 seedlings planted). A 14 percent LSD ($\alpha = 0.05$) was achieved using four replications (50 seedlings per plot) while a 9 percent LSD requires 20 replications (10 seedlings per plot).

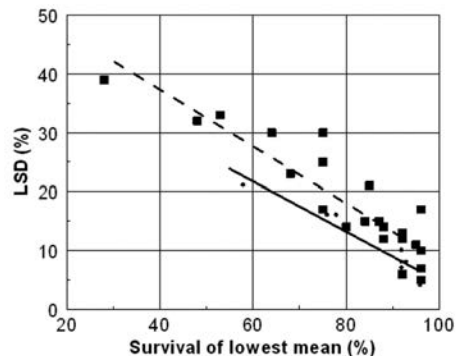


Figure 4—The relationship between the least significant difference (LSD) and seedling survival (for the lowest reported mean out of six treatment means in a randomized complete block design), for stored pine seedlings reported by Jackson and others (2012 - table 5). As survival decreases, the LSD values increased. Each square (dashed line) represents four replications and each dot (solid line) represents 12 replications. There were 30 seedlings per experimental unit.

DISCUSSION

Prior to installing a seedling survival trial, researchers may ask how to design a trial so it might have a chance of detecting an 8 percent difference in survival. The following discusses several options one might consider.

Increasing Power by Increasing Study Area

Sometimes doubling the number of replications will double the study area and hence the number of seedlings planted. Increasing the number of replications (from 4 to 8) will increase power and will decrease the LSD value (fig. 1). The effect of simulated replications is shown in figure 2 and an actual case is illustrated in figure 4. As replication increases, the standard error and LSD values decrease. In contrast, the standard deviations and coefficient of variation increase slightly (from two to seven replications) because more variability is entered into the system as the total number of experimental units increase. In figure 2, a survival difference of 16 percent was detected with four replications while an 8 percent difference was detected with 10 replications.

Reducing the LSD by Reducing the Size of the Experimental Unit

For short-term trials, it is possible to increase the number of replications without increasing the area planted. For example, a study with four blocks and 50 seedlings per experimental unit will require the same number of seedlings as a study with eight blocks and 25 seedlings per experimental unit. The trial with eight replications will have about the same power as one with four replications, but the LSD will likely be smaller (fig. 3).

Do Not Use Pseudoreplication

Pseudoreplication occurs when treatments are not replicated but an analysis of variance (ANOVA) is carried out by assuming sub-samples (or in some cases individual trees) are the same as replication. A major factor that leads to pseudoreplication is an inability (or reluctance) of the author to define the correct experimental unit. Some journal editors do not require ANOVA tables be included in manuscripts. Therefore, one might never know if the error term involved 13 or 2841 degrees of freedom. Reviewers who would normally reject non-replicated trials can be fooled into thinking that a statistical analysis (that involves pseudoreplication) is valid. Although some forestry research involves pseudoreplication (de Souza and others 2016, Dong and Burdett 1986, Kamaluddin and others 2005, Smith 1989), it should not be used as a method to reject a null hypothesis.

Higher Alpha Values Increase Power

A method some silviculturists choose to increase the power of a tree planting test is to raise the alpha value (table 4) and increase the probability of a Type I error. Some wisely use a 0.1 level to reduce the Type II error (Amishev and Fox 2006; Cram and others 2007; Curtis and others 2015; Dean and others 2013; Scott and Stagg 2013; Walker and others 1981, 1985). In some cases an alpha value of 0.15 has been used (Haywood and others 1998, Xydias 1983). Regardless of the alpha value selected, authors should list the actual p-value for treatment effects.

Pre-planned Contrast Tests Increase Power

In most of the studies in table 3, means were compared using a test such as Duncan's multiple range test, Tukey's test and Newman-Keuls test. A few used contrast procedures (Warren 1986) to test for treatment effects. An example of this method is illustrated in a study with 11 nursery treatments and four replications (Blake and South 1991). When survival ranged from 89 percent to 98 percent, an F-test ($p=0.68$) suggested no treatment effect. However, a pre-planned contrast test (comparing only top-pruned treatments) revealed a significant ($\alpha=0.05$) treatment effect.

LSD as an Indication of Statistical Power

Although reporting statistical power is important (Peterman 1990), most BSSRC papers do not provide any indication of the power of the test (i.e., no LSD values and no beta values). However, when comparing studies with the same experimental design, LSD values do provide the reader with some indication of the statistical power (Nemec 1991). In one study (Jackson and others 2012), LSD values were reported when comparing six treatment means. This provided an opportunity to compare published LSD values (fig. 4) with figure 1. As expected, LSD values for pine seedlings (outplanted in sand pits) increased as the percent survival (of the lowest mean) decreased. Increasing the number of replications decreased the LSD values (fig. 4). When the lowest mean was 80 percent, the LSD for four replications was about 18 percent and replicating 12 times reduced this to 14 percent. The predicted LSD values (fig. 2) were similar in magnitude to those in figure 4. We recommend researchers routinely report LSD values when reporting survival means. This will provide the reader with some idea of the power of the test. However, in certain cases, a lower LSD value is not associated with higher statistical power (fig. 3).

Table 4—Effect of alpha value and number of replications on statistical power (1-beta) of experiments with 7 percent standard deviation in survival, (two means of 88 and 80 and standard deviation =7)

ALPHA	4 replications	6 replications	10 replications	15 replications
0.15	0.57	0.70	0.86	0.95
0.1	0.48	0.63	0.81	0.93
0.05	0.36	0.50	0.72	0.87

Table generated using University of British Columbia Web site: <https://www.stat.ubc.ca/~rollin/stats/ssize/n2.html>.

Roofed Survival Trials

When the objective is to test nursery practices or tree planting treatments on first-year survival, then we highly recommend the planting site include a roof that protects seedlings from rainfall. Much time and effort has been wasted designing and installing outdoor studies that end up with adequate rainfall and high survival. For example, in one study (South and others 2012), treatments that received rainfall resulted in a LSD of 5.3 but the difference in survival was only 1 percent (95 percent vs. 96 percent). However, when these seedlings were planted under a roof and exposed to a 4-month drought, the treatment effect was significant ($P = 0.007$; $LSD = 14.6$; means were 28 percent and 74 percent). Approximately one-third of the studies listed in table 3 could have been established in a roofed stress house.

Use figure 1 to Evaluate Your Study

After analyzing a seedling survival trial, researchers may want to evaluate their study design. A simple way is to plot the LSD value ($\alpha = 0.05$) on the Y-axis of figure 1 with the lowest treatment mean on the X-axis. If the point is below the slope-line, your study is above average. However, if the point is above the line, you might need to increase the number of replications in future trials.

CONCLUSIONS

Researchers should establish trials with the knowledge that statistical power is not the same for all response variables. When testing differences in survival, power will decline as average survival decreases (until the average study survival reaches 50 percent). With only four replications, the probability of a Type II error may be high. A treatment that produces a “true” 8 percent increase in survival will likely be declared not significant if the lowest treatment mean is < 80 percent. However, just because a treatment was “non-significant” does not automatically mean the treatment was ineffective. Researchers should remember that they can never “prove” a null hypothesis and therefore it would be unscientific to accept a null hypothesis (even if the experiment had high statistical power). We can only fail to reject the null hypothesis, especially when we establish studies with insufficient replications.

ACKNOWLEDGMENTS

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Pine Bark Beetles

Moderator:

Joshua Adams

Louisiana Tech University

BIOGEOCHEMICAL HOTSPOTS AROUND BARK-BEETLE KILLED TREES

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Natalie A. Clay, Juliet D. Tang, Nicole Hornslein, and Brent L. Chaney

Abstract—Disturbance-induced mortality events in forest ecosystems generate significant hotspots in biogeochemical cycles. These events occur sporadically across the landscape and contribute to large sources of error in terrestrial biosphere carbon models, which have yet to capture the full complexity of biotic and abiotic factors driving ecological processes in the terrestrial environment. The balance between production of stable soil organic matter and respiration from decomposing biomass greatly influences whether temperate forests remain modest carbon sinks or are transformed into carbon sources. In 2015, a field experiment to mimic pine beetle attack was established by girdling loblolly pine trees. Subsequent measurements of throughfall and stemflow for water quantity and quality, transpiration, stem respiration, soil respiration, and soil chemistry were used to quantify the extent of spatial and temporal impacts of tree mortality on carbon budgets. Enhanced fluxes from dying trees primed surrounding soils while decreased tree water use provided additional soil moisture to create biogeochemical hotspots, which could lead to accelerated carbon decomposition and mineralization rates.

INTRODUCTION

Disturbances in forest ecosystems can alter functions like productivity, respiration, and nutrient cycling. Common landscape-scale disturbances such as fire, drought, windstorms, and insect outbreaks increase coarse woody debris inputs and alter the hydrology and biogeochemical processes in forests (Harmon and others 1986, Mikkelsen and others 2012). Disturbances vary in their spatio-temporal extent and can have significant impacts on terrestrial carbon (C) cycling, but they are commonly omitted from terrestrial C models.

A major disturbance agent in the Southeastern United States is the southern pine beetle (*Dendroctonus frontalis*). This native bark beetle species infests many species of southern pine, but most commonly loblolly pine (*Pinus taeda* L.). Southern pine beetles are a particularly aggressive bark beetle species capable of quickly killing a vast number of trees. Outbreaks across the region are estimated to cost timber producers approximately \$43 million annually, although some of these costs may be recovered through timely salvage logging operations (Pye and others 2011). Historically, these outbreaks occurred on 5–7 year cycles with a

duration of 2–3 years, although recent observations indicate a decline in outbreak activity (Asaro and others 2017). Forest management and outbreak reduction and mitigation following bark beetle disturbance have received much attention, but ecosystem impacts like nutrient fluxes have received much less (Clarke and others 2016).

Both water quantity and quality are impacted by bark beetle-killed trees, although much of this research comes from the Western United States due to the extensive damage from mountain pine beetle (*D. ponderosae*) (Kurz and others 2008, Mikkelsen and others 2012, Morehouse and others 2008). The cascade of changes observed by Brouillard and others (2016) include (1) the increase in streamflow due to a reduction in water uptake and transpiration coupled with decreased canopy interception and (2) the release of carbon from biomass decomposition which is returned to the atmosphere via heterotrophic respiration or leaching into surface waters, increasing dissolved organic carbon loads. Typically, this carbon is more aromatic and composed of complex and recalcitrant molecules. The negative charge of dissolved carbon enables the transport of nutrient-rich base

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cations and heavy metals. This, coupled with the more aromatic and recalcitrant nature of these compounds, leads to increased costs of water treatment services downstream (Mikkelsen et al 2012; Brouillard et al 2016).

Water is the dominant pathway of nutrient removal in these ecosystems, and initial rainfall inputs likely contribute to watershed fluxes. Precipitation and nutrients move in distinct pathways through the forest canopy to the forest floor; this water is called throughfall and can impact hydrology and nutrient fluxes throughout a watershed. This pathway may or may not come into contact with tree surfaces, and if it does, residence time is usually minimal. Depending on canopy structure, throughfall can exhibit a large degree of spatial variability (Siegert et al 2016). Stemflow is rainfall that has been captured by the forest canopy and funneled down woody surfaces to be deposited at the base of the tree. Stemflow has much longer residence times and therefore is an important pathway in nutrient cycling. Both of these pathways may become enriched with nutrients and other solutes via washoff of dry deposition from antecedent dry periods or canopy leaching. Precipitation characteristics including magnitude, duration, and intensity are also strong determinants of nutrient flux (Van Stan et al 2016; Nanko et al 2016; Siegert et al 2017). Spots of southern pine beetle-killed trees likely impact these processes.

We hypothesized that bark beetle-killed trees undergoing mortality create biogeochemical hotspots in the vicinity of their trunk due to (1) increased soil moisture from reductions in plant water uptake and increased stemflow production, (2) enhanced canopy-derived inputs of carbon and nitrogen (N), and (3) increased microbial activity and root mortality.

MATERIALS AND METHODS

Study Sites

The study was conducted in a 60-year-old loblolly pine stand in central Mississippi (33.2639°N, 88.8884°W) (fig. 1A). The overstory basal area was 25.5 m² ha⁻¹ with 395 trees ha⁻¹. The midstory had a basal area of 29.4 m² ha⁻¹ with 1,664 trees ha⁻¹ dominated by sweetgum (*Liquidambar styraciflua* L.), red maple (*Acer rubrum* L.), winged elm (*Ulmus alata* Michx.), and oak species (*Quercus* spp.). The soil on this site is a somewhat poorly drained Urbo silt loam, with a depth to the water table of 30–35 cm and occasional flooding (Natural Resources Conservation Service 2015). Average temperatures in summer (June, July, and August) and winter (December, January, and February) are 26.5 °C and 6.9 °C, respectively (30-year mean: Arguez 2010). Total

annual precipitation is 140.3 cm, which falls fairly evenly throughout the year with the lowest rainfall occurring in September (8.6 cm) (Arguez 2010).

Study Design and Treatments

To test our predictions, we established a field study in Summer 2015 and simulated a bark beetle infestation by girdling loblolly pine trees to sever phloem and cambium tissue (Davis et al 2017). Fifteen canopy-dominant loblolly pine trees were selected around a centrally located data logger and randomly allocated to three different treatments: 1) five trees were girdled and received inoculations of bluestain fungus (*Ophiostoma minus*), 2) five trees were girdled and inoculated with agar as negative controls, and 3) five trees were not girdled and received no inoculations as controls (fig. 1B). Southern pine beetles are vectors of bluestain fungus, which is a non-decay fungus that colonizes the vascular system, and may accelerate death due to restrictions of plant water uptake, and preferentially attracts subterranean termites (Little et al 2012; Clay et al 2017).

Measurements

All trees were outfitted with stemflow collars constructed from 2.5 cm inner diameter polyethylene tubing cut longitudinally and sealed around the trunk of each tree above the girdling and inoculation site with aluminum nails and silicone caulk (fig. 1C). Collars drained into 20 L polyethylene bins. Stemflow volumetric flux was measured in the bins and homogenized samples were collected within 24 hours of rainfall events, filtered through a 0.45 µm membrane, and stored at 4 °C until chemical analysis. Dissolved organic carbon (DOC) was analyzed on a Hach DR5000 spectrometer, dissolved organic matter (DOM) absorbance characteristics were analyzed on a Lambda 850 spectrometer, and dissolved nitrogen species [total nitrogen (TN), organic nitrogen (ON), nitrate (NO₃-), and ammonium (NH₄+)] were analyzed on a Bran+Luebbe autoanalyzer. The spectral slope ratio (S_R) is a DOM absorption metric that describes the molecular weight of a sample and is calculated by:

$$S_R = \frac{S_{275-295}}{S_{350-400}} \quad (1)$$

where

$S_{275-295}$ = the slope of the absorption coefficient between wavelengths 275–295 nm

$S_{350-400}$ = the slope of the absorption coefficient between wavelengths 350–400 nm (Helms et al 2008).

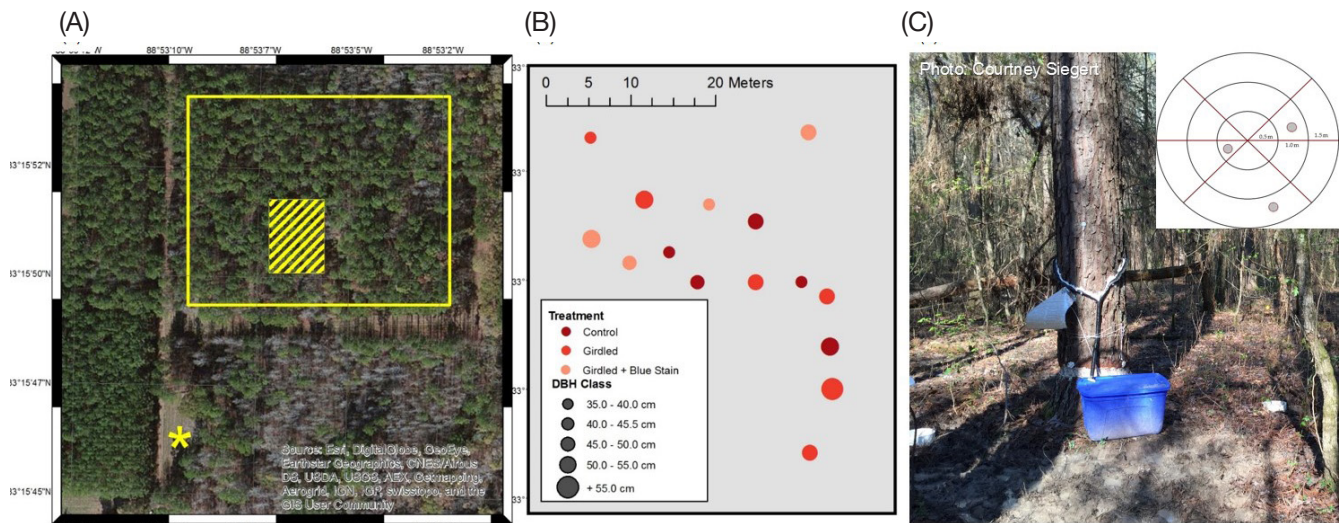


Figure 1—(A) Location of study on the John W. Starr Memorial Forest in central Mississippi denoted by the thatched rectangle. Open precipitation was collected in a nearby clearing (yellow star). (B) Layout of trees selected for the study and their relative diameter at breast height (DBH). (C) Individual tree instrumentation including sapflow sensors hidden under silver radiation shield, stemflow collar draining into collection bin, and soil respiration collars around tree bole. Upper right diagram illustrates soil chemistry and respiration sampling design with study tree located in the middle of the schematic.

The absorption coefficient (a_{254}) is another metric that describes the aromaticity of a compound and is calculated by:

$$a_{254} = \frac{2.303 \times A(\lambda)}{l} \quad (2)$$

where

$A(\lambda)$ = the absorbance at 254 nm

l = the cell path length of the instrument, set to 0.01 m (Green and Blough 1994)

The specific UV absorbance (SUVA) also describes compound aromaticity and standardized for the concentration of DOC in the sample by:

$$SUVA_{254} = \frac{a_{254}}{[DOC]} \quad (3)$$

where

$[DOC]$ = the concentration of dissolved organic carbon in the sample (Weishaar and others 2003).

Sample collection began in fall 2015 and continued through fall 2016 representing nine rainfall events large enough to generate stemflow. DOC and DOM analyses were conducted on a subset of sample dates on November 18, 2015, March 24, 2016, and June 6, 2016.

On all 15 study trees, sapflow probes were installed to measure tree water use following the heat dissipation method (Granier 1987) (fig. 1C). Sapflow probes were connected to a CR1000 datalogger (Campbell Scientific Inc.) and powered by deep cycle batteries and a solar panel located in a nearby canopy gap. Measurements were recorded every 30 seconds and averaged over 30 minutes.

Soil elemental composition and respiration were measured around individual study trees. A circular grid with concentric rings of three 0.5-m intervals split into six sample quadrants was established around each tree (fig. 1C). Soil samples were taken at the initiation of the study in fall 2015 and again in fall 2016 with a 2.5-cm soil auger in the A and B horizons (5 cm and 10 cm, respectively). Two sample points were randomly selected in each of the three distance intervals radiating away from the tree boles. Different random points were used for each measurement date. These samples were dried at room temperature, ground to pass through a 0.1-mm sieve, oven dried at 105 °C for 24 hours, and then stored in air tight Whirl-pak bags. Carbon and nitrogen were determined on a Costech 4010 ECS CHNO elemental analyzer.

In each of the three concentric distance quadrants, a 20-cm inner diameter polychlorinated vinyl tube cut to a length of 10 cm was installed permanently into the soil profile (fig. 1C). Installation locations were randomly generated, but these locations remained fixed

throughout the study to limit disturbance of the soil and root profile. Respiration was measured monthly with a LI8100A Soil Automated Flux Analyzer (Li-Cor Inc) outfitted with a 20-cm survey chamber along with volumetric water content and soil temperature. Respiration data were adjusted based on the average measured depth of each PVC collar. Monthly measurements began in spring 2016 and continued throughout winter 2016/17 for as long as respiration was still measurable.

Data Analysis and Statistics

A two-way factorial design analysis of variance (ANOVA) was used to identify differences in sapflow, stemflow volume, stemflow chemistry, and soil respiration as each responded to girdling treatments, sampling date, and the interacting effects between the two treatments. For soil chemistry, an n-way ANOVA was performed to determine the response of soil nutrient concentrations to interactions between tree treatments, distance from tree bole, and depth in soil profile for individual sampling dates, and additionally between sampling dates in 2015 and 2016. All statistical analyses were performed at $\alpha = 0.05$ unless otherwise indicated in the text.

RESULTS AND DISCUSSION

By fall 2016, eight of the girdled trees were completely dead and had lost all their needles, and the remaining two girdled trees had only one cohort of needles remaining. All five control trees were healthy and had fully foliated canopies.

Mortality Impacts on Canopy Hydrology

Throughout the study, the sapflow data indicated no differences between trees inoculated with bluestain fungi versus those inoculated with agar as controls, leading us to assume that the bluestain inoculations were unsuccessful. As such, the results presented here separate the treatment trees into control (i.e., non-girdled) and girdled trees.

During the first month of data collection, girdled pines displayed significantly higher average daily sapflow than control pines ($p < 0.0001$; fig. 2A). Sapflow rates were not statistically different between control and girdled pines throughout the following fall months. Control pines began to exhibit significantly higher sapflow rates in January of 2016 ($p < 0.001$; fig. 2A), and continued to have at least 25 percent higher average daily sapflow for each month of the study with the exception of June of 2016, where sapflow did not significantly differ between the two treatments ($p > 0.050$). Sapflow of girdled pines decreased at least twofold when comparing the same fall months in 2015 to 2016 (fig. 2A).

Cumulative stemflow volume standardized for basal area ($L\ m^{-2}$) throughout the entire study was not significantly different between treatments (p -value = 0.937), although some differences were observed for individual storms (fig. 2B: November 9, 2015: p -value = 0.063 and December 2, 2015: p -value = 0.010). Control trees generated an average total stemflow of $560\ L\ m^{-2}$ in contrast to girdled trees which generated $415\ L\ m^{-2}$. From June 2016 through December 2016 an extreme drought plagued the study region so no data points for canopy hydrology were obtained in this period.

Mortality Impacts on Canopy Biogeochemistry

Stemflow DOC concentrations were much greater than those observed in rainfall and throughfall ($p = 0.013$), but were not significantly different between treatments (p -value = 0.811) (fig. 3A). In the three storms analyzed for DOC, control trees produced an average total of $818.8\ kg\ ha^{-1}$ compared to girdled trees which produced an average total of $646.9\ kg\ ha^{-1}$ DOC over the three storms analyzed. Stemflow carbon had a higher molecular weight than rainfall or throughfall as indicated by the smaller S_R (fig. 3B) and was higher in aromaticity as indicated by the larger a_{254} (fig. 3C), but when standardized for carbon concentration was not different (fig. 3D). Furthermore, there were no differences in DOM absorbance metrics between treatments (S_R : $p = 0.080$; a_{254} : $p = 0.407$; SUVA $p = 0.607$) although only three storm events have been analyzed thus far (fig. 4). Nitrogen concentrations in stemflow were significantly different by rainfall event (NO_3^- : $p < 0.001$; NH_4^+ : $p < 0.001$; ON: $p < 0.001$) (fig. 4). Cumulatively, the control trees lost more nitrogen in stemflow flux in all forms (NO_3^- : 9.19 vs. $4.42\ kg\ ha^{-1}$; NH_4^+ : 46.34 vs. $29.40\ kg\ ha^{-1}$; ON: 88.53 vs. $56.84\ kg\ ha^{-1}$; TN: 144.06 vs. $90.47\ kg\ ha^{-1}$). The large degree of inter-storm variability in nitrogen flux led to insignificant differences between treatments, although girdled trees were beginning to show larger fluxes at the conclusion of the fall 2016 drought.

Mortality Impacts on Soil Properties

Initially, soils surrounding girdled trees had higher C:N ratios in fall 2015 compared to treatment trees (12.7 vs. 11.7), but in fall 2016 the trend had reversed (12.1 vs. 12.8). In both fall 2015 and 2016, these differences were significant between treatments and between depths, while distance from tree bole was also significant in fall 2016 (table 1). In fall 2015, there were no differences in soil carbon concentrations between treatments, although carbon concentrations decreased with depth (table 1). In fall 2016, carbon concentrations in soils surrounding control trees averaged $0.72\ mg\ C\ mg^{-1}$ soil versus $0.66\ mg\ C\ mg^{-1}$ soil around girdled trees, with

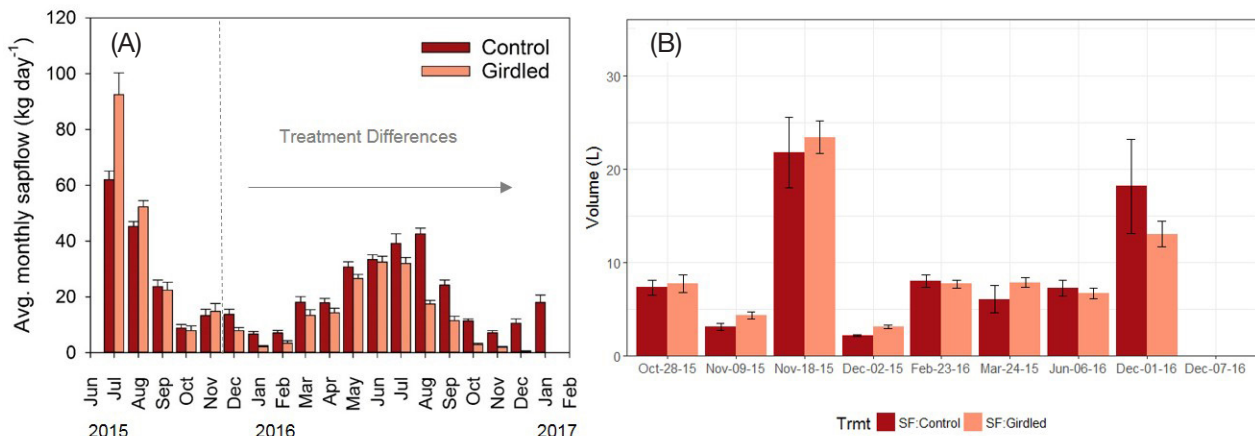


Figure 2—(A) Average monthly sapflow from control and girdled study trees. Differences between treatments were observed beginning in December 2015 and remained significant for the remainder of the study. (B) Stemflow (SF) hydrologic flux from study trees for individual rainfall events. Means + standard errors are given in both graphs.

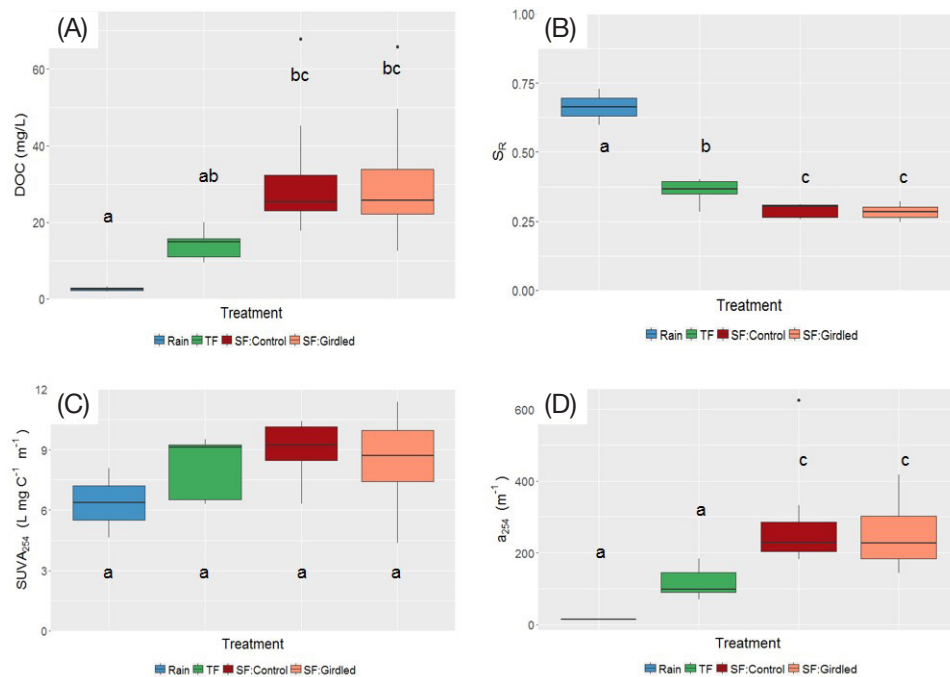


Figure 3—Boxplots of dissolved organic matter characteristics for (A) dissolved organic carbon concentrations; (B) spectral slope ratio; (C) specific UV absorbance; and (D) absorption coefficient for rain, throughfall (TF), and stemflow (SF) from study trees. Significant differences between sample sources are denoted by different letters.

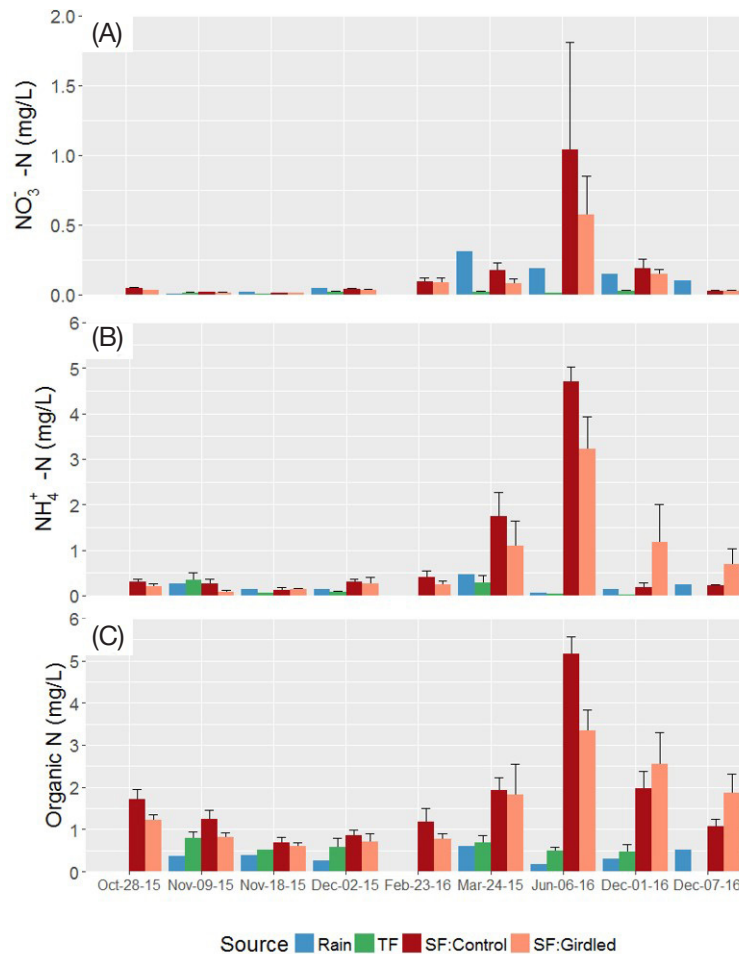


Figure 4—Barplots of dissolved nitrogen characteristics for (A) nitrate-nitrogen; (B) ammonium-nitrogen; and (C) organic nitrogen for rain, throughfall (TF), and stemflow (SF) from study trees during individual rainfall events. Means + standard errors are given.

depth and distances also significant factors (table 1). Comparing the two years, there was a significant interaction effect between treatment and date (table 1), with carbon concentrations remaining stable around girdled trees ($0.65 \text{ mg C mg}^{-1} \text{ soil}$) but increased around control trees (0.59 to $0.72 \text{ mg C mg}^{-1} \text{ soil}$). Similarly, no differences in nitrogen concentrations were observed in fall 2015 between treatments, only between depths, with less nitrogen deeper in the soil profile (table 1). In 2016, distance was also a significant factor along with depth (table 1). Comparing the two measurement years there were still no indications of differences in nitrogen concentrations between treatments (table 1) suggesting that the changes observed in the C:N ratios were driven by changes in soil carbon concentrations and not by nitrogen concentrations. Lastly, while soil respiration decreased from growing season to dormant season in 2016 ($p < 0.00$) and was marginally higher closer to stems than away ($p = 0.091$), there were no significant differences between treatments ($p = 0.154$) (fig. 5).

CONCLUSIONS

In this study, we documented differences in hydrology, canopy-derived biogeochemical fluxes and soil properties following a simulated bark beetle outbreak in a loblolly pine stand. Treatment differences in sapflow and tree water uptake were evident within the first 6 months of study initiation demonstrating the successful girdling efforts to mimic a bark beetle infestation and subsequent mortality. The decreases in stemflow quantity in girdled trees towards the end of the study was surprising and contradictory to our hypothesis, in which the removal of canopy needles would reduce canopy interception and promote stemflow generation (Siegert and Levia 2014). The tree crowns in this loblolly pine plantation were quite small, and removal of intercepting surfaces likely decreased the potential for those crowns to capture rainwater which would then be diverted to stemflow. Additional storms are required to tease out these complex interactions given the variable nature of stemflow generation to storm characteristics

Table 1—Results of n-way factorial analysis of variance on soil biogeochemical properties including carbon concentrations (C), nitrogen concentrations (N), and the ratio of the two (C:N)

	Fall 2015			Fall 2016			Fall 2015 vs. fall 2016		
	C:N	C	N	C:N	C	N	C:N	C	N
Treatment	0.035	0.106	0.763	0.015	0.041	0.205	0.880	0.790	0.768
Distance	0.121	0.095	0.363	0.006	0.004	0.011	0.057	0.035	0.075
Depth	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Date	-	-	-	-	-	-	0.560	0.008	<0.001
Treatment x Distance	0.284	0.421	0.137	0.951	0.991	0.913	0.661	0.546	0.171
Treatment x Depth	0.554	0.439	0.588	0.415	0.331	0.272	0.376	0.230	0.230
Treatment x Date	-	-	-	-	-	-	<0.001	0.007	0.218
Distance x Depth	0.945	0.559	0.285	0.374	0.421	0.578	0.655	0.771	0.919
Distance x Date	-	-	-	-	-	-	0.023	0.020	0.079
Depth x Date	-	-	-	-	-	-	0.326	0.693	0.996
Treatment x Distance x Depth	0.487	0.242	0.241	0.574	0.586	0.841	0.477	0.436	0.615
Treatment x Depth x Date	-	-	-	-	-	-	0.875	0.959	0.704
Treatment x Distance x Date	-	-	-	-	-	-	0.253	0.654	0.602
Depth x Distance x Date	-	-	-	-	-	-	0.516	0.181	0.158
Treatment x Depth x Distance x Date	-	-	-	-	-	-	0.590	0.556	0.728

Treatment: control vs. girdled.

Distance from tree bole: 0.5, 1.0, and 1.5 m.

Depth in soil horizon: A horizon: 0-5 cm; B horizon: 5-10 cm.

Date of sample: fall 2015 and fall 2016.

Bolded values are statistically significant at $\alpha = 0.05$.

(Keim and others 2006, Staelens and others 2008, Van Stan and others 2011). Although mortality was clearly underway in girdled trees, no differences were observed in DOC, DOM quality, or N concentrations in stemflow as hypothesized (Bade and others 2015, Frost and Levia 2014). The inopportune drought that occurred when girdled trees were finally succumbing to mortality represents a significant missed window into canopy biogeochemical processes. Without rainfall during this period, it was impossible to capture these changes, but recovery post-drought will likely provide further insights. In the soils, C:N ratios were greater under control trees than under girdled trees, which was driven by an increase in carbon around control trees and not a decrease around girdled trees. Here, it is likely that

mortality of competitors (i.e., girdled trees) opened up additional growing space and resources which control trees were able to capture and increase belowground biomass growth and subsequent fine root turnover, contributing to increased soil carbon (Rasse and others 2005). While this process was not supported by the soil respiration data, it is possible that heterotrophic respiration from decomposition of belowground biomass on girdled trees was balanced by increased autotrophic respiration from increased belowground biomass production on control trees. In summary, the hydrologic and biogeochemical response of trees undergoing mortality from bark beetle infestations is complex and in real-world circumstances can be complicated further by climatic variability such as drought. In 18 months,

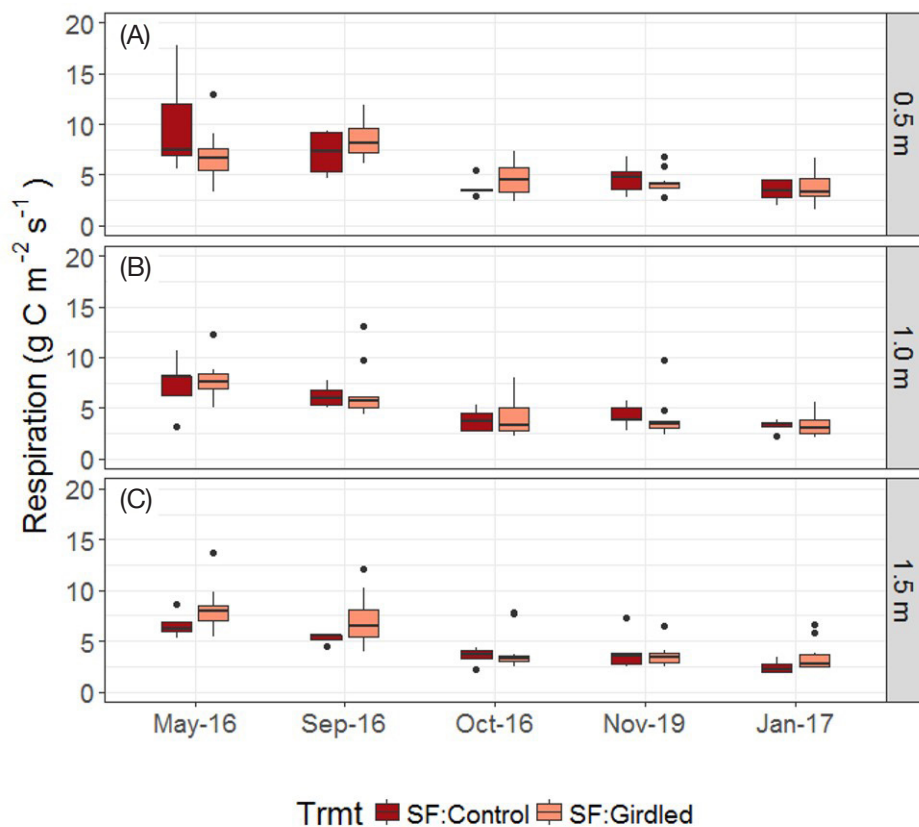


Figure 5—Boxplots of respiration from soils at (A) 0.5 m-, (B) 1.0 m-, and (C) 1.5-m distance from study trees.

we documented changes in canopy-derived hydrologic and biogeochemical fluxes, tree water use, and soil biogeochemical processes although no clear trends were yet apparent. As tree mortality and decomposition of above- and belowground biomass continue, these processes will likely become clearer.

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RESTORING PONDEROSA PINE IN THE DAVIS MOUNTAINS OF WEST TEXAS: IMPACTS OF PLANTING PRACTICES ON SEEDLING SURVIVAL

Lance A. Vickers, James Houser, James Rooni, and James M. Guldin

Abstract—The ponderosa pine forests (*Pinus ponderosa* Laws.) in the Davis Mountains of west Texas recently experienced a major mortality event that resulted, in part, from profound regional drought predisposing trees and stands to mortality from both western pine beetle (*Dendroctonus brevicornis*) and wildfires. To evaluate alternatives for restoration and recovery, the Texas A&M Forest Service (TFS) initiated “Operation Ponderosa” in cooperation with The Nature Conservancy (TNC), on the TNC Davis Mountains Preserve in Jeff Davis County, TX. The loss of overstory pines and lack of natural regeneration pose a considerable challenge to management. A pilot study was commissioned to investigate artificial regeneration of ponderosa pine using containerized seedlings and site preparation alternatives. Early survival was poor, mainly due to below-ground herbivory, which was identified as the principal short-term obstacle to artificial regeneration in the Davis Mountains. The larger question of ponderosa pine recovery, particularly if local climatic conditions become increasingly unfavorable, remains.

INTRODUCTION

Ponderosa pine (*Pinus ponderosa* Laws.) is one of the most important conifers in the United States. With native populations in every state that lies west of the 100°W meridian (except Kansas), it is one of the most widely distributed pines on the continent (Oliver and Ryker 1990). The largest ponderosa pine population in Texas occurs in the Davis Mountains, much of it owned or protected by The Nature Conservancy’s (TNC) Davis Mountains Preserve.

Across much of the range of ponderosa pine, and particularly in the Southwest, “megadisturbances” or major mortality events have weakened or decimated many stands (Millar and Stephenson 2015, Reynolds and others 2013). These events are usually the confluence of economic, environmental, ecological, and policy influences that both individually and collectively have acted as stressors to forest health and vigor over the past century. In the Davis Mountains, for example, forests underwent a densification process that started around the turn of the 20th century (Bataineh 2006, Poulos and others 2013). This process increased piñon-juniper density from approximately 100 trees per acre in 1890 to over 1,100 trees per acre in 2005 (Bataineh 2006). Livestock grazing was likely the only mechanism reducing density through much of the 20th century. A concomitant change in the historic fire regime also

occurred. Fire-return intervals averaged about 5 years before 1937, but fire-free periods increased to 20–40 years thereafter (Poulos and others 2013). Because of the difficult terrain and the sparse distribution of the species across the area, local timber markets are absent, and no sawmill is reported within 300 miles (Prestemon and others 2005).

In the 1990s, these dense forests experienced an extended period of elevated temperatures and profound drought coupled with secondary bark beetle attacks. In 2011–2012, wildfires burned through several stands across the Davis Mountains. This major mortality event (fig. 1) resulted in a reduction of ponderosa pine from an estimate of over 800 trees per acre on parts of the TNC Davis Mountains Preserve in 2004 (Bataineh 2004) to an estimated 17 trees per acre across the Preserve in 2014 [Texas A&M Forest Service (TFS), unpublished data], with bark beetle attacks ongoing. There was a near complete lack of ponderosa pine seedlings observed and a paucity of cones on surviving mature trees 4 years after the wildfires (TFS, unpublished data).

To evaluate alternatives for restoration and recovery of the TNC Davis Mountains Preserve, TFS initiated “Operation Ponderosa,” in cooperation with TNC and other partners. One of the primary goals of Operation Ponderosa was to foster ponderosa pine regeneration,

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Figure 1 — Tree mortality has been catastrophic and widespread across the Davis Mountains Preserve as a result of drought, insect attack, and wildfire. Though all species have been affected by this major mortality event, it has been most notable for ponderosa pine, and many fully-stocked stands have been decimated. (photo by Lance A. Vickers)

by both natural and artificial means. Because science-based silvicultural guidelines are not available for ponderosa pine in the Davis Mountains, several questions regarding the best practices for artificial regeneration exist. These questions include identification of appropriate planting season and early cultural treatments that promote survival and growth. To begin addressing these questions, a planting demonstration study on artificial regeneration of ponderosa pine in the Davis Mountains was commissioned. This demonstration used locally sourced containerized seedlings raised by TFS and a set of low-cost, easily applied site preparation alternatives feasible for the remote, rugged terrain.

METHODS

Site Description

The Davis Mountains of west Texas ($\approx -104.1, 30.7$) are approximately 35 million years old, predominantly igneous in origin, and range in elevation from about 5,000–8,000 feet. Baldy Peak atop Mt. Livermore is the highest point at 8,382 feet in elevation. The Köppen climate type is cold semi-arid, with average minimum temperatures (at 6,790 feet elevation) ranging from 32 to 59 °F (January to July) and average maximum temperatures ranging from 54 °F in January to 85 °F in June. Annual precipitation includes about 19 inches of rainfall and 5 inches of snowfall on average with the majority occurring in a distinct monsoon pattern from June–September.

Vegetation communities in the Davis Mountains are transitional and range from Chihuahuan grasslands to sky island relict forests with increasing elevation (Hinckley 1944). At midelevations piñon woodlands and

oak-piñon-juniper woodlands dominate, whereas at higher elevations ponderosa pine is more prevalent and mixed conifer-hardwood assemblages dominate (see Bataineh and others 2007, Poulos and Camp 2010, and Poulos and others 2007 for more detail).

Planting Demonstration Description

Three stands formerly dominated by ponderosa pine within the TNC Davis Mountains Preserve were used for the Operation Ponderosa planting demonstration. These stands all experienced some degree of ponderosa pine mortality during the wildfires in 2011–2012, and subsequently received a thinning treatment in 2015 that aimed to reduce the density of surviving ponderosa pine competitors not strongly affected by the wildfires (primarily *Juniperus* and *Quercus* spp.). The residual basal area in these three stands averaged 15.3 square feet per acre with ponderosa pine as the chief contributor, making up about 65 percent of the basal area. Other common species were gray oak (*Q. grisea* Lieb.), Emory oak (*Q. emoryi* Leib.), alligator juniper (*J. deppeana* Steud.), and piñon pine (*P. cembroides* Zucc.). Several other comparatively rare oaks were present along with black cherry (*Prunus serotina* var. *virens* McVaugh) and Texas madrone (*Arbutus xalapensis* Kunth).

Soils in these stands were predominantly composed of the Loghouse association (Loamy-skeletal, mixed, superactive, mesic Typic Haplustalfs); soils were uniform within each stand (<http://websoilsurvey.nrcs.usda.gov/>). The Loghouse association typically is a deep and well-drained loam with low available water storage in the profile.

The Operation Ponderosa planting demonstration was a replicated comparison of three weed control treatments and two planting seasons (dormant and monsoon). The dormant season planting (November 2015) included five weed control comparison replicates in each of the three stands in the study. Insufficient stand access and poor nursery survival precluded a full installation (comparable to fall 2015) of the monsoon season planting treatment (August 2016) in all three stands used for the dormant season planting. As a result, only four full replicates were established in two of the stands for the monsoon season planting. Given the within-stand soil uniformity, there was no need for within-stand statistical blocking, and the planting group locations were randomly located within each stand.

For the dormant season planting, 450 containerized (1-0, D40) seedlings developed from local seed sources were used. In the monsoon season planting, the 257 containerized (2-0, D60) seedlings used were of the same cohort as the dormant season seedlings but stepped up from D40 to D60 containers and held longer in the nursery (approximately 1 additional year). The D40 dormant season seedlings had an average root collar diameter of 0.16 inch and stem height of 5.43 inches with an approximate average root:shoot ratio of 2:1. The D40 seedlings were planted to a “first-green” depth, i.e., to the base of live foliage. The larger D60 monsoon season seedlings had an average root collar diameter of 0.23 inch and stem height of 7.12 inches, with an approximate average root:shoot ratio of 2:1, although

there was considerable variation in height, diameter, and overall condition of these older seedlings. Due to rocky soil at depth, the D60 seedlings were planted only to container depth, i.e., no more of the stem was buried at planting than was in the containers.

Planting Details

Dormant season planting occurred on November 17–18, 2015. The local weather was clear and cool (27–63 °F) with relative humidity ranging from 34–41 percent and 14 to 19-mile-per-hour westerly winds reported nearby. There was slight rainfall (approx. 0.1 inch) reported in the area before planting and 2.95 inches reported during the week following planting. Monsoon season planting occurred on August 30–31, 2016. The local weather was overcast and mild (36–70 °F) with high relative humidity (87 percent) and 3 to 4-mile-per-hour northeasterly winds reported. There was approximately 1 inch of rainfall reported in the area during the 2 weeks before planting, about 1 inch during planting, and another 1 inch over 2 weeks following planting.

Localized group structures within uneven-aged stands are commonly observed in frequent-fire forests of the Southwest (Reynolds and others 2013). Field observations in the Davis Mountains Preserve found a consistent pattern of several saplings and small pole-sized trees in groups of approximately four to ten trees behaving more or less as a congeneric group with spacing varying from about 5–20 feet (fig. 2). A triangular



Figure 2—Large saplings and small pole-sized ponderosa pine commonly occurred in a grouped pattern on the Preserve. Field observations found a consistent pattern of groups of approximately four to ten trees behaving more or less as a congeneric group with spacing varying from about 5–20 feet. Localized group structures within uneven-aged stands are commonly observed in frequent-fire forests of the Southwest (Reynolds and others 2013). (photo by Lance A. Vickers)

for CWC) when averaged across all stands (fig. 4). There were no statistical differences in survival between CWC and NWC. Evidence suggests that about 15 percent of the seedlings that did not survive, or 9 percent of all planted seedlings, were lost to desiccation after 13 months. The majority of the mortality was due to below-ground herbivory, likely from pocket gophers (*Thomomys* spp.), which, along with some above-ground browsing, occurred in all stands.

The average survival rate after 4 months for the monsoon season planting was 14 percent. Survival varied somewhat by weed control treatment, with PWC having statistically higher survival rates (25 percent) than the other two treatments (9 percent for NWC, 11 percent for CWC) when averaged across all stands (fig. 4). There were no statistical differences in survival between CWC and NWC. Evidence suggests that about 10 percent of the seedlings that did not survive, or 8 percent of all planted seedlings, were lost to desiccation after 4 months. Similar to the dormant planting, the majority of the mortality in the summer planting has been attributed to gopher herbivory which, along with some above-ground browse, occurred in all stands.

DISCUSSION

The survival rates observed in the planting demonstration were poor, and comparable survival rates have been documented elsewhere in the Southwest under similar conditions. Ouzts and others (2015) reported a 25 percent survival rate for planted ponderosa pine seedlings after 5–8 years across several southwestern stands, and rates from 0–12 percent in 38 percent of stands surveyed. The frequency of planted seedlings completely lost to herbivory in the planting demonstration suggests that it is the principal short-term obstacle to successfully restoring ponderosa pine in the Davis Mountains. Pocket gopher herbivory of planted seedlings is common in many parts of the ponderosa pine range (Barnes 1978, Dingle 1956, Hooven 1971). The survival rates in the planting demonstration (dormant season: 22–34 percent, monsoon season: 9–25 percent) are comparable to the 35 percent survival rates reported by Hooven (1971) after 1 year in areas occupied by pocket gophers. Hooven (1971) reported that survival rates had dropped to 12 percent after 5 years in those areas compared to 87 percent in areas without pocket gophers.

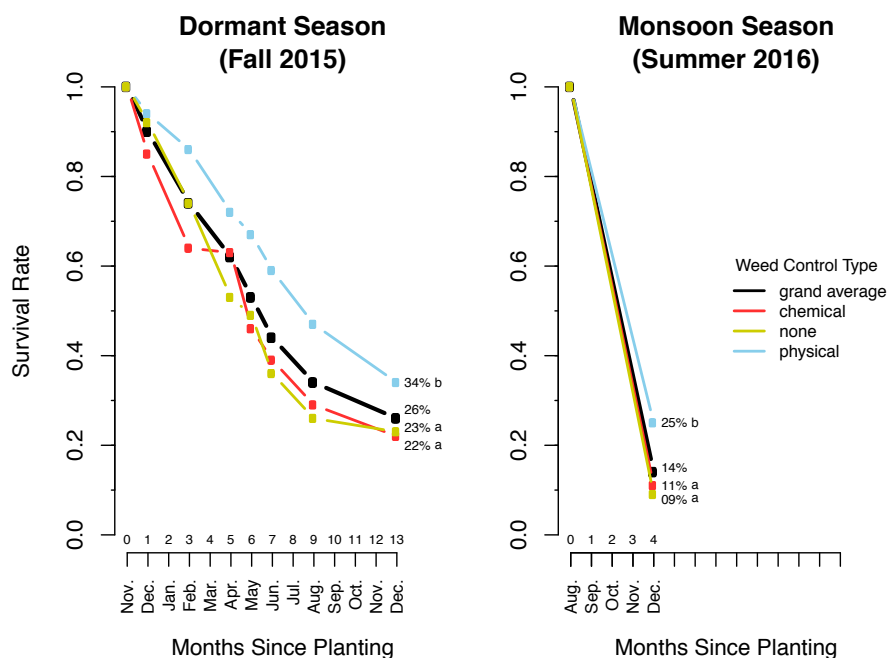


Figure 4—Longitudinal survival curves for the planting demonstration. Treatment means with different letters were statistically different ($\alpha = 0.05$) within a planting season. For the dormant season (left), survival in the Physical Weed Control treatment was statistically greater than both the Chemical and No Weed Control treatments after 13 months. There was no statistical difference in survival between the Chemical and No Weed Control treatments after 13 months in the dormant season planting. In the monsoon season planting (right), survival in the Physical Weed Control treatment was statistically greater than both the Chemical and No Weed Control treatments after 4 months. There was no statistical difference in survival between the Chemical and No Weed Control treatments after 4 months. The vast majority of seedlings lost were attributed to below-ground herbivory, likely from gophers. Note: statistical differences do not necessarily indicate meaningful biological differences.

Successful restoration appears to hinge on identifying treatments to reduce below-ground herbivory, particularly in areas with very loamy soils where herbivory appeared more prevalent. Unfortunately, many common control strategies are seldom effective or feasible without intensive maintenance (Godfrey 1987, Hooven 1971). The higher survival found in the PWC treatment may be attributable to some deterrence offered by the two landscaping staples used to fasten the fibrous mats around the base of seedlings in the PWC treatment. Additional research into the efficacy of this and other below-ground herbivory reduction alternatives in the Davis Mountains Preserve is warranted, and a pilot case study has been planned.

Absent effective deterrence, a less desirable option may be an attempt to compensate for herbivory losses by planting substantially more seedlings than are ultimately desired. Based on the survival rates in the planting demonstration this would mean planting *at least* five times more seedlings than desired. This highlights the second obstacle to restoring ponderosa pine in the Davis Mountains: scarcity of seedlings from both natural and artificial sources. This is in stark contrast to portions of the ponderosa pine range like the Black Hills region where excessive reproduction densities are often a concern (Sheppard and Battaglia 2002). To date, cone and seed collection efforts have been limited to trees and stands within the immediate region of the planting demonstration. Managers may need to consider broadening the seed sources and relying on genetic diversity to allow some individuals to prosper in harsh sites.

The U.S. Forest Service has reported success with summer plantings in Arizona and New Mexico, coinciding with the monsoon season. The amount of desiccation after 13 months in the dormant season seedlings (~9 percent) was observed in only 4 months for the monsoon season seedlings (8 percent). The performance of the monsoon season seedlings 4 months after planting (fig. 4) suggests that monsoon planting is not a reliable option in the Davis Mountains at this time, but that cannot be claimed conclusively from the results as planting season effects were confounded by differences in planting stock (1-0 vs. 2-0), which is somewhat unavoidable when comparing planting seasons. Even in the nursery, the condition and survival of the locally-sourced 2-0 containerized seedlings used for the monsoon season planting were poor. Improved seed sources and nursery practices may yield better results during the monsoon in the Davis Mountains. However, in the immediate future, efforts may be better served by planting 1-0 containerized seedlings abundantly during favorable climatic windows. The

results of the planting demonstration suggest that during El Niño years, dormant season planting could be a viable option if herbivory losses can be avoided. During non-El Niño years, the high winds and limited rainfall of the dormant season may be detrimental. Additional research into the season of planting is recommended.

It is possible that alternative timings, rates, or herbicides could provide more efficacious weed control while avoiding the adverse impacts on ponderosa seedling growth observed in the planting demonstration. Despite the precautionary efforts to minimize damage to the planted dormant season ponderosa seedlings during herbicide application, a few seedlings (10) exhibited some evidence of herbicide damage. Ponderosa pine has been identified as sensitive to several herbicides, including the one used in this application (Oust® XP) despite being labeled for use as herbaceous weed control for ponderosa pine. Continued investigation into the efficacy of alternative chemical weed control treatments with organized trials in the Davis Mountains is suggested.

Though below-ground herbivory has been identified as the principal short-term obstacle to regenerating ponderosa pine in the Davis Mountains, the larger question of ponderosa pine recovery, particularly if local climatic conditions become increasingly unfavorable, remains. Critical components of that question are the identification of target conditions that are both appropriate and achievable and the suite of management options, silvicultural treatments, and timings needed to meet them. The results of the planting demonstration presented here are an important first step in the development of science-based silvicultural guidelines for ponderosa pine restoration efforts in the Davis Mountains.

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CHARACTERIZING TREE MORTALITY AFTER EXTREME DROUGHT AND INSECT OUTBREAKS IN THE SOUTHERN SIERRA NEVADA

Lauren S. Pile, Marc D. Meyer, Ramiro Rojas, and Olivia Roe

Abstract—The interplay of past management practices, higher temperatures, extended drought, and insect outbreaks has resulted in unprecedented levels of tree mortality across the southern Sierra Nevada Mountains of California. To characterize patterns in tree mortality, we collected repeated forest measurements on 255 variable-radius plots during 2015 and 2016. From initial measurements in spring 2015, sugar pine (*Pinus lambertiana*) and ponderosa pine (*P. ponderosa*) mortality increased from 30 percent to 80 percent by summer 2016. Incense cedar (*Calocedrus decurrens*) mortality remained relatively stable in 2015 but increased to 40 percent in summer 2016. There were positive relationships for live crown ratios and survivorship for both sugar pine and white fir (*Abies concolor*). In addition, there was a positive effect of diameter on survivorship for ponderosa pine in spring 2015; however, this relationship was reversed in summer 2016. Our results indicate that the effects of this mortality event were variable among species with initial survivorship positively related to tree size.

INTRODUCTION

In recent years, significant drought-induced tree mortality has occurred throughout the United States (Bendixsen and others 2015, Hember and others 2016, Luce and others 2016). Warmer and drier environments, exemplified by “hotter droughts” (Allen and others 2015), can impact trees through greater metabolic demand, reduced carbon fixation, and increased desiccation and cavitation (Luce and others 2016). Between 2012 and 2017, California experienced a record setting, extreme drought event. In 2015, snow pack was only 5 percent of the historical average due to record high temperatures from January to March (Belmecheri and others 2016). Changes in winter weather conditions such as more precipitation falling as rain (Klos and others 2014, Pierce and others 2008), or earlier melting (Cayan and others 2001, Luce and others 2014, Stewart and others 2005), results in a longer dry season in Mediterranean-type climates subsequently increasing the impacts of reduced annual rainfall (Barnett and others 2005) and increased water deficit (Thorne and others 2015). This recent drought event has resulted in an unprecedented level of tree mortality across the Sierra Nevada Mountains with an estimated loss of 102 million trees in California (USDA Office of Communications 2016). Determining the scope and scale of this mortality event is important for informing future silvicultural implementation and understanding the impact of past management practices.

In the southern Sierra Nevada Mountains of California, fire suppression and timber harvests have caused an increase in the density and extent of small diameter, shade-tolerant species from historic averages (McIntyre and others 2015, Stephens and others 2015). The resulting increase in forest biomass and homogeneity couples with warmer temperatures and extended drought to increase the likelihood of insect outbreaks exceeding natural population thresholds from the scale of trees to entire landscapes (Millar and Stephenson 2015, Raffa and others 2008). In addition, high ambient ozone (O₃) concentrations and elevated nitrogen (N) deposition have also been implicated in a fundamental loss in tree vigor and eventual forest decline (Paoletti and others 2009), and the southern Sierra Nevadas have among the highest exposure indices (Cisneros and others 2010). These combined factors have led to severe canopy moisture stress and epidemic levels of insect outbreaks and subsequent tree mortality (Asner and others 2016).

Bark beetles (Curculionidae: Scolytinae) are major disturbance agents in western forests with a larger spatial impact than forest fires (Hicke and others 2016). Trees under drought stress have reduced defenses to beetle attack and drought can increase the occurrence of different beetle species erupting simultaneously, impacting many tree species (Raffa and others 2005). The magnitude of bark beetle outbreaks has increased

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and expanded in recent years (Raffa and others 2008) compared to our knowledge of their historic frequency, severity, location, and extent (Logan and others 2003, Logan and Powell 2001). Bark beetles can act as an integral agent of natural ecological processes, substantially altering forest structure, composition, and function. However, they can also be a major source of economic loss and a challenge to natural resource policy by altering large-scale biogeophysical processes. Outbreaks may also provide valuable insights into environmental threats arising from anthropogenic change (Kurz and others 2008, Raffa and others 2008).

By 2014, bark beetle-induced mortality was evident on ponderosa (*Pinus ponderosa*) and sugar pine (*P. lambertiana*) at lower elevations (~ 4,000 feet). The goal of this study was to determine the impact of drought and insect mortality on forest stand structure and composition. The objectives of this study were to determine 1) survivorship by species over time and 2) the effect of mortality on tree size. We hypothesized that 1) pine species would have higher levels of mortality than other species due to the impact of drought coupled with

bark beetle outbreaks and 2) larger diameter trees with greater live crown ratios would have higher survivorship due to deeper root systems and larger carbon stores.

MATERIALS AND METHODS

Study Site

The study took place on the High Sierra Ranger District of the Sierra National Forest within the Dinkey Collaborative Forest Landscape Restoration Project (CFLRP) boundary (fig. 1). The Sierra National Forest is located in the southern Sierra Nevada Mountain range of California. The Dinkey CFLRP is located approximately 30 miles east of Fresno, CA. The landscape-level collaborative project was established in 2010, was one of the first projects funded by Title IV of the Omnibus Public Land Management Act of 2009, and lies entirely within the High Sierra Ranger District (Schultz and others 2012). Climate in this area is characterized as montane Mediterranean, with warm, dry summers and cool, wet winters. Most precipitation falls from October to April with 70 percent falling as snow (Bales and others 2011). Soils are in the Gerle-Cagwin families association (NRCS 2009).

Mortality Plot Locations

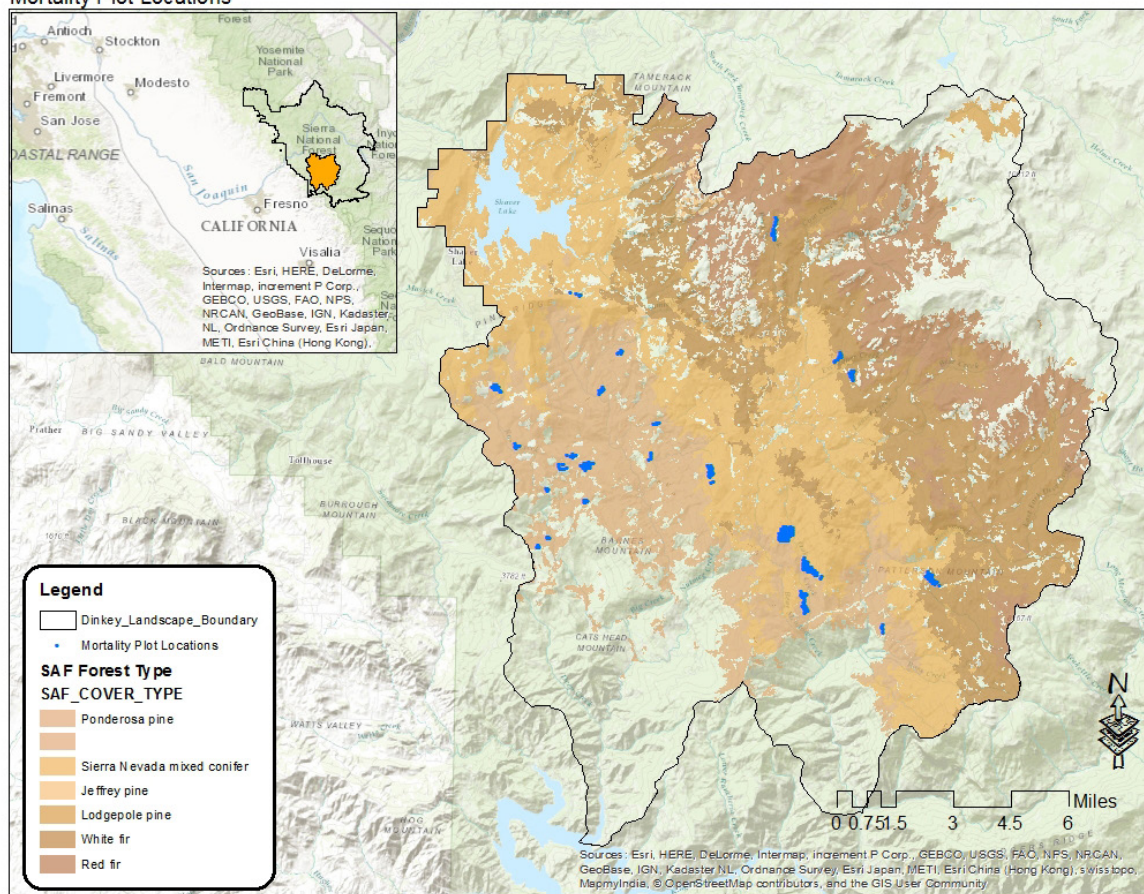


Figure 1—Plot locations on the Sierra National Forest High Sierra Ranger District within the Dinkey Collaborative Forest Landscape Restoration Project (CFLRP) boundary by Society of American Foresters (SAF) cover type.

Plot locations were selected within the Dinkey CFLRP to inform the collaborative planning process for project development and restoration activities as the impact of drought and beetle-related mortality increased across the landscape. To assess our objectives, we established plots outside any recent or active management areas, stratified by forest type, and observed mortality classes based on 2015 aerial detection surveys (ADS). We stratified the ADS into 5 mortality classes (low to very high) based upon dead trees per acre. We limited plot selection to the elevational range of ponderosa pine and mixed conifer zones and stratified by vegetation type, elevational zone, and mortality class. This process identified 25 polygons for measurement with polygons ranging in size from 1 to 10 acres. A randomly placed grid identified plot locations within each of the polygons. This process yielded 255 plots.

Measurements

In the spring of 2015 (SP15), 255 variable-radius plots were established using a 40-factor prism. We permanently tagged trees >5 inches in diameter at breast height (DBH) that were considered “in” the variable-radius plots and recorded each for DBH (inches), height (feet), species, status (dead or alive), and live crown ratio (LCR; percent). Additionally, we established an 11.8-foot offset radius sapling plot to permanently tag and record saplings between 1 and 4.9 inches in DBH. We repeated the measurements in the summer of 2015 (SM15) and the summer of 2016 (SM16).

Data Analyses

Due to relatively small sample sizes, we excluded lodgepole pine (*P. contorta*), Jeffrey pine (*P. jeffreyi*), and canyon live oak (*Quercus chrysolepis*) from the statistical analyses. To assess our first objective, we compared percent survival by species, time (SP15,

SM15, and SM16), and their interaction across the three time periods using a repeated measures analysis of variance (ANOVA). Change in overall density (trees per acre) and density by species across the three time periods was analyzed using a one-way ANOVA. To assess our second objective, the probability of survival in relation to DBH and LCR were analyzed by species using logistic regression with a binary response variable for survivorship (0 dead, 1 alive) for each measurement time using GLIMMIX (SAS® 9.1.3 SAS Institute Inc., Cary, NC). Individual tree was nested within plot as a random factor in the model. Data are reported as means and standard errors of the mean. Where necessary, data were transformed prior to analysis or distributions were transformed within the GLIMMIX model, but values are reported in original scale to aid interpretability. Each *p*-value less than 0.05 was considered evidence of a significant difference.

RESULTS AND DISCUSSION

Extended drought and increasing temperatures have been implicated for large-scale mortality events in the Sierra Nevada previously (Guarín and Taylor 2005). Drought, pollution, and high tree density have been implicated in similar drought- and insect-related mortality events in southern California (Paoletti and others 2009). Increases in the frequency of drought-related mortality are often associated with below average moisture levels that occur over multiple, consecutive years (Guarín and Taylor 2005). In our study, a total of 1,683 trees representing nine species were measured (table 1), and live tree density decreased across the three time periods ($F = 5.17$; $p < 0.01$). Average tree density per acre was 209 ± 15 trees per acre in SP15. In SM15, density remained relatively stable to SP15 at 190 ± 15 trees per acre, however in SM16 the density decreased significantly to 145 ± 15 trees per acre. Ponderosa pine

Table 1—Descriptive statistics of the measurements taken across 255 repeatedly measured plots on the Sierra National Forest by species

Species	Scientific name	n	DBH inches	Height feet
White fir	<i>Abies concolor</i> (Gord. & Glend.) Lindl. ex Hildebr.	526	22.5 ± 12.5	89 ± 42
California red fir	<i>Abies magnifica</i> A. Murray bis	50	26.1 ± 18.7	93 ± 61
Incense cedar	<i>Calocedrus decurrens</i> (Torr.) Florin	419	14.6 ± 12.9	49 ± 36
Lodgepole pine	<i>Pinus contorta</i> Douglas ex Loudon	19	18.2 ± 11.0	77 ± 44
Jeffrey pine	<i>Pinus jeffreyi</i> Balf.	6	35.2 ± 19.7	107 ± 51
Sugar pine	<i>Pinus lambertiana</i> Douglas	117	31.5 ± 16.0	112 ± 47
Ponderosa pine	<i>Pinus ponderosa</i> Lawson & C. Lawson	416	20.7 ± 12.8	87 ± 49
Canyon live oak	<i>Quercus chrysolepis</i> Liebm.	39	5.3 ± 5.8	21 ± 13
California black oak	<i>Quercus kelloggii</i> Newberry	91	19.5 ± 10.7	50 ± 23

n = number sampled.

Diameter at breast height (DBH) and height are averages.

decreased in density only a few months after the initial measurement in SP15, with continued reductions in 2016. White fir density remained relatively stable in SP15 and SM15, however by SM16 the density decreased significantly (table 2). Although there were decreases in the densities of incense cedar (*Calocedrus decurrens*) and sugar pine across the three time periods, these decreases were not significant.

Survivorship

Survivorship was significant by time ($F = 144.6$; $p < 0.01$), species ($F = 48.8$; $p < 0.01$), and their interaction ($F = 30.3$; $p < 0.01$). The interaction was due to the variable individual response of different species across the three time periods (fig. 2). In particular, survivorship decreased for white fir ($F = 194.8$; $p < 0.01$), incense cedar ($F = 40.5$; $p < 0.01$), sugar pine ($F = 60.3$; $p < 0.01$),

and ponderosa pine ($F = 347.9$; $p < 0.01$) over time, but remained stable for California red fir (*A. magnifica*; $F = 0.29$; $p = 0.75$) and California black oak (*Q. kelloggii*; $F = 0.16$; $p = 0.85$). Species-specific responses likely reflect individual tolerances to both abiotic (e.g., drought) and biotic (e.g., insects) factors. The impact of drought predisposes many conifers to insect attack, resulting in higher levels of episodic tree mortality owing to declines in host defenses, increases in host suitability, and increased incidence of insect mass attacks (Mattson and Haack 1987). In comparison, the higher survivorship rates observed in California black oak may reflect its lower susceptibility to insect attack and relatively minor impact of damaging insects, such as the wood-boring carpenterworm (*Prionoxystus robiniae*) or defoliating California oakworm (*Phryganidia californica*) (McDonald 1990), and relatively lower sensitivity to increases in

Table 2—Change in stand density (trees per acre) by species over the three measurement periods

Species	SP15	SM15	SM16	
White fir	103 ± 10 ^a	99 ± 10 ^a	65 ± 10 ^b	$F = 7.3$; $p < 0.01$
California red fir	187 ± 64	187 ± 64	186 ± 64	$F = 0.01$; $p = 0.99$
Incense cedar	138 ± 16	135 ± 16	110 ± 16	$F = 0.92$; $p = 0.40$
Sugar pine	26 ± 6	20 ± 6	18 ± 6	$F = 0.47$; $p = 0.62$
Ponderosa pine	84 ± 10 ^a	61 ± 10 ^{ab}	39 ± 10 ^b	$F = 5.48$; $p < 0.01$
California black oak	58 ± 12	54 ± 12	54 ± 12	$F = 0.04$; $p = 0.96$

SP15 = spring 2015; SM15 = summer 2015; SM16 = summer 2016.

Differences in superscript letters indicate a significant difference between measurement periods within species.

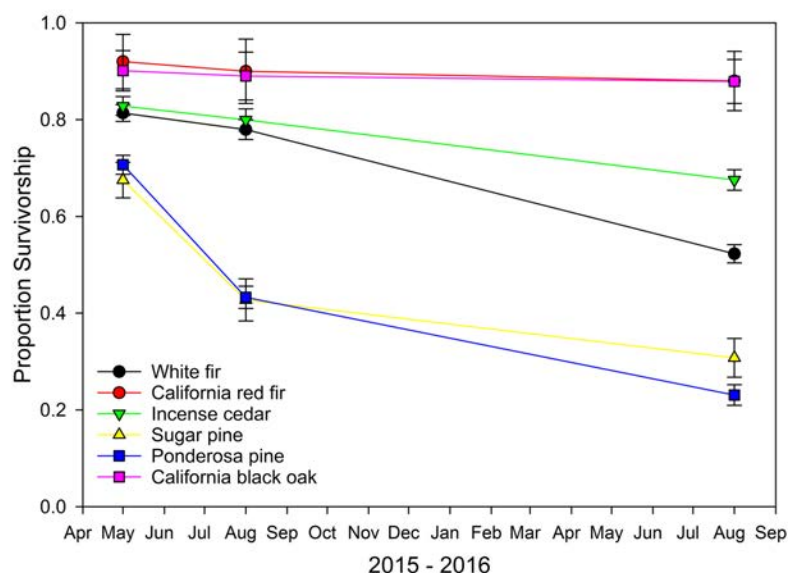


Figure 2—Proportion of survivorship by species during the three measurement periods from spring 2015 to summer 2016 on the Sierra National Forest.

climatic water deficit compared to conifers (McIntyre and others 2015). Previous studies have related drought-induced mortality to insect and pathogen activity coupled with climate change (Logan and others 2003, Millar and Stephenson 2015). Guarín and Taylor (2005) noted that during a drought-related mortality event in California, 62 percent of dead ponderosa pine and 67 percent of incense cedar had beetle galleries or pitch tubes, and 15 percent of dead incense cedar was visibly infected by cedar rust (*Gymnosporangium libocedri*).

The disproportional impact on pine species from the interaction of drought and bark beetles may have severe ecological consequences as it accelerates a transition from fire-tolerant pine species to fire-intolerant and shade-tolerant white fir and incense cedar following historical logging and fire exclusion beginning in the late 1800s. However, white fir and incense cedar were impacted by the mortality event. White fir survivorship decreased from 88 percent in SP15 to 52 percent by SM16. Incense cedar remained relatively stable with 80 percent surviving throughout 2015 and 68 percent surviving in SM16. The impact to pine species was more dramatic. Sugar pine survivorship was reduced from 68 percent in SP15 to 31 percent in SM16. Ponderosa pine

survivorship was reduced from 71 percent in SP15 to 23 percent in SM16. Although the impacts of this mortality event may help to reduce stem densities to levels within the natural range of variation (NRV) prior to fire exclusion, the impact was greatest in large diameter individuals, especially pines (data not reported here), resulting in a significant shift in historic forest stand structure and composition compared to current conditions. This shift will widen the existing deficit of large pines and other conifers in forest landscapes of the southern Sierra Nevada (e.g., Stephens and others 2015).

For several species, the probability of survival could be predicted by either DBH or LCR over the three measurement periods (table 3). Diameter at breast height was significant for determining the probability of survival for incense cedar and California black oak in all three measurement periods (fig. 3; right panel). It was also significant for determining the survival of ponderosa pine in SP15 and SM16, however the relationship of DBH and survivorship was reversed. This reversal indicates the greater survivorship of smaller diameter understory ponderosa pine in some areas compared with larger diameter trees. Live crown ratio was significant for determining the probability of survival for white fir and

Table 3—Species-specific parameter estimates using diameter at breast height (DBH) and live crown ratio (LCR) for logistic binary regression models for each measurement time

Species	Time	DBH estimates			LCR estimates		
		Intercept	Estimate		Intercept	Estimate	
White fir	SP15	1.6068	-0.0059	F = 0.5; <i>p</i> = 0.50	0.0414	0.2621	F = 2.29; <i>p</i> = 0.13
	SM15	1.4291	-0.0073	F = 0.8; <i>p</i> = 0.37	0.1841	0.0962	F = 19.0; <i>p</i> < 0.01
	SM16	-0.0398	0.0059	F = 0.7; <i>p</i> = 0.40	-0.7955	0.0343	F = 31.8; <i>p</i> < 0.01
California red fir	SP15	1.7691	0.0310	F = 0.8; <i>p</i> = 0.37		-	
	SM15	1.9643	0.0097	F = 0.1; <i>p</i> = 0.72	2.3970	0.0288	F = 0.3; <i>p</i> = 0.59
	SM16	1.7793	0.0088	F = 0.1; <i>p</i> = 0.72	2.9975	0.0019	F = 0.0; <i>p</i> = 0.97
Incense cedar	SP15	0.4574	0.1108	F = 32.2; <i>p</i> < 0.01	3.9686	0.0386	F = 0.7; <i>p</i> = 0.40
	SM15	0.4400	0.0855	F = 29.5; <i>p</i> < 0.01	2.7083	0.0132	F = 0.8; <i>p</i> = 0.37
	SM16	-0.0185	0.0589	F = 29.3; <i>p</i> < 0.01	0.8242	0.0190	F = 6.3; <i>p</i> = 0.01
Sugar pine	SP15	0.5337	0.0064	F = 0.3; <i>p</i> = 0.61	-1.7661	0.0532	F = 10.0; <i>p</i> < 0.01
	SM15	-0.6355	0.0108	F = 0.8; <i>p</i> = 0.36		Did not converge	
	SM16	-0.8653	0.0017	F = 0.0; <i>p</i> = 0.89	-2.1345	0.0426	F = 7.9; <i>p</i> < 0.01
Ponderosa pine	SP15	0.5156	0.0182	F = 4.5; <i>p</i> = 0.04	2.9985	0.0569	F = 1.6; <i>p</i> = 0.20
	SM15	-0.2382	-0.0016	F = 0.0; <i>p</i> = 0.83	0.2730	0.0048	F = 0.5; <i>p</i> = 0.49
	SM16	-0.7850	-0.0213	F = 5.1; <i>p</i> = 0.02	-0.5114	-0.0064	F = 0.78; <i>p</i> = 0.38
California black oak	SP15	0.5326	0.1088	F = 4.8; <i>p</i> = 0.03		Did not converge	
	SM15	0.3335	0.1145	F = 5.5; <i>p</i> = 0.02		Did not converge	
	SM16	0.5288	0.0917	F = 4.6; <i>p</i> = 0.04		Did not converge	

Significant (alpha of 0.05) GLIMMIX model effects indicated in **bold**.

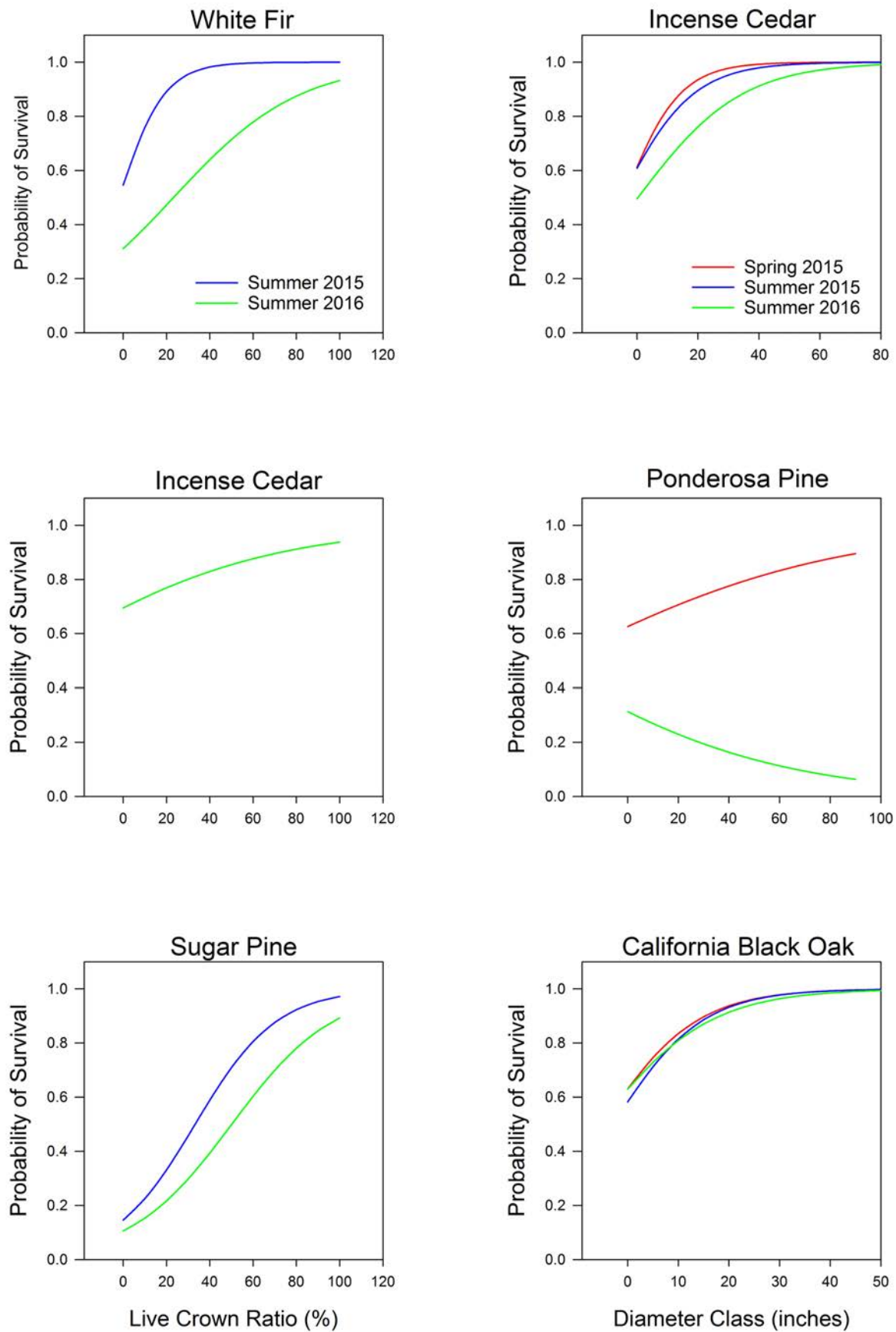


Figure 3—The probability of survival by live crown ratio (LCR in percent; left panel) and diameter at breast height (DBH in inches; right panel) by species and measurement time (red line = spring 2015; blue line = summer 2015; green line = summer 2016).

sugar pine in SM15 and SM16 (fig. 3; left panel). Live crown ratio was also significant for determining the probability of survival for incense cedar in SM16.

Management Implications

Lessons for managers are that treatments that increase tree diameter and LCR may provide for increased survivorship during periods of drought. However, during periods of extended or extreme drought and epidemic insect outbreaks, other landscape-level ecological factors can negate individual tree characteristics. Proactive silvicultural manipulations, such as thinning, may reduce stand-level insect eruptions because they can increase the defensive capacity of a tree by ameliorating the effects of drought, but may be ineffective at reducing landscape-scale eruptions (Fettig and others 2007). Once landscape level thresholds have been breached, no known feasible management action will stop an eruption until hosts are depleted or unseasonably cold temperatures occur over large areas (Raffa and others 2008).

The dramatic loss of mature pine species will require restoration efforts to reforest and promote the ecological conditions necessary for pine to be productive in this landscape. Providing increased growing space for individual trees by increasing stem diameter and maintaining large crowns may help to reduce tree stress. Promoting these characteristics to increase tree vigor has been found to be important for reducing the effects of drought- and insect-related mortality in Mediterranean-type ecosystems (Kolb and others 2016, O'Brien and others 2017). Incorporating a mixed-species approach that includes both pine and other species such as California black oak that have proved more resistant to extreme drought may help increase forest resilience. This will include planting and efforts to control competing vegetation, including shrubs and abundant incense cedar and white fir regeneration, which may include the use of mastication, herbicide, and re-establishing frequent, low intensity fire regime where appropriate. Landscapes that are vertically and structurally heterogeneous including a mixture of species and size classes are likely to result in greater forest resilience. However, as our study shows, individual tree traits such as large DBH and LCR show limited improved survivorship during landscape-level disturbances.

CONCLUSIONS

Our results indicate that trees with larger diameters and LCRs had the greatest initial probability of survival. However, as bark beetle populations reached unprecedented levels, large diameter ponderosa pine were negatively impacted, and mortality was temporally variable among species. In comparison, patterns of high survivorship and low mortality in California black

oak suggest that this species and others (e.g., canyon live oak) may be favored over conifers at low to mid elevations of the Sierra Nevada with increasing climatic water deficit associated with warming climate (McIntyre and others 2015). New silvicultural regimes may be needed in this era of megadisturbances to support the health and vitality of future forests (Millar and Stephenson 2015).

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Loblolly Pine Fertilization

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Southern Research Station

GROWTH OF YOUNG PINE STANDS WITH FERTILIZATION ALONE VERSUS FERTILIZATION PLUS VEGETATION CONTROL

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Abstract—Fertilization and vegetation control are two useful silvicultural tools available to forest managers. Fertilization directly adds nutrients, and vegetation control indirectly improves nutrient and water availability to the crop species. In young pine stands that have not fully captured the site, there has been concern that adding fertilizer will stimulate competing vegetation, which will then outcompete the pines and result in slower pine growth. To determine if this is a valid concern, we tested the hypothesis that there is no difference in pine growth in young stands when applying fertilizer alone or with vegetation control. We installed two treatments at 11 sites with two replications at each site in pine stands across the Southeastern United States. Treatments were fertilization (120 and 12 pounds per acre of elemental nitrogen and phosphorus, respectively, applied every 2 years) and fertilization plus complete sustained vegetation control. Other elements were added if foliar nutrient analysis indicated that they were limiting. Stand age at installation ranged from 2 to 6 years old. Eight years after treatment initiation, no differences in growth were found between the fertilized and fertilized plus vegetation control treatments for diameter, height, basal area, volume, or mortality. Fertilization likely stimulated foliage development, which increased growth and shaded out competing vegetation. However, our results may be influenced by the relatively high rates of nitrogen and phosphorus and additional nutrients added if necessary. In previous studies where one-time applications of 200 and 25 pounds per acre elemental nitrogen and phosphorus, respectively, were added, growth improved when adding vegetation control along with the fertilizer.

INTRODUCTION

Loblolly pine (*Pinus taeda* L.) stands in the Southeastern United States typically have low leaf area levels as a result of nutrient (primarily nitrogen) limitations (Vose and others 1994). These limitations have been ameliorated with mid-rotation applications of 200 and 25 pounds per acre elemental nitrogen and phosphorus, respectively, which result in responses averaging 55 cubic feet per acre per year for 8 years (Fox and others 2007). When one-time vegetation control was applied at the same time as fertilization in mid-rotation loblolly pine stands, treatment responses were additive, indicating that limiting resources other than nitrogen and phosphorus were being ameliorated by the vegetation control treatment (Albaugh and others 2012). However, of the examined sites, a positive response was more common when adding nutrients than when conducting vegetation control. It is likely that in these nutrient-limited mid-rotation stands, the addition of nutrients increased leaf area index such that the additional leaf area shaded out the competing vegetation.

The success in adding nutrients to mid-rotation stands led to an interest in fertilizing younger stands to capture the site more quickly, increase stand productivity, and potentially reduce rotation length. Vegetation control has been included from time of planting with good results. For example, in a regional study, Miller and others (2003) found height gains ranging from 7 to 10 feet at 7 years of age in response to complete sustained herbaceous and woody vegetation control for a range of sites with differing amounts and types of competing vegetation. At age 4 at a site in north-central Florida, fertilization and vegetation control produced similar levels of response when applied alone in loblolly pine stands beginning at planting and additive responses when applied together (Colbert and others 1990). However, in a regional study with similar treatments, there was considerable variation in treatment response, where some sites showed only vegetation control effects at age 5 and others showed only fertilizer effects through age 7 (Will and others 2002). These studies indicated that fertilization could increase growth in young stands, but there was still concern that, at an operational scale, the application of

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fertilizers to young stands which have not fully captured the site would result in competing vegetation taking up these additional nutrients and suppressing rather than improving pine growth.

With this background, our interest was to determine if fertilizer could be applied to young pine stands without stimulating the competing vegetation such that pine growth was reduced. Consequently, we tested the hypothesis that growth as a result of fertilizer and fertilizer plus vegetation control treatments in young pine stands would be the same.

METHODS

We identified 11 sites across the Southeastern United States (table 1) with good operational pine stands that had been site prepared with a mix of chemical or mechanical site preparation. Ten stands were planted in loblolly pine and one (site 11) was in slash pine (*P. elliotii* Engelm.). Stands ranged in age from 2 to 6 years, with Cooperative Research in Forest Fertilization (Jokela and Long 2012) soils groups A, B, C, and E represented. Plots were installed with a measurement plot centered in a larger treatment plot. Measurement plots averaged 0.11 acres and treatment plots averaged 0.45 acres across sites. Between adjacent plots, a minimum buffer of 30 feet was installed. Plots were blocked based on height, basal area, and density to reduce heterogeneity among the plots within and between blocks. There were two blocks per site. Treatments were fertilization (120 and 12 pounds per acre of elemental nitrogen and phosphorus, respectively, applied every 2 years) (F) and

fertilization (120 and 12 pounds per acre of elemental nitrogen and phosphorus, respectively, applied every 2 years) plus complete sustained vegetation control (F+VC). Diammonium phosphate, monoammonium phosphate, urea, and coated urea fertilizer sources were used. To prevent nutrient imbalances, additional nutrients were applied to individual sites based on annual foliar nutrient analyses. After 8 years, total applied nutrients were 480 and 48 pounds per acre of elemental nitrogen and phosphorus, respectively. Vegetation control methods were determined for each site and included mechanical (mowing, weed eaters) and chemical control (spot and directed sprays of glyphosate). Diameter at breast height (4.5 feet) and height were measured prior to treatment and annually for 8 years. Individual tree volume was calculated using an over-bark volume equation for unthinned trees (Tasissa and others 1997) as:

$$V = [0.21949 + (0.00238 * D * D * H)] \quad (1)$$

where

V = individual tree volume in cubic feet per tree

D = diameter at breast height in inches

H = height in feet

The volume increment in year 8 was the volume at the end of year 8 minus the volume prior to treatment.

A mixed model [PROC MIXED (SAS Institute 2002)] was used to test for treatment effects prior to adding

Table 1—Age at initiation, Cooperative Research in Forest Fertilization (CRIFF) soils group, and location for the 11 sites where fertilizer and fertilizer plus complete sustained vegetation control treatments were applied in young pine stands in the Southeastern United States

Site	Age at initiation (years)	CRIFF soils group	Latitude (decimal degrees)	Longitude	County	State
1	6	E	32.91	-86.38	Coosa	AL
2	3	E	34.15	-85.38	Floyd	GA
3	3	E	31.13	-94.46	Angelina	TX
4	2	E	33.81	-82.96	Wilkes	GA
5	4	E	32.70	-88.58	Kemper	MS
6	3	A	32.37	-87.84	Marengo	AL
7	5	E	32.68	-84.74	Talbot	GA
8	2	B	30.48	-93.78	Newton	TX
9	4	E	35.28	-79.94	Montgomery	NC
10	6	B	32.55	-89.64	Montgomery	MS
11	3	C	29.65	-83.17	Dixie	FL

the treatments and 8 years after treatments began. We examined height, diameter, and stand volume prior to adding the treatments and height, diameter, and volume increment (volume at 8 years minus volume at year zero divided by 8) 8 years after treatment began. Fixed effects were treatment (F or F+VC), location (11 sites), and treatment by location. Block was a random effect.

Annual volume growth was calculated for each treatment plot for each year as the volume at the end of the year minus volume at the beginning of the period. We used a linear mixed model to develop a regression equation for the volume increment model:

$$VI_{FVC} = VI_F + S + VI_F \times S \quad (2)$$

where

VI_{FVC} = annual volume increment for the F+VC treatment (cubic feet per acre per year)

VI_F = annual volume increment for the F treatment (cubic feet per acre per year)

S = site

$VI_F \times S$ = the interaction of the annual volume increment for the F treatment and site

Non-significant terms were dropped from the model until all terms were significant. We tested the slope and intercept estimates to see if they were different from 1 and 0, respectively, and estimated the 95 percent confidence intervals around these parameters. All statistical tests were evaluated with alpha equal to 0.05.

RESULTS

No treatment differences were observed for initial diameter, height, and stand volume. Similarly, no treatment differences were found for 8-year diameter,

height, or stand volume increment (table 2 and fig. 1). The only significant term in the volume increment model was VI_F . The intercept for the volume increment model was not different from zero but the slope (0.945) was different from 1. However, the 1:1 line fell within the 95 percent confidence intervals around the volume increment model (fig. 2).

DISCUSSION

We accepted our hypothesis that growth as a result of fertilizer and fertilizer plus vegetation control treatments in young loblolly pine stands was the same. There were no treatment differences in initial stand metrics; consequently, the stands were equal at the start of the study. Eight years after treatment, there were still no differences in stand metrics between the two treatments. We did find a slope <1 for the F+VC and F relationship. In this case, a slope <1 indicates that the F plots had a higher annual volume increment than the F+VC. However, the 1:1 line fell within the 95 percent confidence interval around the regression line suggesting that there is little real difference between the two treatments. Site and site-by-volume increment were not significant factors in the regression model. The loblolly and slash pine sites behaved similarly for the relationship between annual volume increment for the F and F+VC treatments.

Our results were somewhat surprising given the response observed in previous studies where fertilizer and fertilizer plus vegetation control in young stands resulted in an additive treatment response in at least some cases (Colbert and others 1990, Will and others 2002). Fertilization increases leaf area index on nutrient-limited sites, and crown development and expansion can be rapid and extensive (Albaugh and others 2006). Consequently, fertilization alone may eventually stimulate pine leaf area sufficient to shade out competing vegetation. The fertilizer application rate used in our

Table 2—Statistical summary (*p*-values) for growth metrics prior to treatment initiation (initial) and 8 years after treatment initiation for 11 sites in the Southeastern United States where fertilizer and fertilizer plus complete sustained vegetation control treatments were added to young *Pinus taeda* or *P. elliottii* stands

Variable	Treatment	Effect	
		Study	Study x treatment
Initial diameter (inches)	0.310	<0.001	0.541
Initial height (feet)	0.256	<0.001	0.793
Initial stand volume (cubic feet per acre)	0.615	<0.001	0.999
8-year diameter (inches)	0.203	<0.001	0.390
8-year height (feet)	0.121	<0.001	0.714
8-year volume increment (cubic feet per acre per year)	0.386	<0.001	0.990

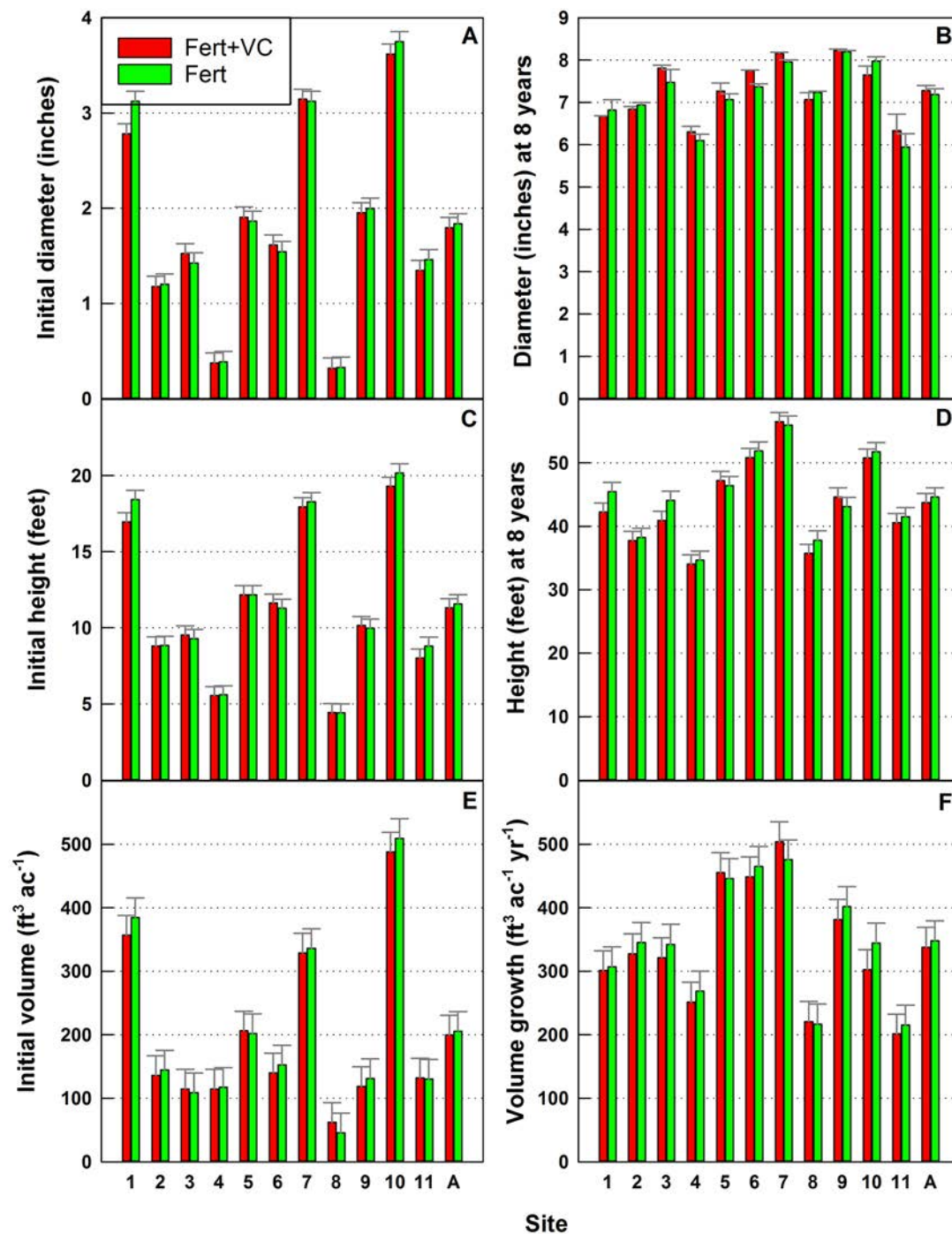


Figure 1—Initial diameter (A), height (C), and stand volume (E) and diameter (B), height (D), and average annual volume increment (F) 8 years after treatment started for the fertilizer only (F) and fertilizer plus vegetation control (F+VC) treatments at 11 sites in the Southeastern United States where fertilizer and fertilizer plus complete sustained vegetation control treatments were applied in young pine stands. Error bars are one standard error; 'A' indicates the average across all 11 sites.

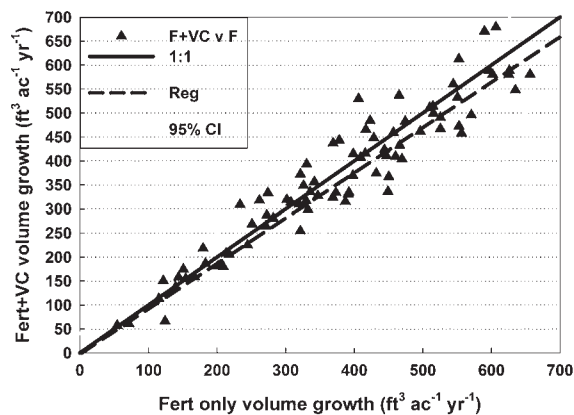


Figure 2—Annual volume growth (cubic feet per acre per year) for 8 years after treatments started for the fertilizer plus vegetation control treatment (F+VC) versus the corresponding fertilizer only treatment (F) at the same site for the 11 sites in the Southeastern United States where fertilizer and fertilizer plus complete sustained vegetation control treatments were applied in young pine stands. Lines shown are the 1:1 line (1:1), the regression (Reg), and plus and minus 95 percent confidence intervals around the regression (95 percent CI).

study (120 and 12 pounds per acre of elemental nitrogen and phosphorus, respectively, applied every 2 years) is a relatively large amount of nutrients to add. After 8 years in our study, we had applied more than twice the current operational application recommendation of 200 and 20 pounds per acre of elemental nitrogen and phosphorus, respectively, applied every 8 years. The relatively high rate of nutrient application may have influenced how rapidly crown expansion occurred and consequently how rapidly the competing vegetation was shaded out in both treatments. The sites selected for our study were considered good operational stands in that they would have had good mechanical or chemical site preparation. This factor may have played a role in our results, in that it is possible there was relatively little competing vegetation when the study started and thus there would be little additional response from the vegetation control. Unfortunately, resources were not available to quantify the competing vegetation at any time during the study, so it is impossible to evaluate this potential effect. Similarly, we did not monitor the effectiveness of the vegetation control; it is possible that vegetation control was not very effective which would have resulted in our observed outcome with no difference between the treatments. Alternatively, although care was taken to avoid damage to the pine from the vegetation control treatments, it is possible that the vegetation control treatments had both a beneficial (elimination of competitors, more available resources) and a detrimental effect (from chemical drift or over application) on the pine where the net result was no change.

From our data alone, one would conclude that on many sites, nutrients may be added to young stands without stimulating the competing vegetation such that there is a loss of pine growth. It is likely that there are sites or conditions where competing vegetation control would be required to get a good response when applying fertilization in young stands. However, if the stands have been well managed from the start and an intensive fertilizer regime is applied, there is a good likelihood of a nutrient response without stimulating the competing vegetation. In support of this argument, there is recent evidence that, in mid-rotation stands with relatively low amounts of competing vegetation, almost all applied nitrogen can be found in the trees, soil, or litter layer with very little found in competing vegetation (Raymond 2016), further supporting the results from this study.

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SOIL RETENTION AND SUBSEQUENT UPTAKE OF NITROGEN 2 YEARS FOLLOWING OPERATIONAL RATES OF FERTILIZATION IN LOBLOLLY PINE

Marshall A. Laviner, Thomas R. Fox, and Jay E. Raymond

Abstract—In this experiment, ^{15}N -labelled fertilizers were used to trace nitrogen uptake, assimilation, and fate in four loblolly pine (*Pinus taeda*) stands fertilized with 224 kg ha^{-1} of elemental nitrogen (N) in the Southeastern United States. The fertilizer treatments included a non-fertilized control, urea, and three different enhanced efficiency N fertilizers. Two years after fertilization, soil from the upper 15 cm in each of five treatments from four different locations was collected. A greenhouse study was established to examine the availability of the residual fertilizer N in the soil. Two bare-root loblolly pine seedlings were planted in each of five replicate pots for a total of 200 seedlings in 100 pots. Total soil N, soil N availability, and seedling N uptake was measured periodically over 15 weeks, and the ^{15}N to ^{14}N ratio was analyzed on soil and seedlings prior to planting and at harvest. Significant ($\alpha = 0.05$) increase in ^{15}N to ^{14}N ratio was found in seedlings 3 weeks after planting in the fertilized treatments. We determined that significant ($\alpha = 0.05$) amounts of residual fertilizer N following operational fertilization was available to loblolly pine seedlings 2 years after application.

INTRODUCTION

Loblolly pine (*Pinus taeda*) is the most widely used plantation species in the Southeastern United States with over 16 million ha in plantations in 2010 (Wear and Greis 2012). Fertilization with nitrogen (N) and phosphorous (P) has been repeatedly shown to increase stem wood volume growth by increasing leaf area which captures more solar radiation and increases net photosynthetic production (Albaugh and others 1998, Carlson and others 2014, Vose and Allen 1988). Between 1998 and 2004, more than 400 000 ha of loblolly pine stands received mid-rotation fertilization with N and P annually (Albaugh and others 2007). With this level of investment in fertilization, it is important to understand how long applied N is available for plant uptake in these forested systems.

A research project was initiated in 2012 to investigate the fate of applied N in loblolly plantations across the Southeast. This study utilized urea enriched with the stable isotope ^{15}N as a tracer (Raymond and others 2016). Urea and three different enhanced efficiency N fertilizers (EEFs) designed to reduce volatilization were tested and compared to an untreated control. After 1 year, Raymond and others (2016) found that 29–39 percent of the applied fertilizer N remained in the soil.

The objectives of this study were to: 1) determine if applied fertilizer N is plant-available in the soil 2 years

after fertilization; and 2) determine if the application of enhanced efficiency fertilizers increases long term plant-available N in the soil more than urea.

METHODS

Field Study Experimental Design

Soils used in this study were collected from a subset of the locations established as part of a large regional fertilization study (Raymond and others 2016). The original study was established as a random complete-block design with five fertilizer treatments at 12 sites supporting mid-rotation loblolly pine plantations across the Southeastern United States. The five fertilizer treatments used in this study were: (1) urea; (2) urea impregnated with N-(n-Butyl) thiophosphoric triamide (NBPT); (3) urea impregnated with NBPT and coated with monoammonium phosphate (CUF); (4) polymer-coated urea (PCU); and (5) a control treatment with no fertilizer added. Urea (46-0-0) was used because it is the most common N fertilizer applied in the Southeastern United States. The enhanced efficiency fertilizers (EEFs) tested in this study were developed to reduce NH_3 volatilization and release fertilizer N slowly to the environment. The NBPT treatment (46-0-0) added NBPT at a rate of 26.7 percent by weight to urea granules to inhibit urease activity. The CUF treatment (39-9-0) also added NBPT to urea granules, which was then coated with an aqueous binder solution of boron and copper sulfate to slow N release. A final coating of CUF was

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added to provide phosphorus (P). The PCU (44-0-0) treatment encapsulated urea granules with a polymer coating containing pores designed to slowly release N (~80 percent) over 120 days. All N treatments were applied at an equivalent rate of 224 kg N ha⁻¹. Because the CUF treatment had P in a coating, P was applied in the other fertilizer treatments at the equivalent rate of 28 kg P ha⁻¹ as triple superphosphate (TSP). The urea in all treatments was enriched with the stable isotope ¹⁵N (0.5 atom percent). Each fertilizer treatment was broadcast applied by hand in individual 100-m² circular plots at each site on the same day between March 26 and April 8, 2012. Details of this study can be found in Raymond and others (2016).

Greenhouse Experimental Design

The four study locations included in this greenhouse experiment were co-located with the PINEMAP Throughfall Exclusion by Fertilization Experiment (Pinemap.org). The stands were located in the Georgia Piedmont (33°37'35" N, 82°47'54" W), Florida Coastal Plain (30°12'22" N, 83°52'12" W), Oklahoma Upper Coastal Plain (34°01'47" N, 94°49'23" W), and Virginia Piedmont (37°27'37" N 78°39'50" W) (fig. 1). The stands ranged in age from 4 to 9 years, and stand density ranged from 789 to 1610 trees ha⁻¹. Will and others (2015) provides detailed site, stand, and climatic descriptions.

In April of 2014, 2 years after initial N fertilization applications, soil from the upper 15 cm of the mineral soil (excluding the forest floor) was collected from each of the five fertilizer treatments. This soil was sieved using a 12-mm screen to remove rocks and roots and to homogenize the sample. Five replicate 4-L plastic pots were filled with soil from each site by treatment combination for a total of 100 pots. Two 1-0 bare-root loblolly pine seedlings were planted in each pot on May 24, 2014. One pot from each site by treatment combination was randomly assigned to each of five benches (blocks) in the greenhouse. Pots were watered every 2 days until drainage was visible from the bottom of the pots. After 21 days, the smaller of the two seedlings in each pot was cut at the ground line. The new growth on the terminal was separated for ¹⁵N analysis. The remaining seedlings were grown in the greenhouse for an additional 12 weeks.

Measurements

Seedling height and ground-line diameter were taken at 0, 21, 69, and 112 days. Soil N concentration and ¹⁵N were assessed at 0 and 112 days. Tissue N concentration and ¹⁵N assessments were also measured at 0, 21, and 112 days.

Tissue samples from the seedlings were dried in a forced air oven at 60 °C. After drying, all samples were coarse



Figure 1—PINEMAP Throughfall Exclusion by Fertilization Experiment study locations where soil was collected. The outlined area on the map represents the natural range for loblolly pine (Pinemap.org).

ground in a Wiley Mill to pass a 2-mm sieve. The organic samples were then homogenized to a fine powder with a ball mill (Retsch® Mixer Mill MM 200, Haan, Germany) for 1 minute at 25 revolutions per second (rps). Mineral soil samples were ball milled for 2 minutes at 25 rps. After ball milling, individual homogenized samples were put in separate tin capsules and weighed on a Mettler-Toledo® MX5 microbalance (Mettler-Toledo, Inc., Columbus, OH, USA). These individually weighed samples were analyzed to determine the $^{15}\text{N}/^{14}\text{N}$ isotope ratio and total N on a coupled elemental analysis-isotope ratio mass spectrometer (IsoPrime 100 EA-IRMS, Isoprime® Ltd., Manchester, UK) All grinding, ball milling, and weighing equipment was cleaned after each sample with ethanol to reduce contamination.

An N availability index using cation and anion exchange membranes was estimated over the period from day 21 to day 112. Cation exchange membranes and anion exchange membranes (GE Power and Water, Trevose, PA, USA) were used to measure ammonium and nitrate concentrations, respectively, and to provide an index of cumulative N availability in the soil solution over the course of their deployment (Cheesman and others 2010). One anion and one cation exchange membrane (10 cm * 5cm) was inserted vertically in each pot after one seedling had been harvested. Exchange membranes were left for 6 weeks and then removed, extracted, and returned to the pots for the final 6-week period. Laboratory procedure followed those described by Cooperband and Logan (1994). Extracts were analyzed for ammonium and nitrate N concentrations using a TRAACS 2000 analytical console (Bran & Luebbe, Norderstedt, Germany).

RESULTS AND DISCUSSION

Initial total soil N was highly variable and ranged from 0.036 percent in the CUF treatment in Georgia to 0.143 percent in the control in Oklahoma (fig. 2). At all sites except Florida, the control treatment had higher total soil N than the fertilized treatments (fig. 2). If we assume that a 15-cm furrow slice weighs 2000 Mg ha⁻¹, then 720 to 2860 kg ha⁻¹ of N is present in these soils. As stated earlier from Raymond and others (2016), 30 percent of the applied fertilizer was present in the soil after 1 year or approximately 60 kg ha⁻¹ of fertilizer N. Therefore similar to the findings of Miller (1981), natural sources of total soil N exceed fertilizer N by a factor of 12 in the lowest case and 47 in the highest case.

N availability index based on the ion exchange membranes was also highly variable (fig. 3), and there were no significant differences ($\alpha = 0.05$) among treatments or sites. The overall site means follow the same trend as total initial total soil N with Florida having the lowest means and Oklahoma the highest. These results support conclusions by many prior researchers (Brady and Weil 2013, Johnson and Todd 1988, Miller and Gardiner 1997) that conventional total soil N and N availability indexes are not very useful for testing or predicting long-term fertilizer response.

The elevated ^{15}N to ^{14}N ratios (fig. 4) of the fertilized soil compared to the controls for each site at day 0 indicate that fertilizer N was still present in the soil 2 years after application. Analysis of ^{15}N to ^{14}N ratios in the new growth on the terminal of seedlings after 21 days indicated significantly elevated ^{15}N above control levels in all of the fertilized treatments (fig. 5). The Georgia and Oklahoma sites had higher ^{15}N in the ESN and NBPT treatments over the urea and CUF. The proportion of N from fertilizer was calculated using the method of Nadelhoffer and Fry (1994). The residual fertilizer N taken up by the seedlings in the 21-day new growth was small compared to the overall N uptake. The percent of the total N taken up by the seedlings derived from the residual ^{15}N -labeled fertilizer averaged 1.1 percent and ranged from 0.25 percent for the urea treatment from Florida to 3.2 percent for the PCU and NBPT treatments from Oklahoma. These results demonstrate that some of the ^{15}N fertilizer applied 2 years prior to the study is still present in a form that is available for loblolly pine uptake. However, when the initial proportion of fertilizer N in the soil is compared to the mean proportion of fertilizer N in the new growth for each site (fig. 6), it is evident that proportionally less fertilizer N is being taken up than is present in the soil. This suggests that the fertilizer N is immobilized in some fashion either by heterotrophs, in other organic compounds, or bound as NH_4^+ to clay or humus (Brady and Weil 2013). Further investigation is needed to determine the means of fertilizer N immobilization and whether more of the fertilizer N can be made plant-available over time.

CONCLUSIONS

Conventional measures of total and “available” soil N are not very useful for evaluating long-term fertilizer response. Fertilizer N is still present in the soil 2 years after fertilization. Some of the fertilizer N present in the soil is plant-available.

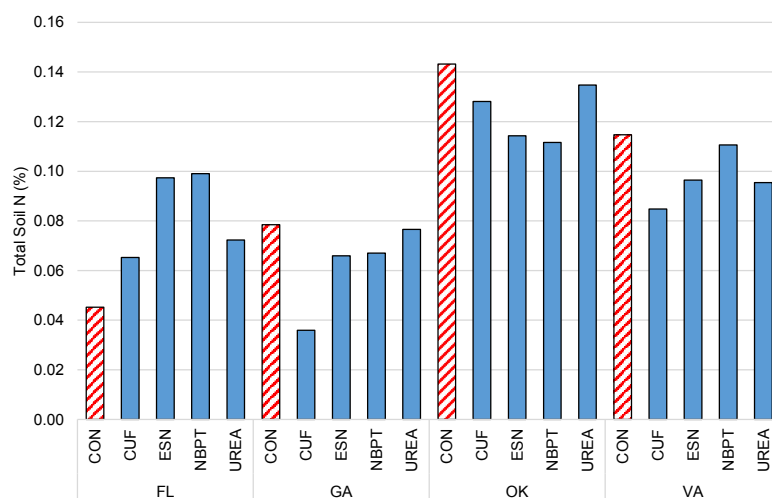


Figure 2—Total soil nitrogen percent at day 0 for each site by fertilizer treatment combination 2 years after fertilizer application to the soil. The hatched bars highlight the control treatment at each site.

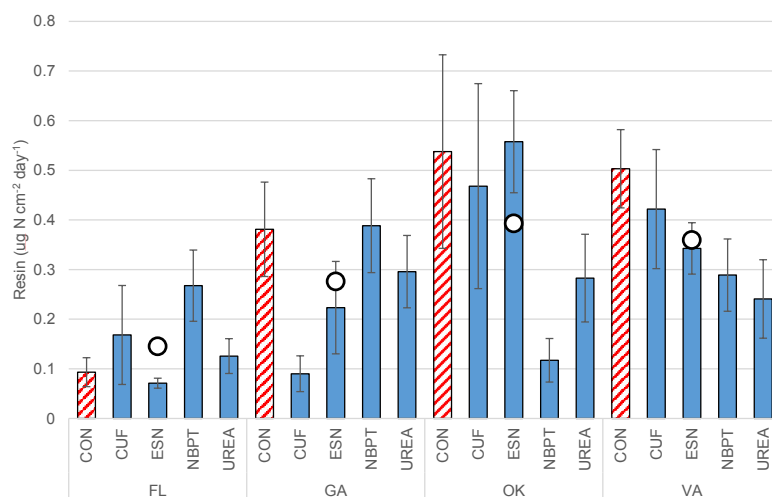


Figure 3—Mean soil nitrogen availability index determined with ion exchange membranes from day 21 to day 112 in ug N cm⁻² day⁻¹. The hatched bars highlight the control treatment at each site. Error bars indicate the standard error (n = 5), and the circles indicate the site mean.

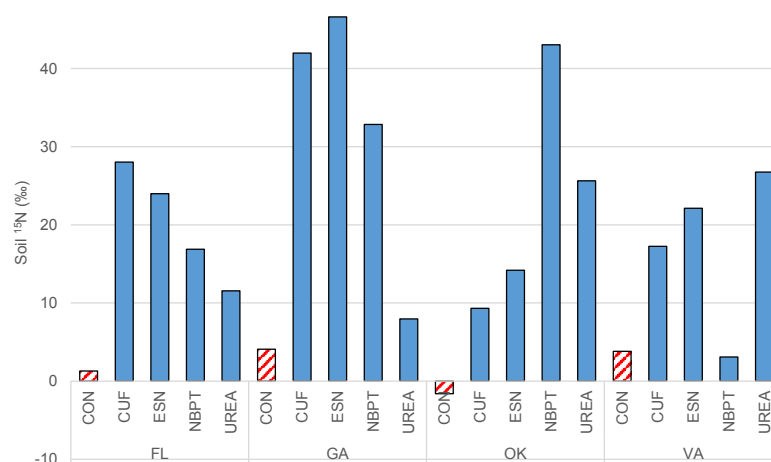


Figure 4—Soil ^{15}N to ^{14}N ratios at day 0 of the greenhouse experiment for each site by fertilizer treatment combination 2 years after fertilizer application to the soil. The hatched bars highlight the control treatment at each site.

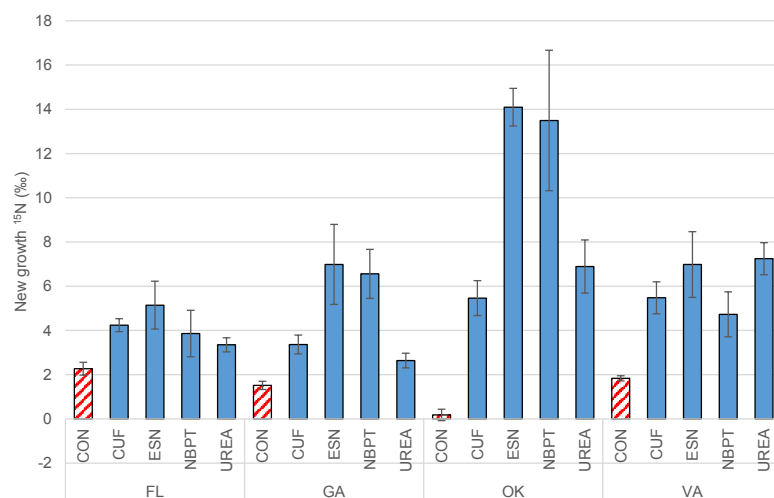


Figure 5—Mean ^{15}N to ^{14}N ratios for tissue from the growing terminal over the first 21 days. The hatched bars highlight the control treatment at each site. Error bars indicate the standard error ($n = 5$).

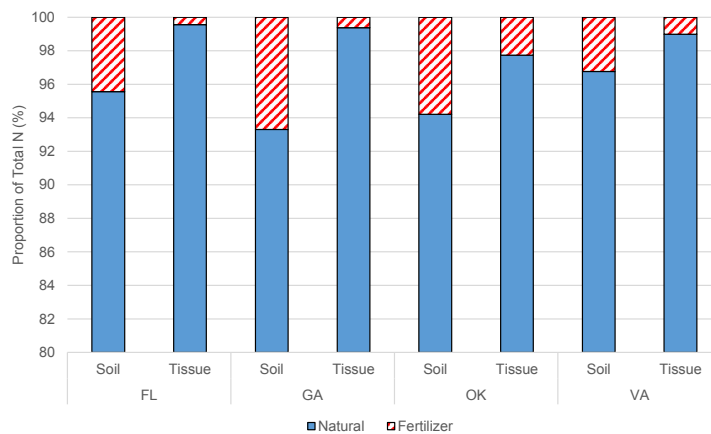


Figure 6—A side-by-side comparison of the proportion of fertilizer N to total N in the soil and new plant tissue for each site.

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EFFECTS OF ANNUAL FERTILIZATION AND COMPETITION CONTROL TREATMENTS ON LOBLOLLY PINE GROWTH THROUGH AGE 25

Stephen M. Kinane and Cristian R. Montes

Abstract—Loblolly pine (*Pinus taeda* L.) plantations represent the largest surface of managed forest in the Southern United States. Over the last 3 decades, forest productivity has been increased as a result of intensive management and improved genetics. Many studies have shown the effects of fertilization and competition control in early ages (1 through 10) on yield of intensively managed loblolly pine stands. However, few studies have looked at the long-term effects of these practices on the inherent site productivity and carrying capacity. The Consortium for Accelerated Pine Plantation Studies (CAPPS), through the Plantation Management Research Cooperative (PMRC), tested if carrying capacity could be increased over the life of the stands in an experimental design with treatments of complete competition control, annual fertilization, and interaction between them. Nine study sites were distributed throughout the Piedmont, Upper Coastal Plain, and Lower Coastal Plain in Georgia, with four replications at each site and tree-level measurements collected each growing season. Results showed higher growth rates on sites receiving annual fertilization and herbicide treatments relative to the control for the first 10 to 15 years. Subsequently, growth rates for this combination treatment diminished at a greater rate than the untreated plot and competition control only. As a result, mortality rates remained fairly constant for all treatments until age 10, after which fertilized treatments showed greater annual mortality. Volume growth rates remained high for treatments without fertilization, showing no evident decline, indicating an increase in site inherent productivity.

INTRODUCTION

Loblolly pine (*Pinus taeda* L.) has been well documented as the most commercially important species of pine in the Southeastern United States. Approximately 45 million acres of established plantations exist throughout the area and an estimated 36 million of those are of loblolly origin (South and Harper 2016). In the Southeastern United States, factors that can lead to limited pine growth include site nutrient deficiencies and interspecific species competition for light and nutrients. Many planted sites use various silvicultural regimes to increase site productivity, including mechanical site preparation, competition control, fertilization, and improved genetics (Allen and others 2005, Gyawali 2015). Four treatment response patterns have been identified to best model a site's response to these different silvicultural treatments. Type A response indicates an increasing volume response with age, attributed to an increase in long-term site resources. Type B volume responses increase with age to a point, reaching an asymptotic level which are maintained through the remainder of the rotation. Type C volume responses show a temporary initial response before declining to no gain. Finally, a Type D volume response shows an initial response before resulting in a negative response (Logan and Shiver 2006). We

hypothesized that type A responses should have an effect on site carrying capacity due to increases in long-term resources.

METHODS

Data Source

The Consortium for Accelerated Pine Production Studies (CAPPS) was established by the Plantation Management Research Cooperative (PMRC) to investigate volume responses to intensive silvicultural regimes. Nine study sites throughout the State of Georgia, established from 1987 to 1995, were analyzed for this study (table 1). A randomized complete block design was implemented for the experimental design. Four 0.15-ha treatment plots were established with interior 0.05-ha measurement plots in each complete block. There was a range of four to six blocks per site with up to three time establishment replicates at each site. Study sites spanned the Piedmont, Upper Coastal Plain, and Lower Coastal Plain. The initial planting density for each site was 1679 trees ha⁻¹ (2.44-m by 2.44-m spacing). Four treatments, control (C), fertilization (F), competition control (H), and combination treatment (HF) were applied at the plot level (table 2). The PMRC field crew collected data annually

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Table 1—Consortium for Accelerated Pine Production Studies study site information

Site	Physiographic region	County	Soil classification	Treatments	Time replicates (number of blocks)
Athens (ATH)	Piedmont	Clarke	Pacolet/Cecil	C,H,F,HF	1989 (2), 1993 (2)
Eatonton-Monitor (BFM)	Piedmont	Putnam	Pacolet/Appling	C,H,F,HF	1988 (2), 1990 (2), 1995 (2)
Eatonton-Powerline (BFP)	Piedmont	Putnam	Cecil	C,H,F,HF	1988(2), 1990 (2), 1995 (2)
Dawsonville-Bottom (DVB)	Piedmont	Dawson	Congaree/Starr	C,H,F,HF	1987 (2), 1989 (2)
Dawsonville-Top (DVT)	Piedmont	Dawson	Hayesville	C,H	1987 (2), 1989 (2)
Thomson (THM)	Upper Coastal Plain	McDuffie	Wagram	C,H	1988 (2), 1990 (2), 1995 (2)
Tifton (TIF)	Upper Coastal Plain	Tift	Pelham/Tifton	C,H,F,HF	1988 (2), 1990 (2)
Waycross-Dry (WCD)	Lower Coastal Plain	Ware	Bonifay/Pacolet	C,H,F,HF	1987(2), 1989 (2),1993 (2)
Waycross-Wet (WCW)	Lower Coastal Plain	Ware	Pelham	C,H,F,HF	1987 (2), 1989 (2),1993 (2)

C = control; H = competition control; F = fertilizer only; HF = combination treatment of competition control and fertilization.

Table 2—Treatment information for Consortium for Accelerated Pine Production Studies study sites

Treatment	Regime
Control (C)	No added treatments other than initial site prep (bedding on Coastal Plain sites, shear, rake, pile, and disc on Piedmont sites)
Fertilization (F)	Spring application of 280 kg ha ⁻¹ DAP, 112 kg ha ⁻¹ KCl; a summer application of 56 kg ha ⁻¹ NH ₄ NO ₃ for the first two growing seasons, followed by early to mid-spring application of 150 kg ha ⁻¹ NH ₄ NO ₃ in growing seasons 3–9. Age 10 treatment was 336 kg ha ⁻¹ of NH ₄ NO ₃ and 140 kg ha ⁻¹ triple super phosphate. Age 11 treatments included 560 kg ha ⁻¹ super rainbow with added micronutrients, and 168 kg ha ⁻¹ of NH ₄ NO ₃ in early spring. Growing seasons 12 and on received 336 kg ha ⁻¹ of NH ₄ NO ₃ in the early spring.
Competition control (H)	Repeated herbicide application to control herbaceous and woody plants
Fertilization + competition control (HF)	Combination of fertilization and competition control treatments

for the first 20 growing seasons. Biennial measurements began in growing season 21. Tree-level measurements include diameter at breast height, total height, and age. Initial planting density allotted 80 trees in the interior measurement plot.

Analysis

Measurement data was summarized at the plot, then treatment level for each site. Volume estimates were made utilizing the PMRC 1996 loblolly yield model for the appropriate physiographic region. Annual growth estimates were calculated from the difference in volume between observation years. Treatment response was calculated using the competition control treatment as a positive control. This method allows for the isolation

of loblolly pine growth from competition and enables the comparison with other treatments' responses. To calculate response, volume for the competition control treatment was subtracted from the treatment of interest's volume.

RESULTS AND DISCUSSION

There was not a consistency in the response type curves for treatments across sites and physiographic regions. Sites in the Piedmont showed higher initial growth rates up to approximately age 10 for plots receiving competition control (fig. 1). Coastal Plain sites exhibited higher initial growth rates for plots receiving fertilization treatments for the first 10 to 15 years. Across all sites, the HF treatment showed higher initial growth

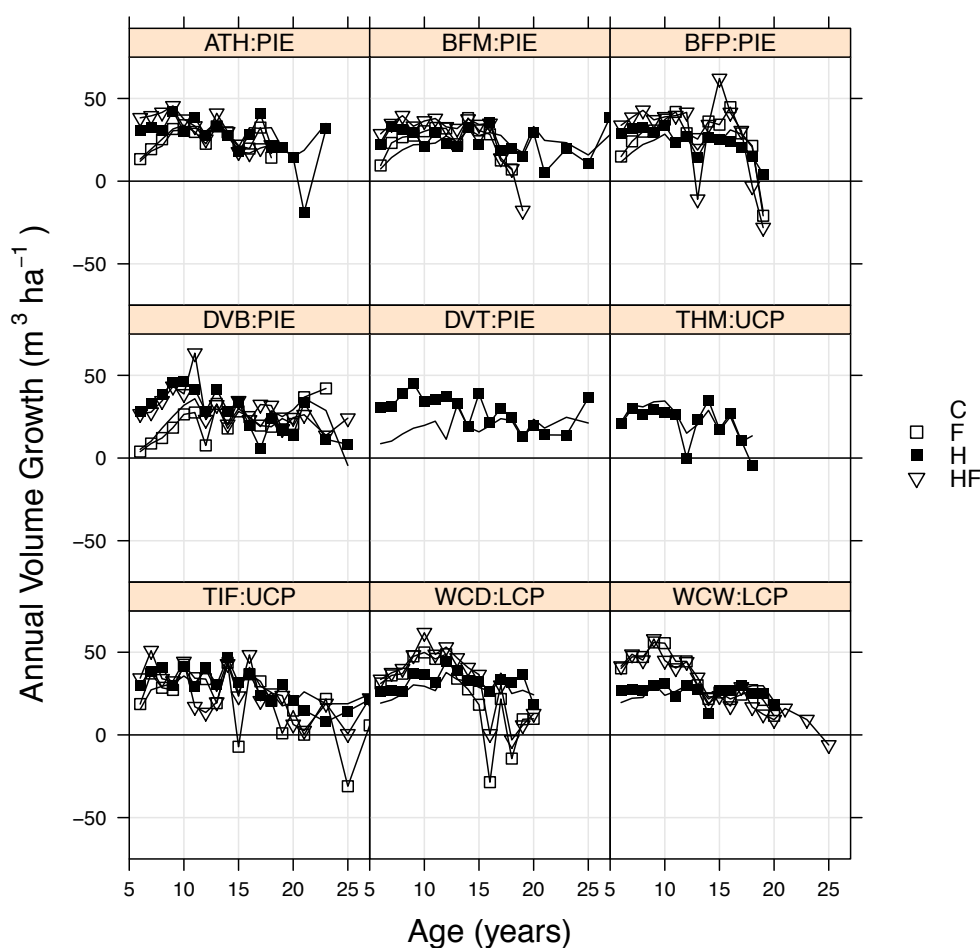


Figure 1—Annual volume growth ($\text{m}^3 \text{ha}^{-1}$) by site and physiographic region of the Consortium for Accelerated Pine Production Studies. ATH: Athens; BFM: Eatonton-Monitor; BFP: Eatonton-Powerline; DVB: Dawsonville-Bottom; DVT: Dawsonville-Top; THM: Thomson; TIF: Tifton; WCD: Waycross-Dry; WCW: Waycross-Wet; PIE: Piedmont; UCP: Upper Coastal Plain; LCP: Lower Coastal Plain. Coastal Plain years after fertilizer application to the soil. The hatched bars highlight the control treatment at each site.

rates. Volume yield indicated a sustained increase in productivity in the long run. Shifts in volume growth after year 15 are the consequence of trees reaching full competition stage with mortality appearing in all plots with higher volumetric growth rates.

Treatment Response

When compared to the competition control treatment, sites receiving the interaction treatment exhibited different response types within and between physiographic regions (fig. 2). Piedmont sites showed initial positive response for treatment plots receiving the interaction treatment followed by a subsequent decline. Lower Coastal Plain sites showed initial positive

responses to treatments containing fertilizer, also followed by a subsequent decline. Sites in Eatonton showed positive responses to the HF treatment. Dawsonville-Bottom showed Type D and no responses in the comparison of C, F, and HF to the H treatments. Negative responses (Type D) were observed in Piedmont sites, including the F treatment in Athens, and C in Athens and Dawsonville (both sites). The Waycross Wet site showed no difference in volume yield response between the competition control (H) and control (C) treatments, but decreases in volume yield response after age 15 can most likely be attributed to tree mortality (fig. 3). This is most likely a result of lack of competition on the site.

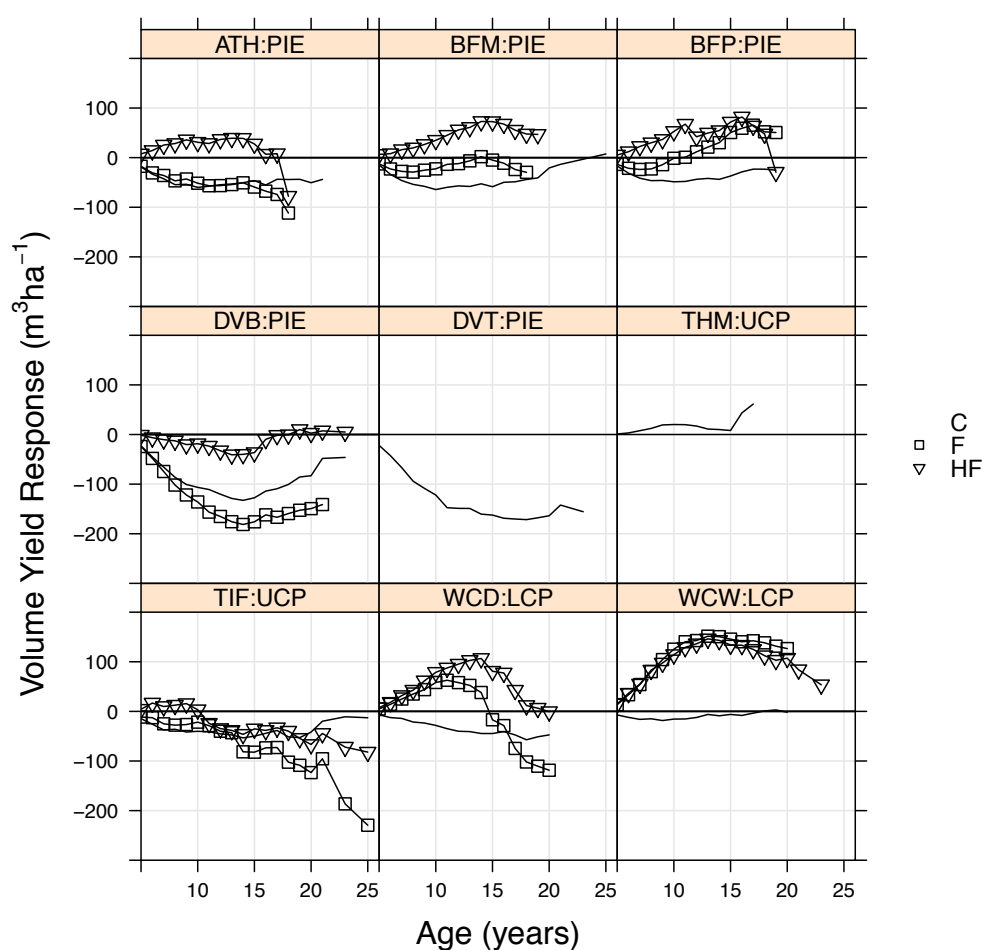


Figure 2—Volume yield response ($\text{m}^3 \text{ha}^{-1}$) for treatments by site as compared to the competition control treatment. ATH: Athens; BFM: Eatonton-Monitor; BFP: Eatonton-Powerline; DVB: Dawsonville-Bottom; DVT: Dawsonville-Top; THM: Thomson; TIF: Tifton; WCD: Waycross-Dry; WCW: Waycross-Wet; PIE: Piedmont; UCP: Upper Coastal Plain; LCP: Lower Coastal Plain.

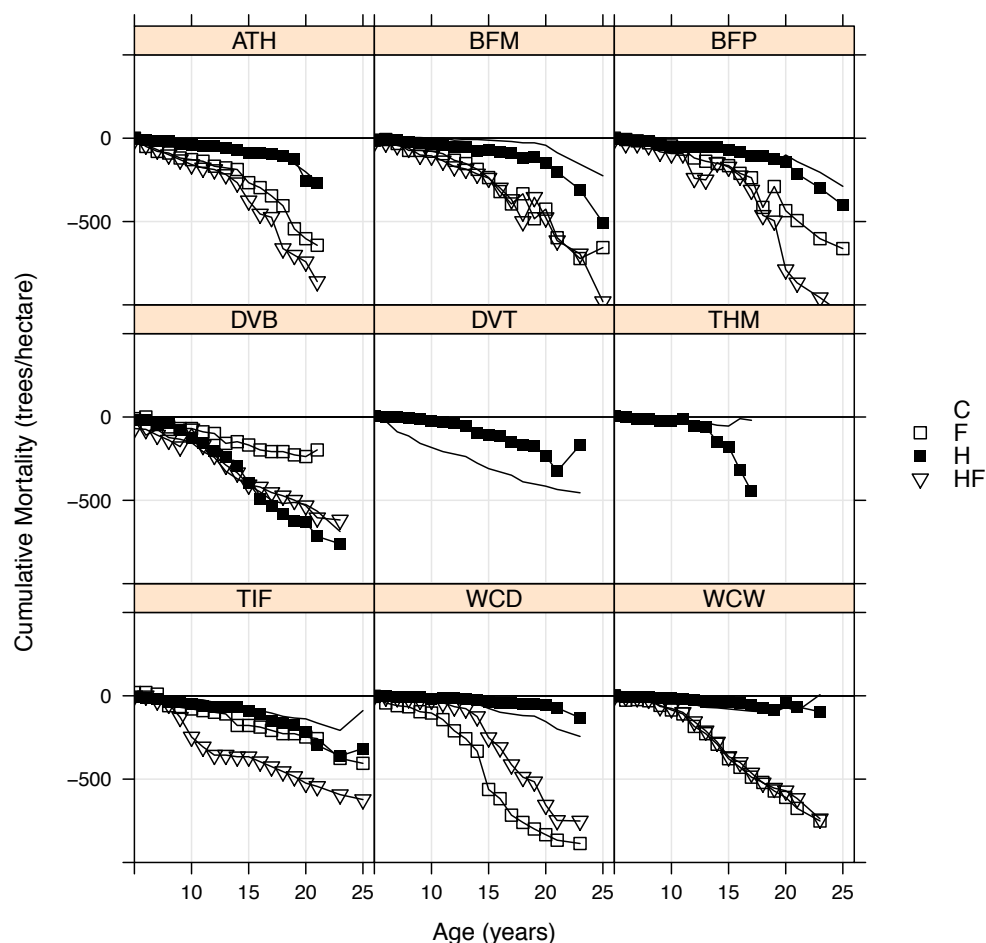


Figure 3—Cumulative mortality (trees ha⁻¹) by site of the Consortium for Accelerated Pine Production Studies. ATH: Athens; BFM: Eatonton-Monitor; BFP: Eatonton-Powerline; DVB: Dawsonville-Bottom; DVT: Dawsonville-Top; THM: Thomson; TIF: Tifton; WCD: Waycross-Dry; WCW: Waycross-Wet.

CONCLUSION

While treatments applied in this study are not operational and intensities will not be seen at scale, results indicate that response trends are not consistent. Knowledge on inherent site properties, such as nutrition levels and competition type and amount, is necessary to select treatments that will result in the best response and not waste resources on ineffective treatments. For sites that showed a relatively slight difference between treatments, such as HF and F on the Waycross-Wet site, we can assume that there was little to no direct competition to be controlled, due to no added response from the competition control component in the HF treatment. With increased growth rates following treatment application, we hypothesize that site carrying capacity must have been increased.

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Afforestation

Moderator:

Don Bragg

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Southern Research Station

VEGETATION CONTROL OPTIONS FOR IMPROVING AFFORESTATION OF A RETIRED SOD FARM IN CENTRAL ARKANSAS

Michael A. Blazier, Hal O. Liechty, and L. Michelle Moore

Abstract—To improve watershed quality, there was interest in converting a retired zoysiagrass (*Zoysia* spp.) sod farm in central Arkansas into hardwood forest. Six vegetation control options for glyphosate and sulfometuron methyl were tested for pre-planting vegetation control to foster water oak (*Quercus nigra* L.) and green ash (*Fraxinus pennsylvanica* Marshall) establishment while protecting water quality at the site. Including sulfometuron with glyphosate in a fall pre-planting application was the most effective for providing vegetation control through the first growing season after planting. A ryegrass (*Lolium* spp.) cover crop suppressed grass and broadleaf competing vegetation in the first growing season after planting, although not to the magnitude and duration as sulfometuron. Horseweed (*Erigeron canadensis* L.) was observed in plots receiving sulfometuron, which prompted an additional trial at this site on herbicides for horseweed control. Clopyralid was most effective among herbicides tested for horseweed control without damaging the water oak and cherrybark oak (*Q. pagoda* Raf.) to which it was applied.

INTRODUCTION

Conversion of retired agricultural land to hardwood forests provides several ecosystem services, but one of the greatest impediments to successful establishment of hardwood seedlings is competition from grasses, non-crop broadleaf plants, vines, shrubs, and trees (Self and Ezell 2015, Stringer and others 2009, Von Althen 1991). Competing vegetation can reduce survival and growth of seedlings by reducing light, moisture, and nutrient availability for the crop species (Self and Ezell 2015, Stringer and others 2009). Suppressing competing vegetation is crucial for improving hardwood survival and growth, and herbicides applied prior to and/or soon after planting are effective at reducing non-crop vegetation (Self and Ezell 2015, Stringer and others 2009). For some site conditions, hardwood-compatible cover crops can be used for suppressing competing vegetation (Stringer and others 2009). Cover crops provide competition suppression by limiting the quantity of weed seeds that reach soil and germinate and by shading out potential weeds (Rentz 2005, Stringer and others 2009). Cover crops used for competition control for hardwood establishment are generally small grains or grasses that do not substantially compete with planted hardwoods for site resources (Stringer and others 2009).

Recent efforts to sustain and improve a watershed in central Arkansas included the conversion of a retired zoysiagrass sod farm into upland hardwood forest.

Due to the management goal of improving water quality through this land use conversion, hardwood establishment protocols required a balance in hardwood establishment efficacy while minimizing erosion potential. The zoysiagrass had a relatively dense root mat that could suppress hardwood establishment, and field portions in which zoysiagrass had been removed and not replanted prior to the conversion project had an array of annual and perennial vegetation. Suppressing this vegetation with herbicides had the potential to foster hardwood establishment. However, broadcast application of soil-active herbicides with relatively long residual activity for competition suppression that could optimize hardwood establishment efficacy may have higher erosion potential due to longer duration of exposed soil. Application of herbicides in a band around each planting row could serve as an alternative to broadcast application that could maintain greater ground coverage of vegetation while providing competition control around seedlings. Other herbicide application factors that can affect the duration of exposed soil include application frequency and residual activity, with soil-active herbicides generally having longer residual suppression of competing vegetation than foliage-active herbicides. Cover crops could serve as an alternative to herbicides in order to maintain ground coverage while reducing competing vegetation; several cover crops have been shown to reduce soil erosion in the initial

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years of stand development in hardwood plantations (Malik and others 2000).

The objective of this study was to determine the effects of several competition suppression treatments on ground coverage trends in the initial years of hardwood forest planted on a retired zoysiagrass sod farm. While conducting this study, horseweed abundance increased within several herbicide-treated plots. This observation led to a second study with the objective of exploring the efficacy of herbicide treatments on horseweed suppression and hardwood tolerance of the herbicides.

MATERIALS AND METHODS

Study Site

A study site was established at a retired sod farm in central Arkansas in February 2013 near the town of Perryville (32° 52' 15.45" N, 92° 43' 47.59" W). Soils for the fields selected for the study were alluvial silt loam soils mapped as Rexor (fine-silty, siliceous, active, thermic Oxyquic Hapludalf), Amy (fine-silty, siliceous, semiactive, thermic Typic Endoaquult), and Sallisaw (fine-silty, siliceous, superactive, thermic Typic Paleudalf) series (USDA SCS 1982). Long-term average annual temperature and precipitation for the region of the study site is 16 °C and 132 cm, respectively (SCIPP 2017).

Treatments and Experimental Design

Primary study—Eight competition control treatments were conducted in 2013; treatments included broadcast and band application of herbicides, single and multiple pre-plant herbicide applications, and a ryegrass cover crop (table 1). Treatments were applied to 0.06-ha plots established in two fields at the sod farm. There were four replications of each treatment in each field applied in a completely randomized design. All herbicides were applied with a tractor-drawn sprayer with shielding to foster application accuracy by reducing drift potential. For treatments that had banded herbicide application,

the sprayer was outfitted with a boom that applied herbicide in a band 1.8 m in width along each planting row. In all treatments that included glyphosate, glyphosate was applied at 11.7 L ha⁻¹ as Accord® XRT (Dow AgroSciences, LLC, Indianapolis, IN). Sulfometuron methyl (hereafter referred to as “sulfometuron”) was applied at 209.7 g ha⁻¹ as Oust® XP (Bayer CropScience LP, Research Triangle Park, NC) in all treatments in which it was included. In August 2013, plots receiving the RYEGRASS treatment were treated with 2241 to 3921 kg ha⁻¹ of lime based on liming recommendations for ryegrass within soil tests performed for each plot. Ryegrass was planted for the RYEGRASS treatment in September 2013 by broadcast seeding 33.6 kg ha⁻¹ of pure live seed.

Plots were planted by hand with water oak and green ash in February 2014. The site was subsoiled to a 51-cm depth in November 2013. Seedlings were planted at 1074 trees ha⁻¹ at a spacing of 3.0 m × 3.0 m. The treatment structure of this study was a split-plot structure, with tree species as sub-plot factor and herbicide treatment as a whole-plot factor.

Horseweed study—In all treatments in the primary study that included sulfometuron, horseweed was observed as the primary vegetation that emerged after the herbicide was applied. To determine whether horseweed could be controlled by either adding an extra herbicide with sulfometuron methyl or using an alternative to sulfometuron, several treatments were conducted. Treatments included: (1) untreated control, (2) sulfometuron as Oust® XP (OUST), (3) sulfometuron as Oust® XP and sulfosulfuron as Outrider® (Valent BioSciences LLC, Libertyville, IL) (OUSTOUT), (4) clopyralid as Transline® (Dow AgroSciences, LLC, Indianapolis, IN) (TRAN), and (5) a pre-mixed granular blend of sulfometuron and metsulfuron methyl (OustExtra®, Bayer CropScience LP, Research Triangle Park, NC) (OUSTX).

Table 1—Pre-planting vegetation control treatments conducted for establishment of green ash and water oak at a retired zoysiagrass sod farm in central Arkansas

Treatment	Herbicide applied April 2013	Herbicide applied September 2013
CONTROL		
RYEGRASS	Broadcast glyphosate	Broadcast glyphosate
BrG	Broadcast glyphosate	
BaG+BaG	Banded glyphosate	Banded glyphosate
BaG+BaGS	Banded glyphosate	Banded glyphosate and sulfometuron methyl
BrG+BaG	Broadcast glyphosate	Banded glyphosate
BrG+BaGS	Broadcast glyphosate	Banded glyphosate and sulfometuron methyl
BrG+BrGS	Broadcast glyphosate	Broadcast glyphosate and sulfometuron methyl

In two fields at the sod farm, 0.02-ha plots were established in September 2014. Soil of one field was mapped as a Sallisaw series gravelly silt loam, and soil in the other field was mapped as a Leadville silt loam (loamy-skeletal, mixed, superactive Ustic Glossocryalf). Treatments were replicated three times in each field. Herbicides applied for all treatments were applied using a tractor-drawn sprayer with shielding to minimize drift. For the OUST and OUSTOUT treatments, sulfometuron was applied in October 2014 at 209.7 g ha⁻¹. Herbicide for the OUSTX treatment was also applied in October 2014 at 279.6 g ha⁻¹. In April 2015, sulfosulfuron was applied for the OUSTOUT treatment and clopyralid was applied for the TRAN treatment at 90.9 g ha⁻¹ and 0.7 L ha⁻¹, respectively.

In January 2015, water oak and cherrybark oak were hand-planted for the study. Cherrybark oak was planted instead of green ash in this trial because in 2014 the emerald ash borer (*Agilus planipennis* Fairmaire) was discovered in central Arkansas, and the recommended forest management practice to curtail ash borer infestation was to avoid planting green ash. Both tree species were planted in each plot as a sub-plot treatment factor.

Measurements

Primary study—In May 2014, September 2014, and June 2015, ground coverage assessments were performed. A 1 m × 1 m quadrat subdivided into 25 equally-sized cells was used to visually determine the percentage of ground covered by total, grass/sedge (hereafter referred to as “grass”), broadleaf forb, and woody vegetation. These measurements were taken at six subsample points per plot along a zigzag pattern within the center of the plot. Data for the subsample points were averaged for each plot for statistical analysis.

Horseweed study—In July 2015, ground coverage measurement was performed with a quadrat as conducted in the primary study. In this trial, horseweed coverage and total number of horseweed plants per quadrat were assessed at each sample point in addition to the same parameters as in the primary study. Horseweed plants per quadrat were also assessed in September 2015. In June 2015 and August 2016, tree damage was measured for each tree by visually assessing each tree and assigning a numerical damage score (table 2).

Statistical Analysis

Analysis of variance (ANOVA) was performed for all parameters in both studies using PROC GLIMMIX of

Table 2—Score system used to evaluate condition of cherrybark oak and water oak seedlings in response to herbicides applied to suppress horseweed at a retired zoysiagrass sod farm in central Arkansas

Tree condition	Tree condition score
No effect, vigorously growing	1
Chlorotic leaves	2
Dead leaves, upper stem	3
Dead leaves, lower stem	4
Dieback at top	5
Dieback to bottom, re-sprouting	6
Dieback to bottom, no re-sprouting	7
Chlorotic leaves, dieback at top	8
Dead	9

the SAS System 9.4 (SAS Institute, Inc., Cary NC) at a significance level of $p = 0.05$. For coverage variables of the primary study, a repeated measures model with an autoregressive correlation structure that included field, herbicide treatment, sampling date, and all possible interactions between the variables as fixed effects was used. Tree species was not considered for these analyses because the ground coverage component of this study was measured for the entire plot rather than the subplot.

For the horseweed study, ANOVA for total horseweed plants m⁻² was performed using the same model as used for the variables of the primary study. For ground coverage, grass coverage, broadleaf coverage, and horseweed coverage, the model used for ANOVA included field, treatment, and their interaction as fixed effects. The ANOVA model for tree damage scores was a repeated measures model with an autoregressive correlation structure that included field, date, species, treatment, and their interactions as fixed effects. The interaction of treatment and species was identified as an error term in the model to account for the split-plot treatment structure.

Means separations were performed for all variables when significant effects were determined by ANOVA. When significant main effects were determined, the LINES option of the LSMEANS statement was used to perform F-protected least significant difference (LSD) means separation. When significant interactions were determined, the SLICEBY and LINES options of the SLICE statement were used to perform F-protected LSD means separation of effects within the interaction.

RESULTS AND DISCUSSION

There were significant date × treatment effects in the analyses of total, grass, and broadleaf coverage in the primary study (figs. 1–3). There were no significant treatment × field effects in the analyses of these parameters in the primary study, so data presented and discussed are averaged for both fields. No vine vegetation was observed in the study. Generally, treatments that included sulfometuron methyl had lower vegetation (total, grass, broadleaf) coverage for a longer time. No treatments affected vegetation coverage by June 2015, which was 19 or 25 months after the vegetation control treatments were conducted.

Only the three treatments that included sulfometuron methyl had lower coverage than the CONTROL treatment in the May 2014 assessment (fig. 1). By the September 2014 assessment, only the BrG+BaGS treatment had lower total coverage than the CONTROL treatment. These results suggest that the greatest overall vegetation control at this site was achieved by broadcast application of glyphosate in the spring prior to planting and including sulfometuron in the herbicide application conducted in the fall prior to planting. Ezell (2002) similarly determined that inclusion of sulfometuron in fall herbicide applications was effective at suppressing vegetation in the following growing season.

There was greater differentiation among treatments in grass coverage than for total coverage (fig. 2). In May 2014, only the RYEGRASS treatment had similar grass coverage relative to the CONTROL. Although similar in coverage, the grass in the RYEGRASS treatment was overwhelmingly ryegrass, whereas the CONTROL treatment had a mixture of grass species. In the May 2014 assessment, the two treatments (BrG+BaGS, BrG+BrGS) that consisted of a spring broadcast application of glyphosate followed by fall application of sulfometuron and glyphosate had the lowest grass coverage. The BaG+BaGS treatment had greater grass coverage than the BrG+BaGS and BrG+BrGS treatments but lower coverage than all other treatments. This finding suggests that although the inclusion of sulfometuron in the fall banded herbicide application provided greater grass control than treatments that included only glyphosate, grass re-encroachment occurred sooner without being preceded by a spring broadcast application of glyphosate. All vegetation control treatments that consisted of only glyphosate had lower grass coverage than the CONTROL treatment in May 2014, but grass coverage of these treatments was greater than all treatments that included sulfometuron. These results illustrate the value of including sulfometuron in the fall pre-planting herbicide application for providing longer-term grass suppression in the year after planting.

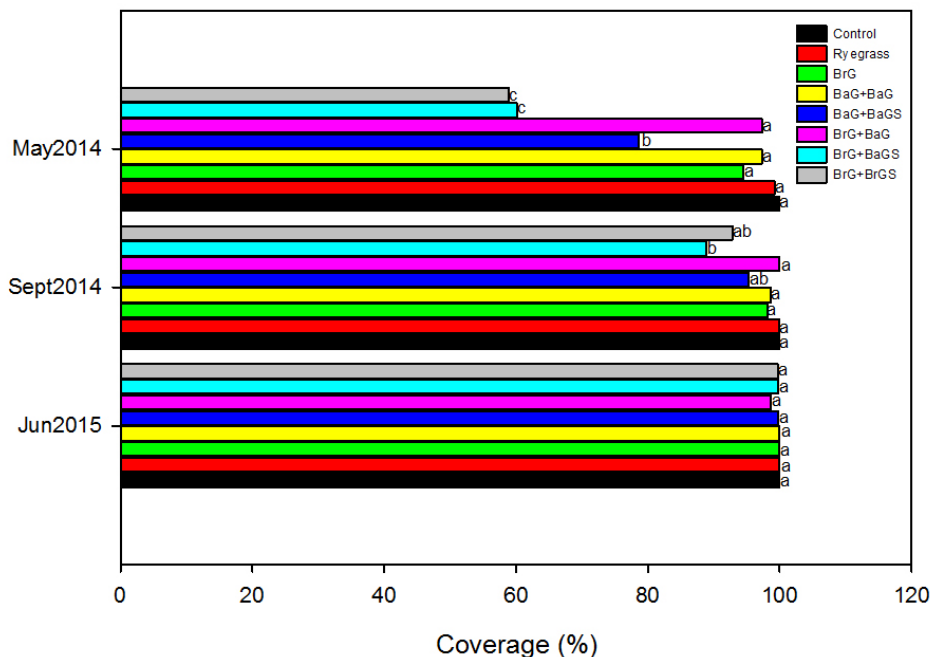


Figure 1 — Total ground coverage by vegetation in response to vegetation control treatments in response to herbicide treatments applied prior to planting of green ash and water oak at a retired zoysiagrass sod farm in central Arkansas. For each date, bars followed by a different letter are significantly different at $p < 0.05$.

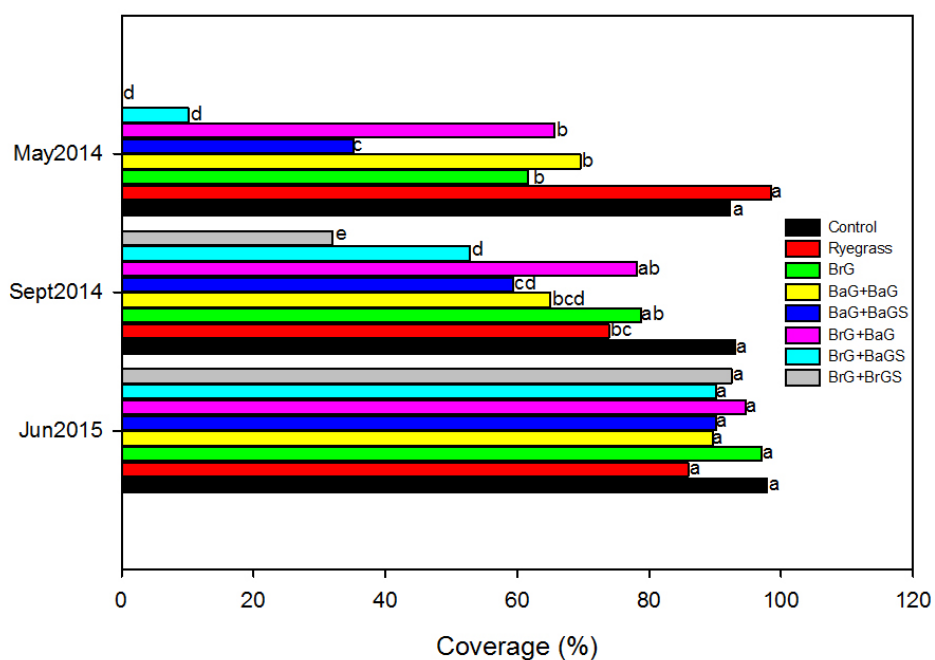


Figure 2—Ground coverage by grass vegetation in response to vegetation control treatments in response to herbicide treatments applied prior to planting of green ash and water oak at a retired zoysiagrass sod farm in central Arkansas. For each date, bars followed by a different letter are significantly different at $p < 0.05$.

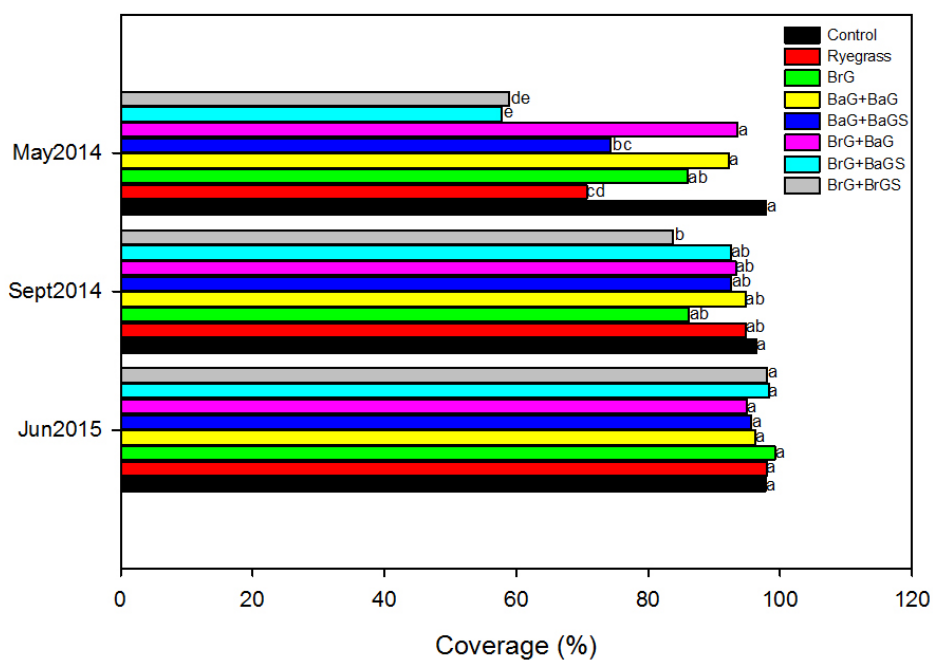


Figure 3—Ground coverage by broadleaf vegetation in response to vegetation control treatments in response to herbicide treatments applied prior to planting of green ash and water oak at a retired zoysiagrass sod farm in central Arkansas. For each date, bars followed by a different letter are significantly different at $p < 0.05$.

By September 2014, the BrG+BrGS treatment had lower grass coverage than all other treatments (fig. 2). Grass coverage of the BrG+BaGS treatment had become higher than that of the BrG+BrGS treatment, although it remained lower than that of the CONTROL, RYEGRASS, BrG, and BrG+BaG treatments. These results suggest that broadcast application of sulfometuron within the fall herbicide application led to the longest-lasting grass suppression throughout the first year following planting. Among the treatments that included only glyphosate, their similarity in grass coverage in all assessments suggests that there was no added benefit to the fall glyphosate application in terms of grass suppression in the year after planting. The RYEGRASS treatment had lower grass coverage than the CONTROL in September 2014. By September the ryegrass had died in this treatment, so this result suggests that it suppressed the establishment of other grasses during the first growing season after planting.

Broadleaf coverage differed among treatments through September 2014, but differences among treatments were not as marked as for grass coverage (fig. 3). In May 2014, similar to grass coverage results, the lowest broadleaf coverage was observed in the two treatments (BrG+BaGS, BrG+BrGS) that consisted of a spring broadcast application of glyphosate followed by a fall herbicide application that included sulfometuron. Also similar to grass coverage results, by September 2014 the BrG+BrGS treatment had lower broadleaf coverage than all other treatments. These results demonstrate that the highest efficacy for non-crop broadleaf suppression

was achieved by pre-planting broadcast applications of glyphosate in the spring followed by a broadcast application that included sulfometuron in the fall. The RYEGRASS treatment had lower broadleaf coverage than the CONTROL and all treatments that included only glyphosate. This result illustrates the efficacy of the ryegrass cover crop for suppressing non-crop broadleaf vegetation, although broadcast applications of glyphosate preceded planting of ryegrass to foster its establishment. The treatments that included only glyphosate proved ineffective for non-crop broadleaf vegetation suppression; broadleaf coverage of all these treatments was similar to the CONTROL in all observations.

In the horseweed study, there were significant field × treatment effects in the analyses of total ground coverage, grass coverage, horseweed coverage, and horseweed plants m⁻² (table 3). A substantial factor contributing to the different treatment rankings by field in this study was likely flooding of the field located on Sallisaw soil. In May 2015, the region received 33 cm of precipitation, with 11 cm occurring in a strong storm on May 11, 2015 (NOAA 2015). The field located on Sallisaw soil (referred to hereafter “Sallisaw field”) was located approximately 130 m from a creek that expanded and submerged the field for days after the May 11, 2015 storm. This inundation may have reduced herbicide activity in the Sallisaw field relative to the field on Leadville soil (henceforth referred to as “Leadville field”), which did not experience flooding.

Table 3—Vegetation coverage (percent) and crop tree condition score (1–9 scale) in response to herbicide treatments applied prior to planting of cherrybark oak and water oak to suppress horseweed at a retired zoysiagrass sod farm in central Arkansas

	Control	TRAN	OUST	OUSTOUT	OUSTX
Sallisaw field					
Total ground coverage	14.6 <i>ab</i>	22.5 <i>a</i>	6.2 <i>b</i>	13.6 <i>b</i>	14.4 <i>ab</i>
Grass coverage	13.7 <i>ab</i>	19.8 <i>a</i>	5.9 <i>a</i>	12.9 <i>ab</i>	14.2 <i>a</i>
Broadleaf coverage	2.4 <i>a</i>	3.8 <i>a</i>	1.3 <i>a</i>	0.2 <i>a</i>	0.1 <i>a</i>
Horseweed coverage	0.3 <i>a</i>	0.0 <i>a</i>	0.1 <i>a</i>	0.1 <i>a</i>	0.0 <i>a</i>
Tree condition score	2.9 <i>d</i>	3.3 <i>cd</i>	4.7 <i>ab</i>	3.9 <i>bc</i>	5.0 <i>a</i>
Leadville field					
Total ground coverage	24.0 <i>a</i>	22.2 <i>a</i>	2.2 <i>b</i>	3.0 <i>b</i>	2.0 <i>b</i>
Grass coverage	22.8 <i>a</i>	22.2 <i>a</i>	1.8 <i>b</i>	0.5 <i>b</i>	1.4 <i>b</i>
Broadleaf coverage	7.7 <i>a</i>	3.1 <i>a</i>	0.4 <i>a</i>	2.4 <i>a</i>	0.6 <i>a</i>
Horseweed coverage	1.3 <i>ab</i>	0.2 <i>b</i>	0.1 <i>b</i>	1.6 <i>a</i>	0.2 <i>b</i>
Tree condition score	1.9 <i>b</i>	2.1 <i>b</i>	2.4 <i>b</i>	2.6 <i>b</i>	4.5 <i>a</i>

For each row, numbers followed by a different letter are significantly different at $p < 0.05$.

Total ground coverage was generally lower with the addition of sulfometuron in the horseweed study (table 3). In the Leadville field, all treatments that included sulfometuron had lower ground coverage than the control and TRAN treatments. In the Sallisaw field, the OUST and OUSTOUT treatments had lower total coverage than the TRAN treatment. The higher total coverage of the TRAN treatment than that of the treatments that included sulfometuron is likely due to lower array of species suppressed by clopyralid relative to sulfometuron. Clopyralid was included in this trial because it is labeled to control horseweed, but clopyralid only suppresses broadleaf and woody brush species whereas sulfometuron can control broadleaf and grass species (Dow AgroSciences 2016, DuPont 2002). This tendency of the TRAN treatment to control only broadleaf vegetation was seen in the grass coverage results (table 3). The TRAN treatment had greater grass coverage than all treatments except the control in the Leadville field and greater grass coverage than the OUST treatment in the Sallisaw field. Clopyralid was not more effective than sulfometuron-containing herbicides at controlling broadleaf vegetation; all treatments had similar broadleaf vegetation.

There were no differences in horseweed coverage among treatments in the Sallisaw field (table 3). In the Leadville field, the OUSTOUT treatment had greater horseweed coverage than the OUSTX, TRAN, and OUST treatments (table 3). These horseweed coverage results are counter-intuitive because the sulfosulfuron component of the OUSTOUT treatment is labeled for horseweed suppression (Monsanto 2006) and horseweed was observed in relative abundance in the primary study that inspired the horseweed trial.

Horseweed coverage was generally low in all treatments, ranging from 0.11 to 1.6 percent. The short time between the heavy rainfall of May 2015 and the July 2015 coverage assessment and the small plot size of the horseweed trial were likely contributors to these results. Horseweed did not establish abundantly in any plots by the time of the coverage assessment. In the horseweed count assessment that was conducted in July and September 2015, there was a significant treatment effect (fig. 4). Across both fields and observation periods, the TRAN and OUSTX treatments had lower horseweed plants m^{-2} than the control treatment. These results suggest that clopyralid and a mixture of sulfometuron and metsulfuron methyl worked best at controlling horseweed at this site.

Despite its efficacy at suppressing horseweed, the OUSTX treatment was most damaging to the tree species tested (table 3). There was a significant date $\times$ treatment $\times$ field effect in the analysis of tree condition scores, with significant treatment $\times$ field effects observed in the June 2015 assessment. In the Leadville field, the OUSTX treatment had higher tree condition scores (higher scores indicate greater damage) than all other treatments. In the Sallisaw field, the OUSTX treatment had higher condition scores than all treatments except the OUST treatment. OustExtra[®] is not labeled for site preparation applications of water oak and cherrybark oak; its inclusion in this trial was to determine whether it could be used to suppress horseweed without damaging these species. The tree damage and horseweed plants m^{-2} results of this study suggest that clopyralid is a better herbicide for suppressing horseweed than OustExtra[®]. However, the inability of clopyralid to control grasses makes it less effective as a site preparation

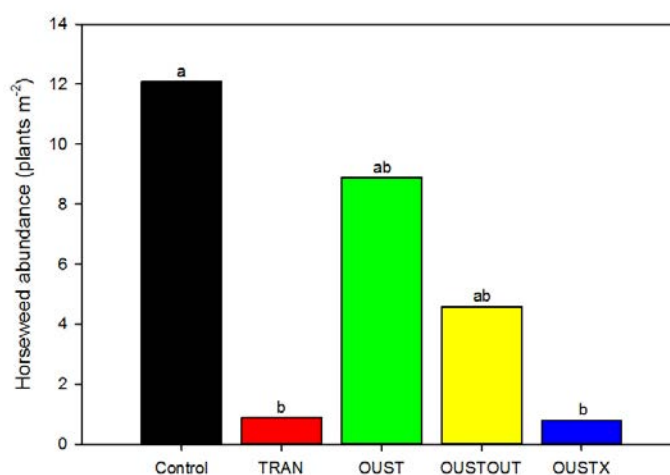


Figure 4—Horseweed abundance in response to vegetation control treatments in response to herbicide treatments applied prior to planting of cherrybark oak and water oak at a retired zoysiagrass sod farm in central Arkansas. Columns headed by a different letter are significantly different at $p < 0.05$.

herbicide. A future study should be devoted to tank mixtures of clopyralid with sulfometuron on cherrybark oak and water oak to determine whether such mixtures will best control an array of vegetation, including horseweed, without damaging these species. Current labeling for Transline® mentions Oust® as a compatible tank mixture, but water oak is not identified as tolerant tree species (Dow AgroSciences 2016).

CONCLUSIONS

Inclusion of sulfometuron in a fall glyphosate application prior to planting had the greatest and longest-lasting vegetation control in the first year after planting the hardwood species at this sod farm. The ryegrass cover crop was a viable option for maintaining organic matter on the soil surface while providing suppression of broadleaf and grass vegetation, although its efficacy for control of both vegetation types was lower than the fall application of sulfometuron. Management decisions between these options would be driven by compromises between competition control efficacy and continual maintenance of organic matter for water quality purposes. Both competition control options had at least 94 percent of the surface covered in living vegetation by May of the first growing season after planting. If relying solely on glyphosate for competition control with these site conditions, a single application in the spring prior to planting was as effective as applying glyphosate in the spring and fall prior to planting. Horseweed was resistant to sulfometuron, but clopyralid appeared effective as controlling horseweed without significantly damaging water oak and cherrybark oak planted at this site. Clopyralid tank mixed with sulfometuron might be an option for controlling horseweed in addition to the species suppressed by sulfometuron, but further testing is necessary to determine effects of this mixture on water oak and cherrybark oak.

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HARDWOOD ESTABLISHMENT ON COMPACTED MINE TAILINGS AFTER SUBSOILING AND NATIVE GROUND COVER SEEDING

Jennifer A. Franklin and Matthew Aldrovandi

Extended abstract—Since 1930, approximately 2.5 million ha have been disturbed by surface mining in the United States. On many mine sites reclaimed since the Surface Mining Control and Reclamation Act of 1977 both compacted soils and aggressive nonnative plant communities are common, and sites may require remediation in order to establish forests (Burger and others 2013). Planted grasses and legumes can compete strongly with planted tree seedlings, and there is a need to identify herbaceous species that are compatible with reforestation (Franklin and others 2012). Browse damage to planted tree seedlings can be extensive in the Southeast and may also be influenced by the composition of herbaceous vegetation.

An experiment was established to test for tree growth and establishment in four vegetation treatments on 8 ha of an old reclaimed mine site in northeastern Tennessee. Soil compaction was first relieved by subsoiling to a depth of 1 m. Bare-root 1-0 seedlings of 15 tree and shrub species were planted on a 2.4-m by 2.4-m spacing in late winter of 2015 (table 1). The herbaceous treatments were applied immediately following tree planting: 1) herbaceous control using an application of glyphosate herbicide in a 0.5-m radius around each planted tree in late spring, 2) seeding with a mixture of herbaceous annuals and perennials preferred by deer, 3) seeding with a mixture of herbaceous annuals and perennials that are not preferred by deer, and 4) untreated control. Pre-treatment composition of vegetation was surveyed in August of 2014 and was dominated by nonnatives with a total herbaceous cover of 93 percent. The composition of herbaceous vegetation was monitored in May and August of 2015.

Herbaceous cover was reduced to 66 percent across all treatments in May, and by August both seeded and control treatments had returned to initial levels (82–93 percent), while herbicide treatments still had significantly reduced cover (78 percent). Subsoiling did not reduce the frequency or dominance of lespedeza (*Lespedeza cuneata*), but did reduce the presence of tall fescue (*Festuca arundinacea*) from 52 percent of plots to 13 percent and its dominance from 34 to 4 percent of plots. At the end of the first growing season, the percentage of tree and shrub seedlings with obvious signs of browse damage varied from 9–64 percent (table 1). Binary logistic regression was used to test the effect of species and treatment on browse damage and survival. Seeding herbaceous species attractive to deer decreased the likelihood of deer browsing damage to planted seedlings, with 33 percent of seedlings being browsed. The highest rates of browse were seen in the treatment with low palatability herbaceous species seeded (45 percent) and herbicide treatment (43 percent), while the control treatment was intermediate with 37 percent of seedlings browsed. Survival and browse damage of trees, but not shrubs, was assessed in February of 2017, at which time survival averaged 38 percent and browse damage was similar to that seen in 2015. The planting of deer-preferred herbaceous species may be a viable strategy to reduce the incidence of browse damage to tree seedlings on young reforestation sites in this region.

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Table 1—Planting density of tree and shrub species, percent survival of trees (Feb. 2017), and percentage of seedlings browsed (Aug. 2015 and Feb. 2017)

Species		Stems/ha	Browse 2015 (%)	Browse 2017 (%)	Survival 2017 (%)
<i>Quercus rubra</i>	northern red oak	82	64	63	38
<i>Quercus muehlenbergii</i>	chinkapin oak	82	39	49	35
<i>Liriodendron tulipifera</i>	yellow poplar	82	9	8	37
<i>Quercus alba</i>	white oak	82	16	25	37
<i>Quercus palustris</i>	pin oak	41	48	55	48
<i>Cornus amomum</i>	silky dogwood	41	43	--	--
<i>Liquidambar styraciflua</i>	sweetgum	41	13	--	--
<i>Physocarpus opulifolius</i>	ninebark	41	40	--	--
<i>Prunus americana</i>	American plum	41	24	---	--
<i>Quercus falcata</i>	southern red oak	41	35	38	32
<i>Cephalanthus occidentalis</i>	buttonbush	41	53	--	--
<i>Callicarpa dichotoma</i>	beautyberry	20	40	--	--
<i>Corylus americana</i>	hazelnut	20	35	--	--
<i>Pinus strobus</i>	eastern white pine	14	4	--	--
<i>Castanea dentata</i>	American chestnut	7	56	66	24

Note: shrubs were not assessed in 2017, and too few sweetgum and pine were measured to report results.

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UNDERSTANDING CARBON SINK-SOURCE RELATIONSHIPS IN SEED ORCHARD LOBLOLLY PINE RAMETS

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James Tule, Phil Dougherty, and Nicholas G. Muir

Abstract—Attributes of shoots and their female reproductive organs from six ramets of a single loblolly pine clone were assessed in 2015 and 2016 at a Louisiana seed orchard. Seasonal developmental patterns were similar between years. Female strobilus buds were visible in December of year 1, and female strobili were pollinated in early March of year 2. Conelets (pollinated female strobili) stopped increasing in length from June of year 2 until March of year 3 when rapid expansion occurred over the next 4 months. In March and April of 2016, 61 percent of female strobili and conelets of the 2017 crop were aborted whereas a few conelets of the 2016 crop were lost. Competition for carbohydrate among various actively growing sinks in the spring may play a critical role in the abortion of female strobili and conelets. Balancing the carbon sink-source relationships among vegetative and reproductive organs should be considered when the goal is to enhance the production and retention of female reproductive organs.

INTRODUCTION

Loblolly pine (*Pinus taeda* L.) is an economically important pine in the Southeast United States due to extensive natural range and intensive tree improvement efforts resulting in faster growth and improved disease resistance. Productivity in plantations of genetically improved loblolly pine has increased 7–12 and 13–21 percent over unimproved seedlots for the first and second generations of tree improvement programs, respectively (Li and others 1999). During the last 2 decades, the third generation of genetically improved seedlings has been deployed, and the estimated plantation productivity increase is even greater, especially when combined with intensive silvicultural practices (McKeand and Bridgewater 1998, McKeand and others 2006). Increasing the formation and retention of female strobili and seed efficiency per cone in clonal seed orchard ramets is a critical part of the tree improvement process.

Correlations exist between the environment and cone and seed production. Dewers and Moehring (1970) showed that loblolly pine trees subjected to April–June irrigation followed by July–September drought increased the number of conelets per tree in a Texas seed orchard. Wenger (1957) reported a correlation between May to July rainfall and cone crop size 2 years later. Soil

fertilization at critical periods throughout the female reproductive cycle is also important (Schmidtling 1983, Wenger 1953). Several studies on shoot and foliage phenology as affected by silvicultural treatments, such as nitrogen fertilization and throughfall control, were conducted on plantation loblolly pine trees (Sword and others 1996, Tang and others 1999, Zhang and others 1997). There is little information, however, on the simultaneous phenology of vegetative shoots and their female reproductive organs in loblolly pine trees grown in plantations or seed orchards. To address this knowledge gap, a study was implemented on clonal loblolly pine in a Louisiana seed orchard in 2015 to evaluate developmental patterns in loblolly pine from the perspective of carbon sink-source relations. Our explicit objectives were: 1) to establish the temporal developmental patterns of vegetative and reproductive organs, and 2) to evaluate silvicultural treatments (soil fertilization and irrigation) for their impact on the production and retention of cones and on seed efficiency.

MATERIALS AND METHODS

Study Area and Experimental Design

The study was conducted at the International Forest Company's (IFCO) clonal seed orchard near Evans, LA. The experimental area is dominated by Hainesville

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fine sand having a 0–2 percent slope that is somewhat excessively drained (NCSS 2016). In January 2015, an experiment with a randomized complete block 2 x 3 factorial design with five blocks was imposed on 30 ramets of loblolly pine clone LSG-62. Blocking was based on competition index. For a given experimental unit (that is, a ramet), its competition index was calculated using equation 1.

$$CI = (CR \times 1.0) + (DR \times 0.7) \quad (1)$$

where

CI = the calculated competition index

CR = the number of adjacent ramets located in four cardinal directions

DR = the number of adjacent ramets located in four diagonals

Treatments were two levels of irrigation (ambient rainfall or ambient rainfall plus drip irrigation at 1090 L per ramet per day between July 10 and September 21, 2015) and three levels of soil fertilization [operational annual broadcast fertilizer application (OFA) in mid-July, OFA plus 28 Kg/ha of phosphorus (P) as triple super phosphate (mid-August), or OFA plus 28 Kg/ha of P as triple super phosphate (mid-August) and 112 Kg/ha of nitrogen (N) as ammonium nitrate (mid-September)]. The 2015 OFA included 115 Kg/ha of N as ammonium nitrate and 1.1 Kg/ha of boron (B) as Solubor®. Fertilizer treatments other than OFA were applied based on the ground area within the dripline of each ramet. The original study was revised in 2016 because no or minimal treatment differences were observed in 2015. The revised experiment is a randomized complete block design with four location-based blocks, three treatments, and two replications. Six of the 30 ramets in the 2015 study were omitted in the new design. Results from these six trees are reported here.

Shoot Selection, Measurements, and Cone Assessment

In January 2015, two first-order branches receiving full sunlight in the southwest quadrant of the upper one-third of the crown of each ramet were identified. Ten main and lateral shoots formed in 2014 were permanently tagged with numbered zip ties for shoot morphology and female reproduction measurements. The phenology of vegetative and reproductive organs was followed at a 4-week interval between February and October. Before needle fascicles emerged, flush length was measured from the base to the tip of the flush, and once needle fascicles began to emerge from their sheaths, only the foliated portion of a flush was measured. Lengths of needles were measured from the fascicle base to the tip. These measurements were repeated for second and third flushes. Day of assessment was expressed

as Julian day (JD) starting in 2015 and continuing through 2016.

One year before the start of this study, shoots in the upper crown of each ramet were bagged in early February 2014 for controlled mass pollination. These ramets were open-pollinated in 2015 and 2016. The number of female strobili on the main shoot and shoots lateral to the main shoot were recorded separately. At each measurement, one randomly selected female strobilus on the main shoot was measured for length. Mature cones were harvested at the end of September in 2015 and 2016. Cone morphological traits and number and dry weight of filled and partially filled seeds were obtained from all harvested cones. Each ovuliferous scale of a female strobilus bears two ovules which may eventually develop into seeds. Scales at the bottom of a female strobilus, although bearing wings, remain closed during pollination. The percent seed efficiency followed those of Bramlett and others (1977) with minor modifications as in equation 2.

$$SE = \frac{\left(\frac{FS}{2}\right) \times 100}{(TS - BUS)} \quad (2)$$

where

SE = the percent seed efficiency

FS = the number of filled seeds

TS = the total number of scales per cone

BUS = the number of base unopened scales

The potential seed efficiency was calculated in equation 3.

$$PSE = \frac{\left(\frac{FS + PFS}{2}\right) \times 100}{(TS - BUS)} \quad (3)$$

where

PSE = the percent potential seed efficiency

FS = the number of filled seeds

PFS = the number of partially filled seeds

TS = the total number of scales per cone

BUS = the number of base unopened scales

Most seeds were gently shaken from the open scales of the oven-dried cones. Some seeds were pried out from unopened scales sealed by resin from insect wounds or from neighboring injured cones.

Statistical Analysis

Results reported here were collected from the six ramets that were excluded from the revised study. Therefore, the 2015 treatment effects cannot be tested. Differences in flush length, flush number per year, and needle length were compared among years using a one-way analysis of variance (ANOVA) with the independent variable year as a class variable. Differences in flush length were compared among groups of shoots bearing different numbers of 2015 flushes using one-way ANOVA, also with year as the independent variable. Means were compared using Duncan's multiple range test at a significance level of 0.05 (SAS 2004). A paired t-test was used to compare the number of cones before and after apparent changes in cone number throughout cone development. Because there were multiple comparisons done on each trend line, a Bonferroni approach was used to maintain the experimentwise error rate for each line by using $p = 0.05/n$ where n was the number of comparisons done on each line.

RESULTS AND DISCUSSION

Vegetative Phenology

The lengths of the first flushes were similar in all years; however, lengths of the 2016 second, third, and all flushes combined were significantly shorter than those of the 2015 flushes (table 1). Needle lengths were similar among years. Needles of the third flush were shorter compared to those formed earlier in the year which can be attributed to cooler temperatures in the fall. These needles did not resume extension during the subsequent spring (data not shown). Compared to results from a loblolly pine plantation in central Louisiana (Tang and others 1999), the seed orchard ramets had longer needles but similar foliated flushes lengths.

Shoots grew more flushes per year in 2014 and 2015 than in 2016 (table 1). Thirty, 23, and 5 percent of shoots grew three flushes in 2014, 2015, and 2016, respectively. Conversely, 2, 5, and 20 percent of shoots had only one flush in 2014, 2015, and 2016, respectively. Decreased flush growth in 2016 can be partially attributed to the many cloudy or rainy days in March and also to the competition from the developing conelets of the 2016 cone crop which was substantial.

During the growing season, vegetative buds either developed into the subsequent flushes or remained dormant through the remainder of the season. These buds overwintered. The 2015 overwinter buds appeared in shoots with one, two, and three flushes in late March, mid-April, and late June of 2015, respectively (fig. 1). The lengths of these overwinter buds continued to increase slightly for the rest of 2015, and they did not differ in lengths until after JD 421 (February 25, 2016). This temporal development is similar to the observations made by Greenwood (1980) on loblolly pine ramets at a North Carolina seed orchard. The appearance of overwinter buds occurred between early June and early July, and the development of vegetative and reproductive primordia continued through the summer at that seed orchard. Lengths of the overwinter resting buds did not differ before spring of 2016 (fig. 1). Once the overwinter buds start elongating as the first flushes in spring of 2016, the final lengths of the first flushes seem to be closely associated with previous year's flush numbers. Shoots with one 2015 flush had shorter 2016 first flushes than those from shoots with two or three 2015 flushes (fig. 1). Total flush lengths in 2015 for shoots with one, two, or three flushes were, 10.4, 23.6, and 49.3 cm, respectively. Fascicles formed in

Table 1—Morphological parameters of vegetative shoots in six loblolly pine ramets in a Louisiana seed orchard

	2014	2015	2016
		<i>per year</i>	
Number of flushes	2.3a	2.2a	1.9b
		<i>length (cm)</i>	
First foliated flush	17.7a	18.3a	14.6a
Second foliated flush	6.8ab	8.6a	4.5b
Third foliated flush	1.9a	2.1a	0.3b
Total foliated flush	26.4a	29.0a	19.4b
First flush needle	18.8b	19.9a	19.1ab
Second flush needle	18.0ab	19.1a	17.9a
Third flush needle	15.3b	17.9a	17.0ab

Means followed by the same letter within a row are not significantly different at the 0.05 level.

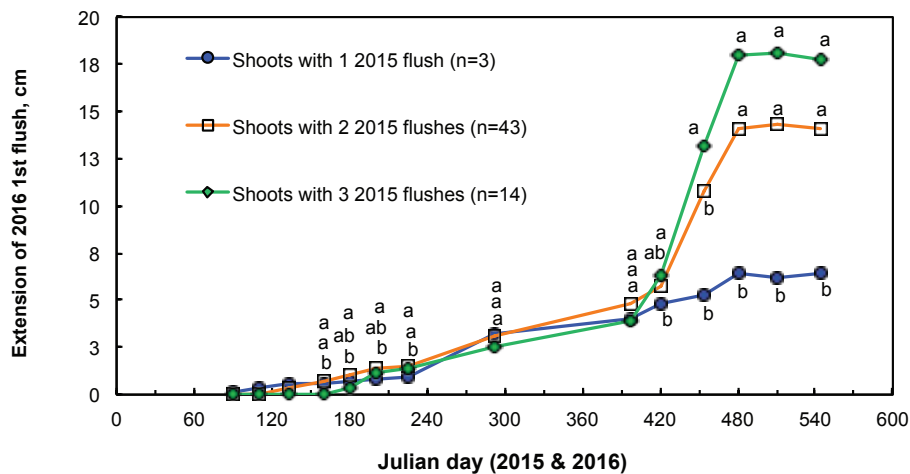


Figure 1—Extension of 2016 first flushes from 2015 overwinter resting buds on shoots with various numbers of 2015 flushes. Ten shoots from each of the six ramets of a single loblolly pine clone were tagged in early 2015 and monitored over time. Means with the same letter within a measurement date were not significantly different at the 0.05 level.

2015 are the main carbohydrate source for the growth of first flushes and their needles and developing female reproductive organs during spring of 2016.

Reproductive Phenology

It takes loblolly pine more than 2 years to complete its life cycle: from the formation of the female strobilus primordia to cone maturation and seed release (Bramlett and O'Gwynn 1980, Bramlett and others 1977, Greenwood 1980, Williams 2009). Male and female strobilus primordia form in early and late summer of year 1 (primordia year), respectively (Greenwood 1980). For clone LSG-62 at the IFCO seed orchard, some male and female strobilus buds were visible in October and November of year 1 (primordia year), respectively. Male and female strobilus buds break in January and February of year 2 (pollination year), respectively. Pollination of female strobili usually occurs in early March for this clone, and the pollinated female strobili become conelets. Fertilization of eggs in the archegonia of conelets does not take place until early June of year 3 (fertilization year) (Williams 2008, 2009). Fertilized conelets of LSG-62 usually mature in late September at the IFCO seed orchard.

Cones that matured in the fall of 2015, 2016, or 2017 were used to represent each crop (fig. 2). No significant loss in 2015 main or lateral cones occurred throughout year 3 (fig. 2). Number of 2016 main cones was sixfold greater than those on the shoots lateral to the main shoots. It was noted that some of the female strobilus buds did not emerge on the lateral shoots until mid-April of 2015, indicating the delayed development of female strobili on the lateral shoots. Overall loss of the 2016 cone crop from JD 90 (March 31, 2015; year 2) to JD 602 (August 24, 2016; year 3) was 16 and 21 percent for main and lateral shoot cones, respectively (fig. 2). An

80-percent retention of the conelets over 2 years can be attributed to the intensive insect control management. Insect damage can cause substantial loss of conelets and cones. In a clonal seed orchard in central Louisiana, a loss of 25–35 percent of female strobili and conelets during March and April of each of the 4 years monitored was attributed to insect damage by McLemore (1977). No apparent insect damage was observed in March and April of 2016 (year 3) at the IFCO seed orchard. The 2016 main cones numbered more than threefold those in 2015 whereas lateral cone numbers were similar in both years (fig. 2).

Maximum numbers of the 2017 main and lateral cones were reached by JD 421 (February 25, 2016; year 2) (fig. 2). However, the number of 2017 main cones decreased 27 and 23 percent in March and April of 2016, respectively. Loss of the 2017 cones from lateral shoots was even greater with 50- and 23-percent loss occurring in March and April of 2016, respectively. In contrast to the 2016 cone crop, the lateral shoots had as large a 2017 cone crop as the main shoots before the March abortion. By JD 602 (August 24, 2016; year 2), the number of 2017 lateral cones was similar to that of 2016 lateral cones, whereas the 2017 main cones were fewer than 50 percent of the 2016 main cones. As stated previously, heavy competition for current photosynthate occurs in March and April among developing flushes and their needles, female strobili, and conelets of previous and current years. Almost no loss in conelets of the 2016 cone crop during March and April of 2016 indicated that they are the stronger sinks of reproductive female organs. Compared to the 2015 season, shorter final lengths of the 2016 first flushes (table 1) implies that suboptimal weather for photosynthesis in March of 2016 caused more severe carbohydrate competition between reproductive and vegetative organs than in other years.

Cones and Seeds

Except for seed efficiency, parameters assessed for cones and seeds were similar between years (table 2). The open-pollinated cones of 2016 had greater seed efficiency and potential seed efficiency compared to the controlled mass pollination cones of 2015. It is generally acknowledged that the controlled mass-pollination cones have a lower seed efficiency than open-pollinated cones (Snyder and Squillace 1966). One way to increase seed efficiency is to increase carbohydrate allocation to partially filled seeds. An increase of 8 percent in seed efficiency would be realized if all partially filled seeds were fully filled (table 2).

Carbon Source and Sink Relationships

The temporal dynamics of loblolly pine carbohydrate sources and sinks were estimated non-destructively using organ length measurements as a surrogate for sink strength. Conventional measures of sink strength, such as net accumulation of dry matter, the synthesis rate of carbon reserves, the import rate of assimilate, or sucrose metabolizing enzyme activities, all require destructive sampling (Ho 1988, Sung and others 1989). Figure 3 presents the development patterns of sink strength in various carbohydrate sinks from JD 40 (February 9, 2015) to JD 544 (June 27, 2016). For clarity, 2015 third flushes and their fascicles were not included

Table 2—Parameters of the controlled mass pollination cone crop of 2015 and the open-pollinated cone crop of 2016 harvested from six loblolly pine ramets in a Louisiana seed orchard

Parameters	2015 crop	2016 crop
Cone (#/10 shoots/ramet)	15.2 (63.8)	33.8 (9.2)
Cone length (cm)	9.8 (7.1)	9.4 (3.2)
Cone dry weight (g)	22.1 (11.3)	20.8 (10.1)
Filled seed (#/cone)	93.7 (19.7)	102.3 (19.4)
Filled seed dry weight (mg/seed)	25.8 (8.1)	24.1 (7.1)
Filled seed dry weight allocation (%)	10.9 (22.9)	11.7 (14.5)
Partially filled seed (#/cone)	24.3 (28.3)	12.8 (71.1)
Partially filled seed dry weight (mg/seed)	3.2 (21.9)	4.4 (11.4)
Seed efficiency (%)	46.0 (8.7)	61.3 (9.6)
Potential seed efficiency (%)	57.6 (7.3)	69.2 (5.0)

Note: values in parentheses are coefficients of variation.

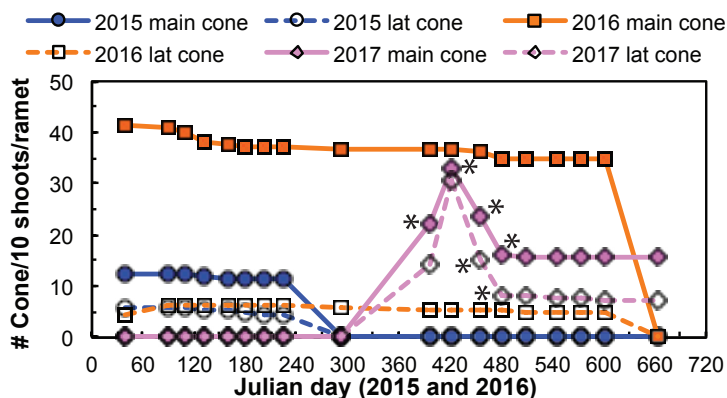


Figure 2—Temporal dynamics in mean numbers of 2015, 2016, and 2017 cone crops from main or lateral shoots in each of the six ramets of a single loblolly pine clone. The same 10 shoots from each ramet were monitored over time. Means with a * were significantly different from their previous data point in time using the Bonferonni paired t-test.

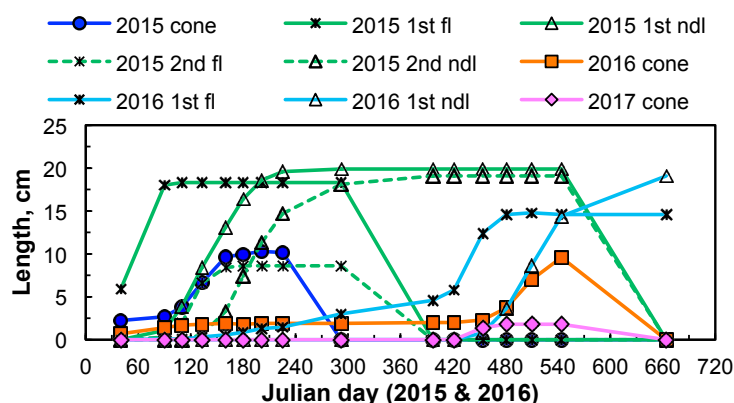


Figure 3—Temporal development patterns of sink strength in various carbohydrate sinks from loblolly pine ramets during 2015 and 2016. Lengths of individual organs were used as a surrogate for sink strength.

in this figure and neither were lateral shoots of the 2016 main shoots. By JD 160 (June 9, 2015), 2015 first flush fascicles reached 65 percent of their final length (fig. 3), became autotrophic, and no longer imported current photosynthate from 1-year-old fascicles (Dickson 1991). Based on the works of Chung and Barnes (1980) in North Carolina, we also predict that loblolly pine decreases the contribution of photosynthate from 1-year-old foliage to expansion of current-year second flushes and their fascicles by mid-August. In this study, second flush fascicles reached 62, 81, and 100 percent of final lengths by JD 201 (July 20, 2015), JD 225 (August 13, 2015), and JD 292 (October 19, 2015), respectively (fig. 3), which, when adjusted for location, is consistent with previous measures of second flush fascicle growth in central Louisiana (Sword and others 1996). In this study at the IFCO seed orchard in southwest Louisiana, ramets displayed more active and faster vegetative growth compared to loblolly pine in cooler areas (Chung and Barnes 1980) or in plantations (Sword and others 1996, Tang and others 1999).

Between February and mid-May of each year (JD 40 to JD 133 in 2015 and JD 397 to JD 510 in 2016), current photosynthate produced by 1-year-old foliage is allocated to all the developing carbohydrate sinks including current-year first flushes and fascicles originated from main and lateral overwinter buds, current-year second flushes, male strobili, female strobili, current-year conelets, and 1-year-old conelets. The abnormally wet and cloudy weather in March of 2016 may have contributed to the heavy abortion of female strobili in March and April for three reasons. First, reduced photosynthate production by the previous year's needles during this period of time may have resulted in strong competition among developing sinks. Second, 1-year-old conelets still rapidly expanded at this time, and their mortality was minor. This suggests a hierarchy in carbon allocation among sinks. In addition

to the aborted 2016 female strobili, the shorter 2016 first flushes may also lose in competition with 1-year-old conelet development. Third, the unusually wet weather in March may have resulted in poor pollination. Compared to other pine species, loblolly pine female strobili can tolerate less than optimal pollination and still be retained by the tree and develop (Williams 2009). In lodgepole pine (*P. contorta* var. *latifolia*), female strobilus drop occurred when less than 80 percent of ovules in a female strobilus were pollinated (Owens and others 2005). The very scarce pollen available to female strobili has resulted in the extensive abortion of female strobili in March 2016. Continued abortion of female strobili in April may have been a direct result of photosynthate allocation to other stronger reproductive and vegetative sinks.

The current-year first flush needles only become a carbohydrate source after June when most of the weaker sinks, female strobili and conelets, had already been aborted. Furthermore, it takes 16 to 17 weeks for loblolly pine seeds to become mature from the fertilization of eggs in early June (Kapik and others 1995, Williams 2009). In other words, the immature cones almost reach their final length before egg fertilization takes place (fig. 3). Therefore, from June to December, needles from current-year first and second flushes are carbon sources for the developing seeds in the immature cones (current-year cone crop) and formation of the female strobilus primordia in overwinter buds (cone crop to be mature 2 years later). During the subsequent spring, these 1-year-old needles are the major contributor for growth of both vegetative and male and female reproductive organs. Thus, the dynamics of source and sink allocations by loblolly pine spans 2 years. This complicates the implementation of efficient silvicultural treatments, such as soil fertilization, to enhance cone production and retention.

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UNDERSTANDING THE FERTILIZATION AND IRRIGATION RESPONSE OF SEED ORCHARD LOBLOLLY PINE

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James Tule, Nicholas G. Muir, and Phil Dougherty

Abstract—A study of loblolly pine (*Pinus taeda* L.) reproduction began in January 2015 at the International Forest Company seed orchard near Evans, LA. Three levels of fertilization and two levels of irrigation were planned to favor reproduction. Fertilization levels were August broadcast application of (1) 112 kg ha⁻¹ nitrogen plus boron (1N), (2) 1N plus August application of 28 kg ha⁻¹ phosphorus (1N1P), or (3) 1N1P plus September application of 112 kg ha⁻¹ nitrogen (2N1P). Irrigation treatments were no irrigation or drip irrigation of 1090 l ramet⁻¹ day⁻¹ between mid-July and late September. Foliar nutrition and vegetative and reproductive growth were not affected by fertilization or irrigation in summer of 2015. This may be attributed to ramet adaptation to edaphic conditions since establishment. Correlation between reproductive carbohydrate demand and photosynthate supply validates the importance of available carbohydrate to new vegetative and reproductive organs. Ramet environment and shoot order are likely to affect these relationships.

INTRODUCTION

Genetic selection has produced commercial sources of loblolly pine (*Pinus taeda* L.) with superior growth and wood quality, providing land managers an opportunity to plant improved seedlings originating from open pollinated (OP) or controlled mass pollinated (CMP) seeds. Relative to woods-run seed sources, OP and CMP seed sources are expensive because of costs incurred during genetic improvement and seed production as well as the year to year variability on seed crops among ramets of a single clone. Operational treatments that increase female strobili emergence and retention have the potential to increase OP and CMP seed production and reduce the cost of genetically improved seedlings.

A 4-year study is underway in the International Forest Company seed orchard near Evans, LA to assess fertilization and irrigation effects on female reproductive processes among ramets of a single clone. Our objectives are twofold. First, we are evaluating operational irrigation and fertilization for their effects on female strobili emergence and retention, cone production, and seed efficiency. Second, in response to these treatments, we are also conducting measurements of vegetative and reproductive phenology, leaf area, and

nutrition dynamics. Together, these data allow a better understanding of the effects on loblolly pine reproduction from interactions among mature foliage as a source of carbohydrate, the carbohydrate and nutritional demands of immature vegetative and reproductive organs, available soil resources, and climate.

Our present purposes are to document soil and climatic characteristics, quarterly stemwood growth, and seasonal foliar nutrition among 30 ramets of a single clone; among a subset of these ramets, we are characterizing early spring relationships between variables representative of carbohydrate supply and both vegetative and reproductive organs. We hypothesize that irrigation and fertilization will favor fascicle and female strobili primordia differentiation in late summer, with corresponding increases in the growth of vegetative and reproductive organs the following spring.

The large-scale longleaf pine restoration effort outlined in the Range-wide Conservation Plan for Longleaf Pine (America's Longleaf 2009) requires a sustained supply of seed. Crane and Barbour (2009) estimated an annual longleaf pine seed demand of 27,500 pounds to meet this need, but reported a range-wide inventory of viable seed at 7,500 pounds. Longleaf pine seed

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supply problems stem, in part, from the infrequency of good or better cone crops. Therefore, a long-range goal of this study is to provide a foundation from which a better understanding of longleaf pine reproduction can be pursued.

MATERIALS AND METHODS

Study Site and Experimental Design

The International Forest Company seed orchard near Evans, LA is located on the flood plain of the Sabine River. The dominant soil series is Hainesville fine sand. The orchard consists of open-grown loblolly pine ramets with a ground cover of frequently mowed warm season grasses and forbs. In December 2014, we permanently identified 30 ramets of loblolly pine clone LSG-62 at age 25 years in orchard 216. We established a randomized complete block, two by three factorial experimental design with five blocks in January 2015. Blocking was based on the presence or absence of neighboring ramets.

Two levels of irrigation treatment were ambient rainfall or ambient rainfall plus drip irrigation under ramet crowns of 1090 l ramet⁻¹ day⁻¹ between July 10 and September 21, 2015. Three levels of planned fertilization treatment were (1) broadcast application of 112 kg ha⁻¹ nitrogen (N) as ammonium nitrate (AN) plus 1.1 kg ha⁻¹ boron (B) as Solubor® in mid-July (1N), (2) 1N plus 28 kg ha⁻¹ ramet⁻¹ of phosphorus (P) as triple super phosphate applied in mid-July (1N1P), and (3) 1N1P plus 112 kg ha⁻¹ ramet⁻¹ N as AN in mid-August (2N1P). The absence of precipitation in July through early August 2015 caused a delay in the application of 1N (applied on August 11, 2015) and the mid-August application of 2N1P (applied on September 9, 2015).

A weather station (Vantage Pro2, Davis Instruments, Vernon Hills, IL) located at the seed orchard office began continuous operation in April 2015. Data between March 29, 2016 and May 25, 2016 were not recorded due to a power outage caused by an unprecedented flood event, but comparable weather station data for these dates were obtained from the National Oceanic and Atmospheric Administration's Lake Charles, LA Regional Airport weather station (National Oceanic and Atmospheric Administration 2017). Measurement variables included air temperature (°C, degrees Celsius) recorded by a 5-minute interval and daily total rainfall (cm). We calculated daily average air temperature, summed total rainfall by month, and estimated monthly water surplus and deficit by the Thornthwaite water balance model (McCabe and Markstrom 2007). We determined values of plant-available water holding

capacity (PAWHC) using a rooting depth of 1.0 m or 2.5 m [PAWHC: 4.7 cm (1.0 m), 11.5 cm (2.5 m)].

In January 2015, we sampled mineral soil by a soil probe to a 15-cm depth at 10 random locations around the crown drip line of each ramet and pooled by ramet. We quantified soil water content at field capacity (~0.03 megapascal, MPa) and permanent wilting point (~1.5 MPa) and estimated PAWHC among experimental ramets by the pressure plate method (Klute 1986). Through services at Waters Agricultural Laboratories in Camilla, GA, we also evaluated soil samples for available P, exchangeable potassium (K), magnesium (Mg), and calcium (Ca); hydrogen soil pH; and concentrations of sulfate-sulfur (S), B, zinc (Zn), manganese (Mn), iron (Fe), and copper (Cu).

Twice, we also sampled soil at a 0–15-cm depth within the study area (two locations in February 2015 and three locations in May 2016) as part of an operational assessment of soil fertility across the seed orchard. Commercial soil fertility analyses at Waters Agricultural Laboratories in Camilla, GA used standard methods but differed by extraction solution. A Mehlich I extract was used for the January 2015 samples across orchard 216, and a Mehlich III extract was used for the February 2015 and May 2016 samples (Mehlich 1984).

Ramet Stem Growth

We measured diameters at breast height (dbh, 1.37 m above the soil surface) of each experimental ramet by diameter tape in December 2014. In January 2015, we fitted experimental ramets with diameter bands at dbh (Keeland and Young 2014). After a 3-week equilibration period, we measured ramet diameters quarterly in February, May, August, and November. In February 2016, it was apparent that the diameter bands did not contain enough leader to measure dbh through 2018. As a result, in February 2016, we measured dbh by both diameter band and diameter tape and then removed the bands. Thereafter, we measured dbh by diameter tape and calculated quarterly dbh growth and daily dbh growth by quarter for each ramet.

Experimental Shoot Leaf Area and Phenology

For each ramet, we identified two or three first or second order branches receiving full sunlight in the upper third of the southwest side of the crown in January 2015. Among these, we permanently identified 10 main or lateral shoots by numbered zip ties. We determined the order of experimental shoots with 1M shoots designated as the terminal bud of a first order branch, 2M shoots designated as the terminal bud of a second order branch, and 2L shoots designated as a lateral bud of a

second order branch. In January 2015, we measured foliated internode and mean fascicle lengths by flush produced in 2014 among experimental shoots. In 3- to 4-week intervals between February 2015 and May 2016, we monitored foliated length by flush of the main shoots, median fascicle length by flush, numbers of female strobili and conelets by main and lateral shoots, median main shoot conelet and cone lengths, and number of lateral shoots with female strobili.

In November 2015, we excised one sample shoot from the north side of the upper third of ramet crowns. For each sample shoot, we assessed main and lateral shoots for flush number, foliated internode length, number of fascicles, and average fascicle all-sided surface area by the water displacement method (Johnson 1984). We estimated leaf area by flush as the product of fascicle number and average fascicle leaf area. By the method of Murthy and Dougherty (1997), we conducted regression analyses by shoot order and flush to establish regression coefficients for estimating flush leaf area by a combination of both foliated flush length and fascicle density. Subsequently, we made predictions of leaf areas by shoot order and flush produced in 2014 and 2015. Estimates of experimental shoot leaf area produced in 2014 utilized fascicle densities of sample shoot foliage produced in 2015. We used the sum of leaf areas produced in 2014 and 2015 to estimate the 2015 peak leaf area of experimental shoots.

In May 2016, due to a change in the experimental design, we excluded six of the 30 experimental ramets from the study. Among excluded ramets, we calculated mean values by shoot order (i.e., 1M, 2M, 2L) and ramet between January 2015 and May 2016 for the following variables: foliated length of the first flush produced in 2016 by late June (Julian date 180), bud length of the second flush produced by late June 2016, foliated length of the total shoot produced in 2015, estimated leaf areas produced in 2014 and 2015, peak leaf area of experimental shoots in 2015, number of second-year conelets in January 2016, number of female strobili emerged from main and lateral shoot buds in February 2016 (Julian date 56), number of first-year conelets (i.e., formerly female strobili in February 2016) remaining in April 2016 (Julian date 116), and numbers of main and lateral shoot female strobili or first-year conelets lost between February and April 2016.

Foliar Nutrition

We evaluated the foliar nutrition of experimental ramets six times between March 2015 and May 2016. Each sample consisted of 25 randomly chosen, healthy fascicles having three needles without lesions

from the first or second flush of 2014 or 2015 in the upper crown excluding the fascicles of experimental shoots. We oven-dried fascicle samples to equilibrium at 70 °C and shipped them to Waters Agricultural Laboratories (Camilla, GA) for macronutrient and micronutrient analyses.

Statistical Analyses

We subjected daily dbh increment by quarter and periodic foliar mineral nutrient concentrations to analyses of variance by a two by three factorial, randomized complete block experimental design with five blocks based on the proximity of neighboring ramets. Main effects were two levels of irrigation and three levels of fertilization. We compared means by the Least Significant Difference test and considered them to be significantly different at an alpha level of 0.05.

Among the six experimental ramets that were excluded from the study in May 2016, we conducted source-sink correlation assessments and determined Pearson correlation coefficients for variables representing the carbohydrate demand of reproductive and vegetative organs (number of second-year conelets per experimental shoot in January 2016, numbers of female strobili or first-year conelets per experimental shoot by main and lateral buds on Julian dates 56 and 116 of 2016, numbers of female strobili or first-year conelets lost from main and lateral buds between Julian dates 56 and 116 of 2016, first flush foliated length by late June 2016, and second flush bud length by late June 2016), and the potential supply of carbohydrate by foliage (2015 foliated shoot length, leaf area of shoots produced in 2015, and peak leaf area in 2015). We considered the F statistics and coefficients of determination to be significant at $p \leq 0.05$.

RESULTS

Climate and Soil

Climate measurements were variable by month and year. During the 16-month period between January 2014 and April 2015, month-to-month differences in water surplus were large (fig. 1A). A 3-month period of water deficit occurred between July and September 2015 with a magnitude that decreased as the root zone increased from 1.0 m to 2.5 m (fig. 1B). Mean daily air temperature was greatest in the summer months of June through August 2015 but was less variable at this time compared to fall and winter (data not shown).

Soil physical properties at the 0- to 15-cm depth showed low water holding capacity of Hainesville sand (table 1). Soil water contents at field capacity (SWC_{fc} , -0.03 MPa),

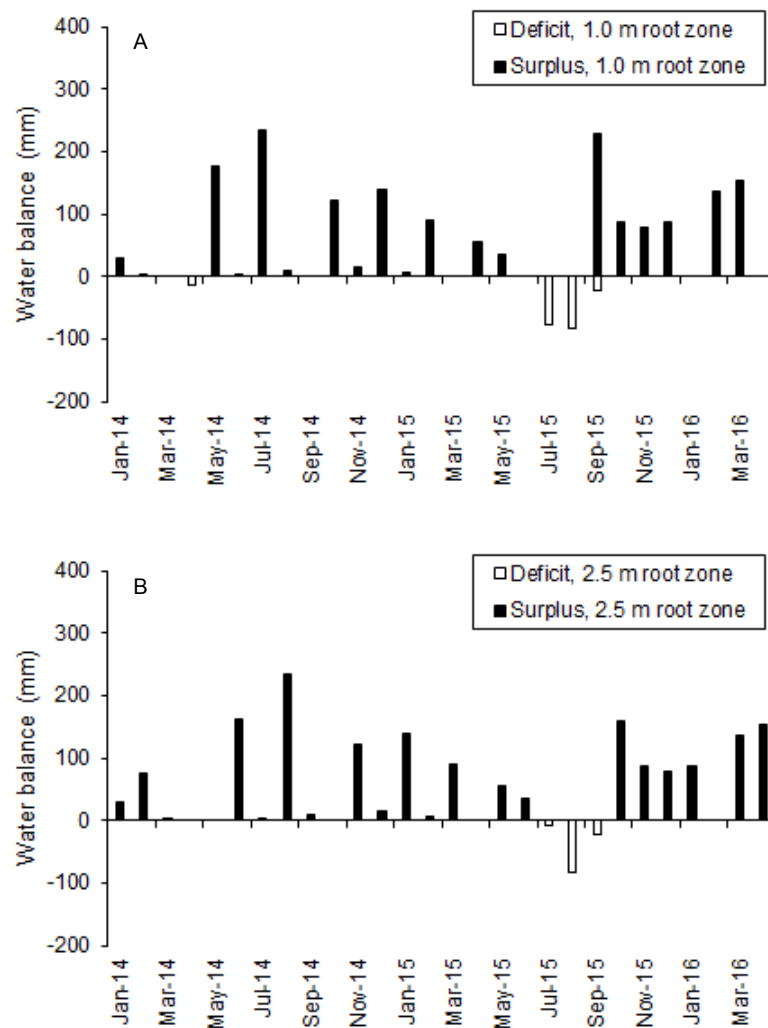


Figure 1—Monthly water surplus and deficit predicted by the Thornthwaite water balance model (McCabe and Markstrom 2007) for the International Forest Company seed orchard near Evans, LA between January 2014 and April 2016 using a root depth of (A) 1 m and (B) 2.5 m.

permanent wilting point (SWC_{wp} , -1.5 MPa), and PAWHC averaged 7.1, 2.7, and 3.4 percent, respectively. In January 2015, surface mineral soil levels of P, K, Mg, and Ca were within or greater than the recommended range for loblolly pine management (Moorhead and Dickens 2002) (table 2). Comparisons among Mehlich III estimated or actual values of soil P, K, Mg, and Ca in January 2015, February 2015, and May 2016 suggested that soil fertility was adequate for loblolly pine but soil P availability was variable.

Ramet Stem Growth and Foliar Nutrition

Mean daily increment of ramet diameter growth was not significantly affected by irrigation or fertilization in the two 3-month quarters of 2015 after the treatments

started or the first 3-month quarter of 2016 (fig. 2). Foliar macro- and micronutrient concentrations by flush, production year, and foliage condition (green or brown/senescent) were not significantly affected by fertilization or irrigation treatments between July 2015 and May 2016. Dormant season K, N, P, Mg, Ca, and S concentrations of green first flush foliage produced in 2014 and 2015 maintained levels at or above the threshold between sufficiency and deficiency for loblolly pine (Albaugh and others 2010) (fig. 3). Dormant season micronutrient concentrations were variable but consistently at or above sufficient levels [B: 7–34 parts per million (ppm), Zn: 21–77 ppm, Mn: 129–541 ppm, Fe: 43–105 ppm, Cu: 2.6–8.8 ppm] (Albaugh and others 2010).

Table 1—Mean mineral soil physical properties of the 0- to 15-cm depth among four spatial blocks of seed orchard 216

Variable ^a	Orchard block ^b			
	2	3	7	8
n	9	7	5	9
Bulk density (g cm ⁻³)	1.37 (0.03)	1.36 (0.02)	1.34 (0.02)	1.31 (0.02)
SWC _{fc} (% g g ⁻¹)	6.9 (0.9)	7.8 (2.0)	6.8 (0.6)	6.9 (0.4)
SWC _{wp} (% g g ⁻¹)	2.4 (0.3)	2.5 (0.3)	3.0 (0.4)	2.9 (0.2)
PAWHC (% g g ⁻¹)	3.6 (1.2)	4.4 (1.9)	2.7 (1.0)	3.0 (0.4)

^a n = number of samples; SWC_{fc} = soil water content at field capacity (-0.03 MPa); SWC_{wp} = soil water content at wilting point (-1.5 MPa); PAWHC = plant-available water holding capacity.

^b Values in parentheses are the standard deviation of the mean.

Table 2—Mineral soil chemical properties of the 0- to 15-cm depth among experimental ramets in orchard 216 in January 2015, and among soil sampled from orchard 216 in February 2015 and May 2016 as part of annual soil fertility assessments

Soil variable ^a	January 2015 mean		Critical value		February 2015 mean	May 2016 mean
	n ^b =30		-		n = 2	n = 3
	Mehlich I value	Mehlich III estimate ^d	Mehlich I value	Mehlich III estimate ^d	Mehlich III value	Mehlich III value
<i>kg ha⁻¹</i>						
P	13 (38) ^c	61	9–13 ^e	27–62	45	48
K	43 (14)	47	21–33 ^e	25–37	57	63
Mg	102 (19)	112	22–37 ^e	45–57	100	124
Ca	772 (25)	740	78–126 ^e	289–319	684	767
S	44 (17)	-	- ^f	-	19	19
B	0.2 (36)	0.8	-	-	0.2	1.0
Zn	6.1 (45)	8.7	-	-	7.1	7.6
Mn	46 (19)	103	-	-	202	323
Fe	35 (28)	-	-	-	277	327
Cu	2.7 (49)	5.3	-	-	4.9	7.1
pH _w	5.5 (4)	-	-	-	6.5	6.2

^a Soil variables are available phosphorus (P), potassium (K), magnesium (Mg), calcium (Ca), sulphur (S), boron (B), zinc (Zn), manganese (Mn), iron (Fe), and copper (Cu); pH_w is soil pH using water as an extract.

^b Number of samples.

^c Values in parentheses are coefficients of variation.

^d The extraction method changed from Mehlich I to Mehlich III between January and February 2015. For comparison among dates, application of available equations that express Mehlich III values by Mehlich I values were applied (Mylavarapu and others 2002) to express means on the basis of a Mehlich III extraction.

^e Critical values delineating insufficiency by Moorhead and Dickens (2002).

Dashes indicate that elements were not evaluated.

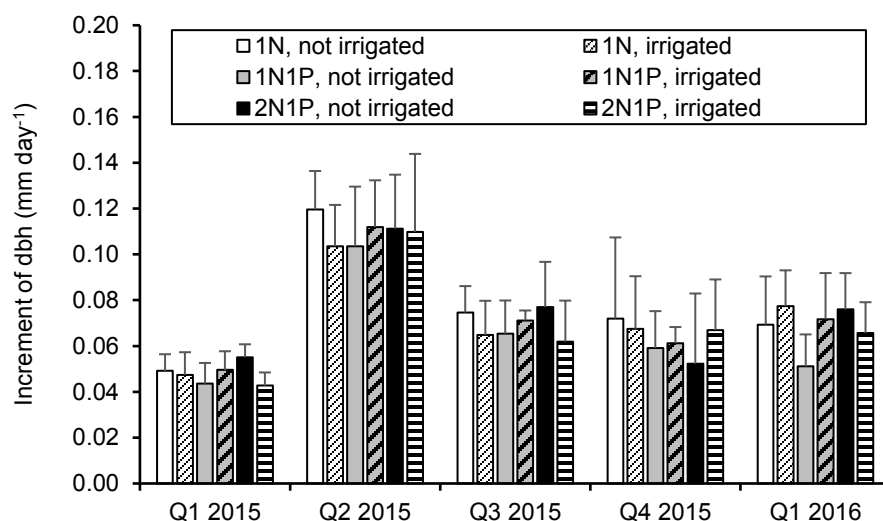


Figure 2—Mean daily increment of ramet diameter growth at breast height (dbh, 1.37 m) of loblolly pine clone LSG-62 estimated by annual quarter in response to two levels of irrigation and three levels of fertilization. The first, second, third, and fourth quarters of 2015 and first quarter of 2016 correspond to December 2014–April 2015, May–July 2015, August–October 2015, November 2015–January 2016, and February–April 2016, respectively. The first and second quarter of 2015 are pretreatment values. Error bars represent one standard error of the mean.

Source-Sink Correlation Assessments

Among the 1M, 2M, and 2L shoot orders of the six experimental ramets excluded from the study in May 2016, we observed significant, positive correlations between mean shoot numbers of main and lateral bud female strobili on Julian date 56 and first-year conelets on Julian date 116 and all shoot means of variables representing carbohydrate supply (2015 foliated shoot length, leaf area of shoots produced in 2015, and peak leaf area in 2015) (table 3). We also observed similar significant, positive relationships for the 2016 first flush foliated length and second flush bud length by late June when correlated with mean shoot foliated length in 2015, mean shoot leaf area produced in 2015, and mean shoot peak leaf area in 2015. Compared to the 2M and 2L shoot orders, the 1M shoot order had greater mean values of 2016 first flush foliated length, 2016 second flush bud length, and 2015 foliated shoot length (fig. 4) as well as 2015 leaf area and 2015 peak leaf area (data not shown). The 1M shoot order also had greater mean shoot numbers of main and lateral bud female strobili on Julian date 56 and first-year conelets on Julian date 116 compared to the 2M and 2L shoot orders (fig. 5). The loss of female strobili or first-year conelets between Julian dates 56 and 116 from the 1M (main: 3.5 shoot⁻¹; lateral: 8.2 shoot⁻¹) and 2M (main: 5.8 shoot⁻¹; lateral: 1.1 shoot⁻¹) shoot orders was larger than those from the 2L (main: 1.1 shoot⁻¹; lateral: 0.5 shoot⁻¹) shoot order.

Percentages of loss for 1M, 2M, and 2L shoots, however, were similar for main (49 ± 6 percent) and lateral buds (73 ± 4 percent). We did not detect significant correlations between second-year conelet number in January 2015 and variables representing carbohydrate supply in 2015.

DISCUSSION

Southern pines are well adapted to sandy soil. Belowground attributes such as a taproot or sinker roots and the horizontal elongation of first order lateral roots form a large network of fine roots and ectomycorrhizae in the upper horizons of the soil profile (Eissenstat and others 1994, Harrington and others 1989). Comparison of the relative areas of foliage and absorbing roots on sites characterized by high and low levels of soil resource supply has demonstrated the plasticity of southern pine root system architecture (Addington and others 2006, Hacke and others 2000). Hacke and others (2000), for example, compared two loblolly pine stands of similar stand densities growing on sand or clay loam soils. They found more than a fivefold greater ratio of root-to-leaf surface area on the sand compared to the clay loam. We anticipated that loblolly pine ramets growing on Hainesville sand exhibited a similar root system architecture and would be effective foragers of irrigation water and nutritional amendments applied to the soil surface.

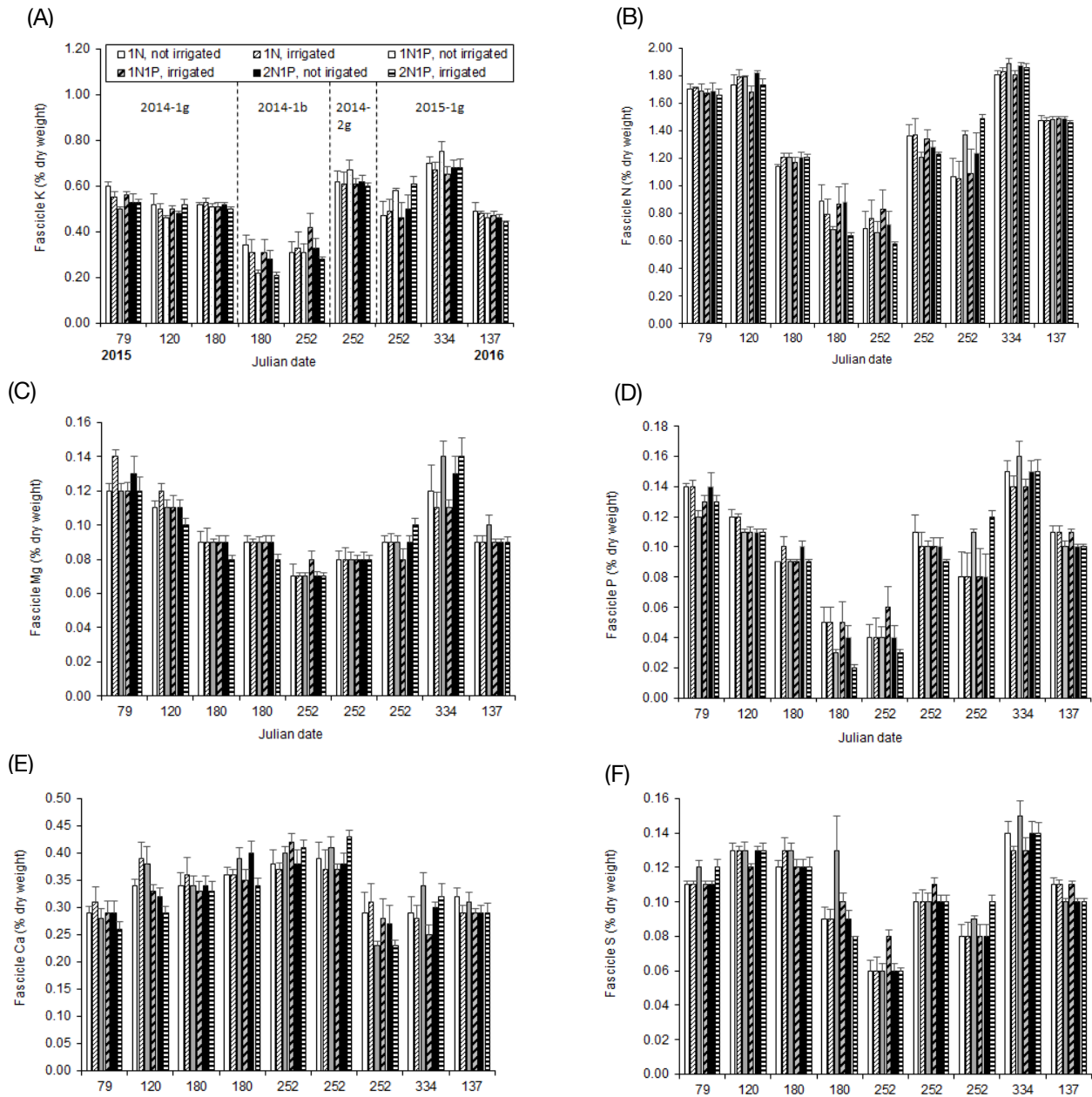


Figure 3—Mean mineral nutrient concentrations of foliage types in response to six combinations of irrigation and summer fertilization treatments. Foliage types include green first and second flush foliage produced in 2014 (2014-1g and 2014-2g, respectively), green first flush foliage produced in 2015 (2015-1g), and brown senescent first flush foliage produced in 2014 (2014-1b) between Julian date 79 of 2015 and Julian date 137 of 2016. For all panels, the dashed lines and abbreviations for foliage type in panel (A) delineate changes in foliage type among all panels. Panels represent information about (A) potassium (K), (B) nitrogen (N), (C) magnesium (Mg), (D) phosphorus (P), (E) calcium (Ca), and (F) sulphur (S). Error bars represent one standard error of the mean.

Table 3—Pearson correlation coefficients between vegetative and reproductive carbohydrate sinks and variables representative of carbohydrate supply^a

Carbohydrate sink variable	Carbohydrate supply variables		
	2015 foliated length (cm)	2015 leaf area (cm ²)	2015 peak leaf area (cm ²)
June 2016 foliated flush length (cm)	0.8241 ^b	0.7770	0.7551
June 2016 second flush bud length (cm)	0.7863	0.7626	0.7813
January 2016 second-year conelets in (#)	0.3633	0.4078	0.3845
February 2016 main bud female strobili (#)	0.9175	0.9101	0.8923
April 2016 main shoot first-year conelets (#)	0.7618	0.7957	0.7225
February 2016 lateral bud female strobili (#)	0.8857	0.9000	0.9033
April 2016 lateral shoot first-year conelets (#)	0.7229	0.7512	0.7209
2016 main bud female strobili or first-year conelet loss (#)	0.8790	-0.8864	-0.9015
2016 lateral bud female strobili or first-year conelet loss (#)	0.8387	-0.8030	-0.8287

^a Data are means by three shoot orders and ramet between January 2015 and May 2016 among six LSG-62 loblolly pine ramets.

^b The F statistics and coefficients of determination were significant at $p < 0.05$ level for all relationships except those between second-year conelets in January 2016 and the three carbohydrate supply variables.

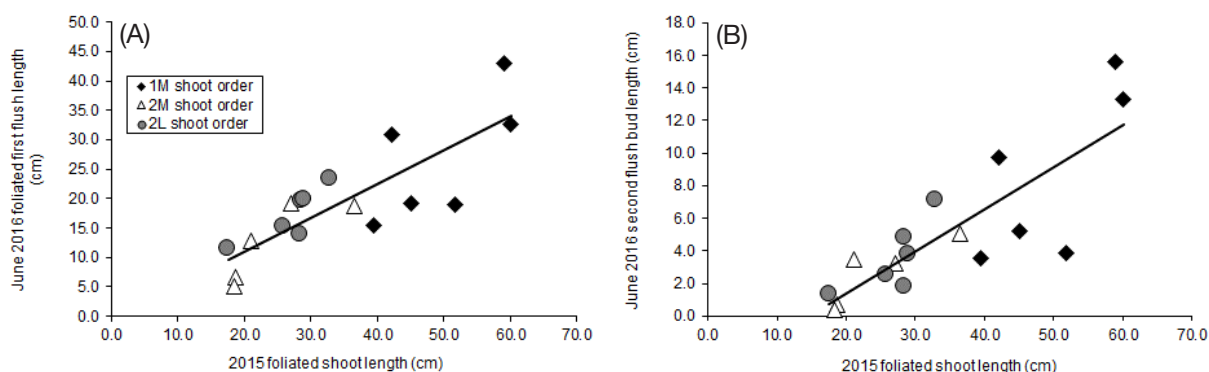


Figure 4—Vegetative carbohydrate sinks evaluated by (A) June 2016 foliated first flush length and (B) June 2016 second flush bud length in relation to final length of the foliated shoot produced in 2015. Values represent means by shoot order and ramet among six ramets evaluated between January 2015 and June 2016.

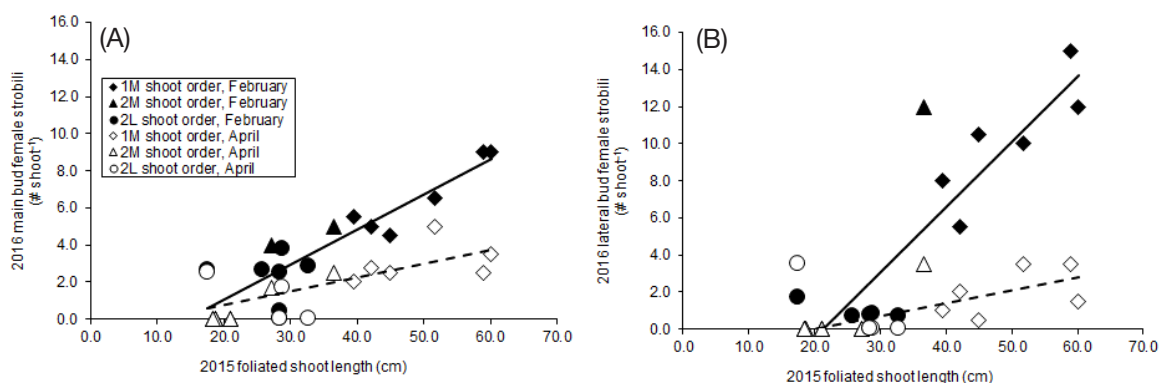


Figure 5—Numbers of female strobili and first-year conelets recorded in February and April 2016, respectively, among (A) main and (B) lateral buds in relation to 2015 final length of the foliated shoot. Values represent means by shoot order and ramet among six ramets evaluated between January 2015 and June 2016.

Pine root system architecture is initiated at germination and continues after planting (Fitter 1996). Past research has shown that vertical and horizontal root system expansion contributes to strong loblolly pine growth on sandy soils (Albaugh and others 2004, Butnor and others 2003, Hacke and others 2000, King and others 1999). Albaugh and others (1998) demonstrated this on a Wakula sand with 94, 84, and 95 percent increases in foliage, stem plus branch, and root system biomass, respectively, in response to annual fertilization between ages 7 and 10 years.

At least two conditions contribute to positive growth responses to fertilization on sandy soils. First, nutritional limitations to growth must be alleviated by fertilization which requires growth to be nutritionally constrained before fertilization. Second, contact between loblolly pine absorbing roots plus mycorrhizae and soil amendments must occur. Absence of these conditions in the present study may have precluded observation of a year 1 fertilization effect on ramet vegetative and reproductive variables. Specifically, in March 2015, the mineral nutrition of first flush fascicles produced in 2014 exceeded dormant season deficiency thresholds (Albaugh and others 2010). Therefore, we did not anticipate a rapid growth response to summer fertilization caused by alleviation of a nutritional deficiency or imbalance. Because the accumulation of leaf area under high light interception controls loblolly pine growth (Fox and others 2006, Vose and Allen 1988), we predicted that summer fertilization would stimulate fascicle development in the overwinter bud leading to an increase in foliage production in spring and summer of the next year, and subsequently, greater annual ramet stemwood growth.

A rogueing operation removed unwanted clones and stumps in orchard 216 in July 2015. We characterized the architecture of these rogued ramet root systems by vertically-oriented lateral roots that accessed water at a depth of 2.5 m. Further observation indicated that rogued root systems did not have an expansive network of lateral roots and mycorrhizae in the upper soil horizons. Rather, first order lateral roots were vertically oriented and had few second or higher order lateral roots emerging from them. Clearly since planting, the rogued ramets had adapted to prolonged periods of dry soil. It is likely that the experimental ramets in the present study have adapted similarly. Therefore, the effect of drip irrigation and fertilization on ramet nutrition and growth may have been limited.

Past research has established the positive relationship between pine reproduction and both crown size and foliage mass (Shoulders 1967, Shoulders 1968). Gains in reproductive potential attributed to foliage production are also demonstrated in our study with significant, positive correlations between female strobili numbers

and variables representative of carbohydrate supply at the time of female strobili primordia differentiation and emergence. It is apparent that carbohydrate imported from mature foliage produced in 2015 also supported first and second flush growth by May 2016. These results emphasize the importance of maximizing the amount of foliage per reproductive shoot in seed orchard ramets. With maturation of the first and second flush of 2016, female strobili primordia differentiation in 2016 and emergence in 2017 are likely to benefit from a greater amount of foliage per shoot.

It became apparent in year 1 that an array of factors had the potential to interfere with the positive relationship between reproductive organ success and carbohydrate supplied by mature foliage. There were, for example, 3 consecutive months of water deficit between July and September 2015. Loblolly pine experiences foliage senescence under prolonged water deficit (Pallardy and others 1995, Tang and others 2004). Because late summer drought and possible acceleration of foliage senescence coincided with normal periods of seed fill and female strobili primordia differentiation (Williams 2008), early foliage senescence may have impacted the carbohydrate available for these reproductive processes. Tang and others (1999) reported a decline in the net photosynthesis of loblolly pine in central Louisiana when air temperatures exceeded 93 °F (34 °C). Therefore, it is also possible that prolonged periods of elevated air temperature in summer reduced the supply of carbohydrate to reproductive sinks such as seeds and female strobili primordia undergoing differentiation. Alternatively, the unique root system architecture of experimental ramets may have accessed deep water and allowed some level of gas exchange to continue in late summer. Under the growth-differentiation-balance hypothesis (Loomis 1932), this scenario would have allowed photosynthate to be allocated to an array of non-growth metabolites.

Distinct differences in amounts of foliage and female strobili by shoot order suggest that shoot size and position in the crown also affect carbohydrate source-sink relationships in reproductive shoots. As such, attention to genetic control of branch number, angle, and distribution in ramet crowns as well as pruning activities that favor fewer but larger reproductive shoots may be warranted. We observed more mature foliage and female strobili and a greater loss of female strobili or first-year conelets between February and April for the 1M shoots compared to 2M and 2L shoots. One factor that may have contributed to these observations is a lower potential for shading of 1M shoots compared to 2M and 2L shoots. This, together with a higher frequency of three flushes among the 1M shoots compared to the 2M and 2L shoots suggests that 1M shoots were better able to support reproductive organs compared to 2M and 2L shoots.

We have compiled environmental information and stemwood growth and foliar nutrition trends among 30 experimental ramets of clone LSG-62 between March 2015 and May 2016. This information provides new knowledge for understanding whole-ramet physiological function at the Evans, LA seed orchard. We have also evaluated a snapshot of the relationships between carbohydrate demand by reproductive organs and the carbohydrate supply by mature foliage among three shoot orders. These observations are providing insight for understanding the importance of current photosynthate and mineral nutrients to female reproductive organs within a single loblolly pine shoot.

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Long-Term Silvicultural Studies

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EXPERIMENTAL FORESTS OF THE SOUTHERN RESEARCH STATION: HIGHLIGHTS OF FOUNDATIONAL SILVICULTURE STUDIES

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Abstract—The U.S. Department of Agriculture Forest Service, Southern Research Station is home to 19 Experimental Forests (EFs) that provide unique opportunities for long-term experimentation and demonstration on forest management practices (fig. 1). Numerous studies on silviculture practices, growth and yield, applied ecology, reforestation, and prescribed fire have originated on EFs and were possible because of partnerships with National Forest System, State and Private Forestry, and other stakeholders. Experimental Forests allow for demonstrations of innovative management practices built on long-term research findings developed in the 20th century to be applied in the 21st. Here, we highlight a selection of foundational studies at EFs in Arkansas, North Carolina, and Louisiana.

Natural Pine Management

Established in 1934, the Crossett EF in southeastern Arkansas conducts research on pine competition control and stand establishment, the rehabilitation of understocked pine stands, and silvicultural practices in natural-origin loblolly (*Pinus taeda*) and shortleaf pine (*P. echinata*). When established, the primary goal of the work at Crossett was to determine if large high-value saw logs could be produced from these cutover, understocked, second-growth southern pine stands (Guldin 2009). Over the years, lessons learned on the Crossett EF have been adapted to help national forests move from intensive plantations toward management that relies on natural regeneration (Guldin 2009). Today, the Crossett EF continues to serve the demonstration and field laboratory needs of researchers, who still follow long-term studies of even- and uneven-aged silviculture, growth and yield, regeneration, and unmanaged stand development in naturally regenerated loblolly and shortleaf pine. In addition, studies of a plus-tree progeny

test (Bragg and others 2016) and pine-hardwood regeneration (Olson and Bragg, in press) have been reactivated, joining new research into the management of mature pine stands for old-growth-like conditions and new tools (eddy covariance tower) in Crossett's ever-growing research portfolio.

Upland Hardwoods

The Henry R. Koen EF is located south of the Buffalo River on the Ozark National Forest. Established in the early 1940s, it was originated to develop principles for forest management. In its first decade, Koen was instrumental in developing best practices for cruising based on accuracy and efficiency. Early studies by Mesavage and Grosenbaugh (1956) determined that plot size and arrangement could be optimized for any given stand. These foundational studies formed the basis for our modern system of hardwood inventory and cruise techniques in upland hardwood forests. Our

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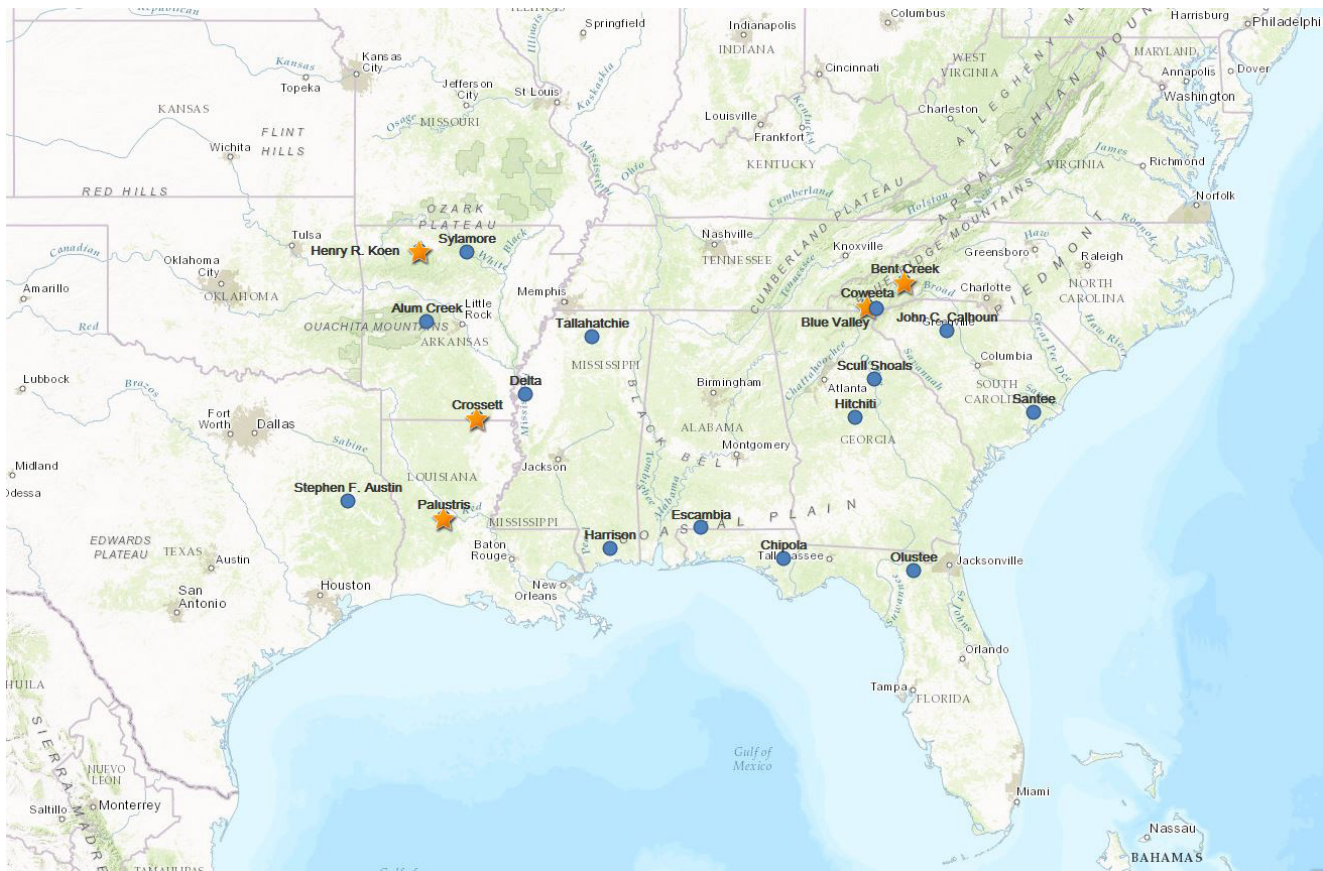


Figure 1—Blue dots mark the location of the nineteen Experimental Forests (EFs) of the Southern Research Station. Yellow stars indicate the locations described in this abstract.

understanding of successional patterns of oaks (*Quercus* spp.), hickory (*Carya* spp.), and shortleaf pine in Ozark uplands comes from the long-term datasets of the Koen. Other research highlights included shortleaf regeneration and seedbed test preparation, stand structure of merchantable white oaks (*Q. alba*), hardwood and pine plantation tests on abandoned agricultural fields, management of eastern red cedar (*Juniperus virginiana*), and loblolly pine suitability in the Ozarks.

Currently, the Koen houses studies that include work on woody species reproduction, stand composition, forest species restoration, quantitative silviculture, and the development of forest management methods. This integrated research program addresses upland hardwood forest dynamics and includes short- and long-term studies of upland hardwood forests at individual tree, stand, landscape, and regional scales.

Bent Creek, the oldest EF in the eastern United States and the second oldest in the country, originated in 1928 on the Pisgah National Forest in western North Carolina. It was established to conduct research on silvicultural practices that would aid in the rehabilitation of cutover, abused lands and promote sustainable forestry, and

to provide a field demonstration of forest management practices. Prior to the 1960s, research consisted of case studies documenting regeneration response to disturbance (primarily clearcutting). Case studies were eventually replaced by replicated, data-intensive studies. At Bent Creek, these focused on species composition and structure, and growth and yield (Loftis 1990). Researchers are currently working on topics including oak restoration, hardwood regeneration, fire ecology, forest stand dynamics, acorn and native forest fruit production, invasive plant species, American chestnut (*Castanea dentata*) restoration, wildlife response to forest management practices, and ecosystem classification.

Forest Structure and Composition

Located in the Southern Appalachians of North Carolina, the Coweeta Hydrologic Laboratory was established in 1934 on the Nantahala National Forest. Early research focused on how natural resource management affected forest hydrology using the paired watershed approach to compare managed and unmanaged forests (Swank and others 2001). At the same time, long-term changes in forest structure and species composition of these deciduous forests were measured at the watershed level

(Elliott and Swank 2008). Silvicultural treatments were implemented to evaluate the management applications of the time including the effects of grazing, exploitive logging, and species conversion on forest watersheds (Elliott and Vose 2011). In addition, Coweeta was instrumental in early studies on road design, clearcutting, cutting methods using cable yarding, partial harvests, and post-disturbance measurements.

Currently, researchers are leading studies that include the effects of wildland fire on ecosystem processes, Eastern hemlock (*Tsuga canadensis*) decline due to the hemlock wooly adelgid (*Adelges tsugae*), measurements of ecosystem carbon dioxide and water flux using an eddy covariance tower, and the continuous monitoring of the long-term climate and streamflow network that has been in place since 1934.

Longleaf Pine

Located in central Louisiana and established in 1935 on the Kisatchie National Forest, the Palustris EF was used for early research on reforestation techniques of the four major southern pine species (Wakeley 1954). Under the umbrella of pine regeneration and management, research included seedling production and quality, stocking and thinning regimes, growth and yield modeling, and site preparation. The Palustris was also heavily involved in pine plantation management with studies on soil quality and management, fire as a management tool, and longleaf pine (*P. palustris*) silviculture.

In 2005, research on the Palustris changed from a pine plantation management focus to one of longleaf pine restoration (Sung and others 2013). Current and future research has been designed to support the Range-wide Conservation Plan for Longleaf Pine (Sword Sayer and others 2010). These topics include: longleaf pine restoration and management, seedling quality and establishment, and flower and cone physiology.

Future of Experimental Forests

As we look to the future, we are working across the EF network to look at the “big” environmental and social issues of the 21st century, using inputs from synthesis work, national assessments, and input from partners to ask questions across multiple scales. As in the 20th century, our forests face new and emerging threats. We can use EFs to develop integrated approaches and strategies to mitigate impacts that will sustain the overall health of forests and grasslands, improve environmental conditions, and reduce economic costs. Experimental forests will continue to be locations with long-term datasets and the ability to sustain experiments and measurements over multi-year periods, places of traditional and novel experiments, valuable

demonstration sites, and sites where monitoring is used for change detection. Experimental Forests can leverage these core attributes and serve as important facilities for collaborative research, partnerships, and platforms for development of new tools and technologies.

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CONTRIBUTION OF SILVICULTURE TO LOBLOLLY PINE GROWTH AND YIELD IN THE SOUTHEASTERN UNITED STATES: A META-ANALYSIS

Héctor I. Restrepo, Bronson P. Bullock, and Cristian R. Montes

Abstract—There has been an increase in loblolly pine production driven by forest management practices like intensive silviculture and improved genetics. Some reported yield gains have been modeled using meta-regression mixed effects models accounting for the potential contribution of the four factors related to forest growth: age, site quality (environment), establishment culture and management, and stand intrinsic characteristics (genetics and initial planting density). The aim of this research was to describe a methodology that allows for the derivation of response equations from yield models for diameter at breast height, stand average height, basal area, and total volume in the Southeastern United States. When compared to low-level silviculture, moderate and intensive silviculture show volume gains at age 20 of 221 and 314 m³/ha, respectively. Likewise, moderate and intensive management consistently performed better over time as compared to low management for all response variables. These management response curves and their associated mathematical expressions can be used to perform financial marginal analyses to improve forest land decision making.

INTRODUCTION

Loblolly pine (*Pinus taeda* L.) is the most commercially important forest species in Southeastern United States. The timber production in this region has been enhanced using genetically improved seedlings and a wide range of silvicultural treatments such as mechanical site preparation, vegetation control, fertilization, and irrigation (Allen and others 2005). These intensive forest management practices together with other key factors like initial planting density, stand age, and environmental conditions result in the expression of forest yield (Clutter and others 1983). Hence, volume gains resulting from intensive management practices can be analyzed by isolating the effect of age, environment, and density using growth and yield models. However, most of the existing models account only for one of the forest growth factors, either management, genetics, or environment. Thus, most of the research on the effect of intensive silviculture on loblolly pine growth and yield is locally restricted and/or density- and genetic composition-dependent. Therefore, those conclusions cannot be easily generalized.

An ideal response generalization would require a large experimental base, covering a wide range of ages, environmental conditions, management practices, and genotypes. The amount of resources involved in such an experimental base may make this kind of research

technically challenging and economically unfeasible. In other research areas (like medicine), scientists have overcome the problem of not having a large experimental base with conclusions drawn out of a large collection of independent studies using meta-analysis. Meta-analysis is a statistical technique utilized to compile information for the purpose of integrating the findings as a rigorous alternative to the traditional narrative discussion (Schwarzer and others 2015).

There is a meta-analysis on forest yield of loblolly pine in the Southeastern United States that accounts for all growth factors (Restrepo and Bullock, in preparation),¹ i.e., effect of age, environment (through physiographic region), genetics, density, and management as explanatory variables for diameter at breast height (DBH), average height (Ht), basal area (BA), and total volume (V). Therefore, those yield models (see footnote 1) can be used to derive silvicultural responses isolating the effect of the remaining factors, which is the purpose of this paper.

¹ Restrepo, H.I.; Bullock, B.P. Manuscript in preparation. Growth and yield of loblolly pine in the Southeastern U.S.: a meta-analysis. On file with: Héctor I. Restrepo, Ph.D. Student, Warnell School of Forestry and Natural Resources, University of Georgia, Athens, GA 30602.

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METHODS

Four-Factor Forest Yield Models

The mean, standard deviation, and number of observations of the response for DBH, Ht, BA, and V were extracted from 22 studies selected out of 500 studies in the Southeastern United States (see footnote 1) in a meta-analysis framework. Included studies constitute a representative sample size of loblolly pine yield (table 1) over a wide range of environmental conditions from 44 counties located in 10 States across the Southeastern United States (fig. 1). The model is termed a four-factor model because it considers covariates from the four factors of forest growth:

- *Age*: age of the stand in years
- *Genetics*: genetically improved categories of loblolly pine [unimproved or unknown (UU), half-sibling (HS), full-sibling (FS), and clone (C)]
- *Mgmt*: management intensity [low (L), moderate (M), and high (H) in quantity and frequency of inputs and applications]
- *Region*: physiographic regions [Upper Coastal Plain and Piedmont together (UCPP), Upper Coastal Plain (UCP), Piedmont (P), and Lower Coastal Plain (LCP)]
- *Density*: surviving density in stems/ha

Forest yield models correspond to the log-transformed Schumacher model (Schumacher 1939) estimated using linear mixed effects models (see footnote 1) (app. A):

$$y_i = \theta + u_i + \epsilon_i \quad (1)$$

$$\epsilon_i \sim N(0, \sigma_i^2); u_i \sim N(0, \tau^2); Cov(\epsilon_i, u_i) = 0$$

where

θ = the fixed effects term

u_i = the random effects term assumed normally distributed with zero mean and variance τ^2

ϵ_i = the error term assumed normally distributed with zero mean and variance σ_i^2 and independent to random effects

An estimator for θ is:

$$\hat{\theta}_{jkl} = \hat{a}_0 + \hat{a}_{1j}(\text{Genetics}) + \hat{a}_{2k}(\text{Mgmt}) + \hat{a}_{3l}(\text{Region}) + \hat{a}_4(\text{Density}) + \frac{1}{\text{Age}} \left[\hat{\beta}_0 + \hat{\beta}_{1j}(\text{Genetics}) + \hat{\beta}_{2k}(\text{Mgmt}) + \hat{\beta}_{3l}(\text{Region}) + \hat{\beta}_4(\text{Density}) \right] \quad (2)$$

where

$\hat{\theta}_{jkl}$ = an estimator of the fixed effects of DBH, Ht, BA, or V in logarithmic units of the genetics j (HS=1, FS=2, C=3) with management regime k (M=1, H=2) in the physiographic region l (UCP=1, P=2, LCP=3)

α (asymptote) = parameter estimate

β (slope) = parameter estimate

Yield model for V did not consider the effect of *Genetics*(FS) and *Genetics*(C) due to the lack of observations of those levels of genetics.

Table 1—Number of observations (combination of treatments), number of measurement plots, and total area of measurements of the selected studies in the meta-analysis of loblolly pine growth and yield in the Southeastern United States (Restrepo and Bullock, in preparation; see footnote 1)

Response variable	No. treatments	No. plots	Area (ha)
DBH	105	1288	79
Ht	97	1344	81
BA	176	1476	70
V	128	1492	259

Source: Restrepo, H.I.; Bullock, B.P. Manuscript in preparation. Growth and yield of loblolly pine in the Southeastern U.S.: a meta-analysis. On file with: Héctor I. Restrepo, Ph.D. Student, Warnell School of Forestry and Natural Resources, University of Georgia, Athens, GA 30602.

DBH = diameter at breast height; Ht = total height; BA = basal area; V = volume.

Silvicultural Responses

Responses for DBH, Ht, BA, and V associated with moderate and high levels of management were derived from the yield model with respect to the low level of management:

$$\begin{aligned}\frac{\partial \hat{\theta}_{jkl}}{\partial Mgmt(M)} &= \left(\hat{a}_{21} + \frac{\hat{\beta}_{21}}{Age} \right) \hat{\theta} \\ \frac{\partial \hat{\theta}_{jkl}}{\partial Mgmt(H)} &= \left(\hat{a}_{22} + \frac{\hat{\beta}_{22}}{Age} \right) \hat{\theta}\end{aligned}\quad (3)$$

These partial derivatives with respect to the low level of management were fixed to HS and UCPP levels of genetics and physiographic region, respectively, and the surviving density based on an arbitrary planting density of 1,500 trees/ha was estimated using the following equation (PMRC 2002):

$$\widehat{Density} = 2.5 + (2.5 - 1500)(1 + 0.68 Age)^{1.46} (1 + Age)^{-135} \exp[-5.9 \times 10^{-4} Age^2] \quad (4)$$

RESULTS AND DISCUSSION

Responses for DBH, BA, Ht, and V were consistently ranked over time from *Mgmt(M)* to *Mgmt(H)* (fig. 2). Thus, at age 20 *Mgmt(M)* added 3.6 cm, 3.3 m, 16 m²/ha, and 221 m³/ha of DBH, Ht, BA, and V, respectively, with respect to *Mgmt(L)*; whereas, at the same age, *Mgmt(H)* added 8.3 cm, 4.5 m, 27 m²/ha, and 314 m³/ha to the corresponding variables with respect to *Mgmt(L)*. Basal area response curves are flat up to age 5 when response curves started exhibiting a linear-looking trend

up to age 20. Partial derivatives of the yield models with respect to *Mgmt(M)* and *Mgmt(H)* are presented in appendix B.

Overall, these responses are consistent with the expected management outcomes. In general, the higher the inputs and the frequency of the applications, the higher the resulting stand growth and yield (Albaugh and others 2004, Aspinwall and others 2011, Borders and others 2004, Roth and others 2007). Moreover, *Mgmt(M)* and *Mgmt(H)* are additive terms to a basic yield curve [*Mgmt(L)*], in a similar way that Pienaar and Rheney (1995) modeled silvicultural treatments.

High levels of inputs in quantity and frequency may adjust to asymptotic response curves, whereas low levels of management exhibit parabolic-looking curves (Snowdon 2002). Thus, there is a possibility that high-order terms of *Mgmt* or interactions such as *Mgmt x Genetics* and/or *Mgmt x Region* were missing in the yield models. The use of first-order terms in the model, as a way to simplify the number of inputs, may cause the management response curves for moderate management to not exhibit a parabolic form and rather attain a peak and then decrease. Despite this mathematical limitation, these management responses give insight into the size of the effect of the three simple levels of management considered here. Hence, economic tradeoffs of operational and intensive forest management can be analyzed. Likewise, since yield models account for the effect of genetics, environment, and density, the model and their derived responses can be also utilized to analyze the effect of a combination of factors.

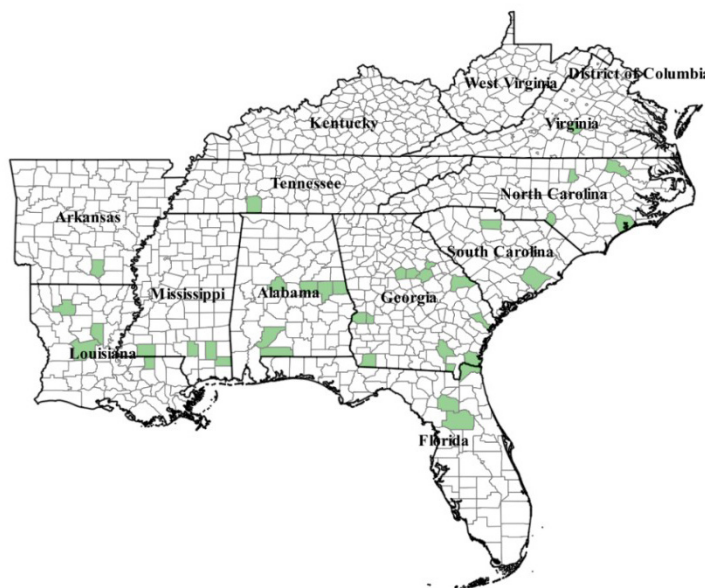


Figure 1—Southeastern U.S. counties in which studies have been conducted that were considered in this research.

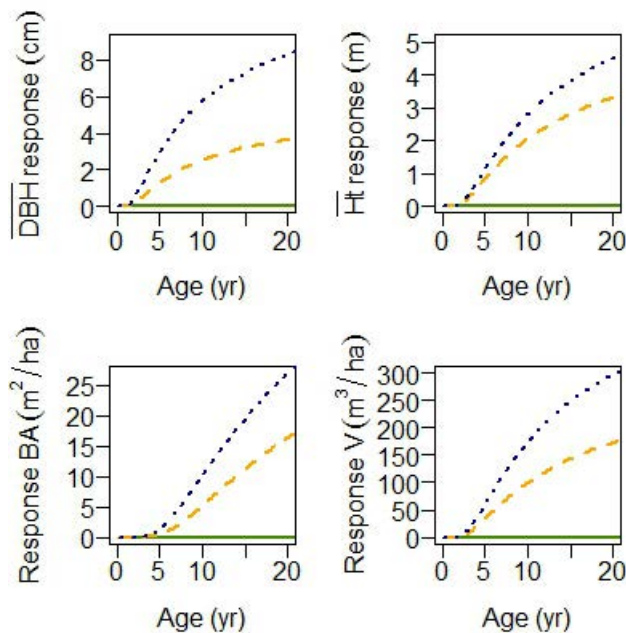


Figure 2—Loblolly pine silvicultural responses relative to low level of management [*Mgmt(L)*] in the Southeastern United States for diameter at breast height (DBH), total height (Ht), basal area (BA), and volume (V) keeping the genetics fixed as half-siblings planted and the physiographic region as Upper Coastal Plain – Piedmont. Dashed line represents the response of moderate management [*Mgmt(M)*], and dotted line represents the response of high management [*Mgmt(H)*].

CONCLUSIONS

Forest growth factors have successfully explained loblolly pine yield (see footnote 1). In those models, moderate and high levels of management were statistically different (superior) to the low level of management. Using this information, partial derivatives were taken to analyze silvicultural response equations. Volume at age 20 for moderate and high levels of management can be as much as 221 and 314 m³/ha higher, respectively, as compared to the low level of management. Hence, yield models that consider the four factors of growth can be used to derive silvicultural responses isolating the effect of genetics, environment, and density. The same framework can be applied to determine a potential volume increase associated with

genetically improved seedlings and differences in yield associated to the environment (physiographic region). This model could be used to perform a financial marginal analysis, characterizing the cost associated with the levels of management regimes and determining the profitability associated with each level.

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APPENDIX A

Summary of loblolly pine yield models for the Southeastern United States using meta-regression (Restrepo and Bullock, in preparation; see footnote 1)

Source	Estimates (log) and significance level ^a							
	DBH		Ht		BA		V	
Intercept	3.67	***	3.46	***	3.48	***	5.67	***
<i>Genetics</i> (HS)					0.29	***		
<i>Genetics</i> (FS)	0.22	***	0.19	***	0.42	***		
<i>Genetics</i> (C)	0.21	**	0.28	*				
<i>Mgmt</i> (M)	0.15	***	0.16	**	0.39	***	0.67	***
<i>Mgmt</i> (H)	0.29	***	0.21	***	0.49	***	0.82	***
<i>Region</i> (UCP)					0.72	***		
<i>Region</i> (LCP)	-0.20	***	-0.71	***				
<i>Density</i>	-3x10 ⁻⁴	***	-1.6x10 ⁻⁴	***	-1.7x10 ⁻⁴	*		
1/ <i>Age</i>	-7.36	***	-9.62	***	-11.53	***	-10.73	***
<i>Genetics</i> (HS):1/ <i>Age</i>								
<i>Mgmt</i> (H):1/ <i>Age</i>					1.31	***		
<i>Region</i> (UCP):1/ <i>Age</i>					-11.52	***		
<i>Region</i> (LCP):1/ <i>Age</i>	1.70	***	4.83	***				
<i>Density</i> :1/ <i>Age</i>	6.4x10 ⁻⁴	***	4.8x10 ⁻⁴	*	1.1x10 ⁻³	***		

^a Significance codes: *** = *p*-value <0.0001; ** = *p*-value <0.001; * = *p*-value <0.01; . = *p*-value <0.05; blank = *p*-value <0.1.

Source: Restrepo, H.I.; Bullock, B.P. Manuscript in preparation. Growth and yield of loblolly pine in the Southeastern U.S.: a meta-analysis. On file with: Héctor I. Restrepo, Ph.D. Student, Warnell School of Forestry and Natural Resources, University of Georgia, Athens, GA 30602.

APPENDIX B

Partial derivatives of the models are:

$$\begin{aligned}
 \frac{\partial DBH}{\partial Mgmt(M)} &= 0.15 \exp \left[3.67 + 0.15 - 3 \times 10^{-4} \text{Density} + \frac{1}{\text{Age}} (-7.36 + 6.4 \times 10^{-4} \text{Density}) \right] \\
 \frac{\partial DBH}{\partial Mgmt(H)} &= 0.29 \exp \left[3.67 + 0.29 - 3 \times 10^{-4} \text{Density} + \frac{1}{\text{Age}} (-7.36 + 6.4 \times 10^{-4} \text{Density}) \right] \\
 \frac{\partial Ht}{\partial Mgmt(M)} &= 0.16 \exp \left[3.46 + 0.16 - 1.6 \times 10^{-4} \text{Density} + \frac{1}{\text{Age}} (-9.62 + 4.8 \times 10^{-4} \text{Density}) \right] \\
 \frac{\partial Ht}{\partial Mgmt(H)} &= 0.21 \exp \left[3.46 + 0.21 - 1.6 \times 10^{-4} \text{Density} + \frac{1}{\text{Age}} (-9.62 + 4.8 \times 10^{-4} \text{Density}) \right] \\
 \frac{\partial BA}{\partial Mgmt(M)} &= 0.39 \exp \left[3.48 + 0.29 + 0.39 + 0.72 - 1.7 \times 10^{-4} \text{Density} + \frac{1}{\text{Age}} (-11.53 - 11.52 + 1.1 \times 10^{-3} \text{Density}) \right] \\
 \frac{\partial BA}{\partial Mgmt(H)} &= \left(0.49 + \frac{1.31}{\text{Age}} \right) \exp \left[3.48 + 0.29 + 0.49 + 0.72 - 1.7 \times 10^{-4} \text{Density} + \frac{1}{\text{Age}} (-11.53 + 1.31 - 11.52 + 1.1 \times 10^{-3} \text{Density}) \right] \\
 \frac{\partial V}{\partial Mgmt(M)} &= 0.67 \exp \left[5.67 + 0.67 + \frac{1}{\text{Age}} (-10.73) \right] \\
 \frac{\partial V}{\partial Mgmt(H)} &= 0.82 \exp \left[5.67 + 0.82 + \frac{1}{\text{Age}} (-10.73) \right]
 \end{aligned}$$

SUPERIOR PINES REVISITED: A PLUS-TREE PROGENY TEST ON THE CROSSETT EXPERIMENTAL FOREST AT A HALF-CENTURY

Don C. Bragg

Abstract—Between 1966 and 1969, Forest Service Plant Geneticist Hoy Grigsby installed the last of his tree improvement studies on the Crossett Experimental Forest (CEF). This research, a series of plus-tree progeny tests of full- and half-sib loblolly pine (*Pinus taeda*), was installed to compare the survival, form, vigor, fusiform rust resistance, and height growth of these families (and CEF “woods-run” sources). Due to a variety of reasons, this study was discontinued in the 1970s. Recently, we have started to investigate the remaining trees in this now 48- to 51-year-old superior pine progeny test. This initial report summarizes what is known about the 1969 outplanting and considers what may be possible for future research. Although past thinnings mean this research can no longer document survivorship or fusiform response, valuable information on growth and yield can still be extracted. For example, comparisons of “winners” and “losers” based on 3-year height outcomes are not consistent with those noted on a more limited sample after 48 years of growth.

INTRODUCTION

Recent reviews have explored the dramatic successes—and some notable failures—of forest genetics and tree improvement programs in loblolly pine (*Pinus taeda*) across the Southern United States (for example, Allen and others 2005, Borders and Bailey 2001). From humble beginnings with “common garden” experiments using field-collected pine seed (Wakeley and Bercaw 1965) to outplantings of mass-controlled pollinations and cloned seedlings, plantations of improved loblolly pine have become the commercial foundation of the most productive timber region in the world (Allen and others 2005). As these programs matured, industry has pursued a research and development trajectory focusing on the selection of loblolly pine for rapid growth, crown ideotypes, and disease resistance using a relatively limited number of genetically improved families.

Over the years, progeny tests have proved invaluable in this process. However, few published results on southern pine progeny/provenance tests older than 30 years can be found in the literature. Examples of longer-term studies include Wakeley and Bercaw (1965), who reported on 35-year results of a common garden study of four geographic sources of loblolly pine planted in southern Louisiana; Wells and Rink (1984), who described loblolly performance in a different provenance test in southern Illinois after 35 years; and Rink and Wells (1988), who compared loblolly pine seed sources with shortleaf pine (*P. echinata*) in southern Illinois 37 years after planting. Loblolly pine in the Southwide

Pine Seed Source Study (SPSSS) (Wells and Wakeley 1966) was measured to age 25 years (Wells 1983). By the late 1980s, very few of the original SPSSS locations remained (Buford 1989), although Schmidting and Froehlich (1993) did report on 37-year results for the Maryland location.

Given that intensively managed loblolly pine stands are rarely grown more than 25 years, knowing the long-term (30+ year) outcomes of progeny tests has not been a priority—especially given the likelihood of new and even more improved families becoming available during that time span. Today, loblolly pine families are usually evaluated using short-term assessments of performance (for example, Farjat and others 2016). Research has shown that the growth performance of different families can be effectively evaluated within the first few years after planting, with early success being maintained through at least mid-rotation (Bridgwater and McKeand 1997, McKeand 1988). However, there are valid questions that cannot be addressed with short-term experiments. For instance, what are the long-term consequences of culling poor early-performing families that may actually be good performers (“Type A” of Bridgwater and McKeand 1997)? What happens if indirect selection results in the choosing of families that ultimately prove unsuccessful under longer rotations (Martin and others 2001)? Could landowners interested in the long-term performance of improved loblolly pine for certain forest products (such as poles or pilings, or high specific gravity wood) or

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non-commodity ecosystem services (such as carbon sequestration) desire families not selected for rapid, early-volume growth?

Since many of the aforementioned longer-term studies yielded interesting and sometimes unexpected results decades after their establishment, it is worth revisiting old progeny tests. Hence, the objective of this limited study is to evaluate the potential of a half-century-old loblolly pine progeny test on the Crossett Experimental Forest (CEF) in southern Arkansas to inform silvicultural researchers about the long-term performance of second-generation, improved loblolly pine from locally derived, superior pines.

METHODS

From 1951 until 1975, a number of tree improvement and forest genetics studies were conducted by staff from the CEF in southern Arkansas (Bragg and others 2016). Between 1966 and 1969, Forest Service Plant Geneticist Hoy Grigsby installed part of a superior pine progeny test in the eastern half of Compartment 3 on the CEF (fig 1). Grigsby chose this approximately 20-acre site because it was relatively level, without any significant drainages, and was dominated by soils common to this portion of the Upper West Gulf Coastal Plain (Bude and Providence silt loams, with a nominal loblolly pine site index of 85 to 90 feet at 50 years; Gill and others 1979). When installed, the existing pine-dominated timber was cleared and the site prepared prior to the first outplantings in February of 1966.

The full- and half-sib seeds came from superior pines identified by trained foresters on the lands of Georgia-Pacific in Ashley and Drew Counties in Arkansas and Morehouse Parish in Louisiana during the preceding decade; open-pollinated seeds (“woods-run”) were collected from the general population of loblolly pines on the CEF (Grigsby 1967–1969). Pine seeds were germinated and raised to 1-year-old seedlings before being planted (bare-root) on 8-foot by 8-foot spacing in a series of replicated blocks (Grigsby 1967–1969). Most seedlings were pure loblolly pine crosses—the few loblolly x shortleaf hybrids included in the 1967 outplantings are not included in this analysis. The Compartment 3 outplantings were established to determine the influence of family on a number of traits, including survival, vigor, growth, form, wood specific gravity, and fusiform rust (*Cronartium quercuum* f. sp. *fusiforme*) susceptibility. The retirement of Project Leader Russ Reynolds in 1969 led to Forest Service’s closing of the CEF, and Grigsby was reassigned to the genetics unit in Saucier, MS (Bragg and others 2016). After Grigsby’s retirement a few years later, Forest Service Plant Geneticist Warren Nance assumed responsibilities for the remaining CEF genetics projects. Shortly thereafter, Nance decided there was insufficient value in continuing the CEF tree improvement studies and formally closed

the projects (Nance 1978). In the decades since, the progeny tests in Compartment 3 have been operationally thinned and salvaged several times but remain largely intact.

This paper focuses on the last outplanting of 28 families (22 full-sibs + 6 half-sibs) produced by crossing superior pines and CEF woods-run pines (table 1). This outplanting (shaded area on fig. 1) was installed in February 1969 using five blocks, each of which contained twenty-nine 48-foot by 48-foot plots with 36 planting points, for a grand total of 5,220 pine seedlings (Grigsby 1967–1969). The original data used for this paper were adapted from a spreadsheet of block-level plot means for the 1969 outplantings assembled by family contained in the study file (dated February 16, 1972). Block-level plot means were treated as replicates; since there were 5 blocks, $n = 5$ for each family for survival percentage, total height (in feet), and fusiform rust occurrence (percent of seedlings showing signs of fusiform). No statistical analysis had been done in 1972, so I conducted an evaluation for some initial perspective. Because these data were not normally distributed and two of the variables of interest were measured in terms of percentages, a Kruskal-Wallis test was conducted to determine if significant ($\alpha = 0.05$) differences among family rankings were present. Post-hoc nonparametric multiple comparisons were also performed to determine which families performed the best (Zar 2010). For his brief unpublished closing report, Nance (1978, his figures 26–28) made a limited statistical assessment when the 1969 outplanted pines were 5 years old.

In January of 2017, a 36-plot subset of the 1969 outplanting was remeasured (Figure 1). This subsection contained at least one block of all families and each live pine in each plot was measured for its diameter at breast height (d.b.h.; to the nearest 0.1 inch) using a diameter tape and total tree height (to the nearest 0.5 foot) using the sine method of height determination (Bragg and others 2011) and a TruPulse 200X laser hypsometer. In addition, merchantable inside-bark volume (V , in cubic feet) was calculated using a local CEF volume equation (Farrar and others 1984):

$$V_{CEF} = -1.41726 - 0.02484 \text{ d.b.h.} + 0.09948 \text{ d.b.h.}^2 + 0.00748 \text{ d.b.h.}^3 + 0.00017 \text{ d.b.h.}^4 \quad [1]$$

and a regional volume equation (Van Deusen and others 1981):

$$V_{VD} = -0.00296 + 0.00193881 \text{ d.b.h.}^2 \text{ HT} \times R \quad [2]$$

where

HT = total tree height

R = a top-diameter conversion ratio.

Table 1—Family labels by Georgia-Pacific district codes^a and superior pine number for the families tested in the 1969 outplanting in Compartment 3 of the Crossett Experimental Forest

----- Full-sibs (F) -----				----- Half-sibs (H) -----		-- Open-pollinated (W) --	
Family	Parent x Parent	Family	Parent x Parent	Family	Parent x wind	Family	wind x wind
F1	BE-12 x YA-01	F12	BE-11 x BE-12	H23	BE-12 x wind	W29	CEF woods-run
F2	BE-12 x CR-04	F13	BE-11 x YA-01	H24	YA-01 x wind		
F3	BE-12 x EA-21	F14	BE-11 x CR-04	H25	CR-04 x wind		
F4	BE-12 x BL-05	F15	BL-05 x YA-01	H26	BL-05 x wind		
F5	CR-14 x BE-12	F16	YA-01 x BE-12	H27	CR-14 x wind		
F6	CR-14 x YA-01	F17	YA-01 x CR-04	H28	BE-11 x wind		
F7	CR-14 x EA-22	F18	YA-01 x BL-05				
F8	CR-14 x BL-42	F19	CR-04 x BE-11				
F9	CR-14 x CR-04	F20	CR-04 x BE-12				
F10	BE-11 x BL-05	F21	CR-04 x CR-14				
F11	BE-11 x EA-21	F22	CR-04 x YA-01				

^a Georgia-Pacific district codes: BE = Berea; BL = Berlin; CR = Crossett; EA = East; YA = Yale Camp.

D.b.h., total tree height, and merchantable volume were then compared between the Crossett woods-run seedlings (Family 29; hereafter, W29) and all other families using one-way ANOVA; Tukey's HSD test for unequal sample sizes ($\alpha = 0.05$) was used to separate means.

RESULTS

1970s Data Reanalysis

Unpublished long-term records from the CEF showed a wet year in 1968 (just over 76 inches of precipitation; the CEF averages about 55.5 inches annually) and slightly drier than average years in 1969 (49.0 inches) and 1970 (53.0 inches), followed by a major drought in 1971 (38.9 inches) and an average year in 1972 (55.8 inches). Not surprisingly, mortality following 3 years post-planting was low, with only two families (H25 and F8) experiencing significantly lower—less than 85 percent—survival (table 2). All but three full-sib families had at least 92 percent survival; all of the half-sib families had between 83 and 89 percent survival, and W29 had an average survival of 92.8 percent. Five-year survivorship reported in Nance (1978) was virtually unchanged for all families.

The considerable variation in height within families resulted in many of them being statistically indistinguishable from each other after 3 years (table 2). On average, the tallest family was F15 at 6.6 feet; families F2 and F6 also averaged at least 6.0 feet tall. Most families had good (5.0 to 5.9 feet) to fair (4.0 to 4.9 feet) height performance after 3 years, and only two (F11 and F12) proved poor (less than 4.0 feet) (tables 2 and 3). At 5.2 feet, W29 was on the lower end of good, but only one family (F15) was significantly taller than W29 after

3 years, and only F11 (at 3.8 feet) was shorter (table 2). Nance (1978, his table 27) reported similar findings; his analysis of height performance at 5 years post-planting noted only five families had exceeded W29 by 5 percent or more and some families averaged more than 25 percent shorter than W29.

In 1972, fusiform rust occurrence was universally low, with only a handful of individual plots at about 6 percent infection, and only a single family (F8) averaging more than 2 percent infected (table 2). It is likely that the higher fusiform infection rate in F8 contributed to this family having the lowest survival rate (54.4 percent) after three growing seasons. Along with 16 other families, W29 showed no evidence of fusiform infection when checked in February of 1972. While it remained relatively modest, the rate of fusiform rust infection increased in all families when evaluated by Nance (1978). Because of this still low rate, Nance (1978) paid little attention to family-based differences; however, his Figure 26 indicated most families had a somewhat more fusiform than W29.

In summary, a reanalysis of the 1972 plot-level data (table 3) show that most tested families had good to excellent survivorship, fair to good height growth, and low to very low fusiform infection rates at 3 years. W29 seedlings performed well after 3 years in the field. Compared to the other 28 families, W29 seedlings had slightly above average survivorship, total height, and lack of fusiform occurrence. Using the performance categories given in table 3, only one family (F6) fell into the excellent category in all three measures of success (survivorship, total height, and fusiform rate). Families F15 and F2 were found in two of three categories,

Table 2—Statistics for the 1969 outplantings in Compartment 3 of the Crossett Experimental Forest, measured in 1972 and summarized by family (using blocks as replicates)

Family	----- Survival -----				----- Total height -----				Fusiform rust occurrence			
	Min.	Max.	Avg. ^a	SD	Min.	Max.	Avg. ^a	SD	Min.	Max.	Avg. ^a	SD
	----- % -----				----- feet -----				----- % -----			
F1	94.4	100.0	98.9 a	2.5	5.2	6.8	5.9 ab	0.6	0.0	0.0	0.0 a	0.0
F2	94.4	100.0	96.7 a	2.3	5.1	7.3	6.0 ab	0.9	0.0	5.9	1.7 a	2.6
F3	88.9	100.0	96.7 a	4.6	5.4	5.9	5.6 abcd	0.2	0.0	2.9	0.6 a	1.3
F4	91.7	100.0	96.7 a	3.6	4.5	6.4	5.3 abcd	0.7	0.0	2.8	0.6 a	1.2
F5	94.4	100.0	97.2 a	2.0	5.0	6.6	5.8 abc	0.6	0.0	2.9	0.6 a	1.3
F6	94.4	100.0	97.2 a	2.8	5.1	7.2	6.0 ab	0.9	0.0	0.0	0.0 a	0.0
F7	61.1	100.0	88.3 a	16.2	4.3	5.5	4.9 bcdef	0.5	0.0	0.0	0.0 a	0.0
F8	25.0	100.0	54.4 b	28.1	4.0	5.9	5.2 bcde	0.8	0.0	6.3	2.2 a	3.1
F9	83.3	100.0	94.4 a	6.5	3.5	5.6	4.8 bcdef	0.8	0.0	0.0	0.0 a	0.0
F10	94.4	100.0	98.3 a	2.5	4.6	5.7	5.2 bcd	0.5	0.0	0.0	0.0 a	0.0
F11	80.6	97.2	87.2 a	7.2	3.3	4.3	3.8 f	0.4	0.0	0.0	0.0 a	0.0
F12	88.9	97.2	93.9 a	3.6	3.3	4.7	3.9 ef	0.6	0.0	0.0	0.0 a	0.0
F13	91.7	100.0	96.7 a	4.6	3.8	4.9	4.5 cdef	0.4	0.0	3.0	1.2 a	1.6
F14	91.7	100.0	96.7 a	3.6	3.8	5.0	4.5 cdef	0.5	0.0	0.0	0.0 a	0.0
F15	88.9	100.0	96.1 a	4.2	5.9	7.2	6.6 a	0.5	0.0	3.1	0.6 a	1.4
F16	94.4	100.0	97.2 a	2.0	5.2	6.6	5.7 abc	0.6	0.0	2.9	1.1 a	1.5
F17	80.6	100.0	94.4 a	7.9	4.4	5.4	5.0 bcdef	0.4	0.0	2.9	0.6 a	1.3
F18	91.7	100.0	94.4 a	3.4	4.4	5.6	4.8 bcdef	0.5	0.0	0.0	0.0 a	0.0
F19	86.1	100.0	92.8 a	5.8	3.7	5.1	4.3 def	0.5	0.0	5.7	1.1 a	2.6
F20	91.7	97.2	93.3 a	2.5	4.4	5.5	4.8 bcdef	0.4	0.0	0.0	0.0 a	0.0
F21	94.4	97.2	96.7 a	1.2	4.0	5.5	4.6 cdef	0.6	0.0	0.0	0.0 a	0.0
F22	91.7	100.0	97.2 a	3.4	4.6	5.7	5.1 bcdef	0.4	0.0	0.0	0.0 a	0.0
H23	80.6	94.4	87.8 a	5.0	3.9	5.2	4.6 cdef	0.5	0.0	0.0	0.0 a	0.0
H24	75.0	97.2	88.3 a	8.2	3.8	4.9	4.3 def	0.4	0.0	0.0	0.0 a	0.0
H25	80.6	88.9	83.9 b	3.0	4.4	5.0	4.7 bcdef	0.2	0.0	0.0	0.0 a	0.0
H26	75.0	97.2	88.9 a	9.4	4.1	5.4	4.6 cdef	0.5	0.0	0.0	0.0 a	0.0
H27	80.6	94.4	88.3 a	6.0	4.5	5.7	5.0 bcdef	0.5	0.0	3.2	0.6 a	1.4
H28	83.3	94.4	88.9 a	4.4	4.5	5.3	4.9 bcdef	0.3	0.0	3.3	1.3 a	1.7
W29	88.9	97.2	92.8 a	4.2	4.8	5.8	5.2 bcde	0.4	0.0	0.0	0.0 a	0.0

^a Averages with the same letters are not significantly different at $\alpha = 0.05$ (Kruskal-Wallis test; non-parametric multiple comparison).

Table 3—Relative ranking of family performance in 1972 using arbitrary categories (the Crossett woods-run family W29 is highlighted in bold)

Performance category	----- Survivorship (%) -----		----- Total height (feet) -----		----- Fusiform (%) -----	
	Range	Families	Range	Families	Range	Families
Excellent	≥95%	F1, F10, F22, F5, F6, F16, F13, F3, F4, F14, F21, F2, F15	≥ 6.0	F15, F2, F6	0.0	F1, F6, F7, F9, F10, F11, F12, F14, F18, F20, F21, F22, H23, H24, H25, H26, W29
Good	90–95%	F18, F17, F9, F12, F20, F19, W29	5.0–6.0	F1, F5, F16, F3, F4, F10, F8, W29 , F22, H27, F17	0.0–0.7	F4, F3, F5, F17, F15, H27
Fair	85–90%	H28, H26, H27, H24, F7, H23, F11	4.0–5.0	H28, F7, F18, F20, F9, H25, F21, H23, H26, F14, F13, F19, H24	0.7–1.4	F16, F19, F13, H28
Poor	<85%	H25, F8	<4.0	F12, F11	>1.4	F2, F8

and most families (including W29) were in at least one excellent category and usually multiple good ratings (table 3). Indeed, in 1972, only F8 seemed to be poorly suited for the CEF location, receiving two poor ratings and one good rating.

Intervening Thinnings and Their Consequences

Although the CEF records are unclear about details, Compartment 3 (including all or part of the progeny tests) were thinned in 1985, 1996, and 2002. The first thinnings in the 1966–1968 plots were implemented in a prearranged pattern; however, it does not appear that the 1969 outplanting was cut in such a fashion, and later thinnings of all outplantings were conducted operationally to improve growth performance. In addition, over the last 40 years, sporadic salvage following other mortality events (for example, lightning, wind, ice, insects, fire) removed some of the planted pines. The thinnings and salvage removals greatly limit any modern-day interpretations of survivorship and fusiform occurrence because they purposefully removed smaller, damaged, and/or diseased individuals.

2017 Data Analysis

Of the 1,296 pines originally planted in this 36-plot subset, 154 remained in January of 2017 (table 4). Although the sample is limited and should be interpreted cautiously, measurements taken in 2017 show changes in performance of the families over time. After 48 years of growth, only one family (H26, averaging 87.9 feet tall)

proved significantly shorter than the four tallest families (F3, F6, F11, F16; these tallest families all averaged at least 96.8 feet) (table 4). While the shortest specimen in H26 had a badly ice-mangled crown (after storms in the 1990s), other less damaged individuals in this half-sib family were not particularly tall, either. By 2017, the Crossett woods-run stock (W29) was on the lower end of height performance, averaging 91.5 feet in total height (table 4), or just about a half-log shorter than the tallest family (F6, at 99.1 feet tall on average). The family with the largest average d.b.h. (F20, at 20.6 inches) was significantly larger than several other families, especially the smallest (F13, 14.2 inches). At 17.2 inches in average d.b.h., W29 remained in the middle of the pack, and was not significantly larger or smaller than any other family (table 4).

Because of the models used to calculate this measure, total inside bark merchantable volume patterns mirrored the trends in d.b.h. According to the CEF (equation [1]) and Van Deusen and others (equation [2]) models, the largest merchantable volumes corresponded most closely with the trees of largest girth (not height). Hence, F20 produced the greatest estimated average merchantable tree volume (over 75 cubic feet), and F13 produced the smallest (less than 36 cubic feet). W29 again fell in the middle, with an average total inside bark merchantable volume of about 51 to 52 cubic feet, and was not significantly different than the other families. Note that for W29, equations [1] and [2] differed only

Table 4—Performance statistics 48 years after planting (January 2017) for the 36-plot subsample of the 1969 outplanting in Compartment 3 of the Crossett Experimental Forest by family

		--- Diameter at breast height ---				----- Total height -----				Total inside bark merchantable volume	
Family	n	Min.	Max.	Average ^a	Std. dev.	Min.	Max.	Average ^a	Std. dev.	$V_{CEF}^{a, b}$	$V_{VD}^{a, b}$
		----- inches -----				----- feet -----				----- cubic feet -----	
F1	9	14.1	19.0	16.9 abcd	1.8	85.5	99.0	95.4 ab	4.6	48.9 abcd	52.2 abcd
F2	5	17.4	20.1	19.0 abc	1.1	89.0	101.5	96.3 ab	4.6	63.8 abc	67.4 abc
F3	5	17.3	20.0	19.1 abc	1.1	94.0	101.0	97.0 a	2.5	64.0 abc	68.1 abc
F4	8	15.0	21.6	17.7 abc	2.1	90.0	104.0	95.4 ab	4.8	54.2 abcd	57.3 abcd
F5	5	16.0	18.7	17.2 abcd	1.1	86.5	99.0	94.2 ab	6.2	50.9 abcd	53.6 abcd
F6	4	13.8	17.1	16.0 bcd	1.6	93.5	104.0	99.1 a	5.4	43.6 bcd	49.0 bcd
F7	5	16.1	19.7	17.5 abcd	1.4	88.5	99.5	94.2 ab	4.0	53.2 abcd	55.7 abcd
F8	3	17.3	20.0	18.6 abcd	1.4	95.0	97.0	95.7 ab	1.2	60.8 abcd	64.0 abcd
F9	4	17.8	22.2	19.9 ab	2.1	94.0	96.0	95.3 ab	1.0	69.8 ab	72.6 ab
F10	4	17.6	20.8	19.2 abc	1.5	88.0	93.5	91.8 ab	2.6	64.7 abc	65.1 abc
F11	5	15.0	16.8	16.1 bcd	0.8	93.0	101.0	96.8 a	3.6	44.2 bcd	48.5 bcd
F12	5	12.9	18.7	16.9 abcd	2.4	88.0	96.0	91.2 ab	3.1	49.3 abcd	50.1 abcd
F13	6	12.6	16.8	14.2 d	1.7	85.0	100.0	93.3 ab	5.2	32.5 d	35.7 d
F14	4	15.7	18.6	16.9 abcd	1.3	94.0	97.0	95.5 ab	1.3	48.8 abcd	52.3 abcd
F15	8	14.4	18.1	16.8 abcd	1.4	94.0	98.5	96.8 ab	1.5	48.3 bcd	52.4 abcd
F16	5	14.1	20.0	16.8 abcd	2.2	96.0	100.0	97.4 a	1.9	48.5 abcd	52.9 abcd
F17	9	15.1	19.4	16.9 abcd	1.3	91.0	102.0	96.2 ab	3.0	49.3 abcd	53.1 abcd
F18	4	17.2	18.9	18.0 abcd	0.9	94.0	100.0	95.8 ab	2.8	56.7 abcd	60.1 abcd
F19	10	14.2	19.9	16.8 abcd	1.9	87.0	97.0	92.2 ab	3.1	48.3 bcd	49.9 bcd
F20	4	19.3	21.1	20.6 a	0.9	93.5	96.0	94.5 ab	1.1	75.6 a	77.8 a
F21	5	14.9	18.0	16.3 bcd	1.3	89.5	95.5	92.1 ab	2.2	45.6 bcd	47.4 bcd
F22	9	14.5	18.6	16.5 abcd	1.3	92.0	98.0	95.6 ab	2.1	46.9 bcd	50.4 abcd
H23	4	18.9	21.0	19.7 ab	1.0	93.0	98.0	95.4 ab	2.1	68.9 ab	71.8 ab
H24	5	14.1	16.7	15.5 cd	1.1	92.0	99.0	96.6 ab	2.8	40.2 cd	44.6 cd
H25	3	16.9	20.2	18.2 abcd	1.8	95.5	97.5	96.2 ab	1.2	57.7 abcd	61.3 abcd
H26	4	13.9	18.6	16.5 bcd	2.0	81.0 ^c	91.0	87.9 b	4.6	46.7 bcd	46.2 bcd
H27	3	17.0	19.9	18.4 abcd	1.5	94.0	95.5	94.5 ab	0.9	59.1 abcd	61.6 abcd
H28	5	15.5	17.0	16.3 bcd	0.7	91.0	97.0	94.5 ab	2.5	45.4 bcd	48.6 bcd
W29	4	15.6	19.6	17.2 abcd	1.8	90.5	93.0	91.5 ab	1.2	51.1 abcd	52.2 abcd
ALL: ^d	154	12.6	22.2	17.2	1.9	81.0	104.0	94.8	3.7	51.7	55.3

^a Average d.b.h., total heights, and total inside bark merchantable volumes with the same letters are not significantly different at $\alpha = 0.05$ (ANOVA; Tukey's HSD for unequal n).

^b Total inside bark merchantable (to 4-inch top) volumes determined with a local equation (Farrar and others 1984; V_{CEF} uses d.b.h. only) and Van Deusen and others (1981) (V_{VD} ; uses d.b.h. and total height).

^c This specimen's crown was severely damaged by an ice storm in the 1990s and never fully recovered apical dominance, as had most other similarly injured trees in this stand.

^d ALL: n = total number of pines sampled for all families; all other variables determined using individual tree measurements, not family-level averages.

slightly from each other in their predictions (51.1 versus 52.2 cubic feet, respectively)—the local model appears well-tuned to the more regional model. However, as is often the case when using a locally derived volume model for trees from outside the calibration source, the addition of height as a variable added several cubic feet to the merchantable volume estimates for the non-local families (table 4).

DISCUSSION AND CONCLUSIONS

Even though one of the original goals of this outplanting was to determine relative susceptibility to fusiform rust, the intervening thinnings mean that we still do not have a good sense of its frequency as a function of family in the Crossett area. Regionally, loblolly pine from the Upper West Gulf Coastal Plain (especially southeastern Arkansas) tend to have a relatively low incidence (less than 10 percent infected), although this also depends on the family being considered (Grigsby 1975a, 1975b; Grigsby 1977; Randolph and others 2015). Short-term differences in fusiform rate were hard to glean from the 1969 outplanting on the CEF, but other longer-duration local progeny tests suggested family-based differences in survival. For example, when planted in other regions and exposed to a wider range of environmental conditions, woods-run loblolly pine from the CEF (and vicinity) had good survivorship and low (less than 20 percent) fusiform infection rates after the first decade, although in some locations survivorship was low and fusiform infection reached 50 percent (Grigsby 1975b). Other studies of fusiform infection indicated that the progeny of superior pines may fare somewhat better than those of conventional woods-run sources (for example, Grigsby 1975a).

From this limited assessment of the tested families, growth performance—as suggested by height—was not consistent between 3 and 48 years post-planting (tables 2 and 4). For example, the family that had the lowest height at 3 years (F11, averaging 3.8 feet) was amongst the tallest at 48 years (averaging 96.8 feet). The Crossett woods-run family (W29) was about average (5.2 feet) at 3 years, but is amongst the shortest today, and other families varied in their eventual outcomes as well (table 4). However, a larger sample of the 1969 outplanting will be required as the height data from 36 plots may be too limited to distinguish between fine-scale site and stand density effects and the influence of family on height performance.

Because bole volume was not determined in 1972, a comparison between short- and long-term patterns in wood accumulation cannot be made. However, the volumes given in table 4 support two other conclusions. First, local volume equations that do not incorporate differences in tree height (and most do not) are inappropriate for comparing volumes between families selected from the local population and those chosen

from more distant seed sources. Local volume equations based solely on d.b.h. (or height, or some other single variable) will probably fail to capture significant differences in allometry and lead to inaccuracies in volumetric predictions that could meaningfully impact predictions (for example, Avery and Burkhart 1983). Second, although this limited sample failed to produce statistically significant volume differences between most families tested, it is clear that the noticeably bigger full- and half-sib families would have yielded more fiber than the woods-run stock, assuming the same number of trees had been planted (and survived) on the same site.

Though the early evaluation of progeny tests has become the standard, this preliminary study suggests this may not be advisable for planted pines to be retained much longer than conventional silvicultural rotations (currently, between 20 and 30 years in the Upper West Gulf Coastal Plain). While further analysis and a larger data set are required, the change in rank order of the most and least “successful” families after 48 years could mean that certain objectives (such as carbon sequestered under long-term contracts) may be better served by a more measured evaluation of growth performance.

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EFFECTS OF GAP SIZE ON NATURAL REGENERATION IN A PINE-HARDWOOD STAND A QUARTER CENTURY AFTER HARVEST

Matthew G. Olson and Don C. Bragg

Abstract—In 1992, an experiment to assess the effect of harvest gap size on natural regeneration of coastal plain mixedwoods was installed in a mature stand on the Crossett Experimental Forest in southeastern Arkansas. Three levels of a gap-size treatment (0.25 acre, 0.625 acre, and 1 acre) were installed in a randomized complete block design with three replications. Gaps were revisited in 2016 to evaluate the effect of gap size on natural regeneration. Gap size significantly ($p < 0.05$) explained variation in pine density (diameter at breast height [d.b.h.] ≥ 3.5 inches), but not the densities of hardwood species groups. Gap size was also significant in a model for pine importance value. Mean separation revealed that pine density and importance value were highest in 1-acre gaps, lowest in 0.25-acre gaps, and intermediate in 0.625-acre gaps. These results provide further support for research indicating gap size plays an important role in natural pine regeneration.

INTRODUCTION

There is growing interest in management strategies that enhance compositional diversity and structural complexity of forests (for example, Puettmann and others 2009). Compared to forests dominated by a single species, mixed-species forests are generally less vulnerable to perturbations from insect pests and microbial pathogens (Drever and others 2006, Jactel and Brockerhoff 2007). Species-rich, structurally heterogeneous forests offer a wide range of ecological niches, which, in turn, can support higher biodiversity (Hunter 1999). More recently, there has been interest in increasing tree species richness and structural heterogeneity as part of a climate change adaptation strategy designed to enhance resiliency and the capacity of forests to adapt to future uncertainty (Messier and others 2013).

In the southern pine region, a mixture of pine and hardwood species (“mixedwood”) may offer several of the aforementioned benefits. For example, mixedwoods have been shown to provide a wider range of forage for both game and non-game wildlife species than stands dominated by either hardwood or pine (Wigley and others 1989). By recruiting both pine and hardwood species into merchantable size classes, mixedwood management can diversify timber resources to supply a wider range of forest product markets than pure stands of either pine or hardwood alone (Zahner and Smalley 1989). Since mixedwoods

can offer a variety of economic and ecological values at a low cost, mixedwood management may be attractive to landowners less interested in intensive forest management.

Multi-aged silviculture also offers considerable management flexibility and can be used to create or maintain structurally complex mixedwood stands. Where the objective is to sustain recruitment of an ecologically diverse assemblage of tree species while retaining mature forest cover, gap-based silviculture methods may be appropriate (Nyland 2016). Although there are some operational challenges associated with gap-based methods, particularly with harvesting of older gap cohorts (Murphy and others 1993), gap-based methods have the flexibility to meet a wide range of landowner objectives. For example, group selection openings can be sized to allow adequate light into the understory to regenerate shade-intolerant species (Marquis 1965) while developing and releasing advance reproduction of shade-tolerant species (Arseneault and others 2011). An additional benefit of partial overstory removal is the ability to help mitigate the negative visual impacts of timber harvesting.

Group selection is an understudied area of silviculture in southern pine-hardwood stands, yet it may be attractive to landowners interested in multi-aged, mixedwood management. To help fill this knowledge gap, a study was initiated to investigate the effectiveness of harvest

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gap size for regenerating pine-hardwood mixtures in the West Gulf Coastal Plain. Early results of this study (Shelton 1998) indicated that gap size had no effect on pine seedling density or stocking in the third year after harvest; however, pine regeneration was more abundant than the oak species in gaps, and pine seedlings were significantly taller in the largest gaps than the smallest gaps (1 acre versus 0.25 acre, respectively). Although this preliminary research provided insights on early gap-phase dynamics in southern pine-hardwood stands, much remains unknown about the long-term outcomes of within-gap development in relation to gap size. The purpose of this followup investigation was to assess natural regeneration and gap-cohort development a quarter century following gap creation.

METHODS

The original experiment (Shelton 1991) was established in a 34-acre mature, pine-hardwood stand on the U.S. Department of Agriculture Forest Service's Crossett Experimental Forest (CEF) in southeastern Arkansas (33°2'36" N, 91°56'22" W). The study site was a broad, somewhat poorly drained upland flat dominated by Bude silt loam soils with a loblolly pine (*Pinus taeda*) 50-year site index of 90 feet. Years earlier (from 1982 to 1984), the stand had been a part of a study of herbicide treatment of certain hardwoods using annual stem injections. In February 1992, the study site received a site preparation burn. In November and December of that year, a gap-size treatment with three levels (0.25 acre, 0.625 acre, and 1 acre) was installed using a randomized complete block design with three replications. Adjacent harvest openings were separated by at least 100 feet, and the matrix between gaps received an improvement cut targeting the retention of 75 square feet per acre in pine sawtimber (no hardwoods were removed from the matrix). Since then, the only activity in this stand has been scattered, low-impact salvage harvesting of pine.

The study site was revisited in the summer of 2016 to evaluate the long-term effect of harvest gap size on the regeneration within gaps. Gap boundaries were delineated based on the locations of mature tree stems at the border of openings (in other words, the expanded gap) and georeferenced using a recreation-grade GPS unit. A complete census of all live trees at least 3.5 inches in diameter at breast height (d.b.h.) was conducted in each gap. For each tallied tree, the species, d.b.h., and amount of vine growth in the crown were recorded.

Analysis of variance (ANOVA) was used to test for an effect of gap size on stem densities of pine [loblolly and shortleaf pine (*P. echinata*)], oak [cherrybark (*Quercus pagoda*), post (*Q. stellata*), southern red (*Q. falcata*), water (*Q. nigra*), white (*Q. alba*), and willow (*Q. phellos*)], sweetgum (*Liquidambar styraciflua*), and other species [mainly black cherry (*Prunus serotina*), blackgum (*Nyssa sylvatica*), eastern hophornbeam (*Ostrya virginiana*), flowering dogwood (*Cornus florida*), hickories (*Carya* spp.), red maple (*Acer rubrum*), sassafras (*Sassafras albidum*), and winged elm (*Ulmus alata*)]. ANOVA models were run to test for a gap-size effect on pine-hardwood composition using the importance value of pine (pine IV) as the response variable. Pine IV was calculated as the average of pine relative density and relative basal area. Pine IV was arcsine square-root transformed to improve normality. Mean separation was performed using Tukey's Honestly Significant Difference test. All analyses were conducted using SAS 9.4 using $\alpha = 0.05$.

RESULTS

Mean area of harvest gaps measured in 2016 were consistently larger than the area of the original gap-size treatments (table 1). This difference arose because the original determination of harvest gap size was based on the opening produced by removing trees within the specified area, and since we could not reconstruct the

Table 1—Description of gap size treatments based on the original study design and measurements of gaps in 2016

Original gap treatment (acres)	Gap size class	2016 Mean gap attributes	
		Area (acres)	Diameter:height ^a
0.25	Small	0.37	1.5
		(0.35-0.39)	(1.3-1.6)
0.625	Medium	0.86	2.0
		(0.79-0.99)	(1.9-2.2)
1	Large	1.34	2.7
		(1.31-1.37)	(2.4-2.9)

^a Diameter:height is the ratio of gap diameter to the average height of co-dominant and dominant trees bordering the gap.

Range is given in parentheses.

exact location of these gaps, we resorted to delineation using the remaining evidence (uncut border trees). In 2016, the mean gap diameter to canopy height ratios of small, medium, and large gaps were 1.5, 2.0, and 2.7, respectively (table 1).

Comparison of diameter distributions in the gaps showed an increasing tree density with increasing gap size (fig. 1). Most of this increasing trend in total density appeared to be attributable to more pines, especially in the small diameter classes. Gap size was a significant factor in ANOVA models explaining pine density ($p = 0.04$) but not in models for oak ($p = 0.50$), sweetgum ($p = 0.38$), or other species ($p = 0.46$). Mean separation indicated that large gaps had a significantly higher density of pine than small gaps (89 versus 14 trees per acre), with an intermediate level of pine density (40 trees per acre) in medium gaps (fig. 2). The slight increases in oak and sweetgum density with increasing gap size were not statistically significant, and the other species group showed no trends with gap size.

Although their numbers were small, pines also increased appreciably in the larger diameter classes with increasing gap size (fig. 1). For instance, in the small gaps, the largest diameter pines were 9 inches in d.b.h. while the medium and large gaps both had pines greater than 12 inches in d.b.h. Increasingly larger gaps had some impact on both the number and size of hardwoods but not as consistently or as pronounced as for pine.

The increasing trend in pine stem density was mirrored by pine IV (fig. 3). Gap size was a significant factor in ANOVA models explaining pine IV ($p = 0.03$), with pine IV significantly higher in large gaps (38 percent) than small gaps (10 percent). No statistically significant differences were detected between pine IV in medium gaps (24 percent) and either of the other gap sizes.

DISCUSSION

One of the original goals of this study was to inform the management of multi-aged pine-hardwood mixtures (Shelton 1991). Therefore, it is also important to consider the development of hardwood species, not just as a competitor of pine but also as a key component of gap cohorts. Similar to what was found for oak regeneration recorded in 1995 on this same study, oak stem density in 2016 was not affected by gap size. Although not individually considered in the earlier study, sweetgum density in 2016 also was unaffected by gap size. However, both oak and sweetgum densities nominally increased along the gap size gradient, suggesting a possible trend of increasing density with gap size, albeit a weak one. Given the general dominance of hardwood regeneration in these gaps, pine regeneration success under this group selection system appears to be the more critical area of concern.

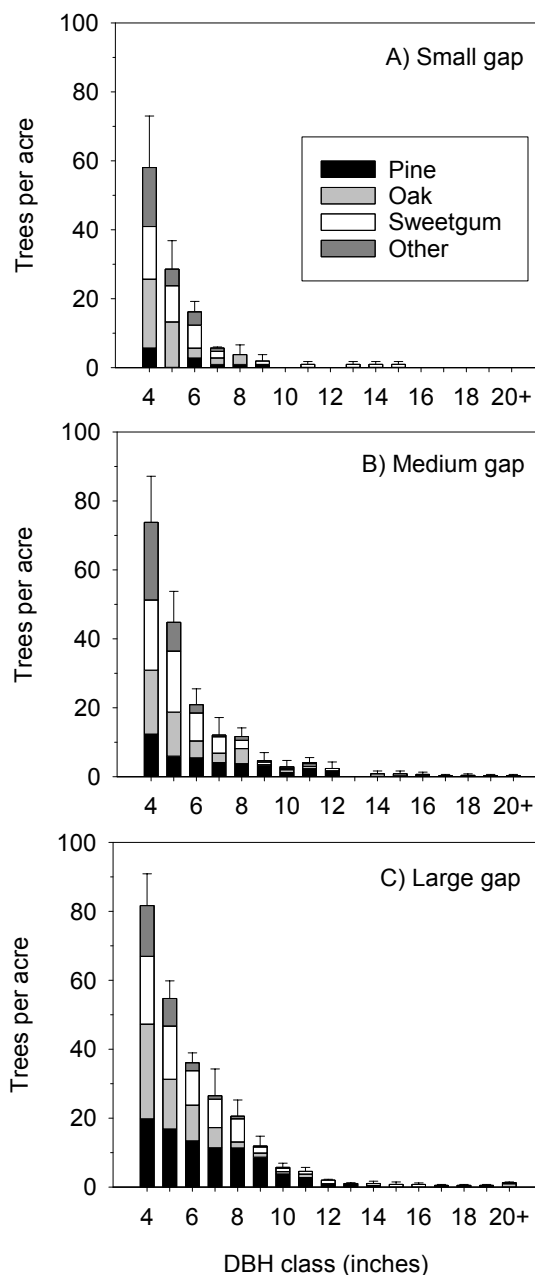


Figure 1—Diameter distribution of small (A), medium (B), and large (C) gap treatments partitioned into pines, oaks, sweetgum, and other species groups. Error bar is equal to one standard error and was calculated from total tree density in each d.b.h. class.

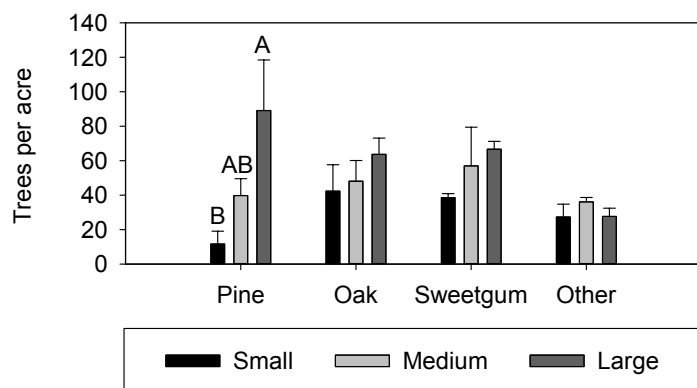


Figure 2—Mean trees per acre of pines, oaks, sweetgum, and other species groups partitioned into gap-size treatments. Error bar is equal to one standard error. Gap treatment means within a species with a different letter are statistically different ($\alpha = 0.05$).

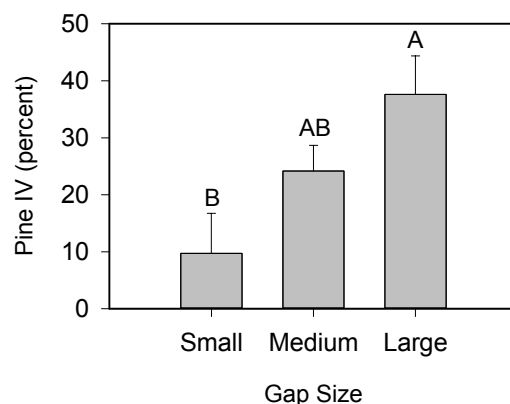


Figure 3—Mean percent pine importance value (IV) of small, medium, and large gap treatments. Error bar is equal to one standard error. Gap treatment means with a different letter are statistically different ($\alpha = 0.05$).

Interestingly, the possibility that pine regeneration may not prove adequate was not immediately apparent. Unlike our findings, third-year results of the original study determined that pine seedling densities were comparable among all gap size treatments (Shelton 1998). A related experiment in northern Louisiana testing the same gap sizes likewise failed to detect an effect of gap size on pine seedling density in the first few years after gap creation (Cain and Shelton 2001). In both of these studies, initial pine seedling densities exceeded 2,500 seedlings per acre, which is considered adequate natural reproduction in uneven-aged forests at that stage of development (Baker and others 1996, Shelton 1998). During the intervening years, attrition of the shade-intolerant pine seedlings was expected, even in the largest gaps. After 24 years, the differential survivorship as a function of gap size can probably be attributed to a combination of enhanced competitiveness of pine within larger canopy openings and higher pine mortality in smaller gaps.

While even relatively small gaps in mixedwood forests can support significant numbers of shade-intolerant pine and hardwood seedlings (for example, Rantis and Johnson 1998), their growth performance will generally increase with gap size. Shelton (1998) noted that pine seedlings were significantly taller in the large gaps than in the small gaps. A different gap study found that pine seedlings were significantly taller in 1/3-acre harvest gaps compared to 1/10-acre harvest gaps, suggesting enhanced growth of southern pine in larger openings (Perry and Waldrop 1995). At 24 years, the greater numbers of larger pines (without concurrent responses by hardwoods) in bigger gaps (fig. 1) provide further support that pines become increasingly competitive with greater resource availability.

The largest openings created for this study still had average pine IV that would have produced a mixedwood composition (between 25 and 75 percent) after 24 years of development. While the composition of medium gaps fell just short of the 25-percent pine threshold, the higher proportion of pine in larger size classes suggests that these gaps may develop naturally into mixedwoods, barring any unexpected mortality. It is apparent in figure 3 that the small gaps have failed to sufficiently recruit a mixedwood gap cohort. The higher rate of mortality in small gaps is certainly related to the greater influence of the surrounding mature overstory on gap light and lower competitiveness of young pine in the more heavily shaded environment of smaller gaps. Silviculturally based light management in mixedwood stands is further challenged by differential impacts of hardwoods and pines on light availability, as other studies in uneven-aged pine-hardwood stands have shown (for example, Guo and Shelton 1998). Although not presented here, the level of vine infestation of tree crowns generally increased with decreasing gap size. Therefore, vine infestation may have also contributed to higher pine mortality in small openings.

CONCLUSIONS

An understanding of gap-cohort development in relation to gap size and other silvicultural treatments (for example, site preparation, artificial regeneration, release) informs the development of gap-based silviculture systems. This study yielded insights on the role of harvest size in the development of this particular mixedwood stand a quarter-century after gap formation. This study suggests that if regenerating a mixedwood composition is desired, harvest gaps may need to be at least 0.625 acres to ensure adequate pine regeneration without further intervention. In medium-

sized gaps, sufficient pine recruitment and retention may be achieved but would benefit from a later tending that preferentially retains pine. However, a late intervention in small gaps is unlikely to produce mixedwood cohorts—earlier treatments to reduce competition from hardwoods and vines would be required to maintain a pine component.

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Bottomland Silviculture

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A PRELIMINARY SYNTHESIS OF GROWTH DATA FOR BOTTOMLAND HARDWOOD SPECIES COMMONLY PLANTED IN THE LOWER MISSISSIPPI ALLUVIAL VALLEY

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Ellen Boerger, and Charles O. Sabatia

Abstract—This study synthesizes published height growth measurements for a range of bottomland hardwood species commonly used in afforestation efforts in the Lower Mississippi Alluvial Valley (LMAV), including a variety of oak and non-oak species, from establishment up to 20–30 years of age. Over this time period, cottonwood outperforms all species, while red oaks (cherrybark, Nuttall, water, and willow), sweetgum, and sycamore show intermediate growth trends. Green ash and swamp chestnut oak show lower growth over the same period. In terms of site quality effects, heavier textured clay soils tend to produce lower growth rates for most species, while higher quality loam soils produce the highest growth. Nonetheless, the lack of published data suggests that an increased sampling effort is needed to improve our knowledge of growth patterns of trees commonly used in afforestation in the LMAV and assist landowners and managers in anticipating stand development and silvicultural treatments.

INTRODUCTION

In recent decades, restoration efforts have established hundreds of thousands of acres of planted hardwood stands throughout the Southeastern United States in an effort to restore bottomland hardwood (BLH) forests on marginal farmland (Allen 1997, Twedt 2004). These efforts have been largely supported by Federal incentive programs that began in the 1980s, particularly the Wetland Reserve Program (WRP) and Conservation Reserve Program (CRP) (Gardiner and others 2004, Schoenholtz 2001, Twedt 2004). The reestablishment of BLH forest cover is meant to reverse many decades of forest loss to agricultural conversion and restore ecosystem services that these forests provide. The intended benefits include improvement in soil and water quality, enhancement of wildlife habitat, and sequestration of carbon, in addition to forest production for biomass or timber (Stanturf and others 2001).

In the Lower Mississippi Alluvial Valley (LMAV), much research has focused on planting approaches and early growth and survival, thereby contributing to more effective establishment methods (Stoll and Frey 2016). Several studies have also reviewed the challenges of afforestation and restoration of BLH forests (e.g., Dey

and others 2010, Groninger 2005, Stanturf and others 2001). While at least one study has quantitatively assessed trends in species and establishment approaches used in hardwood afforestation programs (Schoenholtz and others 2001), to our knowledge no studies have attempted to synthesize published growth data for the wide variety of species used in afforestation efforts, especially across varying site conditions. This represents a particular knowledge gap given that many of the earlier plantings are approaching stand closure and stem exclusion and may benefit from silvicultural treatment. Better growth data, particularly height data, are also needed for modeling carbon storage (Duncanson and others 2015, Shoch and others 2009).

The objectives for this study were to (1) evaluate the availability of measurement data for different bottomland hardwood species and plantation ages, (2) evaluate height growth trends among bottomland hardwood plantation species, and (3) evaluate growth patterns among soil types. These data should be of value for landowners and managers for anticipating stand development, growth and yield, and silvicultural treatments.

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METHODS

For purposes of this study, we gathered data from published, peer-reviewed literature. We performed searches in Scopus and Google Scholar using combinations of search terms that included “bottomland,” “hardwood,” “plantation,” “Mississippi,” “afforestation,” and also the names of BLH species. We also evaluated proceedings of the Biennial Southern Silviculture Research Conference, U.S. Forest Service publications, and related technical reports. Studies selected for the analysis (1) were located in the LMAV or on bottomland sites in areas adjacent to the LMAV (fig. 1) and (2) reported height growth data at a specific age. While a large number of species were identified, we focused primarily on several commonly planted oak species and non-oak species for which there were adequate data. The most common oak species included cherrybark oak (*Quercus pagoda* Raf.), Nuttall oak (*Q. texana* Buckley), water oak (*Q. nigra* L.), willow oak (*Q. phellos* L.), and swamp chestnut oak (*Q. michauxii* Nutt.). The common non-oak species included cottonwood (*Populus deltoides* Bartram ex Marshall), green ash (*Fraxinus pennsylvanica* Marshall), sweetgum (*Liquidambar styraciflua* L.), and sycamore (*Platanus occidentalis* L.).

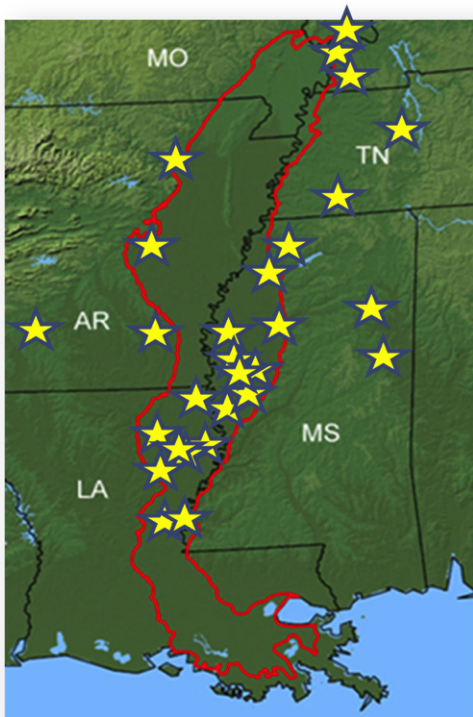


Figure 1—Approximate locations of study sites synthesized from published growth data of bottomland hardwood plantation species used in and around the Lower Mississippi Alluvial Valley. Source: www.lmvsci.gov.

Datapoints were extracted directly from tables where available, and where data were presented in graphical form, datapoints were extracted using DataThief software (Tummers 2006). For purposes of this analysis, individual datapoints were taken directly from plot or replicate means of height or diameter for a given species at a specific age as presented in the individual studies. Studies supplied widely different amounts of usable data. Some studies reported replicate means for multiple species, measured over multiple years, thus providing a large number of datapoints, whereas other studies reported only mean growth values for a limited number of species or ages. In addition to species-age growth data, soil type, site preparation, planting stock, planting density, competition control, and intermediate treatments were also included as available. For the purposes of this paper, a preliminary assessment of age-growth data by species was performed. In addition, height performance across soil types was assessed for a subset of oak species (cherrybark, Nuttall, and water) and non-oak species (green ash, sweetgum, and sycamore) for which sufficient published data were available.

RESULTS AND DISCUSSION

We identified 38 studies that provide quantitative data on height growth performance of woody BLH species that have been assessed in the LMAV or adjacent areas (fig. 1, table 1). There were 13 species that were encountered in this set of studies (fig. 2). However, a subset of species comprised the bulk of the measurements. By far the most frequently measured species in this dataset were several oak species (Nuttall, water, and cherrybark oak) followed by several non-oak species (green ash, sweetgum, sycamore, and cottonwood). These species comprised more than 90 percent of measurements. Other oak species such as Shumard oak (*Quercus shumardii* Buckley var. *shumardii*), willow oak, pin oak (*Q. palustris* Münchh.), swamp chestnut oak, and white oak (*Q. alba*) were less frequent, as was sweet pecan [*Carya illinoensis* (Wangenh.) K. Koch]. Perhaps not surprisingly, the vast majority of measurements have been made on shade-intolerant species, which have been favored because of their rapid growth in open environments (e.g., cottonwood) and/or their wildlife benefits (e.g., red oaks to improve availability of hard mast) (Krinard and Johnson 1980, Lockhart and others 2008). Mid-tolerant to shade-tolerant species were represented principally by green ash and swamp chestnut oak.

Most published height and diameter measurements have been made on stands 10 years age or less (fig. 3). Far fewer measurements have been made in stands of 10–20 years of age, and an even lower number of

Table 1—List of studies providing height growth data for bottomland hardwood species in and around the Lower Mississippi Alluvial Valley

Studies		
Adams and others 2007	Kennedy and others 1987	Miwa 1995
Allen 1990	Krinard 1985	Ozalp and others 1997
Burkett and others 2005	Krinard and Johnson 1975	Patterson and Adams 2003
Carlson and Goelz 1998	Krinard and Johnson 1980	Roth and others 1993
Clatterbuck 2002	Krinard and Johnson 1984	Rousseau 2008
Devine and others 2000	Krinard and Johnson 1988	Self and others 2006
Ezell and Shankle 2004	Krinard and Kennedy 1981	Self and others 2010
Gardiner and others 2004	Krinard and Kennedy 1983	Self and others 2011
Goelz 2001	Krinard and Kennedy 1987	Self and others 2013
Groninger and others 2004	Lockhart and others 2006	Stanturf and others 2009
Gwaze and others 2003	Meadows and Goelz 1993	Stine and others 1994
Jeffreys and others 2010	Meadows and Goelz 2001	Twedt and Wilson 2002
Johnson 1981	Michalek and others 2002	

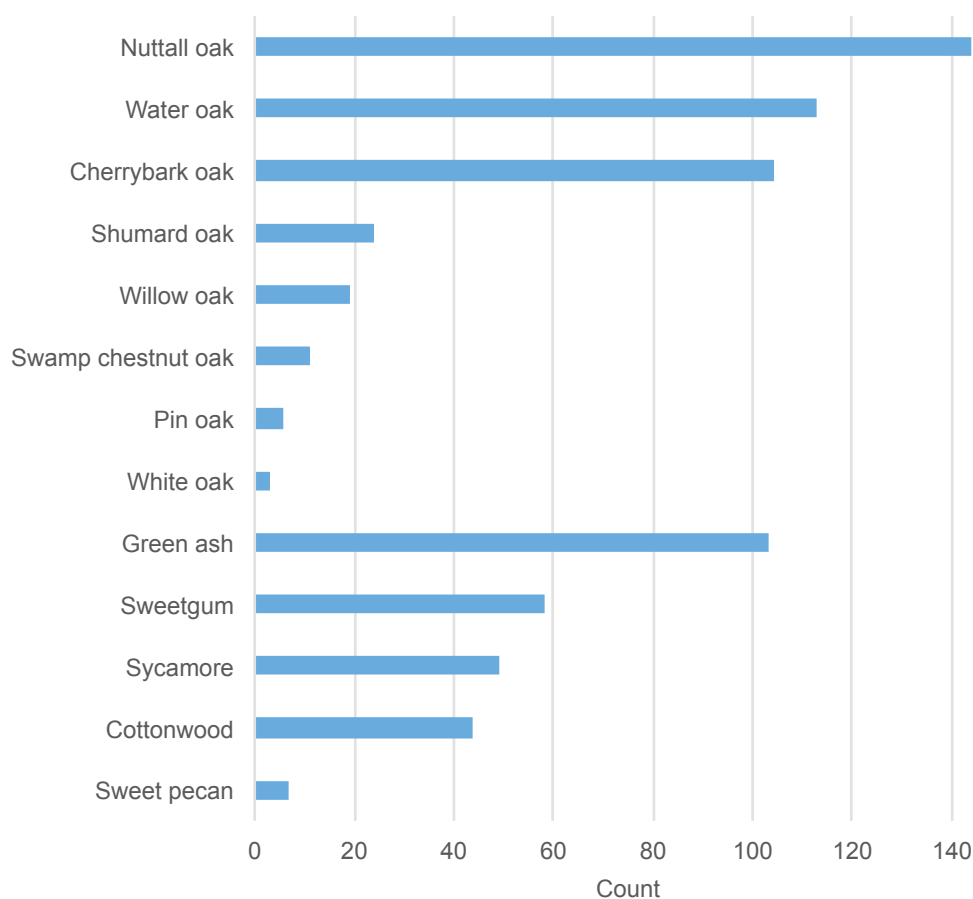


Figure 2—Counts of measurements for different oak and non-oak species synthesized from published growth data of bottomland hardwood plantation species used in and around the Lower Mississippi Alluvial Valley.

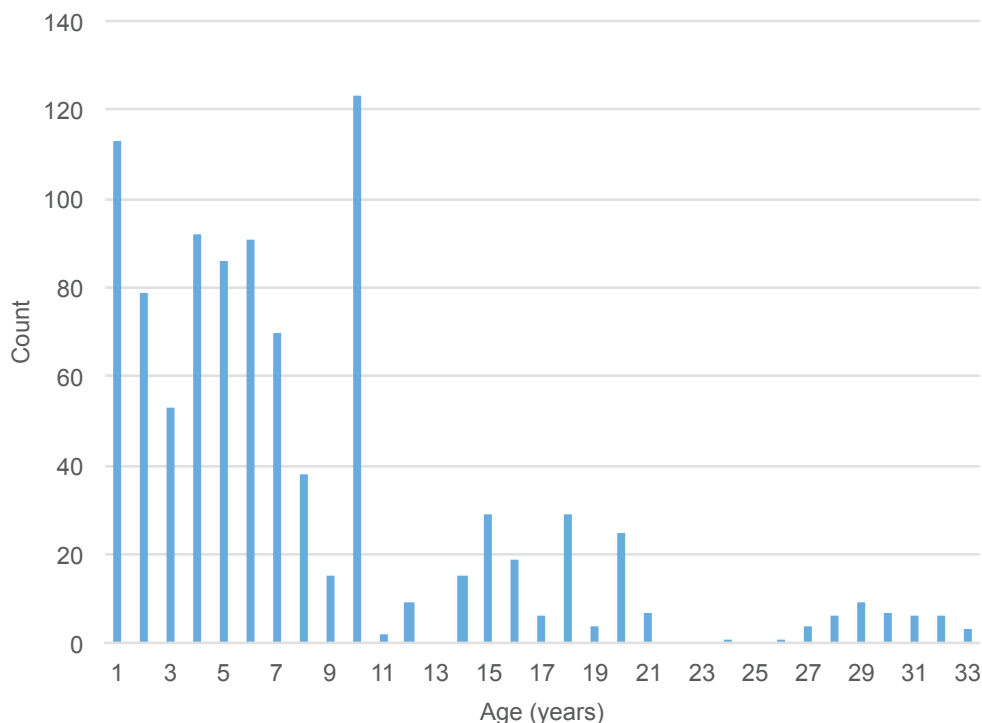


Figure 3—Counts of measurements by age based on published growth data of bottomland hardwood plantation species used in and around the Lower Mississippi Alluvial Valley.

measurements have been made in stands older than 20 years of age. This reflects in part the short history of large-scale planting programs (~30 years), but also the lack of monitoring and measurement of stands beyond the initial years of establishment (Stoll and Frey 2016). Clearly, most studies have focused on establishment and early growth.

Based on linear height growth trends (fig. 4), cottonwood outperforms other BLH plantation species, approaching or exceeding 100 feet by age 20. In contrast, swamp chestnut oak exhibited the lowest height growth, approaching only 30 feet at age 20. Intermediate growth was exhibited by the red oak species (cherrybark, Nuttall, water, willow), sweetgum, and sycamore which approached between 40–50 feet plus at age 20. More specifically, cherrybark oak, sweetgum, and sycamore performed similarly and trended slightly higher than Nuttall, water, and willow oak. Green ash performance was lower than all but swamp chestnut oak, approaching 35 feet by age 20. Outside of cottonwood, the most rapid initial growth was evidenced in sycamore and green ash; however this advantage was not sustained, especially for green ash.

Height growth differences were also evaluated across different soil types for a subset of species for which there were adequate data. Of the soil types evaluated, generally the Falaya and Collins silt loams have higher site quality (as measured by site index) for bottomland hardwood species, with Arkabutla loam soils of

moderate to high site quality, and heavy clay Sharkey soils (a dominant soil type of marginal agricultural land) of lower site quality. Height growth differences were evident when comparing growth of individual species across these different soil types.

Nuttall oak height growth (fig. 5A) was highest in the Falaya silt loam, achieving 55 feet by age 20, which was approximately 25 percent higher growth than in either the Arkabutla loam or Sharkey clay soils beyond age 15. Likewise, water oak height growth (fig. 5B) was also highest in the Falaya silt loam, with height exceeding 60 feet by age 20. Water oak height growth was intermediate in the Arkabutla loam, achieving 45 feet by age 20, while heights were substantially lower in Sharkey clay soils (just above 20 feet by age 18). Cherrybark oak height growth (fig. 5C) was highest in the Falaya silt loam, reaching 60 feet by age 20. Growth was lower in the Arkabutla loam, achieving 47 feet by age 20. Performance of cherrybark oak in Sharkey clay or other heavy clay soils was unavailable, likely because these soils are not considered suitable sites for planting cherrybark oak.

Green ash height growth performance (fig. 6A) was lower than in the oaks, and differed relative to the oak species, showing greater height growth in the Sharkey clay soil, followed by the Collins silt loam, and the Arkabutla loam. Green ash achieved 35 feet in 15 years in the Sharkey soil, but required 18 years in the Collins soil and 20 years in the Arkabutla soil. The lower growth

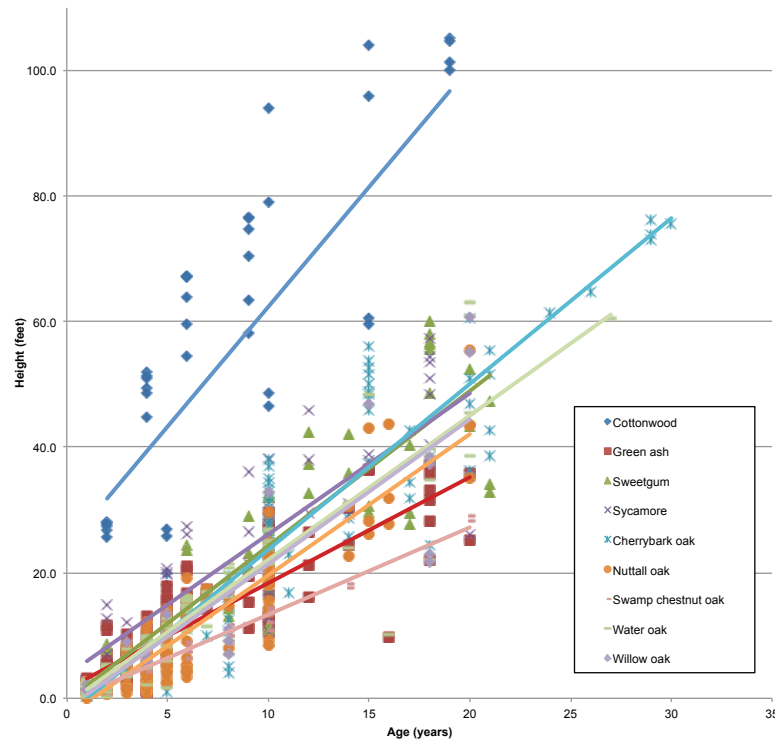


Figure 4—Growth of cherrybark oak, Nuttall oak, water oak, willow oak, swamp chestnut oak, cottonwood, green ash, sweetgum, and sycamore based on published growth data of bottomland hardwood plantation species used in and around the Lower Mississippi Alluvial Valley.

performance in the Arkabutla soil type was notable, given that height performance was more rapid initially (>10 feet versus 5 feet in the other soil types at age 2). Sweetgum height growth (fig. 6B) also diverged among soil types, achieving 60 feet by age 18 in the Collins silt loam, followed by intermediate performance in the Arkabutla loam (>50 feet by age 20) and lower growth in the Sharkey clay soil (30 feet by age 15). As with green ash, sweetgum showed better initial performance in the Arkabutla soil, but this advantage was not sustained. Sycamore height performance (fig. 6C) also diverged with higher growth in the Collins soil (58 feet by age 18), intermediate growth in the Sharkey clay soil (approaching 40 feet by age 15), and lower growth in the Arkabutla soil (~35 feet by age 20). As in both green ash and sweetgum, performance in the Arkabutla soil was higher initially (age 2) but was lower beyond age 10.

CONCLUSIONS

This assessment of published height growth data for bottomland hardwood species used in afforestation efforts in the LMAV yielded several pieces of information. First, measurement data are primarily available for stands less than 10 years of age, with data increasingly limited beyond age 10. Second, while the growth of at

least 13 species has been evaluated at some level, most published measurements have been concentrated on a small group of species that includes cherrybark, Nuttall, and water oak, and cottonwood, green ash, sweetgum, and sycamore. Notably few shade-tolerant to mid-tolerant species have been assessed. Third, based on available data there is varying growth potential among the species. Cottonwood outperforms all species; the red oaks (cherrybark, Nuttall, water, and willow), sweetgum, and sycamore show intermediate growth trends; and green ash and swamp chestnut oak show lower growth over the same period. Growth varies by soil type; heavier textured clay soils generally produce lower growth rates for most species, while higher quality loam soils produce higher growth. Furthermore, growth trajectories differ among species and soil types over time (e.g., rapid initial growth in sycamore and green ash), supporting the need for ongoing measurement. Future research should attempt to increase measurement of stands greater than 10 years of age and across a wider range of species to improve our knowledge of growth patterns of trees commonly used in afforestation in the LMAV. This will be valuable to landowners and managers in anticipating stand development and silvicultural treatments.

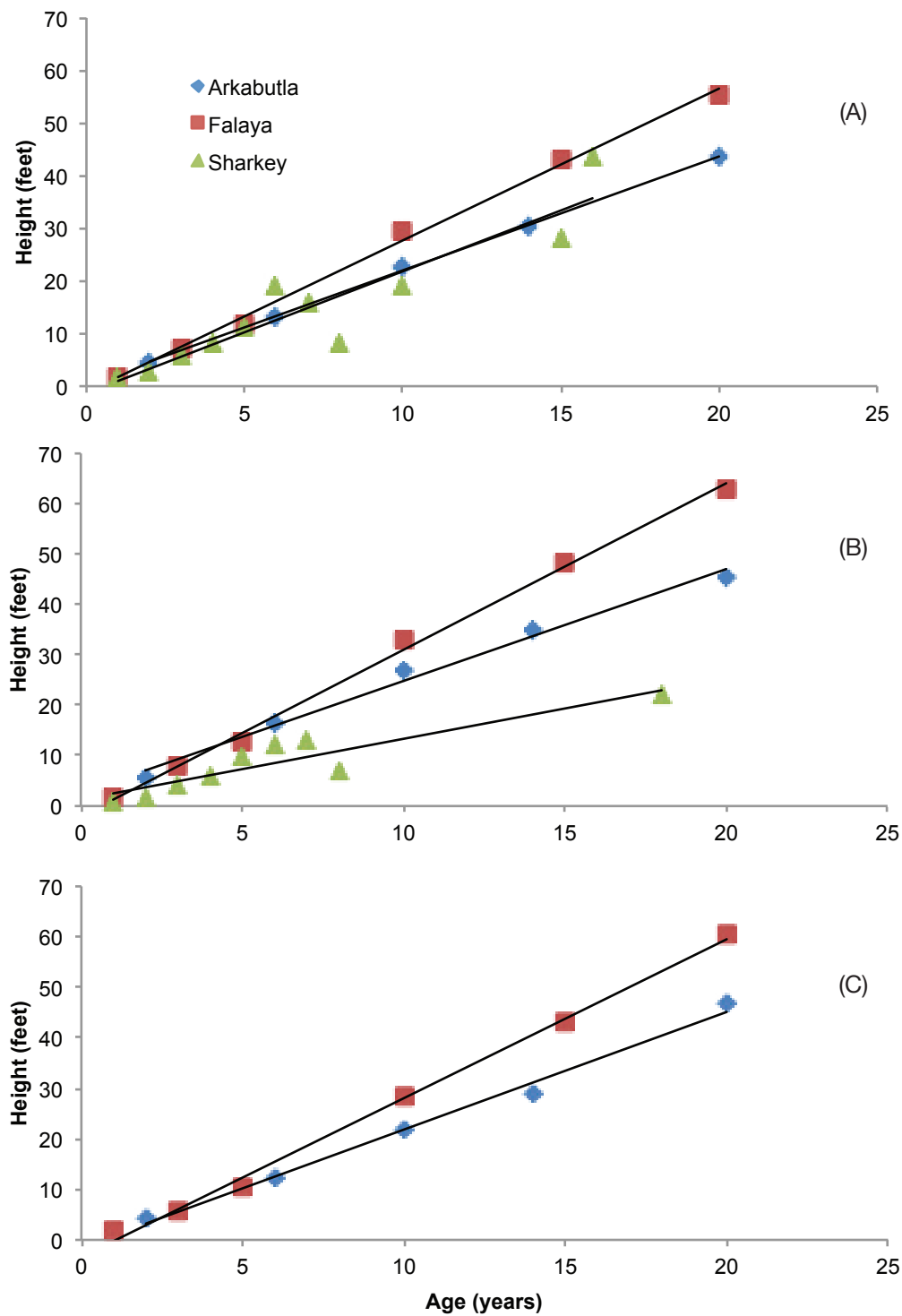


Figure 5—Growth of (A) cherrybark oak, (B) Nuttall oak, and (C) water oak across different soil types based on published growth data of bottomland hardwood plantation species used in and around the Lower Mississippi Alluvial Valley.

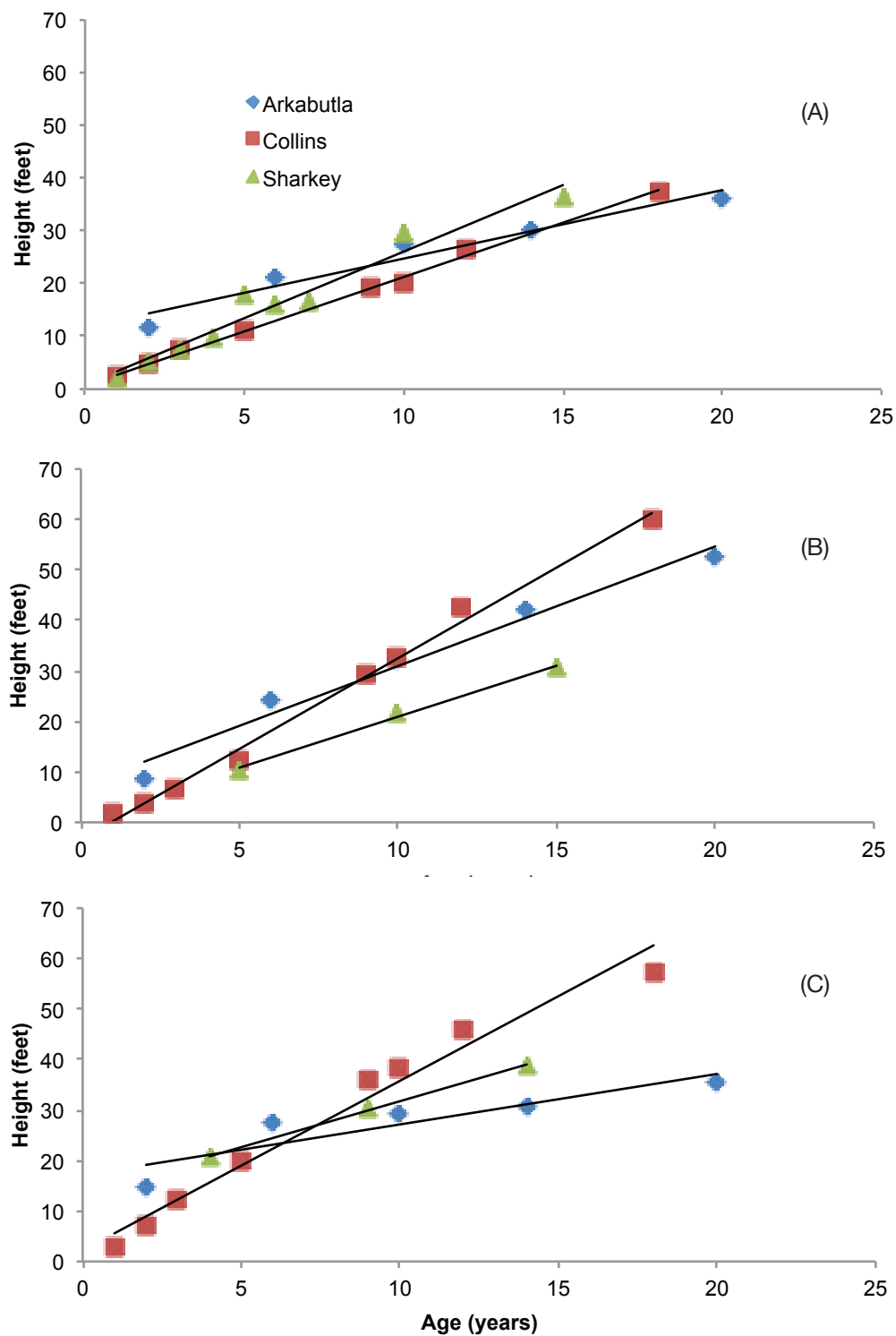


Figure 6—Growth of (A) green ash, (B) sweetgum, and (C) sycamore across different soil types based on published growth data of bottomland hardwood plantation species used in and around the Lower Mississippi Alluvial Valley.

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A COMPARISON OF NUTTALL OAK ESTABLISHMENT METHODS USING IMPROVED AND UNIMPROVED SEEDLINGS, SEEDLING TREATMENTS, AND SITE PREPARATION INTENSITY IN THE LOWER MISSISSIPPI ALLUVIAL VALLEY

Kutcher Kyle Cunningham and H. Christoph Stuhlinger

Abstract—Restoration of bottomland hardwoods in the Lower Mississippi Alluvial Valley (LMAV) has increased over the past 3 decades to restore the resource, increase wildlife habitat, improve water quality, and enhance forest health and production. However, establishment successes of afforestation efforts in this region have been highly variable. This study attempted to employ proven establishment operations with new concepts to further supplement our understanding of oak establishment on bottomland sites. Two locations were established using minimum site preparation and intensive site preparation methods. Cultural factors included varying combinations of subsoiling, chemical site preparation, herbaceous release, and clipping and sheltering treatments to planted 1-0 bare-root seedling stock. An additional opportunity arose to evaluate the impacts on improved and unimproved seedling stock. Significant differences were identified between minimum and intensive site preparation treatments in regard to survival and growth variables. Additional differences were observed in seedling and planting stock treatments at each study site.

INTRODUCTION

Hardwood restoration of bottomland tree species in retired agricultural fields has been a significant practice of varying, but steady, intensity for the past 3 decades. A majority of these hardwood plantations were established in the Lower Mississippi Alluvial Valley (LMAV). The driving force for these efforts has been the soil and water conservation cost-share programs (e.g., Wetland Reserve Program, WRP) initiated in the 1980s, resulting in hundreds of thousands of reforested acres in the LMAV (Lockhart 2008, Schoenholtz and others 2001). High variability in seedling establishment success resulted in a need for improved establishment methods for existing cost-share programs. Ultimately, improvement in our understanding of site factors including species/site relationships, compacted soil layers, and competition control resulted in significantly higher planting success rates (Ezell and others 2007, Ezell and Shankle 2004). This study attempted to further supplement our understanding of site preparation methods through soil amelioration and competition control operations, while developing new ideas about necessity and timing of operations.

Genetic improvement efforts have been conducted on a limited basis for oak species. However, selection of seed from parent trees with desirable traits and ensuring genetic diversity through seed selection methodology are potentially important to successful plantings of oak species (Dey and others 2008). Oak seedling improvement efforts have been successfully conducted by the Arkansas Forestry Commission (AFC). The improved selections were generated from rogued seed orchards for Nuttall oak (*Quercus texana*) and cherrybark oak (*Q. pagoda*). The trees in the orchards were open-pollinated. While total genetic gain is not measurable with such a method, the seedlings are still potentially improved over standard nursery seedlings or “unimproved” seedlings. The largest potential differences stem from the fact that large amounts of acorns may be selected from one single tree of unknown genetic traits for unimproved seedlings. Our study incorporated improved and unimproved seedlings into the design. The goal was to evaluate, in an operational field setting, improved selections against unimproved seedlings. Additionally, we desired to explore the impact cultural treatments could have on the different seedling stock types.

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Finally, individual seedling treatments to promote survival and growth of hardwood seedlings have not been fully explored. Some of these treatments have included top-clipping and the application of tree shelters as independent measures, but few studies have considered both treatments simultaneously. Significant work was conducted evaluating the impact of top-clipping with red oak species such as northern red oak (*Q. rubra*). Top-clipping has been employed at a lesser extent in other oak species, but has proven beneficial in some studies (Dey and others 2008). Some studies have demonstrated the benefits and problems (e.g., cost and maintenance) associated with the use of tree shelters in seedling establishment of bottomland hardwoods (Stuhlinger 2013). Our study was designed to test new methods incorporating top-clipping with short tree shelters to potentially address prior concerns with these treatments.

METHODS

Study Site

Two study sites in the LMAV were selected for use at the University of Arkansas Division of Agriculture Pine Tree Research Station near Forrest City, AR. The retired agricultural sites were located on terrace slope positions along tributaries of the Le'Anguille River. Soil types at both sites were Loftis somewhat poorly drained silt loams. The sites had a potential to be wet during the dormant season, but did not flood regularly. Site analysis suggested Nuttall oak would be a good fit for these site types. Therefore, improved and unimproved Nuttall oak seedlings from the AFC were used in the study.

Vegetative cover at site 1 primarily consisted of native warm season grasses, with lesser components of broadleaf weeds, vines, and woody species. Vegetative cover at site 2 consisted of a mix of competition types including grasses, broadleaf weeds, vines, and woody species. Both sites were assumed to contain compacted soil layers present near the surface, evident by soil probing.

Treatments

Minimum site preparation (MSP) (site factor) – Site 1 was assigned the MSP treatment group. The MSP treatment included mowing in July 2014, followed by an herbicide application in September 2014. Glyphosate (Accord® XRT II) was applied at 2 quarts per acre for the September site preparation treatment. Seedlings were hand-planted in 20-tree row replicates during February 2015.

Intensive site preparation (ISP) (site factor) – Site 2 was assigned the ISP treatment group. The ISP treatment included mowing in July 2014, followed by an herbicide application in September 2014. An attempt was made in November 2014 to subsoil the site; however, excess moisture precluded this treatment.

Instead, each planting location was drilled by auger prior to planting to address subsoil compaction. Seedlings were hand-planted in 20-tree row replicates during February 2015, followed by an herbaceous release using 2 ounces sulfometuron methyl (Oust® XP) per acre 2 weeks after planting.

Seedling Treatments (blocking factor) – At both locations, three seedling treatment factors were applied immediately following planting and herbicide treatment applications. Seedling treatment 1 (CL) included top-clipping seedlings to 3 inches above ground. Seedling treatment 2 (CL+S) included top-clipping seedlings to 3 inches above ground, plus adding a 2-foot tree shelter (Tubex™). Seedling treatment 3 (UNC) included simply planting 1-0 bare-root seedlings in a traditional method.

Planting Stock Treatment (split plot factor) – At both site location and for each blocking factor, 20-tree row plots were established for improved planting stock and unimproved planting stock. At the ISP location, an additional three rows were established with unaltered improved and unimproved stock. The extra rows provided 120 1-0 bare-root improved and 120 1-0 bare-root unimproved seedlings for comparison with reduced soil and competition concerns.

Statistical Analysis

Sample size for analysis varied by treatment factor. Replication in rows was 18 for site factor, 3 or 6 for seedling treatment factor (depending on whether split plot factor was employed), and 9 for the split plot analyses. Each tree was measured for survival, groundline diameter (GLD; inches), and height (feet). Statistical analyses between site locations were performed using paired t-tests for comparisons of survival and growth variables. Paired t-tests were also used for planting stock comparisons. Seedling treatment factors were analyzed using a one-way analysis of variance (ANOVA). Means separation was conducted using a Student-Newman-Keuls (SNK) method. All statistics were performed in SigmaPlot 11.0 and conducted at an alpha 0.05 level.

RESULTS

Site Preparation Intensity

Figure 1 illustrates seedling survival for MSP and ISP. A paired t-test detected a significant difference in overall survival between the two sites ($p = 0.01$). Mean ISP site survival was 6 percent higher than survival on the MSP location.

A paired t-test did not identify a significant difference in height between the two locations ($p = 0.10$). Mean overall height growth at the ISP location was 0.4 feet greater than the MSP location. However, average year

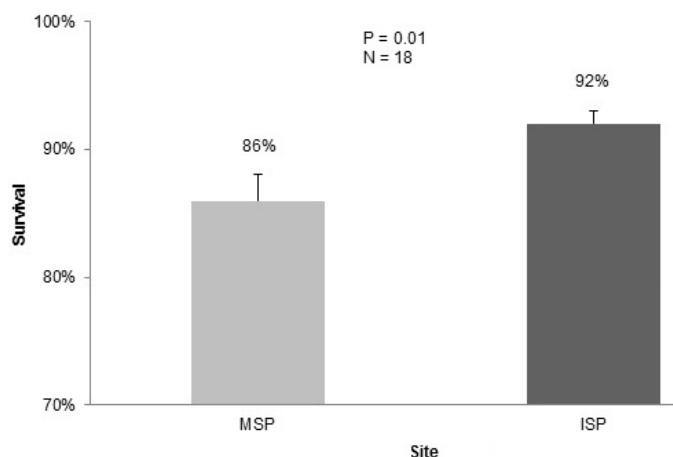


Figure 1—Overall mean survival of Nuttall oak seedlings on the MSP and ISP locations.

2 seedling height was 2.5 feet at both the MSP and ISP sites. A paired t-test identified a significant difference between the two locations in GLD growth ($p = 0.007$). Mean overall GLD growth was 0.04 inches larger on the ISP location versus the MSP location. Mean year 2 GLD was 0.35 inches at the MSP location and 0.45 inches at the ISP location.

Seedling Treatment Factors

ISP location – No significant differences were detected in survival across seedling treatment factors at the ISP location. Year 2 mean survival was 95, 91, and 91

percent, respectively, for CL+S, CL, and UNC seedling treatments at this location.

Figure 2 illustrates mean height data for seedling factors at the ISP site. A one-way ANOVA did not detect a significant difference in year 2 height growth across seedling treatment factors ($p = 0.06$). Mean year 2 total seedling heights were 3.1, 2.5, and 1.3 feet, respectively, for CL+S, CL, and UNC seedling treatments at this location. Furthermore, a one-way ANOVA did not detect a significant difference in year 2 GLD growth across seedling treatment factors. Mean GLD growth was 0.17, 0.19, and 0.20 inches, respectively, for CL+S, CL, and UNC seedling treatments at the ISP location.

MSP location – No significant differences were detected in survival across seedling treatment factors at the MSP location. Year 2 mean survival was 86, 89, and 86 percent, respectively, for CL+S, CL, and UNC seedling treatments at this location.

Figure 3 illustrates mean height data for seedling factors at the MSP site. A one-way ANOVA did not detect a significant difference in year 2 height growth across seedling treatment factors ($p = 0.06$). Mean year 2 total seedling heights were 2.6, 2.3, and 2.6 feet, respectively, for CL+S, CL, and UNC seedling treatments at this location. Furthermore, a one-way ANOVA did not distinguish a significant difference in year 2 GLD growth across seedling treatment factors. Mean GLD growth was 0.15, 0.12, and 0.16 inches, respectively, for CL+S, CL, and UNC seedling treatments at the MSP location.

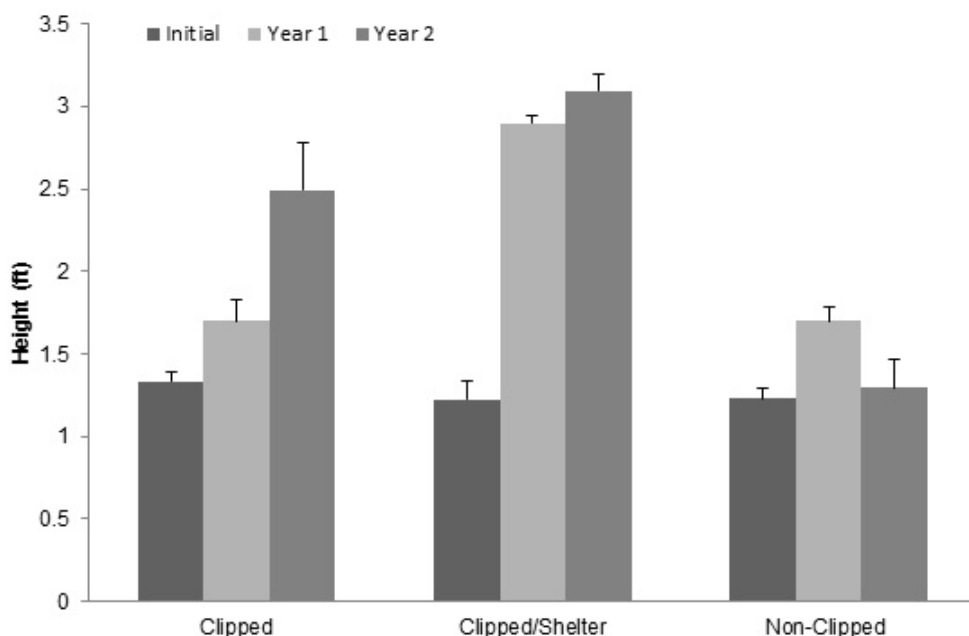


Figure 2—Overall mean heights for seedling treatment factors for Nuttall oak seedlings at the ISP location.

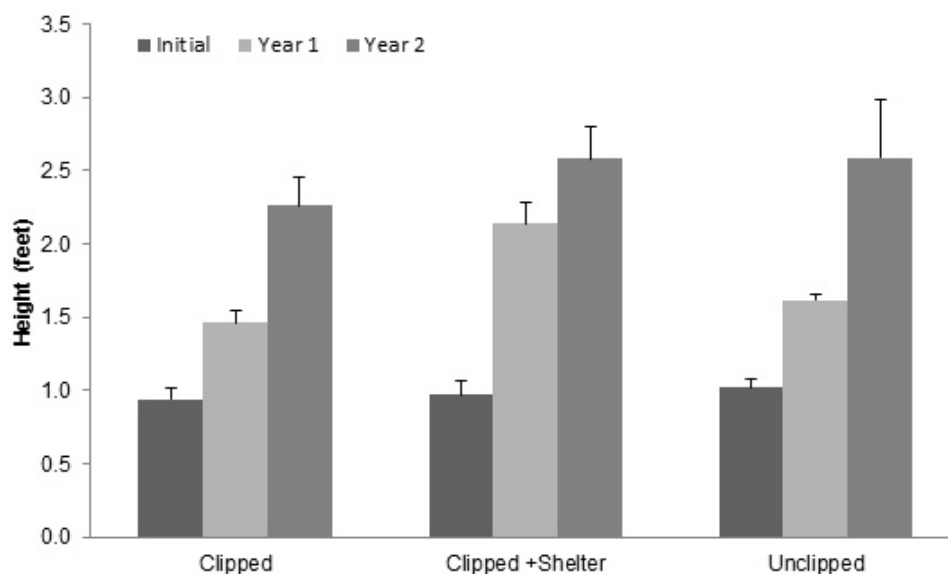


Figure 3—Overall mean heights for seedling treatment factors for Nuttall oak seedlings at the MSP location.

Planting Stock Factors

Figure 4 illustrates mean survival data for seedling factors at the MSP and ISP sites. A paired t-test detected a significant difference between planting stock factors of improved and unimproved at the ISP location. At this location, survival was 94 percent for improved and 89 percent for unimproved. A paired t-test did not detect a significant difference between planting stock factors of improved and unimproved at the MSP location. At the MSP site, survival was 87 percent for improved and 85 percent for unimproved. Seedling

growth analyses between improved and unimproved are not presented.

DISCUSSION

The bulk of statistical significance occurred in the survival and growth analyses across sites. Statistical comparisons for planting stock growth data, improved versus unimproved, were not presented. Differences from genetic factors impacting growth often require several growing seasons to develop. However, some notable differences were beginning to manifest between improved and unimproved seedlings.

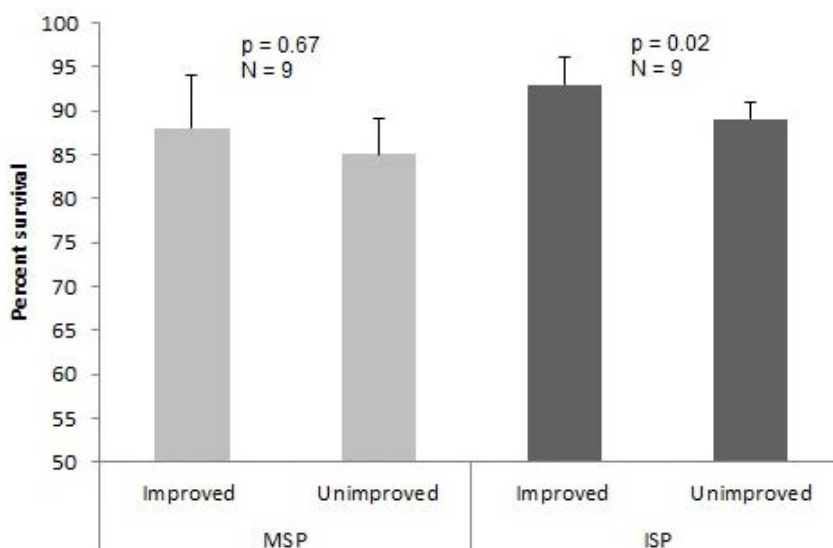


Figure 4—Overall survival between planting stock types at each treatment location.

Survival and growth were significantly better in the ISP treatments compared to the minimal site preparation treatments. While not unexpected and a result demonstrated in previous studies, the results between treatment intensities further validated the more pressing questions regarding the necessity of chemical site preparation treatments in old field scenarios. In this case, the sole application of a foliar active-only herbicide, glyphosate, actually resulted in the release of the herbaceous seed bank in the following season. Survival and growth would have potentially been higher if no site preparation had been conducted and the site would have remained in native warm season grasses. The combination of subsoiling and herbaceous release proved to be a significantly more effective treatment than site preparation alone in this instance.

The analysis between seedling treatment factors provided meaningful and insightful information regarding early growth of planted seedlings. The CL+S treatment demonstrated that first season growth can be greatly increased by its application compared to planting unaltered 1-0 bare-root seedlings. While the CL seedlings closed the gap with the CL+S seedlings in year 2, the greatest benefit and ultimate application of using shelters is the rapid height growth flush initiated in year 1. Furthermore, this effect was primarily present where the intensive cultural treatments were employed. This effect was greatly reduced with MSP inputs. Low predation levels were observed at both locations.

Height growth of 1-0 bare-root Nuttall oak seedlings responded differently at the two sites. At the ISP location, this set of seedlings demonstrated a reduction in height growth over the 2-year period. This dieback pattern is not uncommon in Nuttall oak (Gardiner and others 2010). However, at the MSP location, planted seedlings exhibited substantial 2-year height growth. This could be a microsite effect or potentially an effect of competition intensity, resulting in a height growth response. The rapid growth generated by the combination of intensive site preparation, top-clipping, and addition of a 2-foot tree shelter could prove useful as a method for initiating height growth, a growth response considered atypical of Nuttall oak after transplanting.

Figures 5 and 6 demonstrate the potential effects adequate and inadequate site preparation methods have on seedling survival over time. Survival remained steady at the ISP location from year 1 to year 2. However, survival continued to decrease in year 2 when minimum site preparation occurred. The authors expect this pattern to continue in years 3 through 5 of stand establishment.

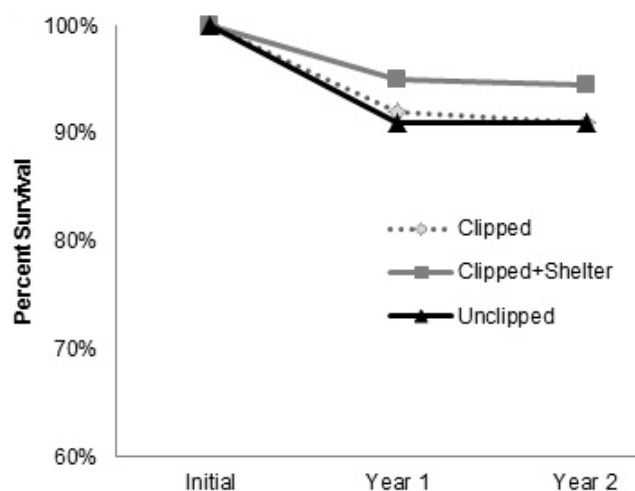


Figure 5—Percent survival by seedling factor at the ISP site.

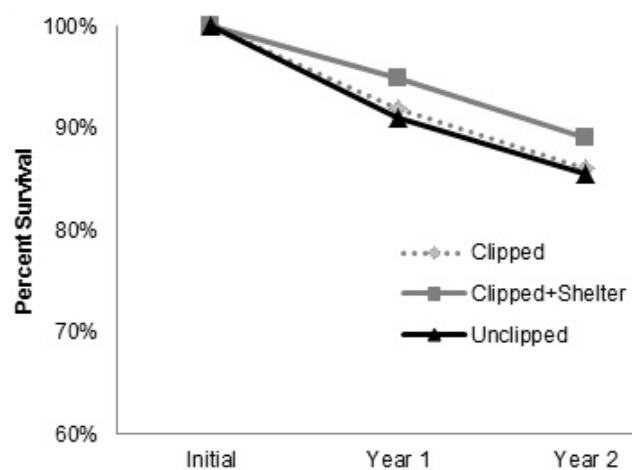


Figure 6—Percent survival by seedling factor at the MSP site.

A question that arose in the course of the study was whether CL+S seedlings were trading off stem diameter and root growth to allocate resources to height growth. The induced rapid height growth pattern was the inverse of the typical allocation of oak seedlings, creating a potential concern for seedling resiliency to adverse environmental conditions. However, an evaluation of GLD did not support this concern. Groundline diameters of CL+S seedlings were equal to or larger than alternative seedling treatment factors. Thus, we concluded that root growth remained adequate for sheltered seedlings (table 1).

Table1—Mean groundline diameter (GLD) growth for minimum site preparation (MSP) and intensive site preparation (ISP) locations by seedling treatment factor

Treatment	GLD growth (inches)	Standard error
MSP location		
CL (unimproved seedlings)	0.16	0.04
CL+S (unimproved seedlings)	0.13	0.03
UNC (unimproved seedlings)	0.17	0.03
CL (improved seedlings)	0.14	0.02
CL+S (improved seedlings)	0.10	0.02
UNC (improved seedlings)	0.15	0.02
ISP location		
CL (unimproved seedlings)	0.14	0.03
CL+S (unimproved seedlings)	0.17	0.03
UNC (unimproved seedlings)	0.21	0.04
CL (improved seedlings)	0.20	0.02
CL+S (improved seedlings)	0.21	0.03
UNC (improved seedlings)	0.18	0.03

CL = seedlings top-clipped to 3 inches above ground;
CL+S = top-clipped seedlings with 2-foot tree shelter;
UNC = unclipped seedlings.

Survival differences between planting stock types were greatest on the ISP location. Improved seedlings had significantly higher survival rates compared to unimproved seedlings. This effect was not observed on the MSP location, with similar survival observed between improved and unimproved seedling stock. A possible explanation is that ameliorating soil and competition concerns adequately allowed for any genetic differences to better express themselves. The authors believe the higher level of competition present and potential soil compaction issues reduced the ability of improved seedlings to express any potential gain in growth traits. Again, growth data comparing planting stock type were

not included in this study, since growth traits could take several years to fully express.

CONCLUSIONS

Addressing soil factors and competition control remain the focus of cultural treatments aimed at increasing survival of planted Nuttall oak seedlings through facilitating root development in the first few growing seasons. The greatest impact from applying the CL+S method was in first year height growth. This rapid growth flush could prove useful in scenarios including areas under heavy predation or at potential risk for growing season flooding. Ultimately, this method could be included in existing methods (cover crops and underplanting) for addressing early height growth concerns of afforestation efforts with Nuttall oak. The ability to utilize 2-foot shelters for one growing season further impacts their use in that they can be reutilized in subsequent plantings, reducing operational costs (a limiting factor for tree shelters).

The improved seedlings in this study performed well in the operational planting. The potential gain in genetic diversity, combined with the ability to obtain at least a mid-parent gain value, help to establish a basis for utilizing improved Nuttall oak in hardwood plantings in the LMAV.

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SILVICULTURAL AND GENETIC INFLUENCES ON PLANTED CYPRESS PRODUCTIVITY

Donald L. Rockwood, Marvin Buchanan, and Monica Ozores-Hampton

Abstract—The potential for baldcypress (*Taxodium distichum* var. *distichum*) and pondcypress (*T. distichum* var. *imbricarium*) plantations has been further evaluated through two silvicultural and genetic studies in Florida. On a flatwoods site, initial bedding+compost, which resulted in better early growth and survival than just bedding, and in turn no bedding, enhanced soil properties and plantation productivity through 16 years with stand basal area averaging 179 square feet per acre at a 10- x 3-foot spacing and 136 square feet per acre at a 10- x 6-foot spacing and associated tree diameters at breast height (DBH) of 5.5 inches and 5.9 inches, respectively. The best progenies increased stand basal area up to 60 percent over these averages. Coppice growth initiated at ~7 years was similar to the original rotation growth, but 9-year-old coppice wood density was less than that of 16-year-old trees. On an irrigated and fertilized sandhills site after 19 years, pondcypress progenies were much smaller than three types of baldcypress (DBH of 6.8 inches versus 10.5, 11.4, and 11.9 inches, with associated stand basal areas of 73 square feet per acre versus 178, 146, and 237 square feet per acre). Seed orchard CO97 composed of 26 pondcypress progenies, 11 baldcypress provenances, 21 baldcypress progenies, and 7 baldcypress clones can annually produce 400,000 or more seed expected to be ~15 percent more productive than unimproved seed. Commercial cypress plantations on non-wetland sites have potential for producing mulchwood and/or sawtimber in multiple rotations of 10 to 25 years.

INTRODUCTION

While taxonomic relationships among three *Taxodium* varieties [*T. distichum* var. *distichum* (L.) Rich (baldcypress - **BC**), *T. distichum* var. *imbricarium* (Nutt.) Croom (pondcypress - **PC**), and *T. distichum* var. *mexicanum* Gordon (Montezuma cypress - **MC**)] remain a source of debate (Tsumura and others 1999), at least one source combines all three into one species with three botanical varieties (Arnold and Denny 2007). The ranges of **BC** and **PC** overlap in forested wetlands throughout the Southeast, and within Florida, they are the most common wetland tree species (Ewel 1990). Net annual harvest of cypress in Florida has often exceeded net annual growth, due largely to harvesting for cypress mulch (Brown 1995). From 1987 to 2013, cypress acreage decreased from 1.40 million acres to 0.96 million acres, and the number of cypress trees decreased from 808.02 million to 568.28 million (Brown and Nowak 2016). Although the associated decline in cypress volume seems to have stabilized recently, the sustainability of the natural cypress resource in Florida, and the industries that depend on it, is of concern.

Silviculture and genetic improvement have potential for significantly increasing productivity of cypress planted on non-wetland sites in Florida. At 1- x 1-m spacing on south Florida muck, a local **BC** source quickly suppressed competing vegetation, was 25.9 feet tall after 7 years, and coppiced consistently from 3 to 7 years of age (Rockwood and Geary 1991). Gaviria (1998) reported encouraging early results from studies of up to 30 seedlots and various cultures on a wide range of sites. In one of a series of genetic and silvicultural studies in Florida (Morse 2003, Rockwood and others 2001), five **BC** provenances planted on south Florida muck differed in tree size but not survival and wood density after 13 years. Individual trees were up to 42.6 feet in height and 7.9 inches in diameter at breast height (DBH) in 13 years, but tree growth rate slowed considerably after 10 years, very likely because of the shallow muck soil and dry climate. Across six studies, nine **BC** provenances and five **BC** checklots were, on average, similar in survival and height to 16 **PC** progenies after as many as

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4 years. Within taxa, individual provenance/progeny differences were significant, but no provenances or progenies were consistently better across sites that ranged from bottomland in northwest Florida to wet and dry flatwoods in northeast Florida to a fertile but poorly drained clay settling area in central Florida. Inconsistent topsoil redistribution hindered the growth of 21 **BC** progenies and two **PC** progenies planted on a reclaimed phosphate mine in northeast Florida in 1998. Bedding+compost (B+C) on a good flatwoods site significantly increased the growth of 30 **BC** and four **PC** progenies compared to bedding alone in studies established in 1999 and 2000. In another flatwoods study planted in 2000, B+C also resulted in better growth and survival than just bedding, which in turn was superior to no bedding. Performance of 13 seedlots in two progeny tests and two commercial plantings on central Florida clay settling areas highlighted the advantages of good site preparation and bedding. An

intensively managed seed orchard with 12 **PC** progenies, 9 **BC** provenances, and 3 **BC** clones produced cones on 5-year-old trees.

This paper updates these findings and extends the assessment of silvicultural and genetic factors that are critical to cypress plantation establishment on non-wetland sites in Florida for the production of mulchwood and sawtimber.

MATERIALS AND METHODS

Two cypress studies in Florida (table 1) involving 65 accessions primarily from Florida (table 2) contribute to this paper. Study SRWC-86 was initially described by Rockwood and others (2001); its B+C treatment involved strip-applying EARTH compost (1.5-0.5-0.5 NPK, www.earthproducts.com) as fourteen ~4-foot-wide, 144-foot-long, 3-inch-high bands, which were subsequently incorporated into fourteen ~2-foot-high

Table 1—Location, planting date, number of BC and PC accessions, total trees, site type, and silvicultural treatments for two studies

Study	Florida location	Plant date	No. of accessions		Total trees	Site type	Silvicultural treatments
			BC	PC			
CO97	Day	12/97	29	19	320	Sandhills; Otela-Penney complex	Fertilizer; irrigation
SRWC-86	Gainesville	2–3/00	13	1	448	Flatwoods; somewhat poorly drained Pomona soil	Bedding+compost; 10 x 3 feet, 10 x 6 feet

Table 2—Type and origin of BC and PC accessions in CO97 and SRWC-86

Type	Origin	CO97	SRWC-86
BC			
Provenance bulk	Arkansas	B5	-
Provenance bulk	Illinois	B7	-
Provenance bulk	Louisiana	B1, B2	-
Provenance bulk	Northwest Florida	B8, B9	-
Provenance bulk	Northeast Florida	B3, B4	-
Provenance bulk	South Florida	B6	-
Individual tree	Northeast Florida	B10, B11, B12, B13, B14, B15, B16, B17, B18, B19, B20, B21, B22, B24, B25, B26, B27, B29, B30, B31	A, B, C, D, E, F, G, H, I, J, L, Ba, Lo
Clone	Various	Senator, Blountstown, Evergreen	-
PC			
Individual tree	Northwest Florida	P1, P2, P3, P7, P8, P9	-
Individual tree	Northeast Florida	P4, P5, P6, P14, P15, P16, P17, P18, P19	K
Individual tree	Central Florida	P10, P11, P12, P13	-

beds before planting 448 trees of 14 accessions in early 2000 (Morse 2003). A ~2-inch-deep, 3-foot-wide top dressing of ERTH compost was applied in August 2001. Representative soil samples were taken from the unbedded (N), bedded only (B), and B+C treatments at planting in 2000 and from the B+C treatment in March 2016. At age 82 months, trees in one replication (seven beds) of the B+C treatment were felled to initiate a coppice rotation. In February 2016 (original age 192 months, coppice age 112 months), to update previous periodic measurements, both replications were remeasured for tree height, DBH, stem quality on a 0 to 5 scale (0 = straight stem, short horizontal branches; 5 = crooked stem, long angular branches, or forking), number of coppice stems equal to ½ of the largest stem (coppiced replication only), and survival and then thinned to the best trees. From a representative 10-tree subset of the felled trees, stem disks were removed at ¼-tree height to determine wood density and moisture content.

The genetic base population CO97 previously described by Rockwood and others (2001) is on sandhills (table 1) and was composed of a different set of diverse accessions, including 10 grafts of the Senator, the world's oldest and largest **BC** until 2012 (table 2), systematically assigned primarily to 24 single-tree plots in 12 replications (four rows of six trees) in a randomized complete block design arranged on eight rows of 40 trees at 15- x 10-foot spacing. In August 2016, CO97 had 32 **BC** accessions (9 provenances, 20 progenies, and 3 clones) and 19 individual **PC** accessions, of which four were systematically allocated to four positions at the ends of the eight rows. Based on August 2016 and previous tree height, DBH, quality, and survival measurements, CO97 was rogued in January 2017 to create a 65-tree orchard with **BC** and **PC** sections.

Stand basal area was also calculated in both studies. Appropriate analyses of variance were conducted on all variables to identify significant silvicultural and genetic factors. As appropriate, means were tested with Duncan's multiple range test.

RESULTS AND DISCUSSION

In SRWC-86, the early beneficial effects of B+C persisted through 16 years (table 3). After 9 months, B+C, involving a relatively low compost rate, was slightly superior to B for growth but not in survival (Rockwood and others 2001). The B treatment was in turn superior for height but not survival to N on this typical flatwoods site after one growing season with much below normal rainfall. However, after 3 years, tree growth and survival in the B and N treatments were so low that both treatments were abandoned. After 16 years, soil properties, especially organic matter and pH, in the B+C treatment were still very favorable, as was tree growth.

More compost may further increase cypress growth. McKinstry (2008), observing mature root growth in the B+C treatment into 12-inch-wide by 12-inch-deep holes filled with municipal solid waste compost mixed with phosphatic clay at 0-, 25-, 50-, 75-, and 100-percent rates, noted that as compost rates increased, soil bulk density significantly decreased and soil porosity increased. The 50-percent compost rate had the most root growth.

Spacing significantly affected tree size and stand development over 16 years (table 4). At 10 x 3 feet in both replications, trees were shorter and thinner at ~7 years than at 10 x 6 feet but had higher stand density. Based on the replication not felled, these differences continued though 16 years as trees at 10- x 6-foot spacing averaged nearly 36 feet tall and 6 inches in DBH, and the trees at 10- x 3-foot spacing had a stand density of nearly 180 square feet per acre. Differences among the 14 progenies for stand basal area, a strong indicator of biomass per acre, were large at both spacings and ages.

Coppice performance was good but also influenced by spacing (table 4). Allowing for the different ages (112 months for coppice versus 82 months for original growth), coppice rotation tree size and stand

Table 3—Soil nutrient levels, organic matter (OM), and pH due to three silvicultural treatments in cypress study SRWC-86 at 0 and/or 16 years

Trt ^a	NH ₄ -N mg/kg	NO ₃ -N mg/kg	OM percent	P mg/ kg	K mg/ kg	Ca mg/ kg	Mg mg/ kg	Zn mg/ kg	Mn mg/ kg	Cu mg/ kg	pH
N ₀	5.7	2.8	2.1	1.9	6.6	73.7	21.6	3.3	0.2	0.1	3.9
B ₀	5.6	2.7	3.6	2.4	9.8	134.0	46.8	2.9	0.6	0.0	3.8
B+C ₀	13	10.4	10.5	71.5	86.1	621.5	122.8	16.2	10.2	0.4	4.1
B+C ₁₆	-	-	4.0	536	57	2968	135	89	6.6	13.3	6.6

^a Treatments: N=none (unbedded); B=bedding only; B+C=bedding+compost.

Table 4—Tree and stand traits by spacing at two original ages and one coppice age for the SRWC-86 B+C treatment before and after thinning

Rotation	Trait	Before thinning		After thinning	
		10- x 3-foot spacing	10- x 6-foot spacing	10- x 3-foot spacing	10- x 6-foot spacing
Original at 82 months	Tree height (ft.) / DBH (in.)	19.7 / 2.8	22.3 / 3.7	-	-
	Trees per acre	1193	645	-	-
	Stand basal area (BA) (ft. ² per acre)	45	38	-	-
	Progeny low–high (BA per acre)	9–91	12–75	-	-
Original at 192 months	Tree height (ft.) / DBH (in.)	34.1 / 5.5	35.8 / 5.9	43.4 / 8.1	42.6 / 7.0
	Wood specific gravity	0.412	0.44	-	-
	Trees per acre	895	635	220	182
	Stand basal area (BA) (ft. ² per acre)	179	136	85	51
	Progeny low–high (BA per acre)	64–271	77–218	-	-
Coppice at 112 months	Tree height (ft.) / DBH (in.)	24.9 / 3.7	27.6 / 4.6	31.2 / 5.0	30.0 / 5.4
	Wood specific gravity	0.371	0.347	-	-
	Trees per acre	1089	311	298	117
	Stems per stool	1.6	1.6	-	-
	Stand basal area (ft. ² per acre)	91	39	42	19

development may actually have surpassed that of the original rotation, e.g., at 10 x 6 feet, tree DBH of 3.7 inches at 82 months versus coppice tree DBH of 4.6 inches at 112 months. As with the unfelled trees, coppice tree size was less at 10 x 3 feet than at 10 x 6 feet, but stand density was higher. The number of coppice stems per stool, 1.6, was favorable for both spacings.

Wood density trends with age appear to follow those of pines (*Pinus* spp.) (table 4). The densities of the older original trees were higher than those of the younger coppice trees at both spacings. Densities at both ages were higher than the average density of 0.330 noted for fast-growing 13-year-old **BC** on muck soil (Rockwood and others 2001) and less than the 0.470 reported by Panshin and others (1964).

In genetic base population CO97, many of the original **PC** and **BC** accessions grew well initially in response to irrigation and weed control on the sandhills site (Morse 2003, Rockwood and others 2001). Overall, **BC** provenances survived best and had the largest trees, while **PC** progenies had the best tree quality. **BC** clones had low survival and the worst tree quality. The range among progeny means was greater than among provenance means, suggesting that more genetic gain could be made by selecting among progenies instead of among provenances. Several **PC** progenies combined good survival with large tree size and desirable quality

characteristics, and individual **BC** and **PC** trees were up to 17 feet tall in 4 years.

At age 19 years, taxa comparisons in the unrogued CO97 changed based on tree data and stand basal area (table 5). Although all four taxa were similar in height, **PC** was smaller in DBH, and **BC** progenies had the poorest quality trees. Seven **BC** provenances had the highest basal area, followed by three **BC** clones and 20 **BC** progenies. Overall, the 19 **PC** progenies were statistically worse than all three **BC** taxa. Still, these **BC** and **PC** comparisons, while updating previous taxa comparisons, are limited for concluding that **BC** outperforms **PC** because this is an atypical site for cypress and representation of **BC** and **PC** is unique.

These observed variations among **BC** and **PC** progenies, as had been noted for cypress elsewhere by Faulkner and Toliver (1983), suggests the potential for genetic gain in each taxon. Liu and others (1990) had noted that **BC** exhibited 95 percent of its genetic variability within populations.

Based on overall accession performances and individual tree attributes, the best 65 trees in CO97 were considerably better than the unrogued trees in three taxa (table 5). The selected trees in each taxon were generally taller, larger in DBH, and much improved in tree quality. The retained 21 **BC** progenies, 7 **BC** clones, and 26 **PC** progenies had 54, 19, and 60 percent, respectively,

higher basal areas than their respective taxa averages before roguing. Due to removing many poorly formed, large trees, the 11 retained **BC** provenance trees actually had 9 percent less basal area. The retained **BC** provenances, **BC** progenies, and **BC** clones were similar in basal area, but the selected trees originating from **PC** progenies were still smaller than all **BC** taxa. Trees of all taxa were notably improved in tree quality, with **BC** progenies improving most to an average quality of 1.3 compared to 3.9 before roguing.

After roguing in January 2017, the distribution of the 65 trees retained across the 1.1-acre site resulted in CO97 having two sections (table 6, figure 1). CO97’s “**BC**” section has 37 trees: 8 from 7 **BC** provenances, 16 from 10 **BC** progenies, 5 of 3 **BC** clones (3 Senators), and 8 from 8 **PC** progenies. The 28 trees in the “**PC**” section are 3 from 3 **BC** provenances, 5 from 4 **BC** progenies, 3 **BC** clones (two Senators), and 17 of 12 **PC** progenies. Collectively, the 36 diverse origins of the 65 CO97 trees are 7 **BC** provenances, 12 **BC** progenies, 3 **BC** clones, and 14 **PC** progenies. Two Senators were previously transplanted in 2013: one to replace the original Senator in Big Tree Park in Seminole County, FL (Babcock 2013) and another to Reiter Park in nearby Longwood, FL (Taylor 2013).

CO97 is now ready to produce commercial improved seed. While limited production of improved seed began with cones produced in Fall 2000 on 5-year-old trees, perhaps due to paclobutrazol treatment (Rockwood and others 2001), CO97’s future production may exceed 400,000 seeds annually, making possible fast-growing, well-formed **BC** and **PC** for mulchwood and/or sawtimber plantations, restoration projects, and urban applications.

Although the estimated genetic gain from CO97 is ~15 percent for stand basal area, CO97 trees [identified by unique accession numbers from 59 to 339 (fig. 1)] will be progeny tested to confirm the realized gain, identify

parents to favor in future collections, and even to rogue CO97 further. A comparison of E1 and E3 seedlots from the “**BC**” and “**PC**” sections of CO97 will document if the two sections influence seedling quality. Progeny tests will also include local, commonly planted **BC** and **PC** seedlots. There’s also potential for vegetatively propagating and testing individual CO97 trees, as is commonly done with cypress elsewhere, particularly China (Creech and others 2011). For example, accession 190 has performed well in clonal trials in Texas (Zhou 2012).

CONCLUSION

These updated results continue to suggest potential for commercial cypress plantations on non-wetland sites in Florida for the production of mulchwood in initial rotations as short as 10 years and/or sawtimber in rotations perhaps as long as 25 years, followed by somewhat shorter coppice rotations. Silvicultural enhancements, notably good site preparation (bedding on flatwoods sites), vegetation control, and nutrition amendments such as compost when and where available, following selection of good quality sites, are essential. Initial spacing influences tree size and stand productivity over time, with close spacing such as 10 x 3 feet limiting tree size and increasing stand basal area without impacting coppice rate. Productivity can also be much further enhanced by using proven genetically superior seedlings from seed orchards and, when economically justified, well-tested clones. At present, **BC** appears more productive than **PC**. Future research includes progeny testing trees in CO97 and documenting response to thinning in SRWC-86.

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Table 5—Number of trees and mean tree height, DBH, quality (Q), and stand basal area by taxon in cypress orchard CO97 before and after roguing at age 228 months

Taxon ^a	Before roguing					After roguing				
	No.	Height feet	DBH inches	Q	Basal area ft. ² per acre	No.	Height feet	DBH inches	Q	Basal area ft. ² per acre
BC Provenances	72	48.3a	11.9a	2.1b	237a	11	51.5a	11.2a	1.7a	215a
BC progenies	134	53.6a	11.4a	3.9a	146b	21	54.9a	11.8a	1.3a	211a
BC clones	22	48.6a	10.5a	3.2ab	178b	7	52.7a	11.2a	1.8a	212a
PC progenies	60	52.0a	6.8b	2.6ab	73c	26	52.1a	8.4b	1.4a	109b
Total	288					65				

^a Taxa means not sharing the same letter for the same trait differ at the 5-percent level.

Table 6—Number of BC and PC accessions retained in the BC and PC sections of cypress orchard CO97

Accession	Number of trees		Total by accession
	CO97 section		
	BC	PC	
BC			
B3	1	1	2
B4	1		1
B5	1		1
B6	2	1	3
B7	1	1	2
B8	1		1
B9	1		1
B11	1		1
B14	2		2
B16		1	1
B19	1		1
B20	2	2	4
B21		1	1
B25	1		1
B26	1	1	2
B27	1		1
B29	3		3
B30	3		3
B31	1		1
E1	3	2	25
E2	1		1
E3	1	1	2
BC Total	29	11	40
PC			
P2	1		1
P3	1	2	3
P4	1		1
P5	1	3	4
P7		1	1
P8		2	2
P9	1	1	2
P10	1	1	2
P11		1	1
P12	1	1	2
P14		1	1
P15		1	1
P16		2	2
P18	1	1	2
PC Total	8	17	25
CO97 Total	37	28	65

		Rows @ 15' Spacing										
		5	6	7	8	9	10	11	12			
Trees @10'	N											N
40			169 B30	206 P2								
39					226 B25	263 P4						
38						262 B19		332 B26				
37						261 B30						
36			167 B6					330 B14	61 E3			
35		127 B3			224 B14							
34				202 B29								
33												
32					59 E1		293 B4			59 E1		
31												
30		123 P18						325 B29				
29				198 B9								
28		121 P5			60 E2							
27					220 B20							
26		119 P3			59 E1			323 B20				
25			156 B11	194 B7				322 B30				
24			155 B6			249 B27	285 B29	321 B31				
23			154 P10									
22				191 P12			283 B5					
21			152 P9	190 B8								
20					59 E1	246 B20						
19					217 P14					343 B26		
18		112 P18				244 B16						
17							278 P10					
16		110 P5						315 B21				
15												
14		108 P3			59 E1	240 B20						
13								312 B7				
12		106 P7			61 E3							
11					213 P16	237 B3						
10		104 P5		179 P12								
9							270 P9			339 P15		
8						234 P3						
7												
6							268 B6					
5					209 P16							
4		99 P5										
3												
2								302 P11				
1			132 P8				264 P8					

Figure 1—Accession number and identification by row-tree location for 65 trees in CO97 after 2017 roguing.

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AN ECONOMIC ANALYSIS OF EVEN- AND UNEVEN-AGED MANAGEMENT IN BOTTOMLAND HARDWOOD FORESTS OF THE LOWER MISSISSIPPI ALLUVIAL VALLEY

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Abstract—A challenge for managers of bottomland hardwood forests is the lack of information about economic tradeoffs among different management approaches. This study evaluated economic tradeoffs, in terms of timber revenue, between even- and uneven-aged management approaches in four common bottomland hardwood forest types in the Lower Mississippi Alluvial Valley (LMAV). Even and uneven-aged management scenarios were simulated using the U.S. Department of Agriculture Forest Service Forest Vegetation Simulator. Data from 107 stands, representing a wide-range of initial conditions, were acquired from the Forest Service Forest Inventory and Analysis (FIA) program. Timber volume outputs under the different scenarios were valued using regional timber price data and evaluated using net present value and equivalent annual annuity measures. As expected, even-aged management generally produced higher timber revenue, but the tradeoff differed among forest types and initial conditions. The magnitude of the tradeoff increased as average diameter increased and was larger for oak-dominated stands. These findings provide guidance to managers and landowners about economic tradeoffs associated with alternative management approaches in common forest types of the LMAV.

INTRODUCTION

The Lower Mississippi Alluvial Valley (LMAV) encompasses the southern extent of the Mississippi River floodplain, an area of nearly 27 million acres extending from southern Illinois to the Gulf of Mexico (Oswalt 2013, Twedt and others 2012). Historically this vast floodplain was covered by bottomland hardwood (BLH) forest; however, agricultural conversion, facilitated by Mississippi River flood control efforts (Stanturf and others 2000), had, by the early 1970s, reduced forest cover to less than 20 percent of its original extent (King and Keeland 1999, Oswalt 2013). A history of fragmentation and high-grading have further degraded its condition (Bowling and Kellison 1983). Increasing concerns about degradation of soil and water quality and habitat loss associated with forest loss prompted significant efforts to improve BLH forest conditions in the LMAV (Stanturf and others 2000). Afforestation of marginal agricultural lands, subsidized primarily through Federal conservation programs under various iterations of the U.S. Farm Bill (i.e., Wetland Reserve Program and Conservation Reserve Program), has managed to restore tree cover on hundreds of thousands of acres

in the region (Gardiner and others 2004, Twedt 2004). At the same time, management of remaining native BLH forests has received increasing scrutiny, with different approaches being advocated depending upon whether timber production, wildlife habitat, soil or water conservation, recreation, or other values are desired.

Evaluating the economic tradeoffs of different forest management approaches in BLH forests is challenging due to complex biophysical conditions and varied management objectives. There are more than 70 different tree species (Putnam and others 1960), which produce structural and compositional diversity, and complex forest stand dynamics (Hodges 1997). Furthermore, the dynamics of past alluvial deposition and frequency and duration of flooding create a wide range of soil and site conditions that affect site productivity and support different forest compositional associations (Hodges 1997). In addition, stand disturbance history creates a range of different stand developmental conditions (Hodges 1997). Ownership of BLH forests is distributed largely among nonindustrial private forest (NIPF) landowners, but also public agencies and

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timberland holding companies. As a consequence, a diversity of management interests exists, and different management approaches are favored depending upon objectives. These typically range from timber production to wildlife habitat to a range of other values (Meadows and Hodges 1997). Timber-oriented management regimes in BLH systems have in the past relied upon selection of individual trees of high value (effectively an uneven-aged selection approach), but have come to favor even-aged management approaches in recent decades as they are seen as more favorable to growing commercially desirable species such as green ash and red oaks (Kellison and Young 1997). Silvicultural systems that are considered most suitable include clearcutting and shelterwood regeneration methods, although group selection may also be possible (Meadows and Stanturf 1997). In contrast, wildlife habitat-oriented approaches tend to prioritize structural diversity, which is considered to be important for many wildlife species, particularly those of high conservation concern such as neotropical migratory birds (Twedt and Somershoe 2009, Twedt and others 2012). Wildlife-oriented forest managers in BLH may thus consider uneven-aged forest management approaches such as single tree or group selection methods more favorable (Meadows and Stanturf 1997), although an array of different multi-aged silvicultural approaches are possible (O'Hara and Ramage 2013).

A significant challenge for landowners and managers in the LMAV is the lack of information about economic tradeoffs among these different management approaches. Recent efforts to manage for enhanced wildlife habitat in BLH forests through partial harvesting and maintenance of continuous cover (e.g., Twedt and others 2012) have raised questions about the economic tradeoffs of alternative management approaches. While recreational values tend to be a high priority among NIPF landowners, the adoption of such novel approaches will often hinge on an understanding of economics. For BLH management this remains an area of great uncertainty for both private landowners and managers alike given the variation in soil types, topography, and stand conditions across the LMAV.

In this study, we evaluated timber revenue tradeoffs of even- and uneven-aged management for a wide range of stand conditions in four common BLH forest types in the LMAV: sweetgum-Nuttall oak-willow oak (*Liquidambar styraciflua*, *Q. texana*, and *Q. phellos*, respectively), overcup oak-water hickory (*Q. lyrata* and *Carya aquatica*, respectively), sycamore-pecan-American elm (*Platanus occidentalis*, *C. illinoensis*, and *Ulmus americana*, respectively), and sugarberry-hackberry-elm-green ash (*Celtis laevigata*, *C. occidentalis*, *Ulmus* spp, and *Fraxinus pennsylvanica*, respectively). We used a modeling approach to simulate forest growth and yield, which was valued using regional timber price data for the Southeastern United States to identify the economically

optimal management regime under even- and uneven-aged scenarios. The objective was to provide landowners and managers with economic information to better evaluate the tradeoffs associated with different silvicultural approaches and thereby allow them to make more informed decisions about management of their BLH forests.

METHODS

Study Area and Stand Conditions

Stand-level data were acquired in 2015 from the Forest Service Forest Inventory and Analysis (FIA) program. The study area was confined to the States of Arkansas, Louisiana, and Mississippi, in which most of the LMAV lies (Sternitzke 1976). Within this study area, bottomland stands designated as Forest Service Ecoregion 234 - Lower Mississippi Riverine Forest Province (Bailey 1995) were identified, and four forest types were selected for the analysis: sweetgum-Nuttall oak-willow oak, overcup oak-water hickory, sycamore-pecan-American elm, and sugarberry-hackberry-elm-green ash forest types (table 1). These dominant forest types represent approximately half of the BLH forest cover in the LMAV (Oswalt 2013). The stand-level data provided by FIA plots covered a wide range of conditions, spanning different site types, stand ages, tree diameter distributions, and stocking conditions. Based on stand characteristics, stands were classified as overstocked (>100 percent), fully stocked (60–100 percent), or understocked (<60 percent) based on the southern bottomland hardwood stocking guide by Goelz (1995). We further classified stands into three site qualities based on sweetgum site index (base age 50): low quality (83 feet), medium quality (99 feet), and high quality (115 feet). Where site index was not provided in the FIA stand-level data, it was estimated based on a regression model that related site productivity class data to site index data (Nepal 2016).

Modeling Approach

We modeled forest stand growth and yield under different silvicultural regimes using the Forest Service Forest Vegetation Simulator (FVS) Southern Variant (Keyser 2008). The FVS is an individual tree, distance-independent growth simulator which can project growth and yield for a wide range of forest types and stand structures and can accommodate a range of silvicultural management scenarios. In the Southern Variant of FVS (FVS/SN), tree growth and mortality relationships are specified for regional conditions. Site index (described above) and regeneration need to be specified by the user. In the modeling scenarios, natural regeneration was supplied manually in FVS/SN after each regeneration treatment (i.e., final harvest or cutting cycle) within each management scenario. For each forest type, the composition and abundance of regeneration were determined by averaging FIA stand-level regeneration data within a forest type (see Nepal 2016).

Table 1—Distribution of forest stands used in Forest Vegetation Simulator (FVS/SN) simulations based on forest type, site quality^a, and stocking conditions^b

Forest type	Site quality	Overstocked	Fully stocked	Understocked	Total
Sweetgum-Nuttall oak-willow oak	High (115 feet)	3	1	1	5
	Medium (99 feet)	4	8	4	16
	Low (83 feet)	6	6	1	13
Overcup-water hickory	High (115 feet)	0	0	1	1
	Medium (99 feet)	8	4	0	12
	Low (83 feet)	5	5	1	11
Sycamore-pecan-American elm	High (112 feet)	1	1	0	2
	Medium (98 feet)	0	5	1	6
	Low (83 feet)	1	1	2	4
Sugarberry-hackberry-elm-green ash	High (110 feet)	0	0	2	2
	Medium (97 feet)	3	6	5	14
	Low (84 feet)	1	12	8	21
Total stands:		32	49	26	107

^a Site quality was determined using Forest Service Forest Inventory and Analysis (FIA) program site index and site productivity class data (see Nepal 2016). Respective site index numbers (dominant height at base age 50) are shown in parentheses above.

^b Stocking conditions were determined based on equations used in the bottomland stocking guide developed by Goelz (1995; fig. 1) using data from Putnam and others (1960).

To constrain the broad spectrum of possible management approaches, we modeled an even-aged scenario and an uneven-aged management scenario (described below). These management scenarios were developed based on published literature and reflect different management paradigms that have been used in BLH management. Furthermore, we chose these management scenarios to capture the extremes of the management spectrum, thereby allowing interpretation at the forest level of mixed approaches that integrate elements of even- and uneven-aged management at the stand level.

In the even-aged management scenario, the existing stands (based on FIA data) were first managed to their financially optimal rotation length to achieve maximum Net Present Value (NPV), based on a management guide (table 2) developed by Goelz and Meadows (1997). Intermediate treatments (i.e., thinnings) were applied to maintain suitable stocking. This type of uniform, even-aged approach is considered economically efficient for commercially desirable hardwood species in the Southern United States (Meadows and Stanturf 1997). After final harvest, a second rotation stand was established using natural regeneration (representing the average densities and composition across the forest type) and was managed through to its financially optimal rotation length. For both rotations, thinning treatments were applied to maintain the stand between the B-line (lower limit) and 100 percent stocking (upper limit), based on the Goelz (1995) BLH stocking guide.

The uneven-aged management scenario followed a “BDq” approach implemented using single tree selection (O’Hara and Gersonde 2004). Target stand conditions for southern BLH forests under the BDq method were determined from Putnam and others (1960). Specifically, their data suggest balanced uneven-aged stand conditions for a managed BLH stand would be achieved with a residual basal area of 68 square feet per acre, maximum tree diameter of 38 inches, and a q-factor of 1.3. These target conditions provide for large trees and structural variability (by diameter class), which represent conditions which may be desirable to wildlife habitat managers (e.g., Twedt and others 2012). Cutting cycles of 5–15 years in length were evaluated to maximize NPV. Natural regeneration was supplied at each cutting cycle, allocated proportionally to the percentage of canopy opening created for regeneration. Each entry removed approximately 20 percent canopy cover, thus 20 percent of the average regeneration for the forest type was provided at each entry (Nepal 2016).

Economic Analysis

Timber yield data (pulp and sawtimber) from each FVS model simulation were valued based on historical hardwood timber price data for the period 2004–2013 for the Southeastern United States reported by TimberMart-South (table 3). Oak sawtimber was valued at \$34.12 per ton, mixed-hardwood sawtimber at \$24.76 per ton, and pulpwood at \$8.43 per ton. Economic outcomes were determined at a 3-percent discount rate. Economic

Table 2—Decisionmaking criteria for even-aged management of BLH forests, adapted from Goelz and Meadows (1997)

Scenario	Stocking conditions	Prescription
Stand <10 years from rotation age		Plan to regenerate when appropriate
Stand >10 years from rotation age	Stocking <100 percent	
	AGS $\geq$ C-10 line	Do nothing
	AGS <C-10 and QMD $\geq$ 16 inches	Consider regeneration
	AGS <C-20 and QMD <16 inches	Consider regeneration
	AGS $\geq$ C-20 and stand >B-line	Consider stand Improvement
	Stand $\leq$ B-line	Do nothing
	Stocking $\geq$ 100 percent	
	AGS >B-line	Thin stand
	AGS $\leq$ B-line and AGS $\geq$ C-10	Timber stand improvement
	AGS <C-10, $\geq$ C-20 and QMD $\geq$ 16 inches	Consider regeneration
	QMD of AGS <16 inches	Timber stand improvement
	AGS $\leq$ C-20	Consider regeneration

AGS = stocking of acceptable growing stock; QMD = quadratic mean diameter; C-10 and C-20 refer to stocking levels below the B-line.

Table 3—Average sawtimber and pulpwood stumpage prices (\$ per ton) by State for the period 2004–2013

State	Sawtimber (\$ per ton)		Pulpwood (\$ per ton)
	Oak	Mixed-hardwood	Hardwood
Arkansas	\$34.39	\$24.18	\$9.16
Mississippi	\$32.00	\$23.27	\$8.29
Louisiana	\$35.98	\$26.82	\$7.83
Average	\$34.12	\$24.76	\$8.43

Timber price data supplied by TimberMart-South.

outcomes under the different management scenarios were estimated in terms of cumulative net present value (NPV), as defined below.

For the even-aged scenario, NPV was determined for the initial rotation of the existing stand, and land expectation value (LEV) was determined for an infinite series of identical rotations starting with the second rotation to calculate LEV (equation 1.1). Cumulative NPV (equation 1.2) for the even-aged management scenario was a summation of NPV from the existing stand and the discounted LEV for the rotations following the initial rotation (fig. 1). These metrics were calculated for a range of possible final harvest ages of the existing stand, in order to achieve the highest cumulative NPV for that stand.

$$LEV = \frac{NFV}{(1+i)^t - 1} \quad (1.1)$$

$$NPV = \frac{NTR + LEV}{(1+i)^k} \quad (1.2)$$

where

LEV = land expectation value for infinite series of identical rotations

NFV = net future value of an identical rotation at year *t*

t = length of rotation

k = number of years remaining in the current rotation

i = interest rate expressed as a decimal

NTR = net timber revenue at *k*th year (monetary value of the conversion period)

NPV = cumulative net present value (monetary value of the conversion period plus *LEV* of future rotations)

For the uneven-aged management scenario, there were two component phases. The initial cutting cycles tended to produce highly variable NPVs, which stabilized over

time (fig. 2). A financially optimal cutting cycle was identified for each stand once this stable (balanced uneven-aged) condition was achieved. NPV of the conversion period harvests was calculated, along with *LEV* (equation 1.3) for the balanced condition assuming average revenue produced in each cutting cycle as perpetual periodic revenue. Cumulative NPV (equation 1.2) for the uneven-aged management was also calculated by summing NPVs from the initial cutting cycles (i.e., conversion period) and discounted *LEV* of the balanced condition.

$$LEV = \frac{R}{(1+i)^t - 1} \quad (1.3)$$

where

LEV = land expectation value of the future managed (balanced uneven-aged) forest

R = net timber revenue received each cutting cycle from future managed forest

t = number of years in the cutting cycle

i = interest rate, expressed as a decimal

The economic tradeoffs between the approaches were assessed by annualizing the returns using an equivalent annual annuity (EAA) (equation 1.4) described by Cafferata and Kemperer (2000). The EAA of the uneven-aged scenario was subtracted from the EAA of the even-aged scenario to produce the EAA difference (equation 1.5). A positive value indicates that the even-aged scenarios outperformed the uneven-aged scenarios, while a negative value indicates that the uneven-aged management scenarios outperformed the even-aged scenarios. Furthermore, the EAA difference can be interpreted as the annualized cost (or benefit) of choosing the less economically optimal approach in \$ per acre per year.

$$EAA = NPV * \text{Discount rate} \quad (1.4)$$

$$EAA \text{ Difference} = \text{Even-aged } EAA - \text{Uneven-aged } EAA \quad (1.5)$$

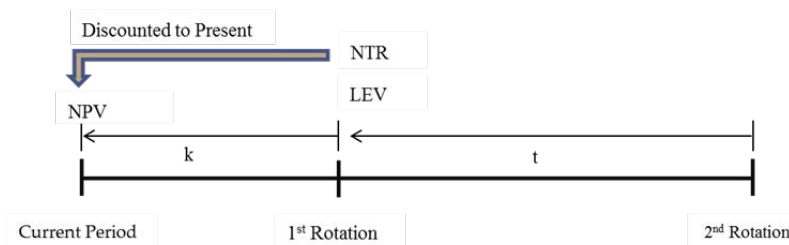


Figure 1—Timeline for cumulative NPV calculation in the even-aged scenario. Cumulative NPV combined the maximized NPV for the first rotation and highest LEV for the second rotation (both discounted to the present).

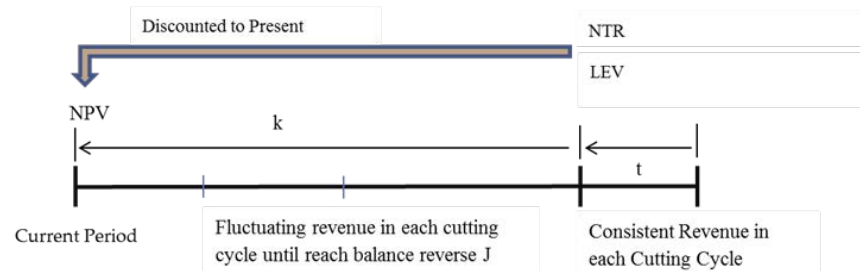


Figure 2—Timeline for cumulative NPV calculation in the uneven-aged scenario. Cumulative NPV combined the NPV for the initial cutting cycles needed to achieve a balanced uneven-aged condition and LEV for subsequent cutting cycles that generated stable revenue (both discounted to the present).

RESULTS

Even-aged Management NPV

Cumulative NPVs increased with higher initial basal area in all forest types. Among the four BLH forest types evaluated, the sweetgum-Nuttall oak-willow oak forest type produced the highest NPVs overall, ranging from approximately \$1,300 per acre to \$9,600 per acre. Trends for the overcup oak-water hickory and sycamore-pecan-American elm forest types were intermediate, ranging from approximately \$1,300 per acre to \$8,300 per acre and from approximately \$1,600 per acre to \$6,100 per acre, respectively. The sugarberry-hackberry-elm-green ash forest type produced lower NPVs compared to the other forest types particularly at higher initial basal areas, ranging from approximately \$700 per acre to \$4,500 per acre (fig. 3).

Uneven-aged Management NPV

NPVs for stands under uneven-aged management followed a similar pattern as stands under even-aged management across all forest types (fig. 4). In most cases, higher NPVs were observed for the sweetgum-Nuttall oak-willow oak forest type, and lower NPVs were observed in the sugarberry-hackberry-elm-green ash forest type. Approximate NPVs ranged from \$800 per acre to \$8,400 per acre for the sweetgum-Nuttall oak-willow oak forest type, \$1,000 per acre to \$7,200 per acre for the overcup oak-water hickory forest type, \$1,200 per acre to \$5,400 per acre for the sycamore-

pecan-American elm forest type, and \$650 per acre to \$4,000 per acre for the sugarberry-hackberry-elm-green ash forest type. Similar to even-aged management scenarios, the NPV trend for the sugarberry-hackberry-elm-green ash forest type did not increase as steeply for stands with higher initial basal areas as compared to other forest types. Overall, NPVs for uneven-aged management were generally lower than in even-aged management.

EAA Tradeoff (\$ per acre per year)

Across all forest types, the EAA difference increased with a larger average diameter (quadratic mean diameter, QMD) (fig. 5). Sweetgum-Nuttall oak-willow oak and overcup oak-water hickory forest types produced higher EAA differences as compared to sycamore-pecan-American elm and sugarberry-hackberry-elm-green ash forest types. Approximate EAA differences ranged from \$3 per acre per year to \$75 per acre per year for the sweetgum-Nuttall oak-willow oak forest type, <\$1 per acre per year to \$46 per acre for the overcup oak-water hickory forest type, -\$2 per acre per year to \$35 per acre per year for the sycamore-pecan-American elm forest type, and -\$15 per acre per year to \$35 per acre per year for the sugarberry-hackberry-elm-green ash forest type. Some stands with lower QMDs in the sugarberry-hackberry-elm-green ash and sycamore-pecan-American elm forest types produced higher EAA difference under uneven-aged management

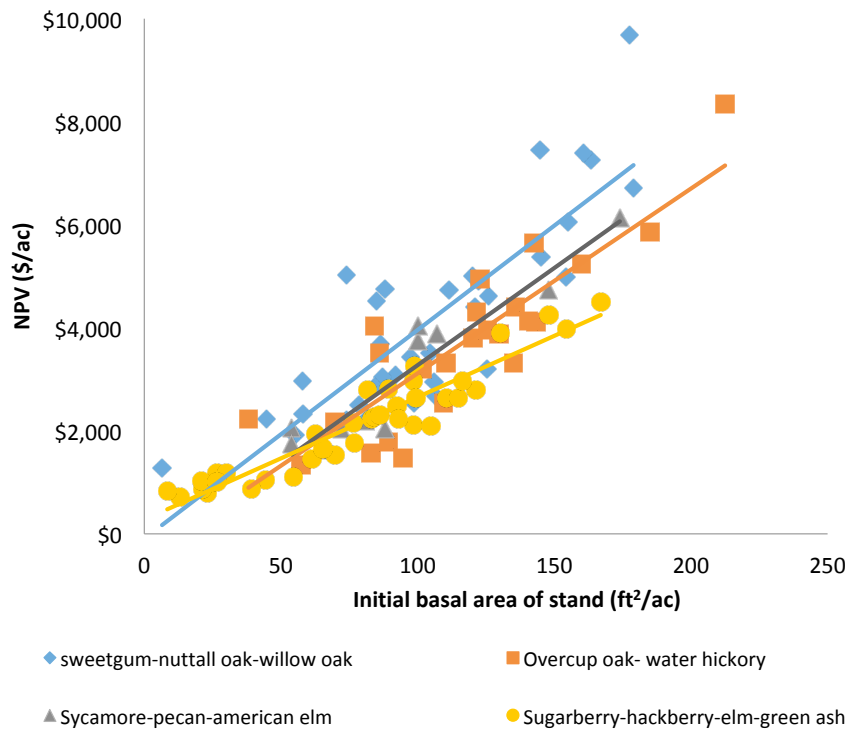


Figure 3—NPVs produced by four different forest types at a 3-percent discount rate under even-aged management.

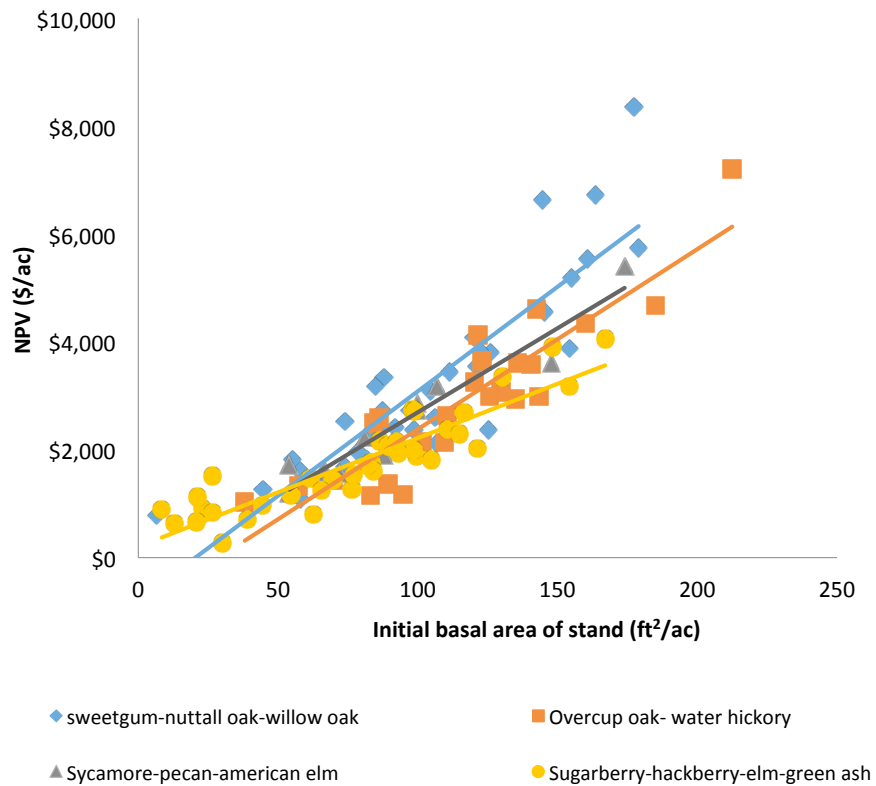


Figure 4—NPVs produced by four different forest types at a 3-percent discount rate under uneven-aged management.

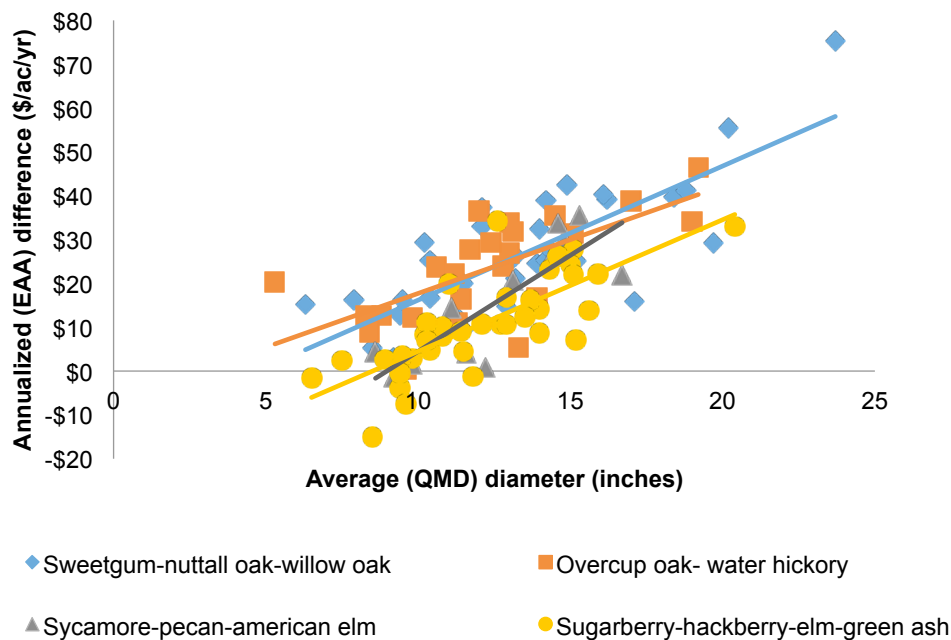


Figure 5—Tradeoff in terms of equivalent annual annuity (EAA) difference (\$ per acre per year) across four forest types at a 3-percent discount rate.

(i.e., negative tradeoffs). Those stands also had lower initial basal areas.

DISCUSSION

This study aimed to quantify economic tradeoffs, specifically in terms of forgone timber revenue, between even- and uneven-aged management in bottomland hardwood forests of the LMAV. The study evaluated a wide range of initial stand conditions. Results from a total of 107 simulated stands for four different forest types in the LMAV showed that even-aged management outperformed uneven-aged management in most cases but not all. However, the magnitude of the tradeoff differed greatly depending on the initial stand conditions. NPVs increased with higher initial basal area in both even- and uneven-aged management because higher basal area stands are ready for harvest sooner in even-aged scenarios or because initial cutting cycles in uneven-aged scenarios yield higher volumes (Nepal and others 2016). The economic tradeoffs in terms of EAA difference were positively correlated with average diameter (QMD) of the initial stand and negatively correlated with the length of rotation of the existing stand in even-aged management. Stands with a higher QMD were likely to perform well under even-aged management, while moderately stocked stands with low or average QMD were likely to produce more competitive NPVs under uneven-aged management.

Among the four forest types, the sweetgum-Nuttall oak-willow oak and the overcup oak-water hickory forest types produced higher NPVs under the even-aged management scenario in all conditions. These forest

types contain and produce higher volumes of higher-valued oak sawtimber resulting in higher NPVs, whereas uneven-aged scenarios produce greater volumes of lower-valued mixed hardwood sawtimber, resulting in lower NPVs (Nepal 2016). That said, several stands from the sugarberry-hackberry-elm-green ash forest type and one stand from the sycamore-pecan-American elm forest type produced higher NPVs under uneven-aged management. Conditions favorable to uneven-aged management occurred in stands of smaller-sized trees (average diameter <10 inches), conditions which were not well represented in the dataset for the oak forest types. It is possible that this more favorable performance of uneven-aged management in stands of smaller-sized trees is a more general phenomenon across forest types; unfortunately, there were limited stand-level data to assess this.

This economic analysis is based solely on valuing timber yields. Management costs were not considered, nor were price premiums for higher value products or price reductions for low quality, all of which could affect the economic performance under even- and uneven-aged management. Our analyses were based on long-term price trends and future product price dynamics could alter these outcomes. Future shifts in sawtimber values could thus alter these outcomes, either narrowing or accentuating the tradeoffs. Likewise, assumptions related to the growth and yield model, particularly regeneration and stand structural targets could affect economic tradeoffs. Sensitivity analyses have suggested that outcomes are not substantially affected by regeneration inputs (Nepal and others 2017). However,

tradeoffs may be sensitive to target stand conditions prescribed in the uneven-aged BDq approach, with a higher q-factor and smaller maximum diameter limit improving the relative performance of uneven-aged management (Nepal and others 2017). Shifts in species composition will also affect economic outcomes. These aspects need further investigation. Finally, future research should also address costs associated with specific forest management practices, and non-timber values such as watershed and wildlife habitat, which are increasingly high priority concerns for BLH forests in the LMAV.

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PERFORMANCE OF EASTERN COTTONWOOD AND HYBRID POPLARS ON ALLUVIAL AND UPLAND SITES IN THE SOUTH

Randall J. Rousseau, Landis B. Herrin, and Oludare S. Ogunlolu

Abstract—Continued emphasis on woody biomass production under short rotation woody crop strategies has focused on both hybrid poplar and eastern cottonwood. The advantages of hybrid poplars in comparison to eastern cottonwood include superior rooting, better wood properties, and the ability to grow well on upland sites. Unfortunately, one specific disease, *Septoria musiva*, which results in stem canker and mortality, has shown to be the most serious impediment to the use of hybrid poplars in the Southern United States. Although disease still ranks as a significant problem in the Southern United States, a limited number of hybrid poplar clones have shown tolerance and changed the thinking about the use of hybrid poplars. In addition, a small population of eastern cottonwood clones has shown high survival and good growth on upland soils. New selections within parent populations may provide an even greater ability to develop more hybrid populations suited to the environment of the Southern United States.

INTRODUCTION

Woody biomass programs in the Southeastern United States have focused primarily on fast-growth hardwoods, which have included *Populus* species and their various hybrids, American sycamore (*Platanus occidentalis* L.), sweetgum (*Liquidambar styraciflua* L.), and Eucalyptus (*Eucalyptus* spp.) as well as loblolly pine (*Pinus taeda* L.). Eastern cottonwood (*Populus deltoides* Bartr. ex Marsh.) has a long history of research efforts in the Mississippi Alluvial Valley (MAV) beginning with the U.S. Forest Service located in Stoneville, MS where collections and testing resulted in genetically superior clonal selections that were used by numerous pulp and paper companies located along the lower portions of the Mississippi River (Stettler and others 1996). Eastern cottonwood, like most hardwood species, is very site-specific and attains its best growth on newly deposited alluvial soils that possess high fertility, good moisture availability during the growing season, and a lack of restrictive layers. In the past, eastern cottonwood genetic programs have been developed for such sites, with little testing on upland soils. There is no doubt that, when planted on alluvial sites along the Mississippi River, eastern cottonwood demonstrates very rapid growth reaching harvestable pulpwood size of approximately 10 inches at diameter at breast height (DBH) and total height of 80 to 90 feet within 8 to 10 years. Although rapid growth is the most positive aspect of eastern cottonwood, the species does not possess extremely

good rooting, which at times has led to overall survival problems since dormant unrooted cuttings are the desired planting stock. These survival problems have shown up when the species has been either moved offsite or deployed under stressful environments.

The Stoneville program took a cursory look at hybrids, but for the area along the MAV, the various hybrid taxa did not fare well, thus eliminating them from further research efforts. In the late 1980s, a limited number of F₁ hybrid poplar clones resulting from the mating of *P. deltoides* and *P. trichocarpa* and these F₁ hybrids backcrossed to eastern cottonwood, developed by the University of Washington and Washington State University, were tested on sites within the MAV. However, all of the clones tested quickly succumbed to *Septoria musiva* stem canker and were eliminated from any further testing in the MAV. It wasn't until the mid-1990s that hybrid poplars were again examined in the South, but in this case the various taxa were being tested on upland sites. The approach at that time was to introduce clones that were performing well in the Pacific Northwest to upland sites in Kentucky, Tennessee, Virginia, and West Virginia. These clones were primarily *P. trichocarpa* x *P. deltoides* taxon (TD), but also included DN34, a *P. deltoides* x *P. nigra* hybrid, and NM6, a *P. nigra* x *P. maximowiczii* hybrid, with the latter two hybrid clones supposedly resistant to *Septoria* stem canker. Results indicated that certain hybrid clones

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from a variety of taxa showed excellent first-year growth and survival on these sites. However, after 9 years the trees again succumbed to Septoria. Additional testing in western Kentucky of a variety of *Populus* hybrids in 1999 examined 10 clones from each of five taxa, which included TD taxon (*P. trichocarpa* x *P. deltoides*), TDD taxon (*P. trichocarpa* x *P. deltoides* hybrid backcrossed to *P. deltoides*), TN taxon (*P. trichocarpa* x *P. nigra*), TM taxon (*P. trichocarpa* x *P. maximowiczii*), and TMM taxon (*P. trichocarpa* x *P. maximowiczii* hybrid backcrossed to *P. maximowiczii*). This new material was the first of its kind in the mid-South area. Differing results clouded the issue as Newcombe and Ostry (2001) stated that TD F₁ hybrids were uniformly susceptible to Septoria stem canker, yet Rousseau and others (2008) showed variable Septoria resistance among clones within the TD F₁ hybrids tested in Kentucky. However, Rousseau and others (2008) identified two of the 10 TD F₁ hybrids that exhibited high survival, good growth, and disease resistance.

METHODS

The 2010 Consolidated Populus Feedstock Trial was a joint effort under the Sun Grant Initiative by the University of Minnesota, Mississippi State University, ArborGen, and GreenWood Resources. All three groups have a history in *Populus* development in various geographic areas throughout the United States. The initial test of this program focused on each group contributing 20 selected clones and each group establishing at least one site in 2010 and one site in 2011. Mississippi State University requested enough clonal material from each group to establish two test sites over the 2-year period, with one to be located on a Mississippi River alluvial site and the second on an upland site in northeast Mississippi approximately 90 miles east of the MAV. The alluvial test site was located in New Madrid County, MO while the upland site was located in Pontotoc County, MS. The experimental design employed was a nested design with three blocks, with clones nested within source in each of the three blocks. Clones were arranged in two-tree row plots and planted at a spacing of 9 by 6 feet with dormant unrooted cuttings. Prior to planting, both test sites were disked and then subsoiled to a depth of 14 inches. The alluvial site was planted April 10, 2010, and the upland test site planted on April 8, 2010. Immediately after planting of each respective test site, the test was treated with a broadcast application of Goal® 2XL at a rate of 64 ounces per acre. Additional herbaceous control during the first year was accomplished by a combination of chemical and mechanical means.

The 2010 Consolidated Populus Feedstock Trial consisted of 80 clones, of which 36 were eastern cottonwood clones and 44 were hybrid poplars. The hybrid clones represented a variety of taxa including

P. deltoides x *P. nigra* (DN), *P. deltoides* x *trichocarpa* (DT), *P. deltoides* x *P. maximowiczii* (DM), *P. trichocarpa* x *P. deltoides* (TD), *P. nigra* x *P. maximowiczii* (NM), and *P. trichocarpa* x *P. nigra* (TN). Both the NM and TN taxa were eliminated from the analysis because each was represented by only a single clone. Both trials were measured on an annual basis with total height measured at age 1 and total height, diameter, and occurrence of stem disease (i.e., Septoria stem canker) measured at ages 2 through 5. Volume outside bark was computed using equations developed by Mohn and Krinard (1971) for small cottonwood in plantations:

$$\text{Total volume outside bark} = 0.21099 + 0.00221(D^2H) \quad (1)$$

where

D = diameter at breast height

H = total tree height

RESULTS

Survival

Age 1 survival of the alluvial test site (table 1) was 7.5 percent lower than that of the upland site and remained lower through the following 4 years. Age 1 survival of the alluvial site was 76.7 percent but dropped each year. Mortality at ages 2, 3, and 4 was between 2.7 and 4.5 percent, however mortality between ages 4 and 5 increased to 21.2 percent lowering overall test survival to 44.9 percent. In comparison, survival of the upland test site (table 2) was 84.2 percent at age 1 and showed a much slower decline in mortality, with survival dropping to 81.0 percent at age 5. Survival was relatively the same at ages 1, 2, and 3 at 84 percent. At age 4, survival dropped only 1 percent and only 2 percent between the ages of 4 and 5.

Examination of survival by taxon provides a more in-depth look at potential contributing factors. On the alluvial site, survival of eastern cottonwood at 64.4 percent was much lower than the hybrid taxa which ranged in survival between 90.3 and 80.6 percent. However, almost immediately, the hybrid taxa survival dropped at age 2 and continued to drop through age 5. The most significant mortality was in the DM taxon where survival dropped from 80.6 percent at age 1 to 69.5, 58.4, 41.7 and 5.6 percent for ages 2, 3, 4, and 5, respectively. By age 5, survival of the DN, DM, DT, and TD hybrid taxa was 36.1, 5.6, 61.1, and 13.0 percent, respectively. The DN, DT, and TD hybrid taxa showed an age 4 survival that was still in the 70-percent range but dropped dramatically for the DN and the TD taxa. The DT taxon dropped the least between ages 4 and 5 but still fell nearly 17 percent. Survival among the taxa

Table 1—Survival, total height, and volume by taxon at ages 1, 3, and 5 years when planted on the alluvial site in New Madrid County, MO

Taxon ^a	Survival percent			Total Height feet			Volume cubic feet	
	Year 1	Year 3	Year 5	Year 1	Year 3	Year 5	Year 3	Year 5
DD	64c	64d	64a	13.3a	27.1a	42.0a	0.9606a	3.0457a
DM	81b	58e	6d	12.7a	22.6b	28.3b	0.5325b	1.0920b
DN	90a	81a	36b	10.4c	20.8c	30.2b	0.3944c	1.1880b
DT	83b	78b	61a	11.5b	21.1c	30.6b	0.5311b	1.4804b
TD	83b	74bc	13c	10.3c	18.8d	27.8b	0.3053c	0.8157b

^a DD = eastern cottonwood; DM = *P. deltoides* x *P. maximowiczii*; DN = *P. deltoides* x *P. nigra*; DT = *P. deltoides* x *P. trichocarpa*; TD = *P. trichocarpa* x *P. deltoides*.

Note: survival, total height, and volume means with the same letter are not significantly different at $p < 0.05$.

Table 2—Survival, total height, and volume by taxon at ages 1, 3, and 5 years when planted on the upland site in Pontotoc County, MO

Taxon ^a	Survival percent			Total Height feet			Volume cubic feet	
	Year 1	Year 3	Year 5	Year 1	Year 3	Year 5	Year 3	Year 5
DD	71d	71d	69d	5.0d	17.5b	28.1a	0.2623b	0.8616a
DM	92b	92bc	78c	8.4a	23.2a	28.9a	0.4371a	1.0268a
DN	98a	98a	92b	6.6bc	14.3c	19.8c	0.1561c	0.2329c
DT	94b	94b	94a	6.7b	18.6b	27.1ab	0.2553b	0.8515a
TD	91b	91b	91b	6.0c	16.8b	24.8b	0.2315b	0.6152b

^a DD = eastern cottonwood; DM = *P. deltoides* x *P. maximowiczii*; DN = *P. deltoides* x *P. nigra*; DT = *P. deltoides* x *P. trichocarpa*; TD = *P. trichocarpa* x *P. deltoides*.

Note: survival, total height, and volume means with the same letter are not significantly different at $p < 0.05$.

on the upland site was similar to that observed on the alluvial site, with survival of eastern cottonwood being much lower than all of the taxa included in the group of hybrid poplars. Age 1 survival of eastern cottonwood was 71.3 percent and remained fairly constant through age 5. The hybrid taxa on the upland site did not exhibit a significant change in survival through time as observed on the alluvial site. Survival of the TD and the DT hybrid taxa was 90.7 and 94.4 percent, respectively, at age 1 and remained the same through age 5. Age 1 survival of the DN taxon was 97.9 percent. Survival remained constant at ages 1, 2, and 3 and dropped slightly at age 4 to 96.4 percent and at age 5 to 93.8 percent. Similar to the survival of the DM taxon observed on the alluvial site, this taxon was again the most impacted, as survival dropped from 91.7 percent at age 1 to 77.8 at age 5.

Total tree height was significantly different between the two sites and, as expected, the alluvial site was taller for all 5 years. Differences increased from a low of 5.9 feet at age 1 to a high of 12.3 feet by age 5. Height growth averaged approximately 7 feet per year, with the exception of the third year which was 3.5 feet. This reduced growth was during a severe drought. Height growth of the upland site was the highest at age 1 but continued to drop each year resulting in growth of only 3.8 feet during the fifth growing season. Diameter and volume followed the trend shown by height, where the alluvial test site exhibited much better growth. The diameter and volume of the alluvial site at age 2 were similar to that of age 5 of the upland site.

When examining the difference between eastern cottonwood and hybrid poplars on both sites at ages 1, 3, and 5 years, the results indicate that eastern cottonwood and hybrid poplars are very different when planted on an alluvial site as compared to an upland site. Height differences between eastern cottonwood and hybrid poplars on the alluvial site at ages 1, 3, and 5 are 2.4, 4.7, and 12.7 feet, respectively. By age 5, eastern cottonwood volumes were nearly three times that of the hybrid poplars on the alluvial site. On the upland site, the eastern cottonwood group of clones was 1.9 feet shorter than the hybrid poplar group. By age 2, this was reversed with the eastern cottonwood group being a foot taller than the hybrid poplar group, and this difference was enlarged by age 5 to 5.3 feet. Volume differences followed the same trend as seen in height with the largest differences observed at age 5, where the eastern cottonwood group (0.8314 cubic feet) was larger than the hybrid poplar group (0.5244 cubic feet).

Breaking this down to the taxon level provides a clearer picture of performance. Rather than looking at each year, the results are shown at ages 1, 3, and 5. The results from the alluvial site show that eastern cottonwood was significantly better than the hybrid taxa for all traits exhibited for early age (i.e., ages 1 and 3) except for survival (table 1). The results from the upland site showed that survival of the hybrid taxa was significantly better than eastern cottonwood (71 percent), with DN being the highest at 98 percent, followed by DT, DM, and TD taxa (table 2). The DM taxon was significantly taller than the other taxa at ages 1 and 2, but by age 5 both eastern cottonwood and the DT hybrid taxon

were very similar. The DN hybrid taxon was significantly shorter with lesser volume than the other taxa. The DM taxon exhibited higher volume at age 3 than eastern cottonwood and all of the other hybrid taxa. By age 5, the DM taxon ranked number one for volume but was not significantly different than the DT hybrid taxon and eastern cottonwood.

The top 10 percent of the test population is represented by eight clones regardless of taxa. The age 5 alluvial test means for DBH, height, and volume were 4.8 inches, 37.0 feet, and 2.352 cubic feet, respectively. The top 10 percent of the alluvial test population exhibited DBH, height, and volume of 5.7 inches, 44.0 feet, and 3.534 cubic feet, respectively. Examination of clones within taxa at age 5 showed that, for the alluvial site, the top eight volume-producing clones were all eastern cottonwood clones (table 3). Clone 414 displayed exceptional growth performance through age 5, with a mean diameter, height, and volume of 7.1 inches, 45.8 feet, and 5.403 cubic feet, respectively. Comparing clone 414 to the next highest ranking clone (i.e., 412), the differences were that 414 was an inch larger in diameter, almost 3 feet taller, and 1.627 cubic feet larger. The age 5 test means for the upland site were considerably lower than the alluvial site for diameter, height, and volume at 2.7 inches, 24.7 feet, and 0.668 cubic feet, respectively. In comparison, the top 10 percent of the test population showed mean diameter, height, and volume of 4.4 inches, 33.9 feet, and 1.496 cubic feet, respectively. Further evaluation of the top eight clones showed that only two clones were hybrids, with the remaining six being eastern cottonwood. The best clone on the upland

Table 3—Survival (SUR), DBH, total height (THT), and volume (VOL) of the top 10 percent of the test population for the alluvial site located in New Madrid County, MO and the upland site located in Pontotoc County, MS

Alluvial					Upland				
----- Age 5 -----					----- Age 5 -----				
Taxon - Clone	SUR %	DBH <i>in.</i>	THT <i>ft.</i>	VOL <i>ft.³</i>	Taxon - Clone	SUR %	DBH <i>in.</i>	THT <i>ft.</i>	VOL <i>ft.³</i>
DD - 414	83	7.1	45.8	5.403	DM - 8019	100	4.5	37.5	1.926
DD - 412	83	6.1	43.0	3.776	DD - 443	83	4.4	35.1	1.800
DD - 66	83	6.0	45.1	3.714	DD - 171	100	4.1	34.4	1.482
DD - 71	100	5.4	45.0	3.334	DD - 142	100	4.2	33.8	1.437
DD - 142	100	5.5	45.4	3.180	DD - 412	83	4.1	33.6	1.376
DD - 31	100	5.5	43.4	3.033	TD - 2128	100	4.0	31.4	1.360
DD - 151	83	5.2	41.0	2.919	DD - 151	100	3.7	31.1	1.336
DD - 184	83	5.3	43.0	2.912	DD - 31	83	3.7	34.2	1.324
Test Mean	45	4.8	37.0	2.352	Test Mean	81	2.7	24.7	0.668

^a DD = eastern cottonwood; DM = *P. deltoides* x *P. maximowiczii*; TD = *P. trichocarpa* x *P. deltoides*.

site was a hybrid poplar 8019, which is a DM hybrid that exhibited means for diameter, height, and volume of 4.5 inches, 37.5 feet, and 1,926 cubic feet, respectively.

DISCUSSION

The fact that the eastern cottonwood outperformed the hybrid poplars on the alluvial test site was expected as previous testing indicated, however it was hopeful that a few of the newly developed hybrids would be able to withstand the formation of *Septoria* stem cankers (Coyle and others 2006, Kaczmarek and others 2013, Rousseau 2014). Based on age 1 survival of both sites, it was also very apparent that the rooting characteristics of the various hybrid poplars are superior to that of eastern cottonwood (May 2012). But, because of hybrid poplars' disease susceptibility, especially to *Septoria*, survival at the alluvial site was severely impacted through age 5. The survival of the hybrid poplars at the upland site was not impacted by *Septoria* until after the third growing season, and even at that time the clones included in the DM taxon were the most affected dropping from 92 percent survival to 78 percent by age 5. Even then, the DM clone 8019 remained at 100 percent survival at age 5, providing hope that within the most affected taxon there is some resistance level to *Septoria*. The level of *Septoria* present on upland sites is probably not to the level found in the MAV where it occurs naturally and where eastern cottonwood is found in large stands. During the 5-year time frame of the upland site, *Septoria* seems to be building, and susceptibility of clones has become more apparent, but the question remains if there are races within the disease and, if so, how much variability in virulence exists among the various races.

Examining the top 10 percent of the 80 clones tested at each site showed that neither the eastern cottonwood clone (414) nor the DM hybrid clone (8019) was among the top 10 percent of the both tests. However, eastern cottonwood clones 142 and 412 were among the top five highest ranking clones for both test sites. Additionally, eastern cottonwood clones 31 and 151 also ranked in the top 10 percent of the population for both test sites. Growth of the taxa and clones can be explained in a couple of ways. A number of clones in the DN taxon were bred and selected for the cooler climate found in Minnesota. Thus, the northern DN hybrid clones are adapted to a much milder environment than either Mississippi or Missouri. It was apparent that these northern hybrid clones were poorly adapted to the longer, warmer growing seasons, especially on the upland site in Mississippi. The clones with the TD, DT,

and DM taxa were bred and selected for growth in the Pacific Northwest, where *Septoria* is not indigenous. Thus, clones within these taxa showed variability in tolerance to the disease. The performance of the eastern cottonwood clones on the upland site was unexpected; generally, eastern cottonwood has not performed well on sites outside of the alluvial area of the Mississippi River. Although growth was suitable on the alluvial site, this area cannot be classified as a marginal site; however, because of historic flooding followed the next year by a severe drought, all of the clones were greatly impacted. The upland site certainly fits the characteristics of a marginal site as it was originally in cotton production but was deemed too poor for that purpose and has been, for a considerable period, in pasture. The growth of the best clones on the upland site was less than half of the volume produced on the alluvial site. This production needs to be higher. The key to better growth on upland sites appears to be a combination of clones of specific taxa that exhibited hybrid vigor, excellent rooting, and increased *Septoria* resistance. Based on this trial, the hybrid poplars belonging to the DT and DM taxa seem to hold the most promise for use in the Southern United States. However, there is a critical need for continued examination of hybrid poplars of both taxa based on growth and disease resistance prior to planting recommendations being finalized. As seen in most *Populus* tree improvement programs around the world, the use of eastern cottonwood as a mainstay in breeding efforts is critical and thus the need for moving forward with a recurrent selection program should be a priority.

SUMMARY

The 2010 Consolidated *Populus* Feedstock Trial provided a view into some of the newest eastern cottonwood and hybrid poplar clones to determine possible use as biomass clones in the Pacific Northwest, Midwest, and Southeastern United States. The two test sites chosen by Mississippi State University represented a site where eastern cottonwood should perform well but is known to have diseases that will be challenging for hybrid poplars and a site where nutrient and moisture levels will be challenging for eastern cottonwood. The results of the study again indicate that hybrid poplars are a very poor choice for any site where the *Septoria* level is high such as in the MAV area. In addition, there is no doubt that upland sites will be challenging in nutrients and moisture. On the positive side, this test indicated that even the most susceptible and quite possibly the fastest-growing taxa showed disease resistance variability among clones.

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HYBRID SWEETGUM RESPONSE TO OUST® XP AT DIFFERENT APPLICATION TIMES FOR PRE-EMERGENT COMPETITION SUPPRESSION

Robert Hane, Joshua Adams, and Michael Blazier

Abstract—Hardwood plantation success can be greatly influenced by competition control. Hybrid sweetgum (*Liquidambar formosana* x *styraciflua*) is an emerging option for short-rotation plantations, but herbicide efficacy is unknown for these new varieties. The hybrids' dormancy timing may make them more vulnerable to pre-emergent herbicide applied at conventional times. In this study we assessed the timing effect of sulfometuron methyl (Oust® XP) application on survival and growth of several hybrid sweetgum varieties at two sites in northern Louisiana to help guide herbicide application timing decisionmaking for these newly available varieties. Four treatments were randomly assigned including no-herbicide control and mid-winter, late winter, and early spring herbicide applications. Overall, 94 percent of the seedlings survived the first growing season. The conventional (late winter) herbicide treatment produced significantly more height growth than early spring and mid-winter treatments, but was not significantly different from the control group.

INTRODUCTION

In Louisiana, 322.5 million cubic feet of pulpwood were processed at pulpwood mills and accounted for about 10.7 percent of all pulpwood production in the Southeastern United States in 2009 (Johnson and others 2011). Most products provided by the pulpwood industry require a mixture of hardwood and softwood pulp in various ratios. The majority of hardwood pulp for this industry in Louisiana comes from the bottomland hardwood forests prevalent throughout the region (Bentley and others 2005). A downside to relying on bottomland hardwood pulp is that these areas can be inaccessible to logging equipment for large portions of the year due to soil saturation or flooding. Due to this uncertainty, many mills will stockpile hardwood material to prevent shortages that could prevent a mill from operating (Kang and Morrell 2000). These stockpiles are expensive to maintain and vulnerable to a host of dangers including insect damage or fungal and bacterial degradation rendering a log unusable to the mill (McMillen and Wengert 1977).

An alternative to stockpiling is to use hardwood species that are adapted to growing on upland sites that are available for longer periods through the year when bottomlands may be inaccessible. Unfortunately, many of the tree species that have adapted to upland sites

may not have the same wood quality or growth rates as their bottomland counterparts. A new hybridization of species may provide a hybrid vigor to address these needs. Faster growing or hardier hybrids may capture more growth on lower quality upland sites and reduce the mortality rate of a planting operation. Sycamore (*Plantanus occidentalis*) has been researched to fill this niche, and many species within genera have been successfully hybridized for this purpose, among them eucalyptus (*Eucalyptus* spp.), poplar (*Populus* spp.), and now sweetgum (*Liquidambar* spp.). Each of these previous trials found limitations, cold tolerance, water needs, and disease vulnerability being the most concerning. Most recently, hybrid sweetgum has emerged as a new option to produce pulpwood on an upland site because of its enhanced growth capabilities as well as its increased wood density and wood quality (Kaczmarek and others 2012). However, research on these hybrids is only in its initiation phases. Indeed, an initial concern is how these hybrids will perform within the target planting range (e.g., north Louisiana) and how they will respond to competition control. Competition control has proven to be a vital part of hardwood plantation management, and now with high-intensity short rotation systems designed to produce pulpwood volume or biomass, competition control is even more critical (Alig and others 2000, Bey and others 1976).

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The responsiveness to competition control measures can play a part in the decisionmaking process of land managers when they are deciding what species to plant (Erdmann 1967). There is complexity in competition control decisionmaking for short rotation hardwoods due to the relative lack of herbicides labeled for broad-spectrum weed suppression that can be applied over actively growing hardwoods. As such, herbicides such as sulfometuron methyl [Oust® XP (DuPont; Wilmington, DE)] that provide such suppression are applied for pre-emergent weed control over dormant hardwoods. Oust® XP is a broad-spectrum herbicide registered for use over hardwood seedlings in low concentrations during dormant season, before bud swell. Oust® XP is soil-activated and works to kill existing vegetation as well as functions as a pre-emergent weed control chemical to continue controlling herbaceous weedy vegetation into the future (Seifert 1993). Pre-emergent applications, typically conducted in late winter, are complicated by weather conditions (notably long periods of rain) that sometimes delay applications until early spring, when hardwoods are beginning to break dormancy (Whitcomb 1999). Another potentially complicating factor for pre-emergent applications to hybrid sweetgum is the hybrid's tendency to maintain quiescent foliage until shortly before bud swell. The tolerance of hybrid sweetgum to sulfometuron methyl is relatively undetermined.

A study was installed at Louisiana Tech University and Louisiana State University (LSU) Ag Center Hill Farm Research Station in northern Louisiana to determine the growth and survival of five hybrid sweetgum varieties in response to sulfometuron methyl application at different times between late winter and early spring. Thus, this study was designed to test the tolerance of each hybrid sweetgum variety to sulfometuron methyl and to observe whether the efficacy of herbicide treatment varies when it is applied during dormancy and into the growing season for competition control.

METHODS

Two sites were used for this study. One was on Louisiana Tech campus (i.e., Louisiana Tech site) and the other was at LSU Ag Center Hill Farm Research Station (i.e., Hill Farm site). The Louisiana Tech study site is largely on an Angie fine sandy loam soil, with a small portion on a Sacul very fine sandy loam, while the Hill Farm site is located on a Darley-Sacul complex, with Darley being on the ridgetop and down the hillside and Sacul being predominant in the bottom below the hill. The Louisiana Tech site was used for horse and cattle hay production since 1990. During this time, horse manure was spread over the field occasionally as fertilizer. Prior to 1990, the Louisiana Tech study site was the interior of a racetrack going back to the late 1970s, during which time Christmas trees were periodically grown on the site. In 2014, the Hill Farm study site was mulched

to remove vegetation left from two failed loblolly pine (*Pinus taeda*) plantings in 2009 and 2010. Prior to these failed plantings, the site had been used for loblolly pine research since the 1960s. In the summer of 2015, the site was fertilized with 150 pounds per acre of diammonium phosphate (DAP). The Louisiana Tech study location did not receive any fertilizer treatment prior to planting because of the site's history of manure fertilization (Scott and others 2004). In preparation for planting, both sites were subsoiled to a 24-inch depth in late summer prior to planting. One week before planting, 3 quarts per acre (2.84 L per acre) of glyphosate were applied [Accord® XRTII (Dow; Indianapolis, IN)] via backpack sprayer to remove any herbaceous vegetation present.

The herbicide tested in this study was sulfometuron methyl (Oust® XP). Oust® XP was applied in a 36-inch wide band using a boom sprayer attached to a tractor at 2 quarts per acre directly over the seedlings. The rate of 2 quarts per acre of Oust® XP was selected in accordance with recommendations for sweetgum based on prior studies (Kushla and Self 2013). Each row was placed into one of four application timing treatment groups. Bud condition was used to separate the treatments. A control group that received no herbaceous weed control was one of the four treatments. In the earliest herbicide application (winter) on January 29, buds were completely dormant. The second application (recommended treatment) was applied on February 17 as the buds began to swell due to the unusually warm winter north Louisiana experienced early in 2016. This middle treatment was recommended by the label of Oust® XP as the best time to apply herbicide without tree mortality while maintaining adequate herbaceous weed control. Our final treatment group (late treatment) received herbicide on March 5 after many of the buds had already broken dormancy, marking the beginning of the growing season.

Both sites were hand-planted in November 2015 with 1-1 sweetgum (*Liquidambar* spp.) varieties provided by ArborGen Corporation, from their nursery located in Shellman, GA. Sweetgum varieties tested included five clonal hybrid sweetgum (*Liquidambar formosana* x *styraciflua*) varieties, four of which are currently available commercially. The varieties currently available are "AGHS 1" through "AGHS 4" in our study. The treatment structure for the study was a split-plot design, with herbicide timing as the whole-plot treatment and variety as the subplot treatment. All treatment combinations were replicated three times at the Hill Farm site and five times at the Louisiana Tech site. Plots were 3 feet by 40 feet in size. Each row of the study had eight trees from all genotypes tested randomly blocked along its 200-foot length. Within each row, there were buffer trees separating the study trees of each genotype from other

groups. The first and last trees of each genotype were considered buffer trees and were removed from the dataset before statistical analyses were conducted.

Height, groundline diameter (GLD), and survival were measured in December 2015. Survival was again measured in July, September, and October of 2016. Height was measured again in July and October of 2016, while GLD was only measured in October of 2016. Height (cm) was measured using a meter stick from the ground level up to the highest living bud present on the seedling. Groundline diameter (mm) was measured with a caliper held at ground level unless roots were exposed above the ground in which case the groundline diameter was taken at the root collar. Growth of individual trees was determined as the difference in height and GLD between December 2015 and all subsequent measurement periods.

Analysis

Plot means of height growth and groundline diameter growth were analyzed independently using a split-plot analysis of variance (ANOVA). Each replication of the main plot containing herbicide treatment was split into five genotype subplots. For both dependent variables, growth was calculated as the difference between the end of the growing season and initial measurements. Once growth of height and groundline diameter were calculated for each tree, all study trees in each plot were averaged together with border trees removed from the dataset before plot averages were created.

Plot means were used in the analysis for significance using independent fixed-variables of location, genotype, and herbicide treatment and all possible interactions among these factors. When significant ($p < 0.05$) terms were found, an F-protected Fisher's Least Significant Difference (LSD) method was used for mean separation.

RESULTS AND DISCUSSION

Survival

Across all herbicide treatments at both locations, 93.9 percent of seedlings survived overall. At Hill Farm, nine trees died before herbicide was applied and another 10 trees before the middle of the summer. At Louisiana Tech, every study tree survived past the mid-July tally, but 24 trees died during the second half of the growing season. By the end of the growing season, 16 more trees had died at Hill Farm, bringing the total to 35 at Hill Farm (fig. 1). Almost half of the mortality occurred in the winter (January) herbicide treatment rows. Closely following the January application of herbicide was a severe rain event across north Louisiana that dropped over 24 inches of rainfall in a 24-hour period. Saturated soil may have increased the efficacy of Oust® XP beyond the intended rate and influenced tree survival (Harvey and others 1985). The lowest mortality across herbicide treatments was observed in the no-herbicide control plots (fig. 2). Hybrid sweetgum variety AGHS3 was the genotype most likely to die across the genetic varieties tested; 24 trees of this variety died, while the next highest mortality count was AGHS2 with 13 trees not surviving through the first growing season (fig. 3).

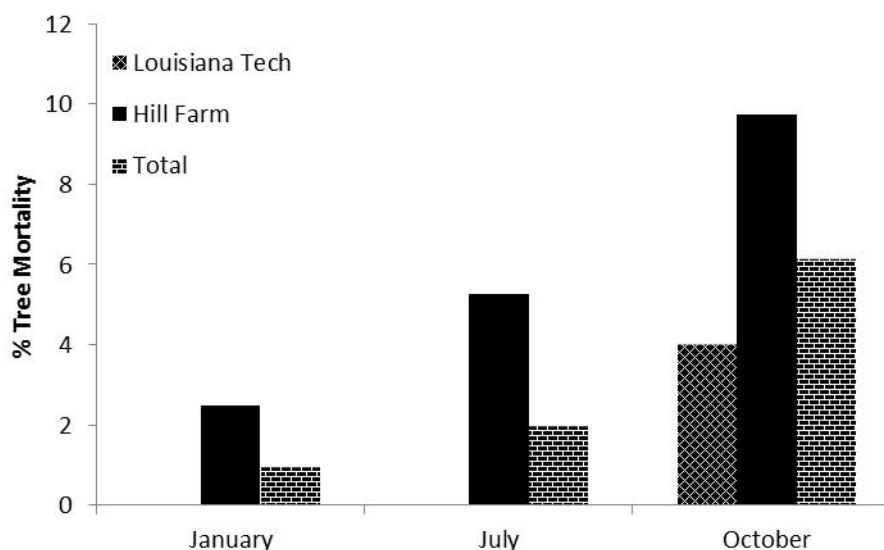


Figure 1—Average tree mortality percentage for each site and overall (Total) across three measurement periods in the growing season. No mortality was observed at Louisiana Tech study site until after the July tally.

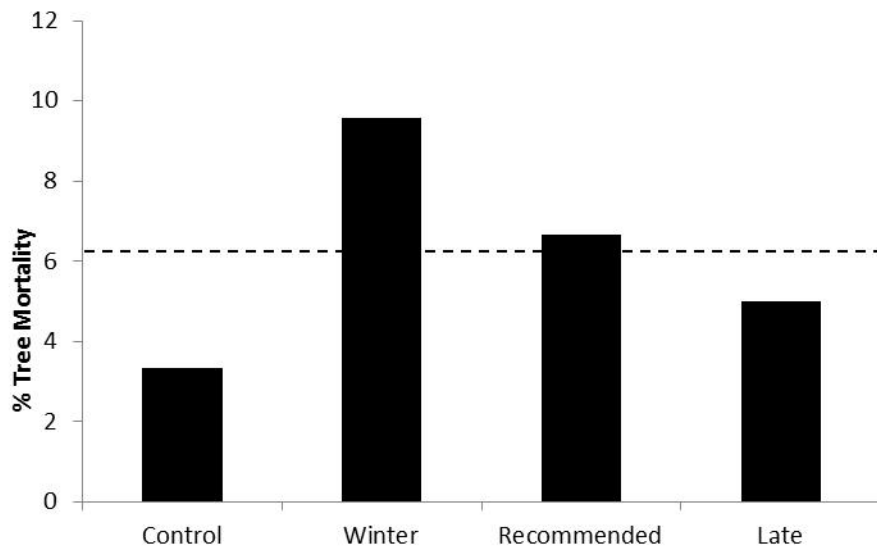


Figure 2—Average percent mortality for each herbicide treatment after one year of establishment. The study's grand average mortality was 6.1 percent indicated by the horizontal dashed line.

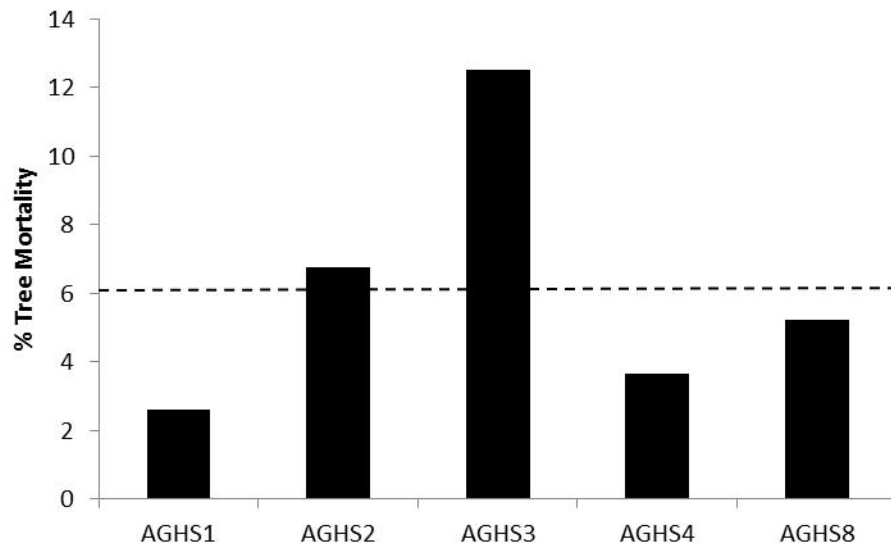


Figure 3—Average percent mortality for each hybrid sweetgum genotype after one year of establishment. The study's grand average mortality was 6.1 percent indicated by the horizontal dashed line.

Groundline Diameter

There were significant location ($p < 0.01$) and herbicide timing ($p < 0.01$) effects on GLD growth. At Louisiana Tech, the seedlings grew an average of 6.63 mm, while seedlings at Hill Farm Research Station only averaged 5.21 mm of growth during the first growing season after planting. Results from initial measurement were also significant for location, which could mean larger trees were randomly selected for planting at Louisiana Tech, and this initial size difference influenced the growing capabilities of the seedlings through the first growing season (fig. 4).

The recommended treatment (February) performed the best (7.06 cm), followed by the winter treatment (January) (6.89 mm), and then the control (5.50 mm) and late (March) (5.43 mm) treatments produced the least diameter growth in the first growing season after planting (fig. 5). The conventional treatment had significantly greater GLD growth than

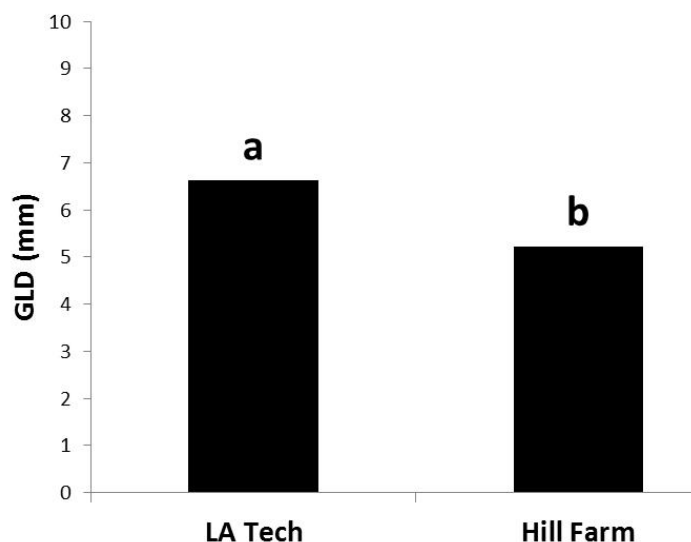


Figure 4—Difference in groundline diameter (mm) between Louisiana Tech and Hill Farm study sites. Differing letters above the bars indicate statically different means using Least Significant Difference mean separation ($\alpha = 0.05$).

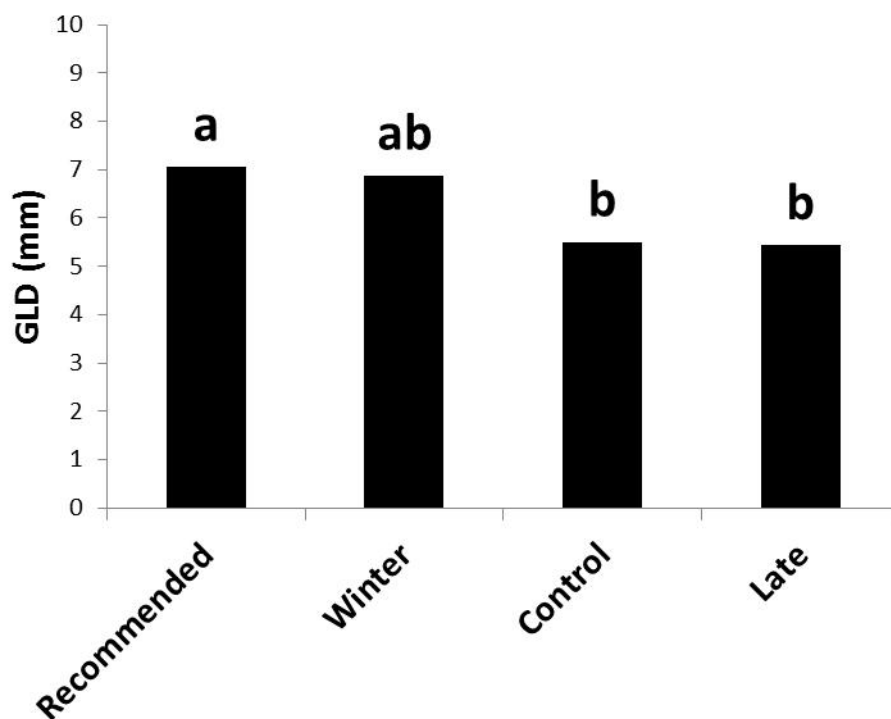


Figure 5—Significant groundline diameter growth changes were seen under various herbicide regimes. Differing letters above the bars indicate statically different means using Least Significant Difference mean separation ($\alpha = 0.05$).

control and late treatments. The early treatment was not significantly different from any other treatment. Our results support previous findings that competition control can increase growth of young hardwood seedlings (Zutter and others 1987).

Height

Significant differences in height growth were observed among the various genotypes ($p < 0.01$) (fig. 6). Three of the varieties produced the most height growth, (46.4, 44.6, 43.5 cm). The fourth variety (AGHS8) grew 33.6 cm in height on average, while the fifth variety (AGHS3) only increased 25.6 cm. Interestingly, the three most vigorous growers are three of the four varieties currently available for purchase commercially.

The interaction between location and herbicide treatment was significant in the ANOVA for height growth (fig. 7). The recommended herbicide and control treatments from both sites were higher than the winter and late herbicide treatments, with Hill Farm showing a larger spread between the treatments than the Louisiana Tech study site. Trees in the recommended herbicide treatment group at Hill Farm grew 48.8 cm in height during their first growing season, the greatest height growth of any treatment across both locations. At Louisiana Tech, the treatment that produced the most height growth was

the recommended herbicide treatment, with an average of 42.9 cm. The winter herbicide application performed the worst at Hill Farm, growing an average of only 28.4 cm during the course of the growing season. Louisiana Tech was more consistent across all treatments than Hill Farm; 7.9 cm separated the highest and the lowest treatments at Louisiana Tech, while 20.4 cm separated the extremes at Hill Farm. This interaction suggests the herbicide reacted differently at the two sites, but it is unclear if this was caused by differences in previous land use, soil texture, or vegetation. Across both sites, the recommended and control treatments outgrew the winter and late treatments, but only the recommended treatments from both sites and the control treatment at Hill Farm could be completely separated from the early and late treatments statistically. The only statistical separation at Louisiana Tech was between the recommended treatment and the late treatment; all other comparisons were not significant. Previous research has shown competition control can increase height growth, but herbicide can also stunt the growth of crop trees at the same time as reducing competition from other species if Oust® XP is not applied according to the label (Cox 2002). The winter (January) herbicide application was applied while some of the seedlings still had leaves from the previous growing season; these leaves could have absorbed some of the herbicide causing the tree

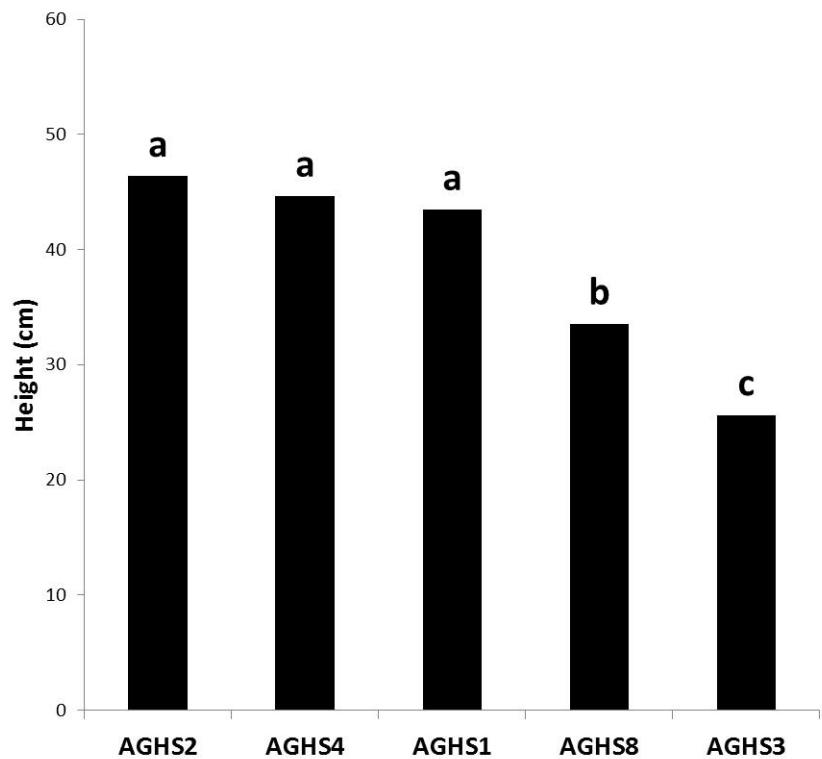


Figure 6—Height growth (cm) differed among genetic varieties significantly. Differing letters above the bars indicate statically different means using Least Significant Difference mean separation ($\alpha = 0.05$).

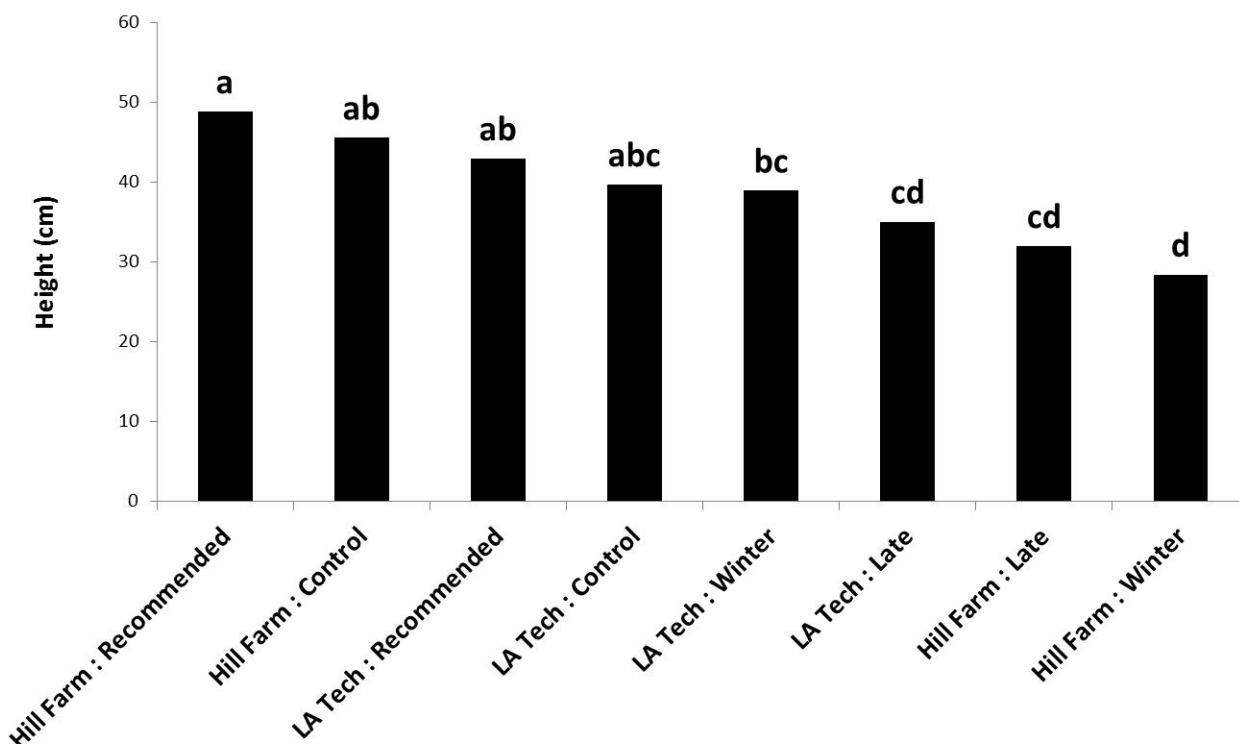


Figure 7—The interaction term among location and herbicide variables was significant in our analysis. The middle herbicide treatment at Hill Farm produced the most height growth, but was not statistically different from the control treatment at Hill Farm or middle and control treatments from Louisiana Tech. Differing letters above the bars indicate statically different means using Least Significant Difference mean separation ($\alpha = 0.05$).

to be stunted. The late herbicide treatment which was applied in March was after bud swell, when the Oust® XP label recommends not spraying because of the risk of damage to the crop trees. While we did not find a mortality difference, this height growth reduction in response to the early March application time relative to the control at the Hill Farm site could be the side effect of spraying herbicide into the growing season (Schweitzer and others 1999).

CONCLUSIONS

We set out to test how herbicide timing affected our five hybrid sweetgum varieties in the areas of mortality and growth in height and diameter during the first growing season. To that end, we found no significant mortality in our trials, but the decreased height growth associated with the herbicide treatments outside the recommended herbicide application could indicate the seedlings were stressed by the herbicide application. These findings could serve as a precaution against spraying hybrid sweetgum seedlings with sulfometuron methyl too early in the winter (before they lose their remaining leaves) and after bud swell (during the growing season).

The significance of our location by herbicide interaction could point towards a site factor that affected the activity of the herbicide. Both sites had a pH between 4.6 (Louisiana Tech) and 4.9 (Hill Farm), and no other known soil factor showed significant variability to cause the trends seen in the location by herbicide interaction. More research needs to be done as to what could have caused these differences on our sites, but this does serve as a caution to always test soil before deciding on what species to plant or what herbicide to apply.

ACKNOWLEDGMENTS

We would like to acknowledge all of the help and support provided by ArborGen on this project in donating the seedlings and providing information from previous studies to guide our quest to further complete our knowledge of the new hybrid sweetgum varieties available. We would also like to thank LSU AgCenter and Hill Farm Research Station for providing a second test location and all the necessary equipment and expert personnel required to implement our herbicide applications. Finally, we would like to thank all of the Louisiana Tech School of Forestry students and faculty who helped lay out plots and plant seedlings at our Louisiana Tech study site.

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Forest Soils and Best Management Practices

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MODELING POTENTIAL EROSION DIFFERENCES OF SMALL TRIBUTARIES IN MANAGED STANDS IN THE BANKHEAD NATIONAL FOREST, ALABAMA

Allison Bohlman, Dawn Lemke, and Andy Scott

Abstract—In the William B. Bankhead National Forest, AL, the Sipsey Fork River Watershed's stream network is composed of many small tributaries that are short in length, ephemeral, and have low average discharges when wet. However, the potential to transport considerable amounts of sediment during significant storm events has not been thoroughly investigated. Sediment runoff was collected from small tributaries that drained recently burned, thinned, and control stands during heavy seasonal storms in early 2016. Water samples were analyzed for Total Suspended Solids. Results from between burned, thinned, and control stands were compared. Observational data were evaluated against predicted potential sediment loads modeled using WEPP (10.3), the geospatial interface program of the Water Erosion Prediction Project (WEPP) - a process-based hillslope and watershed model. The WEPP model performed poorly, and a second erosion potential model was developed to identify important variables that could improve modeling of erosion and sediment runoff from small tributaries. Using ArcMap's ModelBuilder, qualitative rankings of erosion potential were evaluated using the same runoff data. Erosion predictions were positively correlated with observational runoff of small tributaries from the second model.

INTRODUCTION

Forested watersheds in the Southeastern United States are generally well-protected areas that are good sources of clean water and aquatic fauna including fishes, freshwater mussels, and herpetofauna species (Gaines and Creed 2003, Grace 2005, U.S. Environmental Protection Agency 2002). Common management practices in southeastern forests that may contribute to lowered water quality and sedimentation include prescribed burning and timber harvesting (thinning) practices (Sheridan and others 1999, Sun and others 2001). Loss of ground cover, soil disturbance, stand size, and stream proximity are important factors contributing to topsoil erosion and increases in total suspended sediments, nutrients, and turbidity detected in local streams (Sanders and McBroom 2013). While the overall contribution of forest management activities to impairment of southern streams (approximately 8 percent of total continental streams) is low, the potential impact to critical instream habitats or water quality is crucial for conservation efforts in the region.

Best management practices (BMPs) have been implemented by the U.S. Department of Agriculture Forest Service to mitigate and prevent erosion in

forests from prescribed burning and thinning activities. Intermittent streams in the conterminous United States account for about 60 percent of the total river length (Nadeau and Rains 2007). In Alabama, these streams may be a part of a complex hydrologic system that includes karst and cave systems that are poorly understood. Small streams link soil water storage, aquifers, and forest vegetation to the larger stream network within a watershed. Potential soil from disturbance and erosion caused by burning or thinning in forests may be quickly transported downstream, increasing sedimentation and siltation in larger rivers. Few studies have quantified the sediment contribution of intermittent streams in watersheds in Alabama or the greater United States. Additionally, BMPs in Alabama do not directly address how to protect these temporary and small-sized streams when burning or thinning a forest stand. It is unknown if forest management activities such as burning and thinning contribute excess sediment through these small stream channels. The research objective is to assess the impacts of burning and thinning management practices on small forested tributaries using a combination of field and remote sensing techniques.

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MATERIALS AND METHODS

Study Site and Design

The William B. Bankhead National Forest (BNF) is located within the dissected plateau of the Southern Cumberland Plateau in northwest Alabama (fig. 1). Approximately 730 km² in total area, it is a mesic, mixed hardwood-pine forest with highly erodible and well-drained soils. The Sipsey Fork River Watershed drains most of the BNF into Smith Lake, which is located on the southern edge of the BNF in Winston County, AL. Intermittent streams in the BNF are mostly diminutive in size (2m ± 0.5m width), flashy in nature, wet predominantly from December through June, and are dry stream beds in the later summer months of August and September through the fall. Twenty small intermittent streams that directly flow through and solely drain BNF forest stands were selected for water quality sampling at buried road culvert drains. Stream sites were grouped into three treatment types: prescribed burn (n = 7), thin (n = 5), and control (n = 8). Prescribed burns were performed by Forest Service BNF personnel in January through March 2016. Selective harvesting in thin stands occurred in the fall of 2015. Control stands received no management activities that changed the canopy or overall vegetative ground cover. Sites were mapped in ArcMap 10.1, with shapefile BNF boundaries, BNF 2015–2016 burn and thin stands, streams, and a LIDAR-based digital elevation model (DEM). Each sampling site's natural watershed was hand drawn in ArcMap to estimate the surface drainage area using a hill-shaded 10-m DEM overlaid with an ESRI topographic base map (USA Topo Maps 2013).

Stormwater Sampling

Five storms were sampled from January through April 2016. A storm event was sampled if the predicted precipitation was >10 mm and the local meteorological stations forecasted that it would continue for at least 4 hours (The Weather Company 2016). Sites were sampled in mixed-treatment groups for each individual storm. One L of stormwater was collected from the upstream gully side of the culvert every 90 minutes during the storm for a total of three samples from each site. Sample runoff was tested immediately for pH (± 0.2), conductivity (µS, ± 1 percent), temperature (°C, ± 0.2), total dissolved solids (TDS) (ppm, calculated from conductivity and temperature readings), and dissolved oxygen (mg/L, ± 0.2 percent) using a portable multi-meter Pro Plus sonde (YSI Inc.). Turbidity (NTU, ± 0.2 percent) was measured using an Oakton turbidity meter. Each L of stormwater was subsampled with a well-mixed 100 mL subsample stored in a Nasco Whirl-pak®. Total suspended solids (TSS) (mg/L) were determined in the laboratory using three pooled 100-mL water samples collected at each site.

Erosion Modeling and Analysis

The online Water Erosion Prediction Project or WEPP GIS (WEPP) interface model was used to simulate the rate of erosion and sediment yields of the burn, thin, and control watersheds (Flanagan and Frankenberg 2002, National Soil Erosion Research Laboratory 2014). Watersheds were delineated using the allocated online WEPP tools using a preloaded elevation and land use and landcover (LULC). The final set up for the individual WEPP model included climate station selection, defaults for soil and LULC, simulation type, testing years, soil loss tolerance, and certain processing options. Soil and LULC processing were determined by individual grid cells' values. The simulation type selected was for representative hillslopes and channels, also known as the watershed method. The soil loss tolerance (T-value) was set to 3 tonne/ha/year. The model was set for a 10-year analysis estimation of average annual sediment yield (t/ha/year) for each watershed.

The raster files (30-m resolution) were extracted and spatial analyses were conducted within ArcMap (fig. 2). The total sediment loss raster files were extracted and reclassified into four broad erosion classifications. A simple scale of 1 to 4 was devised to estimate erosion potential within each watershed: 1. little to no erosion (modeled soil loss 0–2.5 t/ha/year), 2. slight erosion (modeled soil loss 2.5–5 t/ha/year), 3. moderate erosion (modeled soil loss 5–20 t/ha/year), and 4. heavy erosion (modeled soil loss 20–100 t/ha/year). To gauge the WEPP model's efficiency in modeling small forested streams, an iterative erosion model was proposed. An alternative erosion model (EM) was developed in ArcMap's ModelBuilder to estimate the potential erosion influenced by slope, road proximity, and aspect (fig. 3). Primary input parameters for the EM included the site's individual watershed polygon file, the LIDAR DEM for the BNF, and a Forest Service BNF roads vector line file.

The final EM produced individual delineated watersheds with ranked areas of potential erosion. Overall values for potential erosion rankings were classified on a scale of 1 to 4: (1) little to no erosion (estimated soil loss 0–2.5 t/ha/year), (2) slight erosion (estimated soil loss 2.5–5 t/ha/year), (3) moderate erosion (estimated soil loss 5–20 t/ha/year), and (4) heavy erosion (estimated soil loss 20–100 t/ha/year). This scale is qualitatively similar but cannot be quantitatively compared to the reclassified ranking scale derived from the WEPP model. Percent area for the ranked potential erosion areas was calculated.

Since both models produced separate watershed delineations of each study site, differences in total area (ha) and sub-catchment overlap (spatial agreement with the topography) were examined between the WEPP

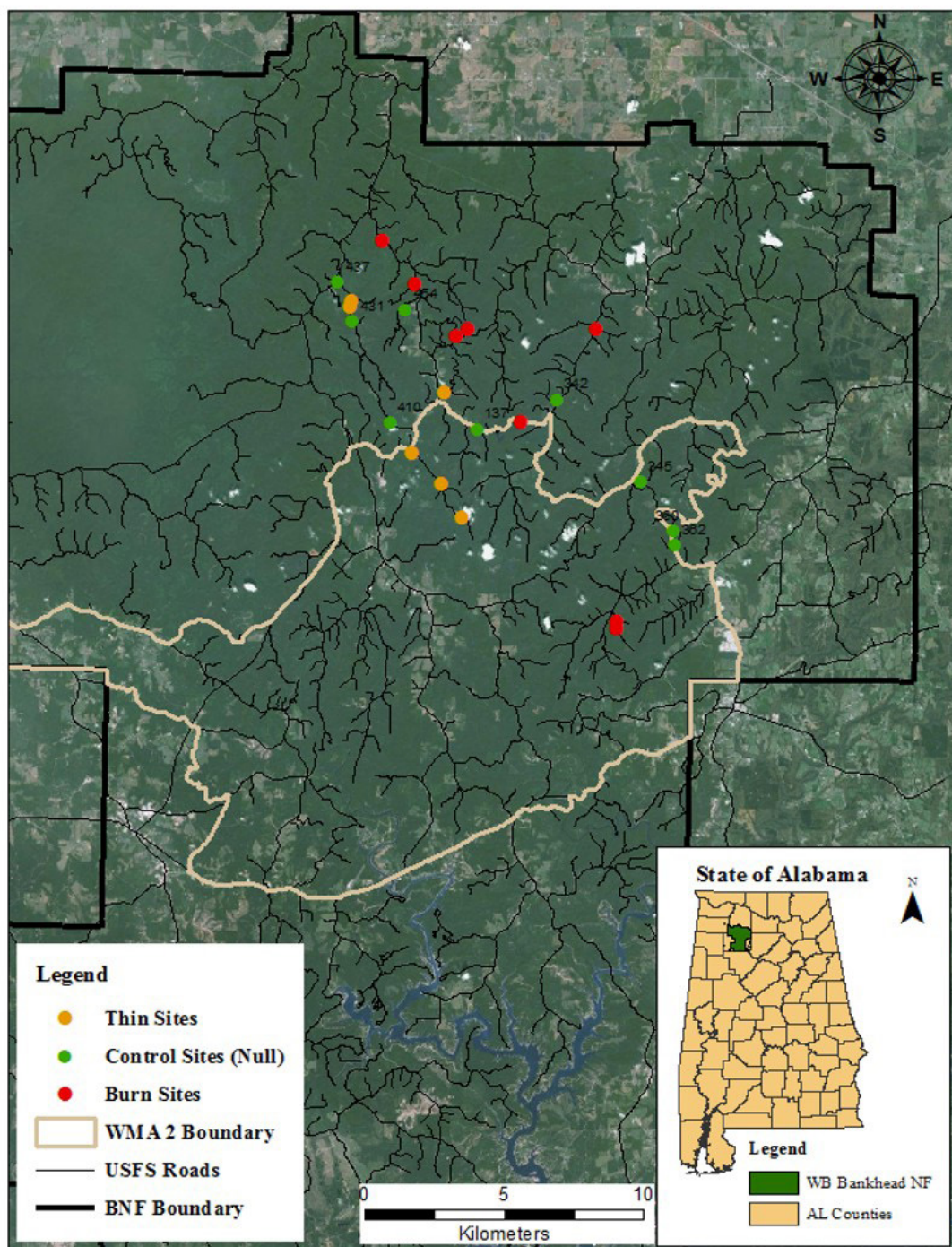


Figure 1—A map of the William B. Bankhead National Forest (BNF) with boundary, primary Forest Service roads, and Wildlife Management Area 2 (WMA 2) boundary, also known as the Black Warrior Wildlife Management Area (Alabama Department of Conservation and Natural Resources and U.S. Department of Agriculture Forest Service). Prescribed burn, thin, and control sites ($n = 20$) for the study are located in the north-central part of the BNF.

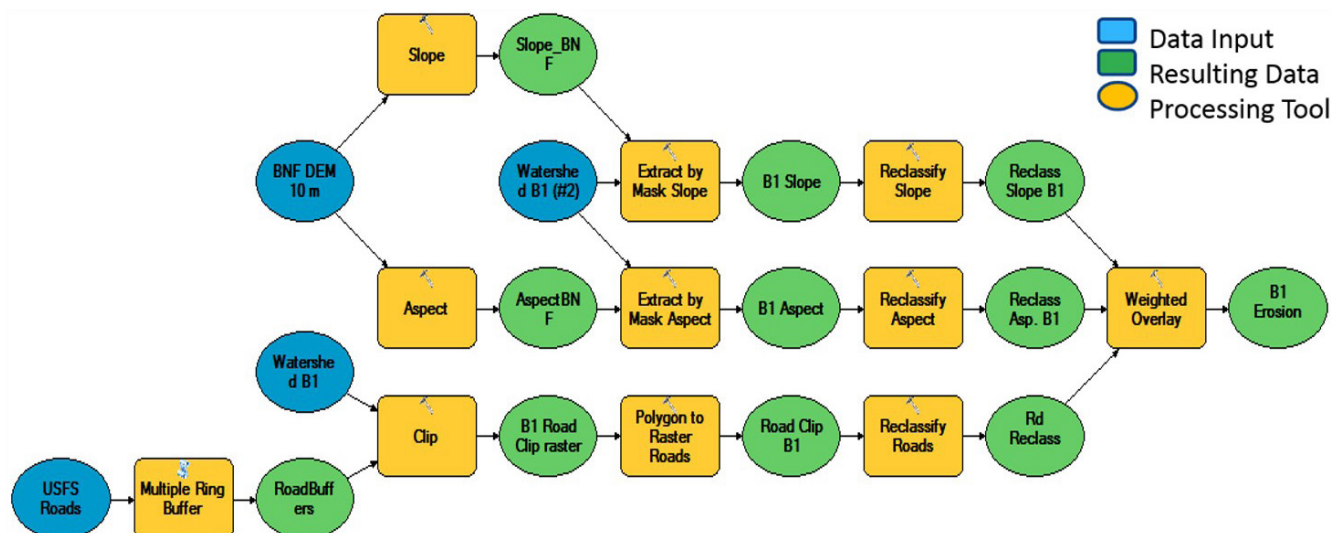


Figure 2—An online Water Erosion Prediction Project (WEPP)-delineated watershed from a thin stand in the William B. Bankhead National Forest (BNF) shapefile extracted and projected over a LIDAR 10-m digital elevation model in ArcMap 10.4. Areas of modeled soil loss (t/ha/year) ranging from 0–36 t/ha/year were modeled over a 10-year timeframe.

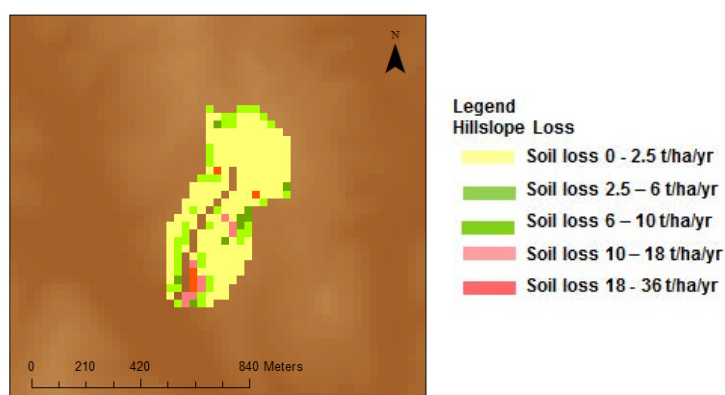


Figure 3—The work schematic for the erosion model built in ArcMap ModelBuilder to estimate the amount of potential erosion in delineated burn, thin, and control watersheds in the William B. Bankhead National Forest (BNF). Data inputs included shapefiles of each watershed, Forest Service roads, and a LIDAR-based 10-m digital elevation model.

and EM. The calculated areas for both models were compared to the topographically drawn watersheds' areas to gauge model accuracy for small intermittent streams. Summary statistics of select water quality parameters (TDS, TSS, and turbidity) and Pearson Correlation analyses were run using the program R Commander (v. 2.3-2), powered by open source statistical software R (v. 3.2.2) (Fox and Bouchet-Valat 2017, R Team 2008).

RESULTS AND DISCUSSION

Measured water quality parameters among the three treatments did not significantly differ ($p = 0.01$). Turbidity and TSS had the highest variance (coefficient

of variance = 1.08 and 1.69). TSS had statistically high measurements (100 percent quartile) of 535.33 mg/L and turbidity of 417.67 NTU, both originating from a control site. The mean areas were 1.53 ha for the natural watersheds, 7.65 ha for the WEPP-delineated watersheds, and 1.65 ha for the EM watersheds. A paired t-test found no significant difference in watershed areas that were hand-drawn and those delineated by the EM [95 percent confidence interval (CI), -3.129 to 0.923, $t(19) = -1.139$, $p = 0.268$]. A significant difference between the WEPP-delineated watershed areas and the hand-drawn watershed areas [95 percent CI, -12.449 to -3.983, $t(19) = -0.062$, $p = 0.0006$] was found. Forty percent of the WEPP watersheds were spatially

dissimilar in total area or location than the EM-delineated areas. Half of all WEPP delineations were projected with opposite flow directions of the natural watershed from the sampling site.

A Pearson correlation analysis ($p = 0.01$) showed a negative relationship between the WEPP model's sediment yield results for each watershed and water quality variables of turbidity (-0.27), TDS (-0.11), and TSS (-0.39). Using the same water quality variables, a positive correlation ($r > 0.8$; $p = 0.01$) was found with the EM model's rankings (standardized by area): turbidity (0.21), TDS (0.29), and TSS (0.33) (table 1). The erosion potential-scaled rankings tabulated for the WEPP and EM models did not show agreement among the predicted burned, thinned, or control treatments. A Spearman rank correlation between the WEPP and EM models' rankings did not find a strong correlation ($r > 0.8$) between the predicted rank values and the three selected water quality variables of turbidity, TDS, and

TSS. WEPP predicted very low amounts of erosion in the treatments. The EM ranks were mixed with higher estimates for slight (2) or moderate (3) erosion potential in the treatments (table 2).

The WEPP model is considered a reliable model for small forested watersheds under 10 ha with permanently flowing streams, but it can overestimate sediment yield and other variables such as discharge (Covert and others 2005, Dun and others 2009). The online WEPP model's preset inputs were a contributing factor for oversized delineations, flow path direction, and spatial positioning of sites. The lower base map resolution (10 m) used in the EM to create smaller stream channels likely increased its delineation accuracy compared to the larger 30-m resolution used by the WEPP. Counter to other popular erosion and hydrological models, the EM model did not include LULC inputs while the WEPP model included both LULC and treatment type (burn, thin, or control stand) for computation. The overall better performance by the EM model through positive correlation with the available observational data suggests that regional landscape characteristics of slope and soil type are more important.

Table 1—Pearson correlation analyses ($p = 0.01$) between two erosion prediction models with observed water quality variables

Water quality parameter	EM model ranking/ Σ area	WEPP model sediment yield
TDS	0.29	-0.11
TSS	0.33	-0.39
Turbidity	0.21	-0.27

Table 2—Calculated mean rankings by treatment type (burn, control, or thin) and standard deviations from each model

Treatment type	Mean erosion ranking (1–4)	Standard deviation
<i>EM model ranking/Σ area</i>		
Burn	2.52	0.42
Control	2.64	0.43
Thin	2.30	0.37
<i>WEPP model sediment yield</i>		
Burn	1.41	0.93
Control	1.73	0.84
Thin	2.00	0.64

CONCLUSIONS

This research's conclusions are limited for the BNF region during a typical winter and early spring season with average monthly precipitation. Total sites ($n = 20$) and low treatment replication currently do not support interpretations for other southeastern forest ecosystems or similar management activities. The BNF management activities of burning and thinning appear to maintain current State and Federal water quality standards. The higher values for turbidity, TSS, and TDS found during storm events in the BNF originated from control stands where there were no human-based stand disturbances recorded.

Improvements in model efficiency for the WEPP should include a 10-m resolution DEM to improve hillslope estimations surrounding smaller intermittent channels. Parameter refinement of the EM's five inputs and calibration is needed to provide further useful information regarding potential erosion. More information is needed to accurately model the connectivity and sediment loading from intermittent streams and their downstream channels. There is an abundance of hydrological data within the Sipsey Fork River Watershed and the greater BNF to supplement future modeling and BMP assessments.

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SOIL EROSION FROM EASTERN HEMLOCK (*TSUGA CANADENSIS*) WINDTHROW MOUNDS FOLLOWING HEMLOCK WOOLLY ADELGID (*ADELGES TSUGAE*) INFESTATIONS IN RIPARIAN AREAS OF THE CHATTOOGA WILD AND SCENIC RIVER

Benjamin T. Poling, C. Andrew Dolloff, W. Michael Aust, and Scott M. Barrett

Abstract—Sediment is a major nonpoint-source pollutant that can decrease the quality of stream waters for humans and aquatic organisms. Hemlock woolly adelgids have infested and decimated eastern hemlocks along the Chattooga Wild and Scenic River, leading to a hypothesis that windthrow mounds of dead hemlocks might have contributed to perceived sediment increases in the Chattooga River. Examination of riparian areas along tributaries of the Chattooga River revealed approximately one hemlock windthrow mound per 3 acres. Erosion estimates indicated that average potential soil erosion from individual windthrow mounds was low (63 pounds per year). Erosion estimates based on pit and mound geometry also indicated low levels of soil disturbance (approximately 1,800 pounds per windthrow mound). Although hemlock decline has potential to cause important ecological changes, low mound frequency, combined with low erosion rates, and riparian buffer protection indicate that hemlock windthrow is a minor source of sediment.

INTRODUCTION

The Chattooga Wild and Scenic River is a 57-mile-long gorge flowing through the southern Blue Ridge Escarpment that was created in 1974 due to its scenic beauty, exceptional fishing, and whitewater rafting opportunities. The Chattooga Wild and Scenic River originates in North Carolina and flows along the border of South Carolina and Georgia within three National Forests (Nantahala, Sumter, and Chattahoochee). Over the previous decade, hemlock woolly adelgid (*Adelges tsugae*) infestations have significantly increased mortality of eastern hemlock (*Tsuga canadensis*) forests throughout the Eastern United States (Evans and others 2011, 2012) and killed the majority of the eastern hemlocks along the Chattooga and its tributaries.

During the same general period, there was a general belief that sediment levels in the Chattooga increased, possibly due to the erosion of windthrow mounds of the dead hemlocks. Other studies of windthrow soil disturbance in the general region support the premise that such mounds could disturb significant areas of soil. Cremeans and Kalisz (1988) found that 2.4 percent of surface area was disturbed by pit and mound topography in the Cumberland Plateau. Greenberg and

McNab (1998) found that 4.3 percent of surface areas were disturbed by pit and mound disturbances following hurricane windbursts in the Appalachian Mountains. Clinton and Baker (2000) found that pit and mound dimensions represented 12 percent of the surface area following a hurricane in the Blue Ridge region. Thus, the pits and mounds created by windthrown hemlocks could provide soil disturbances which may lead to increased erosion and sediment. The objectives of this project were to document the number of hemlock windthrow mounds and to estimate the erosion potential from these mounds in order to determine the potential of such disturbances for sediment pollution.

METHODS

The project had time and budgetary restraints which resulted in the development and use of rapid assessment techniques that could provide insight into the extent and severity of the hemlock windthrow and associated soil disturbances. Many of the bluffs along the actual Chattooga were inaccessible due to safety concerns, so sampling was conducted along five major tributaries of the Chattooga River. The five sampled tributaries were the East Fork Chattooga River, Fowler Creek, Indian Camp Branch, King Creek, and Crane

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Creek. We focused sample efforts on the riparian zone along these tributaries as other studies have found disturbances nearer the streams are most likely to cause sediment increases (Clinton 2011, Lakel and others 2010). Riparian areas were defined as 100 feet on either side of the stream bank (200 feet total width).

Sample plots were large rectangular plots which captured riparian zones along the tributaries. Each sample plot was 200 feet wide (100 feet from each stream bank) and extended 1,640 feet long along the stream meanders. The starting points for sample plots were randomly assigned along the five tributaries, and ArcGIS was used to delineate the stream segments to be sampled. GPS units were used to orient to the starting points. Twenty-five sampling plots were installed along the five tributaries providing an average riparian sampling intensity of 38.7 percent and a range of tributary sampling intensity between 10 and 62.5 percent. All hemlocks within these sample plots were characterized as standing, snapped, or windthrown (e.g., tipped up, pit and mounds). Distance from the windthrow disturbance to the streams was recorded to provide an index of the likelihood of potential erosion becoming sediment.

As part of the hemlock inventory, we modelled the potential erosion rate of each detected windthrow mound by using the Universal Soil Loss Equation as modified for Forestry (USLE-Forest) (Dissmeyer and Foster 1980). The USLE-Forest is an empirical model that uses a rainfall and runoff coefficient (R); soil erosivity (K); slope length and steepness factors (LS); and cover, management, and support factors (CP) to predict potential soil erosion (A) in tons per acre per year. This model has been used to successfully identify erosion-prone areas for a wide variety of forest management activities (Brown and others 2013, Wade and others 2012), thus we used the model to provide an index of potential soil erosion from windthrow disturbance.

In addition to the USLE, we also used simple geometry to estimate the volume of soil disturbance due to windthrow similarly to the method used by Clinton and Baker (2000). We measured the width, length, and depth of the mounds and assumed an ovoid geometry and a soil bulk density of 1.2 g/cm³. This allowed us to estimate the maximum level of soil disturbance due to hemlock windthrow mounds within the watershed.

RESULTS AND DISCUSSIONS

Windthrown trees are often an eye-catching and striking feature in the forest, but we found that windthrown hemlocks actually represented a small proportion (1.5 percent) of the total hemlocks sampled in the five tributaries (table 1). The actual number of windthrown hemlocks was approximately one per every 3 acres, providing an initial indication that such disturbances are relatively minor. Other research on windthrow disturbance in southeastern mountains has indicated that soil disturbances averaged 2.4 percent (Cremeans and Kalisz 1988), 4.3 percent (Greenber and McNab 1998), and 12 percent (Clinton and Baker 2000), but these evaluations followed more acute disturbances caused by windstorms or hurricanes, while our evaluation followed a decade-long chronic disturbance.

The USLE-Forest estimates of potential soil erosion also indicated that soil erosion due to windthrown hemlocks is probably a minor source of sediment (table 2). Our estimates of annual potential soil erosion are similar to those of a mature forest in the Appalachian region (<0.5 tons per acre per year) as presented by Patric (1976), Yoho (1980), and Aust and Blinn (2004). Even the cumulative estimates of erosion from all windthrow mounds in the entire tributary watershed still have estimated erosion quantities that are <13 tons of erosion from the entire 185 acres of riparian forest. Additionally, the average distances of the windthrow mounds were approximately 15 feet from the stream channels. Research in other riparian zones has indicated that even narrow riparian filter strips have the capacity to

Table 1—Results of the eastern hemlock inventory for five tributaries of the Chattooga River

Hemlock disturbance category	Numbers of individual hemlocks in total inventory (185 acres)	Percentage of hemlocks within each disturbance category	Hemlock stems per acre within each disturbance category
Standing dead	3245	83.0	17.5
Snapped	608	15.5	3.3
Windthrow (pit and mound)	58	1.5	0.3
Total for all disturbance categories	3911	100.0	21.1

Table 2—Descriptive statistics for potential soil erosion estimates based on USLE-Forest

Potential soil erosion descriptive statistics	Potential erosion rate
Average per individual windthrow	63.8 pounds per year
Median per individual windthrow	19.6 pounds per year
Maximum per individual windthrow	448.8 pounds per year
Minimum per individual windthrow	0.2 pounds per year
Total estimated erosion from all tributaries	25,183 pounds per year (12.6 tons per year)

trap considerable sediment before it reaches a stream (Clinton 2011, Lakel and others 2010), thus it is doubtful that the majority of eroded material became sediment.

We also measured simple pit and mound geometry in order to estimate the potential of these windthrow disturbances for producing sediment. These calculations indicated that average mass of soil in windthrow mounds was 1,820 pounds, thus the entire mass of soil disturbance from all 58 mounds was approximately 138 tons in all riparian zones. This estimate is the entire mass of displaced soil and is clearly an overestimate of soil erosion, except in instances where a windthrow occurred directly on a stream bank and floodwater displaced the soil.

We do not believe that the windthrow disturbances can account for the increased sediment in the Chattooga. However, during our evaluations we did notice numerous recreationist-created trails on steep slopes that had minimal cover which could provide additional sediment to the streams. We also suspect that some of the sediment may be coming from roads and trails within the watersheds as these anthropogenic disturbances have been found to be common sources of sediment in forested landscapes (Aust and others 2015, Cristan and others 2015). However, the exact sources of sediment are currently unknown, thus future evaluations should be directed to determine the sources. It should also be noted that our investigation does not rule out the possibility that windthrow-related events could provide significant sediment input, particularly in situations of acute disturbances, shallow lithic contacts on steep terrain, or infestations leading to higher proportions of total stand death.

CONCLUSIONS

Our inventory indicates that windthrow mounds are a relatively minor potential source of erosion and sediment. First, the mounds are infrequent, with only one mound occurring on every 3 acres. Secondly, the USLE potential

erosion estimates are quite low, averaging approximately 63 pounds per mound per year. Even maximum potential erosion rates are relatively low. Maximum erosion rates are approximately ¼ ton per mound per year, and these maximum rates are well below those of normal geologic erosion rates. The total potential erosion loss from all mounds in all riparian areas inventoried only equals 12.5 tons per year. This is less than one dump truck of soil from 185 acres of riparian buffers. Finally, the riparian buffers also should trap a significant proportion of the sediment as the average distance from the stream was 15 feet. Lakel and others (2010) found that even narrow buffers (25 feet) were effective in trapping sediment from adjacent clearcut forests. Thus, small isolated pockets of windthrow disturbance within a forest buffer with a relatively intact litter layer should produce only minor sediment to the stream. Similarly, the pit and mound dimension data indicated relatively low masses of disturbed soil due to windthrow mounds. The average soil disturbance mass per mound equaled almost 1 ton of soil, and the total disturbance within all riparian buffers equaled 178 tons per year. It is important to realize that this displaced soil is not analogous to soil erosion, rather it is an index of potential erosion. In summary, the low number of windthrown hemlocks, low estimated erosion rates, and potential erosion trapping of the riparian zones reduce the likelihood that hemlock woolly adelgid-induced hemlock windthrow is a major source of sediment to the Chattooga River.

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EROSION SOURCES AND SEDIMENT PATHWAYS TO STREAMS ASSOCIATED WITH FOREST HARVESTING ACTIVITIES IN NEW ZEALAND

Kristopher Brown and Rien Visser

Abstract—Forest harvesting and associated soil disturbances (e.g., earthworks) can increase catchment sediment yield, with potentially negative consequences for water quality and aquatic habitat. A major challenge for water quality protection is to anticipate where concentrated surface runoff and sediment will reach the stream (a.k.a., a ‘breakthrough’). Improved understanding of sediment sources and pathways can guide forestry practices to control erosion and disrupt concentrated surface runoff. This study involved walking along intermittent and perennial stream channels associated with 23 recent plantation forest harvests throughout New Zealand to quantify the spatial frequency of breakthroughs and characterize their sources. For road-stream crossings that contributed sediment to the stream, the Universal Soil Loss Equation modified for forest land was used to estimate annual sediment delivery rates. Time since harvest completion was typically 2 to 12 months, while harvest area ranged from 4 to 67 ha. Breakthrough frequency for timber harvests using ground-based skidding was 1.9 times that of cable-yarding extraction (6.2 versus 3.3 breakthroughs/stream km). Overall, 73 percent of breakthroughs were associated with concentrated runoff from roads, skid trails, stream crossing approaches, and ruts from machine tracks on hillslopes directed toward streams. Estimates of sediment delivery at road-stream crossings, which accounted for 23 percent of all breakthroughs, were highest for skid trails (median = 70.8 t/ha/year), followed by truck road ditches and running surfaces (3.9 and 1.6 t/ha/year, respectively). These results emphasize the importance of implementing surface cover and adequately spacing water control structures on highly trafficked areas during harvesting activities and site closure. While surface runoff connections between roads and streams cannot be eliminated, the severity of accelerated erosion and their impacts can be minimized.

INTRODUCTION

Forest roads, skid trails, and stream crossings are widely recognized as the predominant sources of surface erosion and sediment delivery associated with forest harvesting operations (Sidle and others 2004, Wemple and others 1996). Despite decades of refinements to forestry best management practices (BMPs), timber harvesting is still associated with short-term increases in catchment sediment yield (Fahey and others 2003). While eliminating sediment delivery through improved BMP technology or implementation may be impossible, opportunities still exist to lessen the severity of sediment inputs that can negatively impact water quality and aquatic habitat.

Many forestry BMP effectiveness studies have focused on reducing erosion (Anderson and Lockaby 2011, Brown and others 2013, Cristan and others 2016), but comparatively less is known about how (and how often) erosion sources become connected to stream channels.

Recent studies have focused on understanding the spatial frequency and physical characteristics of sediment pathways to streams in order to guide site-specific BMP prescriptions for erosion control, sediment trapping, or reducing the volume and velocity of concentrated overland flow (Bowker and others 2010, Croke and Hairsine 2006). For example, Lang and others (2015) and Rivenbark and Jackson (2004) examined ephemeral concentrated flow paths entering streamside management zones (SMZs) (a.k.a., breakthroughs) associated with recent harvesting in the Piedmont physiographic region of the Southeastern United States. On average, one breakthrough occurred for every 6 to 8 harvested ha. However, breakthrough spatial frequency was highly variable from site to site and breakthroughs were non-uniformly distributed throughout forest operational areas. Both studies identified convergent areas (e.g., gullies and swales), surface runoff from roads and skid trails, and road-stream crossings as the most common breakthrough sources. Rivenbark

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and Jackson (2004) found that large contributing areas (mean = 0.4 ha), minimal litter cover, and/or steep slopes characterized the locations where breakthroughs occurred. They concluded that the sediment trapping efficiency of SMZs could be improved by maximizing ground cover, improving road runoff dispersal, increasing resistance to probable surface overland flow paths, and selectively increasing SMZ widths in problem areas.

Management of sediment pathways is particularly relevant for the New Zealand plantation forest industry, which harvests about 30 million m³ of timber [predominantly radiata pine (*Pinus radiata*)] annually (NZFOA 2014). Harvest volumes are predicted to increase to 42 million m³/year by 2025 (NZFOA 2014). Much of the forest land is first-rotation plantation forest on terrain characterized as erodible hill country (Amishev and others 2014), which lacks the road infrastructure needed for harvesting. Annual estimates for new road construction range between 1400 and 2000 km/year for the next decade, the majority of which will be secondary and lower-standard access roads (Fairbrother 2012, Neilson 2012). Concurrent with the potential water quality risk associated with increased harvesting and earthworks, the requirement for water protection in planted forests (and elsewhere) is being raised by environmental regulatory initiatives such as the 2014 National Policy Statement for Freshwater Management (Ministry for the Environment 2014) and the National Environmental Standards for Plantation Forestry (NES-PF) (New Zealand Government 2017). The NES-PF, which came into effect in May 2018, will standardize forestry BMPs across the country, whereas BMPs were previously developed and implemented at a regional level. The timber harvest expansion in New Zealand represents an opportunity to better understand and manage sediment pathways to streams.

The objectives of this study are to: 1) quantify the spatial frequency of breakthroughs associated with recent forest harvesting activities in New Zealand; 2) identify specific sources of breakthroughs and their relative frequency; 3) evaluate hydrologic connectivity and potential rates of sediment delivery at road-stream crossings; 4) evaluate the characteristics of adjacent hillslopes that do and do not contribute sediment to the stream channel; and 5) make recommendations, where applicable, about BMP improvements to reduce the likelihood of breakthroughs.

METHODS

Forestry companies were contacted to select harvest sites that met the following criteria: 1) at least one perennial or intermittent stream within the harvest area, as evidenced by a well-defined, scoured channel; 2) recent harvest (3 to 12 months ago); 3) harvest site to remain in plantation forestry; and 4) harvest area <20 ha. Perennial and intermittent stream channels associated

with 23 recently harvested sites throughout New Zealand were surveyed for evidence of breakthroughs. Sites spanned the following regions: Bay of Plenty (2 sites), Wairarapa (1 site), Tasman (3 sites), Canterbury (10 sites), Otago (5 sites), and Southland (2 sites). Time since harvest completion ranged from 2 to 12 months, except for three sites (also included in this study), which were harvested 2 years beforehand. Breakthrough surveys typically occurred between 2 and 12 months after harvest completion for two reasons: firstly, a 2-month lag time should increase the likelihood that one or more runoff-producing events had occurred prior to the survey, and secondly, recently disturbed soil would be relatively susceptible to water erosion.

Surveys were completed by walking immediately adjacent to stream channels (or next to buffers of native vegetation, if applicable) and looking for evidence of concentrated overland flow (e.g., surface scour) and/or sediment delivery to the stream channel (fig. 1). This is in contrast to Rivenbark and Jackson (2004) and Lang and others (2015), who walked SMZ perimeters searching for evidence of breakthroughs in the Georgia and Virginia Piedmont, where the minimum SMZ width for intermittent streams is 6 and 15 m, respectively, and SMZs are often composed of hardwood forests. In New Zealand's large-scale commercial forests, the silvicultural prescription is typically clearcut harvesting of radiata pine, which can extend to the stream channel if a native forest buffer is not present. However, New Zealand's NES-PF now establishes that harvesting machines can work within 5 to 10 m of perennial streams depending on bankfull channel width, but only at stream crossings or where necessary for directional felling or slash removal. There is also a 10-m buffer requirement for earthworks around perennial streams. Finally, replanting and afforestation of plantation forest must not occur within 5 m of a perennial stream with a bankfull channel width less than 3 m. (New Zealand Government 2017).

Breakthrough flowpaths were followed upslope to identify the erosion source and to describe the hydrologic contributing area associated with the breakthrough. Hillslopes and gullies not contributing to breakthroughs were also measured to compare the geomorphic characteristics of harvest areas that do and do not contribute runoff and sediment to streams. Not including stream crossing approaches, logistic regression was used to predict the likelihood of a breakthrough given upslope contributing area, slope, bare soil percentage, aspect, topography (convergent, divergent, or planar slopes), and the hydrologic influence of roads, skid trails, or machine traffic disturbance (i.e., ruts from mechanized harvesting, slash windrowing, or overland skidder traffic). The presence or absence of roads or skid trails and machine traffic disturbance was evaluated for each data point, and a disturbance ranking was created as follows (1 is lowest, 4 is highest):



Figure 1 — Example of a sediment breakthrough to an ephemeral stream channel. This breakthrough resulted from a landing failure. (photo by Kristopher Brown)

- 1) no roads, skid trails, or machine traffic disturbance;
- 2) machine traffic disturbance only; 3) roads and trails only; 4) both roads/trails and machine traffic disturbance present.

Stream crossing approaches were evaluated separately because these potential breakthrough sources are easily identified, defined (i.e., in terms of contributing hydrologic area), and managed. Runoff and erosion control is important at road-stream crossings because erosion sources can have a relatively direct and unimpeded flowpath to the stream. The Universal Soil Loss Equation modified for forest land (USLE-forest) (Dissmeyer and Foster 1984) was used to estimate annual erosion rates for road-stream crossing approaches that led to breakthroughs. Unlike Lang and others (2015), not all stream crossing approaches were considered to be breakthroughs. Where there was no evidence of surface scour or sediment delivery to the stream channel, as was often the case for overland skid trails with short approaches and slash barriers for water and erosion control, these approaches were considered to be non-breakthroughs. Potential erosion rates were estimated for truck road surfaces, ditches (if applicable), and skid trail surfaces. Field measurements related to the approaches included slope, bare soil percentage, surface roughness, and contributing hydrologic area (i.e., drainage length x width). Drainage length was defined as the distance from the stream to the nearest functioning water control structure (e.g., water bar, cut-out, or cross-drain culvert) or other topographical feature that redirected surface runoff from the road and away from the stream channel. Rainfall erosivity index values (R-values) were selected from a map of R-values

for New Zealand (Haas 2014). For soil erodibility, an intermediate K-value of 0.033 (t/ha/hour / ha/MJ/mm) (Foster and others 1981) was chosen for truck road surfaces because soils associated with the running surface are highly modified. For example, many forest road pavements in New Zealand are composed of a compacted subgrade soil layer underlying an aggregate surface layer (Fairbrother 2011).

RESULTS AND DISCUSSION

Breakthrough Spatial Frequency

Total harvest area per site ranged from 4 to 67 ha, with a median of 23 ha. Timber was extracted with ground-based machines (nine sites), cable yarders (nine sites), and a mix of both (five sites). In total, 23 km of stream channel were traversed in association with 552 harvested ha to count the number of breakthroughs. Overall, 106 breakthroughs were found and the median number of breakthroughs/km of stream was 3.4, or 1 breakthrough for every 6.5 harvested ha. Sites with ground-based extraction had 1.9 times more breakthroughs/km of stream than cable yarding due to higher near-channel disturbances from skid trails. Breakthrough frequency/km of stream ranged from 0 to 23.9 for ground-based extraction, 0 to 9.5 for cable yarding, and 0.5 to 5.3 for sites using both ground-based machines and cable yarders to extract timber (table 1).

Two ground-based harvests accounted for 35 percent of all breakthroughs, suggesting that forest harvesting does not result in ubiquitous sediment inputs, but that one or more 'problem' sites could dominate downstream water quality impacts. At one of these sites, five breakthroughs resulted from surface runoff and fill slope slumps from a

Table 1—Breakthrough spatial frequency per km of stream channel and breakthrough spatial frequency per harvested ha, organized by extraction method and in order of decreasing breakthrough spatial frequency

Region	Channel length <i>km</i>	Harvest area <i>ha</i>	Breakthroughs	Breakthroughs/km	Breakthroughs/ha
Ground-based extraction					
Tasman	0.7	12	16	23.9	1.33
Canterbury	1.9	23	21	10.9	0.91
Southland	0.6	24	5	8.3	0.21
Canterbury	1.4	30	9	6.5	0.30
Tasman	0.7	8	4	6.2	0.51
Otago	1.1	24	6	5.5	0.25
Otago	0.9	17	3	3.3	0.18
Canterbury	0.7	25	2	3.1	0.08
Otago	0.9	12	0	0	0
Cable yarder extraction					
Tasman	0.6	8	6	9.5	0.71
Bay of Plenty	0.6	33	5	8.3	0.15
Canterbury	1.2	23	5	4.2	0.22
Canterbury	0.9	12	3	3.5	0.24
Canterbury	0.6	19	2	3.3	0.10
Canterbury	1.3	27	2	1.5	0.07
Canterbury	0.1	4	0	0	0
Canterbury	0.2	14	0	0	0
Otago	0.7	26	0	0	0
Both ground-based and cable yarder extraction					
Wairarapa	1.5	36	8	5.3	0.22
Canterbury	1.5	47	5	3.4	0.11
Bay of Plenty	1.4	13	2	1.4	0.15
Southland	1.8	47	1	0.6	0.02
Otago	1.9	67	1	0.5	0.01
Summary	23^a	552^a	106^a	3.4^b	0.15^b

^a Sum total.

^b Median value.

truck road and associated stream crossing constructed in highly erodible silty loam soils. An additional 10 breakthroughs resulted from naturally occurring erosion features (i.e., many small land slips, rills and one gully) with no obvious connections to harvesting or road construction activities. At the other problem site, most of the breakthroughs were associated with deep rutting from skid trails and machine traffic near the stream with no provisioning of surface cover during site closure (fig. 2).

Breakthrough Sources

Most breakthroughs (50 percent) were caused by concentrated runoff from roads, skid trails, or machine traffic disturbance on the hillslope (fig. 3). An additional

23 percent of breakthroughs occurred where roads and skid trails crossed streams. Thus, 73 percent of breakthroughs were associated with roads, trails, and machine tracks on the hillslope. These findings reinforce the importance of pre-harvest planning efforts to minimize earthworks, locate roads and stream crossings to avoid steep gradients, and maintain a buffer (e.g., an SMZ or slash barrier) between disturbed soil and streams. Such practices should reduce the occurrence of road-related sediment breakthroughs and reduce the severity of impacts to water quality. Furthermore, road-stream crossings represent easy targets that can be identified during the planning stage and BMPs can be prescribed for water and erosion control.



Figure 2—Photo depicting a badly rutted skid trail post-harvest. (photo by Kristopher Brown)

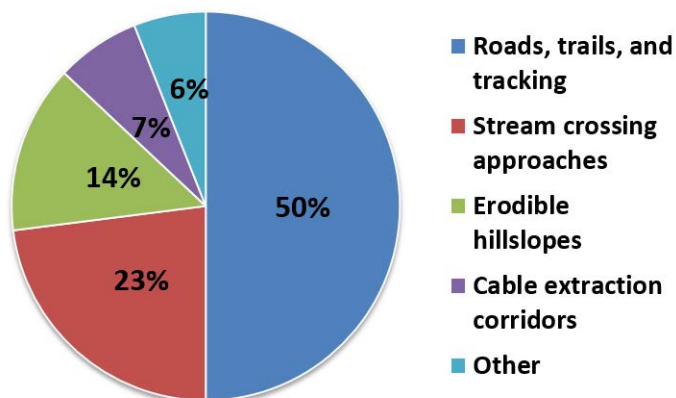


Figure 3—Breakthrough sources and their relative frequency (in percent): n = 53 for roads, trails, and tracking; 24 for stream crossing approaches; 15 for erodible hillslopes; 7 for cable extraction corridors; and 6 for other.

Naturally occurring highly erodible hillslopes generated 14 percent of the overall breakthrough total. These breakthroughs were not necessarily associated with direct harvest disturbance, but instead with small land slips that may have resulted from the vegetation removal. Additional BMPs, such as dry-weather harvesting and slash dispersal on actively eroding areas, may be necessary to avoid exacerbating these erosion sources during harvesting. Seven percent of all breakthroughs were associated with cable yarding corridors that scoured hillslopes as they crossed stream channels. Six percent of all breakthroughs could not be readily classified into any of the aforementioned categories. They included one landing failure (fig. 1), runoff from a saturated bench below a landing, one landing/staging area built over the main stream channel, one uprooted tree near the stream, and two undisturbed gullies that were flowing with water during a storm event.

Connectivity at Road-Stream Crossings

Stream crossing approaches associated with log truck roads (lower-standard spur and secondary roads) delivered concentrated runoff to streams more often (17 of 21 approaches) than those of skid trails (10 of 35 approaches). However, skid trail stream crossings were more frequent. Surfaces of log truck roads are compact by design in order to support heavy loads and prevent water infiltration into the road subgrade. Thus, there is a greater potential for surface runoff generation, even for low-intensity rainfall events. Conversely, skid trail crossings that did not contribute to breakthroughs were typically not bladed (i.e., cut into the hillside), not as compact, and retained some of the protective functions of the forest floor.

The distance from the stream to the nearest water control structure was important for understanding road-to-stream hydrologic connectivity. Median drainage length for road-stream crossings that led to breakthroughs was 73 m, ranging from 5 to 185 m (fig. 4). Conversely, median drainage length for road-stream crossings that did not lead to breakthroughs was 12 m, ranging from 5 to 61 m. These findings suggest that while it may be difficult to eliminate connectivity from log truck roads at stream crossings (due to their inherently compact design), the severity of water quality impacts can be managed by adding water control structures to reduce the drainage length. Sessions (2007) recommended reducing the drainage length at road-stream crossings to approximately 20 m. Of the 21 truck road approaches to stream crossings measured in this study, only 4 had a drainage structure within 20 m of the stream. For skid trail crossings, connectivity could be reduced by using overland (not bladed) skid trails where slope steepness allows, as well as installing cut-outs for water control and using slash or mulch for surface cover during road closure.

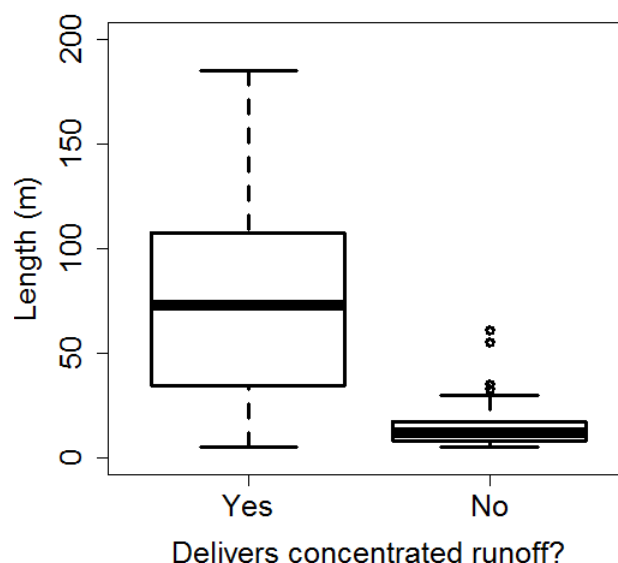


Figure 4—Box and whisker plots (showing the 5th, 25th, 50th, 75th, and 95th percentiles) of stream crossing approach length (m) in relation to whether or not the approach delivered concentrated surface runoff to the stream (n = 27 for 'yes' and n = 29 for 'no'). Approach length was defined as the distance from the stream to the nearest road drainage structure (e.g., water bar, turnout, or culvert cross-drain) or topographic feature that moved surface runoff away from the road.

Potential Erosion Rates for Breakthroughs at Road-Stream Crossings

Erosion estimates ranged from 0.5 to 13.4 t/ha/year for truck road surfaces, 0.9 to 20.1 t/ha/year for truck road ditches, and 5.9 to 314.8 t/ha/year for skid trail surfaces (table 2). The median potential erosion rate for skid trails was 44 times greater than truck road surfaces and 18 times greater than truck road ditches (fig. 5) because skid trails that led to breakthroughs were steep (median = 16.5 percent slope) with 0 to 30 percent surface cover. This finding highlights the importance of closing skid trails properly, especially at stream crossings. Several studies have demonstrated the sediment-reduction efficacy of skid trail closure techniques, such as water bar installation, plus slash or mulch application (Sawyers and others 2012, Vinson 2016, Wade and others 2012).

Truck road surfaces with potential erosion rates <2 t/ha/year had a median bare soil percentage, slope, and drainage length of 29 percent, 4 percent, and 76 m, respectively. Thus, while drainage lengths were relatively long, gentle slopes and/or good surface cover reduced potential erosion rates. Pavement design for lower-standard spur roads in New Zealand typically employs a single 'improved' layer, consisting of compacted basecourse aggregate overlying a compacted subgrade soil. This aggregate layer provided good surface cover. Potential erosion rates were higher in truck road ditches, mainly because ditches were typically unsurfaced.

Table 2—Summary of estimated erosion rates for road-stream crossing approaches for road and skid trail surfaces that led to breakthroughs

Region	Drainage length <i>m</i>	Slope <i>percent</i>	Bare soil <i>percent</i>	Potential erosion <i>t/ha/year</i>
<i>Log truck roads</i>				
Tasman	175	10	30	13.4
Wairarapa	91	12	35	9.0
Otago	185	12	62	7.1
Canterbury	110	9	50	5.5
Southland	105	11	20	3.4
Canterbury	15	9	50	2.9
Otago	79	7	40	2.0
Tasman	51	2	40	1.9
Otago	147	7	27	1.6
Wairarapa	31	3	50	1.6
Wairarapa	49	5	25	1.6
Canterbury	100	5	30	1.3
Canterbury	144	14	20	1.3
Otago	155	7	15	1.2
Southland	50	4	20	0.8
Canterbury	15	2	50	0.6
Otago	180	3	35	0.5
<i>Skid trails</i>				
Tasman	25	52	80	314.8
Tasman	43	33	80	203.7
Tasman	76	17	85	89.8
Tasman	23	16	100	76.6
Tasman	95	12	100	76.6
Bay of Plenty	6	33	70	65.0
Bay of Plenty	5	19	70	23.7
Otago	73	16	80	17.3
Canterbury	51	14	90	15.3
Canterbury	38	10	80	5.9

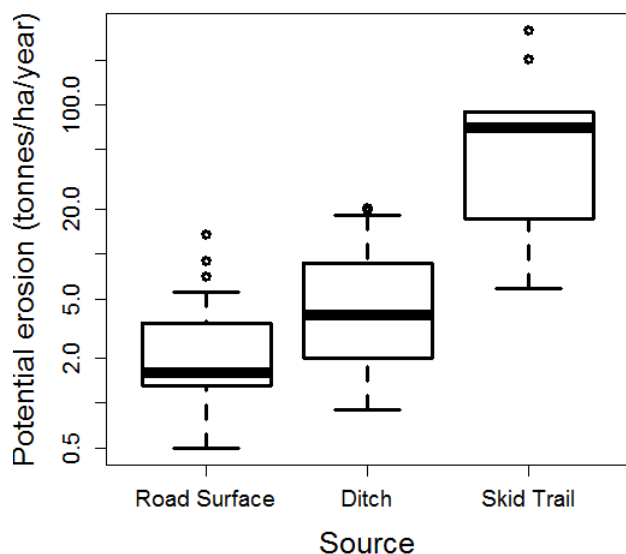


Figure 5—Box and whisker plots (showing the 5th, 25th, 50th, 75th, and 95th percentiles) of potential erosion rates of log truck and skid trail stream crossing approaches (n = 17, 14, and 10, respectively, for truck road surfaces, truck road ditches, and skid trails). Each of these approaches contributed to a breakthrough.

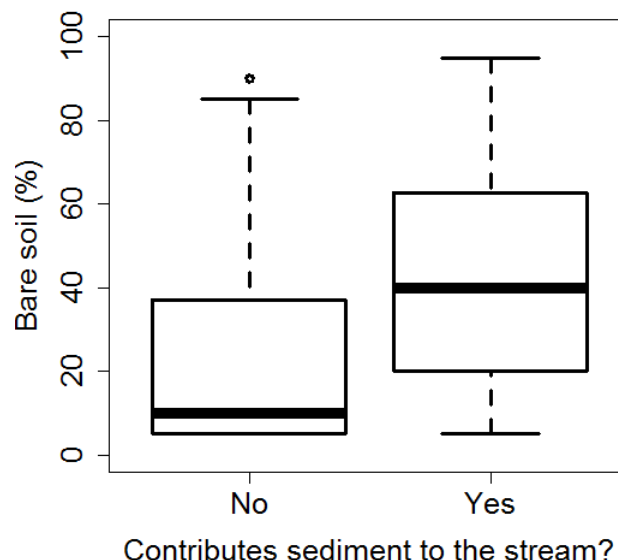


Figure 6—Box and whisker plots (showing the 5th, 25th, 50th, 75th, and 95th percentiles) of bare soil percentage for upslope contributing areas that did (n = 77) and did not (n = 267) lead to a breakthrough.

Characteristics of Upslope Contributing Areas that Led to Breakthroughs

Breakthrough likelihood of occurrence increased with increasing bare soil percentage on upslope contributing areas ($Z = 5.2$, $p < 0.001$) (fig. 6). Hillslopes with no roads, skid trails, or machine traffic disturbance ($Z = -2.4$, $p = 0.02$) or only machine traffic disturbance ($Z = -2.8$, $p = 0.004$) decreased the likelihood of a breakthrough. Median bare soil percentage for upslope areas contributing to breakthroughs was four times higher than those with no evidence of breakthroughs (i.e., 40 vs 10% bare soil).

Breakthrough relative frequency was examined in relation to the severity of disturbance from roads, skid trails, and machine traffic on hillslopes adjacent to the stream (344 total cases). In 186 cases where no roads, skid trails, or machine tracks were present, only 25 breakthroughs were found. In 50 cases where machine tracks, but not roads or skid trails, were present, 10 breakthroughs were found. In 59 cases where roads and skid trails, but not machine tracks, were present, 18 breakthroughs were found. In 49 cases where roads, skid trails, and machine tracks were present, 24 breakthroughs were found. Thus, when more highly compacted or disturbed areas were present upslope, breakthroughs occurred more frequently.

CONCLUSIONS

Negative impacts to water quality can occur when sediment from forest harvesting operations breaks through to stream channels. This study examined

intermittent and perennial stream channels associated with 23 recent plantation forest harvests throughout New Zealand to quantify the spatial frequency of breakthroughs, characterize their sources, and contribute to the improved understanding and management of sediment pathways to streams. Overall, the median number of breakthroughs/km of stream channel was 3.4. Ground-based extraction was typically associated with more soil disturbance from skid trails, thus the median breakthrough spatial frequency was 1.9 times that of cable yarding (i.e., 6.2 versus 3.3 breakthroughs/km of stream). The majority of breakthroughs (73 percent) were associated with truck roads, skid trails, or machine traffic disturbance on hillslopes, which emphasizes the importance of minimizing earthworks, locating roads and stream crossings to avoid steep gradients, maintaining a buffer between disturbed soil and streams, and prescribing BMPs for surface cover and water control.

This study identified two areas for improved BMPs to either reduce road-to-stream connectivity or lessen the severity of water quality impacts from breakthroughs (i.e., implement effective BMPs). The first is to locate road-stream crossings to avoid steep approaches and, if applicable, use water control structures (i.e., cross-drain culverts and cut-outs) to reduce surface runoff volumes that discharge directly to the stream channel. The second is to ensure that temporary stream crossings are closed properly, as potential erosion rates in this study exceeded 80 t/ha/year for steep skid trails with poor water control and surface cover. Closure techniques for skid trails are well-documented and include water

bars and cut-outs, plus some type of surface cover (e.g., slash or mulch application). Both skid trails and temporary log truck roads should be periodically inspected post-harvest to ensure that water control structures are functioning properly.

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RIDGE AND VALLEY HARVESTING EFFECTS ON SOIL PROPERTIES, POTENTIAL EROSION, AND SEDIMENT DELIVERY RATIOS

Brian M. Parkhurst, W. Michael Aust, M. Chad Bolding, and Scott M. Barrett

Extended abstract—Numerous studies have evaluated harvesting influences on soil properties, erosion, and water quality, yet few have developed relationships between erosion and sediment delivery. Our goals were to evaluate forest operation effects on soil properties, erosion potential, and sediment delivery on steep terrain to develop sediment delivery ratios. Erosion rates were estimated using the Universal Soil Loss Equation (USLE) and measured using silt fence sediment traps. Bulk density and saturated hydraulic conductivity were used to evaluate changes in soil properties across the harvest operation

Two adjacent silvicultural systems (clearcut and irregular shelterwood) were harvested in the Ridge and Valley of Virginia. The harvest took place on the Fishburn Forest, Virginia Tech's research forest, located in Montgomery County, VA. The harvest occurred in 2015 and followed traditional harvest methods associated with ground-based operations. A combination of manual felling and a wheeled feller buncher was used in the clearcut. Manual felling was used in the irregular shelterwood. In both harvests, wheeled skidders were used to transport material to centralized landings for processing.

The forest harvest operation was broken into five operational categories including decks, roads, skid trails, harvest area, and streamside management zones (SMZs). Skid trail observations were further divided into bladed and overland categories to evaluate any changes based on degree of disturbance.

Erosion rates and sediment production were evaluated for approximately 50 subwatersheds. Each watershed was mapped and broken into contributing areas based on the five operational categories. Erosion rates and sediment production were calculated based on an area weighted average across each subwatershed. The USLE was used to provide potential erosion rates, and silt fences placed at the watershed outlets were used to capture actual sediment production. Approximately half of the silt fence locations were installed at the edge of the SMZ, and the remainder were installed at 15 m inside the SMZ, the minimum recommended SMZ width.

Soil properties were evaluated across the entire harvest operation using the same broad categories. Soil cores were collected from the upper portion of the soil profile using slide hammers to provide samples. Cores were used to calculate bulk density which was corrected for coarse fragments following standard laboratory procedures. Hydraulic conductivity was calculated by subjecting the soil cores to saturated flow under a constant head following standard laboratory procedures.

Preliminary results for potential erosion rates showed the majority of operational categories experiencing between 2.24–11.21 t/ha/year erosion. The mean erosion rate was 9.66 t/ha/year, the maximum was 67.34 t/ha/year associated with overland skidding, and the minimum was 0.11 t/ha/year associated with a haul road. When potential erosion rates are averaged based on operational category and compared, overland skidding produced significantly more sediment than bladed trails, roads, and harvest area. Because all the samples are from the same location, the major factors in the USLE that contribute to changes in erosion rate are the length and slope factor and the cover practice factor. Overland skid trails may have produced higher erosion rates because of excessive slope length, uncontrolled steep slopes exceeding 10% percent, and a lack of soil cover. Many of these trails followed "logger's choice" methods meaning that they were placed at the discretion of a machine operator. The roads and other more significant soil-disturbing activities likely produced lower erosion rates because of professional layout that controlled length, slope, and ground cover. These disturbances were also subject to Virginia Department of Forestry closeout procedures.

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Actual sediment production results were analyzed for 21 of the 50 silt fence locations. Of the 21 fences observed, only 2 produced appreciable amounts of sediment. A fence draining a predominantly bladed skid trail subwatershed produced 7.3 cm, and a fence draining a predominantly overland skid trail produced 5.6 cm. These results may change when the remainder of the subwatersheds are measured and more time is allowed to pass for followup measurements. These results may suggest that a harvest operation with professional layout and proper closeout will not produce significant sediment.

Soil properties generally showed that operational areas with greater disturbance, more trafficking, and potentially less cover reflect increased soil compaction. Hydraulic conductivity results were not significantly different, but generally followed an inverse pattern to that of soil compaction. Across all operations, bulk density averaged 1.11 g/cm³. The maximum recorded bulk density was 1.50 g/cm³ associated with bladed skidding, and the minimum bulk density of 0.52 g/cm³ was associated with overland skidding. When bulk density was averaged by operational category, roads and bladed trails were significantly higher than overland trails, harvest area, and SMZ regions. Decks were between these groups, but not different than either group. Hydraulic conductivity showed an average of 0.55 cm/hour, a maximum of 17.99 cm/hour for an SMZ, and a minimum of 0.00 cm/hour across numerous categories, but no clear trend emerged from the hydraulic conductivity data.

This research is ongoing and results may change as the study progresses because sample size will increase. Sediment delivery will be analyzed following measurements of the remaining silt fences across the operational categories. These results suggest that a well-planned and monitored timber harvest can be carried out with minimal sediment production and soil impacts concentrated to disturbance areas. Best management practices when implemented properly serve to lessen the impact of a harvest. Planning needs to strive to limit slope length and grade and ensure adequate ground cover during and after harvest operations.

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IMPACTS OF TIMBER HARVEST SOIL DISTURBANCE AND SITE PREPARATION ON SOIL PROPERTIES AND SITE PRODUCTIVITY: LITERATURE REVIEW

Charles M. Neaves III, W. Michael Aust, M. Chad Bolding,
Scott M. Barrett, and Carl C. Trettin

Abstract—Ground-based forest harvesting can potentially alter soil properties and reduce site productivity. Frequent high soil moisture conditions make forested wetlands and coastal plain sites exceptionally prone to these negative impacts. Some sites have adequate natural recovery mechanisms such that productivity over the length of a rotation is not reduced. However, site-specific factors create uncertainty in predicting long-term effects of harvesting disturbance, justifying implementation of site preparation to ameliorate or enhance soil properties and forest productivity. Previous research on harvest disturbance, natural soil and productivity recovery, and site preparation to augment recovery is briefly summarized, with emphasis on southeastern coastal plain sites.

INTRODUCTION

Sustaining forest productivity and forest ecosystem service integrity are guiding principles for sustainable forest management. Globally, the demand for forest ecosystem services is expected to increase simultaneously with decreases in forested land area as a result of human population growth (Burger 2009, Fox 2000). A key ecosystem service provided by forests is timber production, which ensures stable wood and fiber supplies, thereby providing substantial contributions to the economy. Heavy equipment traffic integral to ground-based forest harvesting operations can potentially degrade soil properties and affect productivity and function of forest land (Cambi and others 2015, Miwa and others 2004). For forestry to be sustainable, long-term negative impacts on soil properties must be mitigated (Burger 2009, Fox 2000).

Forested wetlands present a unique management challenge because they are valued for ecosystem services including biogeochemical transformation, hydrology benefits, wildlife habitat, and carbon storage (Richardson 1994). These potentially highly productive sites are exceptionally prone to adverse effects of harvesting equipment traffic due to frequent high soil moisture conditions (Miwa and others 2004,

Richardson 1994). The contiguous United States contain an estimated 44.6 million ha of wetlands, which is approximately one-half of the estimated extent of 89 million ha of wetlands existing prior to European settlement (Dahl 2011). The loss in ecosystem services associated with this decline in wetland area has been acknowledged, and Federal laws have been enacted in an effort to conserve wetlands, including section 404 of the Federal Water Pollution Control Act of 1972 (the Clean Water Act). Wetlands are under jurisdiction of the Clean Water Act if they satisfy defined criteria for hydric soils, hydrophytic vegetation, wetland hydrology, and a nexus to waters of the United States. As a result of wetland creation and restoration efforts, some years show a net gain in wetland area (Dahl 2011). However, it has been suggested that created and restored wetlands do not provide the same ecological and functional value as minimally disturbed wetlands (e.g., Hoeltje and Cole 2007; Hossler and others 2011; Jessop and others 2015; Moreno-Manteos and others 2015, 2012; Richardson 1994; Sutton-Grier and others 2010). Silviculture has been reported as a major cause of decline in forested wetland area between 2004 and 2009 (Dahl 2011), though this is more likely short-term classification as a different type of wetland following harvest rather than conversion to upland. Regardless, it would be

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advantageous to enhance timber production without compromising the jurisdictional status or functional capacity of wetlands.

HARVESTING DISTURBANCE

Modern forest harvesting operations rely upon heavy equipment due to operational and economic efficiencies. Although forestry best management practices recommend avoiding harvesting under wet conditions, the high proportion of timberland located on wet sites, frequent precipitation, and restricted drainage combined with demand for fiber ensure that wet weather harvesting will occur (Aust and Blinn 2004, Miwa and others 2004). Forest harvesting equipment may traffic between 10 and 70 percent of a harvest area during clearcutting, resulting in potential for extensive soil compaction and rutting (Cambi and others 2015). The greatest increases in bulk density and soil strength occur during the first few machine passes. Further compaction becomes more gradual with each additional pass (Gayoso and Iroume 1991, Hatchell and others 1970, Williamson and Neilsen 2000). This indicates that it could be advantageous to minimize the spatial extent of trafficking, restricting damage to a smaller area. The normal and shearing forces generated by traffic interrupt soil particle contact points, potentially eliminating void space and altering soil structure. The degree of disturbance caused by traffic varies with ground pressure exerted by the machine, number of passes, ground cover, soil texture, soil organic matter content, and soil moisture content at the time of disturbance (Cambi and others 2015, Greacen and Sands 1980, Naghdi and others 2016).

EFFECTS OF HARVESTING DISTURBANCE ON SOIL PROPERTIES

Forest harvesting equipment traffic typically increases soil bulk density and decreases macroporosity and hydraulic conductivity. Bulk density increases occur when a mass of soil solids is forced into a smaller volume due to the compression of air-filled pores under a load, or the destruction of water-filled pores via churning (Akram and Kemper 1979, Aust and Lea 1992, Greacen and Sands 1980). Macropores are large enough for water to move freely; therefore, macropore reduction is often linked to decreases in hydraulic conductivity. Aust and others (1998b) observed a bulk density increase from 1.24 to 1.39 Mg m⁻³, macropore space decrease from 15.2 to 5.9 percent, and a hydraulic conductivity decrease from 5.2 to 1.6 cm hr⁻¹ on primary skid trails in wet pine flats in the South Carolina lower coastal plain. At a comparable site, Aust and others (1993) reported similar trends in soil physical property degradation due to ground-based skidding. Gent and others (1983) reported significant increases in bulk density to a depth of 30 cm, and decreases in hydraulic conductivity and macroporosity to a depth of 15 cm in North Carolina coastal plain skid trails. Moehring and Rawls (1970) examined wet weather trafficking with a crawler tractor

on silt loams having a drainage restricting fragipan in Arkansas and reported bulk density increases of 13 percent and macroporosity decreases of 49 percent. Traffic conditions were replicated during the dry season and no significant changes in soil properties were detected. These findings suggest an advantage of operating under dry conditions, but are met with logistic challenges that would arise due to long periods of down time that would occur on poorly drained sites and wetlands.

Soil strength, sometimes referred to as mechanical resistance, provides an index of a soil's load bearing capacity and resistance to root elongation and reflects soil moisture transience (Busscher and others 1997). Soils exhibiting higher bulk densities tend to have higher soil strength when held at the same moisture content (Bradford 1986). Hatchell and others (1970) reported surface soil mechanical resistance values of 107.9, 205.9, 274.6, and 333.4 kPa for undisturbed areas, secondary skid trails, primary skid trails, and decks, respectively at various logging sites in the lower coastal plain of South Carolina and Virginia. A similar trend in soil strength across a traffic gradient was observed by Lockaby and Vidrine (1984) in the Louisiana coastal plain. The values reported are 28.6, 37.4, 42.0, 65.4, 70.7 kPa for undisturbed areas, road borders, secondary roads, primary roads, and decks, respectively. Increasing soil strength with disturbance intensity was also reported by Murphy and Firth (2004) in New Zealand and Jusoff and Majid (1992) in Malaysia. However, harvesting disturbance in wetlands may not always result in increased soil strength because volumetric water content is increased. Carter and others (2007) observed decreasing average soil strength as disturbance intensity increased in a South Carolina wet pine flat. Likewise, Aust and others (1998a) reported no significant increase in soil strength due to compaction in a wet pine flat following harvest. These studies suggest that soil strength changes resulting from harvest traffic are influenced by complex interactions of site-specific soil characteristics and temporally variable moisture content.

Additionally, site hydrology can be altered by timber harvesting. Water table rises following harvests in wet flats attributed to reduced transpiration rates have been reported as 21 cm (Xu and others 2002) and 32 cm (Sun and others 2000). Aust and others (1995, 1993) isolated the effect of soil disturbance on water tables in wet flats and reported rises of 8 to 43 cm. The suggested causes of elevated water tables were reduced hydraulic conductivity, reduced macroporosity, and the walls of ruts which restricted horizontal water movement. Regardless of the mechanism by which water tables rise, it compounds aeration deficits caused by macropore destruction. Startsev and McNabb (2009) investigated the effects of harvest-induced soil compaction on soil morphological indicators of drainage class and aeration

regimes across a drainage gradient in a Canadian boreal forest. Well-drained soils maintained sufficient aeration (based on a 10 percent macroporosity threshold), and drainage class was not affected. Moderately well-drained soils were most prone to become aeration-deficient and undergo a change in drainage class. Imperfectly drained (somewhat poorly drained) and poorly drained soils had restricting aeration values irrespective of compaction level, and morphological indicators did not suggest a change in drainage class.

Skidder trafficking both directly and indirectly influences chemical properties. Naghdi and others (2016) document significantly lower concentrations of organic carbon, phosphorus, nitrogen, and potassium on skidder-trafficked upland soils in Iran relative to adjacent, undisturbed soils. Increasing frequency of machine passes further reduced these nutrient concentrations. It is suggested that skidding directly influenced these chemical properties by displacing the litter and topsoil layers and mixing the topsoil with less fertile subsoil. Soil compaction may indirectly reduce available nutrient concentrations by altering the aeration regime (Greacen and Sands, 1980).

EFFECTS OF ALTERED SOIL PROPERTIES ON TREE ESTABLISHMENT AND GROWTH

Tree root growth is influenced by a variety of soil-related factors, including soil strength, aeration, water, and nutrient availability (Greacen and Sands 1980). Minimally disturbed forested wetland soils typically have low bulk densities and high macropore percentages in surface horizons, creating conditions favorable to tree root growth. In coastal plain wet pine flats, aeration is commonly a limiting factor to root growth (Allen and Campbell 1988). Vomocil and Flocker (1961) identified 10 percent as the macroporosity threshold at which roots have sufficient oxygen availability. Thus, destruction of aeration porosity by traffic under wet conditions has potential to further inhibit site productivity.

Roots larger than soil pores must physically displace soil to grow. If the forces exerted by roots cannot overcome the soil strength, growth will be limited (Greacen and Sands 1980). In a greenhouse study, Mitchell and others (1982) observed root and height growth of loblolly pine (*Pinus taeda*) seedlings inversely proportional to bulk density. Additionally, nutrient deficiency was reported in seedlings planted in soil with a bulk density of 1.8 g cm⁻³. More coarsely textured soils have larger pore radii than fine-textured soils and provide less resistance to root growth. Daddow and Warrington (1983) estimate a growth-limiting bulk density of 1.75 Mg m⁻³ for sandy soils and 1.4 Mg m⁻³ for clays. Sands and others (1979) report 3 MPa as a threshold penetration resistance value, above which radiata pine (*P. radiata*) roots are severely restricted.

Data from Lockaby and Vidrine (1984) demonstrate the effect of traffic intensity on tree survival and growth. Mean loblolly pine heights at age 5 years were 0.79, 1.2, 1.7, 1.7, and 1.95 m for decks, primary roads, secondary roads, road borders, and non-trafficked areas, respectively. Survival rates were 1,347, 1,905, 8,406, 10,092, and 15,696 trees/ha in the same respective order. In a South Carolina wet pine flat, Scheerer (1994) documented mean height and diameter at breast height (DBH) of 2-year-old loblolly pine seedlings as 0.6 m and 0.9 cm in non-trafficked plots. Mean values in primary skid trail plots were 0.2 m and 0.4 cm, for height and DBH, respectively. Primary skid trail plots exhibited significantly higher bulk density and significantly lower macroporosity and hydraulic conductivity. Naghdi and others (2016) exemplify a response of upland hardwoods to different traffic intensities. Seed germination rate, root growth, and height of velvet maple (*Acer velutinum*) decreased significantly with increasing traffic frequency.

In the lower coastal plain of South Carolina and Virginia, Hatchell and others (1970) found lower stocking and growth of naturally regenerated loblolly pine seedlings after one growing season in primary skid trails than undisturbed areas. Impaired aeration was suggested as the primary limiting growth factor, while increased bulk density and soil strength were also suspected of reducing growth. Moehring and Rawls (1970) reported significant reductions in basal area growth for 5 years in response to increased bulk density and decreased macroporosity following wet site trafficking.

Excess moisture stress is a common challenge to regeneration on wet flats (Allen and Campbell 1988). Following harvests, water tables rise in response to reduced transpiration as well as altered hydrologic properties of soil (Aust and others 1995, 1993; Sun and others 2000; Xu and others 2002). Elevated water tables, especially in combination with reduced aeration porosity, may prevent adequate oxygen diffusion in the rooting zone. The rise in water tables following harvest is another rationale for use of mechanical site preparation techniques such as bedding or mounding, which alleviate aeration deficits, allowing planted seedlings to survive until transpiration rates have recovered (Harms and others 1998).

NATURAL RECOVERY AND RESILIENCE OF SOIL PROPERTIES AND PRODUCTIVITY

The extent and timeframe for natural reconciliation of degraded soil properties and forest productivity is site-specific (Miwa and others 2004). Mechanisms attributed to natural recovery include freeze-thaw cycles, wet-dry cycles, shrinking and swelling of clays, bioturbation by soil fauna, rooting activity, and sediment inputs (Greacen and Sands 1980, Hatchell and Ralston 1971, Larson and Allamaras 1971, McKee and others 2012). Freeze-thaw, wet-dry, and shrink-swell cycles can accelerate the

formation of aggregates (Larson and Allamaras 1971). Wet-dry cycles may have a profound impact in wetlands due to cyclic hydrologic fluxes. Soil macrofauna and plant roots form macropores and incorporate organic matter to soil. Although burrowing and root expansion must compress soil in the immediate vicinity, macropores remain and the result is usually a decrease in average bulk density (Larson and Allamaras 1971).

Dickerson (1976) predicted a 12 year period for bulk density and macroporosity to recover to pre-harvest levels in wet weather skidder tracks in the Northern Mississippi coastal plain. Similarly, Hatchell and Ralston (1971) estimated bulk density recovery time of 18 years in the Atlantic Coastal Plain. Rab (2004) reported macroporosity increases of 100 percent 10 years after harvesting on primary skid trails, secondary skid trails, and log landings in an upland setting. However, significant bulk density decreases after 10 years were only found in secondary skid trails. Primary skid trail bulk densities remained 51 percent higher than undisturbed areas.

Some forested wetlands apparently have sufficient recovery mechanisms that make them robust to the effects of wet weather harvesting. McKee and others (2012) reported no negative effects on stand growth and composition 24 years after rutting in a tidal cypress-tupelo wetland. This occurred despite significantly reduced hydraulic conductivity and aeration immediately after harvest (Aust and Lea 1992). The resilience of this site is attributed to the improvement of soil physical properties through the shrinking and swelling of clays, improvement of aeration and nutrient additions from sediment deposition, and accidental hydrologic and microtopographic impacts of traffic that favored the growth of desirable species (McKee and others 2012). Eisenbies and others (2004, 2005) and Passauer and others (2013) concluded that wet weather harvesting was no more detrimental to stand biomass accumulation than dry weather harvesting after 5 and 16 years, respectively, in a South Carolina wet pine flat. Unintentional introduction of microtopography and competition suppression are suggested causes for the successful growth on wet harvested sites. One caveat to this study was the unusually dry season that followed planting, which may have eliminated the excess moisture stress factor that commonly threatens seedling survival on wet pine flats (Allen and Campbell 1988; Eisenbies and others 2004, 2005; Passauer and others 2013). Lang and others (2016) assessed soil physical properties at this site 17 years after harvest and detected no significant differences between wet and dry weather harvest treatments and determined that soil physical properties had generally recovered. Similarly, Tiarks (1990) concluded that wet weather harvesting did not reduce height nor diameter growth of slash pine relative to dry weather harvesting (when sheared) on

Caddo series soil. While soil properties were not directly measured for this study, it is suggested that undisturbed Caddo soils have an inherently low macroporosity percentage so trafficking does not result in reduced drainage or aeration.

MECHANICAL SITE PREPARATION TO ACCELERATE RECOVERY

Some wetland sites resist long-term losses in productivity as a result of altered soil physical properties (Lang and others 2016, McKee and others 2012, Passauer and others 2013). However, well-documented detrimental effects, slow recovery, and overall unpredictability of site-specific growth response to harvest disturbance justify the implementation of site preparation to mitigate potential impacts of wet site harvesting (Aust and others 1998b, Gent and others 1983, Hatchell and others 1970, Miwa and others 2004, Moehring and Rawls 1970, Reisinger and others 1988). Determination of appropriate regeneration and site preparation techniques requires coupling necessary ameliorative practices (Miwa and others 2004) with knowledge of inherent site characteristics and silvicultural objectives (Zhao and others 2009). Historically, common site preparation practices include drainage, chopping, burning, disking, subsoiling, bedding, fertilization, and herbicide application. Benefits provided by site preparation are improved aeration and nutrient availability to roots, competing vegetation control, additions of organic matter to surface horizons, control of logging slash distribution, and manipulation of soil physical and chemical properties (Lof and others 2012, Miwa and others 2004).

Previously, drainage by ditching was practiced on wetland sites to improve growth of pine species by increasing aerated soil depth (Allen and Campbell 1988). Existing drainages can be maintained, but construction of new drainage systems in jurisdictional wetlands is no longer a practical option due to permitting requirements of Federal regulations. Bedding, a more economically and legally feasible alternative to drainage, creates an elevated planting surface that can improve seedling survival and growth on wet sites by improving soil aeration, controlling competing vegetation, incorporating organic matter, and exposing mineral soil (Harms and others 1998, Lof and others 2012, Miwa and others 2004, Reisinger and others 1988). Hatchell (1981) reported survival and growth gains at age 4 in bedded plots relative to disked and non-mechanically prepared plots on poorly drained flats in South Carolina. A difference in growth on compacted and uncompacted bedded plots was not evident, suggesting that bedding mitigated the compaction. At age 12 years, the bedded plots still exhibited superior height and basal area (McKee and Hatchell 1986). McKee and Wilhite (1986) observed the effects of bedding and fertilization on loblolly pine across a drainage gradient and found

survival increases at age 10 years due to bedding were 7 percent, 19 percent, and 15 to 19 percent on moderately well-drained, somewhat poorly drained, and poorly drained sites, respectively. Height gains at age 2 years due to bedding were 33 percent and 18 percent on the moderately well-drained and poorly drained sites, respectively, but gains diminished to 7 percent more than controls by age 10 years on both sites. On poorly drained sites, heights remained 37 to 42 percent greater than controls through age 10. Aust and others (1998b) reported that bedding successfully mitigated the damaging effects of compaction and rutting on loblolly pine survival and growth at age 4 in wet pine flats. Similarly, Eisenbies and others (2004) documented greater biomass accumulation at age 5 on bedded sites than flat planted sites.

Growth gains as a result of bedding may diminish over time. Zhao and others (2009) reported significant slash pine (*P. elliotii*) volume gains due to bedding until an age of 20 years on flatwoods spodosols. At stand age 33, Kyle and others (2005) did not find significantly different productivity levels of loblolly pine among bedded, ditched, and chopped treatments. At the same site, bedded plots exhibited significantly greater productivity metrics than chopped-only plots at age 21 years (Andrews 1993). These declines in tree growth advantages over time on poorly drained sites likely occurred because non-bedded areas eventually develop sufficient evapotranspiration rates to lower the water table and overcome excess moisture limitations (Kyle and others 2005).

Although bedding can provide advantages for tree growth, it may fail to restore or further deteriorate soil physical properties on a short-term basis. Gent and others (1983) measured soil physical properties pre-harvest, post-harvest, and post-site preparation (shear, burn, chop, and bed) in the lower coastal plain of North Carolina. Bedding did not decrease bulk density, but decreased macroporosity and hydraulic conductivity at the soil surface. Aust and others (1998b) reported significantly reduced hydraulic conductivity in non-trafficked, bedded plots relative to non-trafficked, non-site-prepared plots. On trafficked plots, however, bedding increased macropore space compared to non-site-prepared plots. Overall, research suggests that the greatest advantage to bedding is the creation of elevated planting surface microtopography that improves seedling survival by ameliorating excess moisture limitations (Aust and others 1998b, Passauer and others 2013).

Disking has been implemented in attempts to alleviate compaction and rutting associated with forest harvesting. Disking is intended to mechanically loosen soil, incorporate organic matter, and expose mineral soil (Miller and others 2004). On an upland soil in the North Carolina piedmont, Gent and others (1984)

reported successful restoration of bulk density and macroporosity in surface horizons as a result of disking. However, hydraulic conductivity was reduced by disking. On wetland sites, disking may be a much less effective method of mitigating damage to soil physical properties. Aust and others (1998b) found that disking failed to improve surface bulk density, macroporosity, and hydraulic conductivity values in trafficked areas of a wet pine flat. Furthermore, disking significantly decreased macroporosity and hydraulic conductivity in non-trafficked areas. Essentially, disking the undisturbed soil created a puddling effect by eliminating macropores. McKee and Shoulders (1974) found no significant difference in soil redox potential, depth to water table, and total aboveground biomass of slash pine on disked and non-site-prepared plots on medium- to slowly drained soils in Louisiana. The study did not directly measure soil physical properties, but redox potential and depth to water table provide insight to soil aeration. In a study on a similar site, Mann and Derr (1970) also reported no significant height growth gains in slash pine at age 8 years due to disking. However, disking increased loblolly pine heights by an average of 0.67 m at age 8 years.

SUMMARY AND CONCLUSIONS

Forested wetlands provide ecologically and economically valuable services. Expanding human development enforces a need to conserve these services while maintaining timber management as a viable, sustainable option on wetland sites. However, the frequent high moisture content of forested wetland soils makes them particularly vulnerable to soil compaction and rutting under the forces applied by heavy machinery. Compaction and rutting have degrading impacts on soil physical properties which are controlling factors of wetland dynamics and function. Persistence of harvesting impacts on soil properties varies with site-specific factors, and the potential for long-term impairment validates implementation of ameliorative practices to speed the restoration of soil properties and wetland services.

Ideally, forested wetland timber harvesting should be scheduled to avoid high soil moisture conditions. However, this is logistically infeasible due to the prevalence of low lying terrain, long wet seasons, and limited wood storage capacities. Several studies have quantified the effect of harvesting soil disturbance and site preparation on tree growth over time, but few have maintained long-term analysis of soil properties. The few long-term studies that have evaluated long-term site preparation and traffic effects in wetlands invariably found that soil physical properties changed over time. Thus, it would be beneficial to reevaluate site preparation and trafficking studies, while considering inherent site properties that may act as natural repair mechanisms. This may improve the ability to predict long-term impacts

on soil properties and tree growth, allowing forest management decisions to be made efficiently.

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SURVIVAL, GROWTH, AND ESTABLISHMENT OF PLANTED SHORTLEAF PINE AND NATURAL HARDWOOD REGENERATION ON SCARIFIED AREAS IN PARTIALLY CUT STANDS

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Abstract—Establishment of shortleaf pine-hardwood forests on poor to moderate sites in the Cumberland Mountains and Plateau is possible only by planting shortleaf pine seedlings in areas without a seed source. Establishment of small clusters (≤ 0.02 acres) of shortleaf pine at tight spacings on scarified areas such as skid trails following partial harvests may preclude the need for site preparation and release treatments, reintroduce a scattered shortleaf component, and offer a low cost option for landowners interested in mixed stands. Shortleaf pine survival and growth as well as natural regeneration composition and size were assessed in 30 clusters established on former skid trails at a site on Little Brushy Mountain, TN. Overall survival of planted shortleaf seedlings was 53.6 percent, and survival increased on more scarified areas, while differences in soil compaction did not significantly affect seedling survival. Natural hardwood regeneration displayed no differences in stem density or height between scarified and non-scarified areas. Oak spp. and red maple were more prevalent inside scarified areas than outside of them, whereas yellow-poplar was more prevalent outside of scarified areas than within them.

INTRODUCTION

Shortleaf pine (*Pinus echinata*) and associated pine-hardwood forest types have been declining for decades east of the Mississippi River. Data have shown that forest types containing a shortleaf pine component have declined by 4.4 million acres from 1953 to 1997 (South and Buckner 2003). This trend is no different in Tennessee as Forest Service Forest Inventory and Analysis data have shown that removals have exceeded volume growth (Moser and others 2007, Oswalt 2012). Shortleaf pine removals have occurred due to urbanization, pine beetle outbreaks, mesophication, fire suppression, and land use changes (Brose and others 2001, McWilliams and others 1986, Oswalt and others 2016).

Establishment of shortleaf pine on suitable sites in Tennessee will necessitate artificial regeneration in areas without a seed source. Partial harvests (either silviculturally sound methods or high-grading) of even-aged hardwood stands that historically had a shortleaf pine component are common practices by landowners and loggers. Establishment of a new shortleaf pine cohort may be feasible under partial hardwood overstories using cluster plantings, which are small groups (< 0.02 acres) of seedlings planted at small spacings (5 x 5 feet or less) at a variety of

densities and configurations to increase stocking or introduce desirable species to a stand (Anderson 1951, Saha and others 2012). Shortleaf pine's intermediate shade tolerance as a seedling may make it amenable to this type of management (Kabrick and others 2011). Perpetuation of two-aged, mixed stands may also offer landowners more management opportunities as these stands develop depending on local markets and objectives.

Cluster planting on scarified areas such as skid trails in partially harvested stands may be a viable and economical site preparation method to introduce a shortleaf pine component along with natural regeneration of upland hardwood species on these sites. Scarified areas can constitute approximately 20–35 percent of the area within a harvest site depending on topography and harvest type (Buckley and others 2003, Marquis and Bjorkbom 1960, Martin 1988). Scarification can reduce the influence of competing vegetation (Nyland 1996, Prevost 1997) and it can increase soil moisture and temperature, which can improve seedling growth (Löf and others 2012). One drawback to planting on scarified areas could be increased soil compaction and greater soil bulk density effects on seedling survival and growth (e.g., Dickerson 1976).

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OBJECTIVES

The objectives of this study were (1) to investigate the survival and growth rate of 2-year-old shortleaf pine seedlings planted on former skid trails with varying levels of soil scarification and compaction; (2) to determine if shortleaf pine survival and growth on scarified areas can be predicted from a suite of abiotic variables; and (3) to assess and compare the density, composition, and growth of naturally regenerating stems outside and within scarified areas.

SITE DESCRIPTION

The study site was located on a 76-acre property owned by the University of Tennessee Forest Resources Research and Education Center in Morgan County, TN (36.053955° N, -84.435428° W) and was part of a larger ongoing study. This site was located on the western and northwestern flanks of Little Brushy Mountain, which is near the southern terminus of the Cumberland Mountains physiographic province (Smalley 1984). Elevations ranged from 1,280 to 1,840 feet. Parent materials consisted of shale and siltstone, while soils included: Gilpin-Boulin-Petros complex, 25–80 percent slopes, very stony; Lily-Gilpin complex, 20–35 percent slopes; and Gilpin-Petros complex, 20–35 percent slopes. These soils were all rocky silt loams, loams, and clay loams, and depth to bedrock ranged from 25–43 inches. The site index for shortleaf pine is 63 feet at base age 50 years for the Lily-Gilpin complex (Martin 1966, U.S. Department of Agriculture 2017). Mean annual temperature for the area is 55.9 °F, and the region averages 54 inches of precipitation annually with autumn as the driest season (National Oceanic and Atmospheric Administration 2017). The site had remained relatively undisturbed since the university conducted a stand improvement harvest in 1949, except for cyclical southern pine beetle (*Dendroctonus frontalis*) outbreaks, the most recent of which occurred from 1999–2002 across the State (Cassidy 2004).

Vegetation on the site prior to operations consisted of upland hardwoods with only 12 mature shortleaf pine stems across the entire 76 acres. Common overstory species included: black oak (*Quercus velutina*), white oak (*Q. alba*), scarlet oak (*Q. coccinea*), chestnut oak (*Q. montana*), mockernut hickory (*Carya tomentosa*), yellow-poplar (*Liriodendron tulipifera*), and red maple (*Acer rubrum*). Common midstory and understory species included: sourwood (*Oxydendrum arboreum*), blackgum (*Nyssa sylvatica*), serviceberry (*Amelanchier arborea*), white pine (*P. strobus*), and red maple. The site averaged 7,108 board feet per acre sawtimber (Doyle log rule form class 78 and 80). Prior to harvesting operations beginning, oak and hickory retention trees were marked so that 15–20 square feet per acre basal area would remain in these species. A commercial harvest began in

October 2014 and was completed in April 2015. The site averaged 1,552 board feet per acre (Doyle log rule form class 78 and 80) and 52 square feet per acre basal area following the harvest.

METHODOLOGY

During and after harvest operations, thirty 28- x 28-foot clusters were established in skid trails and other scarified areas in three repetitions of 10 clusters at all elevations of the study site. Portions of any cluster that were not scarified by logging machinery were scarified by hand using fire rakes. Within each cluster, all vegetation ≥ 3 feet tall was cut (if present). On April 9, 2015, sixty-four 1-0 stock, bareroot shortleaf pine seedlings purchased from the Tennessee Department of Agriculture Division of Forestry were planted by planting crews at a 4- x 4-foot spacing within each of the 30 clusters.

Soil compaction was measured for 32 of the 64 seedlings within each cluster using a DICKEY-john® soil penetrometer that measured on a pounds-per-square-inch (PSI) scale from 0–350. All compaction measures were taken during appropriate conditions within 24 hours of heavy rainfall events (Duiker 2002), and readings were taken approximately 6 inches from the seedling as soil conditions allowed (lack of rocks to a 12-inch depth). A visual rating system with four levels (level 1 the least scarified and level 4 the most scarified) for the amount of scarification similar to Dyrness (1965) was completed for 32 of the 64 seedlings within each cluster. Compaction levels were defined as follows:

- Level 1 was characterized by only leaf litter removal, and some organic material (O horizon) may have been disturbed (<20 percent). None of the A horizon or deeper horizons in the profile were exposed.
- Level 2 was characterized by leaf litter removal, and the organic and A horizons were incorporated into one another. Incorporation of these horizons was due to machinery traffic. This level was characteristic of areas where only one to three stems were skidded out.
- Level 3 was defined by removal or incorporation of the organic and A horizons, and the Bt horizon(s) was exposed. Machinery compaction was not severe (PSI <250).
- Level 4 was defined by incorporation or removal of the organic and A horizons and exposure of the Bt horizon(s). These areas were more compacted by machinery than the level 3 classification (PSI ≥ 250).

All visual ratings were taken within the square foot around the seedling with the seedling as the center point.

Other measurements taken within each cluster included aspect, slope, and elevation. Photosynthetic photon flux density levels ($\mu\text{mol s}^{-1} \text{m}^{-2}$) were taken using an Accupar Linear PAR/LAI Model PAR-80 ceptometer (Decagon Devices, Pullman, WA) from July to August 2016 at five set points within each cluster and at a fixed height (4 feet) at each point and divided by values taken in areas unshaded by vegetation for a percent sunlight value. All readings were taken during 1045 to 1430 hours on days with suitable weather conditions (Messier and Parent 1997, Parent and Messier 1996). An average value was computed from the five measures taken in each cluster. Shortleaf pine seedlings were assessed for survival and measured for basal diameter and height during December 2016 in all 30 clusters. Natural regeneration ≥ 6 inches but < 1 -inch diameter at breast height was assessed within the center of nine clusters using a $1/300^{\text{th}}$ -acre circular plot, while natural regeneration outside of these clusters was assessed 26.7 feet from the center of the cluster in one $1/300^{\text{th}}$ -acre circular plot located in a non-scarified area. This plot was randomly located parallel to the slope in one of two directions as long as the area was not scarified. Natural regeneration was grouped to species and classified into 6-inch height classifications.

STATISTICAL ANALYSIS

Overall shortleaf pine seedling survival was reported and analyzed by scarification rating and compaction level using chi-square tests ($\alpha = 0.05$). Average shortleaf pine seedling basal diameter and height were reported. Shortleaf pine seedling survival, basal diameter, and height were predicted using multiple linear regression with a random factor (repetition). Independent variables included: compaction (PSI), scarification rating, aspect, elevation, slope, and percent sunlight. Backwards variable elimination was used to choose significant variables, and variables were considered significant at an alpha level of $p = 0.05$. The predictive capabilities of the

multiple regression models for each dependent variable were validated using 25 random holdout samples that each used 25 percent ($n = 240$) of the observations in the dataset. Validation runs correlating observed to predicted values that had R-square values ≥ 0.7 and p -values < 0.05 were considered strong predictors. Natural regeneration density and height comparisons for inside and outside of scarified areas were analyzed using two-sample t-tests ($\alpha = 0.05$). Natural regeneration species composition by percentage and stems per acre are also reported for inside and outside of scarified areas. All statistical analyses were performed using SAS 9.4 (SAS Institute 2012).

RESULTS

Two full growing seasons following planting, shortleaf pine seedling survival was 53.6 percent. No differences in survival were found for survival by compaction level ($p = 0.66$), and survival ranged from 45.0–66.7 percent based on PSI level. Differences in survival by scarification rating were significant ($p = 0.011$) and ranged from 46.0 to 62.4 percent (table 1). Seedling survival was significantly affected by scarification rating, elevation, and slope ($p < 0.05$) (table 2). The 25 holdout sample validation runs found that 24/25 runs had R-square values ≥ 0.7 . This indicates that these three variables may be able to predict shortleaf pine seedling survival. Average basal diameter and standard deviation was 0.4 ± 1.06 inches, and height was 28.5 ± 13.1 inches. The regression results suggest that PSI, light, and elevation all significantly affected basal diameter growth ($p \leq 0.05$) (table 3), yet these variables could not be used for prediction purposes because all of the validation runs had R-square values < 0.7 . Seedling height was significantly affected by light and elevation ($p < 0.05$), yet could not be used for prediction purposes based on the validation runs as only 1/25 of the runs could predict height (table 4).

Table 1—Shortleaf pine seedling survival rate by scarification level two growing seasons after planting for the mixed shortleaf pine-hardwood establishment study in Morgan County, TN

Rating ^a	Survival percent	N
1	52.9	354
2	46.0	179
3	53.3	194
4	62.4	193

^a Based on the Chi-square test, survival differences among the four scarification ratings were found ($p = 0.011$).

Table 2—Shortleaf pine seedling multiple regression predictor variables for survival for the mixed shortleaf pine-hardwood establishment study in Morgan County, TN ($n = 960$)

Effect	Estimate	Standard error	Degrees of freedom	t-value	Prob-t
Rating	0.05	0.018	911	2.7	0.006
PSI	5.89E-06	0.0003	911	0.02	0.9
Light	-0.0008	0.0008	911	-0.9	0.3
Direction	0.001	0.008	911	0.1	0.9
Elevation	-0.0002	0.0001	911	-2.0	0.04
Slope	0.007	0.003	911	2.7	0.007

Table 3—Shortleaf pine seedling multiple regression predictor variables for diameter for the mixed shortleaf pine-hardwood establishment study in Morgan County, TN (n = 515)

Effect	Estimate	Standard error	Degrees of freedom	t-value	Prob-t
Rating	0.01	0.01	465	1.3	0.2
PSI	-0.0003	0.0002	465	-2.0	0.05
Light	0.001	0.0004	465	2.9	0.004
Direction	0.008	0.005	465	1.8	0.07
Elevation	0.0008	0.0002	465	5.3	<0.0001
Slope	0.0003	0.002	465	0.2	0.8

Table 4—Shortleaf pine seedling multiple regression predictor variables for height for the mixed shortleaf pine-hardwood establishment study in Morgan County, TN (n = 515)

Effect	Estimate	Standard error	Degrees of freedom	t-value	Prob-t
Rating	0.008	0.63	465	0.01	0.9
PSI	-0.02	0.01	465	-1.66	0.09
Light	0.06	0.03	465	2.10	0.04
Direction	0.4	0.3	465	1.48	0.1
Elevation	0.03	0.009	465	3.61	0.0003
Slope	-0.02	0.1	465	-0.17	0.9

Average density of natural regeneration was not significantly different between scarified areas and non-scarified areas ($p = 0.662$) (fig. 1). Average height of natural regeneration did not differ between scarified and non-scarified areas either ($p = 0.203$) (fig. 2). Deerberry (*Vaccinium stamineum*) (28 percent), yellow-poplar (10 percent), and sourwood (9 percent) were the three most prevalent individual woody species regenerating in areas exterior to the scarified area and comprised 47 percent of the natural regeneration (fig. 3). Yellow-poplar (20 percent), mountain laurel (*Kalmia latifolia*) (15 percent), scarlet oak (10 percent), and chestnut oak (10 percent) were the four most prevalent individual species regenerating within scarified areas and comprised 55 percent of the natural regeneration (fig. 4).

DISCUSSION AND CONCLUSIONS

Planted shortleaf pine survival tended to increase on areas with higher scarification ratings. Reduced vegetation competition and influence as well as greater soil moisture availability are possible explanations for this trend (Löf and others 2012). Rating levels 1 and 2 did have residual vegetation after the harvest and study establishment were complete. In addition, less herbaceous vegetation cover tended to be present

on levels 3 and 4 during both growing seasons than on levels 1 and 2, though this was not quantitatively assessed. The results for survival by compaction and scarification rating indicate that shortleaf pine can tolerate a wide variety of physical soil properties, because survival rates did not differ by more than 16.4 percent by scarification rating or 21.7 percent by compaction level. The overall survival rate after two growing seasons (53.6 percent) is similar to the overall survival rate in another study that investigated planting bareroot shortleaf pine seedlings under differing levels of partial overstory shade in Missouri. Though scarification was not addressed in the Missouri study, Kabrick and others (2015) found a survival rate of 58.8 percent across different amounts of partial overstory stocking for underplanted 2-year-old, bareroot shortleaf pine seedlings. Seedling survival in this study was likely negatively affected by a severe drought during the second growing season from the third week of August to the last week of November 2016 when precipitation was only about 22.6 percent of the yearly average during the autumn season (National Oceanic and Atmospheric Administration 2017). In addition, redheaded pine sawfly (*Neodiprion lecontei*) defoliation was found on 3.4 percent of living seedlings and is suspected of

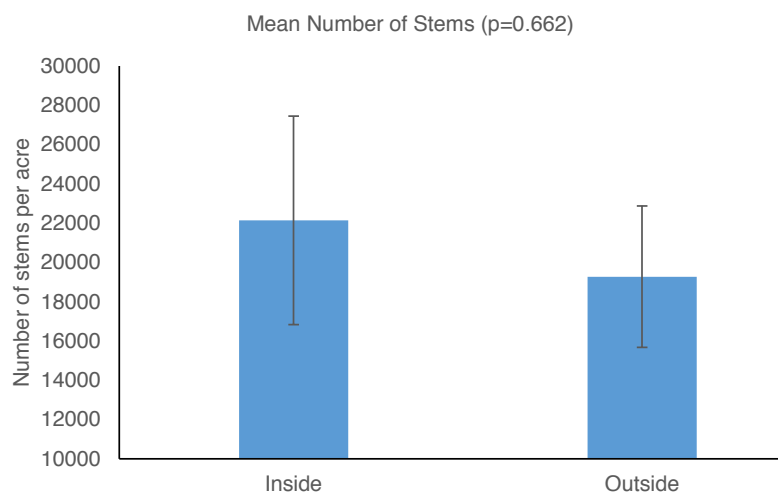


Figure 1—Mean number and standard error of regenerating stems from 6 inches to 1-inch diameter at breast height per acre for the mixed shortleaf pine-hardwood establishment study in Morgan County, TN ($n = 9$ plots).

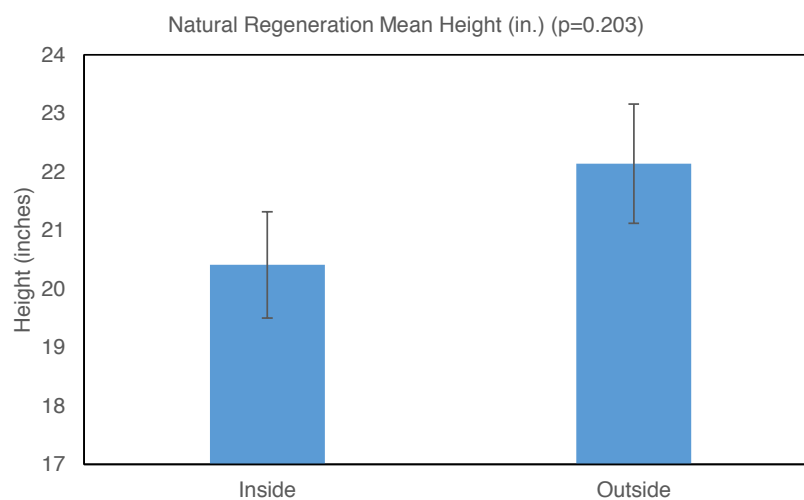


Figure 2—Mean height and standard error of regenerating stems inside and outside of scarified areas based on 6-inch height classes for the mixed shortleaf pine-hardwood establishment study in Morgan County, TN ($n = 9$ plots).

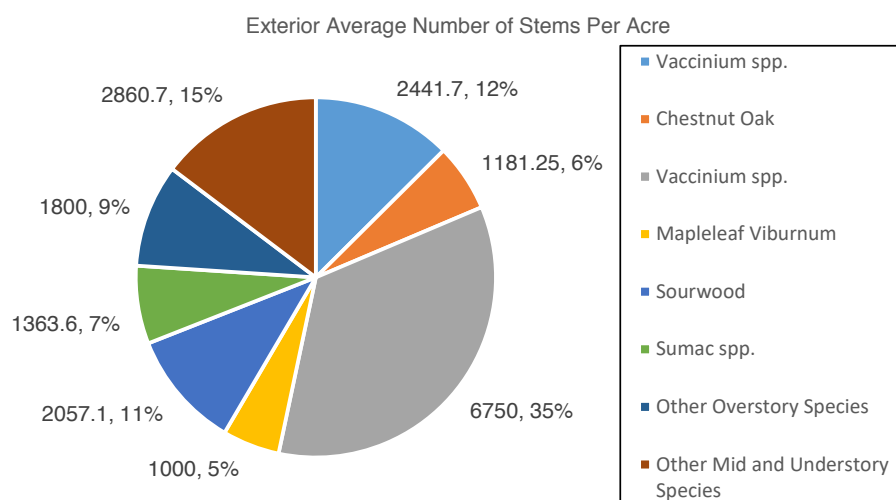


Figure 3—Species composition by number of stems per acre and percentage by species outside of scarified areas for the mixed shortleaf pine-hardwood establishment study in Morgan County, TN (n = 9 plots).

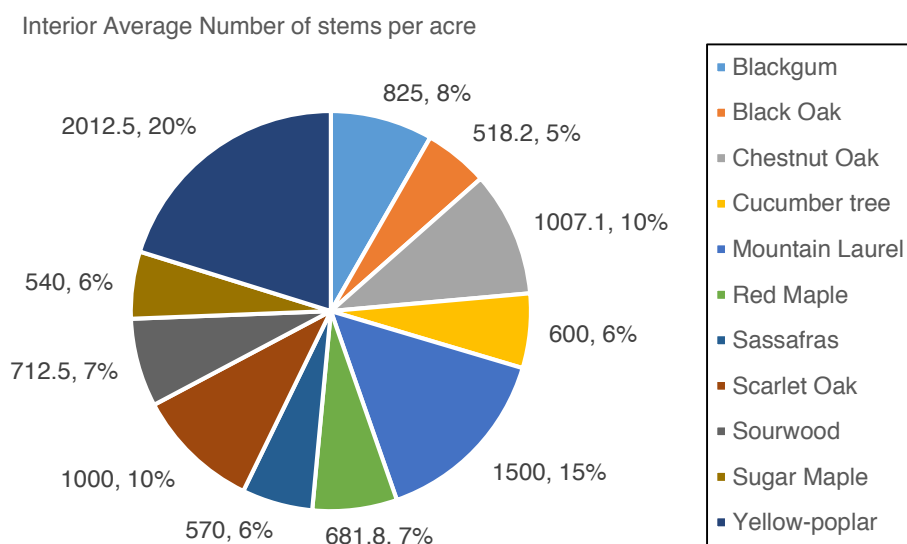


Figure 4—Species composition by number of stems per acre and percentage by species inside of scarified areas for the mixed shortleaf pine-hardwood establishment study in Morgan County, TN (n = 9 plots).

killing some of the non-surviving stems. The regression results indicate that elevation, slope, and scarification rating may be used to predict shortleaf pine survival. Steeper slopes, greater scarifications ratings, and lower elevations tended to be better for seedling survival.

Average annual shortleaf pine seedling height growth (14.2 inches) was at the lower end of normal height growth (1–3 feet) (Williston 1972). Smaller than expected average seedling heights are likely due to the aforementioned environmental stressors and increased soil compaction. Basal diameter growth may have been smaller than expected due to limited primary and lateral root expansion (Carlson and Harrington 1987). Prediction of shortleaf pine height and diameter was not possible based on the validation results, but the overall model did suggest that lower light levels negatively affect growth height and diameter growth, which was supported by Kabrick and others (2011). Increased elevation also tended to positively affect seedling growth for both variables.

No differentiation had occurred yet in number of stems per acre or average stem height for natural regeneration 2 years post-scarification. Scarification tended to increase the abundance of species such as oaks, red maple, and mountain laurel, whereas species such as yellow-poplar and *Viburnum* spp. decreased in abundance within scarified areas (figs. 3 and 4).

Shortleaf pine seedling survival was positively influenced by greater amounts of site preparation scarification associated with logging on this site, and survival after 2 years can be related to elevation, scarification rating, and slope measurements. Natural regeneration density was abundant within scarified and non-scarified areas 2 years following harvest, and scarification may increase the abundance of desirable species regeneration. Future monitoring will be necessary to determine if these trends continue and if scarification left by logging machinery following a partial harvest and planting of shortleaf pine seedlings is a viable method to reintroduce shortleaf pine to suitable sites within the Cumberland Mountains physiographic province.

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Loblolly Pine: Density and Competition Control

Moderator:

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VARIATION IN LOBLOLLY PINE CROWN CHARACTERISTICS BETWEEN TWO GENETIC IDEOTYPES AT AGE 8

Valerie S. West, Randall J. Rousseau, and Scott D. Roberts

ABSTRACT—The drive behind the development of elite loblolly pine (*Pinus taeda* L.) genotypes may include potential increases in stand uniformity and reduction in planting densities and corresponding establishment costs. However, some genotypes produce less desirable crown and branch characteristics than others. The ability to fully realize potential genetic gains is dependent on selecting appropriate combinations of genetic material and silvicultural management. A study was established in 2008 to examine the performance of two related loblolly pine varieties at different initial tree spacing and management intensities. After eight growing seasons, morphological differences in crown structure are apparent between genetics and silvicultural intensity. The crop tree ideotype had, on average, greater crown volume and less acute branch angles. Environment had greater impact on crown characteristics than genotype. Differences due to management intensity were related to crowding from competing vegetation and lower incidence of damage from pine tip moth (*Rhyacionia frustrana*) and sawfly (*Neodiprion* spp.).

INTRODUCTION

Loblolly pine (*Pinus taeda* L.) is the dominant species component of the southern yellow pine timber market. The flexibility of this species in terms of overall production and merchantability has made it an ideal candidate for genetic improvement. Since the establishment of regional tree improvement cooperatives in the 1950s, great advances have been made in volume production, stem form, overall wood quality, and disease resistance from those first open-pollinated selections through the use of single half-sib family blocks and mass-controlled pollination (MCP) full-sib families (Bramlett 2007; Duzan and Williams 1988; Fox and others 2007; Jansson and Li 2004; Jokela and others 2010; Li and others 1999, 2000; McKeand and others 2003, 2006a, 2006b). With each successive increase in yield, a complementary increase in the intensity of site preparation and management becomes essential to the success of achieving the full benefit of the genetic gains (Allen and others 2005, Bettinger and others 2009). As forestry practices began to shift toward an agricultural establishment model of intensive site preparation and management, the expectation of uniformity of product has been the goal of many landowners. The development of varietal materials from the most elite crosses has been undergoing rigorous testing for over

2 decades under a variety of field conditions (Albaugh and others 2016, Gleed and others 1995, Li and others 1991, Pile and others 2016, Wright and Dougherty 2007).

One of the potential benefits of varietal material is the ability to utilize the concept of the ideotype to define the growth habit of a given clone. The ideotype concept summarizes the phenotypic characteristics of the tree's growth habit into categories that establish a standard growth strategy for the tree in a consistent and predictable manner (Dickmann and others 2010, Donald 1968, Martin and others 2001). Three main ideotype categories have been established for trees based on crown characteristics such as branch size, branch angle, number of branches, and tendency to self-prune (Cannell 1978, Donald and Hamblin 1976, Martin and others 2001). The crop ideotype growth strategy is to efficiently exploit locally available resources while experiencing little competition with neighboring trees in terms of site resources. Trees with a crop ideotype tend to have narrow crowns and small branches and grow well without excessively competing for site resources with other similar trees. The crop ideotype is predicted to produce the greatest yield per unit area (Cannell 1978). The competition ideotype, or competitor ideotype, exploits site resources through an expansive crown and

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root system, interferes with the growth of its neighbors, and has the greatest individual tree growth. The isolation ideotype is applied to trees that perform best in young stands when intraspecific competition is minimal. This third ideotype classification was not investigated in this study. Choosing an ideotype to plant relies heavily on the management objectives of the given stand (Zhai and others 2015).

Numerous studies have examined the performance of varietal ideotypes at both the individual tree level and in clonal blocks under a variety of management strategies across the Southern United States. (Albaugh and others 2016, Allen and others 2005, Bettinger and others 2009, Martin and others 2001). Clonal plantations are expected to have greater stand uniformity, but few landowners take into consideration that this uniformity comes at the financial cost of eliminating site-specific resource limitations (Yáñez and others 2015). There may also be an increase in the frequency of genotype by environment interactions when clones are utilized in intensively managed plantations (Li and others 1991, Sierra-Lucero and others 2003). Martin and others (2001) have noted several spatial and temporal scale issues when applying the ideotype concept to tree development. This disconnect between the temporal and spatial scales of inquiry, and those same scales for application, has long been noted as an impediment to the complete understanding of the structural and functional bases of growth knowledge that is necessary for the development of ideotypes (Garcia Villacorta and others 2015, Hinckley and others 1997, Martin and others 2001). The long life span of trees, even under a rotational management regime, is another impediment to tree ideotype development. At each stage of development, trees express traits following growth patterns that differentiate with age class. What is described as 'optimal' resources for growth differs between the seedling, sapling, and mature growth stages of individual trees (Martin and others 2001). Farnsworth and Niklas (1995) observed that plants grow additively; therefore, current morphological traits tend to constrain the future morphology of a plant. The tree canopy structure that is essential to defining an ideotype exhibits spatial and temporal scale challenges. Tree canopies are difficult to study over time due to the age-related variations in the vertical and horizontal aspects of tree canopies which tend to be far more complex than that of annual crop plants (Parker 1995). This makes quantification more difficult for tree canopy architecture than for agronomic crops because there are growth impacts at each morphological phase of stand development (Cochrane and Ford 1978, Ford 1982, Martin and others 2001).

Questions remain as to the impact of less than 'optimal' growth resources on varietal pines. What is the impact of site history on varietal pines? Will management of only

the non-pine competition impact crown growth? Does spacing play a part in maintaining crown ideotype? The present study examines variation in crown architecture with respect to crown length, volume, branch angle, and branch diameter for two related varieties classified as a crop ideotype and competitor ideotype at age 8 under different spacing and management regimes. We hypothesized that each ideotype would adhere to the defined parameters for the given ideotype. We further hypothesized that the competitor ideotype would have greater overall crown volume and larger branch diameters than the crop ideotype regardless of stocking level.

METHODS AND MATERIALS

The study was established in 2008 at Mississippi State University's Coastal Plain Branch Experiment Station near Newton, MS (32° 20' 19"N, 89° 05' 51"W). Soils on the site are a Prentiss very fine sandy loam. The history of this site included agricultural production resulting in a defined Ap soil horizon. Pre-plant treatments included a broadcast application of glyphosate (64 ounces per acre) and 14-inch subsoil tillage in the fall of 2007. A second application of glyphosate (32 ounces per acre) was applied in March of 2008 prior to hand planting with containerized seedlings between April and May of 2008.

The study was set up as a 2 x 2 x 3 factorial split-plot design with four replications. Main effects treatments included two genetic varieties of loblolly pine and two levels of management intensity, with main effects treatment plots split into three subplot initial planting spacings. Trees within the spacing subplots were planted in 64-tree blocks (8 x 8 trees). Two half-sib varietal genotypes of loblolly pine, produced by ArborGen, LLC, were used in this study. One varietal was considered to be a competitor ideotype characterized by a wider crown form, and the other was considered to be a crop tree ideotype with a narrower, compact crown form. The two levels of management included normal intensity (N) and intensive (I). All plots received an additional herbaceous competition control treatment in year 1 through a broadcast application of Oustar® (10 ounces per acre). The intensive management plots were treated with additional competition control and pesticide applications. These applications included tip moth control in the form of a single 20-mg SilvaShield™ tablet (Bayer Environmental Science) in the planting hole at time of planting, PTM™ insecticide (BASF Corp.) injected 3–6 inches deep in the soil adjacent to each tree (0.05 ounces active ingredient per tree) in years 2 and 3 for additional tip moth control, herbaceous competition control in year 2 (1 ounce per acre of Escort®, 16 ounces per acre of Arrow®, 32 ounces per acre of Goal®), and mowing of competing vegetation in year 3. The three initial tree spacing levels were 6 x 14 feet [519 trees per acre (TPA)], 9 x 14 feet (346 TPA), and 16

x 14 feet (194 TPA). Initial height was measured on each seedling immediately following planting. Survival was assessed and heights measured annually following each growing seasons (2008–2016).

In 2016, an assessment of crown characteristics was undertaken to assess the impacts of management intensity and spacing on each of the varietal genotypes. Using an aerial lift, a random subsample of six trees per subplot was measured for branch collar diameter at the stem (cm) and branch angle using the tree stem as the 90° reference point. A branch was randomly selected from each live whorl both within the row and between the rows to a 3-inch top or 10 whorls whichever came first. The length of the live crown (CL) was calculated by subtracting the height to live crown from the total tree height. Crown widths (CW) were measured within and between rows to be used to calculate crown volumes. Crown volume (CV) for each tree was calculated as the volume of a parabolic cone:

$$CV = 2(\pi CW^2 CL) / 15 \quad (1)$$

where

CV is crown volume in cubic feet

CW is the projected area of the crown in square feet

CL is live crown length in feet

Data analyses were performed using a mixed models approach modified for the split-plot design with two main plot effects (PROC MIXED; SAS Institute 2012). Results presented here include the means for crown length, crown width, crown volume, branch angle, and branch diameter at the end of the eighth growing season. Significant results were further analyzed by Tukey-Kramer means tests to determine treatment differences based on a critical value of alpha = 0.05.

RESULTS

The interactions between genotype by management intensity by stocking were statistically chained such that definitive results could not be reasonably obtained. Further analysis for each ideotype by management intensity within a given row spacing treatment was significant for the majority of crown measurement variables (table 1). Trees of both ideotypes under intensive management produced significantly larger crown lengths, crown widths, and corresponding crown volumes than the same ideotypes under normal management intensity. Branch diameters tended to be somewhat larger under intensive management than normal management for both ideotypes. Branch angles were significantly more acute for the crop ideotype under intensive management in comparison to the normal management intensity. The competitor ideotype branch angles remained similar across management intensity level.

Table 1—Mean crown dimensional measurements at age 8 for two half-sib loblolly pine ideotypes in the Upper Coastal Plain of Mississippi at two management intensities and three spacing levels

Ideotype treatment ^a	Within-row spacing	Crown length feet	Crown width feet	Crown volume cubic feet	Branch diameter inches	Branch angle degrees
Intensive management						
Crop	16	20.3b ^b	16.3b	2327.6b	0.61a	50.4a
	9	18.9b	13.5b	1481.5b	0.56	48.3b
	6	17.6ab	11.9b	1068.8b	0.58a	45.9b
Competitor	16	22.9a	17.8a	3124.8a	0.61a	50.7a
	9	20.8a	14.5a	1859.9a	0.54	49.1a
	6	19.6a	12.9a	1393.3a	0.48b	46.5ab
Normal management						
Crop	16	14.2c	11.9c	913.1c	0.59a	46.7b
	9	14.6c	11.2c	891.2c	0.55	46.9c
	6	13.2c	10.1c	641.1c	0.51a	44.1c
Competitor	16	18.4b	15.4b	1962.2b	0.50b	50.9a
	9	17.8b	13.9b	1498.4b	0.53	51.6a
	6	16.9b	12.7ab	1174.6b	0.52a	48.8a

^a All values are statistically chained for ideotype by management by spacing at an alpha = 0.05 significance level.

^b Letters denote statistical differences between ideotype and management intensity levels within a given spacing level at an alpha = 0.05 significance level.

When averaged across spacing levels for each treatment, the intensive management resulted in trees that averaged over 1.1 feet (~24 percent) taller with wider crowns (0.7 feet, 30 percent), longer crowns (0.9 feet, 32 percent), and greater crown volume (5.5 cubic feet, 133 percent) relative to normal management intensity. Increasing management intensity significantly increased crown volume for both ideotypes. On average, the competitor ideotype had a significantly higher crown volume than the crop ideotype (fig. 1). Under intensive management there was no significant difference between crop and competitor ideotypes with respect to branch angle (fig. 2). Under normal management intensity, the competitor ideotype had significantly less acute branch angles than the crop ideotype. Branch diameter significantly increased for both ideotypes when management intensity increased from normal to intensive (fig. 3). As expected, the competitor ideotype had significantly greater branch diameters than the crop ideotype under the same management intensity.

DISCUSSION

The crowns of these trees, especially in the crop ideotype, displayed a variety of phenotypic issues that impacted the overall crown architecture as it related to crown ideotype definitions. There were frequent observations of ramicorn branches in both the crop and competitor ideotypes across management and spacing treatments. This may be directly related to their shared parent which is also well known for ramicorn branching. The competitor ideotype did display a wider crown which spread to occupy space in adjacent tree

crowns both within and between rows at the two higher spacing levels and moving toward crown-to-crown interaction in the lowest stocking level. This is consistent with the definition established by Cannell (1978) for the competitor ideotype. The branch angles and relatively moderate branch diameters of this varietal ideotype also conform to the competitor ideotype. The crop ideotype in this study did have a more compact crown and acute branch angles that allowed the angle of the foliage to potentially intercept light more efficiently, but lacked the smaller branch diameters when compared to the competitor ideotype. This resulted in the purported crop ideotype failing to completely meet the criteria for the crop ideotype (Cannell 1978) but rather falling between the two ideotype categories as what could be considered a “partial” crop ideotype. Furthermore, frequent top dieback observations have been made at this site, especially in the plots with the crop ideotype varietal. Possibly related to the site’s long history of agricultural production, the changes in summer rainfall patterns to more droughty conditions, or a combination of the two, these dieback events may have had a significant impact on the overall crown architecture of the crop ideotype (South and others 2002). Due to the repeated diebacks, the crop ideotype’s crown has assumed a more ovoid crown shape with the branches bending back toward the tree and causing further growth issues. Each time the top dies back the lower branches begin competing for apical dominance resulting in a change in branch angle at the midpoint of the branch. This action has resulted in a more ovoid crown shape with no clearly defined top.

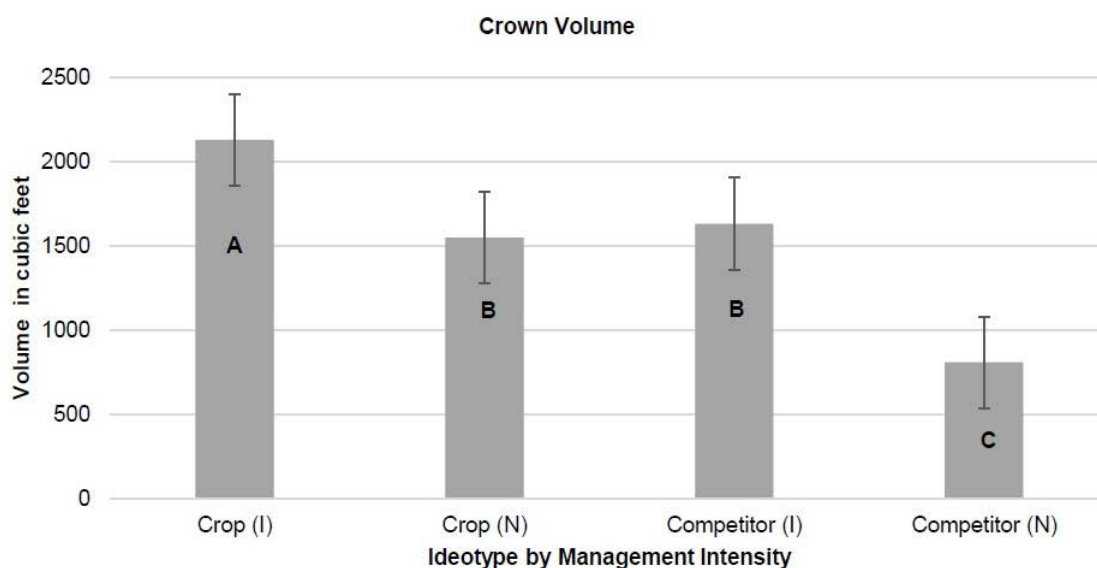


Figure 1—Results of Tukey’s honest significant difference (HSD) test for mean crown volume in cubic feet averaged across stocking levels at age 8 for two half-sib loblolly pine ideotypes in the Upper Coastal Plain of Mississippi at normal (N) and intensive (I) management intensities.

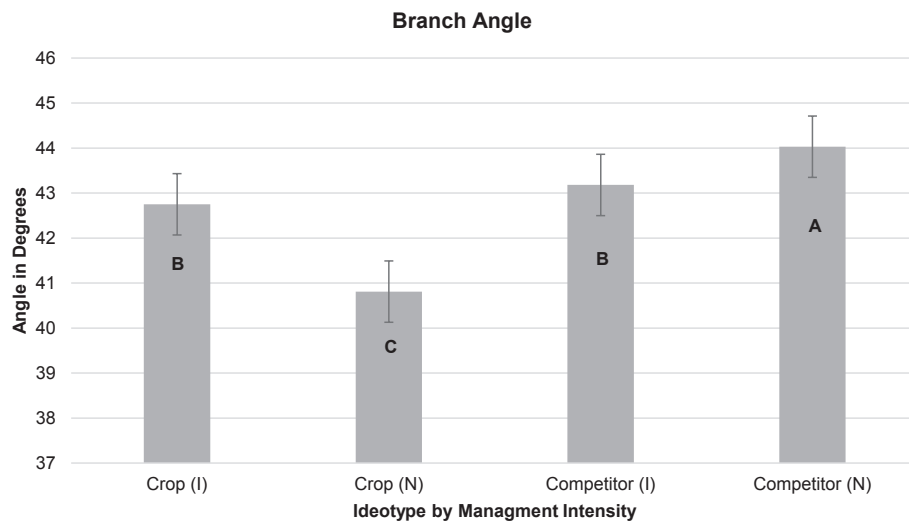


Figure 2—Results of Tukey's HSD test for branch angle at insertion point averaged across stocking levels at age 8 for two half-sib loblolly pine ideotypes in the Upper Coastal Plain of Mississippi at normal (N) and intensive (I) management intensities.

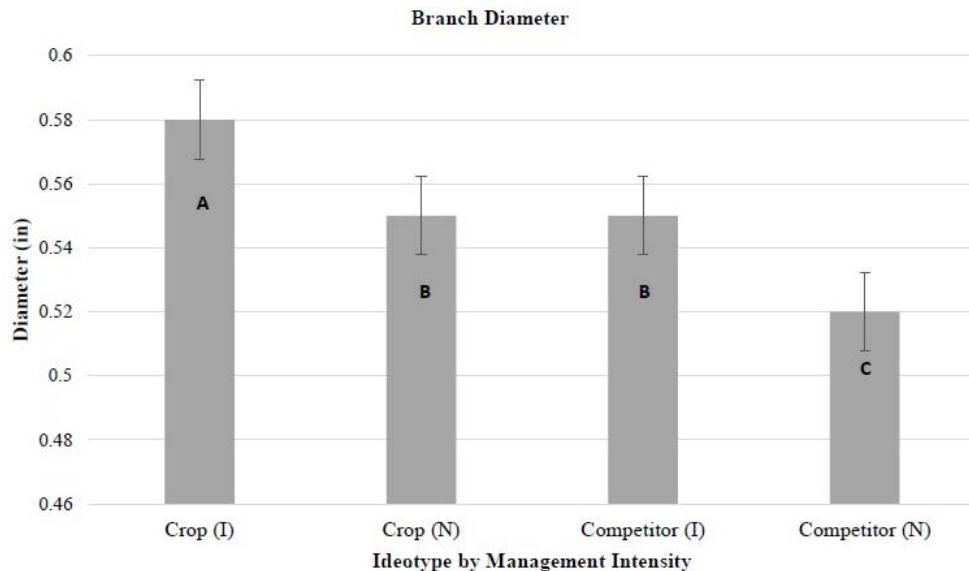


Figure 3—Results of Turkey's HSD test for branch diameter (inches) at insertion point averaged across stocking levels at age 8 for two half-sib loblolly pine ideotypes in the Upper Coastal Plain of Mississippi at normal (N) and intensive (I) management intensities.

CONCLUSION

Intensive management significantly increased crown volume for each ideotype. Competitor ideotype trees had a crown shape that was closer to a parabolic cone while crop ideotype trees had a crown shape that could be better described as ovoid. Branch angle characteristics appear to fit the desired ideotype criteria for the two ideotypes on this site. Branch diameters were larger than anticipated for the crop ideotype and may place this varietal between the two ideotype designations. The multiple top dieback issues and their impact on the crop ideotype and its crown architecture warrants collecting further branch angle data for the crop ideotype as well

as foliage and soil samples across the site to determine possible environmental causes for the top dieback.

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EARLY RESPONSE OF LOBLOLLY PINE TO THINNING IN THE WESTERN GULF REGION

Jason Grogan, Yuhui H. Weng, and Dean W. Coble

Abstract—Early growth responses of loblolly pine (*Pinus taeda* L.) plantations to thinning in the western Gulf region were quantified on nine study sites installed in operational plantations. Three thinning treatment intensities were implemented: light, moderate, and heavy thinning, having 300, 225, and 150 residual trees per acre after thinning, respectively. These were compared to an unthinned control. Individual tree diameter at breast height (DBH) and total tree height were recorded before the thinning and at the end of the first and second growing season after thinning. Thinning treatments improved DBH growth immediately following the thinning, with greater increase on more heavily thinned plots and at the end of the second growing season than the first growing season. Thinning effects on height growth were negligible regardless of thinning intensity and time elapse since thinning. Both thinning responses in DBH and height varied with the initial (pre-thinning) tree size. On the heavily or moderately thinned plots, smaller diameter trees responded more quickly and greater in DBH growth than the intermediate-diameter or larger-diameter trees.

INTRODUCTION

Thinning is one of the most commonly employed silvicultural treatments used to manage loblolly pine (*Pinus taeda* L.) plantations. Responses of loblolly pine to thinning have been well documented (Amateis 2000, Baldwin and others 1989, Ginn and others 1991, Hasenauer and others 1997, Short and Burkhart 1992, Tasissa and Burkhart 1997). These experiments indicate that stem diameter increases consistently with increasing thinning intensity but tree height is influenced very little by stand density.

Our understanding of responses of loblolly pines to thinning is, however, still far from complete. First, most of the above cited studies have focused on long-term response to thinning. In the literature, few studies have investigated immediate response following thinning, and their findings are contradictory. While it is generally accepted that when stands are thinned, time elapses before the effects are evident (Amateis 2000, Ginn and others 1991); others reported an increase in diameter growth increment immediately after thinning (Moschler and others 1989, Tasissa and Burkhart 1997). Second, the above cited early studies have utilized the method of free thinning from below, which does not match the current thinning practice using a combination of

geometric and free thinning techniques. Effects of thinning on tree growth may differ by thinning type (Baldwin and others 1989). Third, most of the above cited studies targeted plantations in the Southeastern United States, and these findings may not be applicable to loblolly pine plantations in the western Gulf region, the western extreme of the southern pine range. Thinning studies on loblolly pine have rarely been carried out in the western Gulf region, although Coble and Grogan (2016) analyzed thinning response in basal area and height growth of residual loblolly pine trees in operationally thinned plantations in east Texas.

Loblolly pine plantations form a significant proportion of forest land in the western Gulf region. Thinning is often applied to regulate growth of residual trees and improve overall stand vigor and stem quality. In order to improve our understanding of thinning response for loblolly pine plantations in this region, the East Texas Pine Plantation Research Program (ETPPRP), a cooperative organization among Stephen F. Austin State University and various industrial forest landowners, initiated a thinning study in 2014 by establishing permanent thinning plots across the region. The objective of this study was to measure and evaluate individual tree early response to thinning in tree height and DBH growth.

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METHODS

In 2014 and 2015, a thinning study was installed in nine loblolly pine plantations. Of these, five were in 2014, and four were in 2015; six were on cutover and site-prepared former stands, and three were on old field sites. These plantations are distributed across east Texas and western Louisiana and were selected following guidelines so that various site qualities were sampled. Initial stand density was uniform across the study at 550 to 605 trees per acre, which is typical for young pine plantations in this region. The stand averages at the time of plot installation were 12.8 years for age (range: 11 to 15), 67 feet at 25 years for site index (range: 60 to 80 feet), 7.24 inches for diameter at breast height (DBH) (range 4.0 to 12.6 inches), and 44.6 feet for total tree height (THT) (range: 12 to 65 feet).

In each plantation, four square, 0.5-acre plots were established. Plots within a plantation had comparable site index, basal area, and number of trees so that the plot-to-plot variation at time of establishment was minimized. Plots were randomly assigned to four thinning treatment categories: no thin (control); and thin to 150 (T150, heavily thinned), thin to 225 (T225, moderately thinned), and thin to 300 (T300, lightly thinned) residual trees per acre residual. Thinning was performed operationally in conjunction with the adjacent stand following current practices, using a combination of geometric and free thinning techniques by removing every fifth row for access and then removing undesirable trees in the remaining rows to meet the thinning target density.

The individual tree diameter at breast height and total height were recorded immediately before thinning, one growing season, and two growing seasons after the thinning. A preliminary analysis showed that there was no difference among plots within a plantation for both traits before thinning.

Data were analyzed by individual tree post-thinning measurement. An analysis of covariance using a mixed model was carried out to test thinning response. The model includes a random factor site (within plantation type), fixed factors of plantation type (cutover vs old field), and thinning intensity and their interactions (some interactions were dropped as their effects were nonsignificant). The initial DBH before thinning was used as a covariate. Where significant ($p < 0.05$) treatment effects were observed, treatment least-square means were calculated. Effect of a thinning treatment over the unthinned control was defined as a thinning response, which was calculated as the difference between the treatment and the control means, and then expressed as the percentage of the control. A positive response suggests an increase in growth from thinning relative to the unthinned control.

RESULTS

Thinning enhanced DBH growth significantly (table 1), more so in the more heavily thinned plots (fig. 1). The Tukey test showed that trees on the T150 and T225 plots had significantly larger DBH than those on the control plots. DBH increment increase was not significant for the T300 plots. Effect of thinning on DBH growth became evident after the first post-thinning growing season and

Table 1—F-values and their significance levels^a of analysis of covariance on individual tree diameter at breast height and total height after the first and second post-thinning (1 & 2 YAT) growing seasons

Trait	Diameter at breast height (DBH)		Total height (THT)	
	1 YAT	2 YAT	1 YAT	2 YAT
Type	7.34*	17.09*	3.24	1.56
TRT	5.17**	14.07**	2.36	2.81
DBH	31119.2**	10780.2**	2190.91**	949.70**
Type*TRT	4.07*	7.33**	1.68	2.82
DBH*TRT	10.81**	19.61**	3.27*	2.14

Type = plantation type; TRT = thinning intensity treatment; DBH = pre-thinning DBH (covariate); Type*TRT = interaction between Type and TRT; DBH*TRT = interaction between DBH and TRT.

^a * = 0.05; ** = 0.01.

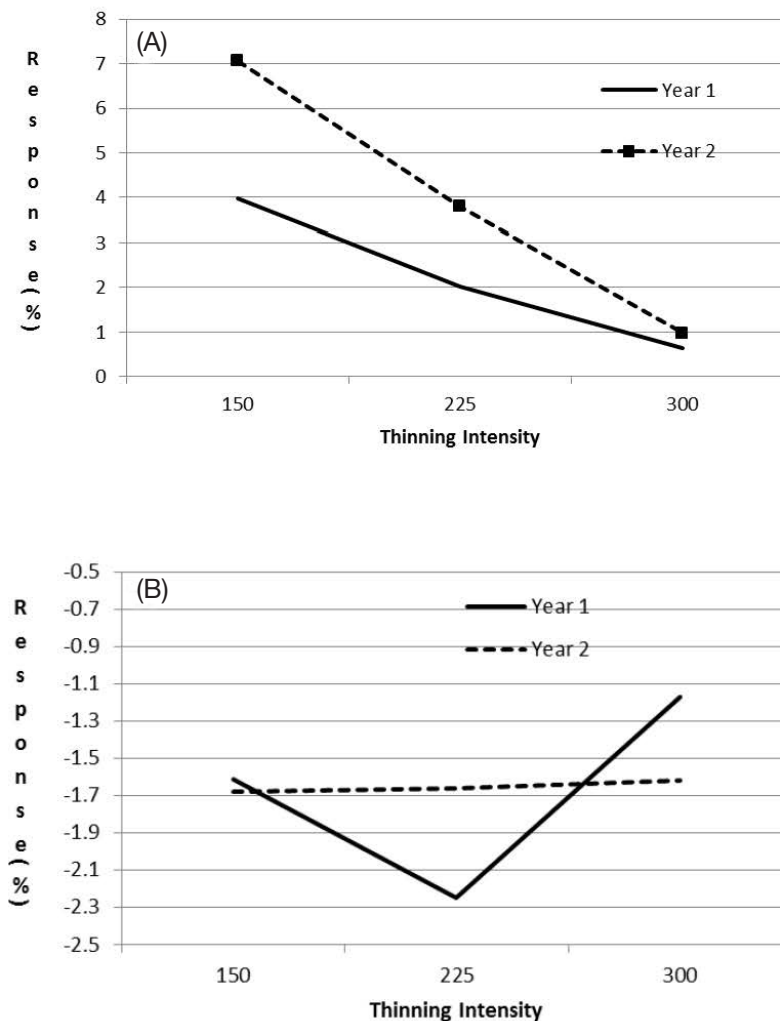


Figure 1—Growth response of individual trees to thinning under various intensities: (A) DBH growth and (B) height growth.

became stronger after 2 years of growth (fig. 1). For example, T150 produced thinning responses of 3.97 percent and 7.07 percent at the end of the first and second post-thinning growing seasons, respectively.

Effects of initial tree size (covariate; pre-thinning DBH) on thinning response were significant and varied with thinning treatment (table 1; fig. 2). The greatest thinning response was found for trees of the 6-inch DBH class for treatments T150 and T225, after 1 year of growth. At the end of the second growing season, the small trees tended to have a greater thinning response than those of the medium-sized or large trees. For T300 plots, trees' thinning responses were independent of DBH class, being around 1.5 percent during the first-year period, but at the end of the second growing season, DBH growth of the small trees substantially declined.

Thinning treatment did not significantly affect THT growth relative to the control (table 1; fig. 1). The covariate, tree DBH before thinning, and its interaction

with thinning treatment, affected thinning response in THT growth significantly (table 1; fig. 3). The thinning responses of the medium-sized trees were similar, being around -2 percent after two growing seasons post-thinning. For the small and the large trees, their thinning responses diversified, depending on thinning intensity. For the small trees, the response improved with increasing thinning intensity, i.e., the thinning response increased from -4 percent for T300 to 3.4 percent for T150 at the end of the first growing season, but the large trees displayed more positive thinning response in the lightly thinned plots than on the heavily thinned plots.

Site type affected DBH growth significantly, and its impact varied with thinning treatment (table 1). Greater responses in DBH growth were observed in the old field sites than in the cutover sites, and the responses became stronger at the end of the second season than the end of the first season. Differences in THT among the treatments were not significant.

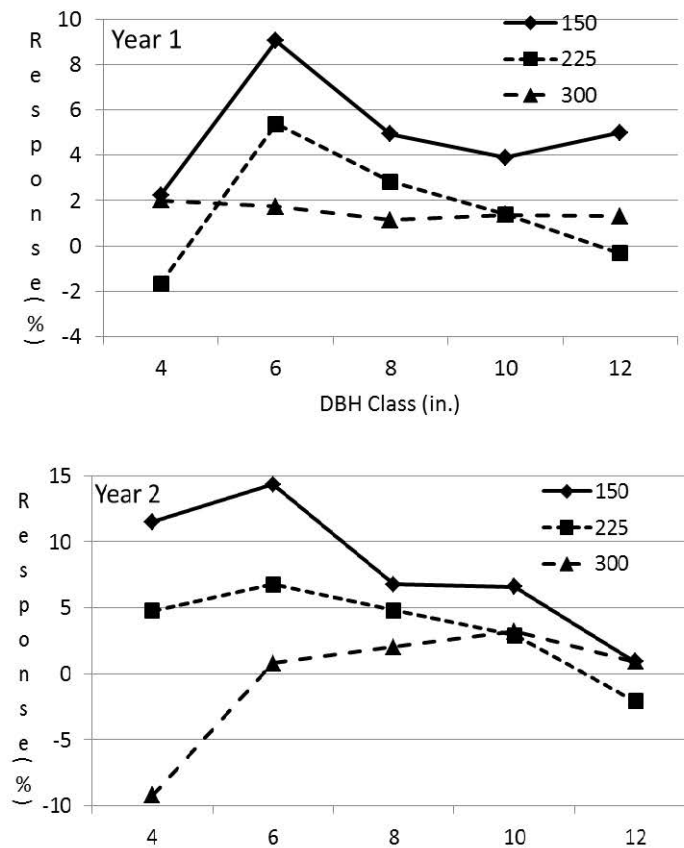


Figure 2—Response in DBH growth of individual trees of various tree sizes to thinning under various intensities.

DISCUSSION

Loblolly pine, a shade-intolerant tree species, requires nearly full sunlight to thrive and grow. In order to improve plantation productivity, thinning is practiced to open the canopy, making more light available for the remaining trees. Thinning also increases nutrient and water availability to residual trees. Other than intensity of thinning and elapsed time since thinning, other factors such as type of thinning, time of thinning, site quality, soil and climatic conditions, and genetic variables may alter response. Since these effects were not considered in this analysis, specific comparisons for combinations of these conditions cannot be made.

Ginn and others (1991) reported that when loblolly pine stands are thinned, time elapses before the effects are evident. However, our results suggest that loblolly pine trees responded to thinning in DBH one growing season following thinning, which is in accordance with previous studies in other regions (Coble and Grogan 2016, Pukkala and others 1998, Tasissa and Burkhart 1997, Valinger 1992). The DBH increment following thinning was greater for the more intensive thinning (Moschler

and others 1989, Tasissa and Burkhart 1997; fig. 1). Thinning effects tend to increase over time until site resources again become limited (crown closure) at which time thinning effects decline (Hynynen 1995, Pienaar and Rheney 1995, Tasissa and Burkhart 1997). This is consistent with our finding that the DBH increment increased from the first growing season to the end of the second growing season (fig. 1). While radial growth is markedly responsive to thinning, height growth is almost unaffected (fig. 1; Ginn and others 1991, Lanner 1985). Various theories have been proposed to explain the contrasting outcome between height growth and radial growth from thinning. After thinning, a tree must first improve its carbohydrate balance through increases in crown diameter and leaf area before it increases its volume growth, which often is at the expense of height growth, resulting in decreases in height growth during the first 2 years after thinning (Haywood 1994). In the long term, the temporary reductions in height growth will decline with time, and eventually trees on thinned plots will become comparable or even taller than those on unthinned counterparts (Brooks and Baily 1992).

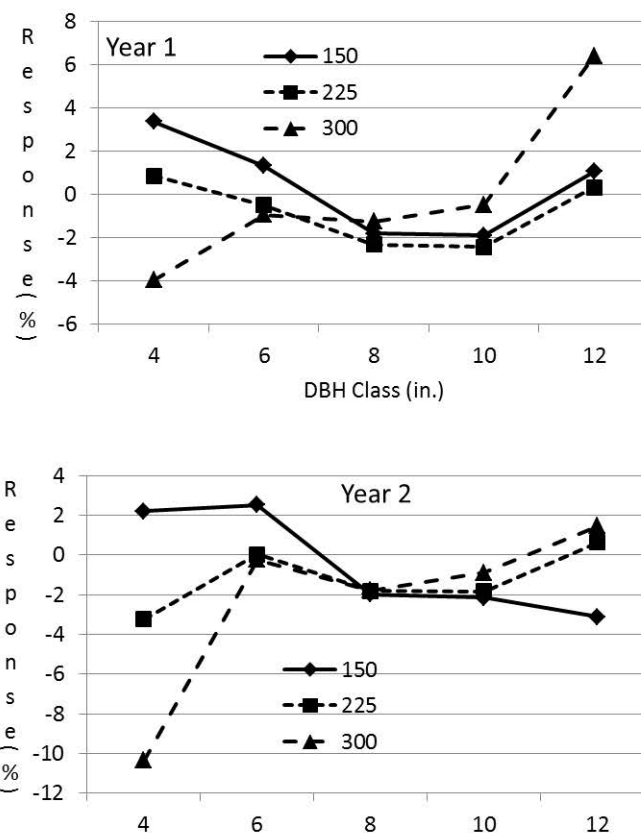


Figure 3—Response in height growth of individual trees of various tree sizes to thinning under various intensities.

How trees of different sizes respond to thinning is of great interest to forest managers and growth and yield modelers (Burkhardt and Tome 2012). Over the 2-year post thinning period, results show that small trees on the heavily thinned plots responded more rapidly and strongly both in DBH and height growth than the medium-sized and large trees (figs. 2 and 3), suggesting the small trees may utilize free growing space more efficiently and create more photosynthate for growth. When thinning intensity was low, such as T300 in this study, the small trees responded negatively to thinning in height growth, while the large trees responded positively. The heavy or moderate thinning might cause substantial thinning stresses to the medium-sized and large trees and, consequently, a negative response to thinning. On the whole, these results may reflect a tradeoff between growing space improvement and thinning shock of a stand after thinning (Harrington & Reukema 1983). How the growth of individual trees of different sizes within a stand respond to thinning has rarely been studied on loblolly pine, but studies on other species showed contradictory results, with some being consistent with

our results (Makinen and Isomaki 2004), while others were not (Eriksson 1987, Hynynen 1995).

In conclusion, our study shows that loblolly pine trees positively respond immediately after thinning and respond strongly to thinning in DBH growth when an intensive thinning was applied. Thinning effects on height growth were negligible, regardless of thinning intensity. Results also show initial tree size, the DBH before thinning, could substantially affect tree growth response to thinning. The study will be monitored and measured in the future, and additional conclusions may be made when more data become available.

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PREDICTING FUTURE VOLUME YIELD AND UNCERTAINTY USING MAXIMUM LIKELIHOOD ESTIMATION

Derrick A. Gallagher, Cristian R. Montes, Bronson P. Bullock, and Michael B. Kane

Abstract—Growth and yield models commonly use the regression residual standard error as a metric of model fit. A common assumption in these models is to have a constant residual error over time. However, it is reasonable to assume errors to increase over time, given accumulated deviations from a yield curve as a consequence of climatic variation from one year to the next. This study tested for equations that include explicit models for errors increasing over time. From those, a linear function using basal area was found to be the best predictor of volume yield error. This study found volume yield prediction uncertainty increased by 0.5839 cubic feet per acre for every 1 square foot per acre increase in basal area.

INTRODUCTION

Accurate estimates of future yield are important for forest managers to quantify the expected monetary value. Several model formulations had been proposed for forest applications. Schumacher and Hall (1933) proposed a logarithmic function for whole-stand yield that has been widely used to predict volume and green weight of multiple pine species. Harrison and Borders (1996) used the logarithm transformation of volume and green weight expressed as a function of whole-stand basal area per acre, dominant height, trees per acre along and interaction variables with age. Different variables and interaction terms were found to be significant by physiographic region in the U.S. Southeast (Harrison et al, 1996). The nonlinear version of the Schumacher (1933) was used by Borders et al (2004) with only one interaction term between basal area and age.

Previous yield models are useful to predict standing volume and green weight, but the uncertainty associated with the estimates is not accounted for. Uncertainty refers to the variance of predictions of future values. Most growth and yield models present a fixed model error value, which is used to derive confidence or prediction intervals to quantify uncertainty. A constant error model implies the uncertainty of those estimates to remain fixed across the temporal and spatial scale. However, it is reasonable to assume errors increase over time, given accumulated deviations from a yield curve as a consequence of climatic variation from one year to the next. This is an observation supported by empirical data

(fig. 2). There is greater variation in volume yield at age 15 as compared to age 5.

When estimating future volume yield, Kangas (1999) described four main sources of prediction error: 1) model misspecification, 2) random estimation errors of model coefficients, 3) residual variation, and 4) errors in the independent variables, and outlined three methods to assess uncertainty of growth and yield predictions. The three methods involve 1) using Monte Carlo simulation to combine residual error from several sources, 2) dividing the error term into fixed and random effects, and 3) modeling the observed past errors. The third method can be used to model the observed variance of volume yield as a function of stand characteristics to account for the temporal and spatial differences. The aim of this study is to outline the procedure to quantify volume yield prediction uncertainty using the maximum likelihood estimation method.

METHODS

Dataset

The data used for this study are from the Western Gulf Culture Density Study (WGCDs) that was established between 2001 and 2003 and managed through 2008 by Texas A&M University and subsequently managed by the Plantation Management Research Cooperative (PMRC) at the University of Georgia. There are 18 installations across the western Gulf region; at each installation, five plots were planted, one each at 200, 450, 700, 950, and 1,200 trees per acre, respectively, and managed with

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a prescribed silviculture regime (fig 1). Measurement plots were surrounded by a minimum 32-foot identically managed buffer. Each site was planted with the best open-pollinated family for that location as determined by the cooperating land manager. Each planting spot was double planted to ensure good survival, with one of the seedlings cut at groundline in the fall following planting as needed to leave one seedling per planting spot. The silviculture regime includes site preparation based on soil type, insecticide application during the first and second growing seasons for tip moth (*Rhyacionia frustrana*) control, herbicide and fertilizer applications during the first growing season, and fertilization at age 9 (table 1). Each installation was assigned to a WGCDS soil group based on drainage class and depth to subsurface restrictive layer (table 2). The distribution of plots across the western Gulf are presented in figure 1. Only the four Lower Coastal Plain (LCP) plots were used in this study. Measurements were recorded annually from age 3 to 6, biennially from age 6 to 12, and every third year thereafter. The age range of the data used in this study is from 3 to 15 years, and only three of the four LCP installations have age 15 measurements.

Diameter at breast height (DBH) was measured on all trees at each measurement. Total height (HT) was measured on all trees >4.5 feet through age 6. After age 6, heights of all trees for the 200-planting density were measured, every other tree for the 450-planting density, and every third tree for the 700-, 950-, and 1,200-planting densities. Heights not measured were predicted using equation 1 with parameters estimated for each plot at every age.

$$\ln(HT) = \beta_0 + \beta_1 * DBH^{-1} + \varepsilon \quad (1)$$

where

HT = height

DBH = diameter at breast height

Dominant trees were classified as non-defective trees with a DBH greater than the plot arithmetic mean at each age interval. Only dominant trees that had both DBH and total height field measurements were included

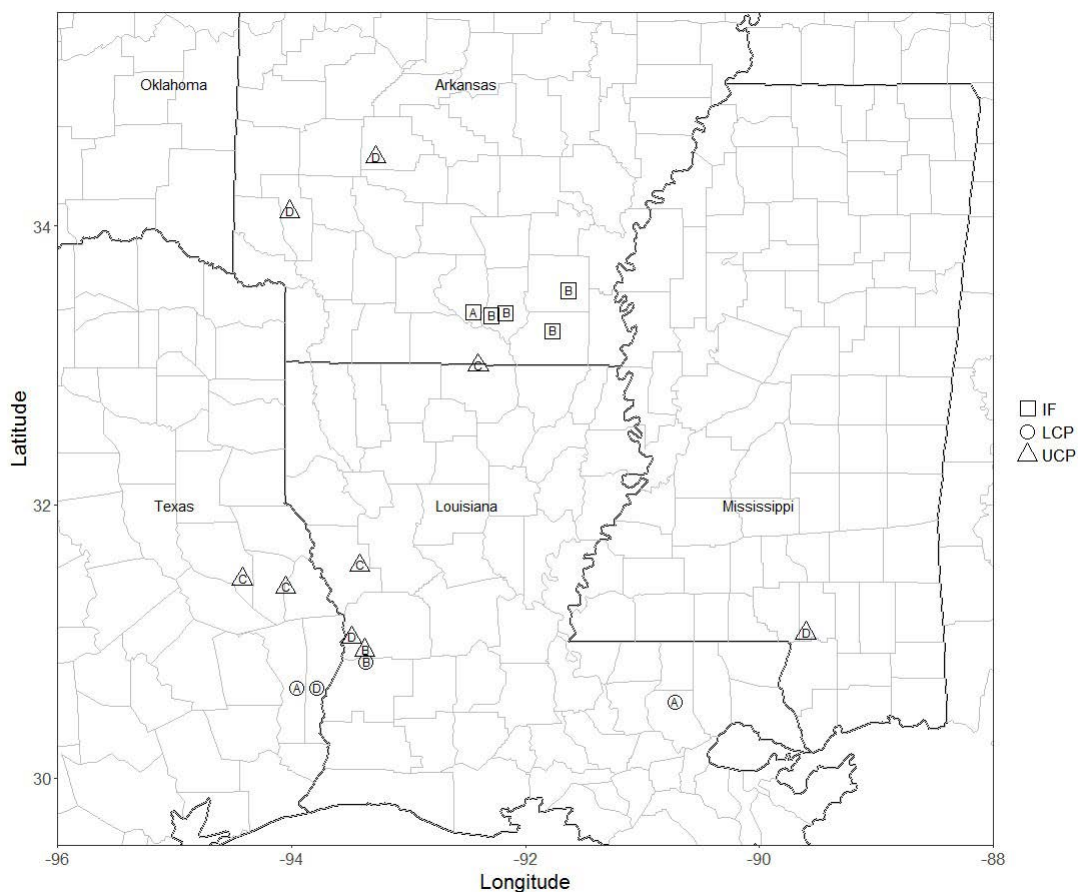


Figure 1—Western Gulf culture density study installations by physiographic region (IF = Interior Flatwoods; LCP = Lower Coastal Plain; UCP = Upper Coastal Plain) and soil group (see table 2).

Table 1—Western Gulf Culture Density Study (WGCDs) silvicultural regime

Activity	Treatment
Site preparation	Soil group A – bedding and ripping Soil group B – bedding only Soil group C – ripping only Soil group D - none
Fertilization	Year 1: 45 pounds N + 50 pounds P per acre Year 10: 200 pounds N + 20 pounds P per acre
Herbicide	Year 1: broadcast
Insecticide	Years 1 and 2: tip moth control

Table 2—Western Gulf Culture Density Study (WGCDs) soil groups

WGCDs soil group	Drainage class	Depth to subsurface restrictive layer inches
A	Poorly—somewhat poorly	<20
B	Poorly—somewhat poorly	>20
C	Moderately well—well	<20
D	Moderately well—well	>20

in the dominant height calculation. Dominant trees with missing heights as a result of subsampling, trees with broken tops, and trees with excessive lean were excluded from the dominant height calculation. This method of dominant height calculation is consistent with standard PMRC procedures.

Individual tree total volume and green weight were calculated using models developed by Lenhart and others (1987) for loblolly pine (*Pinus taeda*) in the western Gulf. Data were summarized by adding individual tree basal area, volume, and green weight per plot. Plot level totals were converted to per-acre estimates based on the known plot sizes. The plot sizes varied based on initial planting density; the plot sizes are 0.2106, 0.1069, 0.1025, 0.1005, and 0.0992 acres for the 200-, 450-, 700-, 950-, and 1,200-trees-per-acre planting densities, respectively. Plot average height, average dominant height, and average diameter were also calculated.

Dominant height ranged from 5 to 17 feet at age 3 and 40 to 67 feet at age 15. Stand average DBH ranged from 0.2 to 2.8 inches at age 3 and 4.0 to 12.5 inches at age 15. Stand DBH exhibited a relationship with initial planting density; stands with lower initial planting densities had a larger average DBH from ages 6 to 15. The mean trend of volume yield is similar for planting densities ranging from 1,200 to 450 trees per acre

(fig. 2). There is significantly less volume per acre in stands planted at 200 trees per acre between ages 8 to 15 (fig. 2). The mean trend was estimated using local regression (LOESS).

Model for Whole Stand Volume Yield

Volume yield per acre is assumed to be normally distributed and expressed as equation 2, which is the nonlinear form of the Schumacher yield function used by Borders and others (2004).

$$\text{Volume} \sim N(V, \sigma^2)$$

$$V = \alpha_0 * HD^{\alpha_1 + \alpha_2/A} * TPA^{\alpha_3 + \alpha_4/A} * BA^{\alpha_5 + \alpha_6/A} + \varepsilon \quad (2)$$

where

V = volume yield

HD = dominant height in feet

TPA = trees per acre

BA = basal area per acre

A = stand age

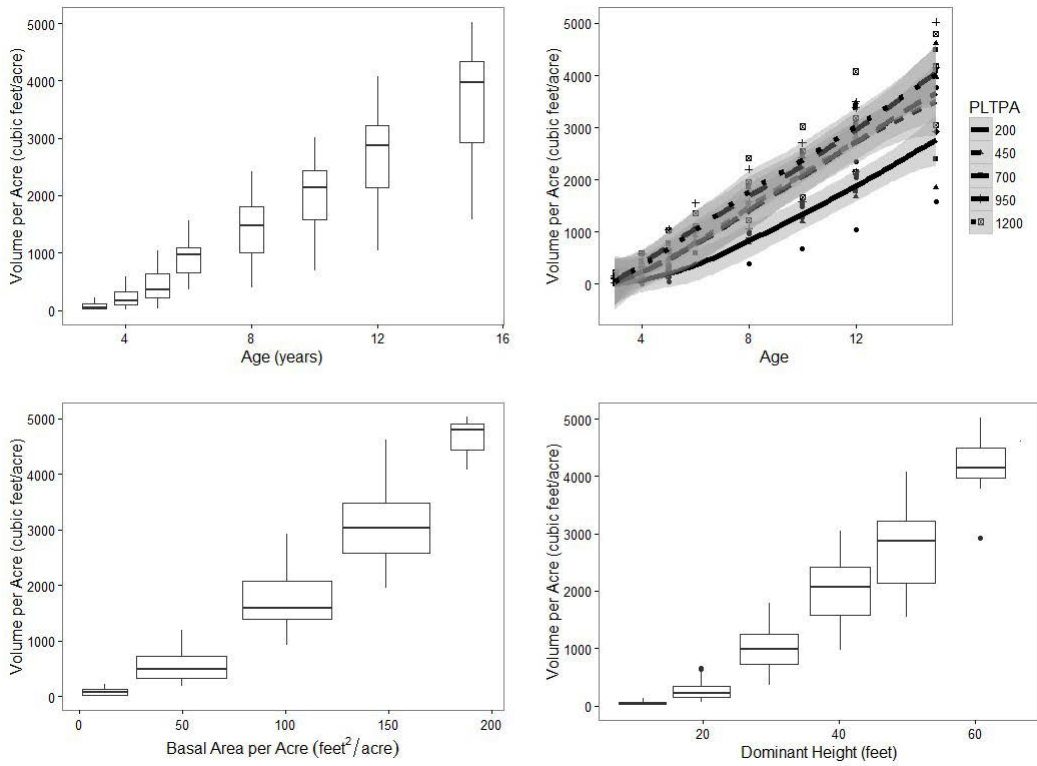


Figure 2—Volume (cubic feet) per acre yield by (A) age, (B) volume by age with LOESS prediction lines and prediction intervals by initial planting density (PLTPA), (C) basal area, and (D) dominant height for all installations.

Models for Future Uncertainty

The 95-percent prediction intervals for the mean volume yield by planting densities exhibit the same width, indicating similar variances between densities (fig. 2). The estimated uncertainty for volume yield predictions should be the same across all densities. Age, basal area, and dominant height exhibit relationships with volume yield variation (fig. 2). Three function types to model uncertainty were compared using the log-likelihood value and the ranking criterion: constant (equation 3), linear (equation 4), and power (equation 5). Parameters estimates must always be positive because negative parameter estimates infer the variance will decrease as the predictor variable increases, which is contradictory to the observed data. Therefore, the parameters are transformed to the exponential scale to ensure the untransformed parameter estimates are always positive. Equation 4 includes a linear combination of BA, HD, and A with no intercept. This mathematical model ensures biological consistency in which the variance of volume yield is zero when either one of the independent variables is zero. Equation 5 is a multiplicative power function including BA, HD, and A.

$$\sigma = (S_1) + \varepsilon \quad (3)$$

$$\sigma = \exp(S_1) * (BA) + \exp(S_2) * (HD) + \exp(S_3) * A + \varepsilon \quad (4)$$

$$\sigma = (BA)^{\exp(S_1)} * (HD)^{\exp(S_2)} * A^{\exp(S_3)} + \varepsilon \quad (5)$$

where

S_i = Parameter estimate for the i^{th} covariate explaining volume yield standard deviation

Maximum Likelihood Procedure

The parameters are estimated simultaneously for the volume yield and prediction variance functions. The negative of the normal log-likelihood function was minimized using the “optim” procedure in R (R Core Team, 2016). The best variance function was determined

using the log-likelihood value as the ranking criterion. Parameter significance was evaluated using t-tests in which the parameter standard errors were derived from the hessian matrix. Parameters found not significant were removed, and the remaining parameters were re-estimated. Model prediction performance and associated estimates of uncertainty were evaluated using graphics. The assumptions of model residuals being normally distributed with mean zero and constant variance were evaluated using a boxplot of the standardized residuals by predicted values.

RESULTS

All parameters in equation 2 were significant except α_6 . The interaction parameter for basal area per acre and age is nearly zero with a standard error larger than the parameter estimate. The parameter was removed, and the final yield function is presented as equation 6.

Basal area per acre was found to be the only significant predictor of the volume yield variation for both equations 4 and 5. The parameter estimates for dominant height and age were approximately zero with large variation. Basal area per acre is the best predictor of volume

yield variation regardless of the equation form. The final recommended models to predict volume yield and the uncertainty of those estimates are equations 6 and 7. The exponential transformed estimate of the S_1 parameter is 0.5839. For each unit increase in basal area per acre, volume yield standard error increases by 0.5839.

$$\hat{V} = \alpha_0 * HD^{\alpha_1 + \alpha_2/A} * TPA^{\alpha_3 + \alpha_4/A} * BA^{\alpha_5} \quad (6)$$

$$\hat{\sigma} = \exp(S_1) * (BA) \quad (7)$$

The estimated residual standard error using equation 3 was 67 cu ft per acre. The residuals standardized using equation 3 exhibit an increasing trend and a constant trend using equation 7 (fig 3). Even though the prediction uncertainty increases as basal area increases, the standardized residuals using equation 7 are homoscedastic.

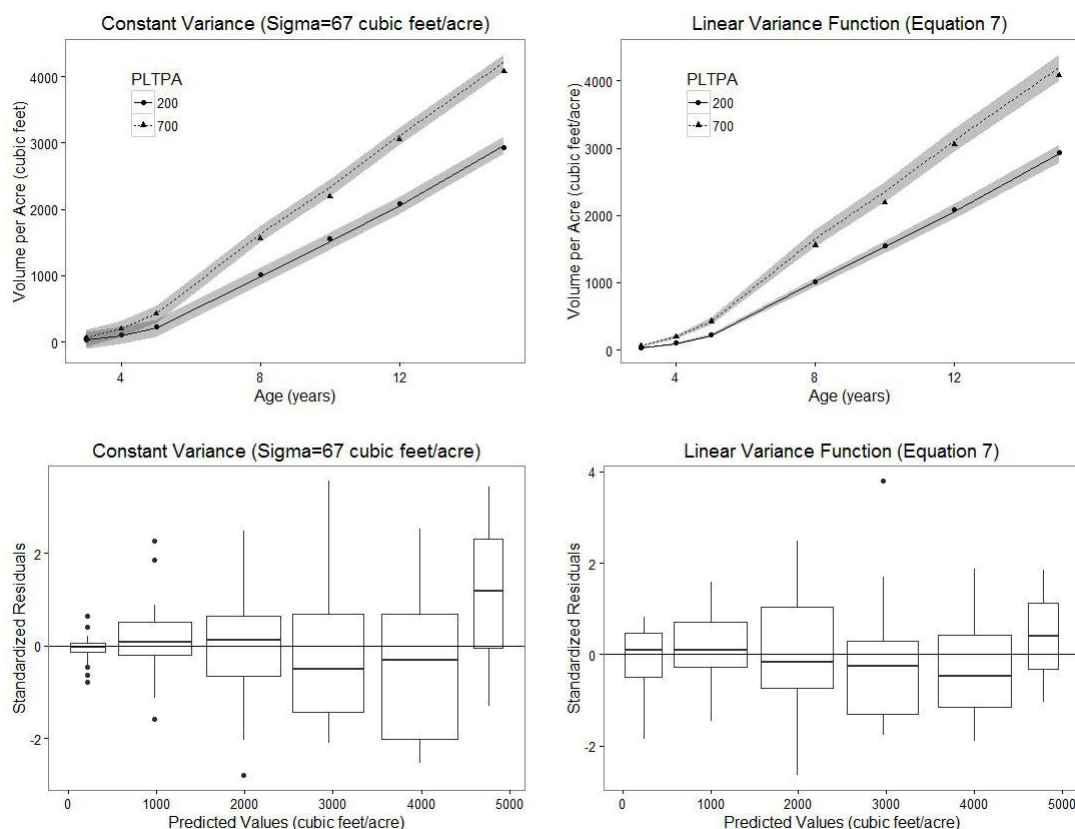


Figure 3—A, B: Equation 6 diagnostics for installation 1 including observed volume (points), equation 6 predicted volume (line), and prediction uncertainty for initial planting densities (PLTPA) of 200 and 700 trees per acre. C, D: Equation 6 residuals standardized using equations 3 and 7.

CONCLUSION

Standard methods of building confidence and prediction intervals using a constant variance to quantify the uncertainty of future estimates is not adequate. A fixed sigma infers the variance is constant across the range of predicted values. This study shows a fixed model error value is not adequate to quantify prediction uncertainty and instead increases as stand basal area increases. Confidence intervals using a constant variance typically include zero at young ages and do not capture the true increase in variability in older stand ages with more accumulation volume.

Using the maximum likelihood method to estimate the parameters of the volume yield prediction and variance functions simultaneously results in more accurate volume predictions and estimates of uncertainty for stand ages 3 to 15. As the stand matures and accumulates more basal area, the stand volume yield variability inherently increases as well. Using an equation to model the uncertainty of volume predictions better accounts for the changes in variability as the stand changes. This study revealed for forest volume yield estimation that basal area is the only predictor required to accurately quantify prediction uncertainty. Further research methods on improving the quantification of uncertainty using maximum likelihood should be investigated.

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Shortleaf Pine Silviculture

Moderator:

Wayne Clatterbuck
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RESTORATION OF SHORLEAF PINE IN THE SOUTHERN UNITED STATES—STRATEGIES AND TACTICS

James M. Guldin and Michael W. Black

Abstract—Shortleaf pine (*Pinus echinata* Mill.) is the most widely distributed and poorly understood of the four major species of southern yellow pine. The area of southern forests dominated by shortleaf pine forest types has declined by more than 50 percent since 1980, with the most dramatic declines found in states east of the Mississippi River. To counteract this decline, the Shortleaf Pine Initiative was launched in the spring of 2013 by a host of partners including the U.S. Department of Agriculture Forest Service, other Federal and State agencies, universities, major conservation organizations, and other private partners in the region. The release of the Shortleaf Pine Restoration Plan in the summer of 2016 outlines a series of optimum restoration strategies, opportunities for coordination among proponents interested in shortleaf pine, and ways for partners to work together. However, geographic conditions and forest types are highly variable across the 23 States where shortleaf pine is found, and as a result, different approaches to restoration will be required in different regions. The management strategies and silvicultural tactics that managers should consider in application to the restoration and management of shortleaf pine in pure and mixed stands across the native range of the species are discussed.

INTRODUCTION

The dominant frequent-fire-adapted southern yellow pine ecosystems in the Southern United States are iconic places that have declined in area and are at risk of further decline. Assessments under two of the Southern Forest Futures forecasts of forest type change from 1950 to 2060 (Wear and Greis 2013) show the scale and scope of the decline. One forecast was built using assumptions of high urbanization, high timber prices, and accelerated rates of pine plantation management. Under those assumptions, the natural pine forest type declines by 59 million acres, the oak-pine forest type declines by 14.5 million acres, and the planted pine type increases by 66 million acres (table 1A). A second forecast was built using assumptions of low urbanization, low timber prices, and decelerating rates of pine plantation establishment. Under those assumptions, the natural pine forest type declines by 48.5 million acres, the oak-pine forest type declines by 9 million acres, and the planted pine type increases by 47 million acres (table 1B). By way of perspective, the draft 2017 Forest Resources report for the 2020 Resources Planning Act reports slightly more than 208 million acres of timberland in the Southern United States (Oswalt and others, In press). In round numbers, by the year 2060, 25 percent of the South's forests will be planted pine stands, and less than 10 percent will be in naturally regenerated pine-dominated stands.

The most substantial declines in acreage of native fire-adapted southern yellow pine ecosystems occurred prior to the 1950 date used in the Forest Futures analysis. The archetypal example of this decline is found in the decline of longleaf pine (*Pinus palustris* Mill.) ecosystems. Estimates are that, prior to European colonization of North America, stands dominated by longleaf pine or mixed pine-oak stands occupied roughly 91 million acres (Frost 1993). Current estimates from Forest Inventory and Analysis show that longleaf-dominant stands occupy 4.3 million acres (Oswalt and others 2012), a decline of 95 percent in its historic area. Since 1970, there has been no net loss of longleaf pine, though there was a slight decline into the 1990s and a recovery since that time. In essence, most of the loss of longleaf pine occurred prior to 1970.

By way of comparison, shortleaf pine (*P. echinata* Mill.) is more widely distributed than longleaf pine, but not as widely dominant historically. Estimates are that, prior to European colonization, stands dominated by shortleaf pine or pine-oak stands covered roughly 70–80 million acres. Today, shortleaf pine or pine-oak stands are dominant on only 6.1 million acres, a decline also of more than 90 percent (Anderson and others 2016). However, unlike the decline in longleaf pine, the decline in shortleaf pine has been more recent; currently, the area of stands dominated by shortleaf pine

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Table 1—Forecast of forest type based on analysis conducted by the Forest Futures Project (Wear and Greis 2013)

(A) Cornerstone Future E.

Assumptions: high rate of urbanization, high timber prices, high rate of planting

Year	Natural pine	Oak-pine	Planted pine
	--million acres--	--million acres--	--million acres--
1950	72.5	28.0	1.0
1980	50.0	30.0	15.0
2010	31.5	22.0	40.0
2060	13.5	13.5	67.0
Net, 1950–2060	-59.0	-14.5	66.0

(B) Cornerstone Future F.

Assumptions: low rate of urbanization, low timber prices, low rate of planting

Year	Natural pine	Oak-pine	Planted pine
	--million acres--	--million acres--	--million acres--
1950	72.5	28.0	1.0
1980	50.0	30.0	15.0
2010	31.5	22.0	40.0
2060	24.0	19.0	48.0
Net, 1950–2060	-48.5	-9.0	47.0

or pine-oak stands have declined by 52 percent since 1980 (Anderson and others 2016). From 1980–2012, a decrease in shortleaf pine and pine-oak types exceeding 500,000 acres has been reported for Alabama, Mississippi, Texas, Arkansas, and Georgia; three other States (Louisiana, Tennessee, and Oklahoma) have had declines between 400,000 and 500,000 acres over that same time period (Anderson and others 2016).

The recent decline for shortleaf pine could be an artifact of the changes in forest management on private lands in the region, especially where the distribution of loblolly pine (*P. taeda* L.) and shortleaf pine are sympatric and management has displaced shortleaf pine with loblolly pine. For example, in the upper west Gulf Coastal Plain of southern Arkansas in the 1980s, woodlands managers with Georgia-Pacific Corporation managed mixed stands of loblolly pine and shortleaf pine of natural origin using the seed tree method, precommercial and commercial thinning, and prescribed fire to rotations of 45 years. That silvicultural system produced an expected mean annual increment of roughly 4 tons per acre annually (Zeide and Sharer 2001). Conversely, Fox and others (2007) describe advances in loblolly pine plantation silviculture that yield roughly 10 tons per acre or more from planted loblolly pine stands on similar sites over a 25-year rotation. It's easy to see why productive forest lands across the South are converted from naturally regenerated stands to planted stands—

especially on lands managed primarily for returns on investment, in areas where the natural range of loblolly and shortleaf pine is sympatric, and by landowners that can afford to make the substantial initial investment in plantation establishment.

Not only are shortleaf pine-dominated stands less prominent on the landscape, but the character of stands that remain has changed because of changing fire regimes. In 1935, U.S. Department of Agriculture (USDA) Forest Service Chief V.A. “Gus” Silcox codified a new agency policy that all forest fires were to be extinguished by 10 a.m. on the day after they were detected (Long 2016). This, in conjunction with the rise in capacity for State forestry agencies after World War II, set the standard for control of wildfires on public and private forest lands nationwide. The message was driven home by the Smokey Bear Wildfire Prevention Program, established in 1944; Smokey's classic message, “Remember... Only YOU Can Prevent Forest Fires,” was established in 1947 (cf. smokeybear.com). But in application to fire-adapted southern yellow pine ecosystems, the message was confusing and impeded resource managers and the public. In addition, the transition from open range to fence laws for domestic livestock during the 20th century played a role in the reduction of surface fires set by farmers and ranchers to improve forage. The result has been a general withdrawal of effective controlled burning to maintain

habitat in mature native stands of southern yellow pines, including shortleaf pine, and replacement of the pine component to a prominent midstory and overstory component of less fire-tolerant hardwoods through succession. Smokey's message was revised in 2001 to say, "Only You Can Prevent Wildfires," in part to reflect the value of controlled burning in southern pine forest types (cf. smokeybear.com).

These two factors, the decline in area of shortleaf pine and pine-hardwood forest types and the exclusion of fire across the landscape, combine to form the crux of the issue—that mature fire-maintained shortleaf pine stands are dramatically underrepresented on the landscape today relative to 200 years ago. Moreover, while 200 years constitute multiple generations in human timeframe, it's within the span of a single generation in shortleaf pine. And when that habitat is underrepresented on the landscape especially within a short ecological timeframe, the species of flora and fauna that are specifically adapted to that habitat become underrepresented as well. For example, west of the Mississippi River and east of the Great Plains, American bison (*Bison bison*) and elk (*Cervus canadensis*) have been largely extirpated in woodlands; species such as northern bobwhite (*Colinus virginianus*), Bachman's sparrow (*Peucaea aestivalis*), and the Diana fritillary butterfly (*Speyeria diana*) have limited distribution; and the red-cockaded woodpecker (*Picoides borealis*) is officially endangered.

THE SHORTLEAF PINE INITIATIVE

In 2010, agencies and conservation leaders across the region convened a Shortleaf Pine Working Group. Two regional meetings were hosted, one in Raleigh, NC, in 2010 and the other in Huntsville, AL, in 2011, which built impetus for a more formal region-wide program on restoration of shortleaf pine. That led to the establishment of the Shortleaf Pine Initiative in 2013; an Advisory Committee was formed consisting of representatives from a dozen public and private organizations including Federal agencies (USDA Forest Service, USDA Natural Resource Conservation Service, and U.S. Department of the Interior Fish and Wildlife Service), State agencies, nongovernmental conservation organizations, universities, forest management organizations, and conservation-based foundations. The Initiative was funded by the Forest Service Region 8, State and Private Forestry, through the University of Tennessee-Knoxville. After a number of regionally based organizational and implementation meetings, the goals and objectives of the Initiative were codified in the Shortleaf Pine Restoration Plan in 2016 (Anderson and others 2016).

The vision and mission statements of the Initiative summarize the scale and complexity of work to be done. The vision is to expand the area of forests and

woodlands dominated by shortleaf pine for the array of economic, ecological, and cultural benefits they provide, through a collaborative partnership effort across the historical range of the species. The mission is to provide the leadership and collaborative partnership framework for the restoration of shortleaf woodlands on a rangewide scale (Anderson and others 2016). Key elements of the restoration plan include 1) a series of optimum restoration strategies that are region-specific, 2) increased needs for coordination among proponents interested in shortleaf pine, and 3) ways for partners to work together.

Strategic Issues

The question of regional effect is important. The dominance of shortleaf pine varies widely across its natural range (fig. 1). The degree to which the species is found in mixture with other pines and hardwoods varies as well. As a result, the silvicultural prescriptions that are needed to manage for the species will be different, especially in the scale of activity across the landscape. A key to understanding the complexity of the challenge in shortleaf pine restoration is the question of managing mixed-species stands. There is a delicate balance in the establishment and development of seedling pines (or sprouts in the case of shortleaf pine) and vigorous sprouting hardwoods. Additional research is needed to develop the appropriate silvicultural prescriptions to ensure that the species desired in mixed stands will survive to the point of ingrowth into merchantable size classes.

Management of shortleaf pine stands in the Ouachita Mountains and Ozark Highlands will be relatively straightforward. In these Interior Highlands, shortleaf pine is the only native pine, and silvicultural systems devoted to managing pure shortleaf pine and pine-hardwood stands have been applied for 7 decades. Clearcutting has been commonly used to manage Ouachita shortleaf pine through the latter part of the 20th century; it's an effective silvicultural system provided that proper site preparation is conducted, especially ripping which markedly improves seedling survival in the very stony soils of the Ouachitas (Brissette and Barnett 2004, Walker 1992). Research from the latter part of the 20th century suggests that containerized shortleaf pine provides marginally better survival and growth than bare-root stock (Barnett and Brissette 2007, Brissette and Barnett 2003), and this appears to be supported by contemporary reports (Bell 2012, Schnake and others 2016). However, shortleaf pine lags far behind loblolly pine, slash pine (*P. elliottii* Engelm.), and even longleaf pine in area planted annually, and some degree of capacity improvement will be needed to broadly expand seed production and nursery propagation for a regional shortleaf pine planting program.

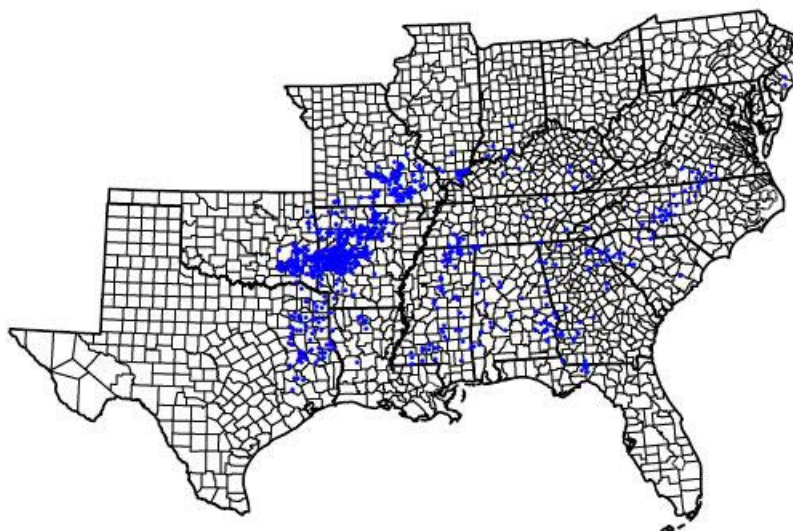


Figure 1—Forest Inventory and Analysis plot locations where shortleaf pine represents more than 50 percent of basal area, representing 3.453 million acres in this condition. Map projection: Lambert Continental. (Map created by J.F. Rosson, Jr., USDA Forest Service, Forest Inventory and Analysis, September 2015)

Natural regeneration under even-aged rotations using the shelterwood method is known to be successful in shortleaf pine (Lawson 1990), especially when prescribed fire is used early in the life of the new age cohorts to stimulate shortleaf pine resprouting and re-establish native understory flora (Guldin 2007). Modifications of the seed tree and shelterwood methods that lead to two-aged stands are ideal for recovery of the endangered red-cockaded woodpecker (Conner and Rudolph 1991, Hedrick and others 1998, Rudolph and Conner 1996). A classic example of this is found in the Ouachita Mountains of western Arkansas and eastern Oklahoma (fig. 2). Active local timber markets allow the commercial sale of trees harvested during thinning in fully stocked sawtimber-sized pine stands. Receipts from that timber sale as well as other funding sources then support removal of the invading hardwood midstory, introduction of periodic prescribed burning at large scale, and even the insertion of artificial nest boxes in residual overstory pines for the endangered woodpecker. In essence, timber sales are the first step in the ecological restoration of extensive areas of shortleaf pine-dominated forests and woodlands, with attendant benefits for both the endangered woodpecker and the panoply of flora and fauna that rely upon open fire-maintained pine woodlands (Guldin 2007).

Restoration of shortleaf pine elsewhere throughout its range is complicated by a host of factors, the foremost of which is that shortleaf pine is not the only native conifer. Most of the silvicultural systems used to manage shortleaf pine in the western part of its range will also favor pitch pine (*P. rigida* Mill.), Virginia pine (*P. virginiana*

Mill.), Table Mountain pine (*P. pungens* Lamb.), longleaf pine, and especially loblolly pine. Planting, whether as part of the clearcutting method, after other kinds of harvesting including high-grading, or to afforest abandoned agricultural lands, is an important part of the toolbox. However, differences in site preparation and release tailored to local conditions will obviously be required among these different kinds of conditions.

Fire is extraordinarily important in maintaining shortleaf pine ecosystems. Shortleaf pine is the only southern pine that will reliably sprout at young ages and small root-collar diameters when top-killed by fire (Lilly and others 2012a, 2012b), an adaptive trait noted more than a century ago (Mattoon 1915). Recent research suggests an increased incidence of hybridization between shortleaf and loblolly pine is, in part, a result of reduced temporal discontinuity between periods of loblolly pollination and shortleaf pine cone receptivity, and perhaps also the widespread planting of loblolly pine across the South (Tauer and others 2012). The importance of surface fires in young stands as an agent to trigger shortleaf pine resprouting while concurrently top-killing hybrids and also volunteer loblolly pine in new mixed-species age cohorts has been shown to be ecologically significant and silviculturally useful (Bradley and others 2016, Stewart and others 2015). However, as mixtures grow more diverse and more complex, another tool in the toolbox for those interested in managing for shortleaf pine is to select for retention of shortleaf preferentially and to remove other pine and hardwood species during the first entries associated with precommercial or commercial thinning.



Figure 2—Management of shortleaf pine on Federal lands in the Ouachita Mountains centers on overstory thinning, hardwood midstory removal, and cyclic prescribed burning; this leads to open fire-maintained shortleaf pine habitat suitable for the endangered red-cockaded woodpecker (cavity tree shown with metal flashing at the base) as well as many other species of flora and fauna that are adapted to these ecological conditions. (photo by James M. Guldin)

Tactics Appropriate for Restoration of Shortleaf Pine

In light of these issues and concerns, a subjective decision model can be developed that gives landowners some guidance about the feasibility and the likelihood of success for managing shortleaf pine. Because these elements depend on a host of factors with which foresters are more likely to be familiar than landowners, the first element is easily stated—landowners who decide to manage shortleaf pine should retain the services of a professional forester who can advise them about the opportunities and challenges that will arise.

The stand must be large enough to allow for cost-effective forest management activities. Two concerns seem paramount in this regard. First, a stand should be large enough to support operable harvests during thinning. Contracted services such as planting, site preparation, and release should be feasible, and the stand should be amenable to the safe execution of prescribed burning. Landowners should take advantage of natural regeneration with shortleaf pine especially if stands currently have mature shortleaf pine in the overstory. It's difficult to conduct many of these treatments if the stand is smaller than 25 acres in size, unless they are part of a larger landscape where treatments can be concurrently conducted.

Prescribed burning is critical for the successful management of new age cohorts of shortleaf pine to reduce the increasing frequency of shortleaf x loblolly pine hybrids (Bradley and others 2016, Stewart and others 2015, Will and others 2013), and perhaps also as a tool to convert offsite loblolly pine stands back to shortleaf pine. Controlled burning will be easier to implement if the site is characterized by homogeneous within-stand topography or lies within a larger landscape that can all be burned. Prescribed fire is more difficult to apply if the stand is isolated or is in terrain that has a high degree of topographic heterogeneity. Some Federal land management programs have a cooperative element that allows Federal fire crews to also burn private lands that are adjacent to Federal ownership as part of a larger landscape fire management plan. However, on private lands generally, the local availability of State or contract burning crews is certainly a constraint.

A key consideration is whether the sites where restoration is planned are within the woodshed of local mills, and whether there are local markets for pine timber. Knowing there is a market for trees harvested during thinning and ultimately for mature shortleaf will be critical to enable a cost-effective restoration program. One of the challenges in shortleaf pine restoration

activity in the northern part of its range, especially east of the Mississippi River and to the north of the natural range of loblolly pine, is the paucity of mills that utilize pine sawtimber and pulpwood.

On the other hand, if the stand is located within the sympatric range of shortleaf and loblolly pine, pine sawmills and pulp mills will be more common because of industrial management of loblolly pine, and shortleaf pine can be merchandised in those markets. But choosing to manage for shortleaf pine within the sympatric range of loblolly pine, especially the establishment of new planted stands, may be difficult for some landowners to justify because of the faster growth rates of loblolly pine, especially over the first 3 decades of stand establishment. Landowners will need other reasons to establish shortleaf pine in preference to loblolly pine—such as to provide landscape-scale species diversity, as part of a wildlife program involving prescribed burning at young ages, or as a hedge against changing climatic conditions especially in the western part of its range.

Obviously, stands with an existing shortleaf pine component are easier to manage than stands where shortleaf pine is rare or absent. But the potential to work with stands where shortleaf pine is a manageable minor component should not be overlooked by landowners and the foresters who advise them. In a study of the rehabilitation of understocked loblolly-shortleaf pine stands on the upper west Gulf Coastal Plain, Baker and Shelton (1998) reported that stands with 30 percent stocking recovered to full stocking in 15 years. Mixed stands where shortleaf is a minor component in the range of 20–50 square feet per acre are candidates for recovery by removing the non-shortleaf component through thinning, followed by judicious midstory treatment and initiation of cyclic prescribed burning. Among the resulting benefits of this approach is the development of new shortleaf pine seedlings and sprouts in the understory, which can develop into a new age cohort as opportunities allow. This could well be the best opportunity and highest priority rangewide to quickly restore shortleaf pine dominance in a stand.

Shortleaf pine can grow in a wide variety of soil types, but it may have a competitive advantage versus other conifers and hardwoods on dry, xeric soil types rather than wet and mesic soil types, especially in the western part of its range. Similarly, past land use is a consideration especially with respect to understory flora. If the site has a history of agricultural use, supplemental restoration will be needed to restore understory flora such as C4 grasses as well as the overstory shortleaf pine component. However, sites on eroded terrain or with a history of littleleaf disease in the Piedmont continue to be a challenge for restoration of shortleaf pine.

DISCUSSION

Shortleaf pine is an iconic southern yellow pine found in pure and mixed stands, but it is gradually being lost in stands and landscapes across the South. Planting will be required for its restoration, but not much shortleaf pine planting is currently done. Substantial effort will be needed by Federal and State agencies as well as the private sector to develop capacity for genetically improved shortleaf pine seed and to expand nursery production and planting of shortleaf pine seedlings. Restoration will be relatively straightforward in States west of the Mississippi, especially the Ouachitas of Arkansas and Oklahoma, where shortleaf is the only native pine, and where local markets for pine pulpwood and sawtimber are well developed. Efforts elsewhere are confounded by lack of markets, the complicated silviculture of mixed-species stands, and the professional preference for loblolly pine because of faster growth rates. Prescribed burning is critical for restoration of pure or mixed shortleaf pine stands, especially at young ages so as to help maintain pure shortleaf pine and to eliminate shortleaf x loblolly pine hybrids. Low-cost restoration should expand to include managing existing stands that have a shortleaf component, even a minor component that could be managed to accentuate its dominance in a given stand. Research is needed to better quantify the advantages of maintaining a dominant shortleaf pine component in mixed species stands, especially within the sympatric range shared by loblolly pine; advantages might include favorable elements of species diversity, distribution of risk in a management portfolio, ability to use cyclic prescribed burning to promote wildlife, and a possible hedge against mortality due to extended drought. Finally, although the strategies and tactics for restoration of shortleaf pine will differ from those used in longleaf pine, those two initiatives are united by a common goal—the restoration of fire-adapted southern pine ecosystems and the fauna and flora that depend upon them, which are underrepresented across the landscape of southern forests.

ACKNOWLEDGMENTS

Thanks to the Advisory Committee of the Shortleaf Pine Initiative for their guidance in development and implementation of regional efforts to improve the management of shortleaf pine. Thanks also to Dr. Rod Will at Oklahoma State University and Dr. Lance Vickers at the University of Missouri-Columbia for a host of illuminating conversations, often in the woods, about the ecology and management of shortleaf pine generally and for their excellent comments and feedback on this manuscript.

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COMPARISON OF 49-YEAR-OLD PLANTATION-GROWN LOBLOLLY AND SHORLEAF PINE IN THE ARKANSAS OZARKS

Matthew G. Olson, William L. Headlee, and H. Christoph Stuhlinger

Abstract—Plantations of pure loblolly pine and pure shortleaf pine grown for 49 years on a site in the Ozark Highlands of Arkansas were compared to test for effects of species on individual tree- and stand-level attributes. Analysis of data collected in 1981 revealed that loblolly pine was significantly larger in diameter and height than shortleaf pine 14 years after establishment ($p < 0.05$). At that time, the planted loblolly pine also supported significantly higher per-acre basal area and stem density. By age 49 (2016), these loblolly pine plantations had dominant stems of significantly larger total height, but no differences in diameter, live crown ratio, basal area, or density were detected. These findings suggest that loblolly pine growth and yield can outperform shortleaf pine at this location north of loblolly pine's native range, at least through a half-century. However, shortleaf pine appears to have narrowed the performance gap noted earlier in stand development.

INTRODUCTION

A common silvicultural objective is to limit plant community composition to species that are best suited to the site while meeting landowner goals (Smith and others 1997). Since reliance on natural regeneration generally favors locally adapted native species, there may be less concern that efforts to naturally regenerate a stand will promote maladapted species. However, in the case of planted stands, successfully meeting forest management goals will depend largely on selecting the most appropriate species for the planting site.

Research comparing performance of species growing under similar conditions helps to inform species selection. Published studies comparing southern pine species suggest differences in performance that vary by region, site, and the species being compared. Loblolly pine (*Pinus taeda*) is the most commercially important and researched pine species native to the Southern United States. Compared to shortleaf pine (*Pinus echinata*), a species that is sympatric with loblolly over much of its range, loblolly pine generally exhibits superior growth at both the individual tree and stand level (Arnold 1978, Branan and Porterfield 1971, Dipesh and other 2015, Lynch and others 2016, Schultz 1997). However, many of these studies only consider development of planted trees during the first few decades (in other words, a period consistent with intensive pine management) (Rink and Wells 1988). Less is known about differences in productivity between

planted loblolly and shortleaf beyond the typical rotation of loblolly pine. However, evidence from a recent study of older, second-growth stands in southern Arkansas indicates that loblolly's growth advantage over shortleaf may not persist beyond the first few decades of stand development (Childs 2016). There is also evidence that shortleaf may outperform loblolly at the stand level when the species are planted well outside of loblolly's natural range (Rink and Wells 1988).

There is growing interest in managing for shortleaf pine and associated natural communities. For instance, the Shortleaf Pine Initiative recently formed in response to concerns over an observed decline of shortleaf pine populations noted across the species' range (Anderson and others 2016). There is also interest in promoting shortleaf pine for addressing forest health concerns, including red oak decline (Law and others 2004), and for enhanced resilience to future climate change (Brandt and others 2014). Where opportunities exist to establish planted shortleaf stands, studies comparing the growth of shortleaf against alternative species will help inform landowners about management-associated tradeoffs.

In the late 1960s, an experiment was installed in the Arkansas Ozarks to compare the performance of plantation-grown loblolly pine and shortleaf pine on a study site north of loblolly's current natural range west of the Mississippi River, but within shortleaf's range. This investigation presented a unique opportunity

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to compare the long-term performance of these two species outside of loblolly's range. The purpose of our study was to compare the growth and development of loblolly and shortleaf pine at two points in time using this same experiment.

METHODS

Study Site

The study site is located at the University of Arkansas Division of Agriculture's Livestock and Forestry Research Station (LFRS) near Batesville, AR. The LFRS is located on the dissected Springfield Plateau physiographic region of the southeastern Ozark Highlands. The experiment was established on a predominantly north-aspect slope underlain by two variants of the Clarksville soils series that differ based on percent slope. Both soils are classified as very cherty silt loams with a shortleaf pine site index at base age 50 years of 55 feet. In the spring of 1967, eight 6-acre plantings of pure loblolly pine and shortleaf pine (four replicates of each) were established in a randomized complete block design. Blocks were designed to capture potential variation associated with slope position. Three blocks were established on the steeper portion of the slope (blocks 1–3), and the fourth was established on the flat above the slope (block 4). Monocultures were planted at a spacing of 6 feet by 8 feet (908 trees per acre). Although it is likely that site preparation and release treatments were applied, we could find no record of any followup treatments to the site.

A commercial harvest of the site occurred in 2006. Most of the area was treated with a thinning that resembled a combination of low and crown thinning and was concentrated in blocks 1–3. Large portions of block 4 were treated with a seed tree establishment cut, while the rest was intentionally left unharvested (the unharvested area included both shortleaf and loblolly plantings).

Analytical Approaches

Analysis was performed on two datasets collected at different times. Data collected in 1981 (stand age 14 years) using a 10-basal area factor (BAF) point cruise were uncovered, providing an opportunity to assess performance at a relatively early age. Diameter at breast height (d.b.h.) and total height (TH) were estimated for only the pines captured from a single point within each experimental unit ($n = 8$). This experiment was revisited in May of 2016 (stand age 49) to inventory what remained of the original study. A 10-BAF point cruise was used to collect data in 2016. Ten points were taken within each of the thinned plantations and six in each unharvested portion of block 4 (hereafter, unthinned). Species and d.b.h. were recorded for each captured

tree. Crown base height and TH were estimated for a dominant or codominant pine (hereafter, dominant) at each point.

Analysis of variance (ANOVA) was used to test for effects of species on tree-level and stand-level attributes at age 14. Specifically, d.b.h. and TH were included as tree-level responses, and basal area per acre (BA) and trees per acre (TPA) were included as stand-level attributes. Analysis of age 49 data was performed separately on thinned and unthinned stands using a combination of ANOVA and t-tests. Tree-level responses (pine only) considered at age 49 were d.b.h., TH, and live crown ratio (LCR). Since the thinning did not appear to be applied uniformly across the study, we were concerned that the thinning may have obscured species effects on stand-level attributes. Therefore, stand-level responses were not included in our assessment of thinned stands. Stand attributes (BA and TPA; pine only) were included in analysis of unthinned stands at age 49. ANOVA was used to test for an effect of species on tree-level attributes in thinned stands. Since the unthinned portion of this study lacked replication, ANOVA was not used to test for species effects. Instead, we considered cruise points within each unthinned stand as independent samples and compared unthinned loblolly and shortleaf stands using t-tests. Post-ANOVA mean separation was performed using Tukey's honestly significant different test. Statistical significance was assessed at $\alpha = 0.05$. Analysis was conducted using SAS 9.4.

RESULTS

Comparison at Age 14

Tree-level ANOVA models detected a significant effect of species on both d.b.h. ($p = 0.002$) and TH ($p = 0.003$). Loblolly pine stems were both larger in diameter and taller than stems of shortleaf pine (34.6 versus 22.8 feet and 6.4 versus 5.4 inches, respectively; fig. 1). A significant species effect was also detected in stand-level ANOVA models of BA ($p = 0.002$) and TPA ($p = 0.011$). Plantations of loblolly pine had higher BA (132.5 square feet per acre) and TPA (645 trees per acre) than shortleaf stands (73.7 square feet per acre and 497 trees per acre, respectively).

Comparison at Age 49

There were no differences found in TPA ($p = 0.605$) or BA ($p = 0.432$) between unthinned loblolly pine and shortleaf pine plantations according to t-tests. Mean TPA and BA were 161 trees per acre and 123.7 square feet per acre in the unthinned loblolly stand and 133 trees per acre and 105.0 square feet per acre in the unthinned shortleaf stand (fig. 2). In unthinned stands, dominant loblolly pine had higher TH than dominant shortleaf pine ($p = 0.003$). Mean TH of dominant loblolly pine and shortleaf pine in

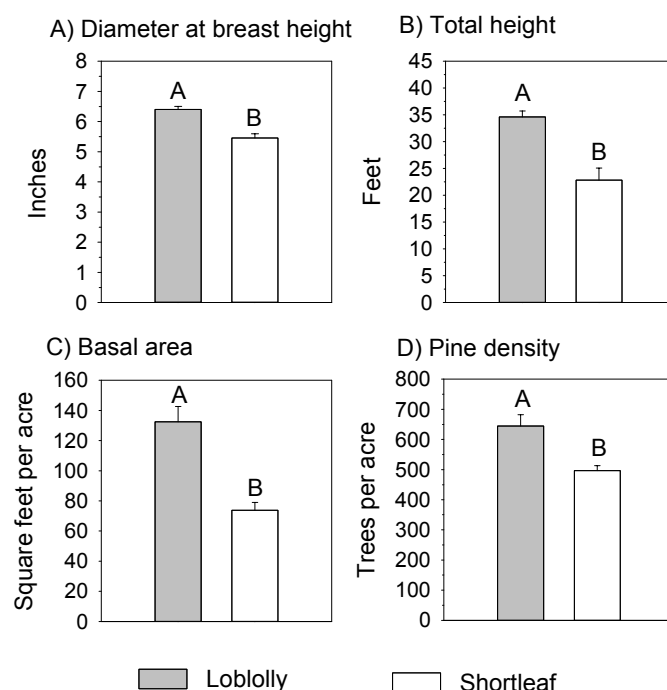


Figure 1—Comparison of loblolly pine and shortleaf pine plantations at stand age 14 based on diameter at breast height (A), total height (B), pine basal area per acre (C), and density of pine trees per acre (D). Means within a comparison with different letters are significantly different.

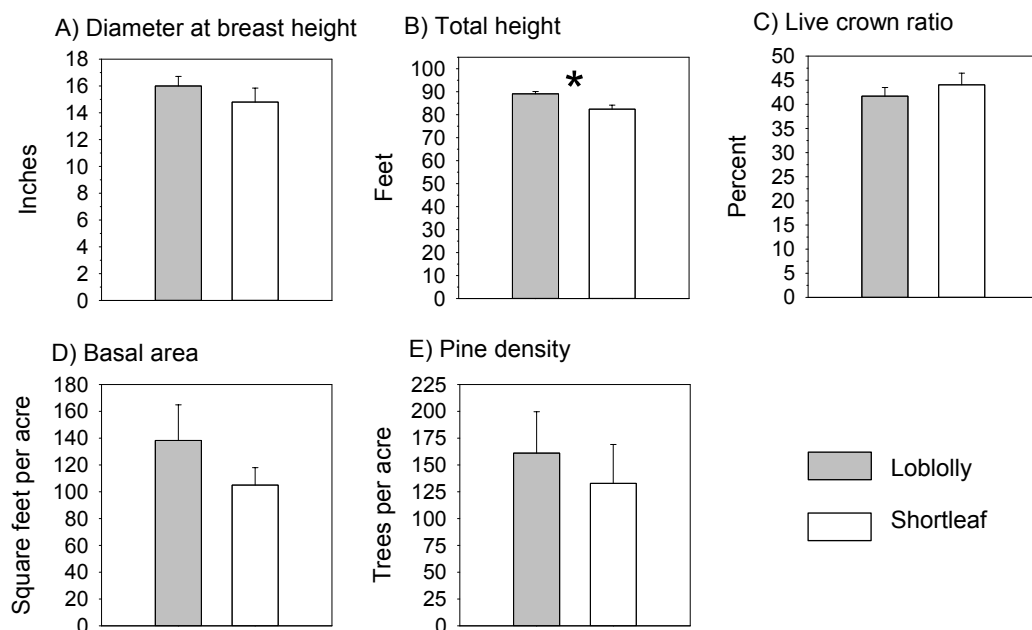


Figure 2—Comparison of unthinned loblolly pine and shortleaf pine plantations at stand age 49 based on diameter at breast height (A), total height (B), live crown ratio (C), pine basal area per acre (D), and density of pine trees per acre (E). An asterisk indicates a significant difference between means.

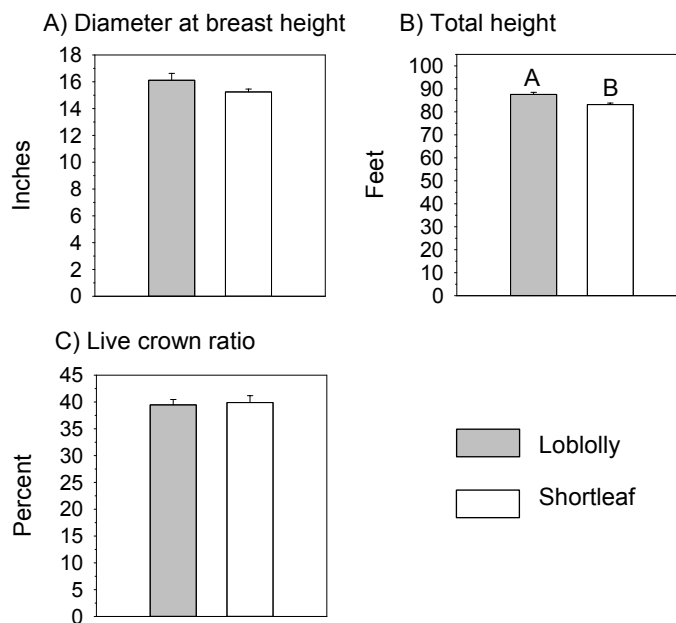


Figure 3—Comparison of thinned loblolly pine and shortleaf pine plantations at stand age 49 based on diameter at breast height (A), total height (B), and live crown ratio (C). Means within a comparison with different letters are significantly different.

the unthinned stands averaged 89.2 feet and 82.5 feet, respectively. There were no differences in d.b.h. (all pines; $p = 0.193$) or LCR (dominant pines; $p = 0.445$) between loblolly (16.0 inches and 41.7 percent) and shortleaf (14.8 inches and 44.1 percent).

In thinned plantations, mean d.b.h. of all pines was 16.1 inches and 15.2 inches for loblolly and shortleaf pine, respectively (fig. 3). Mean LCR of dominant trees was 39.5 percent for loblolly pine and 39.9 percent for shortleaf pine. No effect of species on d.b.h. ($p = 0.194$) or LCR ($p = 0.799$) was detected in ANOVA models. There was a significant effect of species in ANOVA models of TH ($p = 0.014$), with loblolly pine significantly taller than shortleaf pine (87.6 and 83.2 feet respectively).

DISCUSSION

Our results support the general conclusion of past research that loblolly pine usually outperforms shortleaf pine at both the individual tree and stand level (Schultz 1997). In this study, loblolly's growth superiority was clearly evident at 14 years. For example, loblolly on average was 11 feet taller than shortleaf, and the BA of loblolly stands exceeded the BA of shortleaf stands by nearly 60 square feet per acre. These differences in individual-tree growth performance are consistent with our understanding of the silvics of both pine species. Loblolly pine is well known for its rapid early growth and development, which, in combination with its high adaptability to site conditions and genetic variability, largely explains why loblolly is a preferred species for intensive silviculture (Baker and Langdon 1990).

Shortleaf pine, on the other hand, is a slower-growing, drought-tolerant species that is generally associated with xeric, fire-prone growing sites and is rarely used for intensive silviculture (Lawson 1990).

Most of the growth and yield research comparing loblolly pine with other southern pines has been restricted to a timeframe consistent with a high-yield rotation for loblolly pine (20–30 years). Growth studies that go beyond the short-rotation timeframe of past research are needed to provide a temporal scope appropriate for southern pine management on an extended rotation. A recent study comparing growth of mature shortleaf and loblolly in older, second-growth stands on the Coastal Plain of southeast Arkansas found similar periodic growth increment at the tree level during the previous 10-year period (Childs 2016), suggesting that loblolly's growth advantage may not persist beyond the typical loblolly pine rotation. Results observed at age 49 of this study suggest that loblolly was still the dominant performer, yet also indicate that shortleaf pine may have narrowed the growth gap formed earlier in this study. At the individual-tree level, loblolly was significantly larger in d.b.h. at age 14, but not at age 49 in either thinned or unthinned stands. Although loblolly was significantly taller at both stand ages, height differences observed at stand age 49 were less than what was seen at stand age 14. At the stand level, the lack of a species effect on stand-level attributes in 2016 suggests that stands of both species were comparable in year 49. It is worth mentioning that the lack of a significant species effect on 2016 stand-level attributes could be due to lower

statistical power associated with the limited sample size from the unthinned stands. For example, mean BA of unthinned loblolly stands exceeded that of shortleaf stands by 33 square feet per acre, yet no difference was detected. More research comparing growth of loblolly and shortleaf from older plantations is needed to gain a better understanding of the performance of these species when grown on an extended rotation.

Another general conclusion that can be drawn from this study is that loblolly outperformed shortleaf on a site within the natural range of shortleaf, but outside of loblolly's range. A study comparing 37-year-old plantings of shortleaf and loblolly in southern Illinois, a site outside of both species' natural ranges, found that individual loblolly stems were larger than shortleaf, but, due to poor loblolly survival, shortleaf stands supported greater volume (Rink and Wells 1988). High mortality of loblolly pine in this case was likely related to maladaptation of loblolly pine to a site well north and inland of its current natural range (for example, loblolly's susceptibility to ice storm damage). However, the site used for this study is not too far outside of loblolly's range and is still well south of the northern extent of loblolly; therefore, the climate at this site likely is not limiting growth of loblolly pine. Based on site index ($SI_{50} = 55$ feet for shortleaf pine), the study area is considered a low-quality site, indicating that loblolly also outperformed shortleaf on a poor growing site.

The results of this study could be interpreted as evidence in favor of loblolly pine as a southern species to consider for planting in the Arkansas Ozarks for adapting forests to future climate. Although the exploitative growth strategy of loblolly pine could enhance resilience to future environmental uncertainty, the more conservative growth of shortleaf pine, a trait associated with drought tolerance, could be better suited to frequent, intense growing season droughts projected by some climate models (Brandt and others 2014). Furthermore, loblolly's greater susceptibility to ice storm damage should also be considered before planting this species north of its current range (Baker and Langdon 1990). More research is needed to assess impacts of drought and extreme weather on both loblolly pine and shortleaf pine.

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EFFECTS OF ICE DAMAGE ON GROWTH AND SURVIVAL OF SHORLEAF PINE TREES

Pradip Saud, Doug S. Cram, Thomas B. Lynch, and James M. Guldin

Extended abstract—Natural disturbance events, such as ice storms, may last from hours to a few days, but their impacts may create long-term disturbance in forest structure and development (Bragg 2016, Turcotte and others 2012). Following ice storms, the development and survival of individual trees may be affected by numerous factors such as extent of damage, residual density, stand age, vigor, site condition, and species (Bragg and others 2003, Dipesh and others 2015, Saud and others 2016a). In the Southwestern United States, pine species experience considerable crown damage from ice storms since they retain foliage throughout the year and provide large surface area for ice accumulation. Damage is especially likely in younger stands that are densely stocked and have low crown ratio (Guldin 2011).

Shortleaf pine (*Pinus echinata* Mill.), an important commercial tree species in the Southern region, has received little research attention regarding its growth and survival following ice storm damage unlike studies on growth-and-yield modeling (Saud and others 2016b). In December 2000, ice storm events affected millions of forested hectares in Oklahoma, Arkansas, and Texas (Bragg and others 2003, Stevenson and others 2016). Individual trees of even-aged, naturally occurring shortleaf pine forests were damaged. We quantified the damage in permanent growth-and-yield monitoring research plots of shortleaf pine established between 1985–1987 in the Ouachita and Ozark National Forests and subsequently monitored them the following years. A repeated measurement provided an opportunity to investigate variables influencing growth and survival probabilities of individual trees following ice storm damage.

The original dataset consisted of 207 research plots. In 1998–1999, 181 of these plots were thinned. The ice storm damaged trees in 101 plots, 85 of which were thinned plots. Three post-ice storm measurements were recorded in 2000–2001, 2006–2007, and 2012–2014. We used a multiple treatment design analysis to evaluate annual growth response and survival probability of individual trees over a 12-year period. Data analysis consisted of multiple factors including 1) stand basal area [low (7), average (14), moderate (21), and high ($>21 \text{ m}^2 \text{ ha}^{-1}$)]; 2) ice storm effect on plot (ice damage, no ice damage); 3) crown damage [no damage, low (≤ 25 percent), high (>25 percent)]; 4) aspect (east, west, north, and south); and 5) thinning intensity [no thinning, low ($0\text{--}5 \text{ m}^2 \text{ ha}^{-1}$), high ($>5 \text{ m}^2 \text{ ha}^{-1}$)]. A generalized linear mixed model was used to predict annual basal area growth (ABAG) of individual trees selecting the best covariates out of 12 different tree- and stand-level attributes, five factors, and elevation. We used Cox proportional hazard (PH) model for survival analysis assuming right-censored data and tree mortality as the event of interest.

Over a 12-year period, results showed that high crown-damaged trees in high basal area stands had significantly reduced ABAG rates as compared to trees from low crown damage and low basal area. The model suggested that ABAG rates of individual trees were significantly affected by relative spacing, stand age, and stand basal area based on crown damage level. The model also indicated that ice-damaged trees growing on mesic sites (east and north aspects) had better ABAG rates than trees growing on xeric sites (west and south aspects).

Reduced ABAG following an ice storm could be the result of compressed radial growth patterns as observed in dendrochronological study of ice-damaged shortleaf pine by Stevenson and others (2016), a result of reduced photosynthetic capacity due to crown loss. Given the impact of ice damage on height gain on individual trees, crown-damaged trees showed significantly less ABAG than undamaged trees as reported in previous studies (Dipesh and others 2015, Stevenson and others 2016, Turcotte and others 2012). Low crown-damaged trees showed improved ABAG compared to high crown-damaged trees in the subsequent measurements, as reported

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in other studies that the 4- to 5-year interval following the event is enough for the recovery of low crown-damaged trees (Dipesh and others 2015, Stevenson and others 2016). This suggests high crown-damaged trees have slower recovery than low crown-damaged trees. However, crown growth would also vary depending upon stand density, relative spacing, and stand age (Saud and others 2016b).

The greatest individual tree mortality rates over the 12-year period were for ice-damaged plots as compared to no ice-damaged plots. Lower survival rates for crown-damaged trees were correlated with increasing stand basal area, decreasing elevation, and decreasing thinning levels. Over the decade, high crown-damaged trees had substantially reduced survival probability (60 percent) as compared to low crown-damaged trees (80 percent). Ice-damaged trees experienced 2.2 times greater mortality than undamaged trees, and at times, due to random plot effects, some trees were at a fourfold greater relative risk than average-damaged trees. Thus, we conclude that over time low crown-damaged trees will have better survival rates similar to undamaged trees.

Trees occupying dominant-crown positions were more heavily damaged than codominant and suppressed trees. Stand density also might have influenced ice damage risk by collisions from falling neighbor trees as discussed by Bragg and others (2003). Thinned stands experienced higher mortality rates than unthinned stands but the effect was confounded with tree size as observed in plantation forest (Bragg 2016). Further, a study on time-dependent mortality suggested that individual tree attributes such as larger basal area, height, and crown ratio increased survival of ice-damaged trees while crown damage level and stand-level competition decreased survival probability (Saud and others 2016a).

Understanding the response of ice-damaged shortleaf pine trees is important for devising management strategies of damaged stands. We believe these results will be helpful to increase our understanding of effects tree- and stand-level attributes have on resilience and response of individual trees following an ice storm in context to biological and economical optimal rotations.

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Longleaf Pine Silviculture

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RESTORATION OF LONGLEAF PINE IN THE SOUTHERN UNITED STATES: A STATUS REPORT

R. Kevin McIntyre, James M. Guldin, Troy Ettel, Clay Ware, and Kyle Jones

Abstract—In 2009, the America's Longleaf Restoration Initiative set an aggressive goal of increasing the area of ecosystems dominated by longleaf pine (*Pinus palustris*) from 4.29 to 8 million acres by 2025. In 2015, a 5-year review of progress using Forest Inventory and Analysis data showed that gains in longleaf pine acreage were offset by losses and that total longleaf pine acreage remained unchanged since 2010. As a result, Federal, State, and private partners engaged in a review during the summer of 2016 to discuss how to modify or respond to this lack of progress; they agreed to retain the original 8-million-acre goal, and to develop a revised set of strategies to attain that goal. These include efforts to increase restoration and to better understand the causes for the decline of longleaf pine on both public and private lands. Most of these will require changes in agency policy, enhanced restoration through planting and prescribed burning, and developing additional financial and managerial resources for implementation. Key to these efforts will be diversification of longleaf pine silviculture, including novel approaches to managing stands that contain a minor but manageable component of longleaf pine.

INTRODUCTION

Over the last 2 decades, interest in restoration and management of longleaf pine (*Pinus palustris*) ecosystems has substantially increased. Longleaf pine ecosystems have many attributes that are compelling to those managing both public and private lands. They provide opportunities for economic utilization through harvest of timber (especially utility poles and high-quality dimension lumber) and nontimber forest products. They offer outstanding opportunities for wildlife including game species such as white-tailed deer (*Odocoileus virginianus*), eastern wild turkey (*Meleagris gallopavo*), and northern bobwhite quail (*Colinus virginianus*), as well as non-game species of concern such as gopher tortoise (*Gopherus polyphemus*), brown-headed nuthatch (*Sitta pusilla*), and Bachman's sparrow (*Peucaea aestivalis*). Appreciation for longleaf pine also includes extraordinary but less tangible values, including aesthetics and the cultural significance of the iconic role that longleaf pine played in the history of the southeastern landscape. Once the dominant forest type on over about 92 million acres from southeastern Virginia to eastern Texas (Frost 2006), the longleaf pine forest type had been reduced to less than 4 percent of its original extent by the mid-1990s (Outcalt and Sheffield 1996).

Always important to a small cadre of biologists and land managers in the Southeast, the longleaf pine ecosystem began to garner broader attention once the red-cockaded woodpecker (*Leuconotopicus borealis*) and other longleaf-associated species were listed as endangered or threatened under the Endangered Species Act. Through the 1980s and 1990s, interest in saving this remarkable ecosystem continued to grow. There was both a greater emphasis on longleaf pine restoration and management on public lands, as well as increased private sector efforts that were aided by U.S. Department of Agriculture (USDA) incentive programs such as the Conservation Reserve Program and others.

In 2007, a regional working group of 22 public agencies and private organizations formed to develop the America's Longleaf Restoration Initiative (ALRI). A core concept in the development of ALRI was that the task of achieving restoration of longleaf pine ecosystems at a meaningful scale was beyond the capacity of any one agency or organization, and would require a coordinated effort across the historic range of longleaf pine. A stakeholder engagement process facilitated input from a diverse group of conservationists and managers across the Southeast, and a rangewide conservation plan for longleaf pine was released in 2009 (ALRI 2009).

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The broad goal of the conservation plan was to increase the acreage of longleaf pine from 4.29 to 8 million acres by 2025, and more detailed goals related to the manner by which that might occur were also outlined. Although approximately 4.29 million acres existed when the plan was released, only about 1.5 million acres were considered to be in “maintenance” condition, defined by ALRI as containing the desired fire-maintained vegetation structure to provide habitat for longleaf-associated wildlife species. The plan articulated a goal of moving another 1.5 million acres into this category, for a total of 3 million acres in maintenance condition. Other specific goals and objectives outlined in the plan relate to prescribed fire, spatially explicit focal areas, understory restoration, and other considerations. The plan recognized the importance of both public and private lands in achieving the acreage goals for longleaf pine and outlined broad strategies for both.

In 2010, a Memorandum of Understanding (MOU) was signed between the Departments of Agriculture, Interior, and Defense establishing a Federal Coordinating Committee (FCC) to begin implementation of the plan. The MOU called for the establishment of a broader partnership to include State agencies, non-profit conservation organizations, and private sector participants. A stakeholder scoping process was conducted to gather input on the structure and function of this broader partnership and in fall of 2011, the initial meeting of the Longleaf Partnership Council (LPC) was held. The primary purpose of the LPC is to serve as a forum for communication and collaboration in implementation of the plan. The LPC has 33 seats, which are designed to be representative of the diversity of interests in the longleaf pine conservation and management community. Since the founding of the LPC, local implementation teams have formed around each of the focal areas, which are now called Significant Geographic Areas (SGA). These teams consist of local stakeholders and longleaf pine conservationists working collaboratively to manage and restore longleaf pine within their respective SGAs. One important function of the LPC and the local teams is to report restoration accomplishments annually across the range and to critically assess the overall progress towards the goals of the plan (LPC 2014, 2015, 2016).

CURRENT STATUS OF LONGLEAF PINE

The implementation of the ALRI conservation plan began in 2010; coincidentally, 2010 also marked the completion of a full panel of Forest Inventory and Analysis (FIA) plots for longleaf pine. Forest Inventory and Analysis considers two forest types as longleaf pine – longleaf pine and longleaf pine/oak – and plots sampled in these forest types indicated a total of approximately 4.29 million acres in 2010 (Oswalt and others 2012). Also in 2010, the USDA Natural Resources Conservation

Service launched the Longleaf Pine Initiative, which offers incentives and technical assistance to private landowners interested in longleaf pine (USDA NRCS 2017). In 2012, a small working group from the LPC developed a 3-year strategic plan for implementation of the conservation plan that spanned the years 2013–2015 (LPC 2012). This plan set yearly goals for longleaf pine establishment, prescribed fire, and other restoration activities. The year 2012 also marked the beginning of the Longleaf Stewardship Fund, a public-private grant fund, administered by the National Fish and Wildlife Foundation, that provides more than \$4 million annually to support the local implementation teams and on-the-ground restoration activities.

Through these efforts and programs, in 2013 the LPC began collecting data to measure progress towards the ALRI goal. With 3.7 million acres needed to meet the 8-million-acre goal by 2025, an annual average of about 250,000 acres would be required. As 3-year step-down goals were set for the 2013–2015 strategic plan, it was assumed that initial goals from 2013–2015 would realistically be less than that average, but hopefully momentum would grow sufficiently to reach the target by 2025. For 2013–2015, longleaf pine planting goals were initially set at 105,000–130,000 acres, with prescribed fire to benefit longleaf pine at a target of 1.4–1.7 million acres (LPC 2012). These goals were largely achieved – newly-established plantations of longleaf pine exceeded 150,000 acres annually, for a 3-year total of almost 460,000 acres, and prescribed fire gradually increased to attain a 3-year total of 3.9 million acres burned.

Acreage totals for planted stands were calculated using two sources of information. Approximately half of the establishment acreage every year was supported under incentive programs, with related reporting and monitoring. The remaining acreage figures were gleaned from longleaf pine seedling sales data from the Southern Forestry Nursery Cooperative at Auburn University.¹ Assumptions used to calculate acreage from number of seedlings sold were that 1) 90 percent of the seedlings sold were planted, and 2) that seedlings were planted at 650 trees per acre. These assumptions should yield relatively conservative estimates since many landowners plant at lower densities for wildlife benefits. When the establishment data are combined with seedling sales data from 2011 and 2012, it is estimated that the first 5 years of the ALRI resulted in 724,000 acres of newly established longleaf pine which, when added to the 4.3-million-acre estimate in 2012, suggests that there should be nearly 5 million acres of longleaf pine rangewide in 2015.

¹ Personal communication. 2016. Scott Enebak, Director, Southern Nursery Management Cooperative, 3301 Forestry and Wildlife Sciences Building, Auburn University, AL 36849-5418.

Unfortunately, this optimistic accounting of increased establishment does not factor in the reality that longleaf pine acres are being lost to other forest types and land uses. Forest Inventory and Analysis data suggest that between 2010 and 2015, the longleaf pine forest type increased by 204,000 acres, while longleaf pine/oak forest type decreased by 209,000 acres, for a net loss of 5,000 acres. Essentially, the total acreage of longleaf pine-dominated forest types remains unchanged since 2010. America's Longleaf Restoration Initiative data suggest that there were substantially more acres of longleaf pine forest type established than FIA data suggest. This is not necessarily inconsistent – whereas ALRI figures represent a total inventory, FIA numbers are derived from a coarse-scale distribution of sampling plots, of which 20 percent are re-measured annually across a 5-year cycle, creating a potential time lag for reporting. The important point that emerges from the FIA data is that whatever the actual numbers may be, indications are that losses are still equivalent to gains.

These estimates of gains and losses are sobering for those working on the restoration of longleaf pine across the South. Despite tremendous efforts and momentum, longleaf pine acreage remains essentially unchanged over the last 5 years. Clearly, the goal of reaching 8 million acres is even more challenging now. Throughout 2016, the LPC and FCC reviewed this situation and discussed options. Ultimately, the consensus was that the acreage goal and the timeline would remain the same, and the LPC was tasked with developing a proposed framework for a redoubled effort for ALRI.

GAME CHANGERS

In response, a list of seven action areas, or “game changers,” has been developed and proposed. Collectively, these strategies have the potential to significantly accelerate progress toward the goal. A common thread running through the strategies is that they each involve additional effort, and additional resources, dedicated to longleaf pine restoration.

Increased Restoration on Public Lands

Public lands represent perhaps the best investment of resources for longleaf pine restoration. These lands are permanently protected from development, dedicated to long-term management that includes conservation goals, and managed by agencies with natural resource professionals on staff. Within the range of longleaf pine, there are approximately 13.6 million acres of Federal and State-owned public lands (USGS 2016). Even though only a subset of the total public lands are suitable longleaf pine sites, clearly there is room for expansion of longleaf pine acres in the public sector. For example, about 4 million acres of National Forest System (NFS) ownership lies within the historic range of longleaf pine (USGS 2016), but currently, estimates of existing longleaf

pine on NFS lands are approximately 800,000 acres. There are certainly opportunities to add to the current total; for example, the Francis Marion National Forest recently revised its management plan and identified a long-term goal for longleaf pine that doubled the acreage identified in the old plan (USDA Forest Service Southern Region 2016).

Although the USDA Forest Service has taken a leadership role in assessing their public lands resources relative to longleaf pine, there are also significant acreages of other public lands where similar opportunities to expand longleaf pine on suitable sites exist. These include Department of Defense properties, National Wildlife Refuges, State forests, State wildlife management areas, and others. There are currently significant numbers of acres of mixed stands containing longleaf pine, other southern pines, and hardwoods on public lands that could be shifted to longleaf pine dominance with judicious removal of the non-longleaf pine components (Guldin and others 2016). Because the acreages potentially available for longleaf pine restoration on public lands are large, it should be recognized that the time necessary to execute restoration on the ground is likely to exceed the timeline of the ALRI goal. Conversion of such large acreages will involve extensive planning and sequencing of treatments that realistically incorporate the time required to conduct operations, as well as practical considerations such as allowing existing younger stands to reach sufficient maturity for harvest and subsequent conversion to longleaf pine. Assessment of progress towards the goal should acknowledge the time scale inherent in forest management and factor into accounting those acres committed to longleaf pine restoration, but not yet operationally underway or completed.

Increased Restoration on Private Lands

Privately owned forests will be a critical component in reaching the 8-million-acre goal for longleaf pine. In 2012, private ownership controlled approximately 87 percent of the South's forests, with about one-third of that in corporate ownership and two-thirds held by noncorporate or “family forest” owners (Oswalt and others 2014). From 2013–2016, private lands have accounted for 82 percent of the documented longleaf pine establishment, with about half of that acreage supported by incentive programs (LPC 2014, 2015, 2016). Although incentive program support for longleaf pine establishment is substantial, several States typically have demand for these programs that exceeds available funding. We estimate that longleaf pine establishment on private lands will need to at least double to make meaningful progress towards the goal. With stand establishment costs (site preparation and planting) of roughly \$300 per acre, this could mean as much as \$45 million will be needed annually to

support an additional 150,000 acres of establishment on private lands. Although demand for incentive support currently exceeds the supply of funds, even if that much additional funding was available there is no guarantee that demand would increase under current policies that focus on smaller-acreage landowners.

Seek Opportunities to Engage Large-acreage Corporate Landowners in Longleaf Pine Restoration

Corporate landowners include common forestry businesses such as timber investment management organizations (TIMOs) and real estate investment trusts (REITs), but also include family trusts, limited liability corporations, and others. Historically, these entities have not been eligible for most incentives programs due to the programs' limits on adjusted gross income or acreage caps. Furthermore, when judged simply by volume growth and capital value over a short time horizon, the economic performance of longleaf pine investments is lower than investments in faster-growing pines like loblolly and slash pine. As a result, managers often decide to choose species that are economically competitive to satisfy fiduciary obligations to their principals. However, corporate ownerships represent an untapped opportunity to increase longleaf pine establishment on private lands and grow demand for incentive programs. Revision of current Farm Bill policies to allow larger corporate landowners to access these programs could, in many cases, lessen fiduciary concerns for corporate owners. Income from incentive support early in the analytical cycle could minimize or eliminate economic opportunity costs, and thus fiduciary liabilities, associated with managing longleaf pine relative to other species of southern pine. More fully incorporating the long-term nature of forest dynamics and forest management into policy could also foster greater engagement from this group of landowners.

Shift in the Message of Urgency and Importance

One of the major drivers in the longleaf pine restoration efforts has been the desire to benefit the unique suite of plant and, especially, animal species that depend on the habitat provided by the mature, fire-maintained, open structure of well-managed longleaf pine forests and woodlands. Currently, there are 30 species associated with longleaf pine ecosystems that are listed under the Endangered Species Act (ESA) and over 50 additional at-risk species (USFWS 2017). Further listings of longleaf-associated wildlife would bring significant economic costs, particularly for forest industry and private landowners, and could jeopardize the Nation's military readiness by restricting training on Department of Defense installations. For example, the Bonneville Power administration estimated the annual economic impact of salmon conservation efforts at \$350 million for the year 1994 (NRC 1995). A recovery plan that increased the survival odds for the northern spotted owl to 91

percent was estimated to decrease economic welfare by \$33 billion dollars (1990 dollars), with a majority of the impact related to the regional forest products industry (Montgomery and others 1994).

Forestry is a significant component of the South's economy. The total economic output of the region's wood-related sectors in 2009 was approximately \$230 billion (Abt 2013). Successful restoration and management of longleaf pine ecosystems at the scale articulated by the ALRI goal can play a critical role in precluding the need to list many of these species, thus providing justification for the significant investments required and avoiding the economic impact of further listing. Conservation strategies that encourage, rather than discourage, landowners from managing longleaf pine ecosystems and that offer more regulatory certainty would provide better outcomes for both at-risk species and private landowners. Broader articulation of the importance of reducing the need to list additional species is necessary to bolster the case for greater resources and accelerated restoration of longleaf pine ecosystems.

Promote Longleaf Pine Opportunities and Proposals for Gulf Restoration Funding

The 2010 British Petroleum oil spill in the Gulf of Mexico resulted in the largest civil penalty ever assessed in the United States, approaching \$21 billion dollars. As much as two-thirds of these funds are earmarked for natural resource restoration and remediation. Adequate quantity and quality of freshwater is critical to assist in the recovery of nearshore estuarine systems and coastal wetlands (Alber 2002). Fire-maintained, moderately stocked longleaf pine forests use less water than other pine forest types, potentially increasing fresh water downstream (Brantley and others 2017). Strategically located longleaf pine restoration and land protection projects that buffer creeks and rivers can benefit coastal ecosystems by supplying greater quantities of high-quality fresh water. Thus, a strong argument exists for the use of some of these funds to further the ALRI goals. Federal agencies at the departmental level and State governments have some degree of purview over allocation of these funds, and these entities should advocate for funding of longleaf pine restoration projects where appropriate.

Increase Support for Prescribed Burning

Frequent, low-intensity fire is an essential and naturally occurring ecological process that maintains the structure and function of longleaf pine forests and the habitat that they provide. Today, prescribed fire is the land manager's surrogate for this ecological process. To achieve the 8-million-acre goal, maintenance of a fire return interval of no more than 3 years will require annual prescribed burning on an additional million acres above current

levels (estimated at 1.6 million acres in 2016). Assuming an average cost of \$25 per acre, this could represent a cost of \$25 million dollars annually. Beyond direct costs, increased acreage of prescribed fire will require greater capacity for implementation, including more trained personnel, equipment, and agency support. Although realization of the 8-million-acre goal for longleaf pine may be years away, many of the ecological benefits, such as habitat for at-risk species, could be achieved sooner by greater application of prescribed fire to existing mature stands of other pine species in anticipation of their actual conversion to longleaf pine.

Expand Support for Land Protection through Fee Title and Easement Acquisitions

Considerable progress continues to be made in establishing new plantings of longleaf pine, but concurrent losses continue to hold back potential net acreage gains. Some proportion of those losses can be attributed to harvest and land use change. Many of these losses are from older, mature stands and although new plantings may potentially offer the long-term benefits associated with longleaf pine ecosystems, it will take decades for those attributes to develop. Land protection is an important component of an overall conservation portfolio for longleaf pine and is key to maintaining the investment in time that mature longleaf pine represents. Identification of significant vulnerable longleaf pine tracts and prioritization of those sites through land protection programs such as the Forest Service Forest Legacy Program, the Land and Water Conservation Fund, nongovernmental land conservation organizations, and others are needed to slow the loss of these important sites.

SUMMARY

Although the continued loss of longleaf pine has largely offset the acreage gains that ALRI has achieved, in the absence of the ALRI efforts the decline of longleaf pine acreage would have continued. It is hoped that future FIA estimates will begin to reflect gains as the time lag inherent in spreading plot surveys over 5 years catches up to known establishment figures documented by the LPC. Encouragingly, 2016 data from Alabama and North Carolina show net gains of approximately 30,000 acres (Miles 2017). The LPC is also exploring details of losses, which are occurring only in the longleaf pine/oak forest type – are these losses due to land use change such as urban development, conversion to pine plantations, or natural succession to other forest types due to fire suppression? The answer to these questions will help direct efforts to reduce such losses in the future.

Overall, ALRI has been successful and is often held up as a model conservation partnership. These achievements have been sustained by the tremendous commitment and enthusiasm of the longleaf pine conservation and management community, enabled

through moderate amounts of financial support from Federal agencies and the private sector. However, more work and more funds are needed if the 8-million-acre goal is to be met in a timeframe that is reasonably close to that set forth in the ALRI conservation plan. The proposals outlined in these seven game changers are ambitious and represent a significant increase in resources dedicated to the ALRI goal. The justification for this is simply stated—unlike other forest types, longleaf pine forests and their associated biota are extremely underrepresented on the southern forest landscape.

Although the level of funding that these proposals represent may seem unrealistic at first glance, this level of expenditure is not without precedent. For example, faced with the potential Federal listing of the greater sage grouse, the USDA Natural Resources Conservation Service (NRCS) launched the Sage Grouse Initiative (SGI) in 2010. The SGI is a partnership of ranchers, agencies, universities, non-profits, and businesses working to conserve sage brush habitat and its associated wildlife through sustainable ranching. Through Farm Bill programs, the NRCS has dedicated \$751 million to the SGI. In 2015, largely due to the unprecedented conservation partnership of the SGI, the U.S. Department of the Interior Fish and Wildlife Service's status review for the sage grouse determined that protection under the ESA was not warranted and withdrew the species from the candidate species list. Elevating the ALRI to a similar level of support could have similar results for the many species of longleaf-associated wildlife that are currently listed or under consideration for Federal listing, such as the red-cockaded woodpecker, gopher tortoise, gopher frog (*Rana capito*), striped newt (*Notophthalmus perstriatus*), and Louisiana pine snake (*Pituophis ruthveni*).

Longleaf pine ecosystems are among the rarest ecosystems in North America (Noss and others 1995), with many of the wildlife associates similarly imperiled. Although the costs for redoubling efforts to achieve the goals of the conservation plan are high, the costs of continued loss of habitat and the economic impacts of further listing of species under the ESA may be higher (Brown and Shogren 1998). The ambitious ALRI goal was established as a long-term strategy designed to reverse the loss of longleaf pine acreage, establish management regimes that ensure the development of ecosystem values while maintaining working forests, and ultimately restore viable populations of declining wildlife species that help preclude further listings under the ESA. ALRI has demonstrated the ability to make significant progress with relatively modest resources, but we are not on pace to meet the goal. Greater resources will be required to achieve the desired results, and the ALRI track record suggests the initiative is up to the challenge.

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LONGLEAF PINE: RESTORATION OR REFORESTATION?

John R. Brooks and Steven B. Jack

Abstract—For the past 20 years, there has been an interest in increasing the existing acreage of longleaf pine (*Pinus palustris* Mill.) across the Southeast. Due to the constraints of early cost-sharing programs, many of these plantings were established at low densities [400 trees per acre (TPA)], when compared to operational pine planting densities in this region. Visual evaluation of these plantings reveals trees that tend to have much larger crowns with large lower limbs that are not quickly pruned during early stand development, even after crown closure. Utilizing an existing long-term planted longleaf pine growth-and-yield study in southwest Georgia that includes both low (300–400 TPA) and high (800–900 TPA) planting densities, this study attempts to quantify whether these differences in tree development vary by planting density. Results indicate that low-density plantings had a significantly higher frequency and average size of limbs per tree in the first 10 feet of the main bole. In addition, these lower planting densities possessed a higher frequency of branches larger than 1 inch (at the point of bole attachment) than higher density plantings. Although the initial cost-sharing programs had wildlife and ecological objectives, many private landowners had timber objectives as well. These results necessitate the question: Are we simply reforesting this species, or are we restoring stands of this valuable timber species that was once prominent in the South?

INTRODUCTION

There has long been an interest in the reforestation of longleaf pine (*Pinus palustris* Mill.) in the Southern United States, primarily due to the memory of tall, straight stems growing in low-density stands with little or no midstory. However, early planting attempts were often plagued with low survival rates. It was not until the late 1980s when silvicultural prescriptions were refined to ensure successful planting survival. These silvicultural procedures included, but are not limited to, keeping seedlings refrigerated on the planting site and ensuring proper planting depth and adequate site preparation techniques as well as the development of containerized longleaf pine seedlings. By the mid-1990s, the availability of Federal cost-sharing programs piqued the interest of many nonindustrial private forest landowners (NIPF) willing to enroll in such programs. These early cost-sharing programs were focused on longleaf pine ecosystem restoration for the benefit of wildlife species and required wide spacings (low density) to maximize wildlife benefits. Many landowners had dual interests in such programs: to help restore longleaf pine as well as provide a timber crop. As the number of acres planted to longleaf pine continues to increase, early stand development visually does not mirror the memory of this once prominent species. Trees tend to have much larger crowns with large lower limbs that are not quickly pruned during early stand development, even after

crown closure (fig. 1). This study attempts to quantify whether these differences in tree development vary by planting density and to recommend changes that may satisfy the dual nature of the NIPF interest in planting longleaf. Are we simply reforesting this species or are we restoring this valuable timber species that was once prominent in the South?

METHODS

The data used for this analysis are based on a long-term planted longleaf pine growth-and-yield study located in the Flint River Basin of southwest Georgia. This dataset includes two subpopulations: low-density plantings [300–400 trees per acre (TPA)] typical of the Conservation Reserve Program (CRP) requirements in the early 1990s and high-density plantings (800–900 TPA) typical of operational planting densities. A more detailed description of this dataset is provided by Brooks and Jack (2016).

An initial study was conducted to determine whether analysis of photographic images taken of the main bole could accurately identify and determine the location and the branch diameter at the point of attachment for each limb, live or dead, to the main bole. An 8-megapixel digital camera was used on a tripod at a distance of approximately 30 feet to photograph four quadrants of the first 10 feet of the main bole. For each image,

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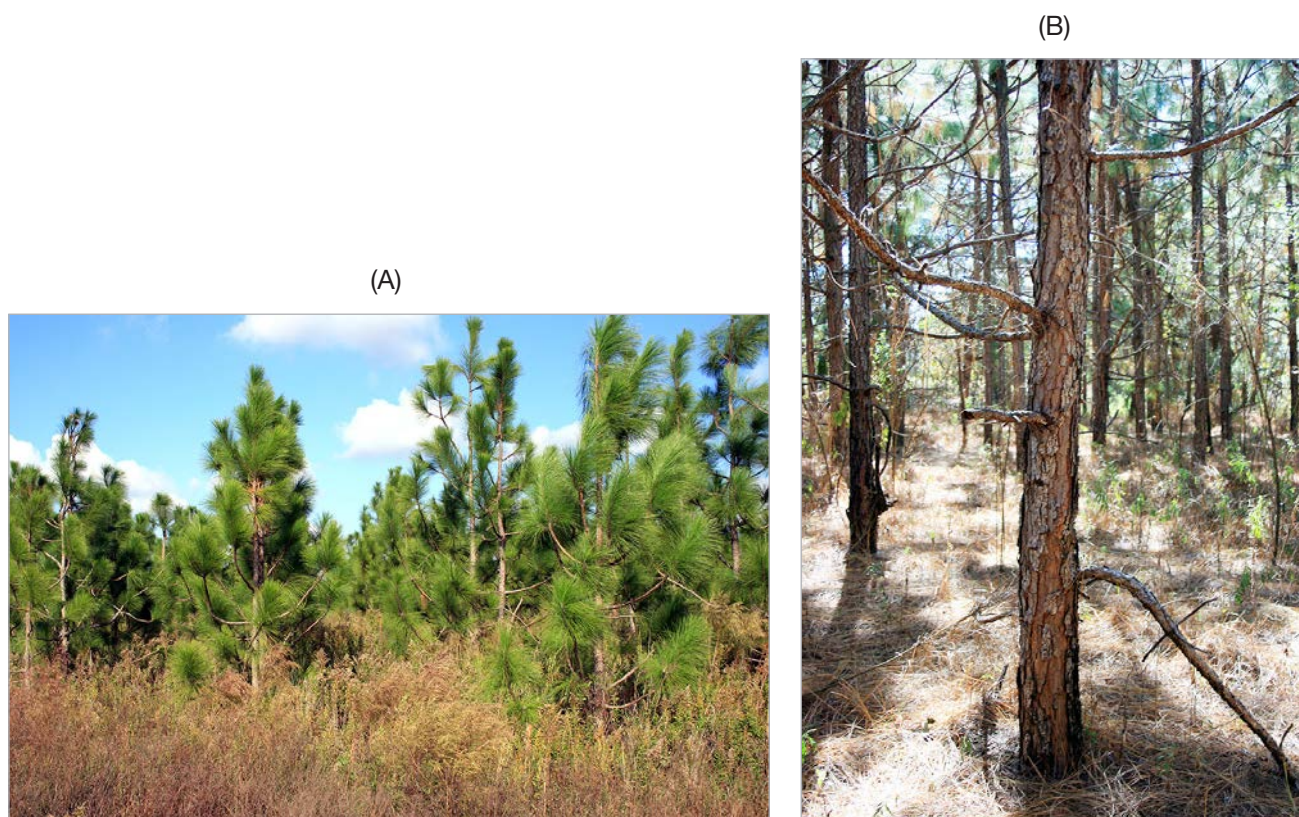


Figure 1 —Low-density planted longleaf (A) and low-density branch development (B).

a 5- x 6-inch target was placed at 4.5 feet above the ground for image calibration (fig. 2). In addition, each branch on each tree was mapped in the field and the diameter subsequently measured using a digital caliper and recorded to the nearest 0.01 inch. One row from a single permanent growth-and-yield plot was selected at random, and every tree in that row was photographed and each branch directly measured.

Based on the successful image analysis test, seven additional stands were selected for further study. As in the preliminary study, one row was selected at random from each permanent plot resulting in data from four low-density and four high-density plantings. A 5- x 6-inch target was again placed at 4.5 feet above the ground for each image for image calibration. Obviously a function of density, the number of sample trees per row ranged from 7 to 19. Stand age ranged from 14 to 25 years. Based on UTHSCSA ImageTool (Version 3.0) image analysis software (UTHSCSA ImageTool 2002), the number and size of each branch attached to the main bole within the first 10 feet were estimated from each photographic

image. Branch diameter was assigned a class size based on branch dimension at bole attachment (table 1). In addition to the permanent growth-and-yield plots, one plot was established in a young natural longleaf stand where six trees were included for comparison purposes.

Data were analyzed using a two-sample t-test (SAS Institute Inc. 2010) for the number of branches and average branch diameter as well as average branch size class. Analysis of the relation between these variables and planting density was reviewed for existing trends.

RESULTS

The initial image analysis investigation was based on one row of trees from a single plot that included seven trees and 58 branch measurements. Average height bias was determined to be 0.04 feet [root mean squared error (RMSE) of 0.21 feet] for height of branch attachment on the bole, and average diameter bias was -0.01 inch (RMSE of 0.16 inches) for branch diameter. These results indicated that the photographic procedure utilized was suitable for the accuracy needed in this study.



Figure 2—Photographic target used for image analysis.

Table 1—Branch size class assigned based on estimated branch diameter

Min. size <i>inches</i>	Max. size <i>inches</i>	Class
0.01	0.50	1
0.51	1.00	2
1.01	1.50	3
1.51	2.00	4
2.01	2.50	5
2.51	3.00	6
3.01	3.50	7

Based on the perceived difference in tree form and branch pattern between natural longleaf regeneration and current plantations, this analysis investigated the differences in average branch size (live or dead), the average number of branches per tree (first 10 feet of the bole), and the average frequency of branch size class per tree as they relate to initial planting density. The low-density plantings had a larger average branch size, a higher number of branches per tree, and a larger number of branches >1 inch in diameter (size class 3 or larger) (fig. 3). The distribution frequency of branch size classes (table 1) by planting density group is displayed in figure 4. The low-density plantings contained more branches in the larger class sizes. In this study, the high-density plantings were more similar to natural stand data in branch number, branch diameter, and branch class distribution.

Because the distribution for average branch diameter and the number of branches per tree are skewed to the right (fig. 3), and these two populations have different variances, two-sample t-tests were conducted using log transformation with the Cochran and Cox (1950) approximation. Average branch size was significantly different ($Pr > |t| = 0.0001$) with the low-density plantings having an average of 1.03 inches and the high-density planting having an average of 0.73 inches. Average number of branches per tree size was significantly different ($Pr > |t| = 0.0001$) with the low-density plantings having an average of 13.16 branches per tree and the high-density planting having an average of 5.65 branches per tree.

CONCLUSIONS

The number and size of branches were larger for low-density plantings, and these branches tend to remain long after crown closure. The low-density plantings (340–410 TPA), typical of CRP plantings common in the mid-1990s, had a significantly higher frequency (13.16) of branches per tree as well as a significantly larger average diameter (1.03 inches) per tree. In addition, these lower planting densities possessed a higher frequency of branches larger than 1 inch (at the point of bole attachment; fig. 4) than higher planting densities (871–940 TPA). If the objective of reforesting longleaf pine in the South is to increase the acreage of this species within its historic range as well as to provide restoration of a valuable sawtimber tree, it begs the question of whether the low-density plantings

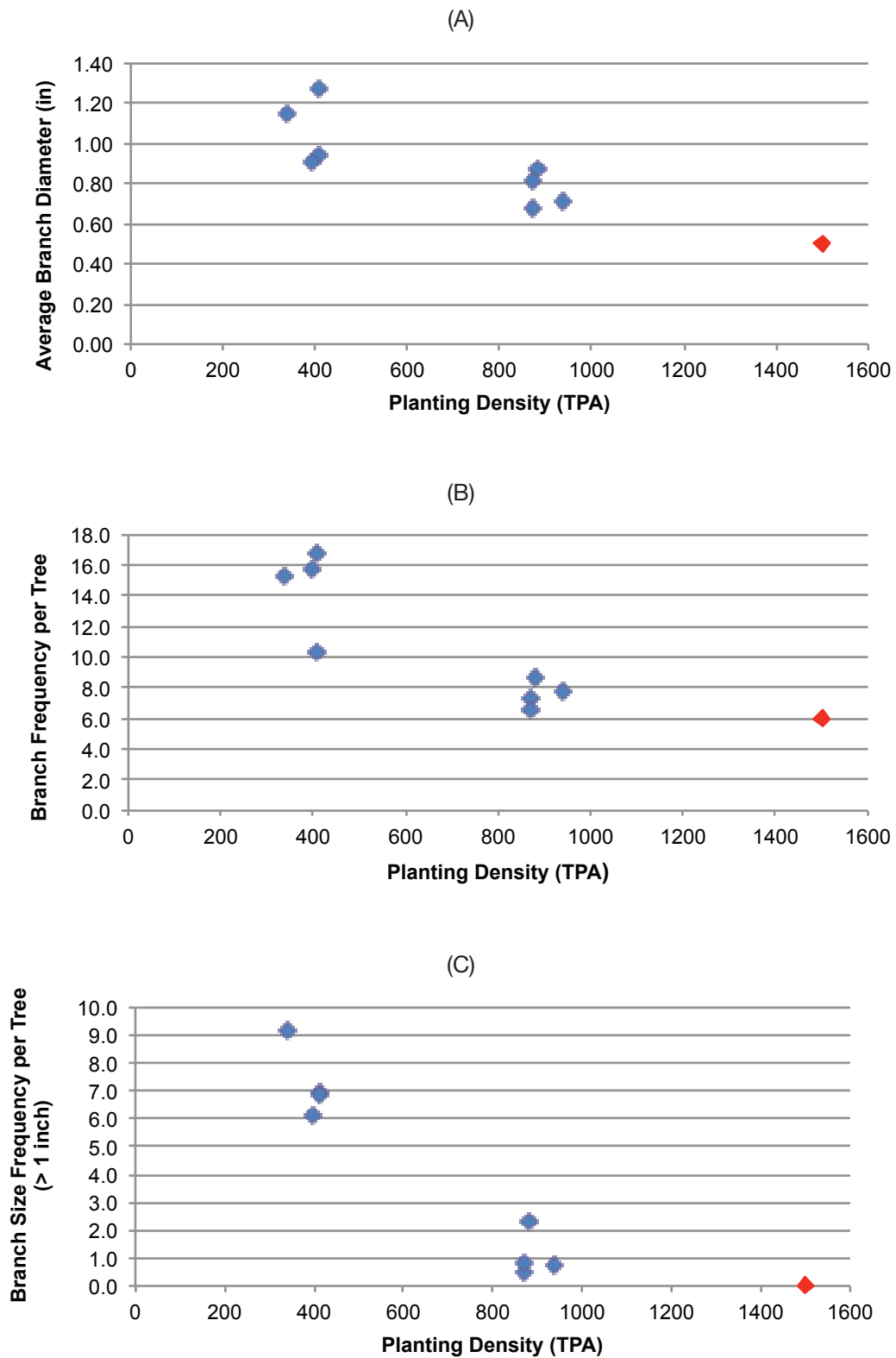


Figure 3—Average branch diameter by planting density (A), average branch frequency per tree (B), and branch size frequency per tree for branches >1-inch diameter (C) for planted and natural longleaf pine (1,500 TPA).

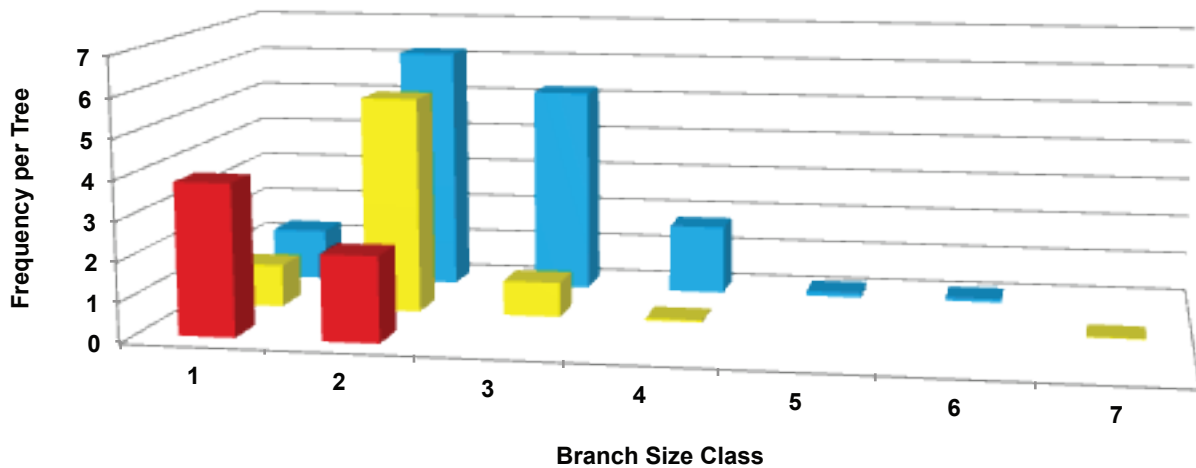


Figure 4—Distribution of branch size class frequency per tree for natural regeneration (red), high-density plantings (gold), and low-density plantings (cyan).

will, in the long run, provide the tree form that many envision for this species based upon tree form observed in natural stands. There is no question that the cost-sharing programs have provided a means to expand the distribution as an estimated 556,000 acres have been planted under these programs since 1998 (Kirkman and others 2017). Whether these plantings provide the timber revenue anticipated by many NIPF landowners is yet to be seen. Observed differences in tree form and branch characteristics greatly impact product classes (e.g., pole class/quality) and thus also potential economic return. Low-density plantings also have a greater risk of severe understocking if planting survival is lower than anticipated. Even with adequate planting survival, these low-density plantings exhibit a heavy branching pattern that remains long after crown closure, resulting in tree form much different than similar-aged high-density plantings.

While many cost-sharing programs promoted low-density plantings to meet strong wildlife objectives, it is our evaluation that the wildlife benefit to these plantings is short-lived, as even at low densities the understory and associated grass communities disappear following crown closure (fig. 5). As an alternative, work conducted by the Joseph W. Jones Ecological Research Center has revealed that warm season grasses can be successfully established when planted in the removal rows following a first thinning (fig. 6). Using this procedure, stands



Figure 5—Five-year-old planted longleaf pine established with 410 TPA and interplanted with wiregrass (A) and the same stand at age 16 (B) with a greatly reduced cover of wiregrass following canopy closure.



Figure 6—Take-out rows seeded with wiregrass and lop-sided Indian grass following first thinning showing grass development 12 years following seeding operations.

can be established at more typical operational planting densities to minimize lateral branch number and size, provide intermediate rotation cash flows for landowners through pine straw harvesting and thinning, and provide opportunities to introduce warm season grasses following thinning that have a higher probability of long-term survival.

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LONG-TERM EFFECTS OF PRESCRIBED FIRE SITE PREPARATION ON LONGLEAF PINE REGENERATION

Mary F. Nieminen and Steven B. Jack

Abstract—Natural regeneration in longleaf pine (*Pinus palustris* Mill.) stands is highly variable and dependent on several site factors, particularly implementation of fire, that affect germination, survival, and growth. A long-term monitoring project was established in 1987 prior to an expected good seed crop in southwest Georgia. One hundred circular plots (0.03 ha) were established with 50 plots burned and 50 plots left unburned prior to seedfall; plots were burned on a biennial cycle following seed establishment. In 2004, 17 years following the initial plot establishment and seedfall event, 79 of the original plots were relocated and surveyed for all longleaf pine regeneration. Seedlings and saplings were measured for total height and diameter (root collar diameter or diameter at breast height, based on height), permanently tagged, and stem mapped for subsequent measurements. Measurements of seedlings were conducted in 2004, 2005, 2006, and 2015. Stem densities differed between burned and unburned plots in all measurement years. Regardless of density, the rates of transition to larger size classes and mortality were markedly similar between initially burned and unburned plots. Though average densities were greater in burned plots, the long-term mortality rate was greater in unburned plots with lower stem densities. Overall, there are continuing long-term effects of prescribed fire site preparation on pine regeneration densities, further supporting the importance of properly planned management and cone crop monitoring for successful recruitment and maintenance of longleaf pine regeneration.

INTRODUCTION

Longleaf pine (*Pinus palustris* Mill.) forests are biologically diverse ecosystems that incorporate a complex suite of environmental processes, both natural and human-driven, that not only maintain plant and animal diversity but also perpetuate the forest type itself (Landers and others 1995). Ecologically functional longleaf pine forests have been reduced to a fraction of their historical extent (Frost 2006), and losses of established longleaf pine stands continue to increase regardless of concerted efforts to conserve these established forests. Though there has been much progress made toward the re-establishment of longleaf pine throughout its historical range (Guldin and others 2016), there are still many questions regarding natural processes of longleaf pine regeneration within established forest stands. Natural regeneration in longleaf pine forests is highly variable and dependent on several site factors that affect germination, survival, and growth. Canopy gap creation and varying gap sizes may mimic natural processes favoring natural regeneration (Gagnon and others 2004); however, there are other factors that can control successful recruitment within gaps such as microsite suitability (including light, rainfall, and nutrients) (Lancaster and Downes 2004; Palik and others 1997, 2003), and timing

of fire introduction (Croker and Boyer 1975, Walker and Wiant 1966), among many other factors. A frequent fire regime and appropriate application of prescribed fire are particularly important to provide suitable seedbed conditions for germination, establishment, and survival of longleaf pine seedlings and for control of competitive understory hardwoods (Brockway and Lewis 1997, Brockway and Outcalt 1998, Croker and Boyer 1975). Long-term growth and size class transitions of seedlings and saplings are not well-documented for longleaf pine, though we know that some seedling stages can be prolonged due to several environmental factors (i.e., long-lived grass stage). By addressing seedling size class transitions and mortality rates at small scales, we may enhance our understanding of how regeneration rates and densities affect our ability to perpetuate longleaf pine stands effectively.

OBJECTIVES

The primary objectives of this study were to: (1) assess differences in small-scale regeneration structure over time; (2) assess rates of seedling growth and transition through the various size classes of longleaf pine regeneration; and (3) assess potential long-term effects of operational prescribed fire on longleaf pine regeneration.

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METHODS

Study site

The study site is located on Ichauway, a 115-km² ecological reserve in the Coastal Plain of southwest Georgia (fig. 1). Ichauway has one of the largest continuous tracts of naturally regenerated second-growth longleaf forest in the Southeast (Kirkman and others 2004). The landscape is managed with low-intensity, primarily dormant season, prescribed fire with a 1–3-year return interval.

Project history

A long-term monitoring project was initiated on Ichauway prior to an expected large seed crop in 1987 to monitor the effects of operational prescribed fire on subsequent establishment of longleaf pine regeneration. One hundred circular plots (0.03 ha) were established with 50 plots burned (i.e., “burned plots”) in August prior to seedfall and 50 plots left unburned (i.e., “unburned plots”); all plots were burned on a biennial cycle following a fire-free period allowing for initial seedling establishment. Plots were distributed across the property under canopy openings surrounded by mature seed-producing longleaf pines with little or no advance regeneration within openings (fig. 1). Monitoring of initial

plots was not formally conducted until 2004, when 79 of the original 100 plots were relocated and surveyed to determine the amount of established regeneration.

Data Collection

In 2004, 79 of the original plots were resurveyed for all longleaf pine regeneration. Seedlings and saplings were measured for total height and diameter [root collar diameter (RCD) and/or diameter at breast height (DBH)], permanently tagged, and stem mapped for subsequent measurements. Seedlings were classified into various size classes by appropriate diameter and height measurements (table 1). Seedling measurements were also conducted in 2005, 2006, and 2015. Mortality, size class distributions, and transition rates were assessed for each sample year following the 2004 sampling period.

Data analyses and summaries

Cumulative stem counts were summarized between burned and unburned plots by sample year. Medians and ranges were reported for central tendencies as data were non-linear and followed a Poisson distribution. Stem densities were assessed for overall differences between plot types across all size classes using a general linear

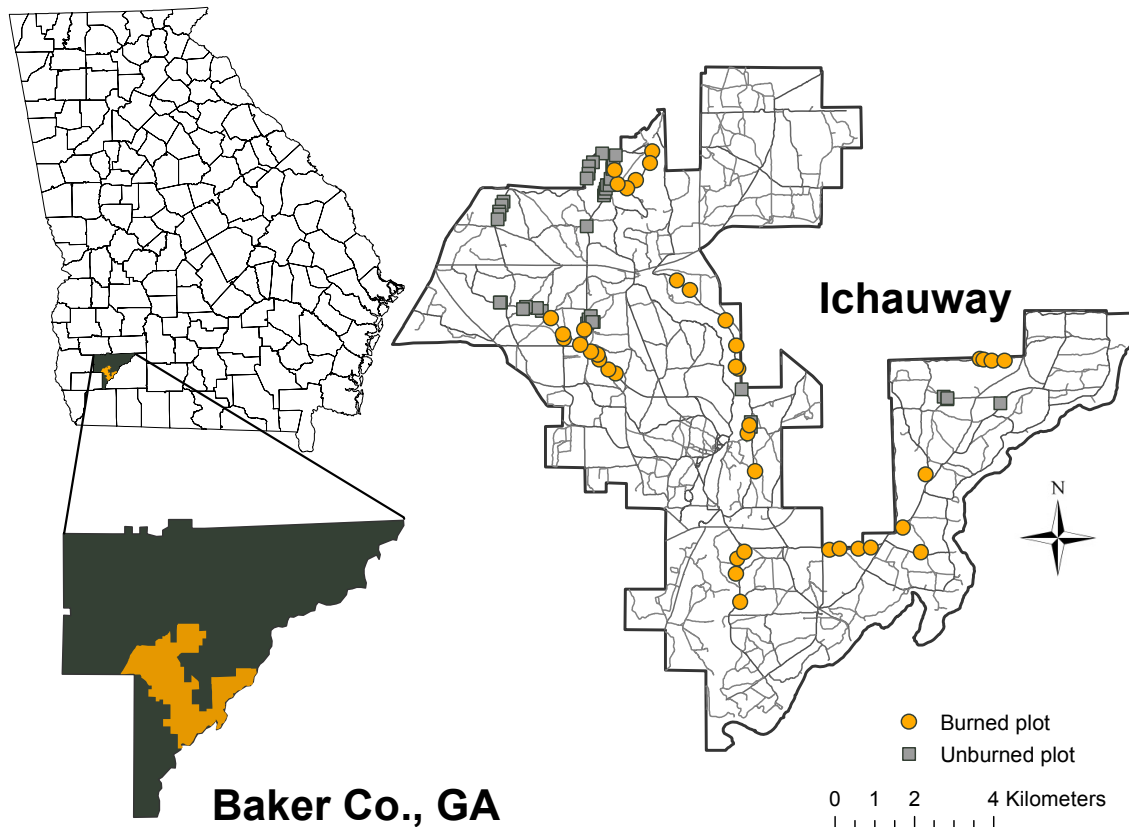


Figure 1—Study site and plot locations on Ichauway in southwest Georgia.

Table 1—Seedling size classifications based on diameter and height thresholds

Size class	Diameter thresholds	Height thresholds
Germinant (germ)	<10 mm RCD	
Grass stage	≥10 mm RCD	<0.5 m
Bolt stage	≥10 mm RCD, < 100 mm DBH	≥0.5 m, <2 m
Sapling stage	<100 mm DBH	≥2 m
Midstory	≥100 mm DBH	≥2 m

DBH = diameter at breast height; RCD = root collar diameter.

model (Proc GENMOD in SAS 9.4). Stem transitioning data were assessed across all plots (i.e., total number seedlings) and were assessed as collectively burned and collectively unburned seedlings. Overall mortality and transition rates by size class were summarized by plot type for each sample period.

RESULTS AND DISCUSSION

Seedling density

Seedling numbers were noticeably different between burned and unburned plots for all sample years (table 2). In 2004, all size classes, except midstory, were significantly different between burned and unburned plots (all p -values <0.0001). In 2005, all size classes, with the exception of germinants and midstory stems, were significantly different including dead stems (all p -values ≤0.0008). In 2006, all size classes, except midstory, were significantly different including dead stems (all p -values ≤0.0169). By 2015, all size classes, with the exception of grass-stage stems, were significantly different including dead stems (all p -values ≤0.0002). Through all sampling years, the sapling stage was consistently greater in average density within burned plots than unburned plots.

Survival and size class transitions

Seedling mortality from 2004 to 2005 was approximately 6 and 9 percent for burned and unburned plots, respectively (fig. 2). A large proportion of seedlings (approximately 80 percent) remained within the initial height class originally recorded during the 2004 sampling period (fig. 2A). Rates of seedling transition into other size classes were approximately 12 and 9 percent of all seedlings within burned and unburned plots, respectively. The majority of established seedlings (i.e., not including germinants) that did not transition into the next size class were within the sapling stage for burned plots and within the grass stage for unburned plots (fig. 2B). Size class transition rates were similar between burned and unburned plot seedlings with a small deviation within the bolt to sapling transition with a slightly greater percentage of stems transitioning in burned plots than unburned plots (fig. 2C). More germinant stems transitioned into grass-stage seedlings in burned plots (fig. 2C).

Table 2—Total number of sampled seedlings by plot type and sample year

Year	Burned ^a	Unburned ^a
2004	3,222 (49; 0–478)	1,531 (4; 0–542)
2005	3,017 (48; 0–448)	1,393 (4; 0–460)
2006	2,760 (43; 0–407)	1,298 (3; 0–439)
2015	1,837 (28; 0–286)	641 (3; 0–202)

^a Plot medians and ranges are given in parentheses.

The mortality rate from 2005 to 2006 was approximately the same for seedlings within burned and unburned plots (8 and 7 percent, respectively; fig. 3). Of the stems living in 2005, the majority remained within the same size class recorded at the beginning of the growth period (i.e., 2005) for both burned and unburned plots (fig. 3A). The majority of established seedlings that did not transition were within the sapling stage for burned plots and grass stage for unburned plots (fig. 3B). Seedling rates of transition into other size classes were approximately 15 and 29 percent of all seedlings within burned and unburned plots, respectively (fig. 3A). Size class transition rates were similar between burned and unburned plots within the three largest stem transitions. More germinants transitioned into grass-stage seedlings in unburned plots than burned plots (fig. 3C).

From 2006 to 2015, the mortality rate was approximately 33 and 51 percent (fig. 4) for burned and unburned plots, respectively. Approximately 44 and 28 percent of surviving seedlings did not transition into the next size class during this period for burned and unburned plots, respectively. Seedling rates of transition into other size classes were approximately 20 percent of all seedlings in both burned and unburned plots. The majority of established stems that did not transition were within the sapling stage for both burned and unburned plots (fig. 4B). Size class transition rates were similar between burned and unburned plot seedlings across all transition classes though differences were present between size classes (fig. 4C).

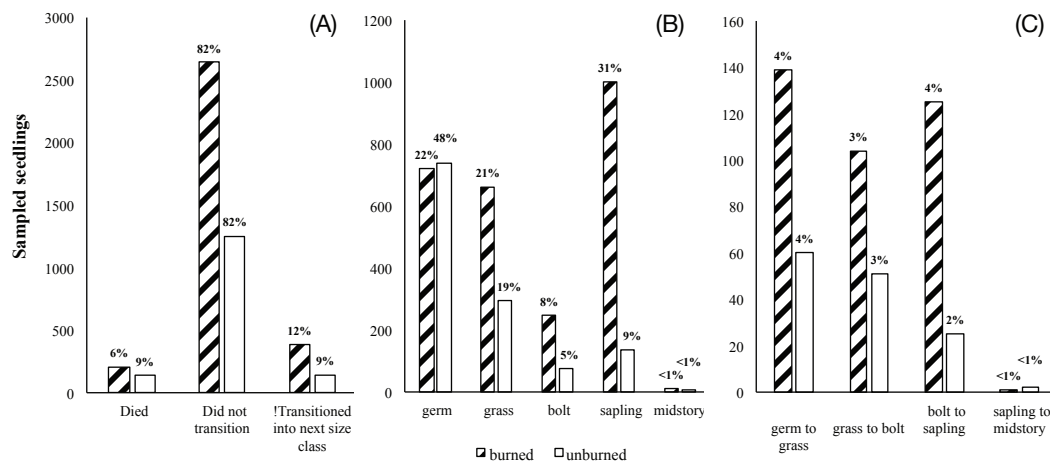


Figure 2—Survival and size class transitioning summary for the 2004–2005 growth period. (A) Overall transition and mortality rates between burned and unburned plot seedlings/saplings; (B) composition of seedlings that did not transition into the next size class for burned and unburned plots; (C) size class transitioning rates of seedlings within burned and unburned plots. Note differences in scale. Percentages within figures 2B and 2C are cumulative for all surviving stems reported in figure 2A.

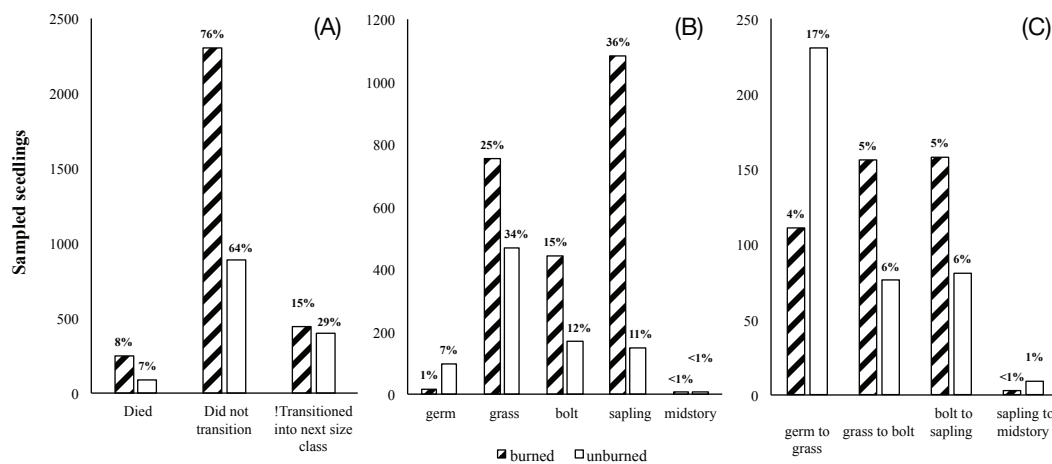


Figure 3—Survival and size class transitioning summary for the 2005–2006 growth period. (A) Overall transition and mortality rates between burned and unburned plot seedlings/saplings; (B) composition of seedlings that did not transition into the next size class within burned and unburned plots; (C) size class transitioning rates of seedlings within burned and unburned plots. Note differences in scale. Percentages within figures 3B and 3C are cumulative for all surviving stems reported in figure 3A.

DISCUSSION

Throughout all sampling periods, there were persistent differences in overall seedling numbers (table 2). With the initial sampling period in 2004, there was a large proportional difference in the number of saplings between burned and unburned plots. Sapling stage regeneration remained the primary size class within burned plots and increased in proportion for each sample period in burned and unburned plots regardless of differences in plot densities. The proportion of seedlings that remained in the sapling stage increased faster for burned plots than for unburned plots, though

the smaller rate of increase for the 2006–2015 growth period was likely due to increased mortality as larger stems competed for resources. In 2004, there were definite differences in plot type structure, but by 2015 these structural differences were nonexistent, though plot densities remained greater in burned plots than unburned plots.

During the 2004–2005 growth period, the general proportional seedling composition (i.e., died or survived) of burned and unburned plot regeneration was surprisingly similar (fig. 2A). The mortality rate was similar between burned and unburned plots for

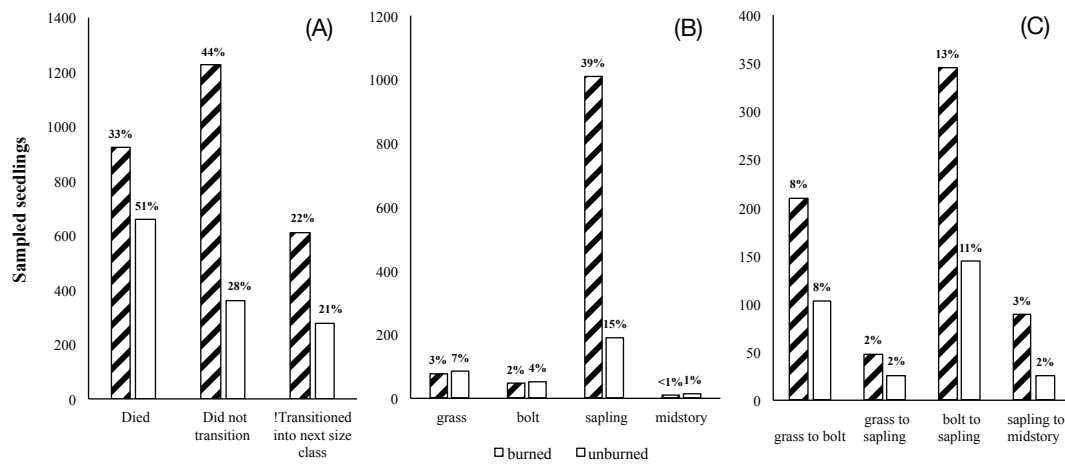


Figure 4—Survival and size class transition summary for the 2006–2015 growth period. (A) Overall transition and mortality rates between burned and unburned plot seedlings/saplings; (B) composition of seedlings that did not transition into the next size class within burned and unburned plots; (C) size class transitioning rates of seedlings within burned and unburned plots. Note differences in scale. Percentages within figures 4B and 4C are cumulative for all surviving stems reported in figure 4A.

the first two growth periods measured (i.e., 2004–2005 and 2005–2006); however, by 2015, the mortality rate was 20 percent greater in the lower density, unburned plots than for the higher density, burned plots. This large disparity in mortality rate was likely a relic of the initial structure sampled in 2004. It has been previously reported that mortality rate decreased as stems attain heights that reduce risk of fire-related damage or mortality (Croker and Boyer 1975), though Wang and others (2016) reported that growth rate and pattern do not affect longleaf pine’s ability to survive fire. Since the majority of stems within burned plots were already within the sapling stage in 2004, it is not surprising that the mortality rate was lower compared to unburned plots which were largely composed of new recruitment and grass-stage stems in 2004.

There were consistently similar rates of transition by established stems between burned and unburned plots (i.e., excluding germinants; figs. 2C, 3C, and 4C), which was an unexpected finding due to the large differences in the overall number of stems sampled and variable densities across plots and between plot types (table 2). Germinants were at similar stem densities regardless of plot type indicating germination was not affected by current conditions — seed availability, seedbed condition, first application of fire, and other management practices—or resource availability during early establishment; however, these factors become increasingly important as development and growth continues (Croker and Boyer 1975, Platt and others 1988). There was a noticeable difference in transitioning germinants between burned and unburned plots. More germinants transitioned into grass-stage seedlings within burned plots during the first growth period (i.e., 2004–2005) compared to unburned plot germinants, whereas

the opposite occurred during the second growth period (i.e., 2005–2006) with more germinants transitioning within unburned plots compared to burned plots. There were no long-term transitions of germinants during the 2006–2015 period indicating that if transitioning into the grass stage does not occur within the first two growing seasons following establishment, fire-related mortality is highly likely (Brockway and others 2006, Croker and Boyer 1975, Grace and Platt 1995, Provencher and others 2001, Walker and Wiant 1966).

In a study previously implemented on Ichauway to determine seedling cohort age (i.e., differentiating stems originating from the 1987 mast year and the 2006 mast year) by conducting ring counts, it was determined that most large stems (i.e., saplings) resulted from the 1987 seed year (unpublished data). These previous findings, in addition to the noticeable difference in the number of surviving saplings within burned and unburned plots, support the hypothesis that the initial fire event for site preparation in 1987 was a significant component in defining the structure of natural regeneration within our plots over time, regardless of limited sampling periods following initial establishment. Several longleaf pine masting events (1996, 1999, 2002, 2005, 2006, 2014; see Brockway 2016) have occurred on Ichauway during the years when additional monitoring was not conducted within the original plots. The importance of additional good masting years is difficult to quantify and increases complexity when defining the importance of initial site preparation and how additional seed events affect overall plot densities and regeneration structure. Reintroduction of fire following seedling establishment continued on a somewhat biennial basis; however, seasonality and frequency of fire varied across Ichauway adding to the complexity of quantifying plot differences.

Seedlings and saplings will continue to be measured for this long-term dataset regarding growth, size class transitioning, and survival of early and advance regeneration of longleaf pine seedlings. Fire frequency and seasonality will be assessed to determine impact of operational prescribed fire on long-term survival and growth of regeneration. Additionally, new seedlings will be mapped into plots as certain criteria are reached in the attempt to assess ingrowth and multi-cohort establishment within natural longleaf pine regeneration groups.

CONCLUSIONS

The initial site-preparation fire was the likely event resulting in long-lasting structural effects due to initial recruitment response that created a disparity in sapling-sized stems between plot types which persisted to the start of sampling in 2004. Significant assumptions have been used to explain the overall establishment and development difference of seedlings for initially burned and unburned plots. These assumptions are well-noted and should be considered when interpreting the relevance of these data for management practices with regard to the establishment of natural regeneration. There were notable differences between burned and unburned plots that remained nearly 30 years after initial plot implementation; however, there are management practices that may have contributed to differences in seedling densities between plot type over time, with the most notable factors being frequency of prescribed fire throughout established plots and seedling dynamics operating within plots of differing densities and structures. Operationally, these differences would be expected because each stand is not managed specifically for successful regeneration of longleaf pine but more for the overall functionality of the forest as a whole, with losses in regeneration being anticipated over time. Overall, there are continuing long-term effects of pre-seedfall prescribed fire on longleaf pine regeneration densities within plots, further supporting the importance of monitoring cone crops and properly timing prescribed fire application for successful recruitment and maintenance of longleaf pine regeneration within established pine stands.

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Ecophysiology

Moderator:

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IMPACTS OF COMPETITION CONTROL ON HARDWOOD MOISTURE AVAILABILITY AND STRESS TWO YEARS FOLLOWING REFORESTATION OF A RETIRED SOD FARM

Hal O. Liechty, Michael A. Blazier, and William Headlee

Extended abstract—The Maumelle River near Little Rock, AR is the primary source water for the largest water utility in Arkansas (Central Arkansas Water). In 2010, Central Arkansas Water purchased the Winrock Grass Farm along the Maumelle River and developed a plan to retire and reforest the sod fields on the property. The University of Arkansas and the LSU AgCenter worked with Central Arkansas Water to evaluate different competition control activities for establishing hardwood seedlings as part of the reforestation effort. We periodically monitored volumetric soil moisture (to a depth of 7.5 cm) in addition to pre-dawn leaf water (Ψ_{PD}) and stomatal conductance (g_w) of first flush foliage collected from water oak (*Quercus nigra* L.) and green ash (*Fraxinus pennsylvanica* Marsh.) seedlings during the second growing season following tree establishment in two fields. Measurements were made in three replicate plots located within three different competition control treatments (NCC, MCC, HCC) in each field. The NCC treatment had no competition control, the MCC (moderate competition control) treatment had competition control prior to planting but none following tree establishment, and the HCC (high competition control) treatment had some level of competition control prior to as well as the first year following planting but complete control competition during the second growing season (see table 1 footnotes). Ψ_{PD} was measured on one tree while g_w was measured on two trees from each species in each plot and measurement period. Volumetric soil moisture was measured within 1 m of the trees used for stomatal conductance. Competing vegetation (forb and grass primarily) biomass was determined on four 0.6-m² subplots within each treatment replicate on 6/3/15 and 8/27/15.

Table 1—Mean^a soil moisture (to a depth of 7.5 cm), leaf pre-dawn water potential, and stomatal conductance for each competition control treatment and measurement period in 2015 during the second growing season following tree establishment

Treatment ^b	5/12	6/9	7/8	8/4	8/25	9/15
-----Percent volumetric soil moisture (7.5 cm)-----						
NCC	32.4 a	14.0 b	27.7 a	6.0 b	17.4 b	11.9 b
MCC	34.2 a	18.9 a	31.1 a	8.3 ab	20.4 ab	13.8 b
HCC	34.1 a	20.9 a	29.9 a	10.9 a	21.8 a	17.2 a
-----Leaf pre-dawn water potential (-Mpa)-----						
NCC	0.64 b	0.31 a	0.38 b	0.44 b	0.56 c	0.49 c
MCC	0.50 a	0.31 a	0.23 a	0.32 b	0.38 b	0.35 b
HCC	0.49 a	0.26 a	0.16 a	0.19 a	0.22 a	0.19 a

^a Treatment means for a specific variable and sampling date with the same letter do not significantly differ ($p = 0.05$).

^b NCC = no competition control; trees planted 2/10/14. MCC = broadcast glyphosate (2.4 l/ha) 4/23/13; banded glyphosate (2.4 l/ha) + sulfometuron methyl (63.8 g/ha) 9/12/13; trees planted 2/10/14. HCC = broadcast glyphosate (2.4 l/ha) 4/23/13; trees planted 2/10/14; mowed competition 7/27/14; directed glyphosate 2.6 percent solution 4/30/15, 5/26/15, 6/30/15, 7/21/15, and 8/18/15.

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Biomass of the forbs and grasses, as expected, was consistently lower in the HCC than the other two treatments (fig. 1). During the June collection date, the biomass sampled in the MCC treatment was only 48 percent of that sampled in the NCC treatment indicating some residual reduction in competition from the herbicide applications applied prior to tree planting. Surface volumetric soil moisture (table 1) was significantly higher with the HCC treatment (22.5 percent) than the NCC treatment (18.4 percent) but not the MCC treatment (21.1 percent). Differences in soil moisture among treatments were greatest during the driest portion of the growing season (table 1). Volumetric soil moisture was significantly ($p = 0.05$) greater in the MCC than NCC treatments during June when differences in competing vegetation biomass were greatest. However, during August when biomass of the competing vegetation was similar in the two treatments, differences in volumetric soil moisture were not significant.

Mean Ψ_{PD} did not significantly ($p = 0.05$) differ between water oak (-0.35 Mpa) and green ash (-0.36 Mpa). The highest Ψ_{PD} for all three treatments occurred during the 5/12/16 measurement period (table 1). This may have been a result of flooding and saturated soil conditions that occurred just prior to this measurement period. In addition, due to a cold spring, leaf emergence was late, and foliage may have not completely matured prior to this initial Ψ_{PD} measurement. During the latter half of the growing season when drier soil conditions occurred (table 1) and competing vegetation biomass was at its highest levels (fig. 1), Ψ_{PD} significantly increased with the intensity of competition control. Ψ_{PD} during the six measurement periods averaged -0.49, -0.35, and -0.20 Mpa for the NCC, MCC, and HCC treatments, respectively. It was apparent that the competition control efforts that occurred prior to planting in the MCC treatment still had an impact on moisture status of the trees in this treatment during the second growing season. Ψ_{PD} appeared to be more sensitive to the impacts of competition control than the surface soil moisture content. Ψ_{PD} may better reflect the moisture availability of the soil column utilized by the trees than our shallow surface moisture measurements. In addition, the competition control treatments may have also impacted morphological or physiological characteristics of the trees so that they could better utilize the available soil moisture in these treatments during the second growing season. These type of changes could result in differences in the moisture status of trees in the MCC and NCC treatments with similar levels of soil moisture availability.

Regardless of the differences in Ψ_{PD} or volumetric soil moisture among treatments, gw was similar among the three competition control treatments during the growing season. Mean gw was significantly higher ($p = 0.05$) for green ash (0.30 mol m⁻² s⁻¹) than water oak (0.25 mol m⁻² s⁻¹). The similar stomatal conductance among treatments during the periods of low Ψ_{PD} suggests the trees growing in the NCC treatment showed no evidence of moisture stress compared to the other two treatments.

In summary, increased levels of competition control resulted in higher soil moisture and pre-dawn leaf water potential during the drier portion of the second growing season following tree establishment. The impact of competition control on tree water status was similar for water oak and green ash seedlings. There was no evidence trees growing without competition control experienced a level of water stress which resulted in reduced stomatal conductance. However, our results suggest that competition control treatments, prior to or following hardwood tree establishment in retired sod fields, might reduce seedling water stress if drought severity exceeded that experienced during the second growing season in our study.

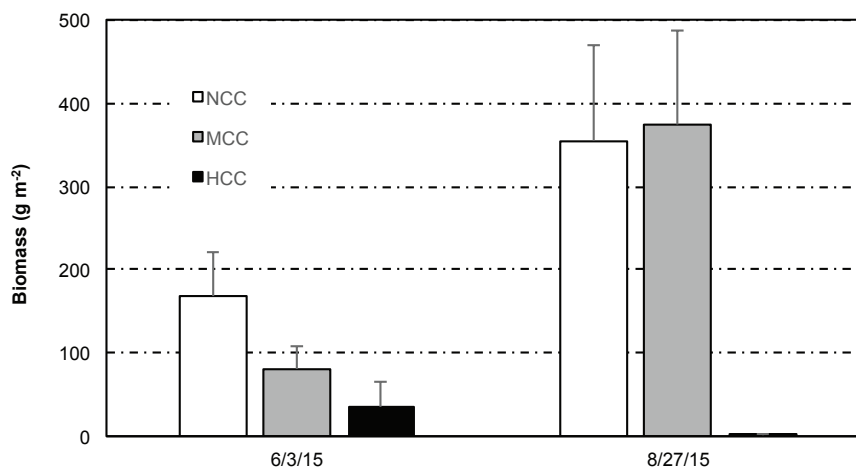


Figure 1—Dry competing vegetation biomass (95 percent confidence interval) collected on two dates in each competition control treatment.

EFFECTS OF COMPETITORS ON SOIL RESPIRATION, AND ITS COMPONENT PARTS, IN LOBLOLLY PINE (*PINUS TAEDA L.*) PLANTATIONS

Maggie Furrow, John Seiler, Brian Strahm, and Michael Aust

Extended abstract—Accurately predicting net ecosystem productivity (NEP) in southeastern loblolly pine plantations is both environmentally and economically important. The economic benefits of increasing NEP can be seen from two positions. The first occurs in sequestering more carbon in timber products at a faster rate and more efficiently and the profit associated with increased sales. Secondly, the possibility of widespread carbon markets in the future entices the timber industry to utilize means of increasing plantations' capacity as carbon sinks. From an environmental perspective, landowners concerned with rising atmospheric carbon dioxide concentrations and climate change can adapt management strategies that maximize carbon sequestration in their forest stands. To increase NEP, one needs to either increase the amount of carbon removed from the atmosphere through photosynthesis or to decrease the amount that is respired back into the atmosphere. It is this variable, respiration, particularly soil respiration, which this study will be examining. Soil respiration fluctuates with the rate of carbon dioxide (CO_2) being respired autotrophically by plants' roots and heterotrophically by living soil organisms. The presence of competitors in the understory—meaning any weeds, grasses, hardwoods, or volunteer pines—can affect the amount of total soil respiration (R_s) and also affect the partitioning of heterotrophic respiration (R_h) and autotrophic respiration (R_a) within total R_s . It is imperative that accurate numbers for these three variables, R_s , R_a , and R_h , be determined as even small changes to these significantly change NEP model outputs. Most NEP models operate around the formula $\text{NEP} = \text{NPP} - R_h$ which exhibits why the piecing apart of R_a and R_h is vital. When comparing previous studies in which R_h was measured we have seen that this variable seems to be affected by the presence of competitors. In one study, R_h was measured in an operational stand in which competitors were present. At this site, R_h accounted for 73 percent of total R_s (McElligott 2016). Another study, done in a competition-free plot showed that R_h made up only 84 percent of total R_s (Heim and others 2015). Figure 1, which was created by the PINEMAP modeling group, demonstrates that just an 11-percent change in R_h accounts for a twofold change in NEP over the span of one rotation.

In order to determine the effects of understory competitors on loblolly pine plantation R_s and its component parts, the following methodology is underway. Two treatment factors, age and competition control, were assigned to each of four blocks. Two loblolly pine plantation ages were selected, the first being young at 3–6 years of age, and the second being older at 18–21 years of age. Analysis of this factor will allow us to predict how soil respiration and the partitioning of its component parts change over time. The second factor, competition control, will allow us to examine how soil respiration varies between plots in which competitors have been left alone, a condition typically found in operational pine plantations, and plots in which all competition has been removed by mechanical and/or chemical means. Plots were established in June of 2016 in the Appomattox-Buckingham State Forest located in the Piedmont physiographic province of Virginia. Prior to installation, all sites had undergone similar management. To measure the effect of competitors, we established two plots, one in which competitors were left untouched and one in the same vicinity in which we created a competitor-free plot. To create these clean plots, we randomly selected a 3-m² area and mechanically and chemically removed all vegetation aside from planted loblolly pine trees. Trees or other plants that fell outside of the 3-m² area but were deemed to possibly have roots encroaching into the plot were also cut. These trees were chosen with the criterion that if twice the height of the tree fell within the plot the tree was cut. The herbicide Garlon® was applied to cut stumps, and glyphosate was applied to smaller plants in order to assure that no further photosynthate would be supplied to competitors' roots. Periodic removal of new competitors is being done to insure that any R_a data collected in these plots will come from only loblolly pine roots within the plot once sufficient time has passed to allow competitors' roots to run out of photosynthate. The current study will compare clean and weedy conditions while controlling for other variables. In order to separate R_a and R_h , we have installed two

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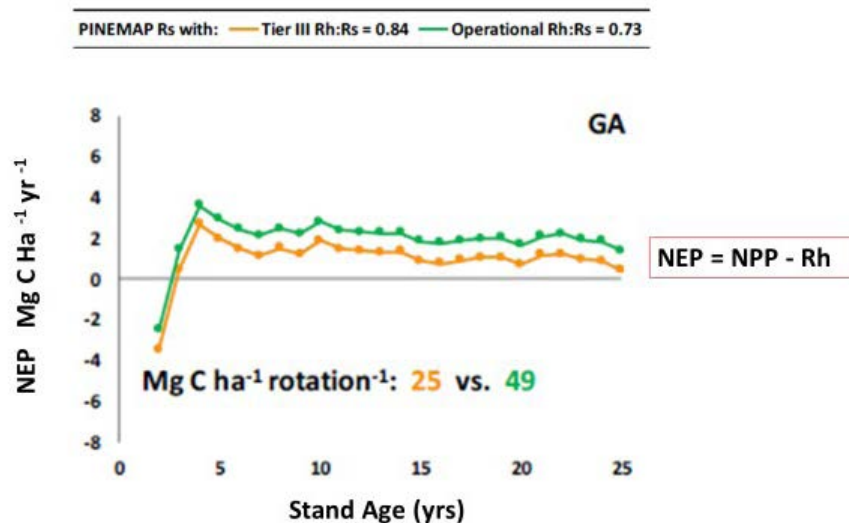


Figure 1—An image of 3-PG (Physiological Principles in Predicting Growth) NEP output showing a nearly doublefold increase in NEP per rotation when Rh:Rs is decreased from 84 percent to 73 percent. (Created at PINEMAP modeling meeting by Evan Brooks.)

root exclusion pipes to a depth of 35 cm in each plot. A study from 2014 determined that after a waiting period of about 90 days after installation, the R_a within the pipe shows a 95 percent decline (Heim and others 2015). After this waiting period, measurements of R_s taken directly over the root exclusion pipe will equal R_h as the influence of R_a will be negligible. In the area outside of the pipe the influence of roots is still present. Subtracting inside pipe measurements from outside measurements will estimate R_a . The same process will be carried out in weedy plots, allowing us to determine the influence of competitors on soil respiration and the partitioning of R_a and R_h in each scenario. Root coring will also be done at the end of the 90-day waiting period. Roots will be separated from soil, divided into pine and non-pine groups, measured, and analyzed for correlations with soil respiration patterns.

At the end of the study, we will examine what effects competitors' roots have on total soil respiration and R_a and R_h . We hypothesize that we will see higher overall R_s levels in weedy plots and decreased Rh:Rs ratios due to the presence of more roots in these sites. Our results will potentially allow NEP modelers to improve accuracy in predicting NEP of operational loblolly pine plantations.

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EFFECTS OF DROUGHT AND FERTILIZATION ON NEEDLE WATER POTENTIALS IN MIDROTATION VIRGINIA PIEDMONT LOBLOLLY PINE (*PINUS TAEDA* L.)

Edward Russell, John Seiler, Chris Maier, Quinn Thomas, and Erik Nilsen

Extended abstract—This study was carried out using an existing 2² factorial field experiment established in an upland, 14-year-old, mid-rotation, loblolly pine plantation located in the Appomattox-Buckingham State Forest of Virginia (latitude 37.443 N, longitude 78.664 W) in the Piedmont physiographic region. The soil at the site is fine-textured, shallow, and acidic, consisting mostly of 1:1 low CEC clay, overlain with an approximately 10-cm O-A layer. The dominant soil series present, as determined using the U.S. Department of Agriculture Natural Resources Conservation Service's Web Soil Survey, are Spears Mountain and Littlejoe (USDA NRCS, no date). From observation of an onsite soil pit, it is evident that there is also a hardpan layer roughly 1 m from the surface in some, if not all, areas. The site was initially planted in 2003 with a seed orchard mix at 1,200 stems ha⁻¹. Following 8 years of growth, natural mortality had reduced the planted density to ~790 stems ha⁻¹. This research site is part of a larger regional network of experimental sites, namely the *Pine Integrated Network: Education, Mitigation, and Adaptation Project (PINEMAP; <http://pinemap.org/>)*. The major goal of this study was to elucidate the effects of fertilization and reduction of available moisture on pre-dawn and midday needle water potentials. It was expected that reduced water availability would reduce needle water potentials in a seasonally dependent manner, but it was unclear the effect that fertilization would have and if there would be any interaction between fertilization and throughfall reduction. It has been suggested by some researchers that fertilization may increase drought stress (Ward and others 2015), so that was tested with respect to needle water potentials.

The layout of the experiment is a completely randomized block design with four blocks. The first treatment factor is fertilization: none versus one-time application of 224 kg N ha⁻¹, 27 kg P ha⁻¹, 52 kg K ha⁻¹, and 1.12 kg ha⁻¹ micronutrient mix (6 percent S, 5 percent Bn, 2 percent Cu, 6 percent Mn, and 5 percent Zn), all applied in April 2012. The second treatment factor is throughfall reduction: none versus ~30 percent removed, concurrently initiated in April 2012 with the fertilization treatments after understory vegetation was removed mechanically and chemically with glyphosate, imazapyr, and metsulfuron. The throughfall reduction was accomplished in eight of the experimental plots by building troughs roughly 1.1 m high made of wood and polyethylene sheeting, which covered ~30 percent of the ground area and carried water away and downhill from the plots. Each square plot is approximately 0.356 ha, including a perimeter buffer strip, which leaves a square internal measurement plot of 0.141 ha. The total number of experimental units is 16, as there are four treatment combinations, with one replicate per block.

During the summer of 2015, values of needle water potential (ψ_L) were collected using a Scholander apparatus by applying the standard technique where excised shoots are subjected to elevated pressure until xylem sap is exuded from the severed tracheids of the stem in question (Scholander and others 1965). Needle water potential values were determined from two shoots representing subsamples from each experimental unit, for a total of eight subsamples from all four blocked replications of the individual treatment combinations per time point. The collection of pre-dawn and midday ψ_L data took place both early and late in the main growing season resulting in 128 individual observations, or 64 differential observations representing the change in ψ_L ($\Delta\psi_L$) for a given day. The summer rainfall was below the 30-year annual average in 2015, when the data were collected. The data were analyzed using a mixed linear modeling approach in the R language for statistical computing (Bates and others 2015).

Fertilizer application did not significantly lower ψ_L at pre-dawn or at midday (fig. 1). Throughfall exclusion did significantly lower needle water potentials, particularly under late season water limitation. The same pattern of significance was seen with the $\Delta\psi_L$ data. This result was consistent when characterized with a simple random

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Needle Water Potential

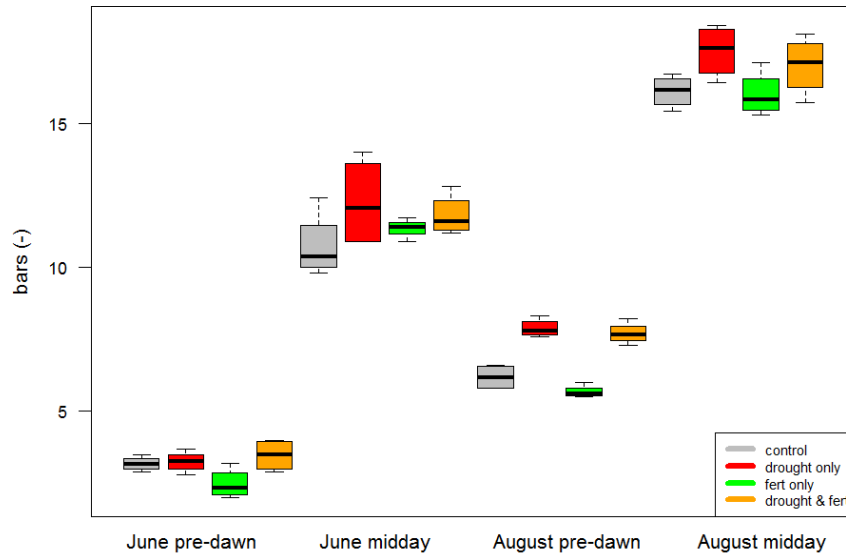


Figure 1—A boxplot showing needle water potential values in units of negative bars for June and August pre-dawn and midday samples. Here, drought refers to the 30 percent throughfall exclusion treatment and fert is shorthand for the fertilizer treatment. The whiskers in the plot extend to the most extreme data point that falls within 1.5 times the interquartile range of the respective data.

effects model where the subsampling in space and time was averaged out and in a more complex hierarchical mixed model where appropriate effects and error terms for the spatial and temporal subsampling were included in the model. As expected, in addition to throughfall exclusion the interactions between time of day and month and between month and throughfall exclusion were statistically significant apropos ψ_L values. The use of needle water potential as an indicator for drought stress has been employed by numerous researchers investigating the physiology and water relations of loblolly pine (Ginn and others 1991, Maggard and others 2016, Tang and others 1999, Tang and others 2004, Will and Teskey 1997). As such, with respect to needle water potentials, it would seem that infrequent fertilizer applications did not result in significant water stress in the trees observed during the course of this study. Furthermore, these results support the assertion that fertilizer application is not likely to significantly endanger plantation trees in the Virginia Piedmont in the near future.

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LIGHT USE EFFICIENCY OF LOBLOLLY PINE (*PINUS TAEDA* L.) IN THE UNITED STATES AND BRAZIL

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Extended abstract—Although the productivity of loblolly pine (*Pinus taeda* L.) has increased over the last several decades due to better silviculture and genetic improvement (Fox and others 2007), the stand density index (SDI) in southeastern U.S. loblolly pine trials has not increased. However, SDI of loblolly pine grown in Hawaii and Brazil is greater than that of loblolly pine grown in the Southeastern United States (Albaugh and others 2015). Loblolly stands in South American countries also maintain considerably higher leaf area index (LAI) (Albaugh and others 2010, Albaugh and others 2015), resulting in substantially lower light at the base of the living crown. This greater leaf area may be related to greater SDI and growth found in Brazil (Yu and others 1999). We hypothesized that needles at the base of the crown in trees in Brazil are more shade tolerant or more efficient in using light than needles at the base of the crown in trees in the United States, in order to maintain the higher leaf area.

To investigate this, we carried out a study in an existing split-split plot experiment with four blocks at Virginia Piedmont (36° 38' 35.32"N, 80° 09' 18.84"W), three blocks at North Carolina Coastal Plain (34° 49' 49.63"N, 78° 35' 18.52"W), and three blocks at Santa Catarina State, Brazil (26° 11' 21.68"S, 49° 29' 48.78"W). Both of the North American installations were established in 2009, and the Brazil site was established in 2011 (Vickers and others 2012). This experiment was part of the Regionwide 20 trial of Forest Productivity Cooperative (<http://forestproductivitycoop.net/>), with silviculture as the whole plot factor (operational vs. intensive), genotype as the split-plot factor (four clonal varieties, one open-pollinated family, and one controlled mass-pollinated family), and three initial stand densities (617, 1,235, and 1,852 trees/ha). We sampled the four clonal varieties (C1, C2, C3, and C4) with intensive silviculture and at a density of 1,852 trees/ha. Two trees were randomly selected from each plot as subsamples. Measurements were taken in January/February 2016 for the Brazil site and August 2016 for the two U.S. sites. Small branches from the lowest living crown were cut off and immediately put into a water bottle for measurement. Light response curves (Loach 1967) were generated on needles for the four varieties to calculate light compensation points and apparent quantum efficiency, indicators for shade tolerance and light use efficiency, respectively. Carbon dioxide (CO₂) assimilation was measured using a portable infrared gas analyzer (LI-6400 system, LI-COR Inc., Lincoln, NE, USA) at light levels of 0, 75, 150, 250, 400, 600, 1000, 1500, and 2000 $\mu\text{mol m}^{-2} \text{s}^{-1}$. Reference CO₂ and block temperature were maintained at 385 $\mu\text{mol m}^{-2} \text{s}^{-1}$ and 25 °C, respectively, for peak measurements. After light curve measurements, chlorophyll content (Chl) of the needles were determined with a CCM-300 chlorophyll meter (Opti-Sciences, Hudson, NH, USA). The two subsamples were averaged for statistical analysis. Two-way ANOVA was used to determine the effects of location and genotype on leaf physiology. A student t-test was used for multiple comparison with $\alpha = 0.1$. Statistical analysis was performed with JMP Pro 10 (SAS Institute, Cary, NC).

Neither location nor genetics affected the apparent quantum efficiency (table 1). The light compensation point (LCP) of needles in Brazil was significantly lower than in Virginia ($p < 0.0001$). Although insignificant, the mean LCP in Brazil was still lower than in North Carolina. Lower LCP indicated better shade tolerance (Valladares and Niinemets 2008). The higher LAI in Brazil was partially attributable to better shade tolerance of needles at the base of the living crown but was not related to the light use efficiency. Similar to other plants growing at lower light levels (Loach 1967), chlorophyll content of the base-crown needles was doubled in Brazil compared to

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Virginia and North Carolina sites. However, the effect of location interacted with genetics for LCP ($p = 0.0639$) and Chl ($p = 0.0843$, Table 1). Clone C1 functioned differently from other clones when grown in Brazil. The chlorophyll content of C1 did not vary among sites while for other clones, the chlorophyll content was doubled or more than doubled in Brazil. Also, the LCP of C1 was significantly lower than other clones ($p = 0.0080$) in Brazil but at the two U.S. sites, no significant difference was found among clones. The significant interaction between genetics and location for LCP and Chl implied that different clones vary in their adaptation ability when planted abroad.

Table 1—Statistics (p -values) for leaf physiological parameters as affected by genotype and location for trees with the same genotype grown in Virginia and North Carolina in the United States and in Brazil

	Genotype	Location	Genotype $\times$ location
Light compensation points ($\mu\text{mol m}^{-2} \text{s}^{-1}$)	0.0691	<0.0001	0.0639
Apparent quantum efficiency	0.2343	0.3099	0.5208
Chlorophyll content (mg m^{-2})	0.1298	<0.0001	0.0843

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Fire

Moderator:

Rebecca Kidd

Stephen F. Austin State University

CAUSES AND COSTS OF BOLE WOUNDS IN HARDWOODS—A SYNOPSIS OF THE LITERATURE

Janice K. Wiedenbeck

Abstract—Silvicultural practices are known to affect the initiation and development of wounds and wound-related defects. Research on partial-cutting-related wounds has focused on residual stand damage, while wound occurrence associated with prescribed fire has been studied much less. This paper reviews published results of occurrence rates, causes, and costs associated with residual tree wounding owing to these two important practices in hardwood forest management.

INTRODUCTION

Tree wounds are caused by a wide range of natural events and anthropogenic activities and occur in tree crowns, on branches, on the bole, or on roots. Existing information mostly relates to bole wounds as they are more visible, accessible, and considered more economically important.

Wind, ice, heavy snow loads, cold temperatures, lightening, wildfire, sunscald, drought, and flooding all contribute to wounding (Seifert and Woeste 2005). Animals also can cause bole wounds and root compaction. Human-caused tree wounding can be accidental (e.g., equipment contacting trees, careless fire and herbicide application) or purposeful (e.g., farm fence construction, maple syrup tapping, branch pruning, and silvicultural activities).

A review of residual stand damage caused by harvest operations (Vasiliauskas 2001) was predominantly softwood focused. Nyland (1986) reviewed pre-1986 studies of logging damage following thinning of even-aged hardwood stands. Dey's (1994) review of logging impacts concentrated on the damage effects to stand structure and summarizes how damage to trees may affect future tree vigor and quality. None of these papers address the residual stand impacts of silvicultural use of fire.

This review focuses on wound occurrence in residual trees in eastern hardwood forests from prescribed fire and forest operations, emphasizing tree quality and value impacts. Tree response to wounding is a critical aspect of tree and wood quality development included in this review. Tree and stand value impacts attributed to

wounding and wound-associated losses in tree vigor and condition are highlighted for stands managed for timber harvest and income.

WOUND OCCURRENCE RATES IN TEMPERATE HARDWOODS

Wound Occurrence, Decay Association, and Likely Causes

An early survey of the amounts and causes of wounding in hardwoods found 47 percent of the trees harvested of eight commercial species in the Appalachian region contained fire wounds. However, harvested trees were "the best trees of the most desirable species" (Hepting and Hedgecock 1937). The critical relationship between basal wounds and decay was elucidated in this paper—only 6 percent of harvested trees lacking basal scars had decay at stump height, compared to 67 percent for trees with wounds.

The entry path of decay fungi in hardwoods was observed in several more recent studies of landscape level wound and decay occurrence rates in the central hardwood region (Berry 1969, 1977, Berry and Beaton 1972a, b). Fire scar associated decay comprised 24 to 48 percent of the infections and accounted for 32 to 63 percent of the affected merchantable volume in these studies. The proportion of decayed trees for which logging damage was the entry path for infection was less certain. Also in the 1970s, bottomland hardwood quality surveys showed 40 percent of harvested trees contained butt rot, with 65 percent of infections attributed to fire scars, indicating fire wounds negatively impacted the quality of 26 percent of trees (McCracken 1977). Most of the non-fire rot infections were attributed to harvesting.

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Berry (1969) and Berry and Beaton (1972a) examined the reliability of visible defect indicators on oaks (*Quercus* spp.) for signifying associated decay—this is key information for timber stand management and sale valuation. In both studies, over 90 percent of open fire scars were associated with underlying decay. Closed fire scar results varied, with 64 percent associated with underlying decay in the Central Hardwood Region (Berry and Beaton 1972a) compared to only 35 percent in Kentucky (Berry 1969). Damaged tops were indicators of wood decay in 50 to 60 percent of trees. Unsound branch stubs indicated decay in 16 to 31 percent of trees. Mechanically injured trees included decay in 10 (Berry 1969) to 26 percent of trees (Berry and Beaton 1972a). These studies were conducted in even-aged, undisturbed stands (except for fire), so damage and decay from forest operations are not represented.

Evidence of tree quality issues affecting tree merchantability in the Northern United States from FIA plots (Morin and others 2016) indicated that 76 percent of trees ≥ 5 inches diameter at breast height (d.b.h.) were free of significant damage and decay was present in only 16 percent of trees. Less than 3 percent of trees had significant recent logging damage, but earlier logging injuries that subsequently appeared as wounds with decay were indistinguishable from other types of decay. The percentage of hardwood trees with major decay ranged from 35 percent for *Fagus* spp. to 10 percent for *Ulmus* spp., with *Quercus* spp. and *Acer* spp. at 11 and 22 percent, respectively. (Morin and others 2016).

Forest Operations Caused Wounds

Over the past half-century, hardwood silviculture to affect stand structure has become widespread. Prescriptions often involve partial cuts to improve the residual stand, promote the regeneration of desired species, generate income while preserving future income, and create structural diversity for wildlife. Although no studies similar to those cited above (sampling tree damage and decay across wide regions) have been conducted since the mid-1970s, scientists have conducted specific studies looking at the impacts of different types of harvest treatments on residual stand attributes, including wounding.

Wounding rates on residual hardwood stems caused by harvesting operations have been reported as low as 8 percent (Dwyer and others 2004) and as high as 68 percent (Meadows 1993); study designs are as varied as these results. Table 1 summarizes 13 unique studies on residual tree bole damage from forest operations. Other forms of damage are not included because several studies did not evaluate crown and root damage, while those that did used different damage classification systems.

Two studies sampled many logging operations across ownerships: one (18 sites) was conducted in northern hardwoods (Cline and others 1991), and one (101 sites) in central Appalachian forests (Hassler and Grushecky 1999). Bole wound occurrence rates after logging for a range of sites and management prescriptions was about 13 percent in both studies. By contrast, for the other 12 studies in table 1, the overall average wound occurrence rate associated with logging activities was about 18 percent higher.

Factors Affecting Wounding Rates and Severity in Forest Operations

Pre- and post-harvest basal area and wounding rates—Residual basal area (RBA) has been examined as a factor affecting the damage levels in several post-harvest residual stand studies (Clatterbuck 2006, Fajvan and others 2002, Lamson and Miller 1982, Miller and others 1984, Nichols and others 1994, Nyland 1986). Lower RBA treatments resulted in higher levels of residual stand damage in two studies (table 1) (Lamson and Miller 1982, Miller and others 1984). Clatterbuck's (2006) study indicated lower RBA resulted in less bole damage but logging operations were judged to be sub-par. Nichols and others (1994) returned mixed results, but found a direct relationship between initial stand basal area and residual tree damage levels. Nyland (1986) offered "The incidence of damage seems to increase with the intensity of thinning operation." Hence, a likely predictor of the level of residual stand damage is the ratio of initial stand basal area and residual stand basal area.

Wounding associated with felling versus skidding—

The distance of residual stems from skid trails was a significant factor in modeling the probability of stem damage in a northern hardwood damage assessment by Nichols and others (1994) comparing manual felling and skidding with a rubber-tired skidder to a mechanical harvest operation using a feller-buncher. The probability of wounding trees located adjacent to skid trails averaged 40 and 60 percent, respectively, for manual and mechanical operations. However, damage to stems caused by feller-buncher skidding declined rapidly as the distance from the skid trails increased with 3 percent of residual trees located 18 feet from skid trails damaged (compared to 15 percent for the manual operation at the same distance; Nichols and others 1994). An evaluation of harvest operations in Missouri (Bruhn and others 2002) included both even- and uneven-aged treatments and showed 74 percent of the injured residual trees (≥ 10 inches d.b.h.) were located within 6 feet of skid trails and 50 percent of trees proximal to primary skid trails suffered bole injuries. A retrospective assessment of 33 non-controlled harvesting operations in West Virginia showed that 62 percent of the damage occurred during skidding (Egan and Baumgras 2003).

Table 1—Forest operations-caused bole wounding of residual hardwood trees

Study authors and year	Site(s)	Forest type(s)	Number of trees	Treatment(s)	Percent bole wounded			Percent with wounds ≥ 100 square inches
					Saw timber	All sizes	Pole-timber	
Dwyer and others (2004)	MO; 9 sites 2 treatments	Oak-hickory and oak-pine	8,901	Uneven-aged: $q=1.5$	---	10	—	—
				Even-aged: intermediate treatment	—	7	—	—
Cline and others (1991)	ME, NH, VT; 18 sites	Northern hardwoods	—	Various as dictated by landowner	—	14	—	—
Hassler and others (1999)	WV; 101 sites	Various throughout State	—	Various as dictated by landowner	9	—	11	4
Clatterbuck (2006)	TN; 1 site, 3 treatments	Oak-hickory	528	12.5% RBA	—	35	—	—
				25% RBA	—	65	—	—
				50% RBA	—	66	—	—
Fajvan and others (2002)	WV; 1 site, 3 treatments	Appalachian hardwood	1,380	12-inch DL	13	—	14	26
				16-inch DL	16	—	19	16
				Shelterwood	22	—	14	16
Johnson and others (1998)	WV; 20 sites	Beech/cherry/maple (11 sites); App. hardwoods (5); oaks (4)	768	Shelterwood:	—	—	—	28
				Mar-Jun	—	58	—	23
				July-Oct	—	46	—	14
				Nov-Feb	—	34	—	—
Kelly (1983)	VT; 4 sites	Northern hardwoods	—	Shelterwood Thinning	52	—	31	10
					74	—	43	12
Lamson and Miller (1982)	WV; 1 site	Cherry-maple	9,350	45% RBA	—	50	—	14
				60% RBA	—	30	—	7
				75% RBA	—	22	—	5
Lamson and others (1985)	WV; 1 site 4 stands	Appalachian hardwood	1,539	q -value=1.3	11	—	13	2
Meadows (1993)	MS; 1 site	Bottomland hardwoods	—	Improvement cut w/ 75% RBA	42	—	37	36
Nichols and others (1994)	ME; 1 site, 4 treatments	Beech-sugar maple	1,394	48% RBA -skidder	43	—	44	—
				80% RBA -skidder	19	—	29	—
				33% RBA feller-bunch	16	—	18	—
				43% RBA feller-bunch	28	—	37	—
Nyland and Gabriel (1971)	NY	Northern hardwoods	—	Uneven-aged thinning	—	16	—	15
Olson and others (2015)	MO; 1 site 20 stands, 2 treatments	Bottomland hardwoods	1,420	CC w/ residuals	—	7	—	—
				BAR: 20-30 ft ² /a	—	9	—	—
Ostrofsky and others (1986)	ME; 2 sites	Paper birch, beech-oak	—	Heavy thin/with whole-tree harvest	—	29	—	—

App.=Appalachian; RBA=residual basal area; CC=clearcut; BAR=basal area removed; DL=diameter-limit cut.

Damage caused by tree felling was assessed by Miller and others (1984) for thinning treatments in central Appalachian mixed oak-cove stands. Overall, between 26 and 34 percent of the residual stems were bent, leaning, or destroyed by tree felling; however, <3 percent of the residual crop trees (≥ 11.0 inches d.b.h.) were affected. Between 13 and 25 percent of residual stems had bole injuries and 21 to 30 percent had broken tops—for crop trees the incidence rates were 6 and 2 percent, respectively.

Season of harvest—Residual trees are at greater risk of suffering harvesting-induced damage and future degradation with spring and early summer harvesting; the percentages of trees with sapwood exposed wounds and with large wounds are higher (Johnson and others 1998). Wound occurrence rates on residuals were 35 and 18 percent after summer and winter harvest operations, respectively, in a Maine study (Nichols and others 1994). The wound width to tree circumference ratio indicated larger wounds resulting from summer harvesting. A study in West Virginia evidences this effect with a large difference in the proportion of trees wounded in spring compared to winter harvesting (58 vs. 34 percent) (Johnson and others 1998). Larger wounds (>100 in²) were significantly more prevalent after spring harvesting (table 1).

Wound location and size—Bole wound size is an important determinant of the probability a tree will recover without significant discoloration and decay. Bole wounds with abraded bark “that remove bark from at least 150 in² of trunk surface have a 50 percent chance of developing decay within 10 years” (Nyland 1986). Hesterberg (1957) determined wounds that exposed underlying wood area of ≥ 100 in² were severe. Most studies have used this Hesterberg measure to distinguish the proportion of bole wounds with a significant probability of decay development. As summarized in Table 1, the overall mean percentage of residual trees having wounds ≥ 100 in² is 16 percent.

Prescribed Fire Caused Wounds

Fire effects studies conducted in the eastern hardwood forests of North America have not focused on the quality and value impacts of prescribed fire on residual sawtimber trees. Instead, the studies focused on the effects of prescribed fire on oak regeneration success, the influence of prescribed fire on tree mortality, and the impacts of prescribed fire on residual stand structure and species dynamics. However, wildfire studies prior to 1960 almost exclusively focused on impacts on overstory trees (Brose 2014).

Fire-caused wounds are difficult to tally soon after occurrence. What can be tallied in the first months after fire is the dimensions of the bark burn/char/scorch. To

determine wound occurrence requires assessment 2 or more years after the fire. Alternatively, wounds have been assessed through time-intense bole dissection of trees already showing evidence of wounding (Dujesiefken and others 2005, Smith and Sutherland 2006, Smith and others 2001). These studies have furthered our understanding of tree response to heat caused injury, wound closure rates, decay occurrence, and fire history.

Table 2 summarizes the prescribed fire studies conducted in the eastern hardwood region that include estimation of bole damage occurrence. Empty cells in table 2 indicate the variability among these studies. Differences in the way damage/scarring is defined and differences in minimum tree sizes measured also exist. Differences in fire temperatures, stand basal areas, topography, etc. are not captured by this summary. Given these caveats, the overall mean proportion of trees damaged in the seven studies included in table 2 is 41 percent with damage proportions ranging from 21 to 66 percent.

Wounding rates and severity associated with prescribed fires—wounding rates and dimensions (table 2) are affected by many factors; fire intensity and duration are among the most important. Unfortunately, wound sizes are missing for most prescribed fire studies that provide wounding rates. Differences among species in overstory tree susceptibility to wounding caused by fire are largely attributable to the level of protection offered by bark. Bark thickness is the key protective factor, but heat transmission properties can vary among species (Hengst and Dawson 1994, Spalt and Reifsnyder 1962) as can bark regrowth rates. The most important component of bark thickness is the thickness of the outer, corky bark portion (Stickel 1941). The low density of the outer bark provides most of the insulation from fire (Hare 1961). Several North American hardwoods have thick bark but the bark has deep longitudinal fissures between bark ridges. Chestnut oak (*Quercus prinus*) is one such species and has been found to develop fire scars under bark fissures where the bark is thinner (Smith and Sutherland 2001, Sutherland and Smith 2000).

The rate at which bark thickens as a tree grows varies by species, with some species being vulnerable to injury from fire much longer than others. The rate of bark thickening as a function of d.b.h. was modeled for 16 hardwood species by Hengst and Dawson (1994). Overall, their results indicated that upland hardwood species produce thicker bark at a younger age than do lowland species.

Species differences in physiological response to wounding are important in limiting degradation over time. How effectively a given species compartmentalizes injuries to limit deleterious effects has been explained

Table 2—Prescribed fire caused wounding of hardwood stems

Study authors and year	Location	Forest type	No. of trees	Prescribed fire treatment	Mortality proportion		Damaged proportion		Mean scar width (inches) or circum (%)
					Over-story	All	Over-story	All	
Wendel and Smith (1986)	Central Appalachians	Oak-hickory	~2,415	Spring	5%	—	66%	—	—
Brose and Van Lear (1999)	VA	Piedmont: oak-hickory	733	Spring	19%	—	31%	—	—
				Summer	0%	—	21%	—	—
				Winter	0%	—	22%	—	—
Paulsell (1957)	MO	Ozark uplands: oak-hickory	1337	Annual burn	5%	—	—	27%	—
				Periodic burn	3%	—	—	34%	—
Guyette and Stambaugh (2004)	TN	Upland hardwoods: post oak	—	Fall 1997	—	—	35%	—	Circum.=10
				Spring 2000	—	—	62%	—	Circum.=12
				Spring 2003	—	—	65%	—	Circum.= 8
Stevenson (2007)	MO	Ozark uplands: oak-hickory-shortleaf pine	3,754	Southwest facing slopes	—	—	—	64%	r. oak: 11 bl. oak: 11 wh. oak: 9
				Northeast facing slopes	—	—	—	52%	post oak: 8 hickory: 7
Ward and Brose (2006)	CT	Eastern oak	3,476	7 prescribed fires	—	—	Oak: 25% Non-oak: 28%	—	—
Wiedenbeck and others (2017)	WV	Mesic upland hardwoods	1,777	Shelterwood-burn - 2 spring fires	—	9%	—	33%	—

Circum.=circumference.

by Hepting (1935), Shigo (1966, 1984), Smith and Sutherland (2001, 2006), and Smith and others (2001). Quick wound closure after injury is a related factor that influences the vulnerability of trees to further wounding; radial growth rates can differ substantially among species. For forest managers using prescribed fires to promote oak regeneration, the application of more than one prescribed fire to affect desired outcomes means that wound closure rates for mid- and overstory trees are particularly meaningful.

One of the first North American studies of eastern hardwood tree species response to fire was conducted by Nelson and others (1933). Their study of fire wounds after a severe wildfire indicated important species differences in wound size. They found that yellow-poplar was distinctly more resistant to wounding by fire than scarlet oak (*Quercus coccinea*) when considering larger diameter trees with other species intermediate in resistance.

Efforts to estimate the impact of a fire on residual tree mortality and quality soon after fire occurrence require that the relationships between bark discoloration and future tree outcomes are identified since it can take 2 years before wound severity and mortality can be detected. Loomis (1973) modeled mortality and wound size based on bark scorch measurements in a broad-scale study of trees in Missouri, Pennsylvania, and West Virginia and predicted the height and width of wounds for red oak (*Quercus rubra*), black oak (*Quercus velutina*), ash (*Fraxinus* spp.), hickory (*Carya* spp.), and scarlet oak would be 90 and 60 percent, respectively, of the height and width of bark blackening measured after a dormant season fire. For white oak (*Quercus alba*), post oak (*Quercus stellata*), and chestnut oak (*Quercus montana*), the predicted height of the wound that would develop was 70 percent of the height of the blackened bark. Species differences in the probability of being injured by fire were explored in a Missouri Ozark forest by Stevenson and others (2008). They

concluded the species-based risk of wounding from fire was: “red oaks > black oak = white oak > post oak = hickories > shortleaf pine” (*Pinus echinata*). These two studies provide evidence that species in the white oak subgenus are more fire tolerant than those in the red oak subgenus.

CONSIDERING LOGGING WOUNDS VERSUS FIRE WOUNDS

What is less known is what, if any differences there may be in tissue damage and wound response for trees subjected to fire wounding compared to abrasion-type wounds. Thicker bark is acknowledged to be more protective than thinner bark. Most studies of this protective capacity have been studies of how bark insulates trees from the heat of fires. That thicker bark also provides greater protection for trees from other types of injuries is logical. Wood and bark strength properties increase with increasing specific gravity, therefore, the outer bark which has lower specific gravity, provides insulation from heat injury but offers less protection from certain types of applied stress.

Smith and Sutherland (1999) dissected tree scars on the lower bole of oaks and concluded that the appearance of historical wounds is not distinctly different for trees injured by fire compared to other wounding agents. They also determined that the presence of charcoal in a wound is not definitive of the wound having been caused by fire. Shigo (1966) observed that an important difference between fire-caused wounds and logging wounds is that fire caused wounds are less variable, typically occurring at the base of the butt log with decay developing upward from the wound.

Closure of Wounds

Wound closure, the process by which exposed xylem is gradually covered by new wood and bark, reestablishes the bark covering of wounded regions to provide the tree needed protection from insects, bacteria, and fungi. Species with faster radial growth are able to close wounds more quickly than slow-growth species. Wounds that are wider require longer to close than narrower wounds. For example, bole wounds in sugar maple have a 50 percent chance of developing decay after 10 years if the wound size is >150 in² but an 80 percent chance if the wound exceeds 250 in² (Hesterberg 1957).

In-woods observation of wound closure rates based on remeasurement of injured stems were conducted by Smith and others (1994) and Jensen and Kabrick (2014). Smith and others (1994) collected remeasurement data 5 and 10 years after wound initiation for bole wounds on 70 to 80 year-old residual trees. For northern red oak, white oak, and yellow-poplar (*Liriodendron tulipifera*), about half of the wounds with initial exposed wood areas ≤100 in² had closed after 5 years. No wound closure was recorded for larger wounds. After 10 years, 88

percent of the smaller wounds but only 19 percent of larger wounds were closed (Smith and others 1994). The wound closure results reported by Jensen and Kabrick (2014) were based on a single resurvey of wounded trees 13 years after harvest operations and included trees ≥4.5 inches d.b.h. Overall, 76 percent of the wounded trees no longer had open bole wounds. The Jensen and Kabrick (2014) sample size permitted species-based comparisons to be made: white oak (92 percent), scarlet oak (82 percent), black oak (58 percent), hickories (36 percent), and shortleaf pine (17 percent) were the most to least successful in closing wounds. For all tree species and wound sizes combined in the Smith and others study, 59 percent of bole wounds were closed after 10 years compared to 76 percent closed after 13 years in the Jensen and Kabrick (2014) study.

Quality, Grade, and Value Impacts Associated With Wounds

A foundational study on the impact of harvest-based wounds on quality and log and lumber grade recovery was conducted by Hesterberg (1957) on sugar maple (*Acer saccharum*). Logs with wounds suffered a mean value loss of 11 percent when evaluated 10 years after wounding, with 9 percent of the logs reduced in grade (Hesterberg 1957). However, the mean value loss of the lumber sawn from these logs was only 3 percent. In another study of sugar maple (Ohman 1970), value losses associated with wounds from harvest operations were lower after 10 years, 1 percent for logs and 3 percent for lumber. Ohman (1970) evaluated yellow birch (*Betula alleghaniensis*) recovery as well and reported 12 and 4 percent log and lumber value losses, respectively.

Estimated log grade impacts for butt logs of residual trees were higher in a study of basal area retention differences in the Southern Appalachians (Clatterbuck 2006) with 45 percent of the bole damaged trees judged to have damage levels sufficient to cause a drop in log grade. Johnson and others (1998) study of shelterwood with reserves stands in the Appalachians evaluated butt log grade changes in residual trees attributable to logging damage with similar results to Hesterberg (1957) and Ohman (1970). About 2 percent of the butt logs lost grade 2 to 5 years after harvesting. With assessments conducted only 2 to 5 years after wound occurrence, further degradation owing to decay development associated with some wounds could be expected.

A study of wildfire impacts on product value based on a range of tree sizes and geographic locations (thus fire intensities) in oak-hickory forests determined value and volume losses were related to wound height, width, age, and tree d.b.h. (Loomis 1973, 1974) with R² levels for log and lumber value loss equations of 0.54 and 0.45, respectively. Loomis (1989) composed a look-up table for estimated volume loss due to basal fire wounds based on his earlier research.

Wildfire caused stumpage value losses for hardwood stands informs the discussion of prescribed fire use in oak-hickory forest types. Reeves and Stringer (2011) determined, based on 10 sets of matched stands with one burned and the other unburned, that the timber value loss from wildfires ranged from 5 to 65 percent with an average loss of 47 percent. Of the estimated loss, 28 percent was attributed to cull volume while 72 percent was related to structural changes including tree mortality and changes in species distribution and size classes of timber (Reeves and Stringer 2011). A coarser approach was taken by Wood (2010) in assessing value loss to sawtimber in West Virginia stands that had undergone from 0 to 6 wildfires. Value decline increased with increasing fire exposure ranging from a 10 percent decline in sawtimber value for stands subject to one wildfire to 53 percent for stands following six wildfires (Wood 2010). The corresponding declines in sawtimber volumes determined by Wood ranged from 6 percent to 55 percent. The stand value effects determined by Reeves and Stringer (2011) and Wood (2010) align well with each other with value losses ranging from 5 percent to 65 percent in the former study and from 6 percent to 55 percent in the latter.

Prescribed fire impacts on the quality and value of residual trees have received little attention. Studies have been conducted on oaks in Missouri (Knapp and others 2017, Marschall and others 2014, Stambaugh and Guyette 2008) and at a single site in West Virginia associated with shelterwood-burn study (Wiedenbeck and Schuler 2014, Wiedenbeck and others 2017). This dearth of information, together with the increased use of prescribed fire in eastern hardwoods, has led to new research activities focused on this subject (Stanis and Saunders 2017).

Analysis of 41 basal tree sections containing fire scars caused by a prescribed fire that occurred 6 years earlier provided averages for scar size (41 cm^2), woundwood volume (24 cm^3), decay volume (27 cm^3) and discolored wood volume (27 cm^3) for the sample of white, black, and scarlet oak stems (Stambaugh and Guyette 2008). Twenty percent of the butt logs with fire scars in this study dropped in grade due to the fire injury. Tree and lumber volume and value loss estimates for red, black, and scarlet oak trees affected by fire were measured by Marschall and others (2014). The average lumber volume and value losses per tree were 4 and 10 percent, respectively, with many boards dropping in lumber grade but not in volume.

Isolating the wood quality changes attributable to two prescribed fires that were conducted 5 and 8 years prior to tree harvest indicated only minor effects of prescribed fire on lumber quality with 16, 13, 12, and 7 percent of the lumber recovered showing any sign

of fire damage for red maple (*Acer rubrum*), red oak, white oak, and yellow-poplar, respectively (Wiedenbeck and Schuler 2014). Minor indicators included mineral stain (red maple), surface checking (red and white oak), and incipient decay. Recent, low intensity fire exposure appears to have very little impact on the quality and value of hardwood lumber recovered from affected stems based on limited research.

The effects of prescribed fire on residual stand value rather than tree value has been given less attention. The one known study was based on Marschall and others' (2014) model. Knapp and others (2017) estimated stand value loss for areas that had a history of prescribed fires to be <3 percent. However, total stumpage value of these stands was substantially lower than the stumpage value of unburned stands—approximately 30 percent (Knapp and others 2017). This higher figure includes structural changes in the stands—species and tree size class shifts—and aligns with the results of Reeves and Stringer (2011) in their study of wildfire impacts. Unfortunately, Knapp and others' (2017) work is based only on red oak species from Missouri.

CONCLUSIONS

A literature review of residual tree wounding from two important eastern hardwood silvicultural practices, timber harvesting practices and prescribed fire, shows a lack of contemporary study related to fire-caused wounds and the effects of prescribed fire on the quality and value of residual trees. Knowledge of wound occurrence is important because of subsequent degradation caused by organisms that invade the tree through exposed sapwood, and the negative effects of tree decay. Multiple surveys of landscape level wounding rates and causes along with evaluations of linked decay amounts and effects, from the 1970s, indicated that fire-caused wounding and decay were more prevalent than wounding from harvesting activities. Since these studies were conducted before the era of prescribed fire use in eastern forests contemporary surveys are needed. Several studies provide information on the timber and product value impacts of wounds associated with mechanical damage. Until recently, studies of the impacts of wounds caused by prescribed fire were lacking. As the knowledge base of prescribed fire impacts on merchantable timber yields grows, comparing the residual tree damage and value impacts of forest operations and prescribed fire will be constructive.

Wounds sustained by residual sawtimber in partial cut harvesting operations are largely the result of skidding activity rather than tree felling. Damage assessments post-logging have been inconsistent in dealing with wounding of roots and tree crowns, but bole wounds to residual sawtimber, especially wounds to the lower

bole, are uniformly reported. Residual tree wounding rates ranged from 7 to 66 percent, with rates of 15 to 35 percent commonly cited.

Wounds create tree health vulnerabilities by exposing the sapwood. Wounds larger than 100 to 150 in² have been categorized as “major.” For wounds caused by logging operations, about 15 percent of residual stems suffer from major wounds. Studies of prescribed fire caused wounds have not used the same classification. Particularly important in silvicultural systems employing prescribed fire is wound width since wider wounds take longer to close. With many regeneration and restoration objectives dependent on multiple prescribed fires, wound closure before repeat fires is important for forests managed for future timber income. Since many pathogens that attack exposed sapwood are moisture dependent, wound shape and location also can affect decay development. Wounds that hold water and wounds in contact with the ground are potentially more vulnerable to decay organisms.

More information is needed on the effects of prescribed fire on the quality and volume of stems of different species and sizes as time elapses before the stems are finally harvested. Published results to-date have been based on small sample sizes in only two locations. For forests managed with timber production and income goals included as priority objectives, this information is essential.

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OVERSTORY TREE MORTALITY AND WOUNDING AFTER THINNING AND PRESCRIBED FIRE IN MIXED PINE-HARWOOD STANDS

Callie Jo Schweitzer, Yong Wang, and Dan Dey

Abstract—The William B. Bankhead National Forest in northcentral Alabama is using active management to shift mixed *Quercus-Pinus* forests toward forests more dominated by upland hardwoods. We studied the impact of three levels of thinning (none, light thin, and heavy thin) and three levels of prescribed fire (none, infrequent fire, and frequent fire) and all combinations in a factorial experimental design to assess overstory (trees >5.5 inches d.b.h.) mortality. All burns were conducted during the dormant season. In all 36 treatment stands, we surveyed five permanent vegetation plots before treatments were initiated and in each subsequent growing season post-burn. Overstory stem density was reduced primarily through the thinning operations, and little mortality was detected. Mortality was significantly greater for unthinned stands, regardless of burning regime, compared to the thinned treatments. Although basal wounding from logging may increase susceptibility to fire damage and mortality, the trees that initially died (after thinning and one fire) did not have bole wounds. Thinning with frequent fire resulted in the greatest number of trees with bole wounds, however, trees that died in these treatments did not have any discernable bole wounds.

INTRODUCTION

The use of prescribed fire to achieve management goals in hardwood forests continues to be scrutinized. Restoring fire as a process in these forests has been suggested as a means to move towards oak-dominated stands (Dey and Hartman 2005, Nowacki and Abrams 2008). However, research has shown that fire's application for reaching desired future conditions, which are weighted towards maintaining or enhancing oak (*Quercus* spp.) in these systems, must be timed to ecosystem stage (Arthur and others 2012, Dey and Schweitzer 2015), and obtaining and favoring oak over other competitors, such as red maple (*Acer rubrum* L.), in the reproduction cohort may prove challenging (Keyser and others 2017, Matlack 2012, Schweitzer and others 2016), whether used alone or in combination with tree harvesting. Fire's longer-term impact on residual stand quality is receiving heightened attention, although past research has addressed this issue.

Reports on stand quality from the early 1900s mention fire as a detriment to growing high quality timber in the southern upland forests of the United States. Concern over the adverse effect fire was having on the composition and structure of the forest was reported early by Zon (1907) who detailed the growth of sprouts under badly burned and slightly burned scenarios. At that time in the Southeast, little effort was made

to keep fire out of the forest, and Zon lamented, "Protection against fire is essential to conservative forest management in the southern Appalachians." Frothingham (1917) noted how the omnipresence of disturbance, including fire and grazing, along with indiscriminant logging and the leaving of poor quality residuals, resulted in second growth forest that then needed management to sustain the timber demand. Frothingham stated, "Recurrent surface fires have badly injured the timber and the forest floor . . . and . . . (fire) is hardly to be recommended until experiments have thoroughly demonstrated its superiority to clearcutting and girdling without burning." These sentiments were echoed by Buell (1928) who stated "The first principle of wise use, control of fire is so well proved that it needs no comment", although no actual research on fire was reported. Zon and Scholz (1929) reported that repeated fire may delay regrowth of a stand indefinitely. However, these statements and conclusions were based more on anecdotal observations and less on actual research on fire damage and mortality.

One of the earliest studies addressing the effects of fire detailed the relationship between tree size and mortality caused by fire for American chestnut and oaks (McCarthy and Sims 1935). Their study showed that smaller saplings, 3 inches diameter at breast height (d.b.h.) and smaller, suffered the most mortality with

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fire, and that susceptibility depended on bark thickness, fire intensity and duration, and crown height above the ground level. In another study of 5,882 cut and scaled trees, those trees wounded at the base had 10 times the amount of butt rot, and 97 percent of those basal wounds were caused by fire according to Hepting and Hedgcock (1935). In their sample, 67 percent of trees with basal fire wounds had butt rot, while only 6 percent of trees without basal wounds had butt rot. These stands were mainly of seedling origin, and they concluded that butt rot in sprout-origin stands occurred independently of fire scars. Roth and Sleeth (1939) confirmed this, reporting from 10 to 40 percent of oak sprouts were butt-rotted as a result of decay transmission from parent stump to sprout. The lasting effects of fire on residual trees was noted and addressed by early silviculture research (Hedlund 1959, Jemison 1946, Paulsell 1957, Wahlenberg 1953).

More recently, it has been realized that the successes in controlling wildfire has come with an ecological consequence of shifting species composition in eastern hardwood and pine forests (Huntley and McGee 1982, Little 1973, Merritt 1979, Sander 1977). This is a major impetus for using prescribed burning in eastern forests, to sustain oak-pine forests and to restore oak-pine natural communities. Consequently, caution on the use of fire is still reported though we lack much research to enable understanding of fire injury to trees and silvicultural methods to minimize damage. Wendel and Smith (1986) showed that overstory trees died or had decreased vigor 5 years after a single fire in an upland hardwood forest in West Virginia. Ward and Stephens (1989) noted that 30 years after a summer wildfire, mortality continued and residual trees had more stem defects compared to unburned areas, although oak regeneration dramatically increased and grew fast in the burned stand. DeSelm and others (1991) reported a doubling of overstory tree mortality compared to controls after 27 years of annual and periodic fires in an upland forest on the Highland Rim in Tennessee. In more recent research, Hutchinson and others (2012) reported a 10 percent decline in overstory trees (> 9 inches d.b.h.) after 2-4 burns over 13 years in mixed oak stands in southern Ohio. Others have reported that single fires have no effect on large diameter stems, such as overstory oaks (Thomas-Van Gundy and others 2015), but multiple fires can eventually cause a reduction in saplings (Arthur and others 2015, Schweitzer and others 2016, Waldrop and others 2008). Research into the mechanism of fire effects and resistance increased our understanding of some of the results of fire on defect and mortality (Butler and Dickinson 2010, Hengst and Dawson 1994, Pomp and others 2008, Smith and Sutherland 1999, 2006), although few studies examined the effect on wood quality (Marschall and others 2014). Collectively, studies on fire effects on tree mortality are scattered spatially, in various forest types and species, include trees of

differing sizes and ages and time of burning, and vary temporally in fire regime such that it is hard to conclude definitively on the effects of future fire applications.

We examined overstory tree mortality and wounding in a mixed pine-hardwood forest that received thinning, prescribed fire and combinations of thinning and fire. These silvicultural techniques are being tested to promote and accelerate the succession of these stands towards dominance of oaks in the overstory. The objective of the study reported here was to (1) examine stand-level overstory tree mortality under thinning, burning and their combination compared to no management disturbance, and (2) examine post-treatment damage to the lower bole of overstory trees as related to disturbance.

METHODS

The 180,000-acre Bankhead National Forest (BNF), in north-central Alabama, is in the Cumberland Plateau Section of the Appalachian Plateaus physiographical province (Fenneman 1938), and study stands are more specifically characterized by the Strongly Dissected Plateau subregion of the Southern Cumberland Plateau, within the Southern Appalachian Highlands (Smalley 1979). These are Plateau tabletop sites with unmanaged pine plantations that have progressed to mixed hardwood-pine stands. Base age 50 site indices for loblolly pine (*Pinus taeda* L.), red oaks [northern red oak (*Q. rubra* L.), black oak (*Q. velutina* Lam.), scarlet oak (*Q. coccinea* Munchh.), and southern red oak (*Q. falcata* Michx.)], and white oaks [white oak (*Q. alba* L.) and chestnut oak (*Q. prinus* L.)] are 75 feet, 65 feet, and 65 feet, respectively (Smalley 1979). Soils are loamy, formed in residuum weathered from sandstones and conglomerates (Smalley 1979). Climate of the region is temperate with mild winters and moderately hot summers with a mean temperature of 55.4 °F and mean precipitation of 59 inches (Smalley 1979). Study stands were located on broad, flat ridges with elevations ranging from 720 to 1,220 feet above sea level.

The BNF study was a randomized complete block design with a 3 by 3 factorial treatment arrangement and four replications of each treatment. Treatment details and implementation are given in table 1. Each treatment was replicated 4 times, for a total of 36 treatment stands. Stand size ranged from 22 to 46 acres. Treatments were representative of management practices described in the BNF's Forest Health and Restoration Project for restoring oak forests and woodlands (USDA Forest Service 2003). Criteria for stand selection were based on species composition, stand size, and stand age. Treatment stands were at least 22 acres in size with basal areas ranging from 122 to 132 square feet per acre. Commercial thinning was conducted by marking from below smaller trees or trees that appeared diseased or damaged; canopy

Table 1—Thinning and prescribed fire treatment designations, residual basal area (std) and stems per acre densities (SPA)(std) after thinning (Time 1), and fire return interval and fires to date, William B. Bankhead National Forest, Alabama

Treatment thin/fire return frequency	Thinning target basal area (ft ² a ⁻¹)	Time 1 basal area (ft ² a ⁻¹)	Time 1 SPA	Fire return interval (years)	Fires to date (number)
No thin/No Rx	No thin	137.5 (20.9)	264 (54)	0	0
No thin/Infrequent Rx	No thin	127.1 (29.1)	275 (69)	9	1
No thin/Frequent Rx	No thin	130.3 (26.9)	317 (76)	3	3
Light thin/No Rx	75	67.5 (18.9)	106 (37)	0	0
Light thin/Infrequent Rx	75	71.7 (16.6)	124 (15)	9	1
Light thin/Frequent Rx	75	64.4 (11.7)	108 (18)	3	3
Heavy thin/No Rx	50	50.6 (9.9)	86 (24)	0	0
Heavy thin/Infrequent Rx	50	49.3 (9.3)	85 (16)	9	1
Heavy thin/Frequent Rx	50	49.9 (9.2)	84 (19)	3	3

trees were also removed to meet target residual basal areas. Hardwoods were preferentially retained. Thinning treatments were completed prior to the initiation of the burning treatments (thinning conducted from June through December).

Prescribed burning was conducted during the dormant season (January through March) using backing fires and strip head fires to ensure that only surface fire occurred. Immediately prior to each fire, we installed 6 to 8 HOBO data recorders (HOBO U12 Series Datalogger, Onset Computer Corporation, Cape Cod, MA) connected to a temperature probe (HOBO TCP6-K12 Probe Thermocouple Sensor, Onset Computer Corporation, Cape Cod, MA) at each vegetation sampling plot (30-48 probes per stand). Installation was based on the design of Iverson and others (2004). Out of the 48 fires in this study, we were only able to obtain fire data for 32 fires, and only for the frequently burned treatments. Ignition type included hand strip firing at approximately 26 feet intervals and aerial ignition for 6 fires; all others were ignited by hand strip firing. All study burns were included as part of a larger target burn area on the BNF, and burn areas ranged from 150 to 3,000 acres. Absolute maximum fire temperatures ranged from 2 °F (27 January 2007) to 575.4 °F (March 16, 2013). On average, the maximum temperature was 203.9 °F (std 145.1 °F) for the first burn, 253.8 °F (std 130.3 °F) for the second burn, and 407.1 °F (std 165.4 °F) for the third burn.

Overstory trees (5.6 inches d.b.h. and greater) were surveyed on five permanent measurement plots (0.2 acre circular plots) systematically located in each stand. Distance and azimuth to all trees were recorded from plot center; species, diameter, tree grade, status, and damage attributes were recorded at pretreatment (time 0), after the thinning and first burn (time 1), the next sampling period followed the second burn (time 2) and the final sampling period followed the third burn (time 3), which was seven growing seasons after pretreatment.

We assessed damage to the lower bole by enumerating each external wound and measuring the wound area (width X length) using a hand-held ruler.

We used an analysis of variance (ANOVA) by implementing PROC MIXED in SAS 9.0 (SAS Institute Inc. 2000), specifying a random effect (block) and a repeated statement (time) with the type of covariance matrix assigned unstructured using TYPE=UN option specified as stand (treatment). The effects were then assigned the between-subject degrees of freedom to provide for better small-sample approximations to the sample distributions. We used DDFM=KENWARDROGER option to perform the degrees of freedom calculations detailed by Kenward and Roger (1997). We used ANOVA to test for differences in overstory basal areas and densities, mortality, and incidence of wounding, among pre- and post-treatment samples (within subject factor) and among treatments (between subject factor). All analyses were conducted at a significance level of $\alpha \leq 0.05$ followed by Tukey's multiple comparison test to detect pair-wise differences.

RESULTS

The response of stand structure and composition to the thinning, and initial sequence of prescribed fire (one fire for the infrequent fire treatment and three fires for the frequent fire treatment) were detailed in Schweitzer and others (2016). Stands had 131.2 square feet per acre of basal area with 290 stems per acre (SPA), on average, at the beginning of the study (table 1). Thinning targeted pines; out of 174 SPA harvested in the light thin, 125 SPA were loblolly pine and 32 SPA were Virginia pine (*P. virginiana* Mill.). For the heavy thin, 132 SPA of loblolly pine and 45 SPA of Virginia pine were removed as the majority of the 199 SPA harvested. There were significant interactions between time and treatment for basal area ($F_{24, 42} = 82.38, P \leq 0.001$) and SPA ($F_{24, 42} = 99.07, P \leq 0.001$). For all thinned treatments, basal area and stem densities for all species were lower at times 1, 2,

and 3 compared to time 0 and there were no differences, within each thinned treatment, between values at times 1, 2, and 3. Residual basal areas and SPA at time 1 averaged 67.9 square feet per acre and 113 SPA for the light thin and 49.9 square feet per acre and 85 SPA for the heavy thin. At time 3, stem density was lowest in stands subjected to heavy thin and frequent prescribed fire (84 SPA) (time 3 $F_{3, 25} = 28.52$, $P \leq 0.001$) and highest in the unthinned stands, regardless of burn treatment (266 SPA no fire; 277 SPA one fire; 333 SPA three fires).

There were no significant differences among treatments for standing dead trees (snags) at the beginning of the study ($F_{3, 25} = 1.36$, $P = 0.20$). Across all 180 measurement plots, only 186 snags were tallied, resulting in an average of 5 snags per acre. Seventy-eight percent of these snags were loblolly pine, 15 percent were Virginia pine, and 7 percent were unidentifiable. These snags were removed from the mortality database, as were any trees that were knocked over during the thinning operation.

Mortality was compared for each time period; we did not collect data immediately after the thin and prior to the burn for the thin and burn treatments. Mortality was calculated for each sample period; dead trees at time 1 were removed from analysis at time 2; dead trees at time 1 and 2 were removed from analysis at time 3. Species that died at time 1 included flowering dogwood (*Cornus florida* L.), eastern hemlock (*Tsuga canadensis* (L.) Carr.), loblolly pine, red maple, scarlet oak, sourwood (*Oxydendrum arboreum* DC), Virginia pine, and yellow-poplar (*Liriodendron tulipifera* L.). The number of dead trees across all treatments was low and ranged from 1 to 4 SPA; there was a difference among treatments ($F_{3, 25} = 3.05$, $P \leq 0.0031$), with unthinned stands having the highest mortality (table 2). The portion of live stems by species across all treatments was 85.1 percent

pinus, 7.2 percent oaks, and 5.3 percent other species combined. The mortality was concentrated in the pines, with 89.3 percent of dead trees loblolly or Virginia pine, and 9.5 percent other species; no overstory oaks died at time 1 (fig. 1).

At time 2, stands either had received another fire or were undisturbed for 4 growing seasons. Mortality was noted for 14 different species, and included black cherry (*Prunus serotina* Ehrh.), bigleaf magnolia (*Magnolia macrophylla* Michx.), black oak, chestnut oak, flowering dogwood, loblolly pine, post oak (*Q. stellata* Wang.), red maple, scarlet oak, southern red oak, sourwood, Virginia pine, white oak and yellow-poplar. Mortality was greatest in the no thin, infrequent fire treatment ($F_{3, 25} = 4.20$, $P \leq 0.0001$) and did not differ among the other treatments (table 2). Mortality in the unthinned stands was dominated by pine species, accounting for 92 percent of the total mortality. In the light thin with infrequent fire, the majority of the mortality was in the other species category (46.2 percent), although the alive stem density was only 5.5 percent other species (fig. 2). Twenty-five percent of the mortality was oaks for the light thin with no fire and heavy thin with no fire treatments (fig. 2).

At time 3, the frequent fire treatments had received a third fire and the other treatments were now undisturbed for 7 growing seasons. Mortality differed among treatments ($F_{3, 25} = 7.11$, $P \leq 0.0001$) and was greatest in the unthinned stands (table 2). Mortality was concentrated in the pines in these treatments, accounted for 91.5 percent of the total mortality, and 83.6 percent of the total live stems in pine (fig. 3). The heavy thin with three fires treatment had 8 SPA of mortality, the next highest level after the unthinned treatments, although it only differed from the unthinned with no fire treatment. A shift in alive stem densities in the heavy thin with frequent fire treatment from pretreatment to time 3 was

Table 2—Mortality stems per acre (std) of trees > 5.5 inches d.b.h., by time and treatment, following thinning and prescribed fire, William B. Bankhead National Forest, Alabama

Treatment thin/fire return frequency	Time 1 post thin and one fire for all treatments	Time 2 post thin and two fires for 3 Rx treatments	Time 3 post thin and three fires for 3 Rx treatments
No thin/No Rx	3.75 (3.93) a	6.75 (6.93) b	17.25 (15.34) a
No thin/Infrequent Rx	2.25 (3.43) ab	13.5 (22.54) a	10.50 (9.45) bc
No thin/Frequent Rx	3.75 (5.10) a	5.25 (6.58) b	13.75 (13.46) ab
Light thin/No Rx	1.75 (2.45) ab	1.00 (3.48) b	2.75 (3.80) d
Light thin/Infrequent Rx	1.25 (2.75) b	3.25 (4.38) b	5.00 (6.07) cd
Light thin/Frequent Rx	0.75 (1.83) b	1.75 (2.94) b	2.50 (4.14) d
Heavy thin/No Rx	0.50 (1.54) b	1.00 (2.62) b	2.00 (2.99) d
Heavy thin/Infrequent Rx	1.50 (3.28) b	2.75 (3.43) b	2.50 (3.44) d
Heavy thin/Frequent Rx	0.75 (2.45) b	2.75 (3.80) b	7.75 (14.55) bcd

Different letters within time column indicates significant difference ($\alpha=0.05$) among treatments.
D.b.h.=diameter at breast height.

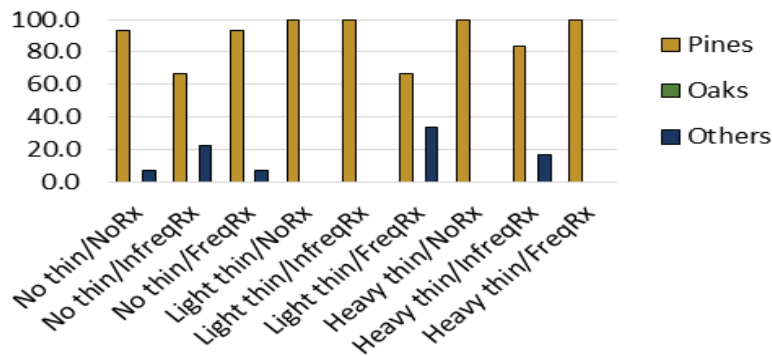


Figure 1—Percent mortality by species groups for each treatment at time 1, one growing season post thinning and post prescribed burn 1, for all treatments on the William B. Bankhead National Forest, Alabama.

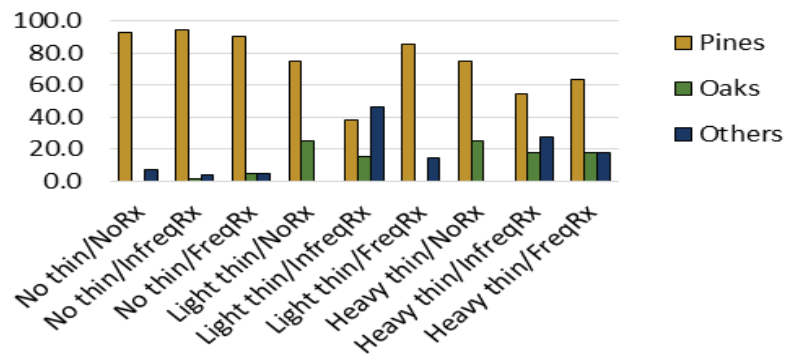


Figure 2—Percent mortality by species groups for each treatment at time 2, post prescribed burn 2 for the frequent Rx treatments, four growing seasons post thin, and post thin and burn for the infrequent Rx treatments, for all treatments on the William B. Bankhead National Forest, Alabama.

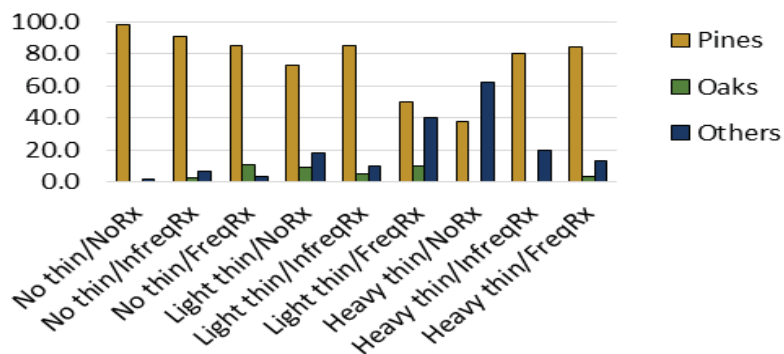


Figure 3—Percent mortality by species groups for each treatment at time 3, one growing season post prescribed burn 3 for the frequent Rx treatments, seven growing seasons post treatment for all other treatments on the William B. Bankhead National Forest, Alabama.

80.7 to 63.7 percent for pines, 8.9 to 22.6 percent for oaks, and 8.0 to 11.6 percent for other species. Over the study period, there was an increase in the proportion of alive oak stems as most mortality was in the pines. Oak mortality was found in all treatments except the unthinned with no fire, heavy thin with no fire, and heavy thin with infrequent fire treatments. The total number of oaks that died was <1 per acre for all three time periods, except for the no thin with frequent fire treatment, which had 1.5 SPA oak death in 7 years. The average diameter of dead oak trees was 7.5 inches.

The amount of visible lower bole damage was minimal across the study. On all 180 measurement plots, we noted 14 different species with damage (table 3). The trees that died in the unthinned treatments at time 1 (no harvesting, two treatments were burned) had no recorded wounds. In general, trees with wounds at time 1 did not die by time 3; across all 180 plots, 6 total trees with damage at time 1 were either knocked over or died by time 3 (0.2 SPA impacted). At time 1, all treatments with thinning had more trees with lower bole damage compared to the unthinned stands ($F_{3,25} = 12.60$, $P \leq 0.0001$) (table 4). In the light thin with frequent burn treatment, there were 16 SPA with damage, the majority of trees with damage were loblolly pine, and the average wound size was 129 square inches. At time 2 and time 3, the thin with frequent burning treatments had the most damage, ranging from 10 to 15 SPA of damaged trees. Damaged trees included oaks (2 SPA, average diameter 9.9 inches), loblolly pine (9 SPA, average diameter 10.9), and less than one SPA of red maple

(average diameter 7.0 inches), sweetgum (*Liquidambar styraciflua* L.) (average diameter 16.1 inches), Virginia pine (average diameter 9.4 inches) and yellow-poplar (average diameter 11.6 inches). We did not note any damage in the unthinned treatments at time 2, but at time 3, after three fires, there were 4 SPA with damage, compared to one SPA of noted damage for the one and no burn treatments.

DISCUSSION

The effect of low intensity prescribed fire on mortality of overstory trees is related to site factors, stand composition and structure, and fire characteristics. Using fire to manipulate the understory in hardwood systems, with a goal to enhance oak recruitment into larger size classes, has been reported with disparate results (Arthur and others 2012, Blankenship and Arthur 2006, Brose 2010, Brose and others 2013, Hutchinson and others 2012, Keyser and others 2017, McEwan and others 2011, Schweitzer and others 2016). Repeated fire will be necessary to alter the stem dynamics in the understory to favor oak over other species, although oak seedlings may be caught in a “fire trap” (Grady and Hoffman 2012) and a fire-free period will be needed to facilitate recruitment (Arthur and others 2015, Thomas-Van Gundy and others 2015). This prescription, of repeated fires with a subsequent fire-free antecedent period, complicates reducing residual tree damage due to fire. Value loss of fire scarred trees can be minimized if trees are harvested within 5 years (Marschall and others 2014, Weidenbeck and Schuler 2014). Application of three (or more) prescribed fires within 5 years is

Table 3—Total number of trees with lower bole damage on 180 measurement plots across all treatments, by species and time, following thinning and prescribed burning on the William B. Bankhead National Forest, Alabama

Damaged tree species	Number of damaged trees tallied on all plots		
	Time 1: post thin and one prescribed fire	Time 2: post second prescribed fire	Time 3: post third prescribed fire
Beech (<i>Fagus grandifolia</i>)	0	1	0
Black cherry (<i>Prunus serotina</i>)	5	1	9
Black oak (<i>Quercus velutina</i>)	2	1	2
Chestnut oak (<i>Quercus prinus</i>)	19	13	15
Loblolly pine (<i>Pinus taeda</i>)	223	127	158
Post oak (<i>Quercus stellata</i>)	0	2	0
Red maple (<i>Acer rubrum</i>)	4	3	9
Scarlet oak (<i>Quercus coccinea</i>)	7	6	17
Sourwood (<i>Oxydendrum arboreum</i>)	1	2	1
Southern red oak (<i>Quercus falcata</i>)	0	0	2
Sweetgum (<i>Liquidambar styraciflua</i>)	0	2	2
Virginia pine (<i>Pinus virginiana</i>)	12	8	8
White oak (<i>Quercus alba</i>)	4	4	6
Yellow-poplar (<i>Liriodendron tulipifera</i>)	5	6	8

Table 4—Number of trees with lower bole damage in stems per acre (std) for trees > 5.5 inches d.b.h., by time and treatment, following thinning and prescribed fire, William B. Bankhead National Forest, Alabama

Treatment Thin/Fire return frequency	Time 1 post thin and one fire	Time 2 post thin and two fires or 3 growing seasons	Time 3 post thin and three fires or 6 growing seasons
No thin/No Rx	0.25 (1.12)d	0.00 (0.00)d	1.25 (2.75)d
No thin/Infrequent Rx	0.50 (1.54)d	0.00 (0.00)d	1.00 (3.48)d
No thin/Frequent Rx	0.00 (0.00)d	0.00 (0.00)d	3.75 (8.87)dc
Light thin/No Rx	13.75 (9.72)ab	7.00 (6.77)bc	7.50 (6.59)c
Light thin/Infrequent Rx	9.50 (9.72)bc	3.75 (3.58)bd	8.25 (7.30)bc
Light thin/Frequent Rx	16.0 (11.77)a	13.25 (12.28)a	14.50 (11.91)a
Heavy thin/No Rx	7.75 (9.24)c	4.50 (5.36)c	5.00 (6.28)cd
Heavy thin/Infrequent Rx	8.25 (7.66)c	6.00 (7.54)bc	5.50 (5.83)cd
Heavy thin/Frequent Rx	14.5 (9.99)ab	9.50 (7.93)ab	12.50 (10.20)ab

Different letters within time column indicates significant difference ($\alpha=0.05$) among treatments.

D.b.h.=diameter at breast height.

challenging in most upland hardwood systems, thus residual trees with fire injury will most likely not meet the <5 year fire-scar residence time.

The stands under study on the BNF are unique for their mixed pine-hardwood composition. The dominance of pine in the overstory of these stands, coupled with the targeted removal of pine in the thinning operation and the higher level of mortality of pine over other species, has assisted the management goal of moving these stands towards greater hardwood (oak) dominance. The highest incidence of overstory tree mortality was documented in the unthinned stands, indicative of the overstocked status of these stands (Gingrich 1967) and stress due to competition. Adding fire to these unthinned stands did not alter the overstory mortality, as stands burned three times had the same mortality as unthinned stands without fire. Tree vigor and species composition are also playing a role in this response (Yaussy and Waldrop 2010). We did not assess tree vigor directly; we used a modified tree grading system that accounted for tree size and surface characteristics (Miller and others 1986), and found that the Virginia pine in these stands were of the lowest grade. Overstory tree mortality did not differ among heavy or light thinning regardless of fire (none, one, or three), and delayed mortality has not been noted at this time. Others have found a 9 to 25 percent decline in density of overstory trees on burned plots (Arthur and others 2015, Brose and Van Lear 1999, Hutchinson and others 2012), and Fan and Dey (2014) reported that overstory basal area continued to increase in stands burned annually to periodically (e.g., every 3 years) for 10 years in the Missouri Ozarks.

The use of prescribed fire and thinning to move these stands towards oak, and perhaps even towards oak woodland structure (Dey and others 2017), can be inferred from the heavy thin with frequent fire treatment. These stands had a relative basal area of oak (white oak,

chestnut oak, and scarlet oak) that was 4.4 percent and a relative oak stem density of 3.4 percent. After thinning, these increased to 10.5 and 14.6 percent, respectively, and after three fires oak relative basal area was 12 percent and oak relative density was 15.7 percent. A concurrent reduction in the density of understory stems has been reported (Schweitzer and others 2016). With continued burning concern exists with regard to the fire-induced damage and mortality that could reduce the overstory oak component as indirect fire mortality. Others have found that overstory trees are often scarred on the lower bole by fire, but mortality from low-intensity burns was low (Dey and Fan 2009, Hutchinson and others 2005, Regelbrugge and Smith 1994).

Damage and value loss related to both wildfire and low-severity prescribed fires have been reported in very few studies, although predictive variables have been detailed (Loomis 1973, 1974). In a study of fire-damaged oaks in the Missouri Ozarks, the proportion of the butt log with defect after fire was shown to increase with increasing size of fire scar, increases in time since fire injury, and decreases in tree diameter, with one third of the volume defect 25 years after the trees (white, black, and scarlet oaks) were fire scared (Stambaugh and Guyette 2008). Reeves and Stringer (2011) found that most loss (value and volume) was due to structural change (tree mortality and size class changes), with about one third of value loss due to degrade and rot. Boards sawn from the side of logs directly impacted by low to medium intensity fire developed defect (mineral stain and some level of decay) that differed depending on species, and lumber loss was to be expected (Wiedenbeck and Schuler 2014). For dimensional red oak lumber, a study of individual tree loss due to fire damage showed volume loss at 3.9 percent and value loss at 10.3 percent after 14 years since fire injury (Marschall and others 2014). Coupling the restoration of fire as a process in these upland hardwood systems with efforts to minimize

residual tree damage will continue to be investigated (Dey and Schweitzer 2015, Sutherland and Smith 2000). We have noted minimal lower bole damage, with most damage found in the most disturbed treatments; trees in treatments that received three fires but were not thinned did have an increase in the incidence of wounding, but the SPA impacted was quite low (4 SPA).

CONCLUSION

Managers in some Southern mixed pine-hardwood forests are using prescribed fire to change stand structure and composition. Although research has been reported documenting the effects of fire on these systems, questions remain as to the long-term impact on forest composition, structure, and health, with mortality rates and wood quality surrogate metrics of forest health and resilience. There exists a conundrum of ensuring the frequency of fire needed to target understory competitors such as red maple, and the need to harvest residual trees quickly following fire to reduce defects caused by rot. In this study, repeated fires that were all conducted in the dormant season have not had appreciable impact on overstory tree mortality or degrade. We found that fire has unequally impacted overstory pine over the more desired oaks, accelerating the management goal of moving these stands away from pine dominance and towards more oak-upland hardwood dominance.

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REHABILITATION OF CUTOVER OAK-HICKORY STANDS USING PRESCRIBED BURNING AND HERBICIDE APPLICATION: DECADAL RESPONSES OF VEGETATION

Mohammad M. Bataineh and Matthew Pelkki

Abstract—High-grading practices in which commercially desirable species and trees are removed without consideration of residual stand conditions or regeneration persist today. Such practices result in degradation of current stand conditions and limit future management alternatives. When these practices are superimposed on forests with a legacy of fire suppression, earlier exploitative cuttings, and high incidence of pests, the consequent synergistic effects alter successional trajectories and increase vulnerability to state shifts. In this study, we evaluate the use of rehabilitation treatments in the form of single and repeated burning, at 3-year intervals, with or without the use of herbicide to improve composition and structure of stands subjected to high-grading in the Arkansas Ozarks. Stem injection reduced black gum, red maple, and hickory from the overstory and allowed for greater recruitment of white oak. Repeated burning, at 3-year intervals, increased red oak sapling abundance at the expense of red maple and in conjunction with stem injection removed large red maple saplings. Repeated burning improved the composition and structure of high-graded stands shifting trajectory away from mesophication and towards greater abundance of oak, especially red oak that has been in steady decline in the Arkansas Ozarks.

INTRODUCTION

The oak-hickory (*Quercus-Carya*) forest type is a major component of the Eastern Deciduous Forest and the Central Hardwood Region (Dyer 2006, Johnson and others 2009). Oak-hickory type covers nearly two thirds of the Central Hardwood acreage with the greatest abundance and largest contiguous area in the Ozark Highlands of southern Missouri and northwestern Arkansas (Barrett 1995, Fei and others 2011). Anthropogenic disturbances, including the use of fire by Native Americans and exploitative cutting, burning, land clearing, and grazing by early European settlers, have shaped these forests (Johnson and others 2009). Frequent and extensive disturbances during early settlement period coupled with sprouting ability and fire-tolerance of oaks have contributed to the dominance of oaks in today's oak-hickory forests (Brose and others 2013, 2014). The fire suppression era that followed has resulted in an altered disturbance regime and compositional shifts toward shade-tolerant, fire-sensitive species (i.e., mesophication), such as red maple (*Acer rubrum*) (Nowacki and Abrams 2008). Widespread declines in oak density and standing volume were reported for the Ozark Highlands during the period 1980-2008 (Fei and others 2011). Decline in oak abundance was the direct result of recruitment failures

and regeneration difficulties that occurred throughout the extensive range of this forest type (Brose and others 2014). Furthermore, episodes of crown dieback followed by canopy tree mortality (i.e., oak decline), oak wilt disease, and outbreaks of native Cerambycid oak borers have all contributed to the overall decline and health concerns of these systems (Haavik and others 2012, Stephen and others 2001, Tainter and Gubler 1973).

High-grading practices in which commercially desirable species and trees are removed without consideration to residual stand conditions or regeneration persist today (Kenefic and others 2014). Such practices result in degradation of current stand conditions and limit future management alternatives. High-grading in historically oak-dominated forests with understories of shade-tolerant, fire-sensitive species, can accelerate mesophication rates (i.e., microenvironmental conditions that reduce fire potential) and increase vulnerability to phase or regime shifts (Nowacki and Abrams 2008). Further acceleration or alternative successional trajectories can occur when high-grading is superimposed on forests with a high incidence of pests. The abundance of recalcitrant mesophytic understories in oak-hickory stands of the Ozarks was well documented in the aftermath of the most recent

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(1990-2000) oak decline episode (Chapman and others 2006, Heitzman 2003). These conditions heighten the need for rehabilitation or restoration treatments in these systems as mature canopy trees succumb to oak decline as well as harvest removals. Rehabilitation entails application of silvicultural treatments to bring about desired stand characteristics, previously degraded through mismanagement (e.g., high-grading) (Kenefic and others 2014). In this study, we evaluate the use of rehabilitation treatments in the form of single and repeated burning, at 3-year intervals, with or without the use of herbicide to improve composition and structure of stands subjected to high-grading in the Arkansas Ozarks. We refer to these treatments as rehabilitation, as opposed to restoration, since our goal is not to mimic reference conditions but to halt or reverse state shifts (Kenefic and others 2014). We use a long-term experiment established in 2004 to elucidate a decade-long period of stand dynamics as a result of these rehabilitation treatments.

MATERIALS AND METHODS

Study Area

The study is part of the University of Arkansas Savoy Research Unit near the town of Savoy in Washington County, AR (fig. 1). The area is within the Ozark Highlands ecoregion which is characterized by highly dissected rolling to gently sloping terrain underlain by fractured dolomite and limestone with abundant karst features (Woods and others 2004). Within the study area, elevation is between 1200-1300 feet above sea level and aspects vary considerably (Natural Resources Conservation Service 1969). The climate is characterized as humid subtropical with hot humid summers and cool to mild winters (Natural Resources Conservation Service 1969). Mean annual temperature is 58 °F and mean minimum and maximum temperatures are 47 °F and 68 °F, respectively. Mean annual rainfall is 46 inches with an additional 6.3 inches in mean snowfall, and annual frost free-period is 180-194 days. Soils of the study area belong mainly to the Clarksville-Nixa-Baxter association and the Captina-Nixa-Pickwick association (fig. 1) (Natural Resources Conservation Service 1969). Forested stands are predominantly oak-hickory with northern red oak (*Quercus rubra*), southern red oak (*Q. falcata*), black oak (*Q. velutina*), blackjack oak (*Q. marilandica*), white oak (*Q. alba*), post oak (*Q. stellata*), bitternut hickory (*Carya cordiformis*), and mockernut hickory (*Carya tomentosa*) as main species (Morris 2007). Red maple (*Acer rubrum*), flowering dogwood (*Cornus florida*), black cherry (*Prunus serotina*), and black gum (*Nyssa sylvatica*) also occur, especially in the mid- and understory layers. Early- to mid-1800s observer accounts describe the area as “barrens interspersed with prairies” and analysis of first land survey bearing/

witness trees document abundance of red oaks (section Lobatae) and relatively open (17-62 trees per acre) stand conditions within the area (Foti 2004).

The experiment was established within an area of 350 acres that was subjected to high-grading in 1996-1998. All merchantable trees >12 inches in diameter at breast height (4.5 feet; DBH) were removed except for black walnut (*Juglans nigra*). Removals occurred mainly on gently sloping ridgetops. The pre-harvest stand originated following exploitative cuts in the early 1900s. Post-harvest residual overstory was composed of low quality white, black, and post oaks as well as black gum, hickory, and low proportions of other red oaks and red maple. Advance regeneration of red maple, black gum, black cherry, and flowering dogwood represented 50-60 percent of total number of stems, whereas oaks represented roughly 25 percent. More details can be found in Morris (2007).

Experimental Design and Treatments

In 2004, four replicates of 2 acres each were established for each of six treatment levels, which were randomly assigned to treatment units. Treatment levels were combinations of a 2 X 3 factorial design that included prescribed burning (3 levels) and herbicide application (2 levels) as main factors. Prescribed burning levels included a single burn treatment, repeated burning at intervals of 3 years, and no burning. Herbicide treatment included herbicide application in fall of 2005 and no herbicide. Initial burning was conducted in 2005 with repeated burns in 2008, 2011/2012, 2014/2015, and 2017. The third and fourth repeated burns had to be delayed to the following year due to weather and other constraints. All burns were dormant season burns with the majority of burns conducted in March or April. For all burns, ground ignition with a drip torch was used. Ignition pattern varied from backing and ring fires to strip-heading fires. Herbicide was applied as Imazapyr (Chopper®, 50 percent aqueous solution) stem injections to all non-oak trees >4 inches in DBH.

Measurements

Within each treatment unit, two circular fixed-area (1/5th acre) overstory (DBH ≥ 4.5 inches) plots were established and permanently marked. Overstory plots within a treatment unit were separated by a distance of 60 feet. Within each overstory plot, two concentric circular fixed-area subplots were established at each of 0°, 120°, and 240° azimuths. Subplots were 1/100th acre and 1/300th acre in area and were used for sapling (height ≥ 4.5 feet and DBH < 4.5 inches) and seedling (1 foot ≤ height < 4.5 feet) sampling, respectively. Subplots were located at 25 feet from overstory plot center. Sampling plots were established in the fall of

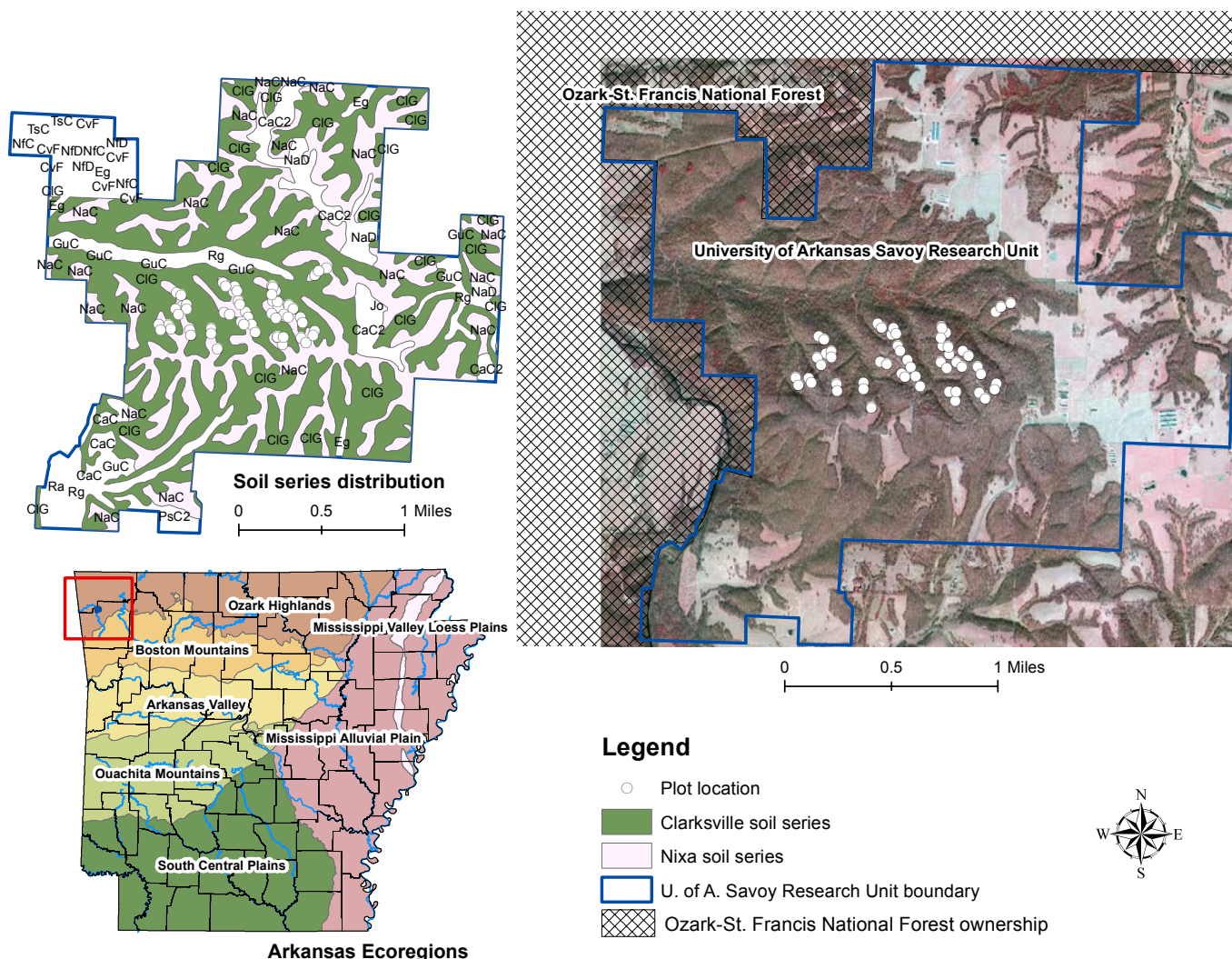


Figure 1—Study area location, bottom left, within the Ozark Highlands ecoregion with plot locations, upper right, and soil series distribution within the area, upper left.

2004 (pre-treatment) and were remeasured in the fall of 2016 (12 years post-treatment) before the last 2017 burn. Overstory trees were measured for DBH, whereas saplings and seedlings were measured for ground-line diameter (GLD). Saplings and seedlings total height (HT) was also measured.

Analytical Approach

Diameters (DBH and GLD) were used to calculate basal area for each species by stratum. Plot level species basal area estimates were divided by total basal area to generate relative basal area estimates. Relative basal area and relative density estimates (trees per acre) were added to calculate importance values (IV) of each species by stratum. Pre- (2004) and post-treatment (2016) data were analyzed separately. The effects of the fixed-factors prescribed burning and herbicide application on the various responses (i.e., density, basal

area, HT, IV) were analyzed using a two-way ANOVA at an alpha level of 0.05. The analysis was conducted using R software version 3.0.2 and the base, car, and plyr packages (R Development Core Team 2013). When significant interaction occurred, tests of simple effects were performed; the effect of prescribed burning was examined at each level of herbicide application (Winer and others 1991). Tukey's multiple comparison procedure was used to separate treatment means whenever an overall significant effect was found. No indication of patterns in residual plots were detected as evidence of violation of assumptions of normality and homogeneity of variance.

RESULTS AND DISCUSSION

Despite previous high-grading removals, pre-treatment (2004) overstory density range was between 131-141 trees per acre (table 1), which is in stark contrast to the

historically open conditions reported by Foti (2004). Not surprisingly, pre-treatment basal area and density reflected low stocking levels, below the B-line, as indicated by Gingrich's stocking charts (Gingrich 1967). Moreover, residuals were undesirable low-quality trees and species (Morris 2007). Pre-treatment overstory structure did not differ between herbicide treatments ($P = 0.40$ - 0.90) or among prescribed burning treatments ($P = 0.73$ - 0.97) and no significant interaction occurred ($P = 0.76$ - 0.91) (table 1). However, single burn treatment units had lower abundance (by 18 percent) of white oak than unburned treatment units (fig. 2). Repeated burn units had higher pre-treatment abundance (by 2 percent) of black cherry compared to unburned. This highlights that removals and post-harvest conditions did not present a confounding effect in comparing rehabilitation treatments—at least for a stand-level assessment as presented here. Stem injection of non-oaks resulted in a 24 percent reduction, over a 12-year period, in overstory trees, mainly in hickory and black gum ($P = 0.02$; table 1; fig. 2). Stem injection also resulted in higher recruitment (by 11 percent) of white oak saplings into the overstory (fig. 2). Overstory basal area and quadratic mean diameter (QMD) did not differ between herbicide treatments ($P = 0.30$ and 0.07 , respectively). Prescribed burning did not result in drastic declines in overstory tree density for single or repeated burning ($P = 0.57$)

and basal area and QMD did not differ among burning treatments ($P = 0.57$ and 0.31 , respectively).

Pre-treatment sapling density, basal area, and total height did not differ among rehabilitation treatments ($P = 0.54$ - 0.92 ; table 2). Also, sapling structural metrics for prescribed burning did not depend on herbicide level ($P = 0.76$ - 0.92). Post-treatment, sapling density was 39 and 52 percent lower for repeated burning than unburned and single burn, respectively ($P < 0.01$). Sapling basal area was also 69 and 60 percent lower for repeated burn compared to unburned and single burn, respectively ($P < 0.01$). Herbicide application did not result in differences in sapling density or basal area ($P = 0.72$ and 0.66 , respectively; table 2). Effect of burning on sapling height 12 years after treatment depended on herbicide treatment ($P < 0.01$), whereas no such dependence was detected for density and basal area ($P = 0.46$ and 0.34 , respectively). Without herbicide, saplings were 24 and 29 percent shorter for single burn compared to unburned and repeated burn, respectively ($P = 0.01$; table 3). With herbicide, saplings were 47 and 31 percent shorter for repeated burn compared to unburned and single burn, respectively ($P < 0.01$). Therefore, repeated burning reduced the density and basal area of saplings but did not reduce their height unless after stem injection to top-kill sprouts of stem-

Table 1—Overstory attributes in 2004 and 2016 by rehabilitation treatment of stem injection (treated and untreated) and prescribed burning (single, repeated at 3-year interval, and untreated) in high-graded oak-hickory stands of the Arkansas Ozarks

Treatment	Density (trees per acre) ^{a, b, c}	Basal area (square feet per acre) ^{a, b, c}	QMD (inches) ^{a, b, c, d}
2004 observation^e			
Herbicide	131 (7)a	61.0 (3.6)a	9.3 (0.2)a
No Herbicide	141 (10)a	61.7 (4.0)a	9.0 (0.2)a
Single Burn	139 (7)A	60.7 (2.4)A	9.0 (0.2)A
Repeated Burn	134 (12)A	62.4 (6.3)A	9.3 (0.2)A
No Burn	135 (13)A	60.9 (4.7)A	9.2 (0.3)A
2016 observation^e			
Herbicide	87 (6)a	58.9 (4.3)a	11.2 (0.4)a
No Herbicide	114 (8)b	64.9 (3.7)a	10.4 (0.2)a
Single Burn	95 (8)A	58.0 (3.7)A	10.7 (0.3)A
Repeated Burn	98 (10)A	65.6 (5.0)A	11.3 (0.5)A
No Burn	108 (12)A	62.0 (6.1)A	10.4 (0.4)A

^a Mean values with standard error in parenthesis.

^b Means followed by the same small letter within a column are not significantly different at the 0.05 level.

^c Means followed by the same capital letter within a column are not significantly different at the 0.05 level.

^d QMD = Quadratic mean diameter.

^e 2004 = pre-treatment; 2016 = 12 years post-treatment.

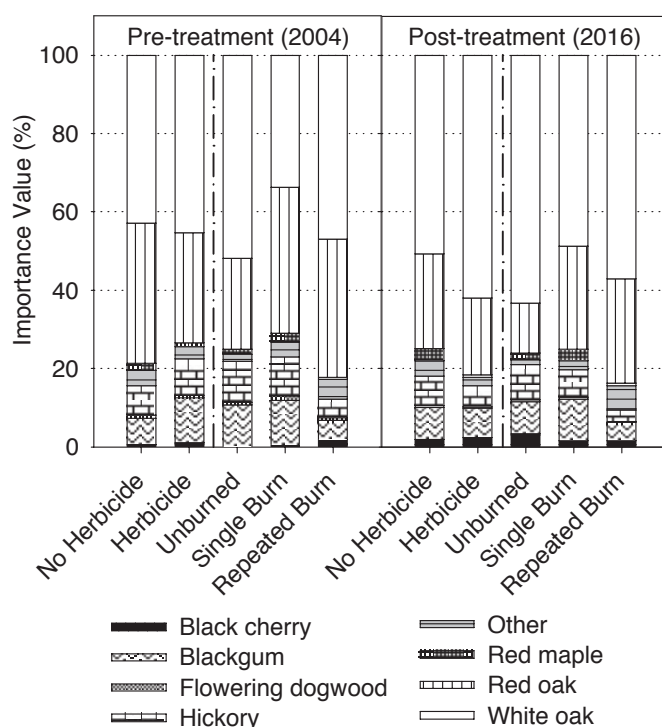


Figure 2—Overstory tree (DBH > 4.5 inches) importance value (sum of relative density and relative basal area) proportion by species group before and after rehabilitation treatments of stem injection and prescribed burning in high-graded oak-hickory stands of the Arkansas Ozarks.

Table 2—Sapling attributes in 2004 and 2016 by rehabilitation treatment of stem injection (treated and untreated) and prescribed burning (single, repeated at 3-year interval, and untreated) in high-graded oak-hickory stands of the Arkansas Ozarks

Treatment	Density (trees per acre) ^{a, b, c}	Basal area (square feet per acre) ^{a, b, c}	Height (feet) ^{a, b, c}
2004 observation^d			
Herbicide	2038 (203)a	34.7 (5.0)a	11.2 (0.5)a
No Herbicide	1946 (190)a	31.5 (2.8)a	11.5 (0.4)a
Single Burn	2032 (315)A	31.9 (4.1)A	11.0 (0.5)A
Repeated Burn	1895 (201)A	29.8 (5.3)A	11.2 (0.6)A
No Burn	2048 (204)A	37.8 (5.2)A	11.9 (0.5)A
2016 observation^d			
Herbicide	1093 (114)a	40.1 (6.8)a	13.8 (1.2)
No Herbicide	1035 (163)a	37.5 (5.6)a	14.6 (0.8)
Single Burn	1411 (189)A	43.6 (5.2)A	12.8 (0.7)
Repeated Burn	674 (96)B	17.4 (4.1)B	13.0 (1.6)
No Burn	1106 (97)A	55.4 (5.7)A	16.8 (0.9)

^a Mean values with standard error in parenthesis.

^b Means followed by the same small letter within a column are not significantly different at the 0.05 level.

^c Means followed by the same capital letter within a column are not significantly different at the 0.05 level.

^d 2004 = pre-treatment; 2016 = 12 years post-treatment.

Table 3—Sapling total height in 2016 by prescribed burning treatment (single, repeated at 3-year interval, and untreated) with or without stem injection in high-graded oak-hickory stands of the Arkansas Ozarks

Treatment	Height (feet) ^{a, b}
Herbicide	
Single Burn	13.8 (1.0)A
Repeated Burn	9.5 (1.4)B
No Burn	18.0 (1.5)B
No herbicide	
Single Burn	11.8 (0.8)A
Repeated Burn	16.6 (1.3)B
No Burn	15.6 (0.7)A

^a Mean values with standard error in parenthesis.

^b Means followed by the same capital letter within a column are not significantly different at the 0.05 level.

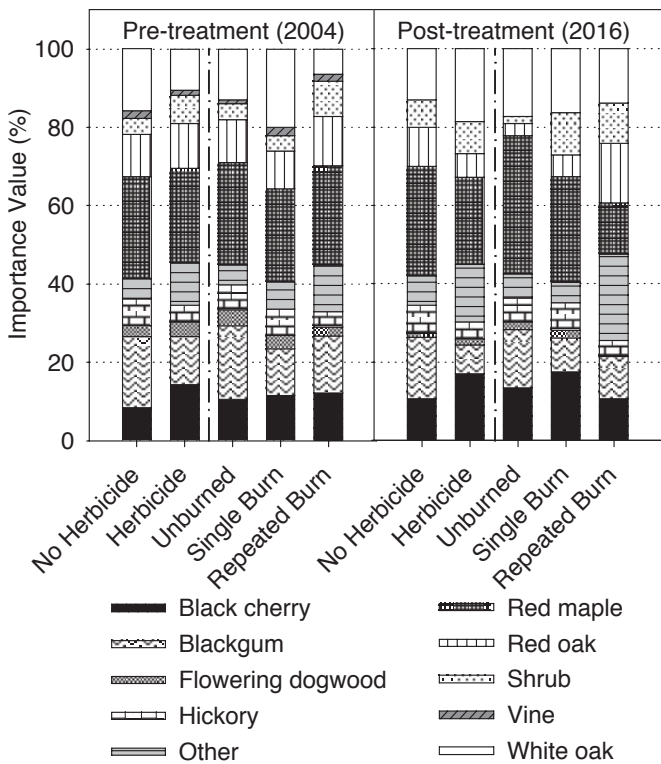


Figure 3—Sapling (height > 4.5 feet and DBH < 4.5 inches) importance value (sum of relative density and relative basal area) proportion by species group before and after rehabilitation treatments of stem injection and prescribed burning in high-graded oak-hickory stands of the Arkansas Ozarks.

injected trees. Red oak importance values were 2-3 times higher for repeated burn compared to unburned and single burn treatments (fig. 3). For repeated burns, red maple abundance was only a third of that of unburned and single burn treatments, indicating that density reduction was mainly for this species (fig. 3). Although post-treatment red maple sapling height did not differ among burn-only treatments ($P = 0.08$), post-treatment height for repeated burn was respectively half and two thirds that of unburned and single burn treatments for herbicide treated units. Pre-treatment red maple sapling height did not differ among rehabilitation treatments ($P > 0.3$). Shrubs such as winged sumac (*Rhus copallinum*), American beautyberry (*Callicarpa americana*), Carolina buckthorn (*Frangula caroliniana*), and rusty blackhaw (*Viburnum rufidulum*) that were at least 4.5 feet tall and less than 4.5 inches in DBH were more prevalent in the single and repeated burns (fig. 3). However, these species were twice as tall in unburned units compared to single and repeated burns. Thus, repeated burns at 3-year intervals appear to be an effective treatment in reducing abundance of red maple, and when combined with stem injection is capable of reducing red maple heights to more manageable levels.

Number of established seedlings, at least 1 foot tall, and their mean heights did not differ between herbicide ($P > 0.34$) or among burning treatments ($P > 0.7$) and there were no significant pre-treatment interactions ($P > 0.30$). Also, no difference in seedling density or height was found among rehabilitation treatments in 2016 ($P > 0.15$). Seedling composition was similar among all pre- and post-rehabilitation treatments, indicating a well-developed and abundant reproduction pool in which oaks shared dominance with more shade-tolerant, fire-sensitive species such as red maple, black gum, black cherry, and flowering dogwood (table 4; fig. 4).

CONCLUSIONS

High-grading practices in oak-hickory forest type of the Arkansas Ozarks have left understocked stands with undesirable, low-quality trees and species in the overstory. A legacy of fire suppression and high incidence of pests promoted the development of recalcitrant mesophytic understories with red maple, black gum, black cherry, and flowering dogwood as main components. These conditions differ drastically

Table 4—Seedling attributes in 2004 and 2016 by rehabilitation treatment of stem injection (treated and untreated) and prescribed burning (single, repeated at 3-year interval, and untreated) in high-graded oak-hickory stands of the Arkansas Ozarks

Treatment	Density (trees per acre) ^{a, b, c}	Height (feet) ^{a, b, c}
2004 observation^d		
Herbicide	1288 (170)a	2.7 (0.1)a
No Herbicide	1563 (242)a	2.5 (0.2)a
Single Burn	1282 (276)A	2.8 (0.1)A
Repeated Burn	1169 (134)A	2.8 (0.2)A
No Burn	1826 (291)A	2.3 (0.1)A
2016 observation^d		
Herbicide	1122 (200)a	2.2 (0.1)a
No Herbicide	1277 (181)a	2.1 (0.1)a
Single Burn	1289 (237)A	2.2 (0.1)A
Repeated Burn	1449 (267)A	2.3 (0.2)A
No Burn	882 (145)A	1.9 (0.1)A

^a Mean values with standard error in parenthesis.

^b Means followed by the same small letter within a column are not significantly different at the 0.05 level.

^c Means followed by the same capital letter within a column are not significantly different at the 0.05 level.

^d 2004 = pre-treatment; 2016 = 12 years post-treatment.

from the historically oak-dominated open stands that covered the area (Foti, 2004, Nowacki and Abrams 2008). In rehabilitating these stands, stem injection of herbicides reduced black gum, red maple, and hickory from the overstory and allowed for greater recruitment of white oak. Repeated burning, at 3-year intervals, increased red oak sapling abundance at the expense of red maple, and stem injection removed large red maple saplings that would have escaped fire in repeated burn only treatments. Open conditions created by repeated burning promoted shrubs such as winged sumac and American beautyberry but kept their height in check. Repeated burning continues to improve the composition and structure of these high-graded stands and appears to shift the mesophication trend towards more abundance of oak, especially red oak that has been in steady decline in the Arkansas Ozarks.

ACKNOWLEDGMENTS

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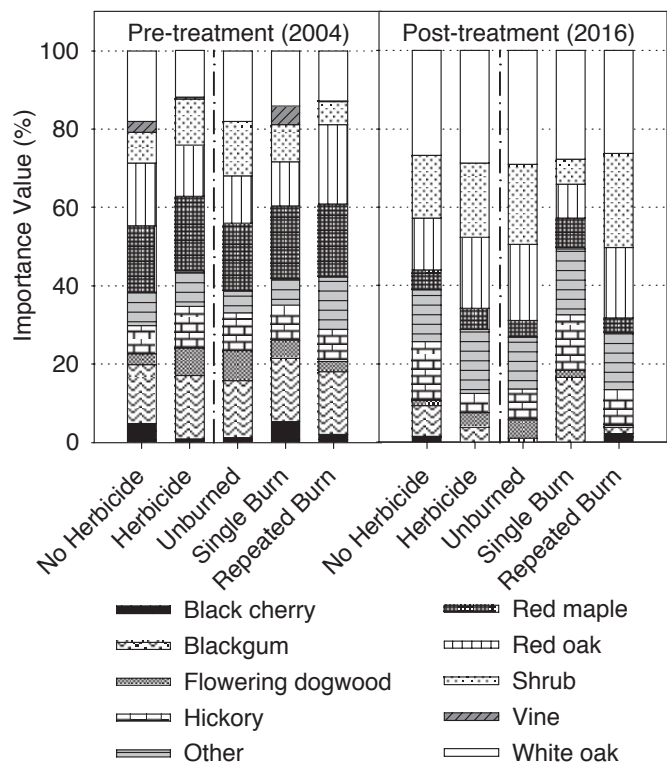


Figure 4—Seedling (1 foot < height < 4.5 feet) importance value (sum of relative density and relative basal area) proportion by species group before and after rehabilitation treatments of stem injection and prescribed burning in high-graded oak-hickory stands of the Arkansas Ozarks.

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LONG-TERM OVERSTORY TREE QUALITY MONITORING THROUGH MULTIPLE PRESCRIBED FIRES IN EASTERN DECIDUOUS FORESTS

Shannon Stanis and Mike R. Saunders

Abstract—A long-term study of the effects of repeated prescribed fire on overstory tree quality is being installed in 20 oak-dominated stands across central Indiana. Each stand is managed with even-aged regeneration systems and designated to receive prescribed fires on 3-5 year cycles until final harvest. Prior to the initial prescribed fire, 20 average quality and 10 highest quality trees in each stand are measured for grade and defects below breast height. During prescribed burns, fire temperatures are monitored with thermal paints and thermocouples at variable heights on some stems. Trees are revisited after prescribed burns to measure initial scorch height; assessment of wounding from fire is conducted two growing seasons after. Here we report on the first post-burn assessment of trees within two stands of this study. Although 37 percent of sample trees sustained fire damage ($n = 59$), no trees had died or received a reduction in U.S. Forest Service tree grade after 2 years.

INTRODUCTION

Prescribed fire is increasingly used in eastern hardwood forests to meet multiple management objectives. Prescribed fire is sometimes used to promote regeneration of oak species (*Quercus* sp.) within shelterwood systems since alternative even- and uneven-age regeneration methods commonly fail (Brose and others 1999, 2013). Many managers, however, hesitate to implement prescribed fire due to a perception that fire injury to overstory trees lowers the quality and volume of timber (Dey and Schweitzer 2015). This perception is likely based on research from wildfire damage (Loomis 1973, Nelson and others 1933, Wood 2010). Wildfire and prescribed fires are not the same; prescribed fires burn at a much lower intensity and more patchily than wildfires, and often leave trees untouched. In fact, foundational work on the effects of prescribed fire on oak timber suggests that there may be minimal volume and economic losses within 5 to 15 years of harvest (Marshall and others 2014, Stambaugh and Guyette 2008, Wiedenbeck and Schuler 2014).

Yet land managers need more accurate and reliable understanding of the effects of prescribed fire on timber quality across the range of site conditions and species, and with varying intensities and timings of prescribed fires. We installed a long-term monitoring study on 20 stands in southern Indiana with the objective of relating prescribed fire behavior to fire scar formation on multiple species groups, from the inception of a periodic, site preparatory prescribed fire regime until final harvest of the overstory trees 10-20 years later. Here, we describe

the methodology of the long-term monitoring study and report upon initial responses of overstory trees in two stands that received the first round of implemented prescribed fires within the study. Specific to these first two stands, we hypothesize that topography of the stand will influence fire behavior. We predict that prescribed fire intensity (i.e., temperature) will be higher on drier, south-facing aspects and that trees on these aspects have more frequent damage. We also hypothesize tree species will differ in their incidence of fire damage.

METHODS

Study Sites

The study is located on both the Hardwood Ecosystem Experiment (HEE), within the Yellowwood and Morgan-Monroe State Forests near Bloomington, IN; and the Naval Service Warfare Center Crane Division (NSWC-Crane) in Martin County, IN. The HEE is a long term, landscape level experiment designed to study the ecological, economic, and social impacts of various timber management practices in Indiana (Swihart and others 2013), whereas NSWC-Crane is part of a long term experiment testing the efficacy of hybrid silvicultural systems to regenerate oak species and to increase structural variability and ecological resiliency. Pertinent to this study, both experiments use prescribed fire as a management tool to promote oak regeneration.

HEE—The HEE lies upon the unglaciated Brown County Hills Section of Indiana's Highland Rim Natural Region. Homoya and others (1985) characterize the region as

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deeply dissected uplands, underlain by siltstone, shale, and sandstone with well-drained, acidic, silt loam soils in the Berks-Gilpin-Weikert association. Upland communities are dominated by oak-hickory species (*Quercus-Carya*), specifically black oak (*Q. velutina*), white oak (*Q. alba*) and shagbark hickory (*C. ovata*), with chestnut oak (*Q. prinus*) on the more xeric sites. Greenbriar (*Smilax* spp.), low growing shrubs (e.g., *Gaylussacia baccata* and *Vaccinium vacillans*), and sedges are characteristic. Small, high gradient ephemeral streams are common throughout and larger streams are primarily medium to low gradient. Mesic north- and east-facing slopes and ravines are commonly dominated by tulip poplar (*Liriodendron tulipifera*), red oak (*Q. rubra*), American beech (*Fagus grandifolia*) and sugar maple (*Acer saccharum*). Dry ridges generally have lower herbaceous species richness compared to mesic sites (Jenkins 2013). There is low incidence of surface fires on these sites within the last 50 years¹ Therefore, most sites have a predominant midstory of sugar maple, beech, and other more tolerant species.

The HEE is comprised of nine 800-1,000 acre management units each with a central 200 acre research core. Units were randomly assigned one of three management regimes: even age with silvicultural clearcuts or two- or three-stage shelterwoods; uneven aged with single-tree selection and several 1.0-5.0 acre patch cuts; and no harvest as an experimental control. Prescribed fire is implemented on 12, approximately 10 acre stands within the even aged management units. The initial fires began in two stands in spring of 2015 on a 4-year rotation with the goal of burning each stand three times before either a silvicultural clearcut or shelterwood harvest in winter 2027-28. Additional stands have been burnt in fall (October-November) and spring (March-April) burn seasons every year since. Over time, these prescribed fires should reduce midstory density and forest floor thickness, expose mineral soil, and eventually lead to a pool of advanced oak regeneration prior to harvest.

NWSC-Crane—The NSWC-Crane was established by the U.S. Navy in 1941 on abandoned farms, pastures and small woodlots; soon after, in the 1950s, active management began to restore the forest and control erosion caused by farm abandonment and the base construction (Osmon 2013). The NSWC-Crane lies within the unglaciated Crawford Upland and Escarpment Sections of the Shawnee Hills Natural Region and is characterized by rugged sandstone or sandstone-capped hills and rockhouses with underlying limestone features occasionally exposed (Homoya and others 1985). Soils are commonly well-drained, acidic silt loams

in the Wellston-Zanesville-Berks Association (Homoya and others 1985). Species composition is very similar to the HEE, although a bit more xeric, with black, white, chestnut, scarlet (*Q. coccinea*) and post (*Q. stellata*) oaks, and pignut (*Carya glabra*) and shagbark hickories. More mesic sites will include American beech, tulip poplar, red oak, sugar maple, and black walnut (*Juglans nigra*). Also like the HEE, sugar maple, beech other shade-tolerant shrub and tree species form a persistent midstory in most stands.

The NWSC-Crane experiment consists of twenty 15-25 acre stands, comprising four replicates of five different silvicultural treatments: i) a two-stage expanding group shelterwood; ii) a three-stage expanding group shelterwood; iii) a two-stage expanding group shelterwood with prescribed fire; iv) a three-stage expanding group shelterwood with prescribed fire; and v) an unharvested, unburnt control. Stands are managed on a 100-year rotation with a 10- year cutting cycle, but cut only during the first half of the rotation. During these entries, approximately 20 percent of the area receives overstory harvest (either 50 percent or 90 percent basal area removal), with additional areas receiving mechanical midstory removal cuts, and/or a final overstory removal cut depending on cycle. Prescribed fire is used in iii) and iv) on a 5-year return interval with goals similar to the HEE, i.e., to reduce density of midstory and other understory competitors and improve seed bed characteristics for accumulation of oak regeneration prior to final overstory harvest. Fires in the NWSC-Crane study occur only in the fall burn season (November) and began in 2014 on the first replicate.

Tree Monitoring

Within each stand of both experiments, an 82 x 82-foot grid is installed with unique names and GPS locations for grid points. Twenty randomly selected grid points, hereafter referred to as fuel points, are monitored for the effects of prescribed fire on dead and live fuels as it relates to fire intensity and consumption. A tree of “average” quality (AQ tree) is randomly chosen near each fuel point (within 39.4 feet) prior to the first prescribed fire in the stand. The tree must meet the following characteristics: (1) be an oak, hickory, maple or tulip poplar species; (2) be >10 inches diameter at breast height (d.b.h.) and have the potential for commercial product (i.e., not obviously cull); and (3) be single-stemmed in the first 10 feet of height. In addition, 10 “high” quality trees (HQ trees) are selected across each stand independent of the grid points. These trees are deemed to be of the highest economic value of all stems in the stand; they are commonly, but not always, large white oak and of U.S. Forest Service (USFS) tree grade 1 (Hanks 1976, Miller and others 1986), externally viewed as marketable prime lumber or veneer stumpage.

¹ Personal communication. 2015. Jim Allen, Property Manager, Yellowwood and Morgan Monroe State Forest, 772 South Yellowwood Road, Nashville, IN 47448.

GPS location, d.b.h., strata, species, condition (e.g., live, leaning, broken tops, etc.), and USFS tree grade are recorded for each tree. All wounds below breast height are recorded by measuring the position on the stem, size, and type (seams, cankers, old branch stubs, etc.). A picture of each tree is taken from plot center (for AQ trees) or 33 feet from the south side of the tree (for HQ trees); close-up pictures of each wound are also taken. Height of bark discoloration (i.e., scorch height) on trees is recorded within 3 days after a prescribed fire.

Timber quality is reevaluated through a damage and wound assessment two growing seasons after a prescribed fire, and again just prior to each prescribed fire the stand will receive. During these assessments, we follow the language outlined by Smith and Sutherland (2001) where damage is an actual or perceived loss of property, value, or usefulness, while injuries are an impairment or loss of function. Wounds are a type of injury with a physical disruption of living tissues and are also considered damaged as they may lower the grade of the resulting timber (Rast and others 1973). Bark discoloration indicates that a tree has been through a fire and may indicate the tree has sustained an injury (Smith and Sutherland 2001). Therefore, we view bark discoloration as damage, as trees with indicators of fire are assessed with less value by timber buyers, but not an injury until an exterior wound is visible.

Any newly acquired wounds are categorized as caused by fire (bark slough, catface, seams, ovals, decay fungus), not caused by fire (no darkening of bark on or near wound), or other (previously missed, previously hidden by bark but now exposed). Bark slough is dead bark disconnected from the bole of the tree. Catfaces (triangle-shaped, open at base of tree) and ovals (oval-shaped, closed at base of tree) have distinct wound ribs. Bark discoloration is categorized as burn, char or scorch, classified according to a combination of Loomis (1974) and Nelson and others (1933) from most to least severe. Burn is bark entirely blackened even with the bottom of bark fissures, and at least partially consumed on the tops and edges of ridges. Char is blacked bark without completely reduced ridges, with sharp edges intact, and fissure bottoms lightly browned or not discolored. Scorch is bark with ridges browned or incompletely blackened, discontinuous pitting of bark ridges, and non-discolored fissures. New pictures are taken of each tree and all wounds and damage at each measurement. Trees are graded again with consideration to both old wounds and any newly acquired damage, fire or otherwise.

Fire Intensity

For each prescribed burn in a stand, fire intensity is measured in several ways. First, two aluminum

pyrometers containing six types of Tempilaq G® thermal indicating liquids (175°F, 250°F, 325°F, 400°F, 600°F, and 800°F) are installed at all grid points in the stand and collected and read within a day after the burn. Second, a single type K thermocouple attached to a 1-channel HOBO datalogger (model US100-014M) is deployed at each of the 20 fuel points in each stand, set to record temperature every 3 seconds during the burn. Third, after each burn and during pyrometer collection, a visual estimate of average tree scorch height (<1 feet, 1-2 feet, 2-4 feet, 4-6 feet, 6-8 feet, 8-10 feet, >10 feet), percent charred litter surface (to nearest 10 percent) and percent mineral soil exposure (to nearest 10 percent) are recorded. Together, these data provide a spatially-explicit model of fire intensity across the entire stand.

To increase the resolution of the fire intensity estimates for tree monitoring, aluminum pyrometers are installed on the uphill and downhill sides of each AQ and HQ tree. In addition, thermocouples are attached to the bole of five randomly selected AQ trees and five randomly selected HQ trees in each stand at heights of approximately 1 feet, 3 feet, 5 feet, and 7 feet, perpendicular to the slope. Thermocouples are attached to a 4-channel HOBO datalogger (model UX120-014M) to record every 3 seconds during the burn.

Weather conditions (temperature, wind speed, humidity) at time of ignition are recorded using standard protocols for fire in the region.

Analysis

This paper presents analysis for only two stands that both received prescribed fire in spring of 2015. Differences between average pyrometer temperatures between the two stands were tested using a Welch two-sample t-test, while differences in the frequency of fire damage (of any type or severity) on AQ trees (1) between the two stands and (2) between tree species groups were tested using chi-square (χ^2) tests. For the latter, we defined five tree species groups: (i) red oak – northern red and black; (ii) maple – sugar and red; (iii) white oaks – chestnut and white, (4) pignut hickory, and (5) tulip poplar. We also tested for differences in frequency of fire damage between units for the HQ trees using a χ^2 test.

In order to determine the relationship between fire temperature and observed AQ and HQ tree damage, we conducted logistic regressions with presence of fire damage on a tree (of any type or severity) as the response variable and pyrometer-derived fire temperature at the tree as the predictor variable. All tree grades were unchanged following prescribed burns and therefore no analysis of tree grade was conducted. All analyses were conducted in R 3.1.4 (R Core Team 2017) and, for all tests, we set alpha at 0.05.

RESULTS AND DISCUSSION

Prescribed Fire

The Indiana DNR Division of Forestry—Fire Headquarters implemented the first prescribed fires of the study on March 31, 2015, on two adjoining stands on the HEE – U3-11 and U3-12. U3-11 is primarily northeast-facing, but includes some north and northwest aspects. U3-12 is primarily southeast-facing, with some east slopes and small areas of northwest slopes (fig. 1A). Firefighters used a backing fire technique along the western ridgetops and several strip fires within the stand to help control flame lengths. The eastern and northern edges of the units follows a ravine. Relative humidity was 35 percent and wind speed was 6-8 miles per hour. It had been five days since the last rain over half an inch. Risk of escape over hand lines was 5-10 percent. The fire received a severity score of 76, placing it in the “basic” prescribed fire category.²

Both units had areas that exceeded the 800 °F pyrometer reading, but as averaged throughout the units, pyrometer temperature of the fire was higher in the southeast facing U3-12 ($466 \pm 22^\circ\text{F}$ [mean $\pm$ standard error]) than in northeast facing U3-11 ($332 \pm 19^\circ\text{F}$) ($t=-4.66$, $df=136$, $P<0.001$) fig. 1B). Scorch heights mirrored temperature patterns throughout the stands. Heights were generally 3-4 feet throughout U3-12 with heights of 4-6 feet in the hottest areas and a few

instances exceeding 10 feet. Consistently throughout U3-11, scorch heights were 1-3 feet, with only a few instances of scorch above 4 feet.

Observed Tree Damage

Overstory tree damage was highly variable, although all sample trees appeared to be in good health and alive after 2 years with no change in crown class or strata. Overall, 37 percent of sample trees ($n = 59$) showed presence of fire damage through bark discoloration or new wounds; average quality (AQ) trees experienced higher damage (44 percent; $n = 39$) than high quality (HQ) trees (25 percent; $n = 20$). Very few new wounds were visible externally. There were six new seams (three on the same tree and the others distributed on three separate trees), eleven instances of bark slough (four instances where bark sloughing was the entire affected area and seven instances where only some of the discolored bark was sloughing), and three examples of decay fungus growing on top of burned bark (three separate individuals). New catfaces or ovals were not observed. Many sample trees sustained bark discoloration of multiple types (e.g., tree with burn, char, and scorch) with 12 instances of burn, 13 of char, and 13 of scorch; burn was generally lowest on the bole with char and then scorch above (table 1). None of the damage resulted in a deduction in USFS tree grade as damage and discoloration were mostly contained in one face and/or <6 feet high if on a second face (Hanks 1976).

² Indiana Division of Forestry. 2015. Fire Headquarters. Unpublished report. On file with: Indiana Division of Forestry, 1217 Deam Lake Rd., Borden, IN 47106.

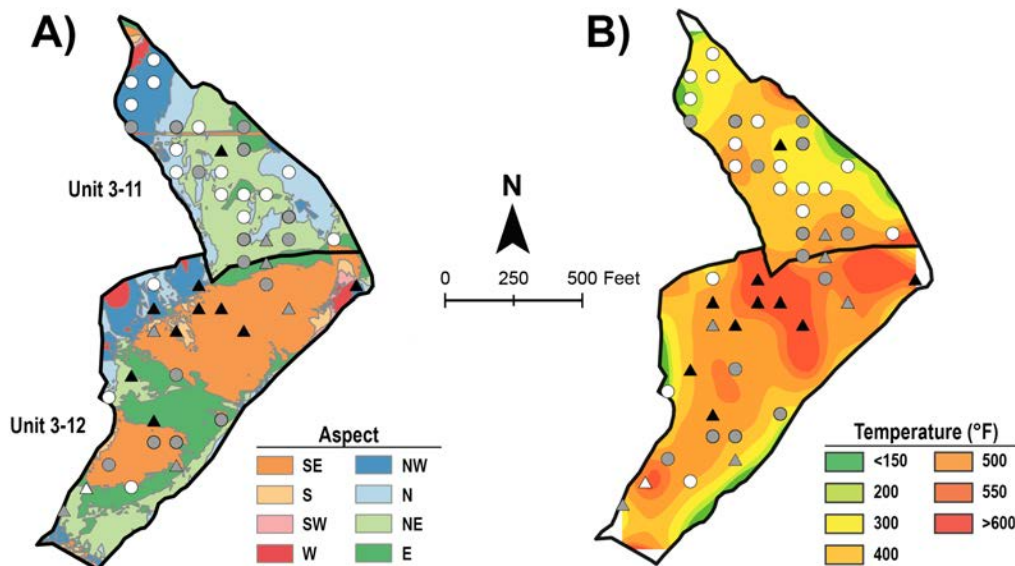


Figure 1—Maps of Units 3-11 and 3-12. Aspect map (A) was GPS-derived; fire intensity map (B) was a kernel estimate derived from pyrometer readings at each grid point. Sample trees are depicted with fire damage (triangles) or without (circles); color of symbol indicates upslope pyrometer readings of <200°F (white), 200-400°F (grey), and >400°F (black).

Table 1—Mean (standard error) length, end height, and width of damage types 2-year post-fire on the Hardwood Ecosystem Experiment sites

Type	<i>n</i>	Length	End height	Width
<i>inches</i>				
Scorch	13	24.9 (3.7)	48.8 (8.4)	24.0 (2.7)
Char	13	26.0 (3.3)	49.7 (6.7)	27.2 (2.9)
Burn	12	38.5 (6.2)	37.3 (6.2)	36.9 (3.4)
Seam	6	5.8 (0.9)	23.9 (3.9)	1.8 (0.6)
Bark slough	4	25.1 (9.6)	25.1 (9.6)	16.6 (4.4)
All types	48	26.2 (2.5)	41.1 (3.0)	24.7 (2.1)

End height is the vertical height at which damage ends on the uphill side of the tree.

After this first prescribed fire, we estimate that there are minimal changes to timber quality and value as injury sizes are small and are generally under 41 inches tall (table 1). Other studies have reported similar findings (Marshall and others 2014, Stambaugh and Guyette 2008, Wiedenbeck and Schuler 2014). However, it is very likely that underneath the discolored (i.e., charred and/or scorched) bark that the cambium has died and new wounds are compartmentalizing (Loomis 1973, Nelson and others 1933, Shigo 1984, Smith and Sutherland 1999)—wounds that are not yet observable after 2 years. The relationship between external discoloration and internal damage is not easy to establish, as trees with discolored bark have been dissected to reveal no internal fire scars (Smith and Sutherland 1999).

The two stands differed in the amount of AQ trees with fire damage; 75 percent (*n* = 20) had damage in U3-12 while only 22 percent (*n* = 19) trees had damage in Unit 3-11 ($\chi^2 = 16.5$; *DF*=1; *P*<0.01). In Unit 3-12, 40 percent (*n*=10) of HQ trees had damage while, in Unit 3-11, 10 percent (*n* = 10) had damage, although these sample sizes were too small to determine statistical significance ($\chi^2 = 2.4$; *df*=1; *P* = 0.12). Aspect likely drove these differences (fig. 1a); aspect influences solar radiation levels where more exposed terrain has drier fuels and increased fuel consumption, temperature, and fire scarring (Dickinson and others 2016, Schwemlein 2006, Stevenson and others 2008). For example, four HQ trees that sustained damage in U3-12 were located in southeast facing, quite exposed portion of the stand; they experienced the highest temperatures at surrounding grid points and on their upslope side (fig. 1b). Further, our northeast-facing stand (U3-11) likely had further reductions in temperatures due to a slight shift in composition towards more mesic species in some parts of the stand. For example, the far northwest corner of U3-11 had significant amounts of red and sugar maple saplings, which creates a fuelbed not conducive to prescribed fire on these sites (Dickinson and others 2016).

We observed some evidence of differences in the frequency of fire damage among species groups for AQ trees ($\chi^2 = 7.6$; *DF*=4; *P*=0.11) (table 2), where all tulip poplars sustained fire damage (*n*=4), but only one hickory sustained fire damage (*n*=5) (table 2). Oaks, of which we had a far greater larger sample, were intermediate in their responses; 36 percent of red oaks (*n*=14) and 50 percent of white oaks (*n*=12) sustained damage. Bark thickness is the likely explanation for these species-specific patterns, having been repeatedly shown to be the greatest predictor of temperature

Table 2—Mean size (range) and frequency of fire damage by species group 2-years post-fire on the Hardwood Ecosystem Experiment sites

Species group	<i>n</i>	D.b.h.	Damaged (percent)
<i>----- Average Quality Trees -----</i>			
Red oak	14	21.0 (12.5 – 28.9)	35
White oak	12	17.5 (13.2 – 22.0)	50
Pignut hickory	6	18.9 (14.7 – 25.9)	16
Maple	3	12.7 (12.3 – 13.1)	33
Tulip poplar	4	21.6 (14.7 – 28.8)	100
Total	39		37
<i>----- High Quality Trees -----</i>			
Red oak	1	21.7	0
White oak	18	23.6 (14.6 – 31.5)	28
Pignut hickory	1	22.5	0
Maple	0	–	–
Tulip poplar	0	–	–
Total	20		25

d.b.h. = diameter at breast height.

– = No sample

reached at the cambium and varying strongly by species and size (Hare 1965, Hengst and Dawson 1994, Martin 1963, Pinard and Huffman 1997, Vines 1968). In the Central Hardwood Region, species in the white oak group have the thickest bark, followed by the red oak group, while beech, poplar, maple and hickory species generally have thinner bark (Sutherland and Smith 2000). Unexpectedly, the hickories in this study were resistant to fire damage; this was likely driven by their large size (table 2), as bark thickness and resistance to fire wounding increase as a tree grows in diameter (Harmon 1984, Hengst and Dawson 1994).

We had some evidence for a fire temperature (i.e., intensity) threshold, but this threshold differed by sample population. AQ trees were more likely than not to sustain damage if their upslope pyrometer temperatures exceeded 407°F ($p < 0.01$) (fig. 2, table 3); the average upslope pyrometer temperature for AQ trees that sustained damage was 564°F $\pm$ 52 while the average temperature for trees without damage was 230°F $\pm$ 35. HQ trees had a damage threshold somewhere between

400°F and 600°F; logistic regression did not converge because of a bifurcation of the sample. All five HQ trees that sustained damage had upslope pyrometer readings of 600°F or 800°F, while the undamaged trees had readings of 400°F or below (fig. 2). These temperatures are similar to those reported by Hengst and Dawson (1994) in their study of simulated fire in 40-year-old plantations. They reported for bur oak (*Q. macrocarpa*), a species with similar bark thickness to white oak, that surface temperatures of 437°F (225°C) would result in cambial temperatures exceeding 140°F (60°F) and cause cambial death and subsequent scar formation. Thinner barked trees, either because of species differences or smaller tree diameter, would obviously reach lethal cambial temperatures at a much lower external temperature (Hengst and Dawson 1994, Pinard and Huffman 1997). Furthermore, longer residence time at lower temperatures could also lead to cambial death and injury (Hare 1961, Vines 1968); future work in this study will use thermocouple-derived data to categorize residence time of fire and relate that to injury. Nevertheless, our pyrometer readings are useful to measure external temperatures and their relations to fire scar formation across the species groups, particularly as death to the cambium becomes evident over time.

CONCLUSION

We found supporting evidence that the aspect of the stand influences fire behavior and tree damage frequency and severity. Although we saw indications of fire damage from these low-intensity prescribed burns, it is still unclear whether this damage will result in long-term injury or loss of economic value. Eventually, this study will monitor between 500-600 trees to build relationships among edaphic factors (i.e., topography and aspect), fire behavior, external indicators of fire damage, and long term development of internal injuries. Further, these models will be validated by tracking sample trees through the milling process upon harvest in 15-25 years to determine the exact effects of multiple prescribed fires on lumber quality and volume. These models can then be used by foresters and other land managers to determine the economic tradeoffs of prescribed fire in relationship to both damage to residual timber and the costs of regeneration efforts for oak species.

Lastly, many recent studies on prescribed fire effects on hardwoods give one general estimate of fire scar indication, calling any extent of discoloration as “char” or “scorch” without specific differences to clarify the severity of the discoloration seen within the stands. The discrepancy between the frequency and severity of fire damage in the initial two stands of our study

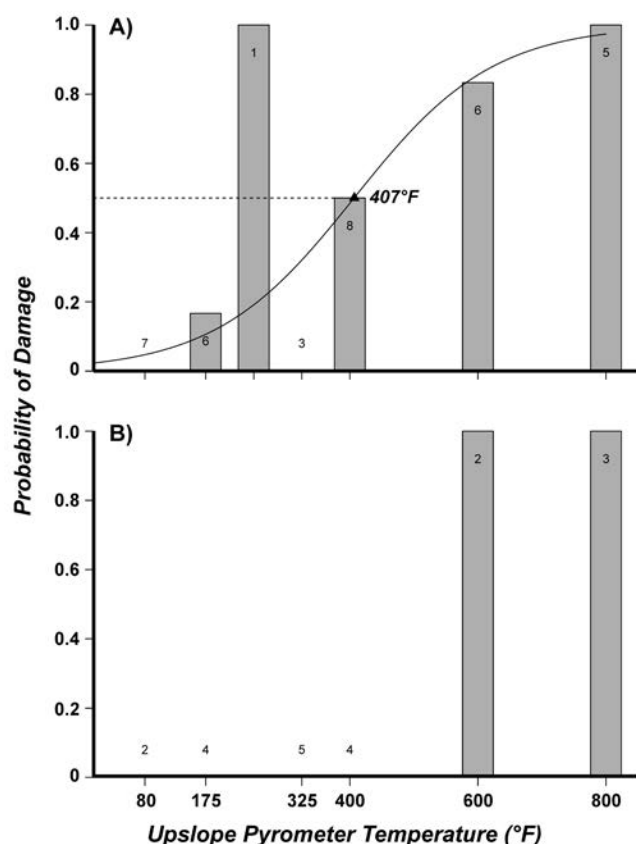


Figure 2—Proportion of damage by upslope pyrometer temperature for AQ (A) and HQ (B) trees. Numbers in bars represent sample size. Probability of damage was fit by logistic regression for AQ trees; 50 percent probability is indicated with a triangle at 407°F ($p < 0.01$).

Table 3—Parameter estimate, standard error, z-value, and p-value of logistic regression relating upslope pyrometer temperature reading (in °F) to fire damage observed 2 years post-fire on AQ trees

Parameter	Estimate	SE	Z	P
Intercept	-3.74	1.2	-3.10	<0.01
Temperature	0.01	0.003	3.13	<0.01

illustrates the need for the careful documentation and descriptions of fire damage and fire scars implemented in our long term post fire assessments, especially as they relate to fire scar formation (Loomis 1973). This study will continue to monitor the fire damage severity to understand how it related to fire scar formation with attention to the fire behavior surrounding the trees.

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PATTERNS OF OVERSTORY MORTALITY IN A SHELTERWOOD-BURN CENTRAL APPALACHIAN FOREST

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Abstract—Prescribed fire is a tool used to regenerate and sustain mixed-oak forests in the central Appalachian region. Two specific applications for prescribed fire are site preparation and the reduction of oak competition after the first removal cut in a shelterwood. This latter use is called the shelterwood-burn method. However, prescribed fire presents additional mortality risk to the stand's overstory trees. In a shelterwood-burn study on the Fernow Experimental Forest in West Virginia, three individual tree factors—species, crown class, and dbh—and three stand manipulation treatments were examined to determine their effects on overstory tree mortality from both background and fire induced causes. A shelterwood treatment and a shelterwood-burn treatment, both site-prepared with prescribed fire, had significant differences in combined background and fire mortality when compared to an unburned, uncut control. Considering fire mortality alone, there was no difference in survival rates between the shelterwood and the shelterwood-burn treatments over the entire study period. Tree diameter influenced both natural mortality and fire-induced mortality. Crown class, species, and treatment were factors in background mortality only.

INTRODUCTION

The change in perception of the disturbance role that fire plays in forests of the Eastern United States has led to an increased utilization of prescribed fire as a tool for regenerating upland oak forests (Brose 2014). A silvicultural prescription that has arisen out of multiple investigative studies of fire in oak forests is the shelterwood-burn technique described by Brose and others (1999a) and Brose and others (1999b). This technique is specifically designed to address lack of recruitment of oak advance regeneration into the overstory. In a shelterwood-burn, prescribed fire is used as a release burn after the first shelterwood removal cut (post-shelterwood burn) to control competition (Brose and Van Lear 1998, Miller and others 2017) and it may optionally be used to help prepare the seedbed for establishment of oak seedlings when advance regeneration is not sufficient (Schuler and others 2010, 2013).

Previous research on prescribed fire in oak forests has examined its effects on advance regeneration in an attempt to better understand prescribed fire's potential use to promote oak forests (Brose 2013, Brose and Van Lear 1998, Brose and others 1999b,

Jackson and Buckley 2004, Van Lear and others 2000). Overstory mortality has been examined in conjunction with advance regeneration mortality from fire (Arthur and others 2015, Barnes and Van Lear 1998, Huddle and Pallardy 1996, Hutchinson and others 2005, Paulsell 1957, Van Lear and others 2000) or reported solely (Wendel and Smith 1986). Due to the relatively recent formulation of the shelterwood-burn technique, examination of the effect of prescribed fire on the survival of the overstory after the post-shelterwood burn has been limited (Brose and Van Lear 1999). In a shelterwood-burn prescription for oaks, roughly 50 percent of the basal area of the overstory is still present when the post-shelterwood burn is conducted. Optionally in a shelterwood-burn, prescribed fire may be used for site preparation to promote advance regeneration before the first cut of the shelterwood. For each application of prescribed fire, there is a risk of overstory tree mortality and it is a concern that should be addressed as these losses could impact mast production, reduce yields of valuable timber, or potentially change forest composition. The cumulative effects of multiple fires is expected to increase the risk. Our objective is to determine the effect that prescribed fire in combination with individual tree characteristics

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have on overstory tree mortality in stands treated with a shelterwood-burn prescription, first looking at combined background and fire induced mortality, then examining the effect of the post-shelterwood burn on the shelterwood treatments.

METHODS

Site Description

The study site is located in West Virginia on the Canoe Run watershed of the Fernow Experimental Forest (39.03° N, 79.67° W). Mean annual precipitation is 57 inches, distributed evenly throughout the year and mean annual temperature is 49 °F which ranges from a monthly mean of 27 °F in January to a monthly mean of 69 °F for July. The study site ranges from 1920-2200 feet in elevation. Five species comprised over 80 percent of the overstory basal area at the start of the study: northern red oak (*Quercus rubra* L.), chestnut oak (*Q. prinus* L.), white oak (*Q. alba* L.), yellow-poplar (*Liriodendron tulipifera* L.) and red maple (*Acer rubrum* L.). Four additional species contributed roughly another 10 percent: American beech (*Fagus grandifolia* Ehrh.), sourwood (*Oxydendrum arboreum* L.), sweet birch (*Betula lenta* L.), and sugar maple (*Acer saccharum* Marsh.).

Experimental Design

The study was established in 1999; for a more detailed description of the larger study please see Schuler and others (2013). Overstory trees were measured periodically on 24 1/2- acre plots. Four plots were assigned as controls, 10 plots were treated as a two-cut shelterwood without a burn (shelterwood) and 10 plots were treated as a two-cut shelterwood-burn (shelterwood-burn). Trees with diameter at breast height (dbh) >5 inches are permanently tagged and periodically remeasured for dbh and status (alive or dead). Mortality status is noted as natural, cut, destroyed, or fire damage. Trees with a mortality status of “cut” or “destroyed” were removed from this analysis. Post-fire inventories were conducted after all burns.

Two prescribed fires were applied to the shelterwood and shelterwood-burn plots prior the first removal cut in each treatment. The first prescribed fire was applied in April 2002 with some plots burned in April 2003 due to unfavorable conditions in 2002. The second prescribed fire occurred in April 2005 with all plots except the control plots burned. The shelterwood cut was performed during the winter of 2009-2010 and reduced the average overstory basal area of both shelterwood treatments from 145 square feet per acre to 62 square feet per acre (Schuler and others 2013). Trees were felled by the Fernow Experimental Forest logging crew with care taken to protect the boles of the residual trees. The post-shelterwood prescribed fire occurred in

the spring of 2014 in only the shelterwood-burn plots. Burn prescriptions were written to result in fires of low to moderate intensity as the continued production of timber products was an objective of the study.

Data Analysis

To determine the effect of prescribed fire on mortality, analysis was divided into two stages, both of which used the Cox Proportional Hazards (CPH) Model (Cox 1972). This model is a better alternative to the more traditional use of logistic regression models for survival analysis because it permits analysis of time to the event vs. the fixed time period required for logistic regression. Parameter estimation occurs simultaneously over the entire period of study and there is no need to consider mortality for arbitrarily fixed time periods, such as 5- or 10-year mortality. Also, the CPH model accounts for the removal of a subject from the sample population due to death (which is one form of censoring).

$$\lambda(t; z) = e^{(z\beta)} \lambda_0(t) \quad (1)$$

where

t = time

z = a 1xp vector of covariates

β = a px1 vector of unknown parameters

λ₀(t) = the baseline hazard function, an unspecified but non-negative function

In the first stage of analysis, combined background and fire-induced mortality was the event analyzed using the CPH model. Four variables were of interest in the model: treatment, species, crown class, and dbh. There were three levels of treatment: control, two-cut shelterwood (two site-preparation burns and first removal cut), and two-cut shelterwood-burn (two site-preparation burns, first removal cut, and one post-shelterwood burn). Considering treatment with these three levels as a factor affecting combined mortality allowed examination of differences in mortality between the unburned control compared to the two shelterwood treatments which were burned. A significant difference among the treatments would indicate additional mortality occurred due to prescribed burning. With a large number of species present and an interest in determining the survival of oaks, the species considered were chestnut oak, northern red oak, white oak, and all others. Due to a conservative designation of the dominant crown class in data measurement, dominant and codominant trees were grouped together, with intermediate and suppressed trees each assigned as unique levels of crown class. All trees >5 inches dbh are considered

overstory trees and included in the survival analysis. In the second stage, the CPH model was again used for survival analysis, however only fire-induced mortality was considered as the event of interest and the unburned control plots were removed from the sample, which permitted consideration of whether the post-shelterwood burn affected mortality. Species, crown class, and dbh were included in the analysis.

All four variables were included in the initial full model, which was analyzed using Proc PHREG in SAS® software Version 9.4 (SAS Institute Inc. 2012). The repeated measures of trees on a plot were accounted for by using the robust sandwich estimator option available in the software (Lin and Wei 1989). Non-significant variables were removed from the model with $\alpha = 0.05$. For any significant class variables, multiple comparisons were conducted using a Tukey-Kramer adjustment (Kramer 1956) with dbh set at the approximate mean value of 13 inches to produce estimated survival probabilities.

RESULTS

Combined Mortality

There was a significant dependency between survival rates for trees on the same plot ($p < 0.0001$) which is indicative of some degree of spatial autocorrelation of mortality for trees on each plot. Therefore, main effects were tested using the robust sandwich estimate for the variance. All of the main effects (crown class, dbh, species, and treatment) in the full model were significant (table 1). Multiple comparisons testing using a Tukey adjustment then proceeded for each of the class variables (table 2). For crown class, dominant/codominant trees had significantly greater survival rates

Table 1—Main effects tests for survival of overstory trees, natural and fire mortality

Effect	df	Wald χ^2	p
Crown class	2	25.36	<0.0001
DBH	1	7.53	0.0061
Species	3	10.27	0.0164
Treatment	2	103.36	<0.0001

DBH=diameter at breast height.

than suppressed trees ($p=0.0060$). This was also the case for intermediate vs. suppressed trees, ($p < 0.0001$). The declining rate of survival as crown class decreases (fig. 1) is expected due to suppression (Oliver and Larson 1996, Smith 1986). Chestnut oak exhibited the highest survival rates and white oak the lowest survival rates (fig. 2). Only the chestnut oak vs. white oak species comparison proved significant ($p=0.0121$). Both the shelterwood and the shelterwood-burn treatments were significantly different from the control, with ($p < 0.0001$) and ($p < 0.0001$), respectively. Survival on the control plots was greater than on both the shelterwood and shelterwood-burn plots (fig. 3). This is attributable to mortality from fire as trees cut or destroyed during harvesting were excluded from analysis. An increase of 1 inch in dbh reduced the mortality probability by 3 to 16 percent [95 percent confidence interval for $\exp(\beta_{DBH})$].

Fire Mortality

Similar to the combined mortality, there was a dependency in survival rates for trees within the same plot, ($p < 0.0001$); thus, the robust sandwich variance

Table 2—Multiple comparisons tests for main effects of the overall survival model

Effect	Effect Level 1	Effect Level 2	Estimated difference	Standard error	p
Crown Class	Dominant/Codominant	Intermediate	-0.18	0.35	0.8639
Crown Class	Dominant/Codominant	Suppressed	-1.04	0.34	0.0060
Crown Class	Intermediate	Suppressed	-0.85	0.19	<0.0001
Species	Chestnut oak	Northern red oak	-0.48	0.31	0.4117
Species	Chestnut oak	White oak	-0.95	0.31	0.0121
Species	Chestnut oak	All Other	-0.28	0.21	0.5509
Species	Northern red oak	White oak	-0.46	0.40	0.6424
Species	Northern red oak	All Other	0.20	0.34	0.9353
Species	White oak	All Other	0.67	0.29	0.0972
Treatment	Shelterwood	Shelterwood-burn	-0.04	0.20	0.9766
Treatment	Shelterwood	Control	2.88	0.29	<0.0001
Treatment	Shelterwood-burn	Control	2.93	0.31	<0.0001

Bold effect levels were significant at the $\alpha=0.05$ level.

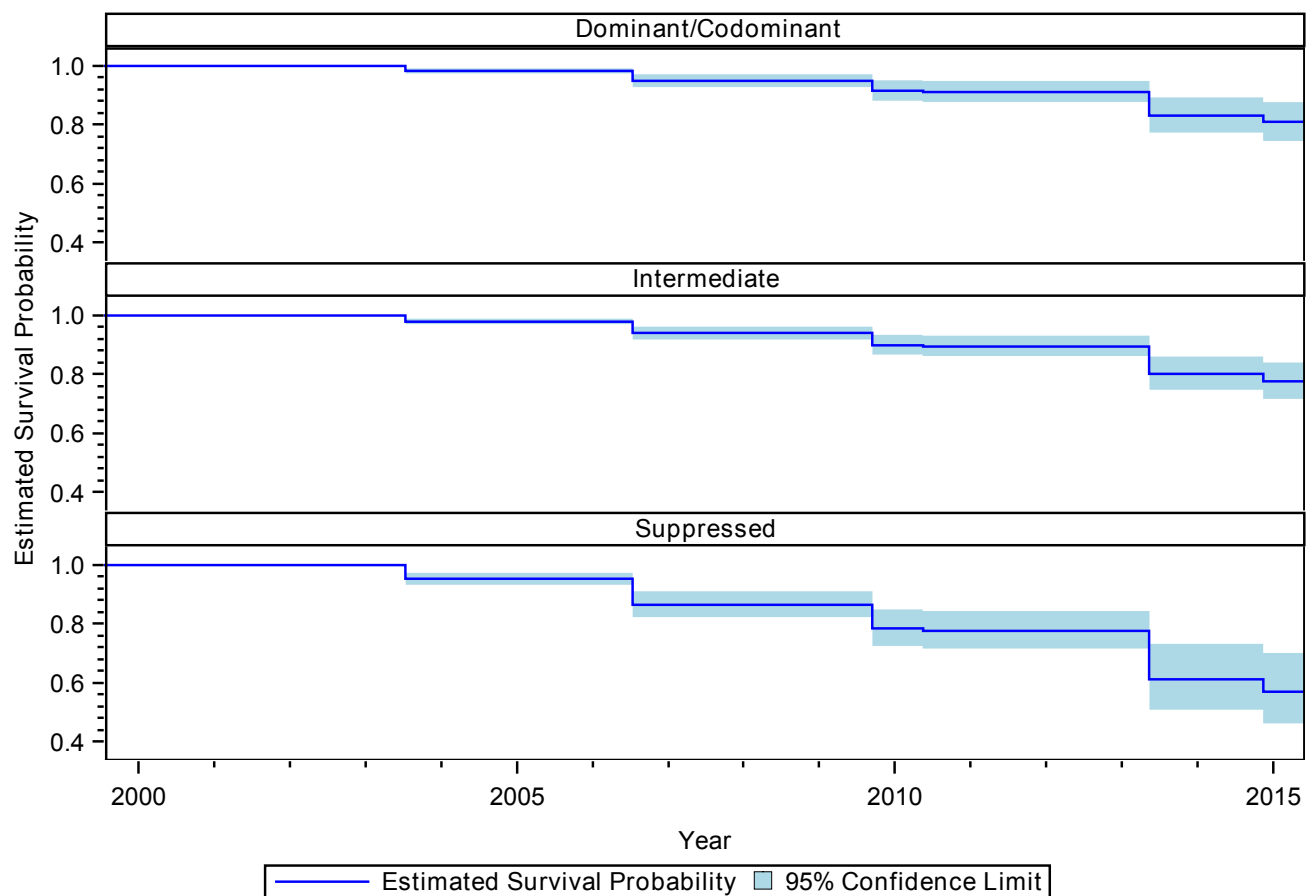


Figure 1—Estimated survival probabilities for each crown class. Estimated probabilities are direct adjusted for each species and treatment, with dbh held at the average value of 13 inches.

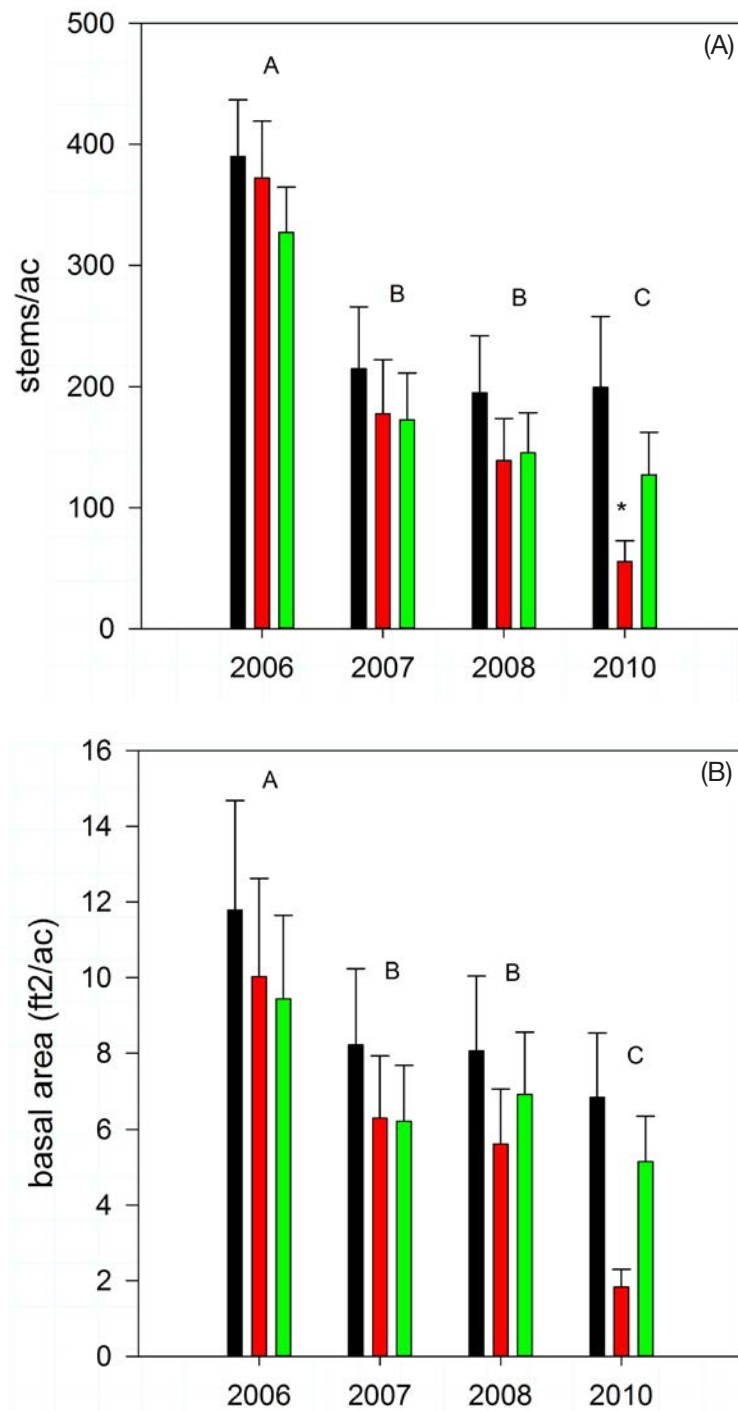


Figure 2— Estimated survival probabilities for each species. Estimated probabilities are direct adjusted for each crown class and treatment, with dbh held at the average value of 13 inches.

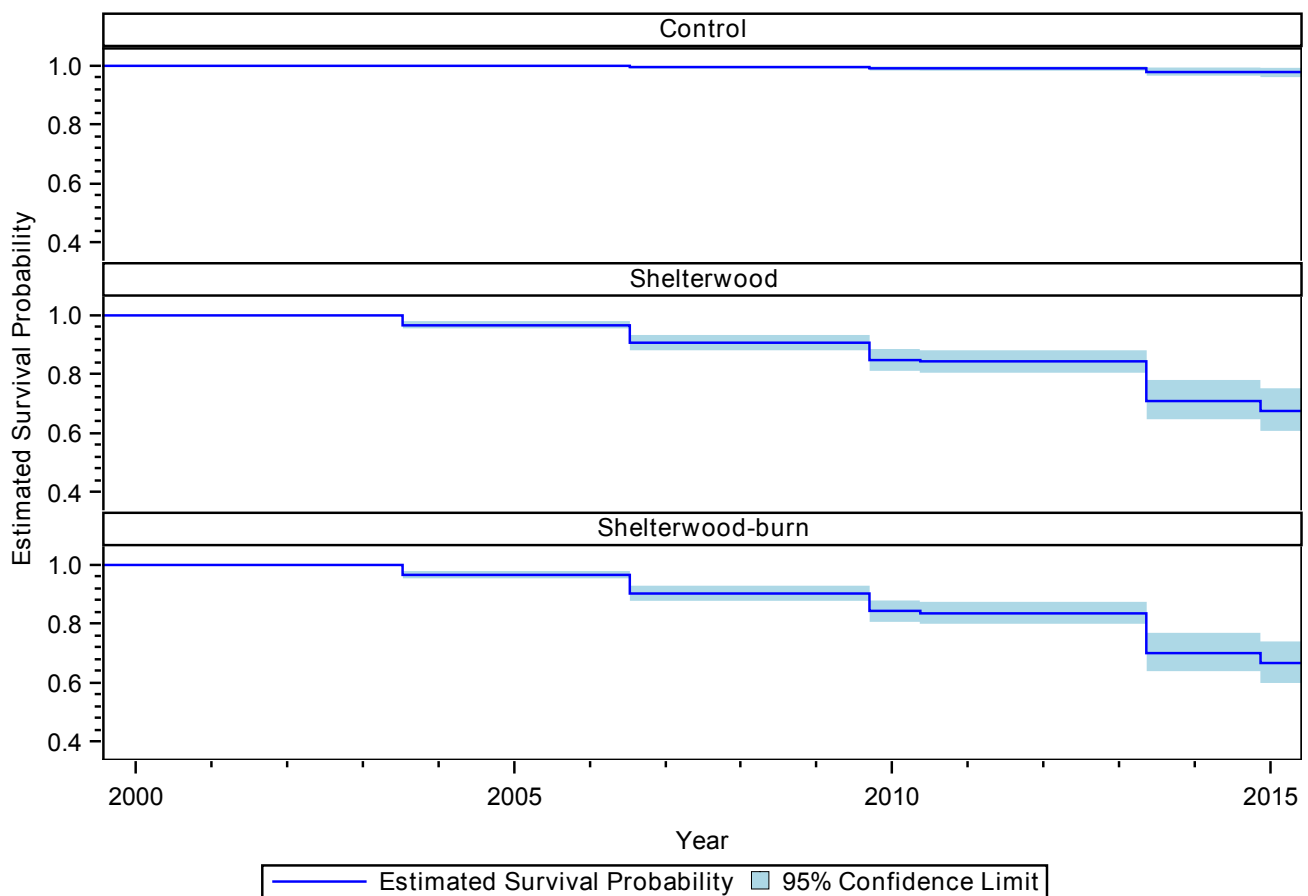


Figure 3— Estimated survival probabilities for each treatment. Estimated probabilities are direct adjusted for each crown class and species, with dbh held at the average value of 13 inches.

estimate was again used to test significance for the main effects. Crown class was not significant ($p=0.7941$) and was dropped from the model. The model was iteratively refit, using backward selection. Both treatment and species were not significant ($p=0.5215$) and ($p=0.0820$), respectively. The lack of a treatment difference indicates that the post-shelterwood burn did not create additional tree mortality. Diameter at breast height was the only significant effect for fire-induced mortality ($p=0.0024$). An increase of 1 inch dbh reduced the probability of dying from 16 to 31 percent [95 percent confidence interval for $\exp(\beta_{DBH})$]. The diameter distribution for all plots treated with prescribed fire (fig. 4) shows that the majority of trees killed by fire were ≤ 10 inches dbh. Further, the shelterwood treatment did not have a prescribed fire after the first removal cut. Mortality shown in figure 4 for the measurement date of November 15, 2014 is solely fire mortality from the shelterwood burn. The number of trees killed is limited and reflects the lack of a significant treatment difference between the shelterwood and the shelterwood burn.

DISCUSSION

As expected, there was a significant increase in overstory tree mortality over background mortality stemming from the application of prescribed fire, particularly in the smaller dbh classes which were the target of the site-preparation burns. Wendel and Smith (1986) observed a similar reduction in the number of overstory stems in a prescribed fire treated stand compared to a control stand in an Appalachian second growth oak-hickory forest. Loucks (2004) did not find a difference in mortality for fire-excluded and burned treatments for trees with dbh >4 inches, however the sample size was very small for that comparison ($n=3$). Paulsell (1957) found in Missouri that for trees with dbh >6.5 inches, mortality was greatest (13.3 percent) over an 8-year period for a control treatment versus an annual-burn treatment (5.2 percent), and a 5-year burn treatment (3.0 percent); none of these studies included a harvest. In a study of eastern hardwoods, Yaussy and Waldrop (2010) found a significant difference

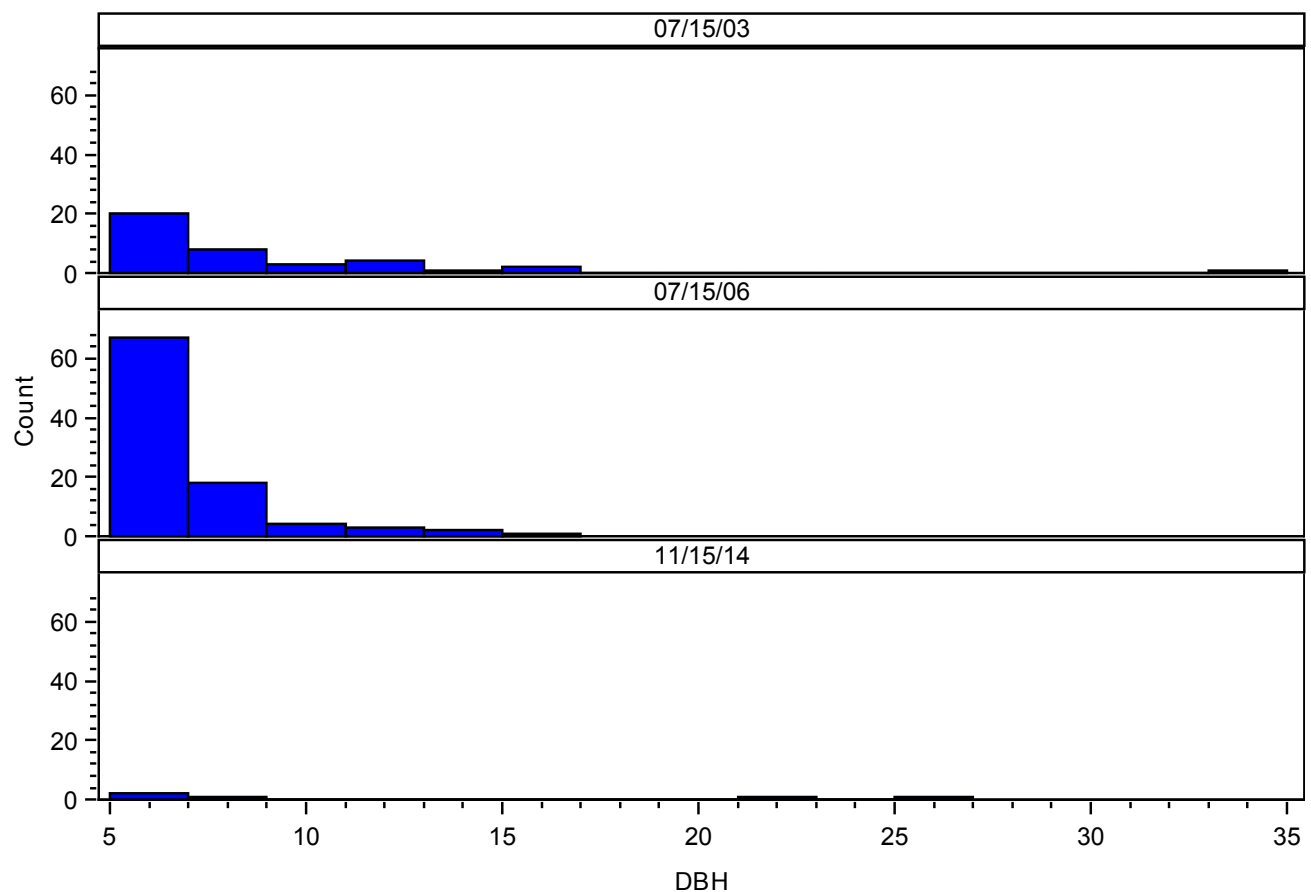


Figure 4—Diameter distributions of fire-killed trees in both shelterwood treatments at post-fire measurement dates.

between four treatments: a control, mechanically treated, prescribed fire, and mechanically treated plus prescribed fire. While no multiple comparisons were performed, the control and the mechanical+burn treatment differed the most at 4 years. In a shelterwood-burn study in Virginia, Brose and Van Lear (1999) reported that mortality was significant greater during a spring burn for all species groups studied (oak, hickory/poplar, and beech/maple) and significantly different for the beech/maple group in summer and winter burns as well. Our study adds further understanding of the effects that multiple prescribed fires and individual tree characteristics have when using the shelterwood-burn method. In particular, we have isolated the effect of the post-shelterwood burn, which has been demonstrated to not significantly affect mortality in the overstory, based on measurements taken 1 year after the prescribed burn. Differences between the control and the two shelterwood treatments for all mortality, therefore, are attributed to the two prescribed fires applied prior to the first removal cut in both shelterwood treatments.

No competitive advantages or disadvantages were discovered for any of the oak species examined in regard to other species present in the study area. This was true for both combined mortality and fire mortality considered alone. While examining species differences for total mortality in a Southern Appalachian oak forest, Greenberg and others (2011) detected a significant difference among oak species. Their study included the three oak species included in our study—northern red, white, and chestnut—and had total mortality rates of 13.8 percent, 10.4 percent, and 5.7 percent, respectively, over a 15-year period. Our results were similar with white oak having higher mortality than chestnut oak; however, mortality for northern red oak in our study did not differ significantly from either white or chestnut oak.

Tree diameter was a significant factor for both combined mortality and fire-induced mortality alone. This is expected for background mortality where the largest dbh trees tend toward greater dominance in the overstory and outcompete smaller trees. In regard to fire, bark

thickness is known to increase with tree diameter and this increase provides additional protection for the cambium from heat damage (Hengst and Dawson 1994, Spalt and Reifsnnyder 1962). For eastern hardwoods, Yaussy and Waldrop (2010) found that a 0.039-inch increase in bark thickness reduced mortality after a prescribed fire by 4.5 percent. Huddle and Pallardy (1996) found that fire accelerated the loss of small diameter trees in all species. Hutchinson, and others (2005) found that small trees (4-10 inches dbh) in study areas with multiple prescribed fires exhibited higher mortality than an unburned control. As was found in our study, Hutchinson and others (2005) determined that mortality for large diameter trees (dbh >10 inches) was not significantly related to fire frequency. Arthur and others (2015) found a relationship of increasing mortality with increasing char height for midstory trees (4-8 inches dbh) on sub-xeric and intermediate landscape positions compared to sub-mesic landscape positions. Also in their study, increasing char heights were correlated with increasing fire temperature. However, char height was not a significant factor for mortality of overstory trees in that study. It appears that the temperature is more of a factor with char height simply a surrogate measure. Given the mortality discrepancy between dbh size classes reported in Arthur and others (2015), a dbh size threshold may have been crossed. In our study, fire mortality subsequent to the two pre-shelterwood cut burns exhibits a reverse j-shaped pattern, with the smallest diameters exhibiting the greatest mortality (fig. 4).

This analysis concerned only mortality and land managers and owners likely have concerns on tree damage, decay, and value loss from the use of prescribed fires. This study area is also used to track wounds, damage, and value loss caused by fire at two points in the study. Seventy-four logs from the shelterwood harvest were tracked from the study site to the mill to evaluate the effects of the two prescribed fires (Wiedenbeck and Schuler 2014). Overall, value loss from decay caused by the two fires in trees assessed 5 years after the last fire was <0.25 percent (Wiedenbeck and Schuler 2014). Differences in species response to fire damage were noted with red maple having a greater level of decay than yellow-poplar (Wiedenbeck and Schuler 2014). With the completion of the shelterwood cut and the post-shelterwood burn, an analysis was made of wounds from all management actions (Wiedenbeck and others 2017). Researchers found that the percentage of trees with fire char and scar was low, 8 percent, after the post-shelterwood fire given the open nature of the stand post-harvest and the thick bark of the mature residual trees (Wiedenbeck and others 2017).

CONCLUSIONS

Our results indicate that increasing tree size decreases the risk of natural and fire mortality by 3 to 16 percent per inch increase in dbh. This is an important consideration for sites for which prescribed fire is used to reduce competition from the seedbed and to establish adequate oak advance regeneration. While mortality differed between the unburned and burned treatments, awareness of this lessened risk of mortality for larger trees may provide guidance to managers wanting to maintain oak in future stands. Knowing that there is reduced mortality for dominant and codominant trees also informs decisions made in the initial shelterwood removal cut. Generally larger diameter dominant and codominant trees are chosen as residuals in a shelterwood harvest and our results show the added benefit of their selection if prescribed fire is also planned. Based on measurements taken 1 year after the post-shelterwood burn, this analysis demonstrates that post-shelterwood controlled burning can be used without additional mortality to the residual overstory trees and provides support for using the shelterwood-burn technique in stands where timber products are an objective for the land manager. The study will continue to be monitored for delayed mortality several years and for other aspects of the study.

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EARLY IMPACTS OF FIRE AND CANOPY GAPS ON SEEDLING AND SAPLING LAYERS: EVIDENCE FOR REVERSING MESOPHICATION?

Melissa A. Thomas-Van Gundy, Thomas M. Schuler, and M. Beth Adams

Abstract—Two prescribed fires and reductions in mid and overstory canopies by herbicide may be starting to reverse the mesophication trend in oak-dominated stands. Before treatment, the sapling layer was about 360 stems per acre and of that, approximately 20 percent was striped maple. The relative abundances of several mesic species increased after treatment (birch and red maple); however, the relative abundance of seedlings of other mesic species showed a temporary reduction after one fire; yellow-poplar seedling abundance dropped after two fires. The relative abundance of non-oak tree seedlings categorized as dry-mesic to xeric (e.g., hickory, sassafras) did increase after the two prescribed fires. Prescribed fire and mid- or overstory reduction treatments occurred at the same time in our study, with little to no existing advanced oak regeneration available to take advantage of favorable conditions created by the treatments. However, our results suggested fire reduced low shade (striped maple) but also shows the disadvantage of creating post-disturbance competitors for oaks (increases in birch and red maple). Two levels of canopy reduction by herbicide treatment conferred no immediate advantage to oak seedlings when combined with fire. For the oak seedlings 2 years post fire, the manipulation of the canopy and midstory by herbicide has not resulted in any benefit over fire alone. We expect that the species composition and structure of the seedling and sapling layers will show greater differences at 5 or 10 years post-fire. In this analysis of almost immediate post-treatment effects, there has not been enough time for oak saplings to develop.

INTRODUCTION

The removal of fire, closed forest canopies, and overabundance of white-tailed deer (*Odocoileus virginianus*) from the eastern hardwood forests has led to an alternative vegetation stable state termed mesophication, whereby shade-tolerant and fire-sensitive tree species, particularly those less prone to herbivory dominate the understory (Campbell and others 2006, Collins and Carson 2003, Nowacki and Abrams 2008). The increase in shade-tolerant and fire-sensitive species in the understory creates shade that leads to a cool and moist microclimate. In turn, these species also tend to produce fuels that are thin, flat, moist, and that rapidly decompose thereby reducing fuel needed to carry a biologically impactful fire (Nowacki and Abrams 2008).

Even with management actions, oak regeneration and eventual accession to the canopy has often proven elusive and less than successful. Oak forests have dominated the Eastern United States for thousands of years (Davis 1981, Watts 1979), yet studies of existing old-growth and second-growth oak-dominated forests show that the replacement of oak overstories with

mesic shade-tolerant species such as sugar maple (*Acer saccharum*) and pioneer species like red maple (*A. rubrum*) are now commonly occurring (Abrams and Downs 1990, Hart and Grissino-Mayer 2009, McGee 1986, Nowacki and Abrams 1992). Although oak seedlings may still be found in the understory and in gap openings, oaks no longer appear to have the ability to persist in the understory as they had in the past (i.e., as much as 54 years on average for northern red oak (*Quercus rubra*)) due to competition for light and herbivory pressure, or become competitive when light conditions are favorable (Kochenderfer and Ford 2008, Rentch and others 2003).

Other factors besides the reduction in fire-return intervals also have occurred in these forests including clearing for agriculture, extensive exploitative logging and slash fires (Stephenson 1993), and the loss of key species from forest disease and insect pathogens (Woods and Shanks 1959). In general, current forests exist under altered conditions from those the original forests developed under, with current forests developing with higher white-tailed deer densities (Rooney 2001), reduced fire frequency (Nowacki and Abrams 2008),

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denser overstory, less flammable understories (Nowacki and Abrams 2008), and smaller canopy gaps (Clebsch and Busing 1989). These have further contributed to depauperate understories dominated by a few tree species that are simultaneously browse-tolerant, shade-tolerant, and fire sensitive (Kain and others 2011, Nuttle and others 2013).

Understory fire has been suggested as a way to change midstory light levels and to favor oak (Lorimer 1989). Generally, oak is a poor competitor on high and medium quality sites, a fact that seems counter-intuitive in view of its current dominance in most Eastern forests (Lorimer 1989). Oaks are not well adapted to low light conditions, although seed will germinate in shade (Crow 1988), and self-replacing oak forests are generally limited to the more xeric sites (Abrams 1992). Oaks do possess many ecophysiological factors that indicate adaptation to fire, such as thick bark on mature trees, ability to form seedling-sprouts hypogeally, ability to stump sprout, deep root system, compartmentalization of stem injuries, and rot resistance (Abrams 1992). Successful oak regeneration relies on advanced reproduction, often as seedling-sprouts and not true seedlings (Johnson and others 2009a).

Fire does top-kill oak seedlings and sprouts, so reduction in their numbers is common immediately after a fire (Brose and others 2014). However, repeated burning and resprouting of oaks and their competitors is expected to create conditions whereby competitors such as red maple and yellow-poplar (*Liriodendron tulipifera*) deplete energy reserves and seed banks faster than oaks due to physiological differences (Lorimer 1985). These insights on fire and oak forest development have led to the increased interest in application of prescribed fire where oaks dominate the overstory.

The mesophication of the eastern forests is an example of the theory of alternative stable states. Applying this theory to fire-adapted forests, it can be argued that the shift caused by the removal of a disturbance regime combined with various other proximate factors is more rapid and harder to reverse on more mesic sites (Nowacki and Abrams 2008). With these regional trends in mind, the effects of fire and canopy reduction on oak regeneration were incorporated in a long-term study on the Fernow Experimental Forest (FEF) in east central West Virginia. The broader study involved the use of prescribed fire and snag creation to create or maintain habitat for tree-roosting bats (Ford and others 2016, Johnson and others 2009b) with no consideration for present or future timber values. Fire and canopy gaps are hypothesized to combat the mesophication occurring in eastern oak forests, particularly those on more mesic locations. If reversal of mesophication is occurring in the study area, the early effects should be discernable in the seedling and sapling layers. We tested the hypothesis

that the species composition of seedlings and saplings has been shifted toward more dry-mesic to xeric species with subsequent decreases in mesic species following two prescribed fires and the creation of canopy gaps.

STUDY AREA

Our research was conducted at the Fernow Experimental Forest (FEF) (fig. 1), an ~4,700-acre experimental forest located in the Unglaciaded Allegheny Mountains subsection of the Appalachian Plateau Physiographic Province in Tucker County, West Virginia (Cleland and others 2007). The study site is Compartment 45 located in the John B. Hollow watershed of the FEF and is referred to as the "John B. Hollow study". Elevations at the study site range from about 2000 to 2600 feet. The underlying geology of the study site is mostly shale and sandstone of the Hampshire formation. Mean annual total precipitation on the FEF is about 57 inches, with maximum average monthly precipitation occurring in June and minimum monthly precipitation in October. Mean annual temperature is 48.5° F, ranging from -0.4° F in January to 69° F in July (Kochenderfer 2006). The FEF was initially logged from 1903 to 1911 with nearly complete removal of all merchantable trees (Schuler and Fajvan 1999). Although the overstory of the FEF is broadly described as mixed mesophytic (Braun 1950), within the study site, the overstory was dominated by northern red oak, chestnut oak (*Q. prinus*), sugar maple, red maple, and yellow-poplar (*Liriodendron tulipifera*) with oak species having larger average diameter at breast height (DBH). In 2006, before treatments were applied, total basal area in the overstory (trees over 5 inches DBH) was about 132 square feet per acre.

METHODS

We conducted spring burns in 2007 and 2008 in Compartment 45 (299 acres), including all sample plots each year (fig. 1). Strip-head-fire techniques were used during the burns with ignition by hand-held drip torches. Since the overarching focus of the study was wildlife habitat (Johnson and others 2009b), greater fire intensity was allowed than if production of timber products were a management goal. No measures of fire intensity were taken, however the resulting mosaic of fire severity includes patches where overstory mortality occurred and other areas with little fire damage (cove or riparian areas). Within the burn unit, a total of 60 randomly located ~66 ft (20 m) radius plots were established and three treatments were assigned in a completely randomized design. After the second prescribed fire in 2008, 49 plots were determined to have been burned; our analysis includes only those 49 plots. On 19 plots, all overstory trees (> 5 inches DBH, with the exception of oak or hickory [*Carya* spp.]; retained as day-roosts for the endangered Indiana bat *Myotis sodalis*; Johnson and others 2010) were deadened with stem-injection application of glyphosate herbicide during the growing

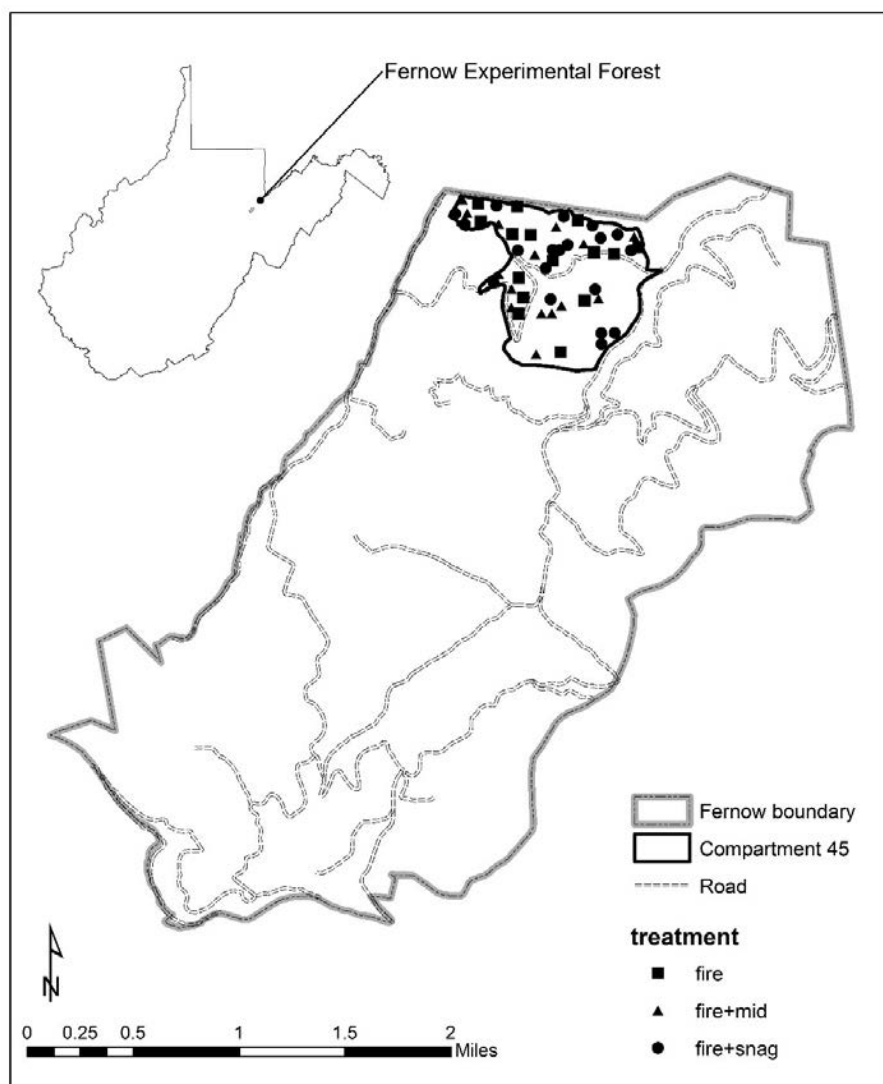


Figure 1—General location of study area with plot locations by treatment type.

season in 2007; this treatment is referred to as fire+snag in this analysis. All midstory trees (5 to 11 inches DBH, with the exception of oak or hickory) were removed by herbicide on another 15 plots (fire+mid treatment). In both these treatments, trees that were not killed by the 2007 application of herbicide were treated again in 2008. We are considering the herbicide treatment to be conducted once (2007) in the sample design and statistical analysis. On the remaining 15 plots no over- or midstory trees were treated with herbicide (fire treatment; fig. 1).

At each plot center, a 1/100th acre plot was established to tally woody vegetation 1 to 5 inches in DBH. Species, DBH, crown class, origin (seedling or sprout), and quality were recorded for each stem. In the four cardinal directions from the main plot center, four milacre plots

were established to sample the seedling regeneration. On these plots, woody vegetation <1 inch DBH and > 6 inches in height was tallied by species, height class, and origin (seedling or sprout).

Statistical Analysis

Seedling relative abundance was calculated for all tree and shrub species by plot (an average of the 4 milacre subplots); no vine species were included in the seedling calculations and blackberry (*Rubus* spp.) was excluded. To test for the effect of prescribed fire and overstory reduction on the species composition of the seedling layer, species were assigned to either mesic or dry-mesic to xeric categories, or were considered separately. Table 1 lists the species considered for each category and those species assessed separately.

Table 1—Species groups used in analysis

Species group	Species/genera
Other mesic trees	White ash (<i>Fraxinus americana</i>)
	Dogwood (<i>Cornus florida</i>)
	Basswood (<i>Tilia americana</i>)
	Sugar maple (<i>Acer saccharum</i>)
	Black cherry (<i>Prunus serotina</i>)
	Fire cherry (<i>P. pensylvanica</i>)
	Serviceberry (<i>Amelanchier</i> spp.)
	Frasier magnolia (<i>Magnolia fraseri</i>)
	Cucumber tree (<i>M. acuminata</i>)
	American beech (<i>Fagus grandifolia</i>)
	Hophornbeam (<i>Ostrya virginiana</i>)
	Hornbeam (<i>Carpinus caroliniana</i>)
Other dry-mesic to xeric trees	Sourwood (<i>Oxydendrum arboreum</i>)
	Blackgum (<i>Nyssa sylvatica</i>)
	Black locust (<i>Robinia pseudoacacia</i>)
	Sassafras (<i>Sassafras albidum</i>)
	American chestnut (<i>Castanea dentata</i>)
	Hickory (<i>Carya</i> spp.)
	White pine (<i>Pinus strobus</i>)
Mesic shrubs	Witch hazel (<i>Hamamelis virginiana</i>)
	Wild hydrangea (<i>Hydrangea arborescens</i>)
	Sumac (<i>Rhus typhina</i>)
	Spicebush (<i>Lindera benzoin</i>)
	Rhododendron (<i>Rhododendron</i> spp.)
	Mapleleaf viburnum (<i>Viburnum acerifolium</i>)
	Deciduous holly (<i>Ilex montana</i>)
	Hercules club (<i>Aralia spinosa</i>)
	Elderberry (<i>Sambucus</i> spp.)
Dry-mesic to xeric shrub	Minniebush (<i>Menziesia pilosa</i>)
	Blueberry (<i>Vaccinium</i> spp.)
	Mountain laurel (<i>Kalmia latifolia</i>)
	Azalea (<i>Azalea</i> spp.)
As individual species or genera	Birch (<i>Betula alleghaniensis</i> and <i>B. lenta</i>)
	Yellow-poplar (<i>Liriodendron tulipifera</i>)
	Striped maple (<i>A. pensylvanicum</i>)
	Red maple (<i>A. rubrum</i>)
	Oak (<i>Quercus</i> spp.)

For the sapling layer, stems per acre and basal area per acre were calculated. Preliminary analyses showed that assessing individual species or descriptive groups of species did not result in conclusive statistical assessment due to low numbers of saplings. Stems per acre by species or species group were calculated for descriptive statistics. The other mesic species category included: eastern hemlock (*Tsuga canadensis*), American beech, black cherry, and sugar maple. The other dry-mesic to xeric category included: black locust, blackgum, and sourwood.

We used a repeated measures completely randomized, generalized linear model with pseudo-maximum likelihood estimation via PROC GLIMMIX (SAS 2012) in a two factor design. Year (2006, 2007, 2008, and 2010) and overstory treatment were the explanatory variables with treatments defined as fire, fire+mid, and fire+snag based on the random assignment of herbicide treatments to the plots. The associated interactions of year and treatment were fixed effects in the model. Pre-treatment year is 2006, with treatments applied in 2007 (fire and herbicide) and 2008 (second fire), with 2010 as most recent post-treatment measurement. We used the exponential distribution with logarithmic link function because relative density data displayed a negative exponential distribution (SAS 2012). In the repeated measures model, covariance structures of autoregressive, heterogeneous autoregressive, or variance components were used depending on response variable to account for the correlation of repeated measures over time. Least square means are displayed; the Tukey-Kramer method was used to adjust p values for multiple orthogonal comparisons. Comparisons between years and among treatments were also made through contrasts of the means of fire only treatment with the means of the other fire and snag creation treatments combined. Contrasts of interest were: 2006 fire vs. 2006 both fire+mid and fire+snag and 2010 fire vs. 2010 both fire+mid and fire+snag.

RESULTS

Although the changes in the overstory were not addressed statistically in this analysis, the herbicide and fire treatments did reduce the overstory basal area. By 2010, basal area in the fire only plots had increased slightly to about 139 square feet per acre, reduced to about 110 square feet per acre in the fire+mid treatment plots, and been reduced to about 79 square feet per acre in the fire+snag treatment plots.

Total stems per acre in saplings were reduced from around 360 stems per acre before treatment to 33 to 205 per acre (depending on treatment) in 2010 (fig. 2a). All treatments showed significant reductions in numbers of stems per acre from 2006 to 2010 ($p=0.0015$ for fire, $p<0.0001$ for fire+mid, and $p=0.0175$ for fire+snag). The fire+mid plots had the lowest mean stems per acre in

2010 but did not differ significantly from other treatments when adjustments were made for multiple comparisons.

Total basal area in saplings of all species followed roughly the same pattern as for stems per acre and was reduced by the treatments (fig.2b), however only the basal area in fire+mid plots was reduced significantly from 2006 to 2010 ($p<0.0001$). For the fire only and fire+snag plots, basal area per acre was not significantly different before treatment and 2010 ($p=0.1442$ for fire only and $p=0.4510$ for fire+snag). In 2010, the basal area in the fire+mid plots averaged only 1.3 square feet per acre but when adjusted for multiple comparisons, there were no differences between treatments in 2010. For both stems per acre and basal area per acre, no significant differences were found in 2010 between the fire plots and the two other treatments combined.

The absolute numbers of birch and red maple saplings declined with fire and mid- or overstory reductions, with slight reductions in relative abundances as well (table 2). There were no yellow-polar or oak (all species combined) saplings in the plots after treatment. Striped maple saplings were reduced in absolute and relative abundance after treatments were applied, dropping to zero in 2008 but increasing to 2 stems per acre in 2010. Surprisingly, saplings categorized as dry-mesic to xeric (other than oaks) also dropped in abundance with treatments although their relative abundance increased slightly. Similarly, saplings of mesic species (other than those already considered) also dropped in absolute numbers but their relative abundance increased after treatments.

Total seedlings increased from about 24,000 per acre before management actions (all treatments combined) to about 41,000 to 49,000 after treatment (table 3; fig 3a) with differences found from 2006 to 2010 for all three treatments combined ($p=0.0009$). Higher stems per acre were found in plots in 2010 where the mid- or overstory was also reduced although differences among treatments were not statistically significant in 2010. The relative abundance of seedlings of all oak species was reduced by all treatments from 2006 to 2010 ($p=0.0004$ for fire, $p=0.0002$ for fire+mid, and $p<0.0001$ for fire+snag), however there was a slight recovery 2 years after the second fire (fig. 3b). In terms of absolute numbers, the number of oak seedlings is about half the pre-treatment level (table 3).

The relative abundance of dry-mesic to xeric shrubs were reduced post treatment compared to pre-treatment levels ($p<0.0001$ for fire, $p=0.0005$ for fire+mid, and $p=0.0177$ for fire+snag), however this group of species only made up approximately 2-4 percent of all seedlings before fire and canopy gaps were created (fig. 3c). In contrast, the seedlings of tree species categorized as dry-mesic to xeric (other than oak species) increased

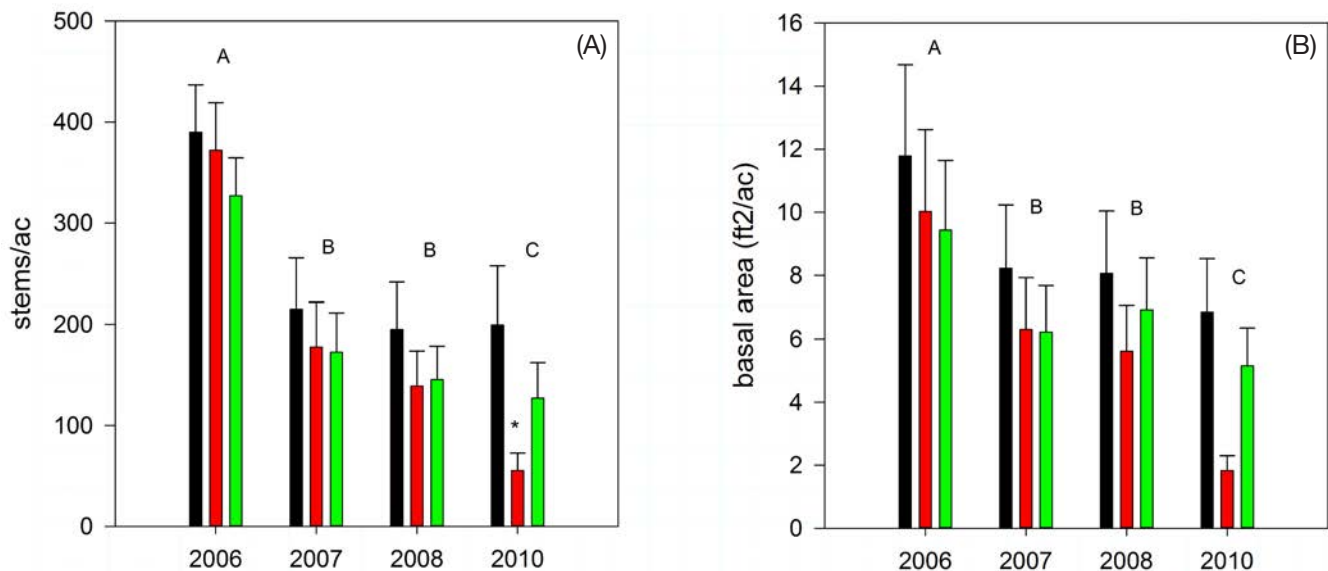


Figure 2—Comparison of least square mean ($\pm$ SE) of total sapling density (a) and total sapling basal area (b) by treatment and year. Means between years with the same letter are not statistically different ($\alpha = 0.05$).

Table 2—Sapling species composition as mean stems per acre (percent total in parenthesis) on the John B. Hollow study site, Fernow Experimental Forest, West Virginia, 2006-2010

Species or group	2006	2007	2008	2010
Birch	20.4 (5.6)	4.1 (2.9)	4.1 (3.4)	6.1 (7.0)
Yellow-poplar	2.0 (0.6)	0.0	0.0	0.0
Striped maple	71.4 (19.8)	6.1 (4.4)	0.0	2.0 (2.3)
Red maple	51.0 (14.1)	32.7 (23.5)	14.3 (12.1)	12.2 (14.0)
Other mesic	175.5 (48.6)	75.5 (54.4)	79.6 (67.2)	53.1 (60.5)
All oaks	0.0	2.0 (1.5)	0.0	0.0
Other dry-mesic to xeric	40.8 (11.3)	18.4 (13.2)	20.4 (17.2)	14.3 (16.3)
Total	361.2	138.8	118.4	87.8

Table 3—Seedling species composition as mean stems per acre (percent total in parenthesis) on the John B. Hollow study site, Fernow Experimental Forest, West Virginia, 2006-2010

Species or group	2006	2007	2008	2010
Birch	270.5 (1.2)	65.1 (0.4)	51.4 (0.1)	3,630.1 (9.0)
Yellow-poplar	407.5 (1.8)	6,126.7 (35.9)	10,729.5 (24.7)	5,301.4 (13.1)
Striped maple	3,434.9 (15.0)	2,256.8 (13.2)	1,455.5 (3.4)	1,106.2 (2.7)
Red maple	2,476.0 (10.8)	3,239.7 (19.0)	5,845.9 (13.5)	11,887.0 (29.3)
Other mesic trees	2,287.7 (10.0)	1,003.4 (5.9)	4,332.2 (10.0)	3,688.4 (9.1)
All oaks	8,664.4 (38.0)	1,804.8 (10.6)	2,260.3 (5.2)	4,438.4 (11.0)
Other dry-mesic to xeric trees	537.7 (2.4)	1,164.4 (6.8)	12,708.9 (29.3)	8,058.2 (19.9)
Mesic shrubs	390.4 (1.7)	123.3 (0.7)	452.1 (1.0)	866.4 (2.1)
Dry-mesic to xeric shrubs	4,356.2 (19.1)	1,291.1 (7.6)	5,568.5 (12.8)	1,547.9 (3.8)
Total seedlings	22,825.3	17,075.3	43,404.1	40,524.0

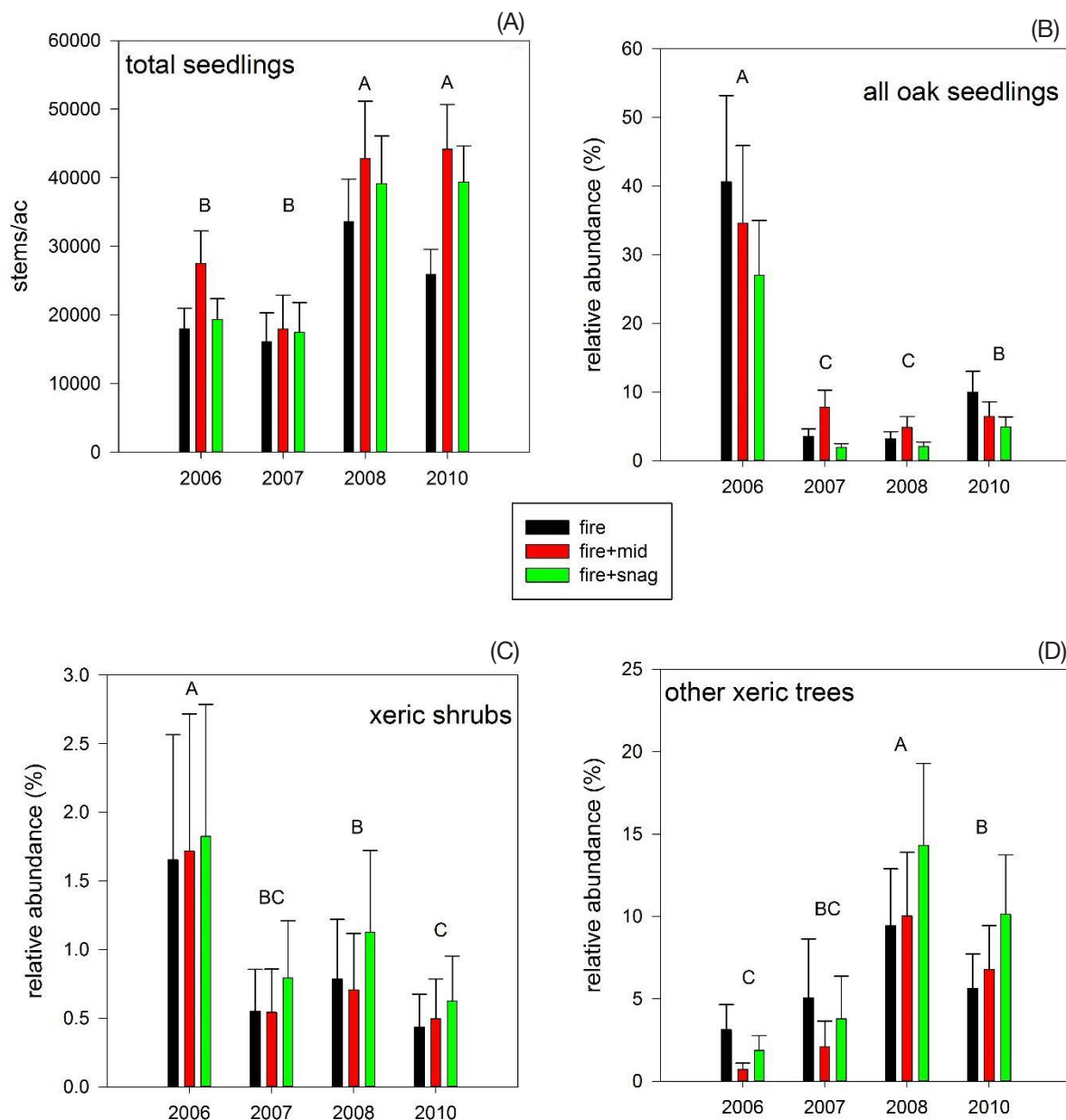


Figure 3—Least square means ($\pm$ SE) of total (a), oak (b), dry-mesic to xeric shrubs (c), and other dry-mesic to xeric tree species (d) seedlings by year and treatment. Means between years with the same letter are not statistically different ($\alpha = 0.05$).

in relative abundance after the two fires and snag creation, increasing from about 1–3 percent abundance to 9–11 percent in 2010 ($p < 0.0001$ for all treatments combined; $p = 0.0021$ for fire+mid) (fig. 3d). There were no statistically significant differences among treatments in any year. This species group increased in absolute numbers (averaged for all treatments) as well, going from about 500 stems per acre in 2006 to over 8,000 in 2010 (table 3).

Birch seedling relative abundance increased dramatically after treatments ($p < 0.0001$ all treatments combined) with the greatest increase found in fire+snag plots, although no statistically significant differences among treatments

were found in 2010 (fig. 4a). Relative abundances of yellow-poplar increased with treatments in 2007 ($p = 0.0004$ for fire, $p < 0.0001$ for fire+mid and fire+snag plots), but are showing decreases after the second prescribed fire (fig. 4b). As happened in the sapling layer, striped maple seedling relative abundances were reduced by prescribed fire ($p < 0.0001$ for fire, $p = 0.0001$ for fire+mid and fire+snag), dropping from around 13 to 16 percent before treatment to 1 to 2 percent in 2008 and 2010 (fig. 4c). In contrast, red maple seedling relative abundances increased over time ($p < 0.0001$ for all treatments combined) and there were no differences between treatments or within treatments over time (fig. 4d).

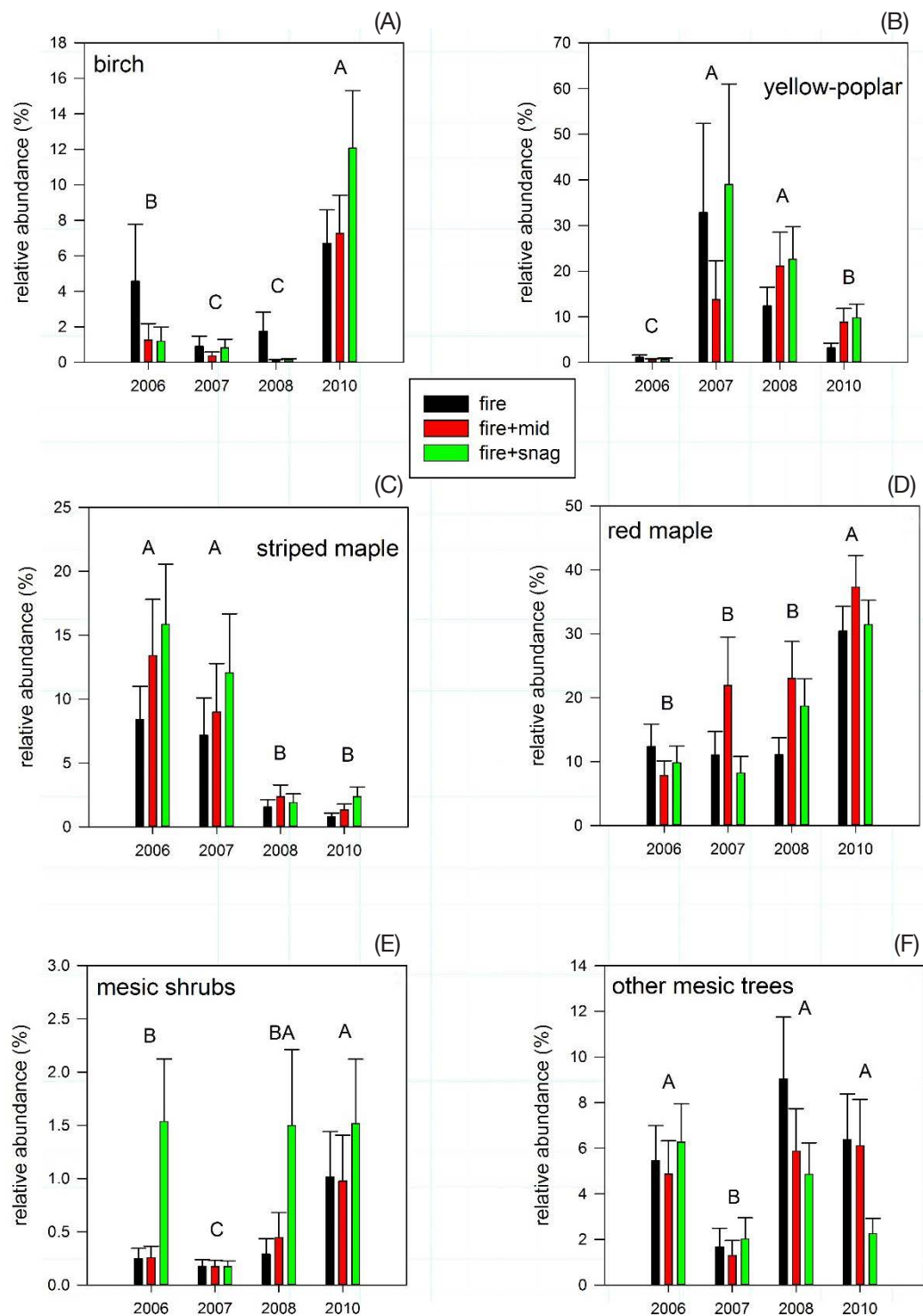


Figure 4—Least square means ($\pm$ SE) of birch (a), yellow-poplar (b), striped maple (c), red maple (d), mesic shrubs (e), and other mesic to xeric tree species (f) seedlings by year and treatment. Means between years with the same letter are not statistically different ($\alpha = 0.05$).

The relative abundances of species categorized as mesic shrubs increased over time ($p=0.0006$ for all treatments combined) and with fire+mid treatment in particular ($p=0.0057$), however, like the dry-mesic to xeric category, mesic shrubs made up a small proportion of the seedling layer even after treatment (fig. 4e). Seedlings of mesic tree species decreased in relative abundance after the first fire ($p<0.0001$) and recovered to pre-treatment levels by 2010, although still < 10 percent of the species composition (fig. 4f). In 2010 the lowest relative abundance for this group was found in the fire+snag plots although no differences between treatments in year 2010 were statistically significant. For seedling groups, no significant differences were found in 2010 between the fire plots and the two other treatments combined.

DISCUSSION

We do find some evidence that two prescribed fires and reductions in mid and overstory canopies by herbicide may be starting to reverse the mesophication trend in these stands. Before treatment, the sapling layer was about 360 stems per acre and of that, approximately 20 percent was striped maple. The dense shade created by striped maple saplings has been removed from the stands as was found on a nearby study site on Canoe Run. After two prescribed fires, the species composition of the sapling layer showed only slight change, but similar to the John B. Hollow study, the stems per acre of saplings was reduced by nearly 90 percent (Schuler and others 2010, 2013). In the John B. Hollow study, sapling species considered dry-mesic to xeric are holding steady in relative abundance (table 2).

Whereas the relative seedling abundances of several mesic species increased after treatment (birch and red maple) the relative abundance of seedlings of other mesic species showed a temporary reduction after one fire and yellow-poplar seedling abundance dropped after two fires. These effects were also found on the nearby Canoe Run study, although two prescribed fires had reduced red maple seedling densities on that site (Schuler and others 2013). The lowest relative abundance of seedlings of mesic tree species (other than birch, yellow-poplar, and red maple) was found in fire+snag plots which is consistent with the hypothesis that fires and canopy gaps are reversing mesophication in these stands.

The relative abundance of tree seedlings categorized as dry-mesic to xeric (other than oaks) did increase after the two prescribed fires. Although there were no differences among treatments in 2010, higher abundances were found in fire+snag plots which, on average, showed the lowest overstory basal area per acre in 2010. Although oak seedling relative abundances were reduced by prescribed fires, other dry-site species appear to be responding to the treatments.

Our results are similar to a prescribed fire study in southeastern Ohio in a very similar forest type where areas were burned either two times or four times although no canopy gaps were created. Early results there found that fires reduced the sapling layer, which was mainly composed of shade tolerant species, and that oak and hickory seedling abundances were not affected by fire (Hutchinson and others 2005a). Similar to our findings in West Virginia, red maple abundances decreased immediately after fire only to rebound within 2 years and yellow-poplar seedling abundances responded with the opposite trend of increasing after fire only to decrease two years post-fire (Hutchinson and others 2005a). In an analysis of the herbaceous layer of the Ohio study, sassafras was found to be an indicator of burned plots (Hutchinson and others 2005b). This species was tallied in the dry-mesic to xeric category in our study and showed a large increase in response to two prescribed fires.

Because oak regeneration requires advanced reproduction, management actions such as prescribed fire can target the pre-disturbance buildup of low, dense shade (Brose and others 2008). Also, post-disturbance, the oak advanced regeneration faces competition from species with regeneration strategies such as seed banking or having lightweight seed that rapidly invades a disturbed area (Schuler and others 2010). Prescribed fire and mid- or overstory reduction treatments occurred at the same time in our study, with little to no existing advanced oak regeneration available to take advantage of possible favorable conditions created by the treatments. However, our results do show the advantage of fire in reducing low shade (striped maple) but also shows the disadvantage of creating post-disturbance competitors for oaks (increases in birch and red maple). Our results further support the findings of others on the need for repeated fires to confer a competitive advantage on oak regeneration and for careful consideration of the timing of management actions (Brose and others 2014, Johnson and others 2009a). Successful oak regeneration incorporating prescribed fire is difficult to predict because the process involves the variable and interconnected factors of species life history traits (both of oaks and competitors), the pre-burn condition of the oak seedlings or seedling-sprouts, and the variability inherent in fire as a management tool (Alexander and others 2008).

Perhaps these modest movements away from mesophication are a function of time since treatment. The latest measurements included in this analysis are from 2010, which was only 2 years post-fire. While the two levels of canopy reduction through herbicide conferred no immediate advantage to oak seedlings when combined with fire, these treatments did create wildlife habitat by direct snag creation. For the oak seedlings 2 years post fire, the manipulation of the

canopy and midstory by herbicide has not resulted in any benefit over fire alone. We expect that the species composition and structure of the seedling and sapling layers will show greater differences at 10 or more years post-fire. In this analysis of almost immediate post-treatment effects, there has not been enough time for oak saplings to develop. We will also be assessing these stands for the need for another fire as others have found more than two fires are needed for oaks to remain competitive (Hutchinson and others 2012) especially given the mesic nature of the study area.

ACKNOWLEDGMENTS

We thank Jim Rentch (West Virginia University), Jamie Schuler (West Virginia University), and Gary Miller (Northern Research Station) for reviews of early drafts of the manuscript. We also thank W. Mark Ford (U.S. Geological Survey, Virginia Cooperative Fish & Wildlife Research Unit) for his work in initiating the larger study and securing funding from Florida Power and Light. And finally, special thanks to Rick Hovatter and Donnie Lowther, forestry technicians on the Fernow Experimental Forest.

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Poster Session

ENHANCING BLACK WALNUT SEEDLING ESTABLISHMENT IN A NORTHWEST ARKANSAS CREEK BOTTOM

William L. Headlee, H. Christoph Stuhlinger, and Amanda M. Foust

Extended abstract—Black walnut (*Juglans nigra*) is a valuable hardwood species that grows best on well-drained sites in creek bottoms (Williams 1990). Natural regeneration of black walnut is sometimes sparse (Williams 1990), so supplemental seedling planting is desirable to establish and produce quality trees. Enhancing early survival and growth of the black walnut seedlings can give them an advantage over competing vegetation and help ensure desired stocking levels. This study compares treatments for black walnut bare-root seedling establishment in a creek bottom along an intermittent unnamed tributary to the Illinois River in northwest Arkansas. The main soil series in the area is a Razort gravelly silt loam, which is a very deep, well-drained soil subject to occasional flooding (WSS 2014).

Thirty-five study plots were installed along a 1.6-km section of woods road adjacent to the creek. Planting holes were dug using a 15-cm-wide skid steer-mounted auger. In late March 2016, 4 black walnut bare-root seedlings were planted approximately 3 m apart at each plot. Each seedling represented one of the four treatments:

- 1) control (no tree shelter or fertilizer)
- 2) tree shelter (122 cm Tubex[®])
- 3) fertilizer tablets (10 g, 20-10-5 tablets, 2 tablets per seedling)
- 4) tree shelter plus fertilizer tablets

Initial and end-of-season seedling height and ground line diameter (GLD) data were collected at each plot. Plots were checked for survival and health issues were noted (e.g., top damage and aphid infestation) one time in each of the following months: June, July, September, and November. Measurements of temperature and humidity both inside and outside of shelters at each plot were also recorded in June and July. HOBO[®] data loggers (Onset Computer Corporation, Bourne, MA) mounted on rebar posts approximately 1 m above the ground in the center of each plot were set to record light intensity readings every hour during June.

For both seedling height and GLD, two-way ANOVA (2 levels of shelter × 2 levels of fertilizer) was used to test for significant differences in end-of-season size, with initial size used as a covariate and observations blocked by plot. For temperature and humidity, two-way ANOVA (2 levels of shelter × 2 levels of time) was used to test for significant differences in growing conditions, with observations blocked by plot. In all cases, the data were analyzed using PROC GLM in SAS[®], and differences were considered statistically significant at $p < 0.05$. When significant treatment differences were indicated by ANOVA, least-squares means were compared using t-tests.

Survival was relatively high (>90 percent) across all treatments, while top damage was relatively low (<6 percent) for all treatments except the control (11.4 percent), as shown in table 1. Aphid infestation was higher for the treatments receiving fertilizer (8.6–40 percent) than those without (0–14.3 percent), which suggests the aphids may have been more attracted to the fertilized trees. Also, aphids were less common on sheltered trees (0–8.6 percent) than unsheltered trees (14.3–40 percent), which suggests the shelters may have made these trees more difficult for the pests to find.

Heights at the end of the growing season were significantly influenced by shelter treatments and by initial heights, but not by fertilizer treatments or by plots. Mean heights of the sheltered trees (92 cm) were significantly higher than unsheltered trees (57 cm), as shown in figure 1A. End of season height showed a positive relationship with initial height. GLD was not significantly influenced by any treatments.

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Table 1—Percent survival, top damage, and aphid infestation by treatment for black walnut seedling establishment in 2016 in northwest Arkansas

Treatment	Survival (%)	Top Damage (%)	Aphids (%)
Control	94.3	11.4	14.3
Shelter	91.4	5.7	0.0
Fertilizer	91.4	5.7	40.0
Shelter and Fertilizer	91.4	2.9	8.6
Overall	92.1	6.4	15.7

Survival is based on end-of-season measurements, top damage is based on observations throughout the first growing season, and aphids were observed to be present only during the July measurements.

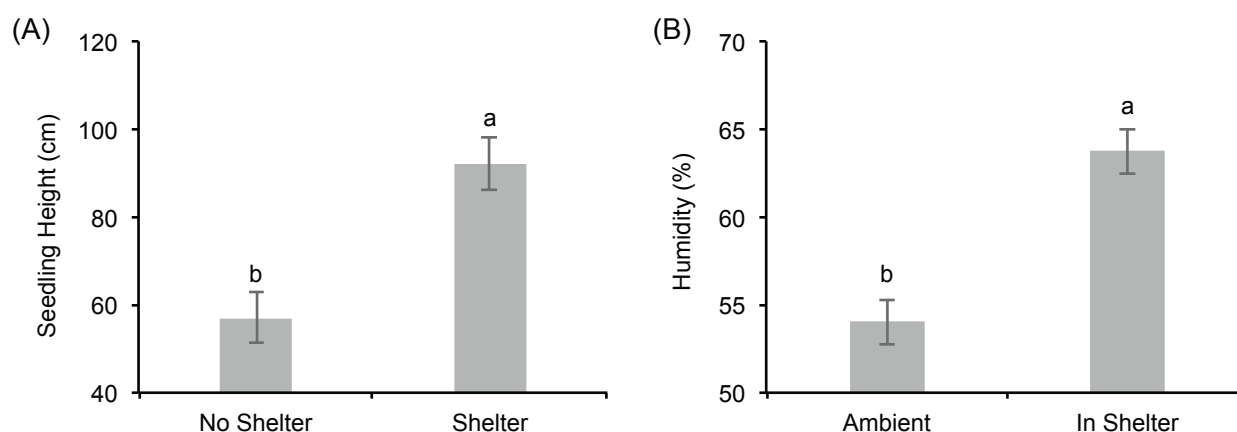


Figure 1—Means and 95 percent confidence intervals for (A) black walnut seedling heights at the end of the first growing season (2016), by shelter treatment, and (B) humidity measurements taken outside (ambient) and inside the tree shelters during June and July, 2016. Different letters above the bars indicate a significant difference at the 0.05 level.

Humidity readings in the plots differed significantly by plot, time, and shelter treatments. The humidity inside the shelters (64 percent) was significantly higher than the ambient humidity (54 percent), as shown in figure 1B. Temperature readings differed significantly by plot and time, but not by shelter treatment. July humidity and temperature were both significantly higher than June. Mean light intensity readings during daylight hours (5:00–21:00) ranged from 11,690 to 86,690 lumens per square meter. However, the differences among plots in light intensity did not translate to differences among plots in final tree height or GLD.

In summary, tree shelters were associated with a significant gain in walnut seedling height at the end of the first growing season, and appear to have altered the growing environment primarily in terms of increased humidity. Fertilized trees were slightly taller than unfertilized trees, and while the differences were nonsignificant in the first year, we plan to monitor growth during the second year to see if the slow-release fertilizer has a delayed effect.

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SEED TREE DISTANCE AND TOPOGRAPHY AFFECT YELLOW-POPLAR SEEDLING DENSITY AFTER PRESCRIBED BURNING AN UPLAND OAK STAND

W. Henry McNab

Abstract—Yellow-poplar (*Liriodendron tulipifera*) seed dispersal has received moderate study, but little has been reported on the pattern of seedling recruitment and site characteristics affecting density, particularly in prescribed burned upland oak stands. The purpose of this study was to determine density of yellow-poplar seedlings 2 years after a growing-season prescribed burn in an upland oak stand in response to environmental and biological factors. Results suggest yellow-poplar seedlings can be established up to 170 m from seed trees, and regeneration densities and heights are greater on concave compared to convex land forms.

INTRODUCTION

Yellow-poplar is a wind-disseminated, shade-intolerant, highly productive, economically valuable, and long-lived pioneer mesophytic species that typically dominates moist slopes and coves in the Central Hardwood Region east of the Mississippi River (Beck and Della-Bianca 1981). Seed production by this species is dependable annually, prolific, accumulates in the forest floor, and can remain viable for up to 7 years (Clark and Boyce 1964). Disturbance of the forest floor, such as from harvesting, can result in establishment of dense yellow-poplar regeneration that grows rapidly in height without the need to establish a well-developed root system, which is necessary for oaks (*Quercus*) and hickories (*Carya*). Although classified as a mesophytic species, new yellow-poplar regeneration can become established on submesic sites with suitable climatic conditions (e.g., above-average precipitation) and compete with other desirable species such as oaks and hickories (Beck and Hooper 1986, McGee 1975). Forest floor disturbance from controlled burning can also initiate yellow-poplar seedling establishment from the residual seed bank (McNab and others 2013, Shearin and others 1972), which could result in changes of species composition on dry sites in the presence of nearby yellow-poplar seed sources. The pattern and density of yellow-poplar seed dissemination has been well studied on unburned sites and found to provide adequate natural regeneration within a distance of about twice the height of seed trees (Whipple 1968). Only sparse information is available on yellow-poplar seedling establishment on upland hardwood sites that have been controlled burned.

In a previous study, I investigated establishment of yellow-poplar regeneration near seed trees following controlled burning in upland oak stands (McNab 2016). That study identified important physical and biological variables affecting yellow-poplar seedling density, but results were limited to distances within 50 m of seed trees. My observations while conducting that study resulted in the investigation reported here, which addresses four questions associated with establishment and growth of yellow-poplar seedlings: (1) is density of regeneration uniformly distributed around seed trees on slopes; (2) is the density of seedlings related to distances beyond 50 m from seed trees; (3) do site variables, particularly topography, influence density of seedlings; and (4) is height of 2-year old seedlings influenced by site factors. The scope of my study was restricted to data from one seed source, and the results should therefore be considered preliminary. The purpose of this study was primarily to identify variables that could be examined in subsequent investigations of broader scope and not to develop predictive models for management applications.

METHODS

Study Area

The study was in the Bent Creek Experimental Forest, located in the Pisgah Ranger District of the Pisgah National Forest (35.5° N 82.6° W). The study area occupied the southwest-facing slope of a low, east-west trending ridge with elevations ranging from 715 to 775 m. Soils are deep (>80 cm), acidic (pH < 5.5), and

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relatively infertile. Annual precipitation averages 125 cm and is uniformly distributed throughout the year, although brief soil moisture deficits may occur during the late growing season. Mean daily temperature ranges from 2.3 °C in January to 22.5 °C in July. Composition of the present timber stand is an overstory mixture of dry-site oaks and hickories and a shade-tolerant midstory consisting primarily of red maple (*Acer rubrum*) and sourwood (*Oxydendron arboreum*). Yellow-poplar is a minor and variable canopy component throughout the stand, occurring with an estimated mean stem density of about one tree ha⁻¹ with greater frequency on lower slopes and in drainages. There is no record or evidence of fire in the study area. The study area is described in more detail in McNab (2016).

Study Design and Field Data

I utilized a small part of an upland oak stand that had been prescribed burned in late April 2013, which was part of a larger study (Keyser and Greenberg 2013). This 5.6-ha stand faces southwest, has an average slope gradient of 15 percent, and extends about 275 m from the crest of a primary ridge downhill across the upper and middle slopes. The land surface consists of broadly rounded secondary ridges separated by typically dry, shallow, wet-weather drains. In October 2014, two growing seasons after the late spring burn treatment, I located two adjacent, dominant yellow-poplar seed trees on an upper slope about 50 m down from the ridge crest. These two trees (hereafter Tree AB) were about 20 m apart, at the same elevation, and formed an oval that served as the center of a single seed source and the focal point from which my sampling design was based. Although yellow-poplar trees were on the opposite, mesic slope from my study area (i.e., northeast side), no other seed-producing yellow-poplar trees were nearby on the dry southwest-facing slope where my study was conducted. My four study questions required data associated with the seed source that were collected using three overlapping sampling designs.

Data for question 1 (i.e., distribution uniformity) were collected in a radius of 50 m from the focal point of Tree AB. The 50-m radius around the seed trees occupied much of the upper slope position of the ridge. The upper slope was slightly convex in profile and had little variability of topography along the contours because the small uphill drainage areas had not caused observable soil erosion. Using random azimuths and distances, I located 29 sample plots (0.45 m²) in a radius of 50 m from Tree AB. I also established a primary transect extending downhill from Tree AB focal point to a distance of 50 m. Three sample plots were established at 10-m intervals along the primary transect: one on each side 10 m at right angles (along the contour) from the transect (fig. 1). Data from the radius and transect sample plots consisted of the number of live yellow-poplar seedlings.

I used a two-sample t-test to test an hypothesis of no difference in the mean number of seedlings per plot in the radius around Tree AB compared to the mean seedling count in the downhill direction in the transect area.

Data for question 2 (i.e., seed source distance) were collected by extending the primary transect used in question 1 downhill across the middle slope, which in comparison with the upper slope displayed increased variation of topography along contours resulting from erosion associated with hillslope hydrology. This transect followed a shallow, linear, wet-weather drainage 220 m across the middle slope, where it ended at a large (diameter at breast height = 35.1 cm, height = 28.6 m) yellow-poplar tree (Tree C) growing in a narrow ravine formed by adjacent hill slopes on each side of the drainage my transect followed. I truncated the primary transect 50 m up slope from Tree C (i.e., 170 m down slope from Tree AB) to reduce the confounding influence of seedlings originating from it. To evaluate the magnitude of seedlings originating from Tree C on my sample data, I fitted a regression to seedling density from an earlier study in this stand (McNab 2016), which included Tree C, and used it to compare predicted with actual seedling densities. Similar to the sampling design for question 1, I established short secondary transects at 10-m intervals along the primary transect. At 50, 100, and 150 m along the primary transect, I extended the secondary transects to 50 m, with sample plots at 10-m intervals. One purpose of the longer secondary transects was to include in the field data variation of topography away from the concave surface of the primary transect, which followed the drainage. Data from these sample plots included density (hereafter DEN, k/ha) of yellow-poplar seedlings and distance (DIS, m) from Tree AB calculated as the hypotenuse of right triangles along the secondary transects. I used simple negative binomial regression to model DEN as a function of DIS because the variance of DEN greatly exceeded the mean of DEN.

Data for question 3 (i.e., site factors) were collected from sample plots on a further truncated section of the primary transect used for question 2. This 120-m section began at the middle slope (50 m down hill from Tree AB), ended 50 m up hill from Tree C, and included all plots established on the secondary transects. Data from sample plots included DEN, DIS, and additional observations consisting of overstory tree basal area (BA, m²/ha), density of non-yellow-poplar tree seedlings (SED, k/ha), density of tree species sprout competition (SPR, k/ha), estimated relative amount present of the pre-burn forest floor litter layer (LIT, percent), and meso-topography (TOP, classified as convex, planar or concave) 10 m around each sample plot. As for question 2, I used multiple negative binomial regression to model DEN as a function of DIS, BA, SED, SPR, LIT, and TOP.

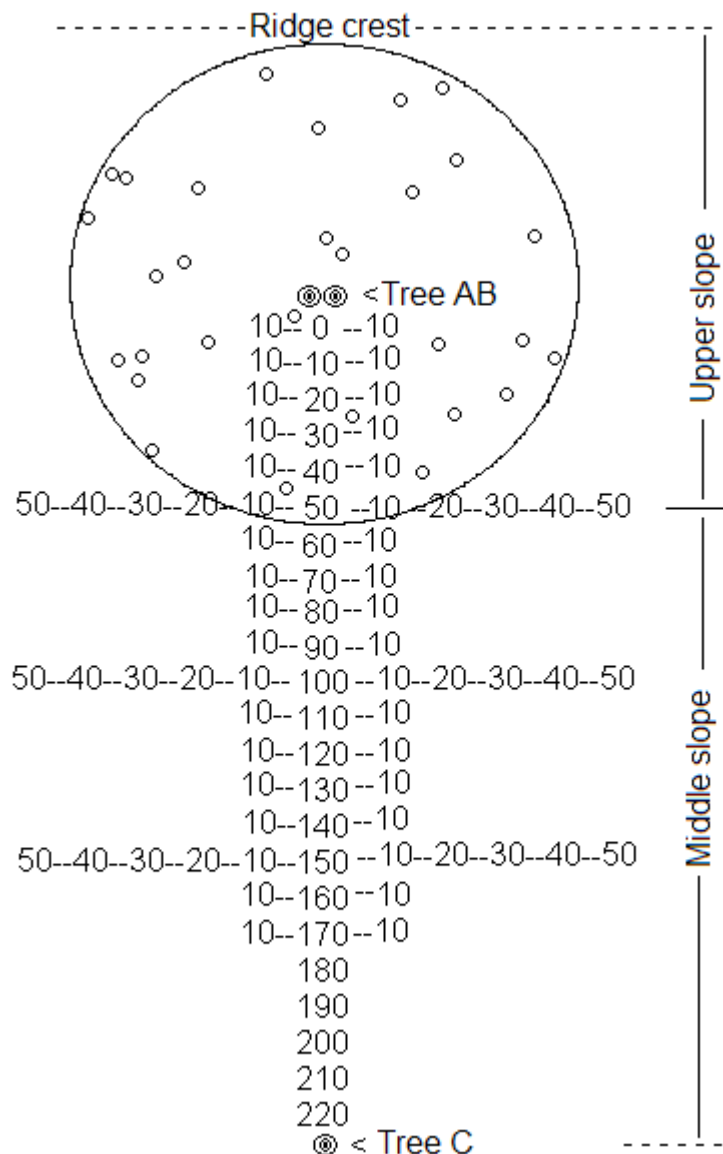


Figure 1 — Design, dimensions, and locations of sample plots in the study area of unit 4 in Bent Creek Experimental Forest. The large circle around the two seed trees (Tree AB) on the upper slope indicates the sample area within a radius of 50 m where small plots (represented by open circles) were established. The numbers 0–220 represent length (m) of the primary sample transect from Tree AB on the upper slope and the next encounter of a yellow-poplar seed tree on the lower slope, at Tree C. Numbers at right angles to the primary transect represent locations of sample plots at designated distances on secondary transects.

The three values of the categorical variable TOP were coded as two dummy variables representing the three subgroups of meso-topography.

Data for question 4 (i.e., seedling height) were obtained from the truncated primary and secondary transects used for question 3. Height (THT, cm) was measured for the tallest dominant yellow-poplar seedling present on each sample plot. Sample site data included that inventoried for question 3, except that TOP was replaced by terrain shape index (TSI), which is a

continuous variable for quantifying micro-topography of the plot itself (McNab 1989). In a manner similar to estimating TSI on large plots with a clinometer, I used a monopod device with top-mounted rotating horizontal cross arm to measure precise distance to the ground surface at the center and four equal quadrants around the plot perimeter and then calculated mean slope gradient, which is equivalent to TSI (McNab 1989). Using histogram plotting, I found the THT data were severely skewed right (i.e., many short and few tall seedlings), which I partially normalized with a square

root transformation. Sample plots lacking seedlings were excluded from the analysis. I used multiple regression to model THT as a function of DEN, BA, SED, SPR, LIT, and TSI.

I used Pearson correlation to examine simple relationships between independent variables to gain insight for development of model formulations, which I made based on my experience and judgment. I used a pseudo R^2 measure of goodness of fit for regression formulations that was calculated using deviance of regression (i.e., $R^2 = 1 - (\text{SS error} / \text{SS total})$, where SS = sum of squares). R statistical software was used for data analysis (R Core Team, 2013). Statistically significant differences were tested at the $\alpha = 0.05$ level.

RESULTS AND DISCUSSION

Field Data

Sample sizes, means, and variation of variables sampled for the four questions are provided in table 1. Diameters at breast height (d.b.h.) of the seed trees were 64.3 and 72.6 cm, and total heights were 31.2 and 36.1 m. Using the d.b.h. relationship of Carvell and Korstian (1955), I estimated that the combined annual sound seed production from Tree AB was approximately 93 thousand per hectare.

Distribution Uniformity

The average number of seedlings per plot for the radius sample at Tree AB was 7.2 compared to 6.1 for the downhill transect (table 1), which was non-significant

Table 1—Number, mean (standard deviation), minimum, and maximum values of selected variables grouped by sample data set of 2-year-old yellow-poplar seedlings at the study site in Bent Creek Experimental Forest

Data set and sample variables	N	Mean (SD)	Minimum	Maximum
Distribution uniformity				
Radius density (seedlings /plot)	29	7.2 (3.2)	2	15
Transect density (seedlings /plot)	54	6.1 (3.3)	1	14
Seed source distance ^a				
Distance from seed source (m)	74	90.9 (48.7)	0	170.3
Yellow-poplar density (K/ha)	74	56.4 (58.6)	0	220.5
Site factors ^b				
Distance from seed source (m)	59	108.2 (37.9)	50.0	170.3
Density yellow-poplar (K/ha)	59	36.6 (37.8)	0.0	161.7
Overstory basal area (m ² /ha)	59	22.5 (5.1)	11.5	32.1
Forest floor litter cover (%)	59	54.9 (16.9)	20	90
Misc. tree seedlings (K/ha)	59	10.3 (8.1)	0	36.7
Misc. tree sprouts (K/ha)	59	9.0 (7.8)	0	29.4
Topography category	59	— ^d	—	—
Seedling height ^c				
Yellow-poplar height (m)	52	0.08 (0.06)	0.03	0.34
Yellow-poplar density (K/ha)	52	41.5 (37.7)	7.3	161.7
Overstory basal area (m ² /ha)	52	22.9 (4.8)	13.8	32.1
Forest floor litter cover (%)	52	55.2 (16.0)	30	90
Misc. tree seedlings (K/ha)	52	10.6 (8.4)	0	36.7
Misc. tree sprouts (K/ha)	52	9.3 (7.7)	0	29.4
Terrain shape index	52	3.6 (5.1)	-5.0	25.8

^a From seed trees (1 m) to 170 m downslope.

^b From upper-middle slope transition (50 m from seed trees) to 170 m downslope.

^c From upper-middle slope transition (50 m from seed trees) to 170 m downslope where yellow-poplar seedlings were present.

^d Not applicable; sample plot meso-topography was classified in three categories: concave, planar, or convex.

($p = 0.13$). Lack of a significant difference between the two subpopulations of seedlings (i.e., radius DEN vs transect DEN) suggests uniform distribution around Tree AB without effects associated with direction, such as resulting from wind or slope, at least in the single transect direction I tested. My results differ from Engle (1960) who did not inventory yellow-poplar seedling density, but observed that seed distribution was strongly affected by direction from seed trees, particularly in reference to the prevailing wind during fall season when mature seeds were disseminated.

Seed Source Distance

Scatter plotting of the full transect field data indicated that DEN declined with increased DIS from Tree AB (fig. 2). A negative binomial regression model based only on DIS accounted for 56 percent of variation in DEN. The model predicts DEN declining from about 200 thousand seedlings/ha 1 m from Tree AB to 9 thousand seedlings/ha 170 m downhill, at the truncated end of the primary transect near Tree C. Under the assumption that yellow-poplar seedlings are associated with presence of seeds, results from my study are similar to findings of Whipple (1968) who reported seed fall for adequate regeneration occurred from two to five times the height of seed trees.

The presence of a single mature yellow-poplar seed tree at the lower end of the primary transect, Tree C, was a confounding source of variation in my study. Several

apparent “outliers” of high DEN values near the end of my truncated transect (see fig. 2) could have included seedlings originating from seeds dispersed by Tree C. I partially accounted for seedlings from Tree C by omitting data from sample plots within 50 m of it. Predicted DEN from Tree C based on data from a previous study (McNab 2016) was higher than actual DEN sample data, which supports my subjective observation of reduced influence from it.

Site Factors

Pearson correlation analysis indicated that DEN was significantly correlated with DIS ($r = -0.63$, $p < 0.01$) and SPT ($r = 0.31$, $n = 59$, $p < 0.05$). I used nonparametric Spearman rank correlation to evaluate the categorical variable TOP, which was significant ($\rho = -0.31$, $n=59$, $p = 0.02$). Multiple negative binomial regression of the truncated transect data revealed DEN was significantly influenced only by DIS and TOP. Predicted seedling DEN was lower on convex land surfaces compared to either planar or concave surfaces (fig. 3). Together these two variables accounted for 55 percent of variation in DEN. If convex sites in this study may be assumed drier than concave sites, then my results are similar to those of Shearin and others (1972) who reported lower density of yellow-poplar seedlings on drier areas of a clearcut and site-prepared hardwood stand in the Piedmont of South Carolina. The positive effects of concave topography on seedling density in my study agree with findings by

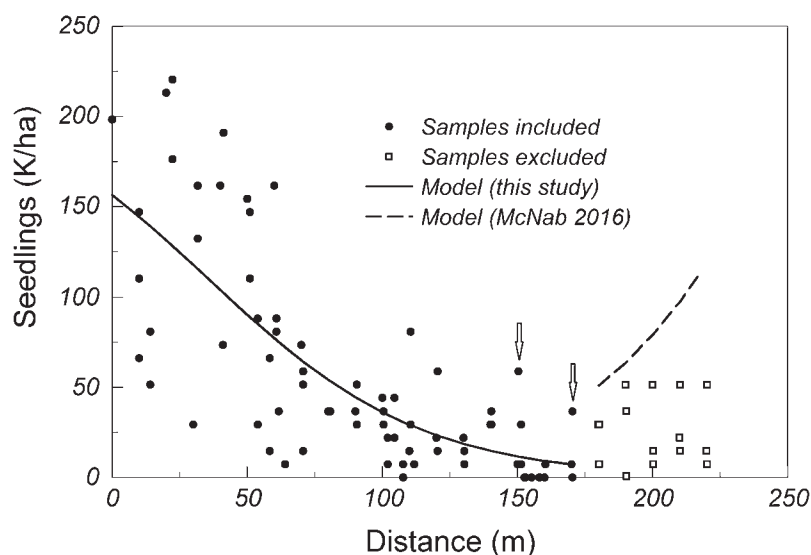


Figure 2—Actual and estimated density of yellow-poplar seedlings in relation to distance from seed trees 2 years following a growing-season prescribed burn in an upland oak stand in Bent Creek Experimental Forest. The solid line is a model fitted to seedlings included in the analysis that were inventoried on 74 sample plots extending 170 m downhill from Tree AB (represented by the closed circles). The open squares are seedling densities inventoried on sample plots within 50 m of Tree C, which were excluded from analysis. The dashed line is a model fitted to data from an earlier study in this area representing seedling density within 50 m of yellow-poplar seed trees (McNab (2016), which provides a means of assessing the confounding effects of Tree C on field data associated with Tree AB. The arrows identify two “outlier” sample plots with high seedling densities that could have resulted from the combination of seeds from both Tree AB and Tree C.

Whipple (1968) who reported greater density of yellow-poplar seedlings in harvested and site-prepared “upland bottoms” compared to other areas.

Seedling Height

Transformed seedling height was significantly correlated with TSI ($r = 0.30$, $n = 52$, $p = 0.05$) and SED ($r = -0.29$, $n = 52$, $p = 0.05$). Multiple linear regression analysis revealed that seedling THT was significantly influenced by TSI ($p < 0.01$) and SED ($p < 0.05$), and produced a model ($p = 0.01$) accounting for 17 percent of the variation in total height. Although TSI alone accounted

for only a small proportion of THT variation ($r^2 = 0.09$), the positive regression trend line indicates seedling heights increased with larger values (i.e., increasing concavity) of TSI (fig. 4). Effects of the relatively weak effect of SED are unclear and could result from an artifact of my non-normal data set (i.e., partially normalized after transformation) or the biological effect on vegetation resulting from competition for light. Dudley and Schmitt (1966) reported a direct relationship of *Impatiens capensis* height with density as a response to competition for light. Similar findings of increased seedling height with higher seedling density were

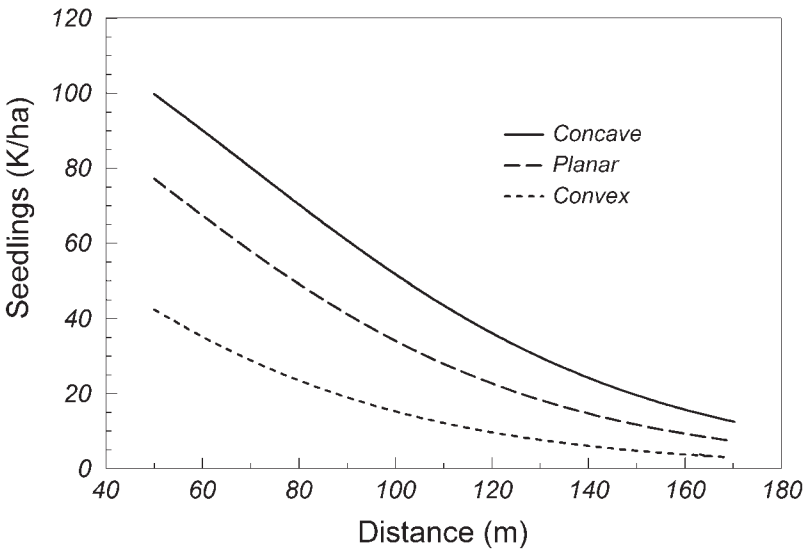


Figure 3— Estimated density of yellow-poplar seedlings in relation to distance from seed trees and category of land surface shape following a growing-season prescribed burn in an upland hardwood stand in Bent Creek Experimental Forest.

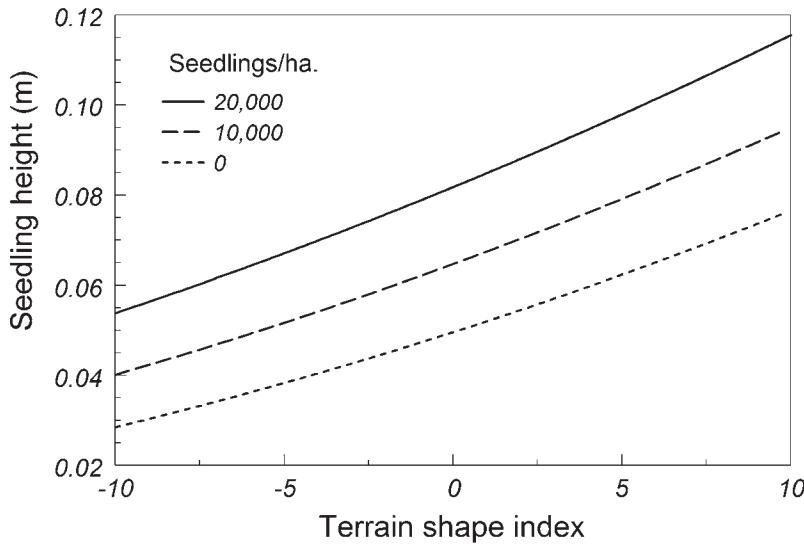


Figure 4— Estimated total height of yellow-poplar seedlings in relation to microtopography and density of miscellaneous tree seedlings 2 years after a growing-season prescribed burn in an upland hardwood stand in Bent Creek Experimental Forest.

reported by McNab and others (2013) in dense, “dog-hair” stands of 5-year-old yellow-poplar regeneration on mesic sites following controlled burning.

SUMMARY AND CONCLUSIONS

Results of this study supplement the sparse information available on the effects of prescribed burning on regeneration of yellow-poplar. Shearin and others (1972) and McNab and others (2013) reported that prescribed burning can increase regeneration of yellow-poplar on mesic sites, apparently by promoting germination of stored seeds in the forest floor. Although moderate research has been directed toward patterns of yellow-poplar seed distribution and quantification of seed production, the literature is largely devoid of results from studies similar to this one that identify topography as a site factor affecting regeneration density.

Results from this study extend findings from my previous research where I studied seedling density within 50 m of seed trees and derived models suggesting dispersal was likely well beyond that distance (McNab 2016). Yellow-poplar seeds are winged samaras that rotate during descent, which slows their rate of fall (McCutchen 1977). Long-distance dispersal of wind disseminated seeds in forests is increased by effects of wind that elevate seeds above the surrounding tree canopy (Nathan and others 2002). During the fall season when yellow-poplar seeds are released, mean annual wind speed peaks and its southerly direction aids in transporting seeds downslope from Tree AB. The combination of fall winds, tall seed trees, and their location on the lee side of the north-trending ridge were likely contributing factors accounting for the long-range dispersal of seeds that germinated into seedlings in my study area. Resource managers can use information from this study to evaluate the potential effects of prescribed burning on establishment of yellow-poplar regeneration at various distances from a seed source.

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COMPARING SHORTLEAF PINE ESTABLISHMENT BY SEED SOWING AND SEEDLING PLANTING ON A POOR ARKANSAS OZARKS SITE

H. Christoph Stuhlinger, Matthew Olson, and Michael McGowan

Extended abstract—Shortleaf pine's (*Pinus echinata*) natural range includes northern Arkansas and is considered to be adaptable to poor, rocky sites (Lawson 1990). Planting seedlings is difficult on rocky sites, so sowing shortleaf pine seed may be a viable alternative to seedling planting. This study/demonstration site compares broadcast sowing and spot sowing during two different times of year with spring bare-root and container seedling planting on a recent clearcut at the University of Arkansas Livestock and Forestry Research Station in the Ozark Highlands of northern Arkansas. First-year survival and growth results are presented.

This 10-acre site was previously wooded (low quality oaks and redcedar (*Juniperus virginiana*)). The soil is mapped as a Clarksville series very cherty silt loam which is somewhat excessively drained (WSS 2014). Site index for shortleaf pine is 55 (base age 50). The site was clearcut in 2013, and harvest debris was bulldozed into piles and burned. Afterwards, the entire site was control burned (results were spotty), followed by an imazapyr foliar application in fall 2015 prior to sowing and planting.

Six treatments were randomly assigned to 0.25-acre units, and treatments were replicated three times in a completely randomized design. The treatments (all improved shortleaf pine) are: 1) spring planted bare-root seedlings, 2) spring planted containerized seedlings, 3) fall spot-sown seed, 4) spring spot-sown seed, 5) fall broadcast seed, and 6) spring broadcast seed. Seedlings were grown at the Arkansas Forestry Commission's (AFC) Baucum Nursery near Little Rock. Seeds were obtained from the AFC seed orchard and had a 96-percent germination rating (39,184 seeds per pound). All seeds were treated with fungicide (42-S Thiram® and Bayleton 50 percent DF®), and spring-sown seeds were also stratified.

Spot-sown seeds were hand sown in late November 2015 (fall sown) and mid-April 2016 (spring sown) on a 10- by 10-foot spacing. Each spot was scraped bare, and then three to five seeds were sown per spot. Seeds were then pressed into the soil with the boot heel. Broadcast seeds were also sown in November 2015 and April 2016 (approximately 0.5 pound per acre). Bare-root and containerized seedlings were hand planted with a planting bar at a 10- by 10-foot spacing in February 2016. The number of spots for the spot-sown blocks and the number of planted seedlings per unit varied from 85 to 96 spots or seedlings.

Planted seedling rows were band-sprayed with Oust XP® in May. Seedling heights and ground line diameters (GLD) were measured on the interior (inside a border row) 48 seedlings in each unit after planting, and again at the end of the first growing season. Seed germination on the spot-sown and broadcast units was also checked at the end of the growing season.

ANOVA was run using the mixed procedure and significance was assessed at $p < 0.05$. Mean separation was performed using Tukey's Honestly Significant Difference. All analyses were run in SAS® 9.4.

Germination rates on the spot-sown and broadcast units were disappointing (table 1). On the spot-sown units, only 38.3 percent of the fall-sown spots had at least one germinant, while only 14.6 percent of the spring-sown spots had germinated. On a per acre basis, this rate corresponds with approximately 100 germinants for spring-sown and 250 for fall-sown plots. For the broadcast units, only 8.3 percent of the fall-sown sampling plots (2 square feet) had at least one germinant, while 10.4 percent of the spring-sown sampling plots had at least one germinant. However, on a per acre basis, that rate still represents about 3,700 germinants for spring broadcast and 3,100 germinants for fall broadcast. Competing weeds and grasses were likely a major limiting factor to germination. Additional seeds may germinate during the second growing season (Lawson 1990).

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Survival was significantly higher for containerized seedlings than bare-root seedlings, both initially (97.9 percent versus 88.9 percent) and at the end of the first growing season (95.8 percent versus 86.1 percent). In general, bare-root seedlings had a larger root system than containerized seedlings at planting, but top shoot sizes were similar. By the end of the first growing season, bare-root seedlings had more than doubled in diameter and almost tripled in height (fig. 1).

Final conclusions about the merits of sowing shortleaf pine seed on this site will be made at the end of the second growing season (fall 2017).

Table 1—Mean seedling stocking for three separate comparisons between fall and spring sowing by method of sowing and method of sowing irrespective of season

Comparison	Treatment	Broadcast (%)	Spot (%)
Season of sowing	Fall	10.4	38.3
	Spring	8.3	14.6
Method of sowing	—	9.4 b	26.5 a

Means with different letters differed statistically at $p < 0.05$.

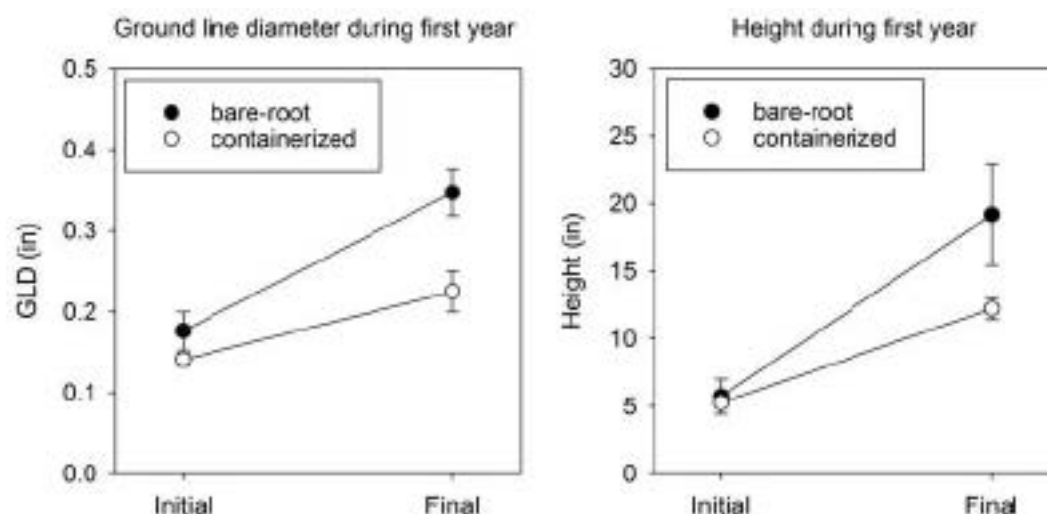


Figure 1—Although initial ground line diameters (GLD) and heights were similar for bare-root and containerized seedlings, the differences became significant (0.05 level) by the end of the first growing season.

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EQUATIONS FOR PREDICTING STAND LEVEL GROWTH AND SURVIVAL FOR CUT-OVER LOBLOLLY PINE PLANTATIONS IN THE MID-GULF REGION OF SOUTHERN UNITED STATES

Binayak Bartaula, Charles O. Sabatia, Thomas G. Matney, and Brent R. Frey

Extended abstract—Dominant height, diameter growth, and survival equations are important components of forest growth and yield modeling systems. Growth and yield equations provide information on future yields, which is used in making management and policy decisions. Stand growth and survival equations for loblolly pine in the mid-Gulf region of southern United States were previously developed by Matney and Farrar (1992). In the current study, improved equations for predicting stand dominant height, minimum and average stand diameters, and number of trees surviving in a stand, were developed. Tree data from 59 additional stands were incorporated in the analyses in addition to accounting for effects of physiographic region and hardwood competition and use of contemporary statistical methods in model fitting. The objectives of the current research were (1) to develop improved stand level height prediction, diameter prediction, and survival prediction equations for loblolly pine in the mid-Gulf region of southern United States; and (2) to identify the best set of models for practical application purposes.

Individual tree diameter and total height data from 377 measurement plots located in 115 stands in the mid-Gulf region of southern United States (Alabama, Arkansas, Louisiana, and Mississippi) were used to develop the equations. The stands ranged in age between 4 and 34 years. Each stand had at least one unthinned control plot and one or two plots that were thinned to 75 percent and to 50 percent of original basal area. Fifty-six of the stands, containing 200 measurement plots, were measured by Mississippi State University Loblolly Pine Growth and Yield Cooperative between 1981 and 1989. The other 59 stands, containing 177 measurements plots, were measured by the Forest Modeling Research Cooperative at Virginia Tech during the same time period. Dominant height; quadratic, arithmetic, and minimum stand diameters; and number of trees per acre were calculated for each plot. Data were then split into two parts that were both used for model fitting and model evaluation. Dominant height prediction models were fitted by nonlinear regression analysis using the MODEL procedure in SAS[®] ETS software. Diameter prediction models were fitted, as an equation system, by 3-stage least squares regression using SAS[®] ETS software. Models for predicting the probability of tree mortality in a stand were fitted by logistic regression using the LOGISTIC procedure in SAS[®] STAT software. Models to predict number of trees surviving in a stand were fitted by nonlinear regression in R software. To identify the best models, fitted models were evaluated based on two-fold cross validation prediction root mean square error. Final parameter estimates for the best models were estimated from the before-splitting data.

The generalized algebraic difference approach (GADA) model based on the Chapman-Richards equation was identified as the best dominant height prediction model. The resulting equation was:

$$H_1 = H_0 \left(\frac{(1 - \exp(-0.070939A_1))^{6.664187}}{(1 - \exp(-0.070939A_0))^{X_0}} \right)^{X_0}, \text{ Fit Index}=0.97, \text{ Standard error of estimate}=2.29 \text{ feet.}$$

where

$$X_0 = 0.5 \left((\ln(H_0) + \sqrt{(\ln(H_0))^2 - 4 \times 6.664187 L_0}) \right)$$

with $L_0 = \ln(1 - \exp(-0.070939A_0))$

In the equation, H_1 is the dominant height at age A_1 , and H_0 is the dominant height at the prediction age A_0 .

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An equation system to predict quadratic mean, arithmetic mean, and minimum stand diameter was constructed. The following were found to be the best equations of the system:

$$QMD = (103.869 - 11.401X_1 - 6.978X_2)N^{-0.194} \exp\left(\frac{-9.329}{\sqrt{H}} - 0.00478A - 0.00222hw\right),$$

Fit Index=0.94, standard error of estimate = 0.45 inch.

$$AMD = QMD - \exp\left(-1.452 - 0.906x \frac{\log(N)}{A}\right), \quad \text{Fit Index=0.99, standard error of estimate = 0.07 inch.}$$

$$DMIN = 0.1662 (AMD)^{0.0297}, \quad \text{Fit Index=0.75, standard error of estimate = 0.75 inch.}$$

In the equation system, *QMD* is the stand quadratic mean diameter, *AMD* the arithmetic mean diameter, and *DMIN* the minimum diameter; X_1 and X_2 are dummy variables for lower coastal plain and upper coastal plain, respectively; N is the number of trees per acre in the stand, A is stand age, and hw is the percentage (0 to 100 scale) of basal area that is hardwood. $X_1 = 1$ if lower coastal plain and zero otherwise. $X_2 = 1$ if upper coastal plain and zero otherwise.

The two-step survival prediction approach proposed by Woollons (1998) was found to work better than the one-step approach used by Matney and Farrar (1992). The equations that were found to work best with the Woollons (1998) approach were the probability-of-mortality equation:

$$p(Mort) = \left(\frac{\exp(-11.2698 + 0.2146A_0 + 0.0750SI + 0.00932N_0)}{1 + \exp(-11.2698 + 0.2146A_0 + 0.0750SI + 0.00932N_0)} \right)^{\frac{1}{3}}$$

and the surviving-number-of-trees equation:

$$N_1 = \left[N_0^{-0.42519} + 0.07121 \left(\frac{SI}{10000} \right)^{0.067368} (Age_1 - Age_0) - 0.02175 \times \ln \left(\frac{A_1}{A_0} \right) \right]^{-\frac{1}{0.42519}}$$

Fit Index=0.94, standard error of estimate = 52 trees per acre

In these equations, $p(Mort)$ is the probability of mortality over a 1-year period for a stand that is A_0 years old, whose base-age-25 site index is SI , and containing N_0 number of trees per acre; N_1 is the number of trees per acre that are expected to be surviving in the stand at age A_1 , before accounting for the probability of mortality in the stand. With the two equations, the number of surviving trees at age A_1 , in a stand containing N_0 trees per acre at age A_0 , is computed as:

$$N_{adj} = N_0 - p(Mort)^{(A_1 - A_0)}(N_0 - N_1)$$

The parameter estimates of the equations developed in the current study exhibited logical behavior and were all significant at $p \leq 0.05$. The equations should be reliable for predicting future dominant height, average and minimum diameters, and number of trees per acre for non-intensively managed loblolly pine stands in the mid-Gulf region of southern United States. The significant dummy variable parameter estimates in diameter prediction equations implied that for stands on sites of the equal productivity potential, diameter growth in stands in the lower Gulf coastal plain may be smaller than that in stands in other regions of the mid-Gulf region in southern United States.

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SIXTEEN-YEAR STAND LEVEL GROWTH AND DEVELOPMENT OF VARIETAL AND NON-VARIETAL LOBLOLLY PINE IN THE ATLANTIC COASTAL PLAIN OF SOUTH CAROLINA

Charles O. Sabatia and Harold E. Burkhart

Extended abstract—Most loblolly pine plantations in southern United States are currently established using genetically improved planting stock with both varietal and non-varietal planting material available for use. Approximately 2 percent of the close to 1 million acres of loblolly pine planted in Southern United States annually are currently established with varietal planting stock of genetically identical trees (McKeand and others 2015). Stands of genetically identical loblolly pine trees are expected to supply high quality products that are more uniform in characteristics. However, the effect of the genetic uniformity on growth and yield characteristics of genetically improved loblolly pine stands is currently not well documented, making it difficult to make reliable growth and yield projections for such stands. In the current study, dominant height, basal area, survival, and inter-tree competition characteristics were compared for non-varietal and for varietal loblolly pine growing in block planting experimental stands at two locations in Charleston County, SC. The objective of the study was to determine the differences/similarities in dominant height and basal area growth, survival, and inter-tree competitive interactions between varietal and non-varietal loblolly pine stands of elite genetic material.

The block planting experiment analyzed in the current study consisted of four genetic entities: an open pollinated (OP) family (family 7-56), a control pollinated (CP) family (7-56 x 11-12), and rooted cutting varieties AA32 and AA93. The experiment was planted in 2000 at two locations. Each location had one replicate of each genetic entry in a block of 100 trees. One of the locations was planted at a spacing of 9 feet by 9 feet (538 trees per acre), while the other location was planted at 12 feet between rows and 7 feet within rows (519 trees per acre). Total control of competing vegetation was implemented in the stands for the entire duration of the study. Individual tree measurements were taken at both locations at the ages of 9, 10, 11, 12, and 16.

Dominant height, basal area development, and stand density versus tree size relationships were compared using graphical plots. The effect of genetic entity on the rate of change of basal area with increasing age, between age 9 and 16, was tested using the linear statistical model:

$$\ln(BA_{ij}) = \left(b_{00} + \sum_{i=1}^3 b_{0i} G_i \right) + b_1 \ln(BA9_{ij}) + \left(b_{20} + \sum_{i=1}^3 b_{2i} G_i \right) \frac{1}{Age} + b_3 Loc_j + \epsilon_{ij} \quad [1]$$

where

$\ln$ is natural logarithm

BA_{ij} is basal area per acre in the i^{th} genetic entity stand in the j^{th} location

$BA9_{ij}$ is the basal area in this stand at age 9

G_i is the effect of the i^{th} genetic entry

Loc_j is the effect of the j^{th} location

ϵ_{ij} is random error with zero expectation, and

bs are parameters.

The effect of genetic entry on neighborhood competition, and hence effect on diameter growth, among individual trees in a stand was investigated using the nonlinear statistical model:

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$$\Delta dbh_{ijk} = \exp(b_0) dbh_{ijk}^{b_1} \exp\left(\left(b_{20} + \sum_{i=1}^3 b_{2i} G_i\right) Cl_{ijk}\right) + \varepsilon_{ijk} \quad [2]$$

where

Δdbh_{ijk} is the age 12 to 16 increase in breast height diameter of the k^{th} tree of the i^{th} genetic entity in the j^{th} location

dbh_{ijk} is the age 12 dbh of the tree

Cl_{ijk} is the neighborhood competition index for the tree, and

ε_{ijk} is random error with zero expectation.

In equation 2, the neighborhood competition index was computed as:

$$Cl_{ijk} = \sum_{l=1}^n \frac{BA_{ijl}}{d_{kl(ij)}} \quad [3]$$

where

BA_{ijl} is the basal area of the l^{th} neighbor to the k^{th} tree, and

$d_{kl(ij)}$ is the distance between the k^{th} tree and its l^{th} neighbor, in the i^{th} genetic entry stand in the j^{th} location

Competitive neighbors for the k^{th} tree were selected by point sampling with a basal area factor 10 prism centered at the subject tree. Equations 1 and 2 were fitted to the data using SAS® software

Dominant height at age 16 ranged between 64 and 73 feet with the varietal stands being, on average, 4 to 6 feet taller than the non-varietal stand (OP and CP). Basal area ranged between 151 and 197 square feet per acre with the non-varietal stands carrying, on average, 15 more square feet of basal area per acre than the varietal stands. Stand density index at age 16 ranged between 293 and 389, which was well above 55 percent of the maximum stand density index for loblolly pine. There was evidence of self-thinning in the stands (fig. 1) with the non-varietal stands exhibiting a slightly more aggressive self-thinning than the varietal stands. Estimates of b_{20} and b_{2i} parameters in equation 1 indicated that between age 9 and age 16, basal area in varietal stands increased faster than it did in non-varietal stands. The faster growth was, however, not sufficient for basal area in varietal stands to catch up with that in non-varietal stands by age 16. In equation 2, estimates of the parameters b_{2i} were not significant. Thus, tree neighborhood competition did not affect individual tree diameter increments differently in varietal versus non-varietal stands. Diameter, dominant height trends, and inter-tree competition trends exhibited in the current study were similar to the trends that have been reported for different, but younger, stands with the same genetic entries (Sabatia and Burkhart 2012, Sabatia and Burkhart 2013).

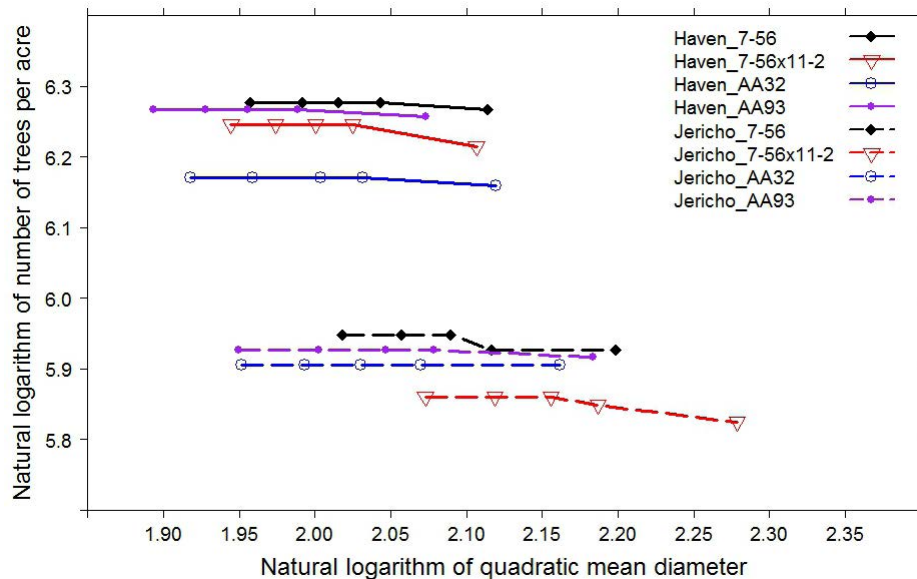


Figure 1—Tree size-density relationships in each of the eight stands used in the current study. The legend prefixes “Haven” and “Jericho” represent the two locations (blocks) of study stands.

ACKNOWLEDGMENTS

The block planting experiment was established by MeadWestvaco and is currently maintained by ArborGen®. Data for earlier ages were provided by the Forest Modeling Research Cooperative at Virginia Tech. Age 16 data collection, and data analysis, were funded by the Forest Modeling Research Cooperative at Virginia Tech under a specific memorandum of understanding with Mississippi State University. Review comments by Scott Roberts of Mississippi State University and Thomas Fox of Virginia Tech helped improve the quality of this publication.

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IMPROVING GROWTH AND YIELD ESTIMATES FROM GROWTH INDEX RATIO METHOD OF STAND TABLE PROJECTION IN BOTTOMLAND RED OAK–SWEETGUM HARDWOOD STANDS

Leah F. Leonard, Charles O. Sabatia, Thomas G. Matney,
Emily B. Schultz, and Theodor D. Leininger

Extended abstract—Forest managers rely on growth information in order to project future timber volumes and financial returns. Growth-index-ratio-based stand table projection (STP_{GI}) is one of the approaches that foresters can use to achieve this. STP_{GI} does not use complex equations that require large amounts of data to parameterize. All that is needed is information on diameter growth of sample trees from the stand in question. Thus, STP_{GI} is easier to develop and apply than more complex modeling methods. It is the projection method that is the best option for stand types where suitable growth-and-yield models do not exist. In STP_{GI} , trees are grouped into diameter classes, and a growth index (GI) ratio for each diameter class, calculated from growth measurements on increment cores, is used to “grow” the trees in the diameter class (Avery and Burkhart 2002). Trees are assumed to be uniformly distributed in each diameter class. STP_{GI} can, however, be inaccurate for predicting future quantity of timber in a stand. Problems occur when the assumption that ‘past growth in a given diameter class is the same as future growth in that class’ does not hold.

Individual tree measurements from 83 red oak–sweetgum mixed species forest stands in minor stream bottoms in Mississippi were used to investigate how well STP_{GI} worked for these forest types. The data were collected from sample plots, of size between 0.1 and 0.4 acre, in 1981, 1988, 1992, and 2005. The average species composition of the stands was: cherrybark oak–9.5 percent, other red oaks–16.9 percent, white oak–3.0 percent, sweetgum–51.9 percent, hickory–3.6 percent, other commercial species–7.2 percent, and non-commercial species–7.7 percent. The main objective of this research was to determine the range of stand ages where STP_{GI} projections were biased and the range where the projections were unbiased. An additional objective was to determine whether applying GI ratio to individual trees rather than diameter classes (a GI-ratio-based individual tree projection) could be a more accurate projection approach.

Year 1993 was used as the projection time. Diameter growths for year 1981 to 1993 (12-year period) were treated as previous diameter growths (“increment cores”) and year 1993 to 2005 (12-year period) was treated as the projection period. Study data did not contain year 1993 diameter measurements. To resolve this, a one-year linear extrapolation was applied to 1988 to 1992 diameter growths to provide year 1993 diameter measurements. One previous diameter growth was randomly sampled per diameter class per species to provide a GI ratio for the class. One-inch diameter classes were used and hence GI ratio for a diameter class was obtained by dividing the previous diameter growth by 1. The GI ratio was then used to project 2005 diameters from 1993 diameters; first by applying the GI ratio to diameter classes ($STPGI$) and, secondly, by applying GI ratio to individual trees (GI ratio individual tree projection). Trees surviving in each diameter class were assumed to be predictable without error, and hence the number of trees per acre observed in 2005 were the ones assigned to the projected diameters/diameter classes. Projected and observed year 2005 diameter distributions were used to evaluate the projection approaches. The evaluation was based on per acre green tons in the observed and projected diameter classes. The green tons were computed for each species in a diameter class using the green weight equations of Clark and others (1985). Percent green weight prediction error for each stand was then computed, for each projection approach, as:

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$$\frac{\sum_{i=1}^n (\hat{W}_i - W_i)}{\sum_{i=1}^n W_i} \times 100 \quad [1]$$

where $\hat{W}_i$ is the projected, and W_i the observed, green tons per acre in the i^{th} diameter class and n is the number of diameter classes in the stand.

Green weight prediction errors for the 83 stands, investigated in the current study, are shown in Figure 1. Errors from GI ratio individual tree projection were not necessarily smaller than those from GI ratio projection based on diameter classes (STP_{GI}). Therefore, applying GI ratio on individual trees rather than diameter classes may not result in more precise stand yield projections in bottomland red oak–sweetgum hardwood stands in Mississippi.

STP_{GI} generally worked well in the 30 to 70 year range (projection errors were generally within 10 percent) but over predicted timber yields for stands that were less than 30 years old and under predicted the yields for those that were older than 70 years. It should, however, be noted that the error trends in Figure 1 were computed under the assumption that number of trees surviving in the stand could be predicted without error. The trends are likely to be different if survival prediction error is taken in to account.

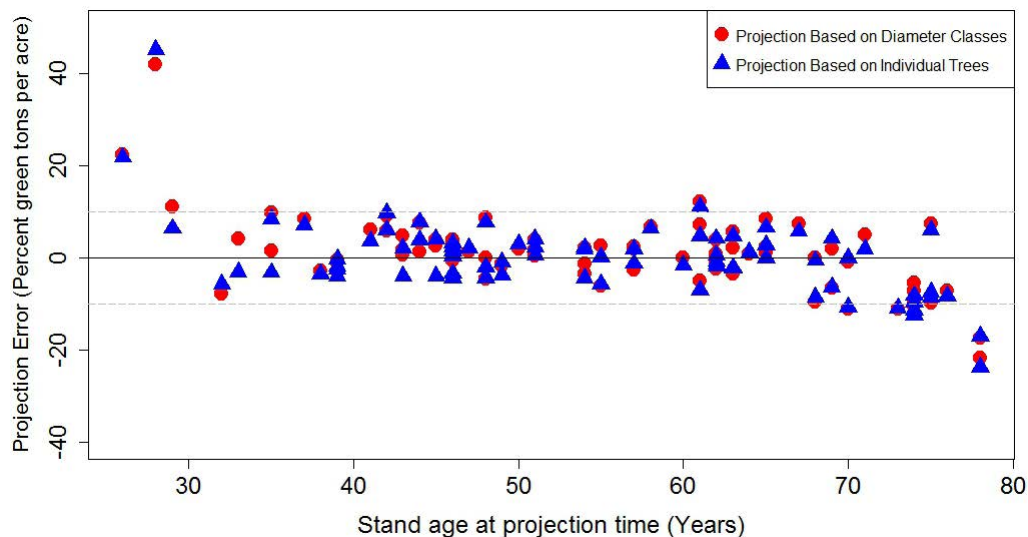


Figure 1—Plots of percent green ton prediction errors from GI-ratio-based diameter growth projection based on diameter classes, and the projection based on individual trees, for bottomland red oak–sweetgum stands in Mississippi. The gray dashed lines indicate the 10-percent projection error limit.

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PLOT SIZE AND PREDICTION MODEL FORM EFFECTS ON STAND DIAMETER DISTRIBUTION RECOVERY METHODS

Josh B. Bankston, Charles O. Sabatia, Thomas G. Matney

Extended abstract—Diameter distribution information of a forest stand is an important source of information on the stand's value. In whole stand forest growth and yield model systems, diameter distribution is predicted by use of models that predict the stand's diameter moments and/or percentiles in conjunction with a probability distribution system to recover the diameter distribution from the predicted moments and/or percentiles. The Weibull probability distribution is widely used in forestry. Recovery methods commonly applied include the method of moments approach (MOM) (Hyink and Moser 1983), the hybrid approach (HYB) (Baldwin and Feduccia 1987), and the percentiles approach (PCT) (Bailey and others 1989). A few studies have compared the recovery approaches to determine the superior approach. An extensive literature search indicates that there has been no agreement as to which approach is superior. In the current study, it was investigated whether the size of plot used in a study could affect the conclusion regarding which approach is superior. Also investigated was whether the moment/percentile prediction model form affects precision of recovered diameter distribution.

Data used in the study consisted of individual tree data from 336 quarter-acre plot re-measurements from 56 loblolly pine stands across Alabama, Arkansas, Louisiana, and Mississippi. Age of the stands ranged between 4 and 34 years, and stand density ranged between 100 and 1213 trees per acre. Some of the stands were thinned. Data collection was done by the Mississippi State University Loblolly Pine Cooperative between 1981 and 1989.

From each 0.25-acre plot re-measurement data, smaller virtual plots of sizes 0.05, 0.1, 0.15, and 0.2 acre were created by proportional (by size of virtual plot) random sampling of the trees, without replacement, from the 0.25-acre plot. This process resulted in five sets of 336 plot observations. Dominant height, relative spacing, and number of trees per acre were computed for each plot. Quadratic mean and arithmetic mean diameters; minimum diameter; and the 25th, 50th, 93rd, and 95th percentiles for each plot were also determined. The following equations were selected for use as the moment/percentile prediction equations:

$$D_{M/P} = \exp\left(b_0 = b_1 RS + b_2 \ln(TPA) + b_3 \ln(HD) + \frac{b_4}{Age}\right) + \epsilon \quad [1]$$

$$D_{M/P} = b_0 TPA^{b_1} \exp\left(\frac{b_2}{\sqrt{HD}} + b_3 Age\right) + \epsilon \quad [2]$$

$$D_{M/P} = b_0 Hd^{b_1} TPA^{b_2} \exp\left(\frac{b_3}{Age}\right) + \epsilon \quad [3]$$

Equations 1, 2, and 3 were adopted from Cao (2004), Matney and Farrar (1992), and Baldwin and Feduccia (1987), respectively. Variables in these equations are:

$D_{M/P}$ = the plot diameter moment or percentile,
 RS = relative spacing,
 TPA = number of trees per acre,
 HD = dominant height,
 b_i (for $i = 1, 2, 3$, or 4) = equation parameters, and
 ϵ = random error with zero expectation.

The equations were fitted, for each moment and percentile, to each of the five data sets, by nonlinear regression using R software.

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The fitted equations 1 to 3 were used to calculate each plot's diameter moment or percentile for the data on which the equation was fitted. A predicted Weibull diameter distribution was then recovered for the plot by MOM, HYB, and then by PCT. The location parameter in MOM and HYB recovery approaches was estimated as one half of the plot minimum diameter.

A total height was then assigned to each observed and each predicted diameter class using the Matney and Farrar (1992) height-diameter equation:

$$H = 5.602 Hd^{1.0021} \exp\left(-1.76969 \left(\frac{QMD}{dbh}\right)^{0.25}\right) \quad [4]$$

Where

H = the average height of the diameter class dbh , and

QMD = the plot's quadratic mean diameter.

Green tons were then computed for each observed and each predicted diameter class using green weight equations of Bullock and Burkhart (2003).

Evaluation of the diameter recovery approaches was done for each of the 15 combinations of plot size, moment/percentile prediction equations, and Weibull diameter distribution recovery approaches, using a green-ton-weighted Reynolds and others (1988) error index that was computed separately for thinned and unthinned plots as:

$$Mean\ Error\ Index = \frac{1}{N} \sum_{i=1}^N \sum_{k=1}^{m_i} |W_{ik} n_{ik} - W_{ik} \hat{n}_{ik}| \quad [5]$$

where

N = the number of thinned/unthinned plots of the 336 in the study,

W_{ik} = the green tons in one tree in the k^{th} diameter class in the i^{th} plot,

n_{ik} = the observed and $\hat{n}_{ik}$ = the predicted number of trees per acre in the k^{th} diameter class of the i^{th} plot for $i = 1$ to N , and

m_i = the number of diameter classes in the i^{th} plot.

The mean error index for the different combinations of plot size, moment/percentile prediction equations, and distribution recovery methods, are summarized in Figure 1 for thinned plots in the data sets. Results for unthinned plots exhibited similar trends, but the MOM, HYB, and PCT trend lines were essentially not different.

From the trends in Figure 1, plot size may not affect the conclusion as to which approach is superior if comparing between MOM and PCT, but it may have some effect if comparing between HYB and PCT. Conclusions as to which approach is superior may not be effected by choice of moment/percentile prediction model. For thinned stands, the PCT approach may be superior to the MOM approach.

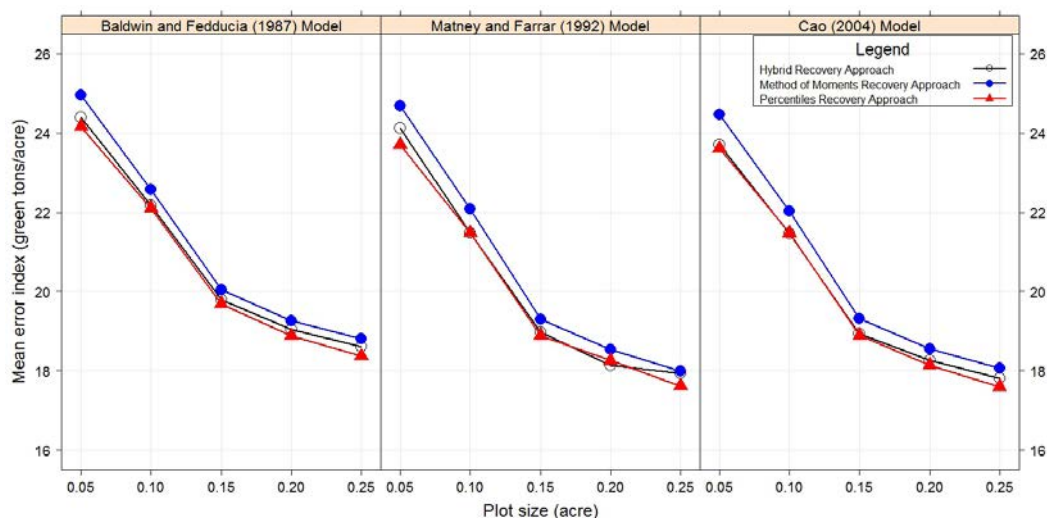


Figure 1—Across-plot-size error index trends for the Weibull diameter distribution recovery approaches investigated for the selected moment/percentile prediction models

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GREEN WEIGHT, TAPER, AND VOLUME EQUATIONS FOR LOBLOLLY PINE IN OKLAHOMA, USA

Will T. Harges and Thomas B. Lynch

Extended abstract—Equations to estimate merchantable green weight, merchantable volume and taper for tree boles based on diameter at breast height (diameter at 4.5 feet, dbh), total tree height, and a specified merchantable diameter or height limit are indispensable to forest inventory. These equations have been developed for loblolly pine (*Pinus taeda* L.) over most of the area where it is grown. Some equations have been developed for specific growing regions within the South, while others are based on data representing most of the South.

There is currently one set of equations available for Oklahoma (Clark and others 1991); however, it is not adequate for some forest management needs because only data from natural stands are included, weight estimation is not included in the original document, and the system is complex. Additionally, because Oklahoma loblolly pine is primarily grown outside the natural range, range-wide prediction systems are likely biased for predictions limited to this smaller growing region. These facts indicate that Oklahoma forest managers will benefit from a loblolly pine stem content and taper prediction system fitted to data from Oklahoma.

To assess this need and, if necessary, ameliorate it, we collected data from 158 trees located in industrial plantations in the Ouachita Mountains of southeastern Oklahoma, USA. This area is near the northwest extreme but outside of the natural range of loblolly pine. The sample trees came from 22 stands across a range of site indices, ages, and tree sizes. Taper data were collected inside and outside bark at 0.5, 1, 2.5, 4.5, and intervals of 4 (feet) thereafter. Tree bole green weight with bark was determined in the field by weighing bolts corresponding to the lengths between taper measurements.

We compared Oklahoma loblolly pine to southwide populations using paired t-tests of the differences between true values and values predicted from southwide equations for green weight and taper. After analyzing the data, we found statistically and practically significant differences (5–10 percent over prediction for mature log green weight). We then used the merchantable-to-total stem content ratio method to estimate merchantable tree content; taper was estimated using compatible equations derived from the merchantable tree content equations. Ratios and total content were fit as a single equation to cumulative tree content profiles consisting of the green weights or volumes of bolts summed from the stump to the points along the stem where taper measurements were taken.

We selected the nonlinear version of the logarithmic total tree content equation (Schumacher and Hall 1933), the exponential merchantable diameter ratio equation (Parresol and others 1987, Van Deusen and others 1981), a new merchantable height ratio equation (Zhao and Kane 2017), as well as a new taper equation (Lynch and others 2017) derived from the merchantable height ratio equation. We compared these equations to several other well-known models using AIC, BIC, the root of the mean square error, mean bias, the pseudo R^2 , parameter standard error size, and normality assessments. We utilized a power of the mean variance structure to model variance for a weighted nonlinear ordinary least squares estimation that could account for non-constant error variance in our data. We used this procedure to develop prediction equations for loblolly pine merchantable outside bark green weight as well as merchantable inside and outside bark volume. We used parameters from the merchantable height based volume equation in the taper equations to predict inside and outside bark taper. This work produced equations that have good predictive ability for trees across a wide range of conditions present in the Ouachita Mountains of southeastern Oklahoma and that are simple to use.

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OPTIMAL SAMPLE SIZE OR POINT SAMPLING FACTOR BASED ON THE COST-PLUS-LOSS CRITERION

Thomas B. Lynch

Extended abstract—The cost-plus-loss principle was used to develop methodology for simultaneously determining optimal sample size and optimal plot size or point sampling factor using the Fairfield-Smith relationship between plot size and the variance among plots. The expected absolute value of the difference between true mean and sample mean volumes per hectare was used to develop a loss function in terms of United States Dollar (USD) value. Sampling costs included USD values of plot measurement and travel time costs. The resulting cost-plus-loss function was minimized by differentiating with respect to plot size and sample size. The optimal plot and sample size values were determined by setting these differentials equal to zero and solving. Example solutions are presented based on realistic stumpage and cost values for the Southern United States.

At least three major approaches have been taken to the determination of optimum sample size for forest inventories: determination of sample size needed to achieve a specified standard error of the mean, determination of sample size that minimizes the standard error of the mean for a fixed total cost, and determination of sample size that minimizes total cost-plus-loss. Determination of the sample size needed to achieve a specified standard error of the mean is probably the most commonly discussed method and is usually presented in forest mensuration texts. The method is often presented as determination of the sample size required to achieve a specified error with a specified confidence level. Another approach to determination of sample size is to minimize the standard error of the mean for a given total sampling budget. This approach has been applied in forestry by Mandallaz (2008). The principle of minimizing standard error subject to a fixed total sampling budget has been used in agriculture. Hamilton (1979) investigated the approach to sample size determination of minimizing the cost-plus-loss for forest sampling plans. Burkhart and others (1978) also used the cost-plus-loss framework to determine the best sampling intensity for inventories to be used for multiple resource planning. Zeide (1980) developed a technique for simultaneously determining optimal plot and sample size using a relationship between plot size and variance due to Freese (1961), which is a special case of the relationship between plot size and sample size presented by Fairfield-Smith (1938).

A cost-plus-loss function was developed by expressing the loss as the USD dollar value of the expected absolute value loss with a normal distribution. The variance of this normal distribution was expressed as a function of plot size using the Fairfield-Smith relationship between plots size and variance among plots (Smith 1938). Plot establishment and travel costs were added to the loss functions. The plot cost was developed by using Zeide's (1980) function for plot measurement time as a function of plot size. This plot cost function was multiplied by the number of plots and the sampling wage rate. An expression for travel time was also added to the loss function and the plot costs. Zeide (1980) indicated that the average travel distance for a sample of n plots will be the square root of the product of total tract size and the number of plots. This distance was multiplied by the average rate of travel and the wage rate for sampling personnel. Fixed costs were not included because they do not affect the optimal plot and sample size calculations. This is due to the fact that the mathematical derivative of fixed cost with respect to plot size or sample size is zero.

Reasonable values from the literature were used to parametrize the plot measurement time function and to determine wage rates and stumpage values which would be realistic for the Southern United States. The cost-plus-loss function was minimized by differentiating the cost-plus-loss function with respect to plot size and sample size. The cost-plus-loss function can also be minimized directly using appropriate computing software or within widely available spreadsheet applications. The examples indicate that the cost-plus-loss criterion tends to result in higher sampling intensities than would be typical in the Southern United States. For the example scenarios developed, the cost-plus-loss surface is rather flat near the optimal values of sample and plot size, so

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there are a wide range of combinations of plot and sample size that are associated with cost-plus-loss values close to optimum. Although the cost-plus-loss criterion results in somewhat high sampling intensities, sampling costs under this criterion were only about half of one percent of timber value for the scenarios tested. This level of sampling might be justified in situations where very precise valuations are desired, such as inventories for timber sales.

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ALTERNATIVE COLLECTION METHODS FOR RED SPRUCE (*PICEA RUBENS*) CONES

John R. Butnor, Kurt H. Johnsen, Robert Eaton,
Thomas Christensen, and Chris A. Maier

Abstract—Commercially available pole pruners, hydraulic lifts, firearms, and certified tree climbing personnel are the most common means of collecting single source seeds from individual trees. These methods are highly productive at suitable locations, but have serious shortcomings for collecting red spruce (*Picea rubens* Sarg.) cones at high elevations in both the southern highlands and Northeastern States. Rough, inaccessible terrain precludes hydraulic lifts and makes collections by tree climbers' time consuming, while commercial pole pruners are difficult to handle once extended beyond 6–7 m. We present a novel pole pruner that is capable of reaching heights of up to 12 m, expanding the range and ease of use over commonly available pruners. Alternative collection methods are also presented, including cutting cone bearing limbs with specialized chainsaw chains and shaking cones loose from trees with ropes from heights of up to 25 m. Strategies for setting lines and removing and securing cones with ropes are offered.

INTRODUCTION

For restoration or general purpose plantation stock, most conifer seed are produced in open pollinated seed orchards or collected from the ground in forest stands. These seeds are orchard mixes from different parent trees with a variety of traits. In order to identify specific heritable traits, such as growth rates, phenology, crown structure, etc., it is helpful to distinguish between different families or offspring sharing a common mother tree. Commercially available pole pruners, hydraulic lifts, firearms, and certified tree climbing personnel are the most common means of collecting single source (half-sib) seeds from individual trees. Red spruce (*Picea rubens* Sarg.) are usually found in mid to high elevation stands across its range, and many remote populations are accessible only by foot. Rough terrain precludes hydraulic lifts and makes collections by climbers time consuming, while commercial pole pruners are difficult to handle once extended to lengths of 6–7 m. Red spruce cone are usually viable by late August to early September. The resinous cones (fig. 1) mature rapidly in mid to late September and begin to release seed. This short collection window coincides with fall foliage and recreation by the public in scenic mountain areas, which may be incompatible with shooting cones out of trees with firearms, especially in State and National parks.

OUR APPROACH

Red spruce requires substantial light exposure to stimulate cone production. Red spruce seedlings and advanced regeneration are shade tolerant and may survive for decades in the understory (Harlow and others 1978), but shaded trees grow very slowly and generally do not produce cones. Stunted trees on mountain tops or trees in sunny locations with extensive live crowns that touch the ground may have easy-to-reach cones by the age of 30 to 40 years (Harlow and others 1978). Conversely, in closed canopy forests under heavy competition for light, cones may be present at the very top of co-dominant trees and not visible to individuals on the ground. On quality sites, trees may reach 45 m in height with cones located near the top of the tree (fig. 2). Depending on accessibility of cones, single source seed are collected by: 1) hand if close to the ground, 2) ground collection if there is no possibility of cones



Figure 1—Resinous red spruce (*Picea rubens*) cone photographed at Mount Mitchell State Park, North Carolina, in September 2016. (Photo by Chris Maier)

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originating from other trees, 3) pole pruner, 4) setting lines and sawing branches, or 5) setting lines and shaking branches. Whenever cones are not accessible by hand, pole pruners are the most expedient option.

Long-Reach Pruner

Commercially available pole pruners are usually telescoping, reaching 2–5 m, or sectional where ~2-m locking poles may be connected in series as high as the user deems prudent. Once extended to >6 m, the poles become heavy and very flexible, limiting their utility. To extend the range of the pole pruners, we constructed a 12-m long-reach pruner by putting a pruning head atop a telescoping tree height pole (fig. 3). The unit is best deployed with at least two people as it gets unwieldy when fully extended. The pole is raised through the canopy, passing by and through lower branches which give the pole some support. In dense canopies, a spotter that can see the cone branch directs the individual manipulating the pole to the desired branch. It is imperative that the pole be raised up and down vertically; any attempt to lower the pole horizontally will lead to loss of control and damage to the equipment. Telescoping poles ranging from 14 m to 31 m are now commercially available [e.g., Geodata Systems Management Inc., Berea, OH (<http://www.geodatasys.com/>)]; shorter, heavy duty poles (<14 m) may be custom outfitted with a pruner head and saw. Given that red

spruce have stiff, strong lateral branches, it is common to have cut branches hang or get lodged in lower branches, so planning for this situation is necessary.

Sawing Cone Bearing Branches

Beyond the range of long-reach pole pruners (12–14 m), we developed a technique to collect cones as high as 25 m using ropes and specially constructed chainsaw chains. On branches at least 3 cm in diameter, a throwline is set using a slingshot and shot bag (fig. 4). The thin throwline is replaced with a strong 5-mm cord attached to a specialty saw chain (PockeTech HighLimb2-48 inch). The chain resembles a common saw chain, but it has bladed chisels on both sides of the chain and in both directions (fig. 5). This is a vast improvement over chains that only have chisels on one side of the chain. Early attempts to cut high branches were complicated by partially cut branches flopping over and releasing the saw, or having cut branches getting caught in other branches on the way down. To alleviate this problem, a tension line was set directly on the cone bearing branch (fig. 4). The sawing rope is gently pulled back and forth to cut the branch, and the tension line holds the branch straight. The tension line may also be used to retrieve branches over cliffs or ravines. The saw needs to be sharpened by hand daily to ensure quality cuts.



Figure 2—Tall red spruce (*Picea rubens*) on Hunter Mountain, New York. Cones are clustered near the top of the tree (see inset). (Photo by John Butnor)



Figure 3—Long-reach pole pruner constructed by attaching a pole pruner cutting head to a ~12-m telescoping tree height pole. (Photo by Tom Christensen)



Figure 4— Steps to retrieve cone bearing branches with rope-operated saw chains: 1) set throwline over the target branch with a shot-bag and sling shot; 2) replace throwline with strong 5-mm cord tied to saw chain; 3) if possible, set a tension line on the cone bearing branch; 4) gently pull saw back and forth, making cuts in both directions with a sharp chain; and 5) use tension line to support branch as it is cut and retrieve severed branch. (Photos by Chris Maier, John Butnor, and G. Scott Bryan)



Figure 5— Specialty saw chain (PockeTech HighLimb2-48 inch) that has bladed chisels on both sides of the chain and in both directions. (Photo by John Butnor)

Shaking Cone Bearing Branches

When a single branch cannot be tethered, the throwline may be sent over the top of the tree and replaced by a stronger 5-mm cord to facilitate shaking or whipping branches with rope to dislodge cones. Red spruce cones are firmly attached to the branches, but it is possible to shake or strip them loose with some effort. To catch the cones and insure they came from the subject tree, lightweight spun polyester row-cover fabric was placed on the ground (~3 x 20 m). Shaking cones from trees is a demanding, high energy endeavor. Many dislodged cones are ejected into the forest and not onto the fabric, but this method still leads to some cones being recovered. In the 2016 field season, the number of cones recovered via shaking ranged from 1 to 120 per tree. Whipping branches with the rope can help dislodge firmly attached cones. To magnify the force, tie the rope to a 1–2 m long “whip stick”—with some practice, the generated force is considerable.

CONCLUSIONS

Depending on accessibility of cones, single source seed were collected by: 1) hand if close to the ground,

2) ground collection if there is no possibility of cones originating from other trees, 3) pole pruner, 4) setting lines and sawing branches, or 5) setting lines and shaking branches. In the 2016 field season in North Carolina, Tennessee, and Virginia, we collected cones from 115 trees that yielded approximately 200,000 half-sib seeds from rugged mountainous terrain. These strategies were developed by trial and error in the field, and we hope this publication will help others with similar collection needs. We also tested radio-controlled chain saws, rope launchers powered by blank ammunition, and pruner heads on ropes (without poles); we do not recommend them in any way.

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FERTILIZATION AND IRRIGATION EFFECTS ON SOIL CO₂ CONCENTRATION AND EFFLUX IN A 16-YEAR-OLD LOBLOLLY PINE STAND

Peter H. Anderson and Christopher A. Maier

Abstract—Soil CO₂ efflux (S_f) rates are the second largest carbon flux in forest ecosystems. Fertilization and, to a lesser degree, irrigation are increasingly used to raise productivity in managed pine stands. To understand how management effects forest floor S_f , we compared seasonal variations in concurrently measured S_f and soil CO₂ concentration profile (pCO₂) in a 16-year-old loblolly pine plantation. The study design was a 2 x 2 factorial treatment combination of fertilization (no addition vs. complete nutrition) and irrigation (no addition vs. well-watered) replicated four times. S_f was measured with a commercially available chamber-based system, and pCO₂ was measured with an infrared gas analyzer in wells installed at five depths (10–100 cm). S_f was significantly increased by irrigation compared to control and fertilized treatments, and pCO₂ increased with soil depth. Treatments had no effect on pCO₂ measured at shallow depths (<15 cm); however, pCO₂ was significantly greater in fertilized treatments at depths >30 cm. Effluxes calculated from pCO₂ gradients at 10 and 50 cm were well correlated with measured S_f . Measurements of soil pCO₂ provide an inexpensive and rapid method for measuring soil carbon efflux.

INTRODUCTION

Loblolly pine (*Pinus taeda*) is used extensively for commercial timber and fiber production in the Southern United States. Fertilization is increasingly being used to raise productivity in managed pine stands. In a span of 10 years from 1988 to 1998, industrial forest fertilization increased from 16,200 ha to 344,250 ha (NCSFNC 2000). Increased forest productivity through intensive management (i.e., harvest, bedding, fertilization, and vegetation control) has been proposed as a method for slowing the rate of atmospheric CO₂ increases (IPCC 1996).

Carbon dioxide evolution from soils is a combined product of the metabolic activity of plant roots, and both free living and symbiotic heterotrophs. Forest floor soil CO₂ efflux (S_f) rates are the second largest carbon flux in the global carbon cycle and the largest terrestrial contributor (Raich and Schelsinger 1992). Changes in soil resource availability may affect S_f rates by altering root density, changing microbial populations, and modifying metabolic activity. In addition to biotic shifts in response to nutrient amendments, many abiotic factors contribute to changes in soil carbon pools and fluxes. Soil temperature and moisture account for much of the seasonal and diurnal variation in S_f (Davidson and others 1998). Determining the physical and biological factors

that regulate S_f is essential for understanding how forest management affects soil carbon storage. We measured S_f and soil CO₂ concentration (pCO₂) in a 16-year-old pine plantation under varying fertilization and irrigation regimes. The objectives were to 1) examine the effects of fertilization and irrigation on S_f and soil pCO₂, and 2) compare measured soil surface S_f with calculated soil CO₂ efflux (S_{fx}) estimated from soil pCO₂ gradients.

MATERIALS AND METHODS

Site Description

The study site is located 16 km north of Laurinburg in Scotland County, NC. The area is known as the Carolina Sandhills due to its deep, sandy soil (90–95 percent sand), and its relatively high elevation (100–200 m) when compared to the adjoining Coastal Plain region. The site was planted with loblolly pine in 1985 and thinned to 1260 trees per hectare in 1992 when fertilization treatments began. More details on the Southeast Tree Research and Education Site (SETRES) can be found in Albaugh and others (1998) and Maier and others (2004).

Experimental Design

The study was a 2 x 2 factorial combination of nutrition (no addition vs. complete nutrition) and irrigation (no addition vs. well-watered) replicated four times. The

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objective of the fertilizer treatment was to maintain 'optimum' foliar nitrogen levels (1.3 percent) while other nutrients were held in balance. The objective of the irrigation treatment was to maintain soil water content >40 percent field capacity (Albaugh and others 1998) during the growing season (April–October). Annual fertilization treatments began in 1992 and irrigation began in 1993. Tree and stand characteristics in January 2001 are summarized in table 1.

Measurements of S_f were made using the Licor 6400 portable photosynthesis system and a 6400-09 Soil CO_2 flux chamber at four locations per plot. Soil temperature was measured at 15 cm and soil volumetric water content in the top 50 cm using a time domain reflectometry. Measurements were conducted five times from June 14, 2000 to March 1, 2001. At each location, soil pCO_2 was measured at 10- and 50-cm soil depth. Soil pCO_2 was measured through a 3/8-inch teflon tube attached to a porous air stone with an infra-red gas analyzer (EGM-2, P. P. Systems, Haverhill, MA) scaled to 10,000 ppm. In addition, soil pCO_2 profile was measured at 10, 15, 30, 50, and 100 cm at one location in each plot.

Soil CO_2 efflux was estimated from pCO_2 measurements using the soil CO_2 profile method as outlined by deJong and Schappert (1972). This method assumes that S_f is dominated by diffusion and thus controlled by CO_2 partial pressures in the soil atmosphere and the diffusivity of CO_2 in the soil. If diffusion is assumed to be at steady state, then Fick's first law of diffusion can calculate soil surface CO_2 flux as:

$$S_{f,x} = -D_s (\partial C / \partial Z) \quad (1)$$

where

$S_{f,x}$ = the soil CO_2 flux ($\mu\text{mol m}^{-2} \text{s}^{-1}$) estimated from pCO_2 measured at depth x (10 and 50 cm),
 D_s = the diffusion coefficient in soil ($\text{m}^2 \text{s}^{-1}$),
 C = CO_2 concentration ($\mu\text{mol m}^{-3}$), and
 Z = soil depth (cm).

Soil CO_2 diffusion is strongly affected by soil moisture. We estimated D_s following Rolston (1986):

$$D_s = (\text{Peff}^{10/3} / E^2) * D_a \quad (2)$$

where

E = total soil porosity estimated from soil bulk density (1.5 and 1.7 g cm^{-3} at 10 and 50 cm, respectively, Abrahamson and others 1998),
 Peff = effective soil porosity (E - volumetric soil water content), and
 D_a = the rate of CO_2 diffusion in air ($\text{cm}^2 \text{s}^{-1}$).

Treatment effects on S_f , pCO_2 , $S_{f,x}$, soil temperature, and soil moisture were analyzed using analysis of variance. Repeated measures were used to analyze time series data (PROC MIXED, SAS version 9.4, SAS Institute, Cary, North Carolina) using a first order autoregressive covariance structure. Treatment differences were determined with LSMEANS statement using Tukey's adjustment.

RESULTS

There were significant treatment and treatment by time effects ($p < 0.001$) on soil moisture, soil temperature, S_f , pCO_2 and q_{50} (table 2, fig. 1). Volumetric soil moisture was significantly higher in the irrigated treatments during the growing season (Day of Year, DOY 150–250) (table 3, fig. 1A). Soil temperature was significantly lower in the fertilized treatments; however, there was a significant fertilizer x DOY interaction where treatment differences were only significant during growing season (fig. 1B). This interaction was likely due to higher leaf area moderating soil temperature in the fertilized treatments. There was a significant irrigation x fertilization interaction on S_f . Irrigation significantly increased S_f relative to the control treatment, but had no effect on S_f in the fertilized treatments (table 3). There was a significant irrigation by DOY effect on S_f . This was probably due to the lack of irrigation treatment during the winter months (November–March).

Table 1—Mean diameter at breast height, tree height, basal area, coarse root biomass, and taproot biomass in control, irrigated, fertilized, and fertilized and irrigated treatments

Variable	Control	Irrigated	Fertilized	Fertilized and irrigated
Diameter at breast height (cm)	13.4	14.4	17.7	18.8
Tree height (m)	8.7	9.7	11.5	12.7
Basal area (m^2/ha)	17.7	21.1	31.8	34.5
Course root biomass (Mg/ha)	2.4	2.8	4.0	4.3
Tap root biomass (Mg/ha)	8.7	9.9	23.3	25.7

Notes: measurements were taken in January of 2001.

Table 2—Statistics (p values) for soil CO₂ flux (S_f , $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$), soil temperature (T_{soil} , °C), volumetric soil moisture (%), soil CO₂ concentration (pCO₂), and calculated soil CO₂ efflux from pCO₂ (equation 1) at 10 ($S_{f,10}$) and 50 ($S_{f,50}$) cm soil depth

Treatment	T_{soil}	Soil moisture	S_f	pCO ₂ (10 cm)	pCO ₂ (50 cm)	$S_{f,10}$	$S_{f,50}$
Irrigation (I)	0.071	0.006	0.020	0.108	0.343	0.948	0.906
Fertilization (F)	<0.001	0.089	0.696	0.633	0.003	0.879	0.011
I x F	0.785	0.215	0.039	0.401	0.397	0.083	0.145
DOY (D) ^a	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
I x D	0.352	0.164	0.002	0.484	0.456	0.840	0.638
F x D	<0.001	0.092	0.338	0.426	0.617	0.640	0.933
I x F x D	0.410	0.442	0.467	0.964	0.531	0.427	0.441

^a Day of Year (DOY).

Due to the sandy nature of the soil, pCO₂ rarely exceeded 10 $\mu\text{mol mol}^{-1}$. There were no treatment effects on pCO₂ measured at 10 cm; however, pCO₂ at 50 cm was 25-percent higher in the in the fertilized treatments (tables 2 and 3). There were strong seasonal trends in S_f and pCO₂ where S_f and pCO₂ were greater during the growing season (fig. 1C, D, and E). There was a significant fertilization by depth interaction ($p=0.028$) on the pCO₂ profile (fig. 2). There were no treatment effects on pCO₂ at depths <30 cm; however, pCO₂ was significantly greater in the fertilized treatments at depths >30 cm. There were no significant treatment effects on calculated $S_{f,10}$; however, $S_{f,50}$ were significantly greater in the fertilized treatments (table 3). Calculated $S_{f,x}$ was validated by comparing simultaneously measured pCO₂ and S_f measured with the Licor 6400. There was a linear relationship between $S_{f,x}$ and measured S_f (fig. 3A and B). Surface S_f was overestimated by $S_{f,10}$ and $S_{f,50}$ at low efflux rates, and $S_{f,50}$ underestimated S_f at high efflux rates. Model fits were similar between S_f and $S_{f,10}$ and $S_{f,50}$.

DISCUSSION AND CONCLUSIONS

Soil S_f rates were generally lower than that measured in the same treatments 5 years earlier (age 11, Maier and Kress 2000). The differences were probably a result of lower soil moisture during this study. There was a significant irrigation by fertilization interaction on S_f , where irrigation increased S_f , but only in the non-fertilized treatment. The effect of irrigation was limited to the summer months as there were no treatment differences during the winter and early spring. These results were similar to those found by Maier and others (2004).

The fertilization treatments had an almost 2.4-fold greater coarse root biomass than non-fertilized

treatments (Albaugh and others 2004). Increased root biomass in the fertilized treatments probably drove the 25- to 30-percent greater pCO₂ and $S_{f,50}$ observed in these treatments. Nevertheless, increased root biomass and pCO₂ at the deeper depths had no effect on surface soil CO₂ efflux. This may be because surface CO₂ efflux is driven primarily by CO₂ production at shallow soil depths. Drewett and others (2005) estimated S_f from soil pCO₂ profiles and concluded that 85 percent of the soil surface S_f is generated from CO₂ production in the top 30 cm of soil. Furthermore, root respiration may contribute only a small fraction of S_f in young loblolly pine stands. McElligot and others (2016) determined that heterotrophic respiration accounted for 59–82 percent of S_f in loblolly pine plantations that ranged in age from 3 to 18 years.

Soil pCO₂ increased with soil depth, but rarely exceeded 10 $\mu\text{mol mol}^{-1}$. Soil pCO₂ profiles were similar to those observed in other temperate forest ecosystems (Drewett and others 2005, Fang and Moncrieff 1998). Calculated $S_{f,x}$ correlated well with measured S_f and pCO₂ at both depths produced similar estimates of S_f . The soil CO₂ profile method assumes steady state conditions and can produce large errors if the CO₂ gradients are rapidly changing, such as after rainfall events (Liang and others 2004). In addition, calculations of S_f from pCO₂ are very sensitive to soil CO₂ diffusivity (equation 2) (Jassal and others 2005). Tang and others (2003) found that S_f estimated from pCO₂ profiles varied from a 9-percent underestimate to an 18-percent overestimate, depending on the model used to estimate D_s . Measurements of soil pCO₂ gradients provide an inexpensive, non-invasive, and easy method for measuring soil carbon efflux. The accuracy of the method will depend on correctly estimating soil CO₂ diffusivity under changing environmental conditions.

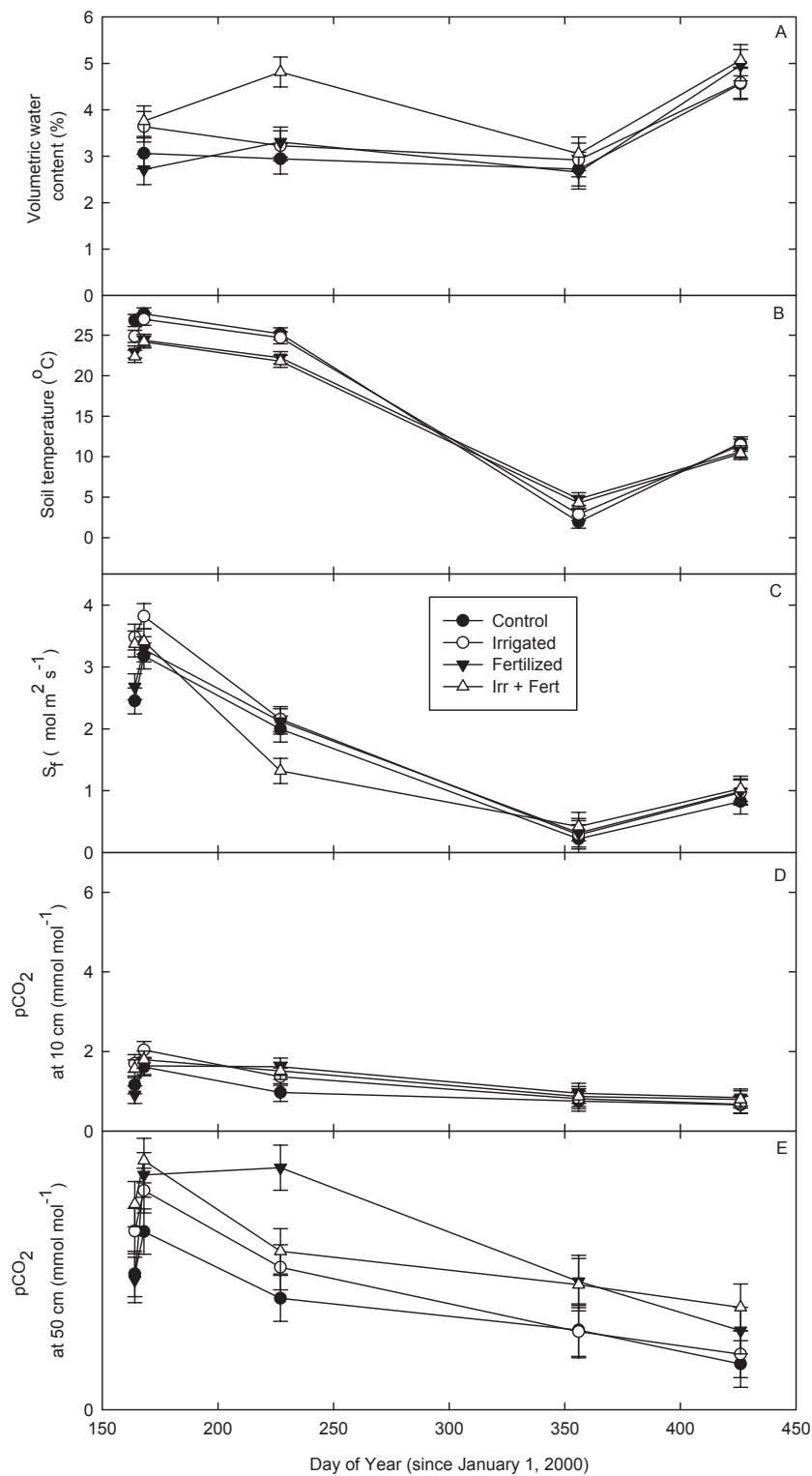


Figure 1—Seasonal variation of volumetric soil moisture (A), soil temperature (B), soil CO_2 efflux (S_f) (C), and soil CO_2 concentration (pCO_2) measured at 10 cm (D) and 50 cm (E) soil depth in control, irrigated, fertilized, and fertilized and irrigated plots. Data are LSMEANS and standard error.

Table 3—Soil CO₂ flux (S_f , $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$), soil temperature (T_{soil} , °C), volumetric soil moisture (%), soil CO₂ concentration ($p\text{CO}_2$, $\mu\text{mol mol}^{-1}$) and calculated soil CO₂ efflux from $p\text{CO}_2$ (equation 1) at 10 ($S_{f,10}$) and 50 ($S_{f,50}$) cm soil depth in control, irrigated, fertilized, and fertilized and irrigated treatments

Variable	Control	Irrigated	Fertilized	Fertilized and irrigated	SE
Soil Moisture	3.32 ^a	3.59 ^{ab}	3.41 ^a	4.17 ^b	0.24
T_{soil}	18.7 ^a	18.2 ^a	17.0 ^b	16.6 ^b	0.7
S_f	1.73 ^a	2.14 ^b	1.87 ^{ab}	1.91 ^{ab}	0.13
$p\text{CO}_2$ (10 cm)	1.03 ^a	1.32 ^a	1.19 ^a	1.31 ^a	1.08
$p\text{CO}_2$ (50 cm)	2.77 ^a	3.39 ^{ab}	4.09 ^b	4.22 ^b	3.84
$S_{f,10}$	1.63 a	2.14 a	2.16 a	1.69 a	0.30
$S_{f,50}$	1.35 a	1.68 ab	2.33 b	1.95 ab	0.26

Note: data are LSMEANS and standard error. Means within a row with a different letter are significantly different ($P < 0.05$).

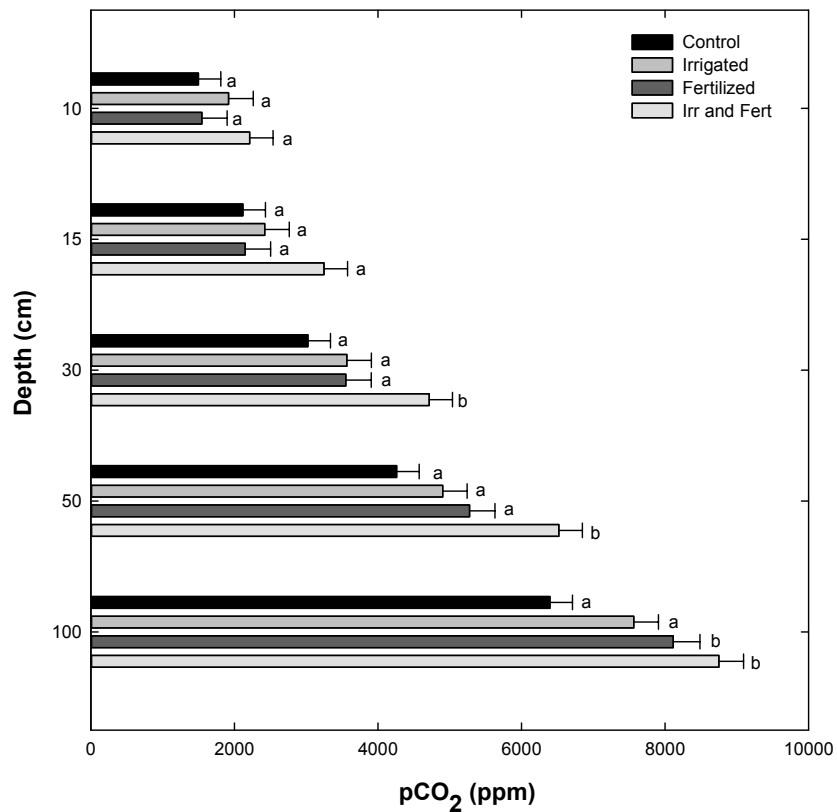


Figure 2—Mean soil CO₂ concentration ($p\text{CO}_2$) of control, irrigated, fertilized, and irrigated and fertilized treatments at 10, 15, 30, 50, and 100 cm soil depth. Different letters represent significant treatment differences by depth at the $p < 0.05$ level.

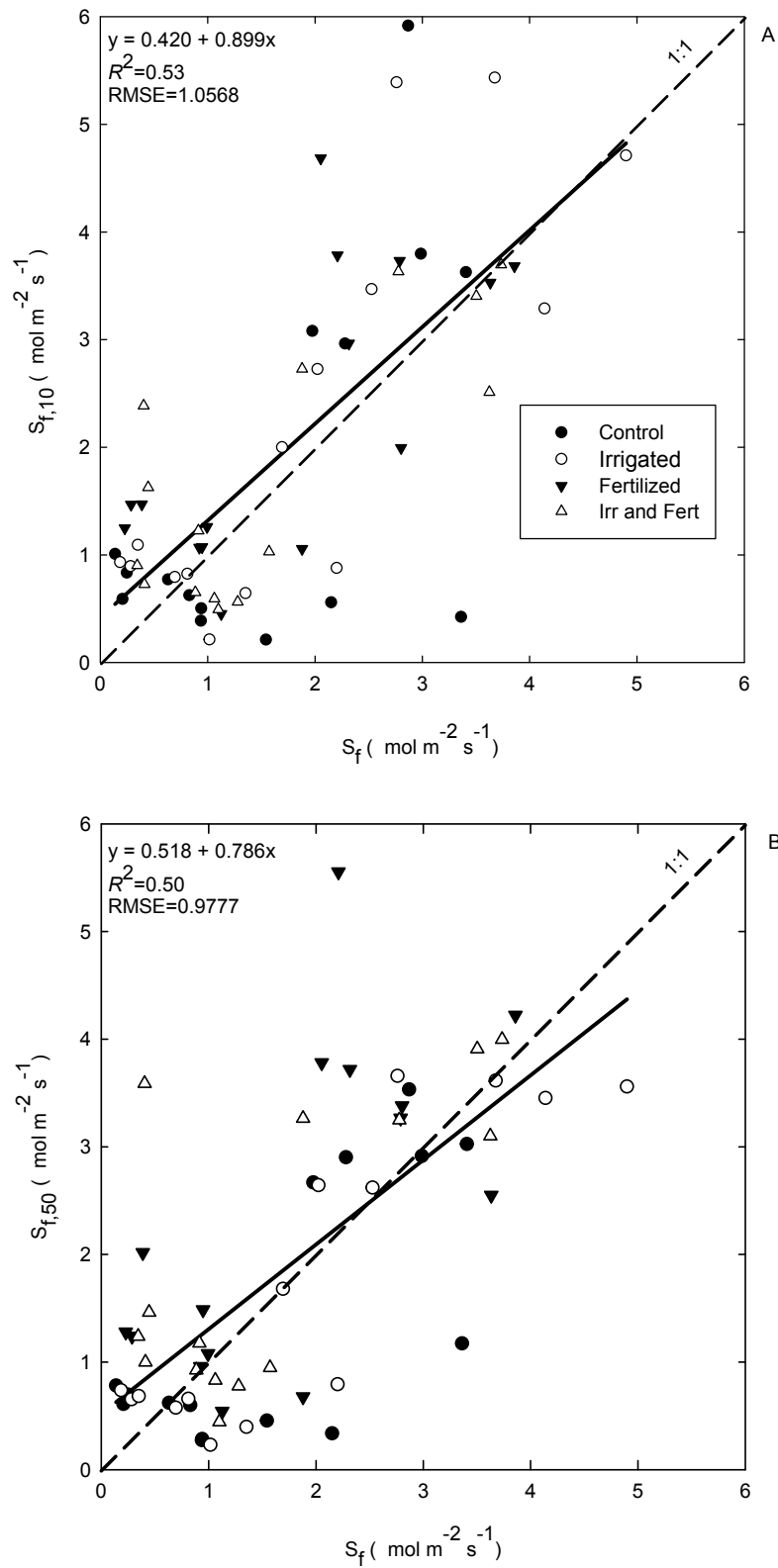


Figure 3—Calculated soil CO₂ efflux ($S_{f,x}$) from soil CO₂ concentration (pCO_2), at 10 cm (A) and 50 cm (B) versus measured surface soil CO₂ efflux (S_f). Each point represents the mean of four observations within a treatment plot.

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NINE-YEAR RESULTS FROM A *PAULOWNIA* FIELD TRIAL OF THREE SPECIES IN THE SOUTHERN APPALACHIANS

W. Henry McNab, Erik C. Berg, and Anne E. Suratt

Abstract—Three species of *Paulownia* (*P. elongata*, *P. fortunei*, and *P. tomentosa*) were evaluated for attained survival and diameter breast height (d.b.h.) after 9 years in the Southern Appalachian Mountains of western North Carolina. Because species of *Paulownia* vary in their cold-hardiness and moisture requirements, the primary purpose of this study was to compare their survival and diameter growth under the temperature and precipitation conditions of a region where they had not been previously evaluated. Particularly important was comparison of the documented invasive species, *P. tomentosa*, with the other two species that have not been reported as invasive. Mean survival differed little among species and averaged about 28 percent overall. Mean d.b.h. of *P. fortunei* was 4.5 inches, which was smaller than that for either *P. elongata* (6.7 inches) or *P. tomentosa* (6.8 inches). Preliminary results from this field trial suggest little difference in performance among the species.

INTRODUCTION

The genus *Paulownia* includes seven species native to the temperate region of eastern Asia, primarily China. One species, *P. tomentosa*, has been naturalized throughout the Eastern United States following its introduction in the 1840s as an ornamental (Snow 2015). Certain silvical characteristics of *P. tomentosa*, including prolific production of wind-disseminated seeds, rapid juvenile height growth, abundant basal sprouts following top kill, and the ability to colonize xeric sites, have led to its classification as an invasive species in the United States (Hemmerly 1989, Kuppinger 2008, Miller and others 2010, Snow 2015) and Austria (Franz 2007). However, these same characteristics, when combined with its desirable wood properties, have resulted in increasing attention to *P. tomentosa* as an important species for short rotation, high-quality saw log production (Clatterbuck and Hodges 2004, Henning 1989, Kays and others 1997), coppice biomass production (Beckjord and McIntosh 1983, Rad 2015), agroforestry (Mueller and others 2001), antibacterial drugs (Smejkal 2008), phytoremediation (Doumet and others 2008), flavanones (Asai and others 2008), pulp (Olsen 1985), and bioenergy (Yadav and others 2013).

Other species of *Paulownia* have not been reported as invasive, and some have potential productivity exceeding *P. tomentosa* (Kuppinger 2008). Although much is known about the temperature- and moisture-stress tolerance and the rapid growth rate of the deep-rooted *P. tomentosa*, relatively little has been reported for

other species, particularly in response to extreme levels of temperature and precipitation. Beyond considerable study in their native range in China, two species of *Paulownia*, *P. elongata* and *P. fortunei*, have received trial evaluations of performance in Spain (Zuazo and others 2013), Iran (Rad 2015), Brazil (Zeni and Simberloff 2013), South Africa (Donald 1990), Turkey (Ayan and others 2006) and Australia (Sun and Dickerson 1997). In many of those studies, performance of *P. elongata* and *P. fortunei* were superior to *P. tomentosa*. In several trials in the Coastal Plain and Piedmont of the Southern United States, short-term evaluations of *Paulownia* species other than *P. tomentosa* have reported similar or better survival and growth (Bergman 2003, Dong and Buijtenen 1994). The primary purpose of our study was long-term evaluation of *P. elongata* and *P. fortunei* survival and growth in comparison to *P. tomentosa* in the cool climate of the southern Appalachian Mountains. The scope of our study was limited to a single site and did not encompass the range of sites and soils present in the southern Appalachian Mountains. Also, our study did not include cultural treatments, such as coppicing, fertilization, irrigation, weed control, and stem pruning, which are typically part of intensive management in commercial plantings. Presented in this report are final results of attained survival and diameter breast height (d.b.h.) at 9 years of age.

STUDY SITE AND METHODS

The study was established in the Bent Creek Experimental Forest, within the Pisgah National

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Forest, which is located in the southern Appalachian Mountains physiographic province (35.5 °N, 82.6 °W). This area has a warm, humid continental climate. Annual temperature averages 55.8 °F and varies from 41.2 °F in winter to 70.5 °F in summer. Annual precipitation averages 47.8 inches and is evenly distributed among seasons, with occasional soil moisture deficits of several weeks duration during late summer. Growing season length averages 157 days, extending from late April to middle October.

The study site was a low elevation, southwest-facing slope that had been cleared of upland hardwood vegetation (primarily *Quercus* spp. and *Carya* spp.) two years earlier to allow disposal of sediments removed from nearby Lake Powhatan. The 13-acre lake receives sediments from the third-order stream, Bent Creek, which drains a forested watershed of about 6 square miles. Sediments are derived primarily from soils formed from gneiss and schist formations that have eroded from roads and occasionally from debris avalanches (Neary and others 1986). Surface soils in the Lake Powhatan watershed are classified as fine to coarse textured loams and sandy loams with < 33 percent clay content. The saturated sediments were allowed to drain for a year and were then leveled to form a uniform layer of soil material approximately 2 feet thick with a slope gradient of about 15 percent. The soil material was characterized from samples systematically collected to a depth of 6 inches throughout the study area for determination of pH, carbon, organic matter, nitrogen, and texture. The Bouyoucos hydrometer method was used for particle size analysis of oven-dried, 2-mm sieved samples. Air

temperature and precipitation were recorded daily at a standard weather station located 2 miles northeast of the study area, at about the same elevation.

Study treatments consisted of measuring the survival and growth response of three species of *Paulownia*—*P. elongata*, *P. fortunei*, and *P. tomentosa*—to the prevailing edaphic and climatic conditions of the planting site. In March 1995, we hand planted 150, 1-year-old containerized seedlings of each species from a commercial nursery on a spacing of 11 feet between rows and 7 feet between seedlings within each row. Vegetation on the planting site consisted primarily of a sparse cover of native grasses and herbs originating from wind-blown seeds from the surrounding (< 50 feet distance) forested area. A corrugated plastic collar (4 inches diameter x 6 inches tall) was placed around each seedling at planting for water conservation, weed control, protection from rodent girdling, and support of the relatively weak succulent stem and root system from breakage by strong winds during thunderstorms. Seedlings received no fertilization, cultivation, or weed control during the study. Attained survival and d.b.h. were summarized by species after 9 years of growth, when the study ended in December 2003.

RESULTS AND DISCUSSION

Climate and Soil Environments

Winter temperature and summer precipitation during the 9 years of the study were near the 30-year normal (table 1); however, several extreme events occurred that provided environmental stresses desirable for our species trials. The 9-year minimum temperature of -5 °F

Table 1—Mean, minimum, and maximum climatic and edaphic properties of the study site in the Bent Creek Experimental Forest, near Asheville, NC

Variable	N ^d	Mean (SD)	Minimum	Maximum
Climatic variables (1995-2003)				
Winter temperature (°F) ^a	9	39.2 (1.83)	35.70	42.65
Summer precipitation (inches) ^b	9	14.3 (5.42)	5.2	23.70
Edaphic properties ^c				
Acidity (pH)	9	4.27 (0.029)	4.25	4.32
Carbon (%)	9	0.026 (0.004)	0.021	0.032
Organic matter (%)	9	0.005 (0.0006)	0.004	0.006
Nitrogen (ppm)	9	369.44 (81.962)	256.00	478.00

^a Winter season includes January-March. The 30-year normal winter season temperature during the 9 years of the study was 40.7 °F. The minimum daily temperature recorded during the study was -5 °F;

^b Summer season includes July-September. The 30-year normal summer season precipitation during the 9 years of the study was 12.36 inches. The minimum monthly precipitation recorded during the study was 0.25 inch.

^c To depth of 6 inches.

^d N; 9 years for climatic variables or 9 samples for edaphic variables.

SD = standard deviation.

occurred in February 1996, after the second year of growth for the seedlings. Variation of cold-temperature hardiness of species in their native China is not well known; however, because species have been widely planted, Donald (1990) reported that *P. tomentosa* has greatest cold tolerance (-4°F) and *P. elongata* is least tolerant (14°F). Minimum temperatures occurring during our study were near or exceeded values reported as critical for each of the three species. Except for late frosts during the early growing season (Mitchem and others 2002), results of *Paulownia* trials seldom specify temperature as a factor affecting survival and growth, which suggests other characteristics of sites are of greater importance, such as soil fertility and moisture availability. Summer precipitation during the study averaged 14.3 inches, which was 2 inches above normal. Drought, however, occurred during the third growing season, 1998, resulting in a summer season precipitation of 5.2 inches, including one month in which only 0.25 inch was recorded. Mature *P. tomentosa* and other species are relatively drought tolerant as a result of their deep root systems (Donald 1990, Reynolds and others 2009), which are well suited when combined with shallow-rooted, inter-planted agricultural crops in agroforestry systems (Mueller and others 2001). Compared with other species of *Paulownia*, *P. elongata* has been reported as most drought tolerant (Llino-Sotelo and others 2010).

The soil material of our planting site is classified as a loam with mean coarse material $>2\text{ mm}$ of 7.0 (SD = 3.2) percent. The loam soil consisted of 49 (3.5) percent sand, 21 (1.2) percent clay and 30 (2.6) percent silt. Mean soil pH was 4.3, which is characterized as extremely acidic; N content was low, and organic matter was nearly zero (table 1). Soil quality of our planting site is classified as low using criteria of organic matter and nitrogen (Schoenholtz and others 2000) and in comparison with other *Paulownia* studies in the Virginia Piedmont (Mitchem and others 2002). Fertility requirements are minimal for survival and growth of *Paulownia* as indicated by successful establishment of several species for restoration of mine spoils (Jiang and others 2012, Tang and others 1994). Although low in fertility, the dredged sediments at our planting site were also low in clay content and well drained, which

is suitable for *Paulownia* survivorship and growth (Kay and others 1998). Darmody and others (2004) found that dredged lake sediments can be highly productive for agricultural crops when properly fertilized.

Survival

Mean survival at age 9 averaged 28.7 percent overall and was similar among species (table 2). The lower level of survival for *P. tomentosa* in our study was similar to that for a 6-year species trial in Texas by Dong and Builigi (1994), who cited late spring frosts as the likely cause. Unlike our findings of low survival for *P. fortunei*, however, Dong and Builigi (1994) reported high survival (60 percent) after 6 years. Bergmann (2003) reported relatively high mean survival (>70 percent) after 5 years for *P. elongata* and *P. fortunei* in the Piedmont of North Carolina and also found superior survival of rooted stem cuttings compared to seedlings (Bergmann 1998). In the Virginia Piedmont, Mitchem and others (1999) suggested soil moisture stress was likely the primary factor for low survival of *Paulownia* seedlings. Low moisture was probably not the reason for the high mortality that occurred during the first growing season after planting in our study because precipitation was above the 9-year average, and weed competition was not severe.

The plastic collars installed at planting had been firmly imbedded in soil; this action eliminated the possibility of “chimney” drying and mortality of seedlings which can occur when gaps are left at collar bases. Further, we did not observe rodent damage to seedlings. However, some species of *Paulownia* have been reported as susceptible to fatal bacterial and fungal diseases, spores of which could have been present in the lake sediments on the planting site (Mehrotra 1997). Also, *Paulownia* is sensitive to late spring frosts (Personal communication, W.C. Clatterbuck. 2017. Professor, Ellington Plant Sciences Building, University of Tennessee, Knoxville, TN 37996) that can cause top dieback of young seedlings and result in increased levels of mortality (Dong and Builigi 1994). During the study period, a low temperature of 29°F was recorded in early May 1996, which could have been a factor that reduced survival when the seedlings were beginning their second growing season. We have no satisfactory explanation for the low survival of all species in our study.

Table 2—Mean survival (SE) and d.b.h. (SE) attained by three species of *Paulownia* 9 years after planting in the Bent Creek Experimental Forest, near Asheville, NC

Species	Survival (%)	D.b.h. (inches)
<i>P. elongata</i>	34.3 (4.03)	6.7 (0.42)
<i>P. fortunei</i>	30.2 (4.06)	4.5 (0.46)
<i>P. tomentosa</i>	21.9 (3.44)	6.8 (0.51)
All	28.7 (2.22)	6.0 (0.46)

Diameter

Mean d.b.h. attained at age 9 averaged 6.0 inches overall and was similar among species (table 2). *P. fortunei* averaged 4.5 inches compared to *P. elongata* (6.7 inches) or *P. tomentosa* (6.8 inches). Variation of mean tree d.b.h. was also similar among species (0.46 inches). Our study was not designed to associate precipitation or soil moisture with tree size. However, because diameter growth of some species varies directly with fertility (Rad and Mirkala 2015), the diameter attained by trees in our study was likely affected by the relatively infertile lake sediments.

Overall mean d.b.h. of *Paulownia* in our study averaged 6.0 inches at 9 years, or a mean annual increment (MAI) of 0.67 inch. For other studies in the South, Mitchem and others (2002) and Bergmann (1998) reported MAI of approximately 1.3 inches for 5-year-old *P. elongata* in the Piedmont of Virginia. In the Piedmont of North Carolina, Bergmann (2003) found MAI of 1.7 inches for *P. elongata* at 4 years. Beckjord and McIntosh (1993) and Bergman (2003) reported that diameter of all *Paulownia* species responded to fertilization, particularly increased nitrogen. In our study, the primary factor affecting d.b.h. was probably low fertility of the lake sediments of our site, which was about half that of a typical upland hardwood site (Mitchem and others 2002). Another explanation for the smaller tree sizes in our study could be a shorter growing season resulting from the cooler climate of the southern Appalachians compared to the Piedmont of North Carolina (Bergmann 2003) or east Texas (Dong and Buijten 1994).

CONCLUSIONS

Results of our study revealed low survival at age 9 among all species, but we were unable to determine if it was associated with temperature, soil moisture, or fertility. Our study was not designed to determine critical levels of minimum temperature or moisture that are important for survival of the three species of *Paulownia*. However, the levels of temperature and precipitation encountered are near the extremes likely for this part of the southern Appalachians and therefore provide useful information on species selection. Most surprising was the satisfactory performance of *P. elongata*, which has been reported as less tolerant of low temperatures compared to *P. tomentosa*. The relatively good performance of *P. elongata* was also noteworthy because it has not been reported as invasive. Although *P. tomentosa* has many desirable qualities, this species is not recommended for planting because of its reported invasive characteristics on certain sites. The lake sediment soils of our planting site were not typical of the southern Appalachians; however, the purpose of our study was an initial assessment of species performance and not to determine potential growth. Overall, our

preliminary assessment indicated similar attained survival and diameter among the three species at 9 years of age.

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CATION RETENTION SOIL ADDITIVE INFLUENCES ON LOBLOLLY PINE PLANTATION RESPONSE TO FERTILIZATION IN CENTRAL LOUISIANA

Michael A. Blazier, Ed Poole, and Mickey Rachal

Extended abstract—Loblolly pine (*Pinus taeda* L.) plantations in the Southeastern United States are fertilized to ameliorate N and P deficiencies that are relatively common in soils of the region (Allen 1987, Fox and others 2007, Jokela 2004). A relatively unexplored method for improving efficacy of forest plantation fertilization is the application of amendments that improve cation retention. A recently developed example of such products is Capture® (BioDirt, Inc., Cumming, GA). Capture® is a polymer originally developed as a crystal growth inhibitor for pipelines, but its propensity for binding cations led to its development as a soil amendment to improve cation availability for plant uptake. The biodegradable and non-toxic polymer has been tested by its manufacturer in turfgrass and horticultural plant trials, but it has never been tested in forests to the authors' knowledge. The Capture® polymer could improve the efficacy of fertilization in loblolly pine plantations by promoting retention of applied cations within the upper soil profile, which could foster a more pronounced growth increase of loblolly pine. The polymer can also increase concentrations of other cations in the upper soil profile, which could provide additional loblolly pine growth increases. Due to the complexes it forms with Ca, Capture® also has potential to improve P fertilization efficacy by reducing precipitation of applied soluble P with Ca, thereby improving bioavailable P availability. The objective of this study was to test the effects of the Capture® polymer on tree growth and soil nutrients when applied to a loblolly pine plantation in central Louisiana.

A study site was established in a newly planted loblolly pine plantation near Dry Prong, LA (31.60865261 N, 92.62854479 W) in May 2014. Soil of the site was a Sacul series well-drained sandy loam (fine, mixed, active thermic Aquic Hapludult; the Sacul series has broad extent in the western coastal plain of Arkansas, Louisiana, and Texas as well as in the southern coastal plain of Mississippi and Alabama (USDA SCS 1986, USDA NRCS 2015). Three treatments were conducted for the study: an untreated control (C), fertilization (F), and fertilization with the Capture® amendment (FA). Treatments were applied to 0.08-ha plots; treatments were replicated two times each. A randomized complete block design, with slope as a blocking factor, was used for the experimental design. For the F and FA treatments, fertilizer was applied by ATV-mounted spreaders in July 2014 as diammonium phosphate at 280 kg ha⁻¹. For the FA treatment, Capture® was applied two weeks after fertilization by backpack sprayer at 4.7 L ha⁻¹.

In October 2014, total height and groundline diameter were measured on 20 trees per plot; measured trees were within a centrally located measurement zone in each plot. In June 2014 and September 2015, soil samples were collected to a 30-cm depth using punch augers. Soil samples were analyzed for pH (1:1 soil:water), P, K, Ca, Mg, Na, S, Cu, and Zn (Mehlich 3 extraction) concentrations (McLean 1982, Mehlich 1984).

Height and groundline diameter were both significantly ($P < 0.10$) affected by treatments (fig. 1), with height and groundline diameter of the F treatment being lower than that of the C and FA treatments. These results were likely affected by Nantucket pine tip moth (*Rhyacionia frustrana* Comstock) infestation of the site, with greater infestation observed in the F treatment. Among all soil nutrients, P, K, and Cu were affected ($P < 0.10$) by treatments. Phosphorus concentrations of the F (11.2 mg kg⁻¹) and FA (17.4 mg kg⁻¹) were greater than that of the C (7.3 mg kg⁻¹) treatment. For K, the FA treatment had higher concentrations (47.8 mg kg⁻¹) than the F treatment (34.8 mg kg⁻¹). Concentrations of Cu were higher for the FA treatment (0.59 mg kg⁻¹) than the C treatment (0.50 mg kg⁻¹). Analyses of variance (ANOVA) of soil nutrient concentrations were complicated for some nutrients by relatively high standard errors associated with highly different means for upper and lower slope measurements within treatments. The low replication of this preliminary study reduced the ability to account for such slope differences with ANOVA. Soil concentration means by treatment and slope position are informative for P (table 1). On the upper slope, mean soil test P concentration of the FA treatment was 56-percent lower than that of the C treatment, but on the lower slope, mean soil test P concentration of the FA treatment was 5.3 and

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2.3 times greater than that of the C and F treatments, respectively. The 3 months following F and FA treatments had precipitation 44-percent greater than the long-term average (SCIPP 2017), which may have affected Capture[®] and mobility along the upper slope and reduced its efficacy. Results of this small-scale preliminary study provide modest evidence of the potential for Capture[®] polymer to improve availability of P and cations in soil, but further studies longer in duration, more highly replicated, and over a greater array of site conditions are needed to better understand the utility of the polymer in loblolly pine plantation fertilization.

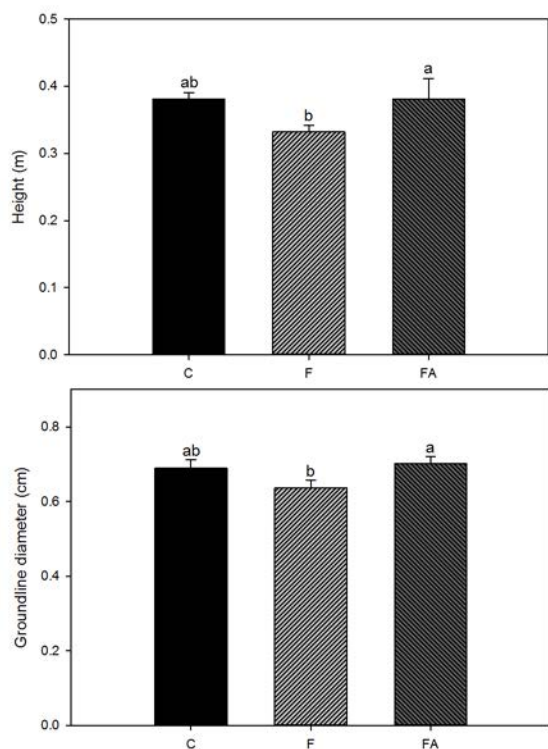


Figure 1—Total height and groundline diameter of a loblolly pine plantation in central Louisiana 3 months after treatment in response to untreated control (C), fertilization with diammonium phosphate (F), and fertilization with diammonium phosphate with a cation-retention soil amendment (FA).

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Table 1—Mean soil nutrient concentrations to 30 cm in a loblolly pine plantation in central Louisiana 15 months after treatment in response to untreated control (C), fertilization with diammonium phosphate (F), and fertilization with diammonium phosphate with a cation-retention soil amendment (FA)

Slope position	Treatment	P	Nutrient concentration (mg kg ⁻¹)						
			K	Ca	Mg	Na	S	Cu	Zn
Upper	C	9.0	30.1	265.3	63.2	9.5	7.0	0.5	2.2
Upper	F	9.2	33.7	387.4	51.8	20.4	6.7	0.6	0.8
Upper	FA	3.9	26.2	242.4	53.0	10.3	6.3	0.5	1.0
Lower	C	5.8	68.2	314.8	113.5	10.6	8.9	0.7	1.7
Lower	F	13.4	45.7	376.6	64.8	14.2	7.0	0.6	1.9
Lower	FA	30.8	45.7	393.1	78.0	13.0	7.1	0.6	1.5

MATERIAL COSTS OF FORESTRY BEST MANAGEMENT PRACTICES ACROSS NORTH CAROLINA

A.J. Lang, W.A. Coats, Tom A. Gerow, and W.A. Swartley

Abstract—Forestry best management practices (BMPs) are effective and practical methods used to reduce nonpoint source pollution to a level compatible with water quality goals. Pollution caused by increases in nutrients, chemicals, organics, and temperature are addressed by forestry BMPs, but sediment is the primary pollutant associated with forest operations in the Southeastern United States (Aust and others 1996). Therefore, most BMPs are designed to reduce or prevent accelerated soil erosion. North Carolina uses a quasi-regulatory approach to manage nonpoint source pollution from silvicultural operations. Forestry-related, land-disturbing activities must comply with the standards codified in the State's *Forest Practice Guidelines Related to Water Quality* (abbreviated FPGs). While the use of BMPs is non-regulatory, observations have shown that liberal implementation of BMPs can often result in compliance with the FPG standards. Subsequently, outreach and education that promotes forest management and water quality protection is a primary mission of the North Carolina Forest Service (NCFS).

NCFS identified monetary costs associated with recommended BMPs and preharvest planning aids as potential areas for program improvement. BMP costs can be difficult to assign because available resources and site-specific conditions are unique to each forest operation. However, estimating BMP costs are part of the pre-harvest planning process and can help estimate the profitability of forest management (Conrad and others 2012). Material costs are an important component to the overall BMP costs and can be obtained through personal experience, contractor bids, or spreadsheets. Therefore, the purpose of our poster was to provide practicing North Carolina foresters, loggers, and landowners with estimates of BMP material costs associated with North Carolina's recommended forestry BMPs. Secondary objectives were to highlight NCFS Forest Preharvest Planning Tool (FPPT) and to educate stakeholders about the benefits of effective forestry BMPs.

NCFS contacted approximately 100 stone quarries, farm supply stores, and other businesses specializing in erosion control across North Carolina and requested current (February 2017) material costs for several BMP materials recommended in the 2006 Forestry BMP manual. Costs were categorized by region (Mountains, Piedmont, Coastal Plain) and averaged among similar products (table 1). Generally, prices for seed, straw,

silt fence, straw logs, geotextile fabric, and wooden bridgemats were similar among the three regions. This was likely due to the widespread availability of those materials across the State. Gravel and steel bridgemats had noticeably different values among regions. Gravel price differences can be partially explained by the limited supply of the stone resources in the Coastal Plain. The differences in price for steel bridgemats was likely due to the differences in panel design and type of steel used to build the bridgemats.

Values reported in table 1 can be incorporated into spreadsheets and adjusted over time to estimate BMP costs. An example of an applicable spreadsheet is the Virginia Tech Road and Skid Trail Cost Method developed by Conrad and others (2012). The results of the BMP material cost survey and a link to the Virginia Tech Road and Skid Trail Cost Method spreadsheet are available on the NCFS website (http://ncforestservice.gov/water_quality/bmp_costs.htm). This product will continue to be updated as new information becomes available.

Estimates of BMP implementation cost require users to have basic knowledge of the local terrain that can be obtained through office and field reconnaissance. NCFS provides a free, publicly available, and easy-to-use web-based tool (FPPT) to assist in forest operational planning.

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Table 1—Average price (2017 US\$) for best management practices (BMP) materials recommended within 2006 North Carolina Forestry BMP manual by Mountain, Piedmont, and Coastal Plain regions of North Carolina

Seed/Material/Equipment	Quantity	Mountains	Piedmont	Coastal Plain
Spring Seed Mix	Creeping red clover (20 lbs ac ⁻¹) Red clover (10 lbs ac ⁻¹) Oats (1 bag ac ⁻¹)	\$76.50 ^a	\$71.60 ^a	\$67.54 ^a
Summer Seed Mix	Browntop Millet (25 lbs ac ⁻¹)	\$23.66 ^a	\$20.39 ^a	\$20.50 ^a
Early Fall Seed Mix	Creeping red clover (20 lbs ac ⁻¹) Red clover (10 lbs ac ⁻¹) Wheat (1 bag ac ⁻¹)	\$71.27 ^a	\$69.46 ^a	\$80.44 ^a
Late Fall Seed Mix	Creeping red clover (20 lbs ac ⁻¹) Annual ryegrass (10 lbs ac ⁻¹) Rye (1 bag ac ⁻¹)	\$130.34 ^a	\$128.26 ^a	\$140.33 ^a
Winter Seed Mix	Annual ryegrass (20 lbs ac ⁻¹)	\$15.60 ^a	\$13.50 ^a	\$14.71 ^a
Straw Mulch	(1 Bale)	\$5.73	\$5.41	\$5.75
Straw Matting	8-ft by 90-ft	\$44.10	\$41.92	\$48.63
Silt Fence	100-ft Pre-staked	\$27.54	\$26.19	\$26.00
ABC-spec Stone	Non-delivered ton	\$19.79	\$21.00	\$24.82
2-3-inch Size Stone	Non-delivered ton	\$23.54	\$26.49	\$33.74
3-5-inch Size Stone	Non-delivered ton	\$24.46	\$28.25	\$37.75
6-8-inch Size Stone	Non-delivered ton	\$25.14	\$29.13	\$43.52
Geotextile fabric	300-ft by 15-ft	\$329.17	\$313.75	\$313.75
30-ft Steel Bridgemat Set	3 panels	n/a	\$12,209.00	\$16,127.50
25-ft Wooden Bridgemat Set	3 panels	\$2,685.00	\$4,000.00	\$3,654.00
Plastic Road Surface Matting	1000 ft ²	\$3,000.00	\$3,000.00	\$3,000.00
Motor grader 138 hp	Hourly rate	\$200.00	\$200.00	\$200.00
Bulldozer 96 hp	Hourly rate	\$105.00	\$105.00	\$105.00
Excavator 80-125 hp	Hourly rate	\$97.50	\$97.50	\$97.50

^a Seed prices based on 50-pound bag and adjusted to fit recommended application rate

This online tool is a new way for forest operators and forest landowners to create customizable maps and receive site-specific reports using GIS data. Users can define management areas, overlay harvest features (skid trails, roads, log decks, etc.) and generate detailed site and soil reports. Site summaries provide information on applicable regulations, tract size, and links to online BMP technical assistance. The soil summary report gives users Natural Resources Conservation Service soils map and soil suitability ratings. The FPPT is an information gathering and planning tool, but should still be supplemented with a pre-harvest site inspection prior to forest operations. Figure 1 depicts an example of a FPPT map product.

In addition to the FPPT, the NCFS provides several water-related services/programs:

- Loaning of steel bridgemats for loggers to cross waterways
- Water quality-themed training through the NC ProLogger Program, and other workshops
- Stream identification and determination
- FPG compliance inspections
- BMP implementation monitoring site assessments

Harvest Tract Example



5/15/2017 4:10:55 PM N.C. Forest Preharvest Planning Tool

[Website](#)



Figure 1—An example of a customized map product produced by the Forest Preharvest Planning Tool.

North Carolina forest landowners and other stakeholders are encouraged to contact NCFS with forest-related questions and assistance with management plans.

Overall, the benefits of implementing BMPs can include improved or maintained water quality, sustained site productivity, compliance with rules and regulations, enhanced contribution to the conservation of freshwater aquatic wildlife, public and landowner approval, professional improvement, fewer days lost to weather-related down time, and decreased wear and tear on equipment. Material cost information and the Virginia Tech Forest Road Cost Method spreadsheet provide stakeholders with a starting point for estimating BMP cost.

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A COMPUTER PROGRAM TO PREDICT THE QUALITY OF LONGLEAF PINE SEED CROPS

Daniel J. Leduc and Shi-Jean S. Sung

Abstract—Longleaf pine (*Pinus palustris* Mill.) has good seed years at irregular intervals. Although previous researchers found significant relationships between weather variables and size of the cone crop for a given year, they have stopped short of developing a predictive model. In this study, seed crops were classified as bumper, good to fair, and poor to failed. A canonical discriminant analysis based on weather data was performed to develop a classification function. We then developed a computer program that implemented the results of this canonical discriminant analysis to predict the class of cone crop for a given year. This prediction can be made as early as 18 months prior to seed maturity. This model should greatly help in planning site preparation or seed harvesting activities.

INTRODUCTION

Longleaf pine (*Pinus palustris* Mill.) has long been known to produce irregular cone crops. As far back as 1922, a Forest Service, U.S. Department of Agriculture report stated that longleaf pine bears seed in good quantities only once every 5 to 7 years (Mattoon 1922). This irregularity complicates management efforts in natural regeneration and even potential seed harvests for artificial regeneration.

Given this variation, it would be useful to predict what the quality of the cone crop might be for a given year. Many authors have noticed a correlation between weather and longleaf pine cone crops (Chen and others 2016, Leduc and others 2016, Pederson 1999, Shoulders 1967). Leduc and others (2016) found many weak but sometimes significant correlations between the current year cone crop and weather variables for the preceding three years. Furthermore, they used the results of a canonical discriminant analysis to show that even the nonsignificant variables contributed to an observable difference in classification between bumper, fair-to-good, and poor-to-failed seed crops. We sought to make it easy to use the many weather variables to generate actual concrete predictions for longleaf pine cone crop quality.

METHODS

Data

Longleaf pine cones have been counted during the spring of each year since 1958 at the Escambia Experimental Forest in Alabama, and this count

was expanded to nine other locations across the South in subsequent years. This dataset ranges from Louisiana to North Carolina (Leduc and others 2016) and is maintained by the Southern Research Station at Auburn, AL. We obtained monthly values for average temperature, high temperature, low temperature, cooling-degree days, heating-degree days, precipitation, and Palmer drought severity index (PDSI) for each location and year since 1958 (NOAA 2014). Using these variables together resulted in 390 observations from which a model was developed.

Model Development

We wanted to find an empirical predictive model that would allow us to utilize many input variables to classify cone crops into bumper, good-to-fair, and poor-to-failed classes (see table 1 for class definitions). Several methodologies were tested. Among the modeling methods tested were genetic algorithms (System Dynamics International 1997) and neural networks (NeuralWare 1991). However, these methods proved unsatisfactory. We then used more traditional statistics

Table 1—Definitions of longleaf pine cone crop quality used in this study

Crop quality	Cones per tree
Bumper crop	≥100
Fair-to-good crop	25 to 99
Poor-to-failed crop	<25

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including canonical discriminant analysis (SAS 2004). The initial model was built to predict cone crop quality with data up to March of the crop (that is, seed maturity) year; subsequent models were built to use data only up to December of the previous year (that is, the pollination year), September of the pollination year, and June of the pollination year. The intention was that the best model would use the latest data available, but models with less data could be used when a longer planning horizon for natural regeneration and/or cone harvest was desired at the cost of some loss of accuracy.

RESULTS

The Model

Two canonical vectors were required to classify the seed crop into the three crop quality classes defined in table 1. We checked model quality by predicting the crops for all of the data used to create the model. While this is not ideal, it allowed us to use all of our limited data to develop the model. The percent of successful predictions is shown in table 2. Figure 1 shows the canonical scores for each of the observations in the dataset when the full 27 months of data is used in the model. The three classes are distinctly different for the most part, but there is some overlap that results in uncertainty for the predictions.

In order to use the results of the canonical discriminant analysis, the vector of canonical scores is multiplied by the vector of standardized weather data to get a canonical score of the data. Each of the weather variables is standardized by subtracting its mean value and dividing by its standard deviation. The result of this multiplication is a model canonical score. We have two vectors of canonical scores, so we calculated two model canonical scores and these can be visualized as coordinates on an x-y graph as shown in figure 2. The next step is to see how far the calculated score is from

the mean values for each of the cone crop classes. This distance is calculated as a Euclidian distance (ED) as shown in equation 1:

$$ED = \sqrt{X^2 + Y^2} \quad (1)$$

where X and Y are the respective distances in the horizontal and vertical directions from the calculated score to the mean score of a given class.

For example, figure 2 shows the crop would be considered bumper since the calculated ED is closest to the bumper mean, but one would accept this conclusion tentatively since the score is also close to the mean for the poor-to-failed class.

The Computer Program

Description—The calculations to predict a cone crop class are not difficult, but they are numerous. In the 27-month model (June of crop year), 185 variables are standardized by subtracting their means and dividing by their standard deviations, multiplied by the vectors for the canonical scores one and two, and the results summed. The calculations are not complex but are sufficiently tedious to make manual calculation unlikely. For this reason, a program was written in Visual BASIC® (2012 Microsoft Corporation) to perform the calculations. This program is called LongCones, and it is available on the Southern Research Station Web site (<https://srs.fs.usda.gov/longleaf/tools/>) as a tool of the Restoring and Managing Longleaf Pine Ecosystems unit.

Requirements—This program was written on Windows 7® and should also run on the more recent versions of Windows®. The program was written for a screen resolution of 1920x1200, but the windows can be scaled. An internet connection is needed to update the weather data.

Table 2—Successful crop quality predictions using various models

Model	Overall	Bumper	Good to fair	Poor to failed
<i>correct prediction percentage</i>				
March of seed year (27-month)	90	95	82	92
December of pollination year (24-month)	87	95	78	90
September of pollination year (21-month)	85	90	79	87
June of pollination year (18-month)	82	95	76	85

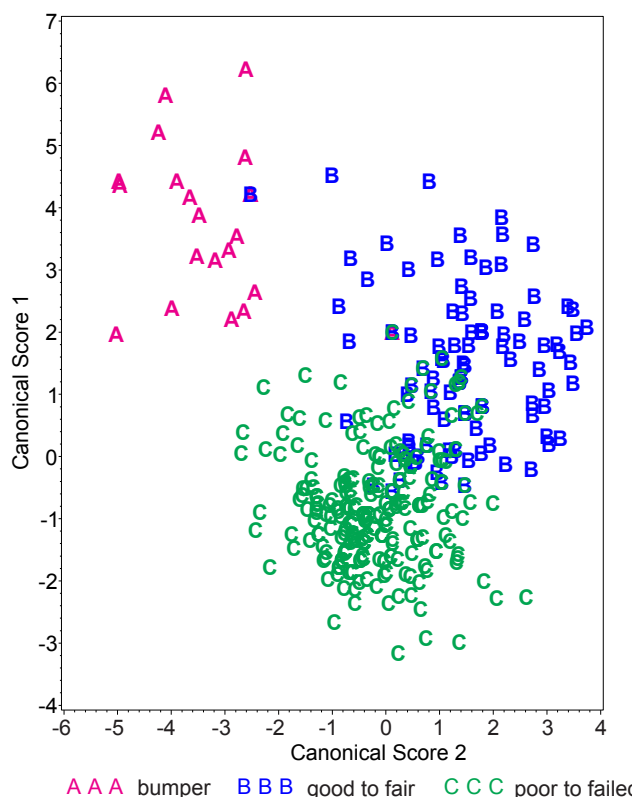


Figure 1—Results of canonical discriminant analysis showing that the calculated scores for each of the three classes of cone crops are mostly distinct from each other.

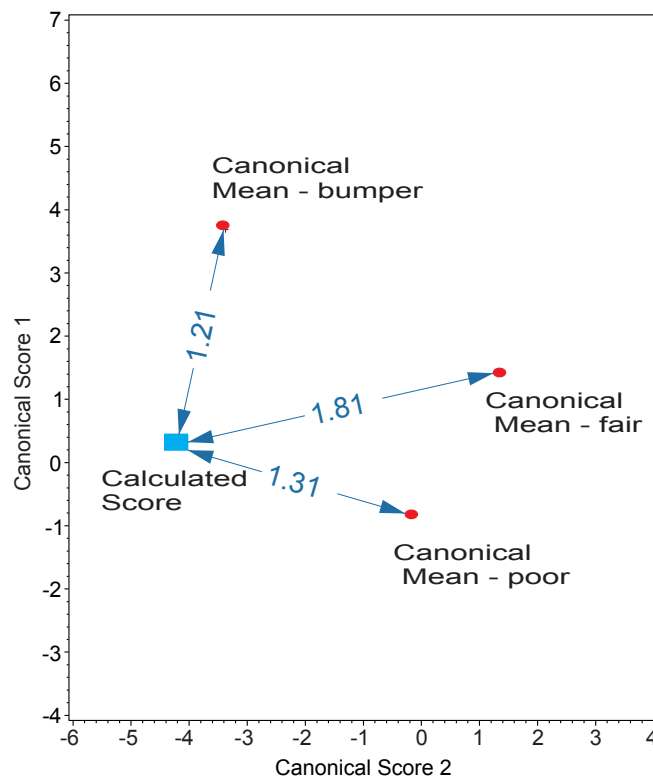


Figure 2—Graphical representation of how a new cone crop class is determined by the model. A new canonical score is calculated by multiplying canonical vectors one and two by the standardized input variables. The two numbers obtained are used as coordinates, and the distance from these coordinates to each of the class means is determined. The closest class is determined to be the new predicted class.

User's guide—Once the program has been installed using the longcones-setup.exe program downloaded from the Web site, the user can begin to use it by double-clicking on the icon. This will bring up a brief introductory splash screen followed by a screen that looks like figure 3. The largest part of this screen is occupied by a map of the areas where this model might be applicable. However, only the climate divisions highlighted in green actually contain stands used in developing this model. (The map shown in figure 3 is simply a reference.) The user must select the State, climate division, and year of interest using the buttons and text boxes below the map. When done, the user simply clicks on the “Get Data” button to continue.

This program uses data files from the NOAA website (<https://www1.ncdc.noaa.gov/pub/data/cirs/climdiv/>) copied to the user's computer at C:\ProgramData\LongCones. NOAA updates these files about once per month, and the user can update the local files by clicking on the button “Update Weather Data.”

Once the user has clicked on “Get Data,” the screen will go blank for a few seconds while the appropriate weather data is loaded. This resulting screen (fig. 4) shows the user all of the available weather data for the climate division and year for which they are trying to predict a cone crop. The data is shown for reference and can be changed by the user. Users can use more specific local data or do a sensitivity analysis to determine the effects of individual variables. Only available data will be shown, and this can affect the predictions that can be made. One oddity is that the months of July and August do not show heating degree days (HDD); this is because, in all of the historical data used, these values were always zero. Since this constant value of zero is problematic with the canonical discriminant function, HDD values for July and August were not used in the model. When the user is satisfied with the data shown, a single click on the “Calculate” button will do the necessary calculations to show model results. The user can also click “Go Back” to select different climate data or “Quit” to end the program.

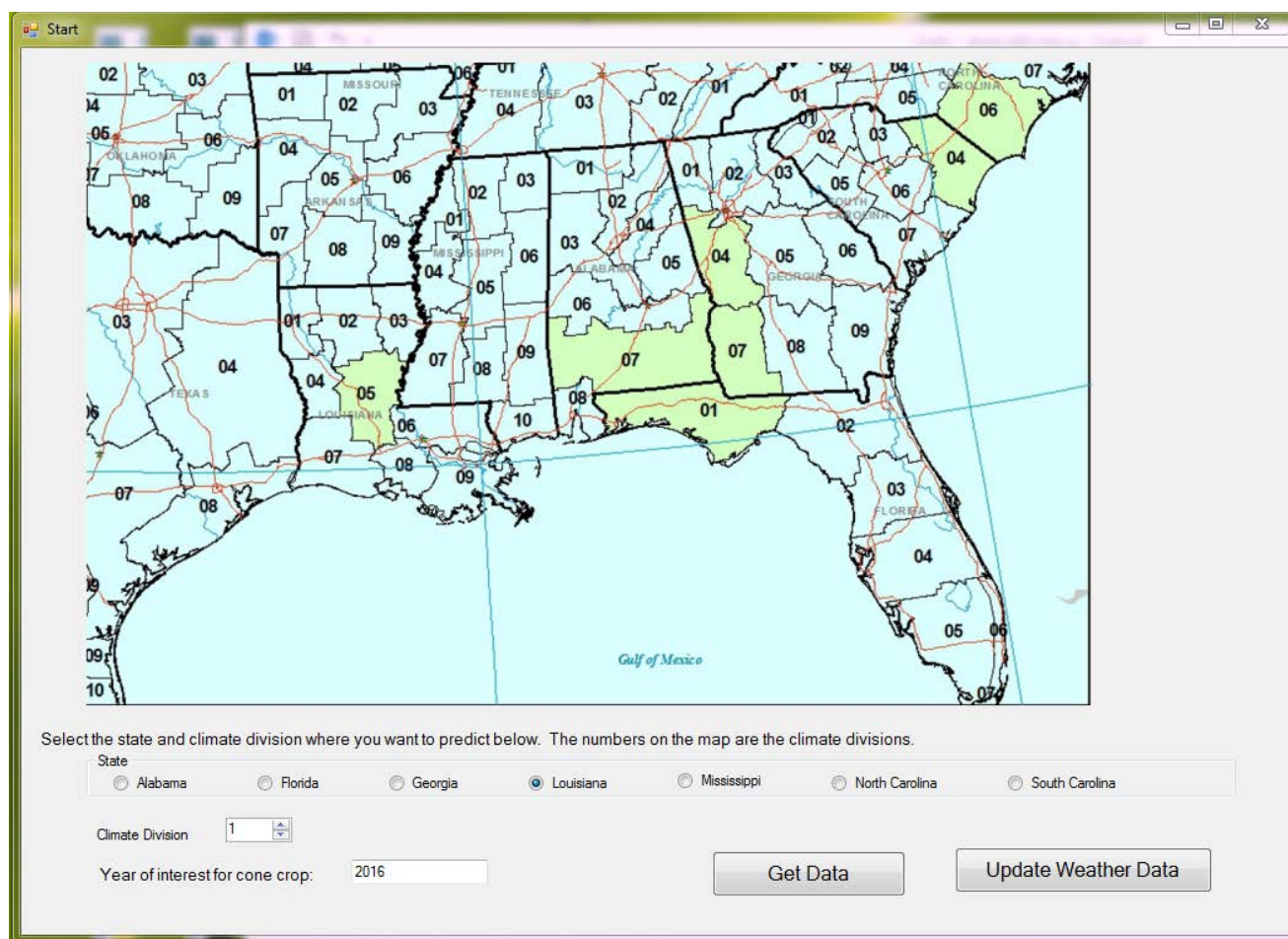


Figure 3—Opening screen from the program LongCones. This is the screen where the user will pick the location of interest and the prediction year.

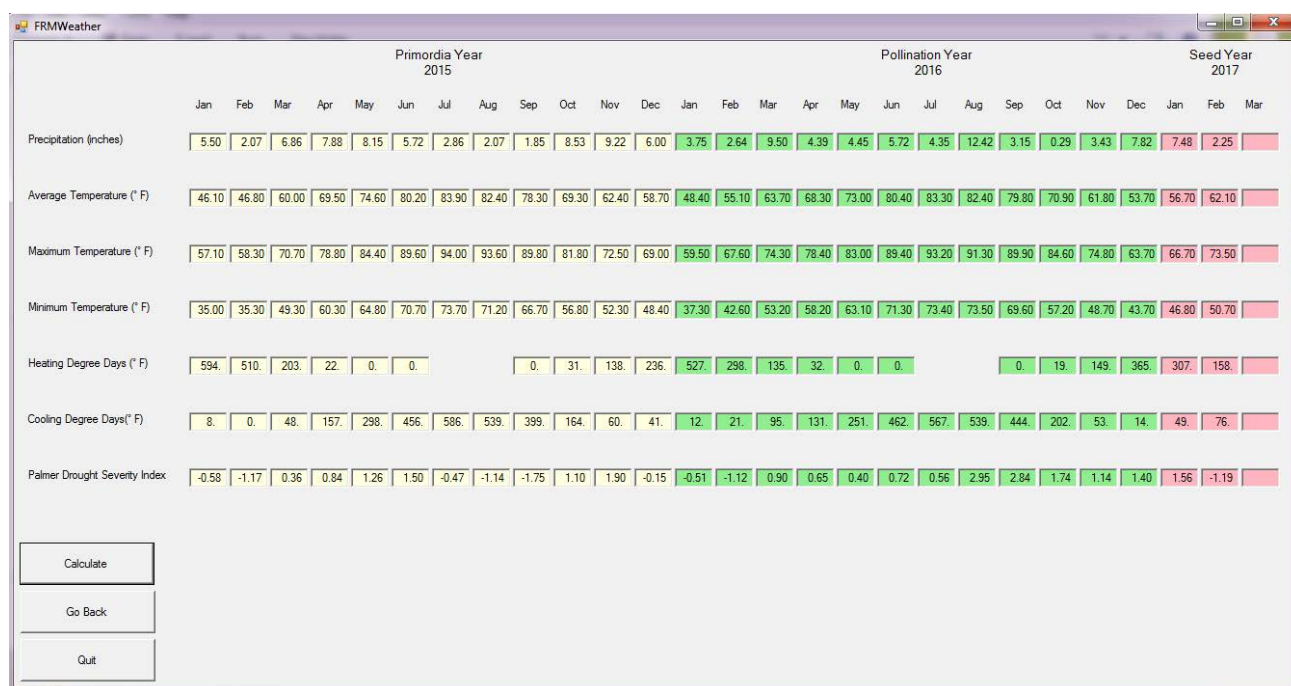


Figure 4—Weather data screen from the program LongCones. This is a presentation of the climate data for the location and timeframe from which the user will make model predictions. The data can be edited or the “Calculate” button can be clicked to make predictions.

Figure 5 shows the final results of the program. There are four models, namely: (1) a model that only uses data up through June of the pollination year, (2) a model that uses data through September of the pollination year, (3) a model that uses data up through December of the pollination year, and (4) a model that uses data up through March of the seed year. In figure 5, only three results are shown since the data for March of 2017 was not yet available. Results are only presented for those models that have sufficient data for the calculation.

In addition to estimated crop quality class, the weather data screen (fig. 5) also presents distances from the mean of all classes. These can be used to judge how confident the user can be in the results. In the example, using data only through June of the pollination year resulted in the prediction of a bumper crop, but the distances to the means indicated that while the bumper class mean was closest, the poor-to-failed class mean was also very close. With the additional data added in the next two models, through September and through December of the pollination year, the distance from the bumper mean increases while the distance to the poor-to-failed mean decreases. Another check on the model is to see how many of the four models agree. In validation, having four models in agreement increased the reliability by 31 percent over having three models in agreement.

DISCUSSION

We envision that this model will be used primarily to predict quality of cone crops, which will enhance

longleaf pine regeneration planning. However, as with all models, caution must be taken since there is an element of error. At its best in March of the seed year, the model is correct 90 percent of the time, but in June of the previous year it is only correct 82 percent of the time.

Aside from the practical predictions that can be made, another use for this model is testing effects of variables on cone crops. In predicting the 2017 cone crop for climate division 5 in Louisiana, the June of the pollination year model predicted a bumper crop, but subsequent models predicted a poor-to-failed crop. The 1984 crop for the same location was actually a bumper crop, and all of the models predicted that it would be a bumper crop. Using the 1984 data as a guide and a little trial and error, it was discovered that simply changing the average temperature for November of the pollination year from 61.8 to 57.7 °F made the December model predict a bumper crop. The September model still predicted a poor-to-failed crop unless the rainfall of August of the pollination year was changed from 12.42 to 5.56 inches. Changing both variables resulted in all of the available models predicting a bumper crop. Unfortunately, the user of this model might have difficulty in reproducing the above result. NOAA updates climate data values monthly, and these updates do more than just add to the data collection. They also adjust values for the most recent two calendar years (NOAA 2014). The data files current at this writing do not produce the phenomenon described above.



Figure 5—Weather data screen from the program LongCones after the “Calculate” button has been clicked. Results for the models with sufficient input data are shown.

The implications of the sensitivity of the model to small changes is something that needs to be investigated further. By changing the August rainfall from 12.42 to 5.56 inches, a change from an extreme value to a moderate value, the change in distance to a bumper crop changed from 3.4 to 1.3. However, by changing November temperature from 61.8 to 57.7 °F, a moderate change, the ED to the bumper crop mean changed from 6.0 to 47.7 and the ED to the poor-to-failed crop changed from 1.1 to 52.6. Thus, bumper class is the closest result, but at that distance it might not be totally reliable. The threshold for the distance rendering the model meaningless is another area for future investigation.

Finally, the model might be criticized for over parameterization since the full model for March of the seed year (27-month) has 370 parameters and is based on 390 observations. Previous work (Leduc and others 2016) has shown that all of these variables are necessary to get class separation, as shown in figure 1. Thus, the model may not be as applicable in a general sense as is desired. However, the models that terminate in earlier months have fewer parameters and the same number of observations. The June of the pollination year model (18-month) has only 248 parameters, which gives some freedom for generalization. In applying this model, one should consider the variation in models depending on endpoint to help judge reliability.

CONCLUSION

A model was developed that predicts longleaf pine crops and could be a useful tool in planning regeneration strategies. The computer program makes the model easy to implement. However, the model is only the starting point for investigating prediction reliability and potentially finding strategies to increase future cone crops.

ACKNOWLEDGMENTS

We thank the many people who have contributed longleaf pine cone counts since 1958 and especially Dale G. Brockway who made this data available to us.

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WITHIN-TREE VARIABILITY IN WOOD QUALITY PARAMETERS FOR MATURE LONGLEAF PINE

Chi-Leung So, Thomas L. Eberhardt, and Daniel J. Leduc

Abstract—Mature longleaf pine (*Pinus palustris* Mill.) trees were harvested from a spacing, thinning, and pruning study on the Palustris Experimental Forest, LA, to assess wood quality parameters of earlywood specific gravity (SG), latewood SG, ring SG, and latewood percent using X-ray densitometry. For each of ten 70-year-old trees used in the study, 2-inch thick disks were cut every 2 feet from the stump cut at 0.5 feet. A strip of wood was cut from bark to bark, and through the pith, to afford “cores” from bark to pith for the northern and southern cardinal directions. This sampling scheme provided the opportunity to compare wood properties determined at breast height to whole-tree area-weighted values. Only a few effects of silvicultural treatment were significant. As expected for spacing, an increase in ring width appeared to accompany wider planting densities. Significant differences were observed in whole-tree values as compared with those at breast height alone, in particular with north-south differences related to ring and latewood SG.

INTRODUCTION

The southern pine forest was predominated by longleaf pine (*Pinus palustris* Mill.) prior to colonial settlement of the United States. Through fire suppression activities and natural seeding on abandoned agricultural lands, the occurrence of loblolly pine (*Pinus taeda* L.) increased (Fox and others 2007, Stanturf and others 2002).

Widespread planting further changed the landscape to where now more than half of the volume of southern pine timber is loblolly pine (Shiver and others 2000). Studies on the productivity, variability, and utilization of the southern pines have therefore focused primarily on loblolly pine. However, as efforts to restore longleaf pine ecosystems continue, it is anticipated that longleaf pine timber will be sufficiently available for utilization (Landers and others 1995), thus, warranting assessments of physical and mechanical properties of the currently available longleaf pine merchantable timber.

Prior to a timber sale from a spacing, thinning, and pruning study site on the Palustris Experimental Forest, longleaf pine trees were made available for destructive sampling. Preliminary results provided tree property maps of predicted SG values for a few individual trees (So and others 2010); this was accomplished via multivariate analysis of near infrared spectroscopy data collected by scanning pith to bark wood specimens. Additional measurements were performed on the study

trees to determine inner and outer bark thicknesses along the length of the bole (Eberhardt 2013, 2015). Measurements of bark thicknesses were conducted to provide data for each of the cardinal directions (north, south, etc.). Results of paired t-tests showed statistically-significant differences in bark thickness measurements for the northern and southern quadrants, but not for the eastern and western quadrants (Eberhardt 2013). Given the appearance of limited north vs. south SG asymmetries in the aforementioned individual tree property maps (So and others 2010), the present study was conducted to determine if significant differences could be determined by the direct determination of SG by X-ray densitometry.

MATERIALS AND METHODS

Ten 70-year-old longleaf pine trees were harvested. The study site was located on the Palustris Experimental Forest, LA (N31.176°, W92.677°). Trees were sampled across the different silvicultural treatments (spacing, thinning, and pruning), affording a range of diameters at breast height from 5.7 to 19.6 inches (table 1). Total heights varied from 57.7 to 90.3 feet. Reviewing the treatments, spacing was at four levels (4.3, 5.2, 6.2, 13.1 feet), thinning was to a variety of basal areas (e.g., 100 square feet per acre), and trees were either pruned or left unpruned.

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Table 1—General characteristics of 70-year-old longleaf pine trees used in study

Tree Number	Diameter at Breast Height	Height	Height to Live Crown	Diameter at Live Crown	Basal Area	Spacing	Pruning
	(inches)	(feet)	(feet)	(inches)	(feet ² /acre)	(feet × feet)	
1	19.6	83.6	38.8	16.1	100	6.2	unpruned
2	13.0	89.5	58.7	6.8	186	6.2	pruned
4	14.5	86.4	50.5	9.3	100	5.2	unpruned
7	5.7	57.7	46.0	2.3	162	4.3	pruned
8	16.8	87.4	48.5	12.1	142	13.1	pruned
9	16.6	86.4	60.3	8.7	96	5.2	unpruned
11	10.3	90.3	65.7	5.2	151	5.2	pruned
12	13.7	75.0	42.9	8.2	151	5.2	pruned
13	13.4	83.2	50.7	10.7	112	6.2	pruned
17	11.3	85.8	64.8	5.0	158	5.2	pruned

Although the treatments for the harvested trees were known (table 1), it was well beyond the scope of the current study to make any definitive assessments of treatment effects on tree growth and wood quality parameters due to the limited number of trees subjected to destructive sampling. That stated, ANOVAs were generated simply to observe any trends in the wood quality parameters, classifying the data by spacing and pruning. The thinning data were also classified, using current basal areas (<111, 111-155, >155 square feet per acre), with these criteria affording a minimum of three trees in each basal area class.

Prior to felling, the trees were marked to identify the northern cardinal direction. Trees were felled, de-limbed, marked again, and then bucked with a chainsaw to produce 2-inch thick disks cut at a height of 0.5 feet and then every 2 feet along the bole, including at breast height. The northern cardinal direction was again marked on the top of each disk, after which a 0.5-inch thick wood slice was sectioned along the north-south direction, through the center of each disk, encompassing the pith. The wood slices were then extracted by steeping in acetone at room temperature over a 2-week period, with the solvent exchanged every 2nd or 3rd day. Wood slices were placed in wooden core holders, dried (50 °C, 24 hours), and then permanently glued into place. Mounted specimens were sawn into 2.3-mm thick strips, from bark to pith, leaving the transverse surface of the core exposed and bordered by adhering wood strips remaining from the core holders. Densitometry was employed to collect wood quality parameters (e.g., ring SG, latewood percent) and ring widths; this was performed using a Quintek Measurement Systems (Knoxville, TN) X-ray densitometer as described in Eberhardt and Samuelson (2015). Briefly, SG measurements were determined at 0.06-mm intervals,

scanning along the radial direction from bark to pith. A SG of 0.480 was used to differentiate between earlywood and latewood zones (Antony and others, 2012; Clark and others, 2006; Koubaa and others, 2002).

Wood quality parameters were collected from the 'whole tree' (all disks), as well as from the breast height disk alone for each tree. These parameters were weighted by ring area to obtain mean area-weighted whole-core values for each tree, both at breast height, and to determine whole-tree values, further weighting the data by the transverse area of each disk. The effects of spacing, pruning, and basal area class on wood quality data were tested by analysis of variance (ANOVA) using SAS (SAS 2004), while the effect of cardinal direction was tested by a paired t-test using SAS (SAS 2004), both with significance at a $P < 0.05$ level.

RESULTS AND DISCUSSION

The effects of thinning and pruning treatments on the wood quality parameters are shown in table 2 for the breast height and whole-tree values. Significant differences ($P < 0.05$) were not observed for the breast height data. Similar results were obtained with whole-tree (all disks in a tree) values (table 2) with the exception being ring SG showing a significant difference ($P = 0.008$) with basal area. ANOVA results for the effect of spacing on the wood quality parameters are shown in table 3. Note that the spacing effects may have been particularly affected by having only one tree representing the lowest and highest spacings. Similar to the other treatments, most of the results show no significant differences at breast height or on a whole-tree basis.

A major factor affecting wood quality parameters for the whole tree, as compared with those only at breast height, is juvenility. The wood formed by trees changes

Table 2—Wood quality data ANOVA for effect of pruning and basal area including mean values, standard deviations (in parentheses), and probabilities (*P*) for breast height and whole tree

			Wood Quality Parameter				
Treatment	Class	<i>N</i>	Ring SG	Latewood SG	Earlywood SG	Latewood Percent	Ring Width (mm)
Breast Height							
Pruning	pruned	7	0.575 (0.041)	0.766 (0.035)	0.348 (0.019)	54.1 (6.5)	2.71 (0.82)
	unpruned	3	0.587 (0.011)	0.780 (0.009)	0.346 (0.016)	55.9 (2.4)	2.87 (0.33)
	<i>P</i>		0.635	0.539	0.869	0.678	0.768
Basal Area	<111	3	0.587 (0.011)	0.780 (0.009)	0.346 (0.016)	55.9 (2.4)	2.87 (0.33)
	111-155	4	0.577 (0.050)	0.763 (0.017)	0.349 (0.024)	54.8 (8.8)	3.06 (0.76)
	>155	3	0.573 (0.036)	0.771 (0.056)	0.347 (0.017)	53.2 (3.0)	2.25 (0.78)
	<i>P</i>		0.895	0.796	0.980	0.869	0.332
Whole Tree							
Pruning	pruned	7	0.526 (0.010)	0.726 (0.026)	0.337 (0.018)	48.6 (3.5)	2.63 (0.69)
	unpruned	3	0.541 (0.007)	0.758 (0.019)	0.333 (0.015)	49.0 (0.5)	2.91 (0.58)
	<i>P</i>		0.051	0.090	0.757	0.877	0.549
Basal Area	<111	3	0.541 (0.007)	0.758 (0.019)	0.333 (0.015)	49.0 (0.5)	2.91 (0.58)
	111-155	4	0.519 (0.006)	0.719 (0.023)	0.334 (0.019)	48.0 (4.5)	2.91 (0.63)
	>155	3	0.535 (0.007)	0.736 (0.030)	0.341 (0.019)	49.4 (2.3)	2.24 (0.66)
	<i>P</i>		0.008	0.178	0.853	0.845	0.360

Significance (*P* < 0.05) is shown in bold.

with age, with the inner wood, so-called juvenile wood, having different chemical, physical, and mechanical properties than that of wood formed later in the life of the tree, so-called mature wood. Intuitively, there is not an abrupt change from juvenile wood to mature wood, thus the term transition wood may be used to classify that wood that is neither juvenile, nor mature. Further complicating matters, annual rings assigned to juvenile wood are not specifically set. In the southern pines, juvenile wood can vary not only between species, but other factors such as region, let alone differences resulting from the use of different criteria (e.g., microfibril angle, specific gravity, latewood percent) for demarcation (Clark 2006). Juvenility in loblolly pine has been well studied while that for longleaf pine has not. Since juvenility can last as long as 20 years for loblolly pine, we used a cambial age of 20 years for each disk to generally classify the wood quality data into predominantly juvenile or mature wood zones.

The effect of spacing on the wood quality parameters from juvenile (≤ 20 years) and mature (≥ 21 years) wood zones is shown in table 3. A significant difference is observed at breast height in the juvenile wood ($P=0.045$), in which the lowest spacing (4.3 feet $\times$ 4.3 feet) is significantly different from the other spacings, though this may have been an anomaly, affected by sample size. The most striking differences for the effect of spacing are observed with latewood percent, in which significant

differences are observed with the whole-core and mature wood data, both on a whole-tree basis (table 3); however, this should be interpreted with caution given that the seemingly higher latewood percent values at the highest and lowest spacings were provided by only one tree for each. As expected for spacing, an increase in ring width appeared to accompany wider planting densities. Notwithstanding, many reports in the literature draw conclusions on mean values and not statistical comparisons. Results presented here do allow some degree of comparison.

The primary objective of this study was to assess whether there were significant differences between wood quality parameters between the northern and southern cardinal directions. The process of collecting tree cores is often not specified in the literature, presumably because it may be deemed inconsequential. Thus, a core entry point may always be taken in a single cardinal direction, at a position in which a knowledgeable field worker can generate a core from bark to pith (i.e., at the widest diameter), or simply randomly. Knowing that bark thickness has been shown to differ between the northern and southern cardinal directions, but not between the eastern and western cardinal directions (Eberhardt 2013, 2015), a strip of wood was cut from each disk, bark to bark, and through the pith, from south to north (So and others 2010).

Table 3—Wood quality data ANOVA for effect of spacing including mean values, standard deviations (in parentheses), and probabilities (*P*) for breast height and whole tree

			Wood Quality Parameter				
Sample	Spacing	N	Ring SG	Latewood SG	Earlywood SG	Latewood Percent	Ring Width (mm)
Breast Height							
Whole	4.3	1	0.542 (—)	0.707 (—)	0.360 (—)	52.5 (—)	1.52 (—)
Core	5.2	5	0.574 (0.034)	0.777 (0.024)	0.344 (0.020)	53.0 (4.8)	2.68 (0.69)
	6.2	3	0.577 (0.011)	0.775 (0.024)	0.343 (0.018)	54.2 (3.1)	3.06 (0.18)
	13.1	1	0.644 (—)	0.787 (—)	0.362 (—)	66.7 (—)	3.47 (—)
	P		0.171	0.137	0.737	0.118	0.170
	Juvenile	4.3	1	0.519 (—)	0.680 (—)	0.361 (—)	50.1 (—)
Core	5.2	5	0.575 (0.040)	0.784 (0.026)	0.357 (0.021)	51.1 (6.3)	4.08 (1.16)
	6.2	3	0.574 (0.002)	0.781 (0.023)	0.351 (0.017)	51.2 (0.9)	4.56 (0.50)
	13.1	1	0.600 (—)	0.765 (—)	0.364 (—)	59.4 (—)	5.60 (—)
	P		0.404	0.045	0.927	0.542	0.173
	Mature	4.3	1	0.561 (—)	0.730 (—)	0.359 (—)	54.6 (—)
Core	5.2	5	0.574 (0.033)	0.772 (0.026)	0.339 (0.021)	54.0 (4.0)	1.90 (0.31)
	6.2	3	0.578 (0.017)	0.773 (0.028)	0.337 (0.018)	55.6 (5.4)	2.18 (0.28)
	13.1	1	0.672 (—)	0.801 (—)	0.360 (—)	71.3 (—)	2.13 (—)
	P		0.084	0.387	0.617	0.065	0.083
	Whole Tree						
Whole	4.3	1	0.535 (—)	0.701 (—)	0.360 (—)	51.7 (—)	1.60 (—)
Core	5.2	5	0.532 (0.017)	0.747 (0.025)	0.334 (0.019)	47.8 (2.0)	2.56 (0.57)
	6.2	3	0.528 (0.005)	0.744 (0.013)	0.333 (0.012)	47.3 (1.5)	3.12 (0.27)
	13.1	1	0.526 (—)	0.689 (—)	0.330 (—)	54.6 (—)	3.34 (—)
	P		0.952	0.116	0.538	0.045	0.110
	Juvenile	4.3	1	0.511 (—)	0.690 (—)	0.354 (—)	47.1 (—)
Core	5.2	5	0.523 (0.019)	0.746 (0.024)	0.340 (0.020)	45.2 (2.8)	3.64 (0.92)
	6.2	3	0.517 (0.006)	0.745 (0.010)	0.336 (0.013)	43.8 (1.8)	4.42 (0.40)
	13.1	1	0.497 (—)	0.681 (—)	0.329 (—)	47.4 (—)	4.89 (—)
	P		0.523	0.054	0.788	0.549	0.127
	Mature	4.3	1	0.567 (—)	0.715 (—)	0.369 (—)	57.9 (—)
Core	5.2	5	0.538 (0.018)	0.745 (0.031)	0.330 (0.018)	49.9 (2.3)	1.69 (0.29)
	6.2	3	0.536 (0.005)	0.744 (0.015)	0.330 (0.012)	49.8 (1.2)	2.06 (0.66)
	13.1	1	0.547 (—)	0.695 (—)	0.330 (—)	59.5 (—)	2.26 (—)
	P		0.358	0.377	0.268	0.008	0.198

Significance ($P < 0.05$) is shown in bold.

— = No standard deviation value ($N=1$)

Wood quality parameters measured at breast height are shown in table 4. The lack of a significant difference between the northern and southern directions is the most consistent finding. However, even without significance, the trend for the whole core shows that the northern direction has equal or higher values for all of the wood quality parameters except for earlywood SG. Plotting of the individual data points was carried out to determine if there were any differences from pith to bark that would be lost by comparisons from the whole-core data (fig. 1). Line plots were originally generated (not shown), but trends were difficult to discern, thus smoothing was applied to facilitate the visualization of any possible differences. In general terms, similar plots are obtained for the two cardinal directions, save for

the appearance of greater deviation moving further out, through the mature wood zone, towards the bark, while less variation is observed in the transition zone between juvenile and mature wood (fig. 1). Nevertheless, the data in table 4 were still classified into juvenile and mature wood zones. The resultant breast height values also show no significant differences between the northern and southern directions. The same trend of equal or higher values from the northern direction is still true, except that latewood percent is slightly higher for the southern direction in juvenile wood. This might well be the result of the higher earlywood SG in the southern direction causing more of the wood to be classified as latewood. The clearest north-south differences between juvenile and mature wood data arise from ring width

Table 4—Wood quality data paired t-test for effect of direction including mean values, standard deviations (in parentheses), and probabilities (*P*) for breast height and whole tree

Sample	Direction	Wood Quality Parameter				
		Ring SG	Latewood SG	Earlywood SG	Latewood Percent	Ring Width (mm)
Breast Height						
Whole	North	0.580 (0.031)	0.774 (0.028)	0.346 (0.020)	54.8 (5.0)	2.81 (0.73)
Core	South	0.577 (0.040)	0.767 (0.038)	0.349 (0.017)	54.5 (6.9)	2.68 (0.67)
	difference	0.003 (0.020)	0.007 (0.028)	0.003 (0.014)	0.3 (5.0)	0.13 (0.35)
	<i>P</i>	0.650	0.479	0.526	0.860	0.267
Juvenile	North	0.572 (0.038)	0.777 (0.041)	0.354 (0.021)	51.4 (6.8)	4.15 (1.21)
Core	South	0.571 (0.042)	0.765 (0.045)	0.358 (0.019)	52.1 (7.4)	4.15 (1.21)
	difference	0.001 (0.043)	0.013 (0.036)	0.004 (0.020)	0.7 (10.0)	0.00 (0.58)
	<i>P</i>	0.950	0.291	0.539	0.825	0.995
Mature	North	0.585 (0.038)	0.772 (0.031)	0.342 (0.021)	56.7 (5.8)	1.99 (0.44)
Core	South	0.581 (0.044)	0.770 (0.035)	0.343 (0.018)	55.7 (7.6)	1.82 (0.42)
	difference	0.004 (0.023)	0.002 (0.034)	0.001 (0.144)	1.0 (3.6)	0.17 (0.27)
	<i>P</i>	0.577	0.887	0.803	0.426	0.084
Whole Tree						
Whole	North	0.535 (0.013)	0.743 (0.029)	0.335 (0.016)	48.9 (3.2)	2.73 (0.65)
Core	South	0.525 (0.011)	0.727 (0.030)	0.337 (0.017)	48.4 (2.6)	2.69 (0.63)
	difference	0.010 (0.008)	0.016 (0.018)	0.001 (0.004)	0.5 (1.9)	0.04 (0.07)
	<i>P</i>	0.002	0.019	0.333	0.449	0.096
Juvenile	North	0.518 (0.016)	0.739 (0.030)	0.338 (0.016)	44.6 (2.1)	3.85 (1.00)
Core	South	0.517 (0.017)	0.727 (0.032)	0.340 (0.016)	45.7 (3.2)	3.84 (1.00)
	difference	0.002 (0.008)	0.012 (0.008)	0.002 (0.005)	1.1 (2.4)	0.01 (0.10)
	<i>P</i>	0.511	0.001	0.285	0.206	0.733
Mature	North	0.548 (0.021)	0.746 (0.033)	0.333 (0.018)	52.2 (4.6)	1.83 (0.54)
Core	South	0.532 (0.012)	0.725 (0.032)	0.335 (0.020)	50.8 (3.5)	1.71 (0.53)
	difference	0.016 (0.016)	0.021 (0.029)	0.001 (0.006)	1.4 (2.4)	0.12 (0.18)
	<i>P</i>	0.012	0.051	0.505	0.095	0.058

Significance (*P* < 0.05) is shown in bold.

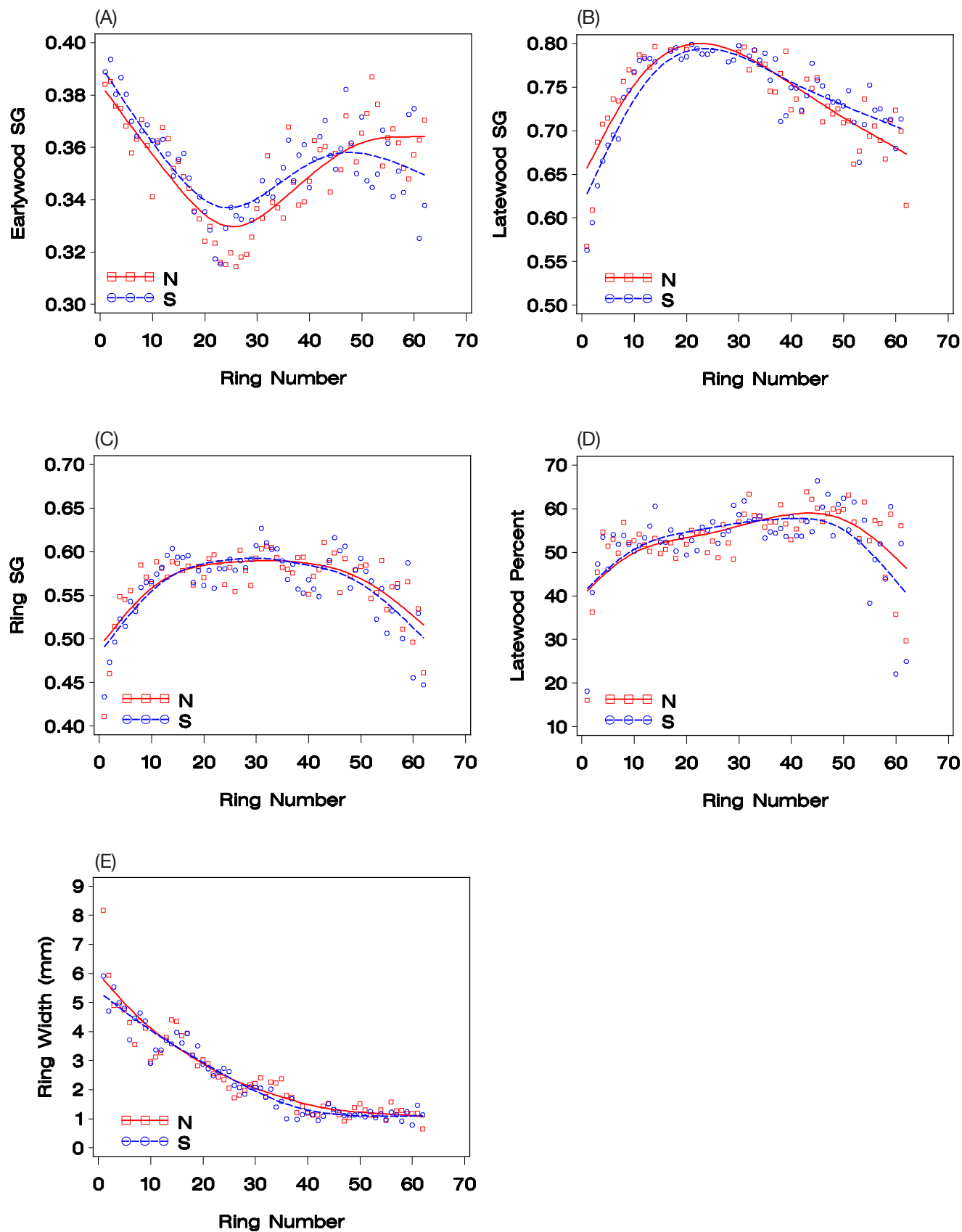


Figure 1—Plot of breast height values of (A) earlywood SG, (B) latewood SG, (C) ring SG, (D) latewood percent and (E) ring width in both the north and south cardinal directions.

data with identical values for the juvenile wood while differences (approaching significance) exist for the mature wood.

Whole-tree values for the wood quality parameters appear lower than those determined at breast height alone (table 4). Intuitively, this is because the data are weighted by the contribution from each wood disk and thus, an increasing contribution from the juvenile wood zone going upwards to the top of the tree is generally expected. These data mirror the trends shown at breast height, but some of the north-south differences are now showing up as significant. For the whole-core data, significantly higher values are now obtained for ring SG and latewood SG in the northern direction; similar results are also obtained for the mature wood data. The ring width observation between juvenile and mature wood match that observed at breast height. Furthermore, ring width for the whole core trends with a seemingly higher mean value in the northern direction compared to the southern direction, following the same trend of a significantly higher thickness for the bark in the northern direction (Eberhardt 2013, 2015). The caveat, herein, is assuming that the asymmetry in wood properties between the northern and southern directions is specifically related to a biological compass applicable to all trees. Indeed, it is impossible to even suggest any causes for the observed differences; however, the results presented here suggest that it would be prudent to be consistent in sampling to avoid any data artifacts that may manifest from repeating patterns of wood property asymmetry in trees.

CONCLUSION

The variability in wood quality parameters for mature longleaf pine trees was investigated using X-ray densitometry. The effect of cardinal direction and treatment (spacing, thinning, and pruning) was assessed, although the number of trees subjected to destructive testing was very limited. The trees were assessed at breast height and on a whole-tree basis. Few significant differences were observed with treatment. Significant differences based on cardinal direction were mostly observed in whole-tree values rather than those at breast height alone, in particular with north-south differences related to ring and latewood SG.

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The 19th Biennial Southern Silvicultural Research Conference was held in Blacksburg, VA, March 14–16, 2017. This conference provided a forum for silviculturists, researchers, and practitioners to exchange the latest research information on the ecology and management of southern forests. Of the 70 oral presentations and 55 posters presented during the conference, 56 papers and 16 poster abstracts were submitted for these proceedings. The papers cover 15 topics, which include Oak Silviculture, Invasive Species, Disturbance and Damaging Agents, Forest Mensuration and Modeling, Pine Bark Beetle, Loblolly Pine Fertilization, Afforestation, Long-Term Silvicultural Studies, Bottomland Silviculture, Forest Soils and Best Management Practices, Loblolly Pine: Density & Competition Control, Shortleaf Pine Silviculture, Longleaf Pine Silviculture, Ecophysiology, and Fire. Papers and abstracts from oral and poster presentations provide managers with the latest research from universities, government agencies, and natural resource agencies.

Keywords: Afforestation, BMP, bottomland, disturbance, ecophysiology, fertilization, fire, forest mensuration, invasive, loblolly pine, longleaf pine, modeling, oak, pine bark beetles, shortleaf pine, silviculture, soil.



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